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Polypores, *Agrobacterium* and ivy damage on Hungarian ancient trees

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Abstract

Ancient trees are important habitats, confer vital ecological roles and function as cultural legacies. Old trees with large girth are keystone structures in various ecosystems. We aim to present which species amongst the greatest Hungarian trees (and some other phanerophyte plants) are damaged by polypores (the most important agents of wood decay), *Agrobacterium tumefaciens* (usually causing root tumour) or ivy (competing against the native vegetation and causing windthrow damage) and at what extent and frequency; and whether there is a relationship between these types of damage and the origin of the species (native or adventive) or its situation (solitary or surrounded by other trees). We measured 2,000 trees, belonging to 29 native and 43 non-native species. Polypore infection could be detected in 12.2% of the observed 531 settlements, 22.8% are damaged by *Agrobacterium* and 29.6% by ivy, while 51.2% by other types of pests and diseases. Altogether, one third of the observed 2000 ancient or veteran trees suffered from one or more types of damage. A total of 33.5% of the native species (519 specimens out of 1550) and 28.7% of the adventives (129 trees out of 450) are damaged by any (or more than one) of the mentioned infections or ivy. Mostly, damage occurred to those old trees that stand in a park or forest, while the single (solitary) trees were usually healthy. The most infected regions are the western and south-western counties, while the Northern Hungarian Mountain Range is much less affected, despite its great sample size. Low damage was detected in the Great Hungarian Plain, but the number of sample areas and veteran trees was also low here. The damage to old trees remains without any management or healing in Hungary, since the only effective solution would be prevention.

Keywords

environmental history, notable tree, pest, plant protection, veteran tree, non-native

Introduction

Ancient trees have large girth and astonishing ecological value. The age when a tree can be considered ancient is species specific (Hartel and Plieninger 2014). They contain varied microhabitats, such as hollows or hollowing trunks or branches, cavities, wood mould, decaying wood in the crown, flaking bark which support exceptional numbers of specialised species including fungi, lichens, birds, small mammals and endangered species of wood-living insects (Ranius and Jansson 2000; Read 2000; Sverdrup-Thygeson et al. 2010; Bergman et al. 2012). This is why ancient tree-based systems are considered as global hotspots of biodiversity (Buse et al. 2010) and the old trees are keystone structures in natural, agricultural and urban ecosystems (Gibbons et al. 2008). Their great size and age provide ecological niches of value to specialised flora and fauna that cannot be provided by younger, smaller trees (Lindenmayer et al. 2014) and function as cultural-emotional legacies as well, linking the past to the present (Manning et al. 2006; Lonsdale 2013; Eriksson 2018). The preservation of landscapes, where there is still a high density of ancient trees, should be a priority for all European countries (Zapponi et al. 2017).

One of the reasons why we are so concerned about the ancient trees in Hungary is that we believe that they are of great importance when considering nature values and qualities in forests, agricultural landscapes, as well as cultural heritage and landscape features. This is why it is inevitable that we review their general data (girth, perimeter, height etc.), ethnographical and historical relations, health status and endangering factors to plan their active conservation. In Hungary, the first calls for the protection of ancient, giant trees date back to the early 20th century (e.g. Rapaics 1929). The respect towards them has led to the protection of several ancient trees, most of them within settlements (Tardy 1996). Their main data are registered in online databases (Pósfai Gy 2019, Monumental Trees 2019, Hungarian Monumental Trees 2019). The greatest Hungarian database (Pósfai Gy 2019) started as a private website, but as a good example of citizen science, everyone can send new data; however, these will be uploaded only after validation (visited and measured on the spot) by the founder. Overviews on the cult of the oak genus and presentation of some remarkable Hungarian oak trees were provided by Szakonyi (2018), while in the case of lime species and sweet chestnut, see our previous articles (Takács and Malatinszky 2012; Takács et al. 2015). Iváncsics and Filepné (2019) provide explanations to visual tree assessment.

There are thousands of ancient trees in Hungary and they need special care and protection. However, most of them are situated in hidden areas, far from parks and gardens, thus, without any special attention being given to their health state, maintenance or care. Our aim is to present which specimens and species amongst the greatest Hungarian trees (and other phanerophyte plants, such as black elder and hawthorn species) are damaged by well-known pests and diseases, such as polypores, *Agrobacterium* or ivy and to what extent and frequency. As far as we know, there has been no similar research so far. Our study covers every Hungarian region, about 2,000 very old, sizeable trees, some of them being champions with the largest species-specific girth or height in the country.

Polypores are a group of basidiomycetes fungi that form fruiting bodies with pores or tubes on the underside. They inhabit tree trunks or branches consuming the wood and, thus, they are the most important agents of wood decay. Even though saproxylic fungi act as keystone species in forest ecosystems (Moose et al. 2019), sustaining, for example, beetle communities (Andrési and Tuba 2018), several polypore species are serious pathogens of plantation trees and are major causes of timber spoilage. The most common polypore in Hungary is the tinder fungus (*Fomes fomentarius*), a stem decay fungal plant pathogen of beech and other deciduous trees, such as birch, poplar, willow and oak species (Igmándy 1991). Its mycelium penetrates the wood of trees through damaged bark or broken branches, causing white rot in the host. It continues to live on trees long after they have died, changing from a parasite to a decomposer (Baum et al. 2003). In Hungary, other polypores of significance are *Fuscoporia torulosa*, *Inonotus cuticularis*, *Daedaleopsis confragosa*, *D. c.* var. *tricolor*, *Phellinus igniarius*, *Ph. tremulae* and *Ganoderma applanatum* (Vasas and Locsmándi 2010; Gerhardt 2017).

Bacteria and viruses usually do not play a significant role in the diseases of old trees. However, injuries to the infected tissues are variable (Bartosiewicz and Siewniak 1979). *Agrobacterium* is a genus of Gram-negative bacteria that causes tumours in plants. *A. tumefaciens* causes crown-gall disease in plants, which is a tumour-like growth or gall on the infected plant, often at the junction between the root and the shoot. *A. rhizogenes* induces root tumours in apple and its relatives. Old trees are infected by *A. tumefaciens*, usually causing root tumour, but even shoots might be stressed in case of poplars, oaks, limes, sycamores and willows (Glits and Folk 2000). It may attack any tree species in Hungary, but the visible tumours on the shoot system are most remarkable on poplars (Szabó 2003). This pathogen penetrates the wood of trees through damaged bark. The bacterial tumour-inducing (Ti) plasmid penetrates into the genome of the host (Atherly 2004).

Ivies (*Hedera* spp.) create a dense, shade-tolerant evergreen cover that can spread through underground rhizomes and above-ground runners quickly and outcompete the native vegetation. In Europe, the harm caused by common ivy (*Hedera helix*) is generally minor, although trees climbed by ivy (as high as 25 metres) can suffer windthrow damage (Gencsi and Vancsura 1997). Oaks with their crown almost completely wrapped by ivy are quite common. Trees may show clear symptoms of decline, such as shoot withering and progressive crown reduction (Garfì and Ficarrotta 2003). Tree characteristics and spatial patterns of species significantly influence ivy distribution. Preferred hosts are large, isolated trees (Castagneri et al. 2013). Protection against ivy is cost and labour-intensive and usually remains without a real result (English Heritage 2010).

We aimed to select those types of damage that can easily be recognised, in order to use them in citizen science activities in the future for the large trees that now remain unattended. This is why we focused on polypores, *Agrobacterium* and ivy infections.

Methods

Inventory of general data of ancient trees

There is only one thorough online database that lists the greatest trees in Hungary, providing species, settlement, GPS coordinates, girth and year of its measurement (Pósfai Gy 2019). Our original aim was to observe every specimen listed in this database. Their total number was 700 in 2008, when we started our study. Meanwhile, this number now exceeds 3500 for 2020 and is still growing. This is why we present actual data of altogether 2000 trees that were listed in this database during our measurements between 2008 and 2017. For each specimen, we measured girth (at 1.3 m height), smallest trunk diameter (at 1.3 m height), smallest crown diameter and height, using a measuring tape, Waldmeister forestry caliper and Haglöf clinometer. Due to the high number of observed specimens, there were no repetitions.

We described the health status on a 5-points scale (1 = dead, 2 = bad condition, 3 = fair condition, 4 = good condition, 5 = excellent condition), based on the status of the crown, diseases, breaks, hollows and maintenance. For example, if only a decayed trunk has remained without branches and bark, then this value is 1. If there are pests or other organisms that cause damage or there is a hollow instead of the crown, but the tree is still able to grow new shoots (ie. is still vital), then this value is 3. While in case of no pests or other organisms that would cause damage and healthy bark and crown without broken branches, this value is 5. Accessibility was expressed by considering the quality (usability) of roads leading to each specimen, presence or absence of informative signs, distance from settlements or roads etc. (1 = very difficult, 2 = poor, 3 = medium, 4 = good, 5 = excellent). We did not aim to measure the age of the trees (due to technical difficulties, risk of damage etc.).

The rates of polypores, *Agrobacterium* and ivy damage are also described on a 5-point scale (Table 1). The majority of observed specimens are not damaged at all, thus, these trees are not presented here.

Measuring health status

To measure the health status of each tree, we used a Fakopp Arborsonic 3D Acoustic Tomograph that estimates the velocity at which sound is conducted through the tree. Acoustic tomography has been used for *in situ* inspection of trees since 1986 (Arciniegas et al. 2014). The tomograph distributes and receives sound waves across 10 sensors placed evenly around the circumference of a tree. The sensors are connected to a laptop. Each sensor is tapped a minimum of three times, an action which propagates sound across the tree. The Fakopp Arborsonic software records the length of time taken for the sound to be received by each sensor and this depends on the density of the substrate (Thompson et al. 2016). Then the software renders three-dimensional computer models of the internal tree density (Figs 21–23). The position of the sensors is important, but it does not markedly affect the result. It is no use measuring on the ground or

Table 1. Determination of each damage category.

Polypores infection categories	
1 = not infected	No conks on trunk, branches, visible roots
2 = sparsely infected	One active or few dead conks
3 = slightly infected	Less than 5 conks
4 = moderately infected	5 to 10, small conks
5 = acutely infected	More than 10 healthy conks
***Agrobacterium* infection categories**	
1 = not infected	No tumour on trunk, branches, visible roots
2 = sparsely infected	One tumour, with less than 20 cm diameter
3 = slightly infected	One tumour, with 20 to 40 cm diameter
4 = moderately infected	Several small tumours (galls)
5 = acutely infected	Totally spread on the tree
Ivy damage categories	
1 = not damaged	No shoots on trunk, branches, visible roots
2 = sparsely damaged	Few young shoots, up to 2 m height on the tree
3 = slightly damaged	Few young shoots, up to 4 m height on the tree
4 = moderately damaged	Young or thin shoots appear even on branches and totally cover the trunk
5 = acutely damaged	Total cover

at the first branches, because it may be misleading. The device is suitable for measuring trees up to 500 cm girth (the cables are too short). However, most of the tree species are greater when they are old. This is why we basically chose old wild pears, since it is a highly valuable species and its girth does not exceed this size even at higher age.

Studied tree specimens

We studied 2000 tree specimens in 531 Hungarian settlements (Fig. 1). Although there are significantly less observed trees on the eastern and south-eastern parts of the country (i.e. the Great Hungarian Plain), we consider this study to be representative, since the mentioned area belongs to the steppe climate zone, with a significantly lower rate of forested areas. Observed species and the minimum girth (at 1.3 m height) required for the study of each specimen are shown in Table 2. We included whether they are native or not, their number in the only thorough database (Pósfai Gy 2019) and the number of measured trees and the presence of pests and other organisms that cause damage (in case of more than one, they are shown in a separate column).

Results

Almost all (98%) of those old trees that are infected only by polypores (i.e. without other damage) are surrounded by other trees in a park or a forest (Table 2). This high ratio is understandable when we consider the fact that most of the affected trees are beech, since we found only a couple of solitary trees of this species. A total of 70 out of 400 measured beech trees (i.e. 17.5%) were infected by polypores (including the

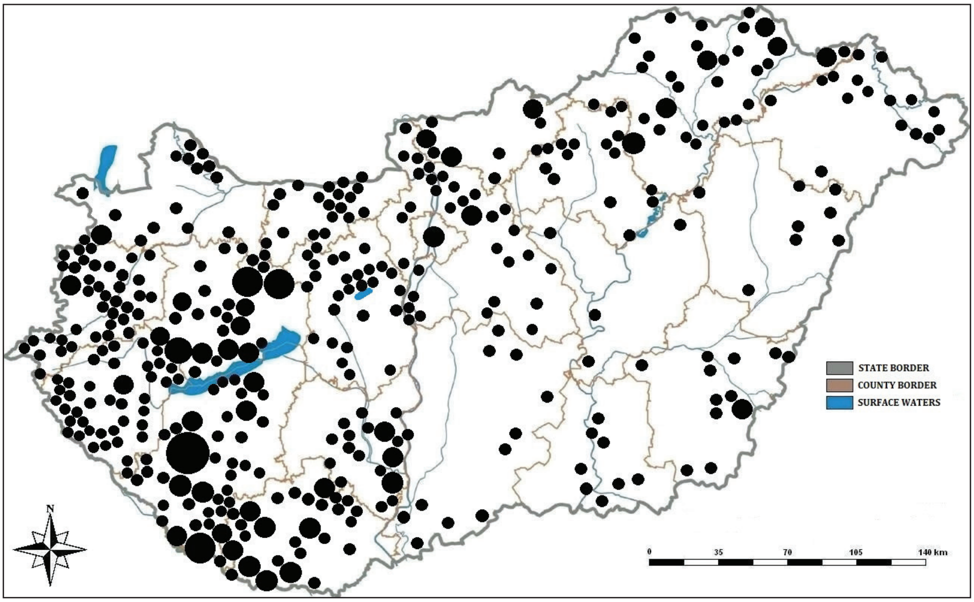

Figure 1. The situation of the observed settlements on the map of Hungary. A small dot indicates one settlement, a medium dot two or three settlements, while large dots refer to at least five settlements situated close to each other.

Table 2. List of the observed species and their main data. N = native, A = adventive [those taxa whose native nature is still under dispute in Hungary (*Abies alba*, *Castanea sativa*, *Juglans regia*, *Quercus frainetto*) and those that are native only in a limited percentage of the country, but occur in a much greater area (*Pinus sylvestris*, *Sorbus domestica*, *Taxus baccata* and *Tilia tomentosa*) are also here (based on Bartha 2000)]; *Agrob.* = *Agrobacterium* infection, S = single tree, F = surrounded by other trees in a park or forest, Comb. = combined infection/damage.

Species	N/A	Min. girth (cm)	Number in the database of Pósfai	Number of measured trees S/F		Polypore		*Agrobact.*		Ivy		Comb.		Measured/ damaged (%)
						S	F	S	F	S	F	S	F	
Abies alba	A	300	6	1	4		1				3			80
Abies cephalonica	A	300	1		1						1			100
Abies numidica	A	300	3		1						1			100
Acer campestre	N	300	68	7	34				1	1	16	1		46.3
Acer negundo	A	300	19	2	7				1	1	2			44.4
Acer platanoides	N	300	37	2	18						7		1	40
Acer pseudoplatanus	N	300	35	2	15							2	3	29.4
Acer saccharinum	A	300	43	5	22		2		1	1	1			18.5
Aesculus flava	A	400	1		1									0
Aesculus hippocastanum	A	400	23	1	13						2		1	21.4
Ailanthus altissima	A	300	11	4	4						1			12.5
Alnus glutinosa	N	300	71		17				2		4		1	41.2
Betula pendula	N	200	25		7				2					28.6
Calocedrus decurrens	A	300	9		7						2			28.6
Carpinus betulus	N	300	171	2	74		6		2		9		3	26.3
Castanea sativa	A	500	96	17	36				3	4	3			18.9
Catalpa bignonioides	A	200	7	1	3						2			50
Cedrus deodora	A	500	1		1									0

Species	N/A	Min. girth (cm)	Number in the database of Pósfai	Number of measured trees S/F		Polypore		*Agrobact.*		Ivy		Comb.		Measured/ damaged (%)
						S	F	S	F	S	F	S	F	
Cedrus libani	A	400	1		1									0
Celtis occidentalis	A	300	55	7	24		1				7			25.8
Corylus colurna	A	300	9	4	4				1					12.5
Crataegus monogyna	N	100	9	1	1									0
Fagus sylvatica	N	400	529	3	397		66		8		26		4	26
Fraxinus angustifolia ssp. *pannonica*	N	400	42	1	12						4		1	38.5
Fraxinus excelsior	N	400	62	6	31		1	1			7		1	27
Ginkgo biloba	A	400	10	2	8				1	1	5			70
Gleditsia triacanthos	A	300	6		2									0
Gymnocladus dioicus	A	300	7		1						1			100
Hedera helix	N	50	3	2	1									0
Juglans nigra	A	300	11	1	6				1					14.3
Juglans regia	A	300	4	1										0
Larix decidua	A	300	8		5						1			20
Liriodendron tulipifera	A	400	11	1	6						2			28.6
Maclura pomifera	A	300	2		2				1		1			100
Magnolia acuminata	A	200	4		3						3			100
Morus alba	A	400	20	9	3					1				8.3
Paulownia tomentosa	A	300	16	1	3					1	1			50
Picea abies	A	300	46	2	20					1	1			9.1
Pinus nigra	A	300	17	3	12						6			40
Pinus sylvestris	A	300	20	1	5						2			33.3
Pinus strobus	A	300	6		1									0
Platanus × acerifolia	A	600	69	11	47			3	4		10	1	1	32.8
Populus alba	N	600	42	4	11				3		2			33.3
Populus × canescens	N	600	33	3	9						5			41.7
Populus nigra	N	600	395	25	174	1	1	8	91	4	5		13	61.8
Prunus avium	N	300	32	4	10				2	2	1			35.7
Pseudotsuga menziesii	A	300	60	1	4		1							20
Pterocaria stenoptera	A	600	2		1									0
Pyrus pyraster	N	300	34	6	16						2			9.1
Quercus cerris	N	500	56	6	23		1		4		1		1	24.1
Quercus frainetto	A	400	2		1						1			100
Quercus petraea	N	500	26	2	9		1				4			45.5
Quercus pubescens	N	400	4	3	1									0
Quercus robur	N	500	573	95	284	1	12	3	22	12	67	2	13	34.8
Quercus rubra	A	400	9		3		1				1			66.7
Robinia pseudoacacia	A	300	58	5	14									0
Salix alba	N	600	179	8	70		2	1	6		2			14.1
Salix caprea	N	200	2		1									0
Sambucus nigra	N	100	5	1	1									0
Sequoiadendron giganteum	A	500	21	2	12				1		3		1	35.7
Sophora japonica	A	400	28	7	14					2	10			57.1
Sorbus domestica	A	200	14	4	1					1				20
Sorbus torminalis	N	200	13		7						1		1	28.6
Taxodium distichum	A	300	41	1	23						9			37.5
Taxus baccata	A	200	36	4	9						2			15.4
Thuja plicata	A	300	7		3									0
Tilia cordata	N	400	73	19	40		1		5	1	9			27.1
Tilia platyphyllos	N	400	78	10	45		1		6		7	2		29.1
Tilia tomentosa	A	400	18	8	6		1				4			35.7
Ulmus glabra	N	400	4		2						1			50
Ulmus laevis	N	400	41	1	25				2		6			30.8
Ulmus minor	N	400	7	1	1						1			50
Total	–	–	**3487**	**320**	**1680**	**2**	**99**	**16**	**170**	**33**	**275**	**8**	**45**	**32.4**

combined infections as well). Only seven specimens of non-native trees were infected by polypores, two of them being silver maple (*Acer saccharinum*).

A total of 170 (91.4%) of those old trees that are infected only by *Agrobacterium* stand in a (relatively) closed forest or park (Table 2). Only 16 solitary trees were damaged by this infection, half of them being black poplar (*Populus nigra*). Including the combined infections as well, *Agrobacterium* was documented on 115 out of 226 measured black poplars (i.e. 51%). Seventeen non-native trees were infected by *Agrobacterium*, most of them (seven) being sycamores.

We found ivy on 33 solitary trees, meaning 10.7%, compared to 275 specimens within stands. Mostly oaks were damaged (101, meaning 28.5%) and ivy appeared on 23.6% of the measured oaks. The highest rate, however, was seen in case of the maple genus, being 31 out of 114 measured maple trees (27.2%) (combined case included for both genera). A total of 101 non-native trees were affected, mostly (in 12 cases) the Japanese pagoda tree (*Sophora japonica*).

Two or three types of damage were documented on 53 old trees. They belong to the worst health category, most of them having almost died. A total of 84.9% (i.e. 45 trees) stand in a group or forest, while only 8 solitary trees are affected by combined infections (Table 2).

In the following text, we present one example per damage category from our database.

Polypores infection

Not infected

- species: common oak (*Quercus robur*)
- locality: Battonya-Tompapuszta (Békés County, SE Hungary)
- habitat: grassland, abandoned farmyard
- girth: 574 cm
- trunk diameter: 1.5 m
- crown diameter: 30 m
- height: 18 m

Branches properly cut in the past; however, irregular fractures are seen lately. The area has been abandoned and unattended for several years. No visible rot, fungi, moss, *Agrobacterium* or ivy (Fig. 2).

Sparsely infected

- species: beech (*Fagus sylvatica*)
- locality: Gadány (Somogy County, SW Hungary)
- habitat: woody area, along a stream
- girth: 552 cm

Figure 2. Not infected *Quercus robur*.

- trunk diameter: 2.5 m
- crown diameter: 50 m
- height: 30 m

Not managed woody area. Sparse polypore infection: one conk of tinder fungus (*Fomes fomentarius*). The conk is healthy, fresh. The tree is in a fair health state besides the polypore, its foliage is lush and healthy and the branches are not dry (Fig. 3).

Slightly infected

- species: Norway maple (*Acer platanoides*)
- locality: Hencse (Somogy County, SW Hungary)
- habitat: previously Palace Park, currently Golf Club
- girth: 497 cm
- trunk diameter: 1.5 m
- crown diameter: 30 m
- height: 17 m

Its locality can be visited with permission. Old yews (*Taxus baccata*) surrounding. Ivy on its trunk, wasps in the hollow of the former branch. More than two fresh, healthy polypore conks on the lower part of the trunk. Although the park is well managed, no attention is paid to this infection (Fig. 4).

Moderately infected

- species: silver fir (*Abies alba*)
- locality: Fehérvárcsurgó (Fejér County, Central Hungary)
- habitat: Palace Park
- girth: 429 cm
- trunk diameter: 1.5 m
- crown diameter: 20 m
- height: 22 m

Several huge veteran trees in the Palace Park. This specimen is probably the oldest silver fir in Hungary (next to the pebble road leading through the park). Slight ivy infection. More than five tiny polypore conks, some of them several years old with dry conk (Fig. 5).

Acutely infected

- species: beech (*Fagus sylvatica*)
- locality: Becsehely (Zala County, W Hungary)

Figure 3. Sparsely infected *Fagus sylvatica*.

- habitat: young beech forest, with some old beech trees
- girth: 487 cm
- trunk diameter: 1.5 m
- crown diameter: 30 m
- height: 27 m

Figure 4. Slightly infected *Acer platanoides*

Figure 5. Moderately infected *Abies alba*.

Extended beech forests in the territory, with several old beech and hornbeam specimens on a plot of relatively-young trees. More than ten polypore conks on the trunk, moreover, the infection has appeared even on the branches. The area is not managed, with significant blackberry (*Rubus fruticosus*) coverage (Fig. 6).

Figure 6. Acutely infected *Fagus sylvatica*.

Polypore infection could be detected in 65 out of the observed 531 settlements, 123 per 2000 trees. This means 12.24% of the studied areas and 6.15% of the measured trees (Fig. 7). Polypores seem to be missing from veteran trees of the Great Hungarian Plain, but we have to add that the main host of the most frequent polypore, i.e. beech (*Fagus sylvatica*), is also missing from the area (although other polypores could occur). Veszprém, Zala, Baranya and Somogy Counties (i.e. South-Western and Central-Western) are moderately infected.

Agrobacterium infection

Not infected

- species: sweet chestnut (*Castanea sativa*)
- locality: Bak (Zala County, W Hungary)
- habitat: between wine cellars, private area
- girth: 606 cm
- trunk diameter: 2 m
- crown diameter: 20 m
- height: 20 m

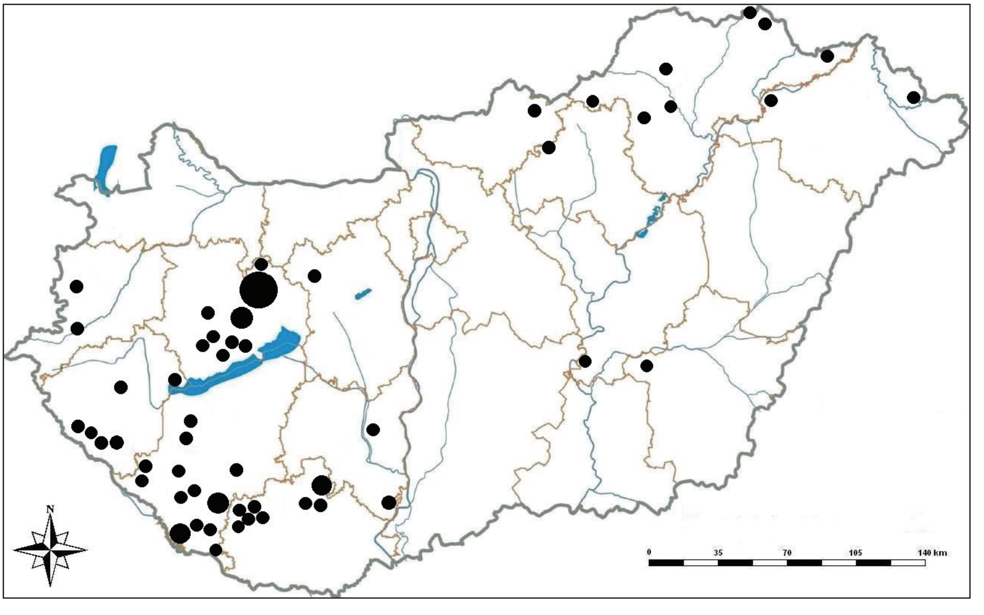

Figure 7. Settlements that host large trees infected by polypores (Hungary). A small dot indicates one settlement, a medium dot two or three settlements, while large dots refer to at least five settlements situated close to each other.

Walnut and chestnut trees were commonly planted around wine yards and cellars during the past centuries. According to its information board, this tree is 400 years old, under nature protection. No *Agrobacterium* and slight ivy infection. Branches cut back, without any rot (Fig. 8).

Figure 8. Not infected *Castanea sativa*.

Sparsely infected

- species: common oak (*Quercus robur*)
- locality: Káld (Vas County, W Hungary)
- habitat: young Scots Pine forest, clearing
- girth: 539 cm
- trunk diameter: 2 m
- crown diameter: 40 m
- height: 22 m

Enormous oak tree in a young Scots Pine stand. A tumour-like gall with 25 cm diameter makes its unique feature, with no other tumours. High coverage of mosses and ants on the tree, dense blackberry around (Fig. 9).

Slightly infected

- species: London plane (*Platanus* × *acerifolia*)
- locality: Pölöske (Zala County, W Hungary)
- habitat: backyard of a family house
- girth: 758 cm
- trunk diameter: 3 m
- crown diameter: 40 m
- height: 28 m

Private area at the end of Dózsa György Street in Pölöske village, officially nature protected, can be visited with permission. Two huge London planes, one is slightly infected with ivy and a tumour-like gall with 40 cm diameter at the junction between the root and the shoot (Fig. 10).

Moderately infected

- species: common oak (*Quercus robur*)
- locality: Nagyrécse (Zala County, W Hungary)
- habitat: young oak forest
- girth: 500 cm
- trunk diameter: 1.5 m
- crown diameter: 35 m
- height: 23 m

Figure 9. Sparsely infected *Quercus robur*.

Figure 10. Slightly infected *Platanus* × *acerifolia*.

Numerous old oaks in a young stand. No tumours on the trunk, but more than ten galls in the crown, none of them exceeding 10 cm in diameter. Several galls had fallen on the ground (Fig. 11).

Figure 11. Moderately infected *Quercus robur*.

Acutely infected

- species: black poplar (*Populus nigra*)
- locality: Tiszacsege (Hajdú-Bihar County, E Hungary)
- habitat: ferry station, holiday houses

- girth: 826 cm
- trunk diameter: 3 m
- crown diameter: 10 m
- height: 11 m

Huge, but highly infected poplar. Its original bark is missing, the whole trunk is covered by tumours, branches are cut back every year due to the galls since its crown is not safe from the infection. Its notable trunk diameter is a consequence of its disease (Fig. 12).

Agrobacterium infection could be detected in 121 places out of the observed 531 settlements, this means 22.79%. Amongst the studied veteran trees, 217 are infected by Agrobacterium, meaning 10.85% (Fig. 13). Considering the age and vulnerability of the trees and the fact that this disease cannot be managed easily, this rate is high. The most infected territories are Zala and Vas Counties (i.e. Western Hungary) and the South-Western region (Somogy and Baranya Counties). A notable result is that Borsod-Abaúj-Zemplén (Northern-Hungary) is almost free from this kind of infection, despite the high number of studied trees.

Ivy damage

Not damaged

- species: black locust (*Robinia pseudoacacia*)
- locality: Bábolna (Komárom-Esztergom County, NW Hungary)
- habitat: Bábolna National Stud area
- girth: 649 cm
- trunk diameter: 2.5 m
- crown diameter: 25 m
- height: 19 m

Hungary's oldest black locust specimen, planted in 1710. Unfavourable health state, but not damaged by ivy or other damage. Branches fixed to the trunk with a belt, trunk filled artificially (Fig. 14).

Sparsely damaged

- species: Hungarian oak (*Quercus frainetto*)
- locality: Deszk (Csongrád County, SE Hungary)
- habitat: hospital park
- girth: 561 cm
- trunk diameter: 1.5 m
- crown diameter: 30 m
- height: 18 m

Figure 12. Acutely infected *Populus nigra*.

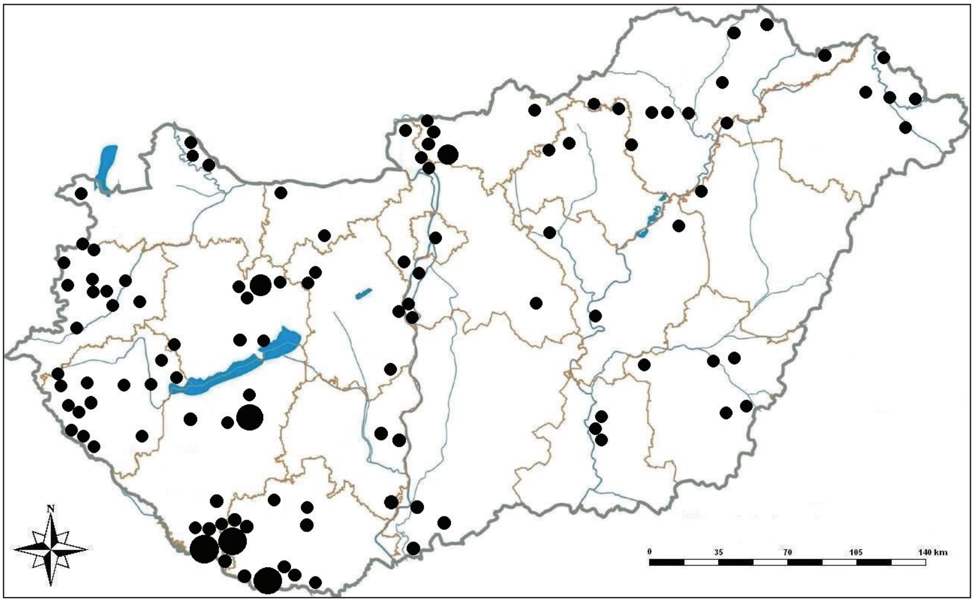

Figure 13. Settlements that host large trees infected by *Agrobacterium* (Hungary). A small dot indicates one settlement, while larger dots refer to more settlements situated close to each other.

Thin ivy sprouts (runners), up to 1.5 m height, with tiny leaves. They are currently not causing any serious injury, but now is the time for protection, as later interventions may not be effective (Fig. 15).

Slightly damaged

- species: large-leaved linden (*Tilia platyphyllos*)
- locality: Pápa (Veszprém County, W Hungary)
- habitat: Castle Park
- girth: 420 cm
- trunk diameter: 1.5 m
- crown diameter: 30 m
- height: 23 m

The central part of the Castle Park is characterised by an old linden tree. Its trunk is densely covered by ivy up to 3 m height. Ivy leaves are still small and fresh and do not overgrow the foliage; thus, its elimination is still possible (Fig. 16).

Figure 14. Not damaged *Robinia pseudoacacia*.

Moderately damaged

- species: common oak (*Quercus robur*)
- locality: Zsennye (Vas County, W Hungary)
- habitat: Palace Park

Figure 15. Sparsely damaged *Quercus frainetto*.

Figure 16. Slightly damaged *Tilia platyphyllos*.

- girth: 535 cm
- trunk diameter: 2 m
- crown diameter: 30 m
- height: 25 m

Numerous notable oaks rule the (not properly managed) park of the Bezerédj Palace in Zsennye, all of them damaged by ivy, even in their foliage. Leaves are great and sprouts form constant foliage, although not reaching the tips. Protection would be very complicated at this stage (Fig. 17).

Acutely damaged

- species: common oak (*Quercus robur*)
- locality: Szőcsénypuszta (Somogy County, SW Hungary)
- habitat: lumberyard
- girth: 573 cm
- trunk diameter: 2 m
- crown diameter: 30 m
- height: 18 m

Hard to see and measure due to enormous ivy mass and the tree has grown on to the fence. The trunk is totally hidden by ivy sprouts and leaves, which reach the tips. Protection is impossible at this stage (Fig. 18).

A total of 157 of the observed 531 settlements and 353 of the measured 2000 trees showed ivy damage, meaning 29.57% and 17.65%, respectively (Fig. 19). The highest rate was detected in Somogy, Zala, Baranya, Veszprém and Vas Counties (i.e. Western and South-Western Hungary). The Northern Hungarian Mountain Range is almost free from ivy damage, despite the high number of samples. North-Western, Eastern and Southern Hungary are also almost free of ivy.

Other damage

Besides those mentioned above, we also noticed some other types of damage. Most of them are general, such as mistletoe species (*Loranthus* sp., *Viscum* sp.), mosses and lichens. Some other, less well-known pests and diseases that we detected on ancient trees are old man's beard (*Clematis vitalba*), wild grapes (e.g. *Parthenocissus inserta*), hedge bindweed (*Calystegia sepium*), dewberries (*Rubus* spp.), epiphyte lichens and mosses, honey fungus (*Armillaria mellea*), pear-shaped puffball (*Lycoperdon pyriforme*), gall wasps (*Cynipidae*), mite that forms the lime nail gall (*Eriophyes tiliae*), chestnut gall wasp (*Dryocosmus kuriphilus*), beech gall midge (*Mikiola fagi*), firebug (*Pyrrhocoris apterus*), horse-chestnut leaf miner (*Cameraria ohridella*), ants (*Formicidae*), European hornet (*Vespa crabro*), woodpeckers (*Picinae*) and several species of games (*Cervus elaphus*, *Sus scrofa*).

Figure 17. Moderately damaged *Quercus robur*.

Figure 18. Acutely damaged *Quercus robur*.

A total of 51.22% of the observed 272 settlements host ancient trees that are damaged by one or some of the above listed damage-causing organisms. A total of 31.70% of the measured trees (i.e. 634 trees) are damaged by one or combined organisms (Fig. 20). Veteran trees in the Great Hungarian Plain are almost free of damage,

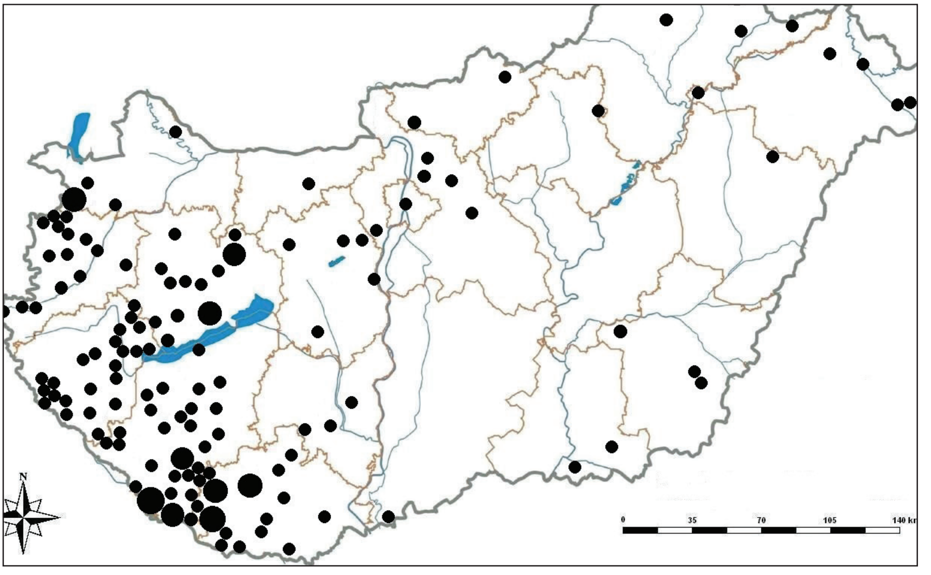

Figure 19. Settlements that host large trees damaged by ivy (Hungary). A small dot indicates one settlement, while larger dots refer to more settlements situated close to each other.

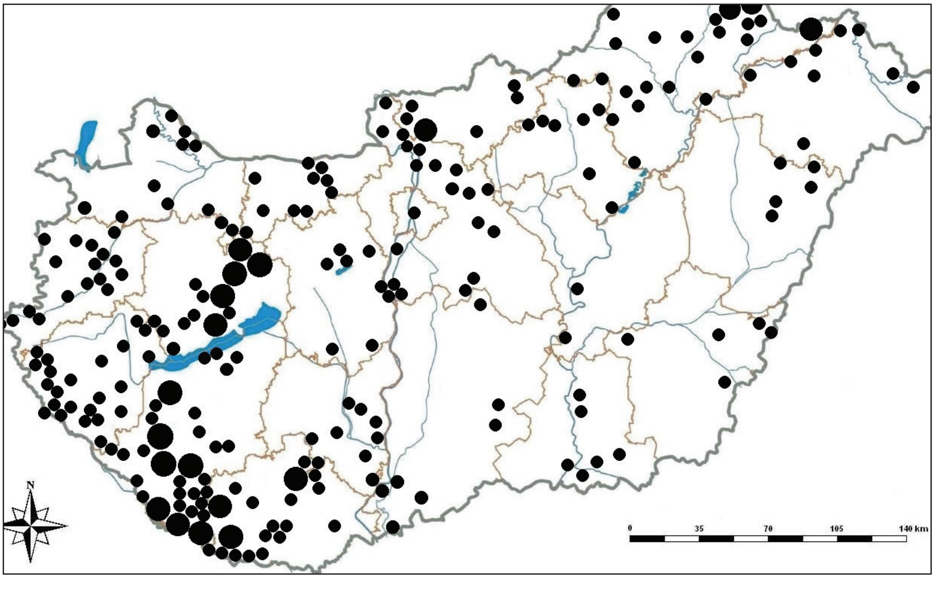

Figure 20. Settlements that host large trees suffered by other types of damage (Hungary). A small dot indicates one settlement, while larger dots refer to more settlements situated close to each other.

probably due to the low sample size. The most damaged regions are Somogy, Baranya and Veszprém Counties (i.e. Western, South-Western Hungary), each with a large sample size.

Health status

We measured the health status of 7 trees with a Fakopp Arborsonic 3D Acoustic Tomograph in 2012, at different layers and heights (Table 3). Different shapes (roller trunk or fork shape) and measurability (sprout, shrub) justified different measurement methods.

We present one example per decay status category from our database.

Slight decay status

- species: black walnut (*Juglans nigra*)
- locality: Martonvásár (Fejer County, C Hungary)
- habitat: little island in the middle of the castle park
- girth: 444 cm
- trunk diameter: 1.5 m
- crown diameter: 25 m
- height: 25 m

Sensors were placed at 30 and 130 cm height (i.e. two layers). Measured decay: 17 and 2%, respectively. The average decay level of this black walnut is 9.5%, i.e. very low (Fig. 21).

Medium decay status

- species: wild pear (*Pyrus pyraster*)
- locality: Gödöllő (Pest County, C Hungary)
- habitat: Botanical Garden
- girth: 317 cm
- trunk diameter: 1 m
- crown diameter: 15 m
- height: 12 m

Sensors were placed at 5 heights: 30, 70, 110, 150 and 190 cm. Measured decay at the different layers: 57, 41, 25, 26 and 42%. The average decay level of this wild pear is 38.2%, i.e. medium level (Fig. 22).

Strong decay status

- species: Turkish hazel (*Corylus colurna*)
- locality: Gyöngyös (Heves County, N Hungary)
- habitat: Castle Park

Table 3. Decay status of trees measured with a tomograph.

Locality	Species	Girth (CM)	Number of layers	Decay status per layers	Status/ Average (%)
Csokonyavisonta	(%)	400	2	1: 2; 2: 30	16 (slight)
Csokonyavisonta	*Pyrus pyraster*	322	2	1: 41; 2: 39	40 (medium)
Gödöllő	*Pyrus pyraster*	317	5	1: 57; 2: 41; 3: 25; 4: 26; 5: 42	38.2 (medium)
Gyöngyös	*Corylus colurna*	426	2	1: 69; 2: 70	69.5 (strong)
Kaposvár-Kaposfüred	*Pyrus pyraster*	321	2	1: 54; 2: 53	53.5 (strong)
Martonvásár	*Juglans nigra*	444	2	1:17; 2: 2	9.5 (slight)
Túristvándi	*Pyrus pyraster*	329	2	1: 72; 2: 76	74 (strong)

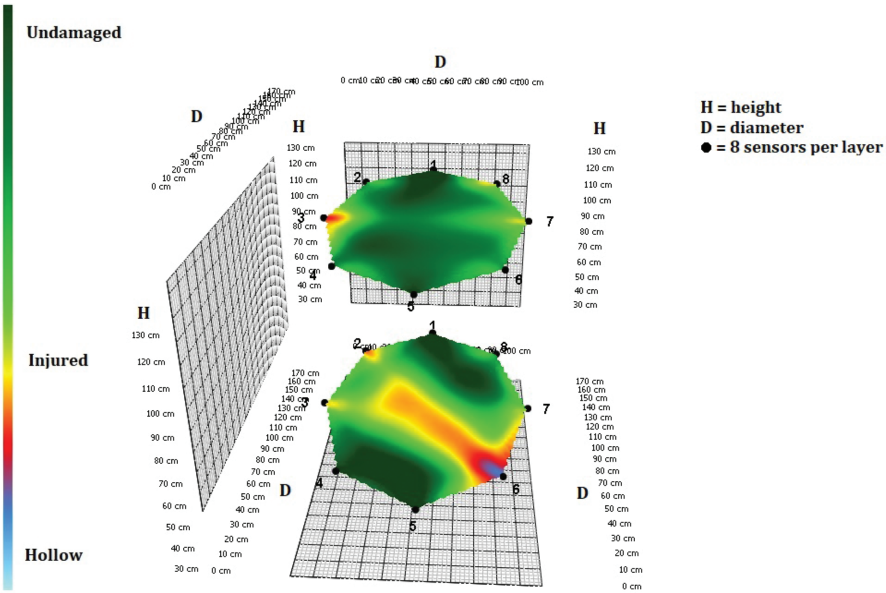

Figure 21. Three-dimensional computer model of internal tree density of the black walnut in Martonvásár, Hungary (at two different heights).

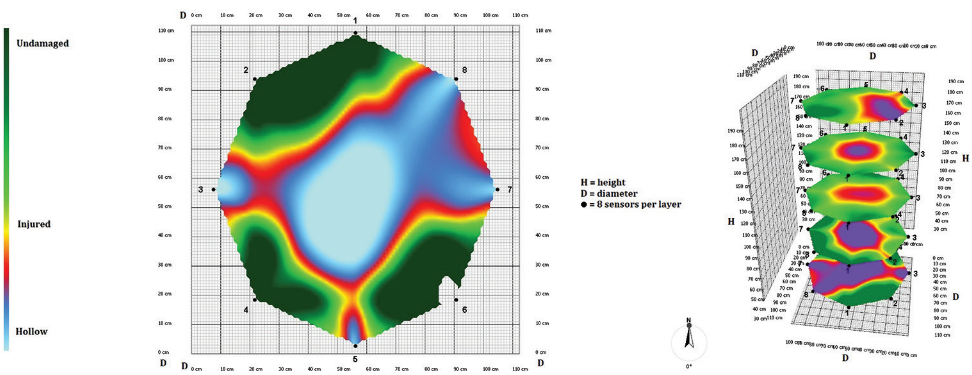

Figure 22. Three-dimensional computer model of internal tree density of the old wild pear in Gödöllő, Hungary (left: layer at 30 cm height; right: 3D-image at 5 different heights).

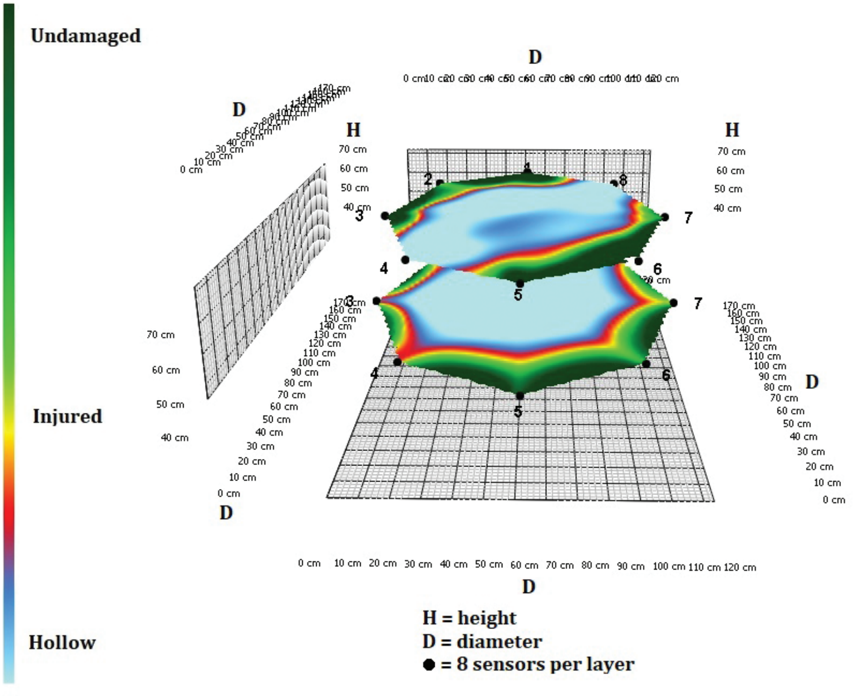

Figure 23. Three-dimensional computer model of internal tree density of the Turkish hazel in Gyöngyös, Hungary (at two different heights).

- girth: 426 cm
- trunk diameter: 1.5 m
- crown diameter: 20 m
- height: 13 m

Sensors were placed at 35 and 70 cm height (i.e. two layers). Measured decay: 69 and 70%, respectively. The average decay level of this Turkish hazel is 69.5%, i.e. considerably high (Fig. 23).

Conclusions

The number of veteran trees in the online database of Pósfai (Pósfai Gy 2019) is constantly growing (from 700 in 2008 onto 3,500 in 2020) and this reflects the popularity of the large trees, emerging citizen science activities and is a consequence of our digital word as well. From this perspective, we should mention the European Tree of the Year competition as well. This programme started in 2002 and Hungarian trees have been involved since 2010. The emotional value, i.e., the role of the tree in the everyday life of the local community plays an essential role in this competition. Most of the Hungarian winners are sizable trees and thus, we have measured them as well.

Based on our results, we state that one third of the observed ancient trees (648 out of 2,000) suffer from polypores, *Agrobacterium* or ivy.

Polypores were detected on altogether 16 species. The well-known *Agrobacterium* that usually infects poplars was found on 23 different species. Ivy was documented from 56 tree species.

Amongst the observed 72 tree species, beech was the most infected by polypores. Seventy out of 400 (17.5%) measured beech trees were damaged by polypores and 56.7% of all documented damage affected the beech species. Our map suggests that the damage caused by polypores is not more remarkable than other pests and diseases. However, when we add the other fungal species (honey fungus, pear-shaped puffball etc.), then this taxonomic group (i.e. fungi tribes) significantly affect the state of the Hungarian ancient trees. Protection against polypores is not successful in Hungary (as in other countries). Literature sources on the description of polypores in Hungary (Gerhardt 2017, Szabó 2003, Vasas and Locsmándi 2010) mention almost every tree genus that we found to be infected by polypores, except for the *Celtis*, the *Pseudotsuga* and the *Sequoiadendron* genera, on which we found these infections as well. However, as these sources usually write generally about exotic trees in parks and gardens, the infections found on the mentioned three genera cannot be evaluated as new scientific results.

Altogether, 217 trees were infected by *Agrobacterium* species. Not surprisingly, mostly poplars suffered from these bacteria, 51% of the observed specimens (115 out of 226); while 53% of all the documented injuries belong to this pest. Hungarian literature (Szabó 2003) mentions basically the poplars in case of old trees, but still we found the bacteria on altogether 23 species. There are no efforts in order to protect the ancient trees against *Agrobacterium*, only in case of young trees in tree nurseries (Dreistadt 2001). Its only benefit (besides its use in gene techniques) is for the timber industry, that the cutting of a processed tumour-like growth or gall from a poplar is beautiful in an ornamental sense (Thomas and Schumann 1993).

Ivy was found on 353 trees, 101 of them are oaks (28.5%). Thirty one of the 114 observed maples were damaged by ivy (27.2%). From a nature conservation aspect, it is harder to evaluate ivy as absolutely negative (such as in case of polypores or *Agrobacterium*). Although it harms the tree with shading, it might host nesting birds and carries aesthetic value as well, especially in parks and palace gardens. Protection against ivy usually means cutting its shoots to dry out, but the dead biomass remains on the tree.

The measured diseases and damage-causing organisms usually attack those trees that are surrounded by other trees, ensuring a good chance for pest reproduction. Almost no solitary trees were damaged amongst the ornamental non-native species, since their old specimens usually appear in parks and arboretums, surrounded by other trees. However, it is obvious, even in case of the oaks, maples and wild pear (i.e. those species that usually stand as solitary), that the presented damage mostly appears in denser stands.

We measured 29 native and 43 non-native tree species. A total of 1550 out of 2000 measured specimens are native. All the three damage types were mostly documented on native trees, but the thorough rate of damaged trees is about the same in case of native and adventive species. A total of 33.5% of the native specimens (519 out of 1550) and 28.7% of the adventives (129 trees out of 450) are damaged by any (or more than one) of the mentioned infections or ivy. However, we have to add that, in case of ivy (and without combined infection), 101 out of 308 trees are non-native. This phenomenon can be explained by the fact that ivy appears mostly in arboretums, castle and mansion parks and town parks, where the highest rate of non-native, ornamental trees are planted. Preventive measures are well-known in the case of polypores and ivy. To protect old trees against polypores, we should avoid scars on the bark. When a forested plot is harvested, a few monumental trees are usually left (e.g. in order to renew the forest with their acorns). These specimens face a lot more pests and altered environmental conditions (e.g. stagnating water in the scars and hollows serves the appearance of pests). Ivy damage can be avoided by eliminating its young sprouts and shoots from the trunk. Unfortunately, ivy is usually realised only in a later stage, when it starts to overgrow the foliage. Tree protection is almost impossible at this stage. In case of *Agrobacterium*, prevention means the use of healthy products from the nursery garden, but this is obviously not relevant for greater trees.

In order to measure the health status, we could use the Fakopp Arborsonic 3D Acoustic Tomograph only for a couple of trees due to its weight (10 kg including laptop) and time-consuming assemblage (e.g. placing its sensors on the tree). Its main defect is that it cannot be used for trees above 450–500 cm girth (cables are too short) and thus, we could not measure the decay status of the observed oaks, willows, poplars and sycamores. Thus, we state that the Fakopp 3D Tomograph is suitable for measuring the health status of the old trees as well, but only in case of those species that do not exceed 500 cm girth, such as wild pear, maples and hornbeam.

It can be stated that the old trees are usually not covered by nature protection areas in Hungary, since most of them stand in isolated, hardly reachable places or in the middle of pastures and meadows, sometimes croplands. Lack of protection is mainly the result of economic factors, as their management, pest protection etc. is not economically viable for the owners, foresters or park gardeners; other reasons may lie in the lack of adequate knowledge or missing responsibilities. There is no case that one tree specimen is protected at national level, although there is an alley that is protected via IUCN IV category (nature conservation area) on its own. However, some local municipalities protect one or more concrete ancient trees in their own municipal decrees, based on the right given to them by the Hungarian Nature Conservation Act, referring to IUCN category III (natural monument). Some examples for local-level protection are the oaks in Kétújfalu, the Turkish hazel in Pécs, the giant lime tree in Szőkedencs, the '1000-year-old oak' in Zsennye or the sweet chestnuts in Surd (Nature conservation areas under local protection in Hungary 2018). Some trees that we measured stand in castle (or mansion) parks that are protected at national (e.g. Alcsútdoboz and Gödöllő) or local level (e.g. Lengyel and Sellye). That type of protection does not mean pest control, but it can help draw attention to preserving the condition of those trees.

We recommend preserving the state of the current trees, even if it is hard to improve. One should improve their protection against the main pests with preventative measures and, if possible, during the first and second damage level. In case of more severe damage, there is practically no solution. In case of *Agrobacterium* infecting the tree, we are already late even when realising the first infection level.

Protection against the presented pests and diseases is very complicated and, in practice, almost impossible. The Hungarian practice shows that these types of damage remain without any management or healing due to lack of time, financial background or energy, but mostly because the only effective solution would be prevention.

While the age of trees is generally not a precondition to being emotionally important for the local community, many of the trees documented are in fact amongst the country's oldest. We, therefore, concluded that old-growth assets have a considerable intrinsic worth that can and should be valued.

Only a very small proportion of the greatest Hungarian trees are covered by local nature protection and even some of the protected ones are close to death. The main causes of this negative phenomenon are the lack of caring or management, the presented pests and diseases or environmental factors, such as storm, wind or frost damage.

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References

Andrési R, Tuba K (2018) Comparative study of the *Fomes fomentarius* and *Trametes gibbosa* beetle communities in Hidegvíz Valley, Sopron Mts., Hungary. Community Ecology 19(2): 141–147. https://doi.org/10.1556/168.2018.19.2.6

Arciniegas A, Prieto F, Brancheriau L, Lasaygues P (2014) Literature review of acoustic and ultrasonic tomography in standing trees. Trees (Berlin) 28(6): 1559–1567. https://doi.org/10.1007/s00468-014-1062-6

Atherly AG (2004) *Agrobacterium*, *Rhizobium*, and other gram-negative soil bacteria. Transformation of Plants and Soil Microorganisms 3: 23–33. https://doi.org/10.1017/CBO9780511752261.005

Bartha D (2000) Black List. Adventive Trees and Shrubs in Hungary. Published by the author. Sopron, 31 pp.

Bartosiewicz A, Siewniak M (1979) Öreg Fák, Díszfák Ápolása. Mezőgazdasági Kiadó, Budapest, 182 pp.

Baum S, Sieber T, Schwarze F, Fink S (2003) Latent infections of *Fomes fomentarius* in the xylem of European beech (*Fagus sylvatica*). Mycological Progress 2(2): 141–148. https://doi.org/10.1007/s11557-006-0052-5

Bergman KO, Jansson N, Claesson K, Palmer MW, Milberg P (2012) How much and at what scale? Multiscale analyses as decision support for conservation of saproxylic oak beetles. Forest Ecology and Management 265: 133–141. https://doi.org/10.1016/j.foreco.2011.10.030

Buse J, Levanony T, Timm A, Dayan T, Assmann T (2010) Saproxylic beetle assemblages in the Mediterranean region: Impact of forest management on richness and structure. Forest Ecology and Management 259(8): 1376–1384. https://doi.org/10.1016/j.foreco.2010.01.004

Castagneri D, Garbarino M, Nola P (2013) Host preference and growth patterns of ivy (Hedera helix L.) in a temperate alluvial forest. Plant Ecology 214(1): 1–9. https://doi.org/10.1007/s11258-012-0130-5

Dreistadt SH (2001) Integrated Pest Management for Floriculture and Nurseries. University of California, 102–104.

English Heritage (2010) Ivy on Walls. Seminar Reports, 62 pp.

Eriksson O (2018) What is biological cultural heritage and why should we care about it? An example from Swedish rural landscapes and forests. Nature Conservation 28: 1–32. https://doi.org/10.3897/natureconservation.28.25067

Garfi G, Ficarrotta S (2003) Influence of ivy (Hedera helix L.) on the growth of downy oak (Quercus pubescens sl) in the Monte Carcaci nature reserve (central-western Sicily). Ecologia Mediterranea: Revue internationale d'écologie méditerranéenne. International Journal of Mediterranean Ecology 29(1): 5–14. https://doi.org/10.3406/ecmed.2003.1524

Gencsi L, Vancsura R (1997) Dendrológia. Mezőgazda Kiadó, Budapest, 223–226.

Gerhardt E (2017) Gombászok Kézikönyve. Cser Kiadó, Budapest, 720 pp.

Gibbons P, Lindenmayer DB, Fischer J, Manning AD, Weinberg A, Seddon J, Ryan P, Barrett G (2008) The future of scattered trees in agricultural landscapes. Conservation Biology 22(5): 1309–1319. https://doi.org/10.1111/j.1523-1739.2008.00997.x

Glits M, Folk Gy (2000) Kertészeti Növénykórtan. 3., Átdolgozott és Bővített Kiadás. Mezőgazda Kiadó, Budapest, 582 pp.

Hartel T, Plieninger T (2014) European wood pastures in transition. A social-ecological approach. Routledge, New York, 322 pp. https://doi.org/10.4324/9780203797082

Hungarian Monumental Trees (2019) http://oregfak.emk.nyme.hu [Accessed on 11.10.2019]

Igmándy Z (1991) A Magyar Erdők Taplógombái. Akadémiai Kiadó, Budapest, 112 pp.

Iváncsics V, Filepné KK (2019) Assessment methodology of green infrastructure – the case of Keszthely town. Hungarian Journal of Landscape Ecology 17(2): 193–208.

Lindenmayer DB, Laurance WF, Franklin JF, Likens GE, Banks SC, Blanchard W, Gibbons P, Ikin K, Blair D, McBurney L, Manning AD, Stein JAR (2014) New policies for old trees: Averting a global crisis in a keystone ecological structure. Conservation Letters 7(1): 61–69. https://doi.org/10.1111/conl.12013

Lonsdale D (2013) The recognition of functional units as an aid to tree management, with particular reference to veteran trees. Arboricultural Journal 35(4): 188–201. https://doi.org/10.1080/03071375.2013.883214

Manning AD, Fisher J, Lindenmayer DB (2006) Scattered trees are keystone structures – implications for conservation. Biological Conservation 132(3): 11–321. https://doi.org/10.1016/j.biocon.2006.04.023

Monumental Trees (2019) http://monumentaltrees.com [Accessed on 10.09.2019]

Moose RA, Schigel D, Kirby LJ, Shumskaya M (2019) Dead wood fungi in North America: An insight into research and conservation potential. Nature Conservation 32: 1–17. https://doi.org/10.3897/natureconservation.32.30875

Nature conservation areas under local protection in Hungary (2018) http://www.termeszetvedelem.hu/helyi-jelentosegu-vedett-termeszeti-teruletek [Accessed on 11.20.2018]

Pósfai Gy (2019) Thickest trees of Hungary http://www.dendromania.hu/index.php?old=foold [Accessed on 18.12.2019]

Ranius T, Jansson N (2000) The influence of forest regrowth, original canopy cover and tree size on saproxylic beetles associated with old oaks. Biological Conservation 95(1): 85–94. https://doi.org/10.1016/S0006-3207(00)00007-0

Rapaics R (1929) Öreg fák, ősi legendák. Természettudományi Közlöny 61: 721–735.

Read H (2000) Veteran Trees: a Guide to Good Management, 169 pp. http://publications.naturalengland.org.uk/publication/75035

Sverdrup-Thygeson A, Skarpaas O, Odegaard F (2010) Hollow oaks and beetle conservation: The significance of the surroundings. Biodiversity and Conservation 19(3): 837–852. https://doi.org/10.1007/s10531-009-9739-7

Szabó I (2003) Erdei Fák Betegségei. Szaktudás Kiadó Ház Zrt, Budapest, 316 pp.

Szakonyi Z Sz (2018) An overview on the cult of oak genus and presentation of some remarkable Hungarian oak trees. Hungarian Journal of Landscape Ecology 16(1): 35–43.

Takács M, Malatinszky Á (2012) An overview on the cult of sweet chestnut and presentation of the greatest Hungarian sweet chestnut trees. Hungarian Journal of Landscape Ecology 10(2): 457–466.

Takács M, Mravcsik Z, Malatinszky Á (2015) Legendary lime trees of the Carpathian Basin. Annals of Faculty of Engineering Hunedoara – International Journal of Engineering 13(1): 29–32.

Tardy J (1996) Magyarországi Települések Védett Természeti Értékei. Mezőgazda Kiadó, Budapest, 665 pp.

Thomas MG, Schumann DR (1993) Income Opportunities in Special Forest Products. Agriculture Information Bulletin U.S. Department of Agriculture, Washington, 206 pp.

Thompson KET, Bankoff RJ, Louis Jr EE, Perry GH (2016) Deadwood structural properties may influence Aye-Aye (*Daubentonia madagascariensis*) Extractive foraging behavior. International Journal of Primatology 37(2): 281–295. https://doi.org/10.1007/s10764-016-9901-5

Vasas G, Locsmándi Cs (2010) Gyakoribb Gombáink. Műszaki Kiadó, Budapest, 204 pp.

Zapponi L, Mazza G, Farina A, Fedrigoli L, Mazzocchi F, Roversi PF, Sabbatini Peverieri G, Mason F (2017) The role of monumental trees for the preservation of saproxylic biodiversity: re-thinking their management in cultural landscapes. In: Campanaro A, Hardersen S, Sabbatini Peverieri G, Carpaneto GM (Eds) Monitoring of saproxylic beetles and other insects protected in the European Union. Nature Conservation 19: 231–243. https://doi.org/10.3897/natureconservation.19.12464

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RESEARCH ARTICLE

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Revised criteria system for a national assessment of threatened habitats in Germany

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Abstract

The Red List of threatened habitat types in Germany was first published in 1994 and it is updated approximately every ten years. In 2017 the third version was published by the German Federal Agency for Nature Conservation. In the course of the revision, the criteria system was also extended. In doing so, an attempt was made to find a compromise between the consideration of international developments that had taken place and existing national requirements. In particular, short-term developments should become visible through the German Red List status. In addition to 'National long-term Threat', the valuation now also includes 'Current Trend' and 'Rarity'. Following the IUCN's approach, the collapse risk is now represented on the basis of several criteria. However, in contrast to the IUCN procedure, where the worst evaluated criterion is determinative for Red List status, in our procedure all criteria are included in the evaluation. To counteract misleading signal-effects for management decisions, all significant criteria have an influence on the resulting German Red List status (RLG). They are combined in an assessment scheme. In order to map the overall risk of loss, both the long-term threat as a historical reference value and furthermore the current trend must have an influence on RLG. As a result, 65% of habitat types have differing risk of loss.

Keywords

collapse risk, criteria, ecosystem, endangered habitats, nature conservation, risk of loss

Introduction

The protection of biotopes, which aims to preserve a habitat and its complete biocenosis, has become a core instrument of nature conservation in Europe since the 1970s (e.g. Erz 1971; Blab 1984; Kaule 1986; Blab et al. 1993, 1995; Riecken et al. 1994; Essl et al. 2002). Global standards of Red List categories and criteria for ecosystems have recently been proposed by the International Union for Conservation of Nature (IUCN) (Bland et al. 2017). In Europe, Red Lists of biotopes/habitats (on the varied use of the terms 'biotope', 'habitat' and 'ecosystem' see chapter 'Terms and basic concepts') have a noteworthy tradition in several countries (for a comprehensive overview see Rodwell et al. 2013; Savio and Gaudillat 2015; Finck et al. 2017; IUCN 2019). Several European countries have developed distinct national specific assessment systems (e.g. Riecken et al. 1994, 2006; Essl et al. 2002; Dimopoulos et al. 2005; Doniţă et al. 2005; Petrella et al. 2005; Raunio et al. 2008; Härtel et al. 2009; Essl and Egger 2010; Biserkov et al. 2015; Finck et al. 2017).

Consequently, recently developed national approaches had to find a balance between national specific requirements and international comparability (e.g. Delarze et al. 2015, 2016; Finck et al. 2017; Kontula and Raunio 2019). Therefore, instead of exclusively assessing the long-term threat to habitats, the most recently published Red Lists assess different symptoms of the overall 'ecosystem collapse risk' (cf. Bland et al. 2017, 2018). The evaluation of the 'collapse risk' requires the assessment of the condition of an ecosystem type over different time periods – from historical to future trend. Furthermore, the analysis has to consider both reduction in area and in ecosystem quality. The European Commission has funded a comprehensive project to develop a 'European Red List of Habitats' (Gubbay et al. 2016; Janssen et al. 2016) which is based on the IUCN approach. However, Gubbay et al. (2016) and Janssen et al. (2016) also had to allow for European-specific modifications in the application of the IUCN criteria. Therefore, for example, the criteria which assess functional symptoms (degradation of ecological processes: Criteria C/D) have been combined in this project because it has been impossible to separate biotic and abiotic degradation processes.

In 2017, a third updated edition of the 'German Red List of threatened habitats' was published (Finck et al. 2017). The evaluation system was revised in the course of the new edition. The following main considerations were taken into account: (1) the new criteria and categories should clearly relate to those used in previous editions and thereby allow for comparisons to earlier editions (Riecken et al. 1994, 2006); (2) as far as they are also relevant for habitats, existing updated national standards for Red List assessments of species (Ludwig et al. 2009) should be considered (e.g. assessment scheme, consideration of different time frames, and specific risk factors like rarity); (3) and, as many aspects as possible should be taken into consideration from both the IUCN concept (Keith et al. 2013; Bland et al. 2017) and the approaches currently used for the European Red List of Habitats (Gubbay et al. 2016; Janssen et al. 2016).

Red Lists of habitats are characterised by their direct spatial reference and are therefore explicitly focused on landscape planning and actors in the field of habitat management and practical nature conservation. In Germany, the Red List also serves as a technical basis for legal biotope protection. Therefore, the Red List status by itself should indicate current needs for action and also success in nature conservation, thus functioning as a basis for political decisions which concern the prioritisation of nature conservation measures (Gigante et al. 2018; Alaniz et al. 2019). Taking into account the above cited considerations, the criteria system of the German Red List has been considerably revised to indicate the threat to habitat types in Germany under current national threat conditions. Accordingly, an evaluation scheme was developed in which the short-term trend also has a clear influence on the resulting Red List status.

The objective of this paper is to present the recently revised assessment procedure for habitat types in Germany and to contribute to further discussion of an appropriate method for Red List assessment for habitats (see also Janssen et al. 2016).

Terms and basic concepts

Blab et al. (1995) defined a 'biotope type' as an idealised type, derived from similar biotopes in the field, having specific ecological, unique, and more or less constant environmental conditions for animal and plant life. For practical use, the definition is restricted to a certain minimum size, which can still be mapped in the field. The IUCN uses the term 'ecosystem' as a classification unit. The definition of the country-specific terms 'habitat' or 'biotope' used in Europe includes both biotic and abiotic elements, as well as ecological and spatio-functional interactions (see Riecken et al. 1994, 2006) and is therefore comparable to the definition of an 'ecosystem' (e.g. Bland et al. 2017). Following the common usage in other European countries, we will hereafter use the term 'habitat' instead of 'biotope', which is actually the common expression in Germany and functions as an applied mapping unit (cf. Rodwell et al. 2013).

The Red List assessment is based on a complete standard list of habitat types occurring in Germany (Riecken et al. 2003). This list covers the entire range of the German landscape – pristine (cf. BfN 2010), technical (e.g. buildings and transport infrastructure) and cultural habitat types. All these types partly represent the biodiversity of the cultural European landscape (cf. Agnoletti and Rotherham 2015). Only minor modifications of the standard list have been introduced for inland habitats in the third edition to consider advanced knowledge. However, for the marine standard habitat list a complete revision was necessary following new international standards (HELCOM 2013; Finck et al. 2017).

The German Red List of habitats is revised in an approximately ten-year-evaluation cycle. Experience has shown that sufficient monitoring data are available from the federal states within this period. In addition, improvements and deteriorations in the state of conservation can be observed within this period as a result of current risk factors. As proposed by the IUCN (Keith et al. 2015), our assessment system evaluates the overall 'risk of loss' of ecosystems, which manifests itself in the collapse of ecosystems.

In the third edition of the German Red List, we derive the 'German Red List status' (RLG) by combining three different criteria. Since the publication of the first edition of the Red List in 1994, the criteria system for assessing the overall risk has been continuously enhanced. This development is justified in many ways, including by improved knowledge, a better data basis, and new international standards. In earlier editions, only the long-term trend with information on changes in area and quality was included in the overall assessment (Riecken et al. 1994, 2006). The current trend was introduced in 2006 as additional information; the rarity was first assigned as a Red list category (cf. Riecken et al. 2006). In the latest edition, the 'Current Trend' (T) and the 'Rarity' (R) were introduced as further criteria which can positively or negatively influence the degree of endangerment of a habitat based on the 'National Long-term Threat'(nTH). By taking into account these two new criteria (which represent habitat conditions within time windows of the recent past, present and near future), current successes and negative developments are now directly represented by RLG (Fig. 1).

Categories are specified by verbal-descriptive definitions since evaluations for several habitats are still based on expert judgement. There was a broad national consensus that it is not possible to exclusively derive individual threat categories from quantitative values as proposed by Rodríguez et al. (2011, 2015). However, even for well-known habitat types the available knowledge is far from sufficient to compile all the required quantitative data. It remains to be seen whether these deficiencies in data

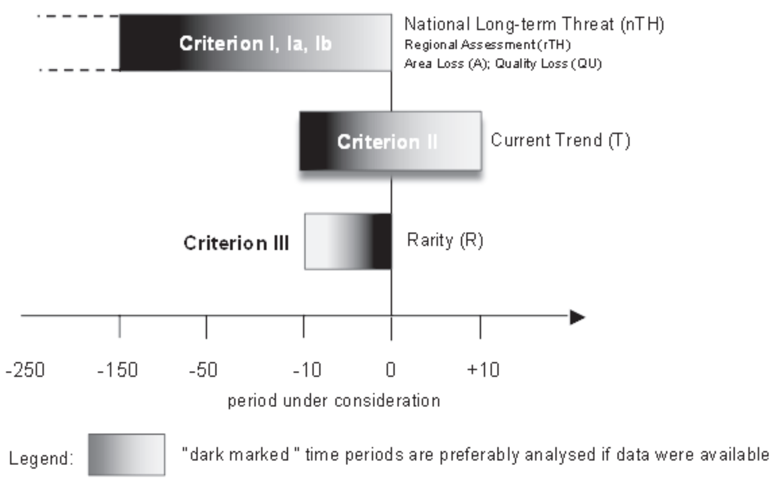

Figure 1. Time frames of the three red listing criteria of the German Red List (Finck et al. 2017). For the long-term evaluation (nTH), mainly anthropogenic spatial (sub criterion AL) and qualitative (sub criterion QUL) changes over the last 50–150 years (sliding time frame) are assessed for the major regional landscape units. The estimation of the 'Current Trend' (T) is based on development over the last ten years and a forecast for the near future (maximum ten years). A higher risk of loss is basically assumed for habitat types which are 'Extremely Rare' at present (R). The latter includes both 'natural' rarity as well as rarity as a result of human impact.

can be resolved in the future, and whether quantitative data can then also be used as a basis in the national Red List of threatened habitats. Verbal descriptive categories can be particularly useful for countries for which complete quantitative information on the occurrence of habitats is not available.

Methods of Red List-assessment for threatened habitats in Germany

Evaluation procedure

To counteract misleading signal-effects for management decisions, we established a mechanism in the assessment procedure to ensure that all significant criteria have an influence on the resulting RLG. Thus, RLG is determined by a step-by-step evaluation procedure (Fig. 2).

Regional assessment: Regional Long-term Threat

For the long-term risk assessment mainly anthropogenic spatial (sub criterion AL) and qualitative (sub criterion QUL) changes over the last 50–150 years (sliding time frame) are assessed for the major landscape regions (see Fig. 3). For this purpose, the

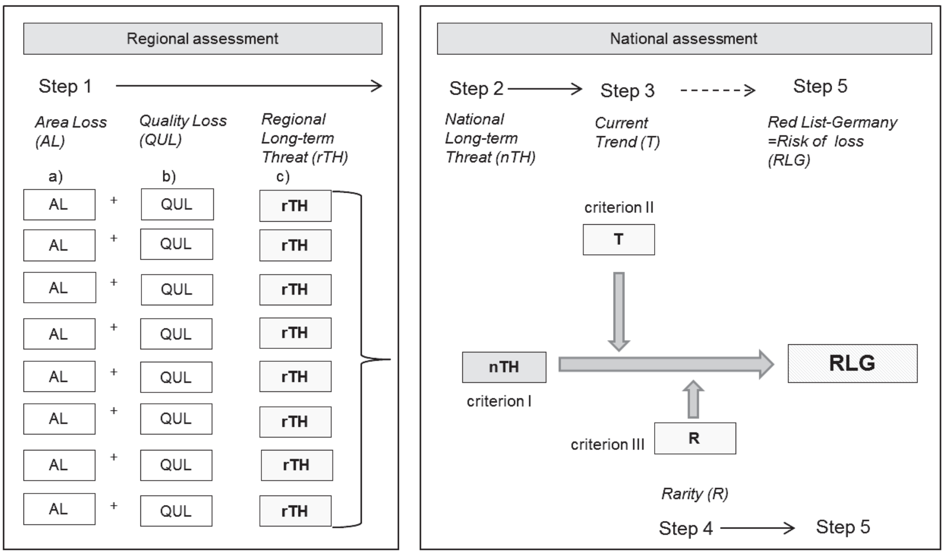

Figure 2. Stepwise Red List assessment for habitat types in Germany. The 'National Long-term Threat' (nTH) is derived from the 'Regional Long-term Threats' (rTH) of eight major landscape regions (Step 1, 2) (see Fig. 3). After that, the degree of endangerment is upgraded or devalued consecutively, first by applying criterion T (Step 3) and then criterion R (Step 4). RLG represents the overall 'Risk of loss' (Step 5) (cf. BfN 2017).

time period between 1850 and 1950 is set as the reference. In most cases, an earlier reference stage cannot be used due to insufficient data. Hence, the considered reference period does not represent the pristine stage of nature as still existed in the Middle Ages in greater parts of Europe. Specific to the habitat, the initial phase of industrialisation (~1850) or rather the situation before the massive intensification in agriculture after the Second World War started (~1950) was chosen.

A similar reference period for the assessment of the long-term threat in Germany is used by Ludwig et al. (2009) for species, and also in Red Lists of habitats from several German federal states which were used as data sources (e.g. Buder and Uhlemann 2010; Von Hengel and Westhus 2011; Zimmermann et al. 2011; Von Drachenfels 2012). The IUCN uses an earlier reference period for the long-term trend; here the relative changes since 1750 are considered (Bland et al. 2017).

For each of the defined eight major landscape regions (see Fig. 3) a risk assessment is performed with regard to the two sub-criteria AL and QUL. Subsequently, consolidation of these sub-criteria into the 'Regional Long-term Threat' (rTH) is carried out (see Fig. 2, step 1). Following the 'precautionary principle', the highest risk category obtained by any of the two sub-criteria is used as the overall rTH. The verbal-descriptive definitions for the categories of the sub criteria AL and QUL, and the overall categories for rTH, are presented in Table 1 as they also correspond to criterion I (nTH), which only differs in spatial scale of assessment.

The sub-criterion AL represents the estimated long-term loss in area of occupancy and the decline in number of sites of habitats (by demolition, building activities, changes in land use, etc.). AL has been described in detail by Blab et al. (1995) (here criterion I). Hereby, the historical ideal condition that belongs to a habitat concerning total area and site density is used as a hypothetical reference to assess threats. In fact, this ideal situation currently rarely exists for any habitat type and can only be described in approximation. In some well-documented cases (e.g. bogs, heathland, ponds, hedges, and unmodified running waters) precise data for the net loss of area are available over a longer period. However, in most cases additional expert judgement is needed to assess this sub-criterion.

Apart from direct loss of total area and decrease in number of sites, habitats can be threatened in particular by qualitative changes and deterioration represented by sub-criterion QUL. Typically, this has adverse effects on the abiotic conditions as well as on the structural appearance, the typical set of characteristic species, and on ecological interactions (see Blab et al. 1995). As the discrepancy from a habitat's ideal or (semi)-natural state increases, it becomes more endangered. An 'ideal state' in quality for each habitat type must be elaborated to serve as a reference with regard to all relevant parameters (essential for the value and possible colonisation of the habitat type by typical species). This reference has to consider, among other parameters, the historic conditions, known abiotic requirements, and ecological requirements of typical animal or plant species or plant communities. However, this is linked to methodical problems. In a number of cases the 'ideal' or 'historic' state is not sufficiently known or can only be described in general terms. Therefore, expert judgement is additionally needed to assess QUL. Given that it is often difficult or impossible to separate biotic and abiotic deg-

Figure 3. Map of the regions delimited for the regional threat classification of habitats (major landscape regions; red outlines). For ecological characterisation, Germany can be subdivided into natural units. The figure is based on the system of Meynen and Schmithüsen et al. (1953–1962). For the application in the habitats directive (Natura 2000) and the risk assessment of habitats Ssymank, A (1994) has restructured and generalised the system. The classification of major landscape units is based on physiographic units (black outlines; for a reference list see Annex V.6, Finck et al. [2017]) according to Ssymank et al. (1998) and Petersen et al. (2003).

Table 1. Verbal descriptive definition of criterion I 'National Long-term Threat' (nTH). Following the 'precautionary principle', the highest risk category obtained by any of the two sub-criteria AL and QUL is defined as the overall value of rTH and subsequently nTH.

Criterion I: National long-term threat (nTH)		**Sub-criterion Ia: Area loss (AL)**	**Sub-criterion Ib: Quality loss (QUL)**
Scale of assessment:			
Step 1: regional scale (rTH); Step 2: upscaling to national scale (nTH)		Regional	Regional
Category	Description	Definition: verbal-descriptive	Definition: verbal-descriptive
0	Collapsed	Types of habitats which were previously present in the area considered but today can no longer be proven to exist.	Types of habitats with their quality affected so severely that typical or natural variants are completely destroyed.
1	Critically Endangered	Types of habitats of which only a small part of the original area still exists. With the causes of threat continuing and without any activities for protection and management, complete destruction has to be expected in the near future.	Types of habitats with their quality being negatively affected in nearly their whole range, so that typical or natural variants are only left in one or very few sub regions and threatened by complete destruction in a short time.
2	Endangered	Types of habitats with a heavy decline in area in nearly the whole region considered or already extinct in several (sub) regions.	Types of habitats with their quality being negatively affected in a way that – a decline of typical variants can be stated in nearly the whole area of interest or – typical variants already became extinct in several (sub)regions.
3	Vulnerable	Types of habitats with negative development of area over a broad range of the considered region, or locally extinct at numerous sites.	Types of habitats with their quality being negatively affected in a way that – a decline of typical variants in several sub regions can be stated or – typical variants already became locally extinct at numerous sites.
V	Near Threatened	Types of habitats with negative development (also in the long term), thus being potentially threatened by loss of area if not already threatened according to categories 1–3.	Not defined in the German assessment
*	Least Concern	Presumably not endangered at present	
?	Data Deficient	Classification not possible because of insufficient data	
#	Evaluation not reasonable	These are types of habitats that – although they may show declining tendencies – are considered 'undesirable' from a nature conservation point of view. Examples would be forests of non-native tree species, arable fields on peat soil, or certain degeneration stages of fens and bogs.	
–	Not Evaluated	No corresponding category in the national assessment; all types have been evaluated based on a complete reference list for Germany	

radation processes as proposed by Rodríguez et al. (2011), theses aspects are combined to QUL in the German Red List. This corresponds to a similar approach e.g. in the European Red List of Habitats (Janssen et al. 2016) and in the Red List of Ecosystems of Switzerland (Delarze et al. 2016).

Criterion I: National Long-term Threat

The assessment of nTH in the current edition corresponds to the overall Red List category of the second edition because in 2006 only nTH was considered to determine RLG (Riecken et al. 2006). The assessment is based on an upscaling from rTH to nTH, i.e. from the regional to the national scale (Fig. 2, step 2). Median values of all rTH values for every habitat type are calculated (of a maximum of seven terrestrial regions, i.e. all regions where the habitat type is present).

If regions differ extremely in rTH, the most representative region(s) for each habitat turned the balance. The reference period corresponds to that of rTH. For nTH the categories and definitions remain largely unchanged compared to earlier editions of the Red List Germany (Table 1). However, in contrast to previous editions, intermediate values (1–2, 2–3) are no longer used in this context. The evaluation of nTH is the starting value underlying the Red List assessment scheme (Table 3), whereas the following criteria T and R 'only' cause an upward or downward revaluation of the category.

Criterion II: Current Trend

The 'Current Trend' (T) in total area (and number of sites) is assigned at the national level. The estimation of T is based on development over the last ten years and a forecast for the near future (maximum ten years). This period corresponds to the updating cycle of the Red List Germany. A comparable criterion is used in Germany for the Red List assessment of species, but without the future assessment (short-term population trend, cf. Ludwig et al. 2009). The reporting format for the main results of the monitoring referred to in Article 11 of the European Union (EU) Habitats Directive for habitat types in Annex I also considers short-term trends over a similar time horizon (sliding window over 12 years, cf. DG Environment 2017.). A criterion with a similar idea was also integrated into the Finnish assessment (Kontula and Raunio 2009). They estimate the 'projected quantitative and qualitative change in the near future (criterion A2/B2)' in a time frame of 20–30 years. Bland et al. (2017) use a 50-year period for short-time assessments. T is included in the overall assessment of RLG in this new edition for the first time. Considering the availability of data sources, five categories are used (Table 2). For all endangered and near threatened types of habitats, which show a negative short-term trend, the threat category increases by half a value based on the assessment of nTH. For endangered/near threatened types of habitats which have a stable trend the threat category decreases (i.e. improves) by half a value because we interpret stabilisation as success of nature conservation activities. For endangered habitat types with a currently positive short-term trend, the threat category improves by one value (Fig. 2, step 3).

Criterion III: Rarity

In the revised assessment scheme, a higher risk of loss is basically assumed for habitat types which are extremely rare. They are characterised through very few or very small occurrences and are therefore usually very sensitive to the loss of individual sites since

Table 2. Definition of criterion II 'Current Trend' (T) and implication for the risk assessment procedure.

Symbol	Category	Definition	Change in threat category (based on nTH)
↓	Negative	In the last ten years, a decrease in the total stock of the total area, or at least in large parts of the area, can be observed and is likely to continue in the coming years.	– 0.5
→	Stable	The total area has been largely constant over the past ten years. However, local and regional differences in development are possible. No other trend is expected for the coming years.	+ 0.5
↑	Positive	In the past ten years, the increase in the total area of these types of habitats as a whole, or at least in large parts of the area, is likely to continue in the next few years.	+ 1.0
?	Data Deficient	Classification not possible	no change in threat category
#	Evaluation not reasonable	Types of habitats showing declining tendencies, but are 'undesirable' from the point of view of nature conservation.	no change in threat category

one single event or a critical hazard could destroy the whole inventory (cf. Williams et al. 2015). In the German procedure criterion R functions as a regulating upgrading factor. The main objective of this assessment is to emphasise the higher risk of loss of extremely rare habitats. A similar approach was introduced by Kontula and Raunio (2009), even though thresholds and the degree of differentiation differ due to the specificity of national data sources and natural conditions. The IUCN sets graded thresholds of 'restricted geographic distribution' which are only decisive if defined threat conditions are given (e.g. continuing decline, inferred threatening processes, low number of locations) (IUCN 2016).

Criterion R is not classified in a full system from widespread to extremely rare. All types of habitats are examined and classified as either 'Extremely Rare' or 'Not Extremely Rare'. All types which had been assessed as category 'R' (extremely rare) for the Red List status in the second edition (Riecken et al. 2006) were transferred to the category 'Extremely Rare' of criterion R. Furthermore, extremely rare types of habitats were derived from the area sums of the related Natura 2000 habitat types from the national report of 2013 (Ellwanger et al. 2015) (reporting obligation under Article 17 of the EU Habitats Directive). A maximum threshold for 'Extremely Rare' in terms of area size was set at a total area of 500 hectares in Germany. The corresponding assessment tightens RLG by half a value (Fig. 2, step 4).

Summary of symptoms of risk – Risk of Loss

RLG describes the overall 'Risk of Loss' under current national threat conditions. Based on nTH, criteria T and R have a downgrading or an upgrading effect (Fig. 2, step 5). All possible evaluation constellations are defined in the assessment scheme (Table 3). In principle, only long-term endangered habitat types and types classified in the early warning stage are taken into account in the assessment scheme. For non-

Table 3. Assessment scheme for determining the German Red List status (RLG). For the overall classification, three criteria are applied stepwise from left to right (National Long-term Threat [nTH], Current Trend [T], Rarity [R]).

Criterion I		Criterion II		Change in category	Interim value	Criterion III		Change in category	RLG
National Long-term Threat	**0**		→	+/-0	0		n/s		**0**
	1	Current Trend	↓	-0.5	1!	Rarity	x	-0.5	**1!**
							–		**1!**
			→	+0.5	1–2		x	-0.5	**1**
							–		**1–2**
			#, ?	+/-0	1		x	-0.5	**1!**
							–		**1**
			↑	+1	2		x	-0.5	**1–2**
							–		**2**
	2	Current Trend	↓	-0.5	1–2	Rarity	x	-0.5	**1**
							–		**1–2**
			→	+0.5	2–3		x	-0.5	**2**
							–		**2–3**
			#, ?	+/-0	2		x	-0.5	**1–2**
							–		**2**
			↑	+1	3		x	-0.5	**2–3**
							–		**3**
	3	Current Trend	↓	-0.5	2–3	Rarity	x	-0.5	**2**
							–		**2–3**
			→	+0.5	3–V		x	-0.5	**3**
							–		**V–3**
			#, ?	+/-0	3		x	-0.5	**2–3**
							–		**3**
			↑	+1	V		x	-0.5	**V–3**
							–		**V**
	V	Current Trend	↓	-0.5	3–V	Rarity	x	-0.5	**3**
							–		**V–3**
			→	+/-0	V		x	-0.5	**V–3**
							–		**V**
			#, ?	+/-0	V		x	-0.5	**V–3**
							–		**V**
			↑	+1	*		x	-0.5	**V**
							–		*
	*	Categories are not changed by the evaluation scheme							
	?								
	#								

endangered types, types with unknown threat-status, and types not relevant for nature conservation purposes, nTH corresponds to RLG. Due to the algorithm used, intermediate values can also occur. The stepwise assessment results in a wider spread of Red List categories (Table 4). The (verbal-descriptive) definitions of the Red List categories are derived from the possible combinations of the individual criteria according to the evaluation scheme (see Tables 1, 2).

The categories 'Imminently Threatened By Complete Destruction' (1!) and 'Imminently Threatened' (V–3) are newly introduced. These new categories represent both extremes of 'collapse risk' in the German approach.

Table 4. Categories of the German Red List status (RLG). The (verbal-descriptive) definitions of the Red List categories are derived from the possible combinations of the individual criteria according to the evaluation scheme (see Table 1, 2). The categories 'Imminently Threatened By Complete Destruction' (1!) and 'Imminently Threatened' (V–3) are newly introduced. These new categories represent both extremes of 'collapse risk' in the German approach.

German Red List status (RLG) Category	Description
0	Collapsed (CO)
1!	Imminently Threatened By Complete Destruction
1	Critically Endangered (CR)
1–2	Endangered (EN) to Critically Endangered (CR)
2	Endangered (EN)
2–3	Vulnerable (VU) to Endangered (EN)
3	Vulnerable (VU)
3–V	Imminently Threatened
V	Near Threatened (NT)
*	Least Concern (LC)
#	Evaluation not reasonable
?	Data Deficient (DD)

Results of the first-time application of the Assessment scheme

German Red List of Habitats 2017

The revised assessment system has been tested and applied in the current edition of the 'German Red List of threatened habitats' (Finck et al. 2017) (Table 5). The assessment covers a total of 863 marine, coastal, inland water, open terrestrial, shrubs/trees/forests, and alpine types of habitats in Germany (not considering so called 'technical habitats'). While two-thirds (65.1%, n = 562) of the assessed habitat types were assigned with different degrees of 'risk of loss' (Red List categories '0' to '3–V'), 24.7% (213) are currently of 'Least Concern'. Thirteen marine types of habitats (1.5%), mainly characterised by the European oyster (*Ostrea edulis*) or Honeycomb worm reefs (*Sabellaria* sp.), had to be classified as 'Collapsed' (category 0). Comparing the main habitat groups in Germany, the proportion of threatened coastal habitats (RLG categories 0 to 3–V) is the highest (82.8%). Alpine (58.8%) and marine (52.5%) habitat types represent the least threatened habitat groups. Inland waters (76.4%), open terrestrial habitats (68.8%), and shrubs/trees/forests (69.5%) show proportions of threatened habitat types above the average (65.1%). Open terrestrial habitats represent a significant proportion of habitat types classified in the highest threat category '1!' (16.3%). Intensive land use still represents the main threat factor especially for open terrestrial habitats and (to a lesser extent) forest habitats. A detailed analysis of major threat factors for habitat types in Germany was published in 2019 (Heinze et al. 2019).

Effects of applying the new assessment scheme

The application of the assessment scheme results in a clear spread of the realised categories for the Red List-status. Only a total of 101 (17.1%) of the long-term

Table 5. Assessment results for RLG 2017 (Finck et al. 2017). Number and proportions of habitat types assessed in the categories of German Red List status are given by the main groups of habitat types. Cat = Red Listing Category; T = Number of Types.

Cat RLG	Marine habitats		Coastal habitats		Inland waters		Open terrestrial habitats		Shrubs, trees & forests		Alpine habitats		All habitats (minus tech.)		Technical habitats[†]		All habitats	
	T	%	T	%	T	%	T	%	T	%	T	%	T	%	T	%	T	%
0	13	4.7	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0	**13**	**1.5**	0	0.0	13	1.4
1!	3	1.1	4	6.9	7	5.7	33	16.3	3	2.0	2	3.9	**52**	**6.0**	0	0.0	52	5.5
1	3	1.1	2	3.4	8	6.5	3	1.5	4	2.6	1	2.0	**21**	**2.4**	0	0.0	21	2.2
1–2	3	1.1	7	12.1	31	25.2	48	23.8	22	14.6	1	2.0	**112**	**13.0**	2	2.7	114	12.2
2	22	7.9	5	8.6	4	3.3	2	1.0	5	3.3	7	13.7	**45**	**5.2**	0	0.0	45	4.8
2–3	28	10.1	13	22.4	24	19.5	38	18.8	43	28.5	4	7.8	**150**	**17.4**	6	8.0	156	16.6
3	19	6.8	3	5.2	3	2.4	1	0.5	2	1.3	1	2.0	**29**	**3.4**	0	0.0	29	3.1
3–V	55	19.8	14	24.1	17	13.8	14	6.9	26	17.2	14	27.5	**140**	**16.2**	4	5.3	144	15.4
V	20	7.2	3	5.2	1	0.8	2	1.0	1	0.7	1	2.0	**28**	**3.2**	0	0.0	28	3.0
*	80	28.8	7	12.1	24	19.5	51	25.2	32	21.2	19	37.3	**213**	**24.7**	21	28.0	234	24.9
?	9	3.2	0	0.0	0	0.0	0	0.0	0	0.0	1	2.0	**10**	**1.2**	0	0.0	10	1.1
#	23	8.3	0	0.0	4	3.3	10	5.0	13	8.6	0	0.0	**50**	**5.8**	42	56.0	92	9.8
Σ (all)	278	100	58	100	123	100	202	100	151	100	51	100	**863**	**100**	75	100	938	100
Risk of Loss (Σ cat. 0 to 3–V)	146	52.5	48	82.8	94	76.4	139	68.8	105	69.5	30	58.8	**562**	**65.1**	12	16	574	61.2

[†] Technical habitats: Group of anthropogenic habitats (e.g. buildings, roads, landfills) which have generally less significance for nature conservation. In special cases they can function as substitute habitats for species which are adapted to habitat conditions of settlement areas. Threats to this habitat group are mostly characterised by intensification of usage (e.g. sealing), restoration or demolition of old, historic buildings.

endangered habitat types (nTH = 0, 1, 2, 3, V) were classified in the same category for the overall RLG-status (Fig. 4). The newly introduced categories '1!' and '3–V' are frequently used: 16.2% of all assessed habitat types were classified in category '3–V'. 6% had to be assessed in category '1!' (Table 5). Two thirds of the habitat types, which are valued as 'Critically Endangered' (1) for nTH had to be upgraded to the category '1!' for RLG (Fig. 4, second bar). In contrast, almost 60% of habitat types that were assessed as 'Vulnerable' (3) according to nTH could be downgraded to category '3–V' or V, respectively, as they had a stable or positive short-term trend (Fig. 4, fourth bar).

Case study – raised bogs

The IUCN criteria catalogue (Keith et al. 2013) was applied to the national situation in Germany for raised bogs by Riecken et al. (2013). The overall status was assessed to be 'Critically endangered'. This result corresponded exactly to the national assessment at that time (Riecken et al. 2006). The condition of bogs is even better represented by the new methodology. The degradation started at the beginning of industrialisation during the 18th century, especially in the North-western lowlands of Germany but also in other parts of Europe. Bogs were drained and the peat was cut, dried, and trans-

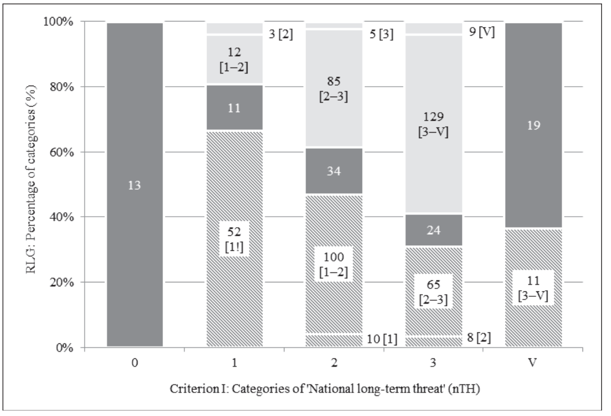

Figure 4. Spread of RLG-values (y-axis) by applying the matrix algorithm (Table 3). The analysis is based on nTH (criterion I, x-axis) for habitat types in Germany. Full dark pigmented bar = no change in category; full light-coloured bar = downgrading of threat category value; brindled bar = upgrading of threat category value; Labels: number of attributive habitat-types in the [resulting RLG-category].

ported so that only about 1–2% of the original area is preserved today (LLUR 2012; Ellwanger et al. 2015). The remaining sites are of relatively small size and isolated from other stocks. In the assessment period of the current edition of the German Red List, the long-term threat situation (rTH) of 'raised bogs' did not change significantly in most landscape regions (Table 6). Nevertheless, agricultural utilisation of former bogs continues and has increased in intensity during recent decades (Rath and Buchwald 2010). Additionally, climate-induced changes in abiotic conditions are having an increasingly negative impact (Essl and Rabitsch 2013). Therefore, T is still classified as negative. Applying the new assessment scheme, RLG had to be upgraded to the highest threat category ('1!'). Compared to the European assessment (Janssen et al. 2016), the situation in Germany is much more critical (European assessment [EU 28] for the decisive criterion A3/historical decline: EN). For the evaluation of the revised assessment of 'raised bogs', it must be considered that 'raised bogs' are characterised by very slow regeneration ability ('RE', Table 6, symbol 'N'). The regeneration ability was estimated for each habitat type. The result is 'additional information' and does not influence the assessment of the degree of threat (Blab et al. 1995). For this reason, no major improvements in the 'Long-term threat' can be expected in the near future. Only limited areas are available for bog restoration in Germany. With these preconditions, 'raised bogs' may always remain in a high long-term risk category in Germany. Criterion T was integrated into the evaluation process in order to be able to display current trends. The method-inherent increased Red List status in 2017 reveals an acute need for action to counteract specific causes of threat. Differences in the rTH show that the situation for characteristic habitat-subtypes in the North-western and North-eastern lowlands and the highland regions is even worse than for (subtypes of) the alpine region. Especially in the Alps, impacts of climate change and anthropogenic use can be observed, but so far have not changed the threat situation of 'raised bogs'. However, there may be a threshold for observable detriments, which has not yet been reached.

Table 6. Red List assessment for 'raised bogs' and Beech (mixed) forest in 2017. Regional Red List categories are presented for all major landscape regions. Code – hierarchical coding for database applications; A – Areas Loss; QU – Quality Loss; rTH – Regional Long-term Threat; nTH – National Long-term Threat; T – Current Trend; RLG – German Red List status; RE – Regeneration Ability: B-K – regeneration 'conditionally possible' to 'hardly possible'; N – 'not regenerable'; Major landscape region (see Blab et al. 1995): NW-Low – North-western lowlands; NE-Low – North-eastern lowlands; W-Upl. – Western highlands; E-Upl. – Eastern highlands; SW-Upl. – South-western highlands; Alp. Fh. – Alpine foothills; Alps – Alps * intermediate values are no longer used for nTH in 2017.

Code	Habitat-Type	NW-Low			NE-Low			W-Upl.			E-Upl.			SW-Upl.			Alp. Fh.			Alps			nTH	n TH	T	T	RLG	RE
		A	QU	rTH	A	QU	rTH	A	QU	rTH	A	QU	rTH	A	QU	rTH	A	QU	rTH	A	QU	rTH	2006	2017	2006	2017	2017	
36.01	Raised bogs (largely intact)	1	1	**1**	1	1	**1**	1	1	**1**	1	2	**1**	1	1	**1**	2	2	**2**	2	3	**2**	1	1	↓	↓	**1!**	N
43.07.04	Beech (mixed) forest on moist, base-deficient sites	2	2	**2**	3	3	**3**	2	3	**2**	3	3	**3**	2	3	**2**	2	3	**2**	3	3	**3**	2–3*	**2**	↑	↑	**3**	B-K

Case study – beech forests

In the German habitat classification used for the Red List, pristine woods are not separated but are assessed together with their utilized variants. There is no database available which describes different pristine Central European forest types in detail. The risk assessment of forest habitat types therefore represents a weighted median of the existing stands (structure-rich old-forest, young age-class forest, etc.). The degree of naturalness (richness of structure, mixed forest, old wood, deadwood, stratification of different age classes) is weighted by the risk assessment through the quality criterion (QUL). In many cases, the specific ground layer is also well-developed in woodland areas which are used by forestry, so that a classification of the forest habitat type is possible. The Long-term Threat to 'beech (mixed) forests on moist, base-deficient sites (*Fagus sylvatica*)' has not changed since the last assessment period and is still classified as being 'Endangered (EN/2)'. Thus, the continued positive short-term trend has not yet affected the long-term threat assessment. However, this type is experiencing an improvement of a full threat category from EN (2) to VU (3) for RLG (Table 6) as the area of beech forests in Germany has been continuously increasing in recent decades. A general reorientation in forest management in recent decades has contributed to a significant increase in native broadleaved forests in Germany (BMEL 2016). Former main threats such as 'reforestation with non-autochthonous trees' have decreased, at least in protected areas. The current downgrading of the Red List status reflects these efforts. Nevertheless, the legal protection of beech forests within Natura 2000 sites may not be sufficient to reach biodiversity goals if intensive forestry continues in large parts of protected areas (Panek 2016). Therefore, programmes were initiated to increase non-intervention management areas (e.g. EU Biodiversity Strategy, National Strategy on Biological Diversity). To continue the positive development, additional focus must be set on the habitat quality as well.

Discussion, conclusions and perspectives

The following discussion focuses on terrestrial and limnic habitats, as more detailed knowledge about most marine habitats has only recently become available.

Dealing with data availability

In contrast to the Red Lists of species, the underlying data for habitat threat are not collected by volunteer scientists but exclusively in the context of monitoring obligations (e.g. EU Habitats Directive) or in the course of habitat mapping by the federal states. Thus, the national Red List assessment in Germany mostly relies on regional data sources collected by federal state administrations. Marine habitats are an exception, because here the German Federal Agency for Nature Conservation is the directly responsible nature conservation authority. Data collection in the 'Exclusive Economic Zone' is therefore carried out by the Federal Government and in coastal areas by the Federal States. Even though data from current habitat mappings were not available for all federal states in the current Red List, the existing baselines provides a good overview of all major landscape regions (see Fig. 3). However, there are still considerable differences in the actuality, evaluation and mapping methodology (Kaiser et al. 2013). In addition, classification systems of habitat types of the sixteen federal states are not completely comparable. For this reason, data originating from federal states often cannot be transferred directly. Thus, a supplementary, case-by-case expert assessment was often necessary up to now. However, there are approaches to minimise problems with data availability and transferability. On the one hand, there are efforts on the part of the federal states to standardise mapping (e.g. in a benchmark paper; Beck et al. 2013). On the other hand, a universally applicable standard list of habitat types is being developed in a current research project which aims to establish a nationwide, uniform random sample of habitat mapping (BfN 2018). This will make it easier to match the data originating from the federal states in the future. In addition, the planned nationwide random samples would represent a kind of calibration. The establishment of frequent, standardised ecosystem monitoring could function as a solid regular database for the assessment of changes in the actual threat situation of many habitat types. Great efforts have also been made in recent years in the classification and monitoring of marine habitat types (Finck et al. 2017; BfN 2019). There is still a lack of national standardised monitoring data for a wide range of habitat types occurring in Germany and Europe. However, through the monitoring obligations of the EU Habitats Directive, a standardised tool is available at least for habitats which correspond to types in Annex I of the Directive. Nevertheless, it is not always possible to assign the types defined in the Habitats Directive directly to the German standard list of habitats due to different definitions or development goals. A crucial question remains: to what extent the habitat data, which were collected under the Habitats Directive, can be used to draw conclusions about the current frequency, distribution, and quality of habitat types within the framework of the national Red List. At least we tried to keep the European types as distinct as possible in order to guarantee a transferability of the data into the German standard list. Nonetheless, summarising the given data into a nationwide Red list is usually a standardisation step for which expert assessment remains necessary.

Comparisons with the previous edition of the Red List (Riecken et al. 2006) are only possible to a limited extent for the individual criteria due to the changed evaluation methodology. Unfortunately, changes in methodology are always at the expense of comparability; at least the determination of the individual criteria nTH and T was kept unchanged.

Relevance of a historic reference value

As a first step in our assessment procedure the long-term threat situation in area and quality is always assessed – if detailed databases are lacking –, based on expert estimation. In today's intensively cultivated landscape, we can assume that the historical conditions of many habitat types with significance for biodiversity were more favourable. Therefore, a comparison with the 'historical *more* ideal condition' of habitat types is the starting point of our Red List assessment. This rationale can be confirmed by the application of IUCN criteria for the 'European Red List of Habitats' (Janssen et al. 2016). In some European countries, only limited data for long-term trends were available, so that criterion A3 (reduction in geographic distribution since 1750) was assessed as being 'data deficient' (cf. Janssen et al. 2016; Biró et al. 2017). Following the 'precautionary principle', the resulting Red List category is therefore based on one or two criteria which often only reflect short-term threat situations and can therefore be misleading to an overrated positive or negative evaluation compared to the 'historical condition' of the habitat. For some habitat types the reference period already reveals a depleted situation. In a study from Hungary, Biró et al. (2017) have shown that the number of highly endangered habitat types increase dramatically if the long-term trend is taken into account. To deal with the possibility of 'earlier decline', Kontula and Raunio (2009) proposed tightening the assessment in a sub-step of their stepwise procedure (here criterion A3, B3). Since 'early decline' represents a temporal shift in the historical 'ideal state' for particular habitat types, this factor is taken into account in the long-term assessment of the German procedure by setting a sliding time frame. Thus, the selection of an adequate historical reference period for Red List assessments is also a question of the specific history of landscape development, as well as national nature conservation objectives. In Germany, for example, the preservation of extensively used semi-natural habitat types is a legal objective regulated in the Federal Nature Conservation Act. This is one of the reasons why we chose a later and dynamic long-term reference period than the IUCN. Since habitats are dynamic systems, which typically do not disappear but rather replace each other, vulnerability must be assessed individually for each type. For this reason, we state that reference periods (particularly long-term evaluations) can also differ for habitat groups. Overrated positive evaluations through the IUCN method may particularly apply to European forest habitats, which have experienced severe historical losses but are currently increasing or stable (see case study beech forests).

Signal effect of the short-time control value

By extending the criteria system, RLG is particularly intended to reveal successes in nature conservation and the need for action. In this context, we agree with the argumentation of Delarze et al. (2016), that the objective of national Red Lists is to demonstrate current trends and to indicate needs for action. This effect becomes clear in various ways by analysing the results of the current German Red List. For example, in an alarming way, many of the open terrestrial habitats were assigned to the highest threat category '1!' (16.3%, Table 5). There is still an ongoing negative trend, especially for many terrestrial open landscape habitat types, mainly caused by the intensification of agriculture accompanied by grassland loss and levelling of site conditions, which results in a severe loss of extensively used rural habitats (Heinze et al. 2019). The loss of biodiversity in the cultural landscape is also a topical issue in European politics. Here the result of the Red List fits into the general picture. The Common Agricultural Policy severely impacts biodiversity and ecosystem services (Simoncini et al. 2019). The "Red List tool" must therefore also be able to reveal short-term changes in intensively used landscapes, which can change very quickly due to initial agricultural policy conditions.

The urgent need for an accentuation of "critically endangered" as well as currently declining habitat types is clearly illustrated by case study of raised bog ecosystems. On the other hand, widespread beech forest habitat types are experiencing an improvement of a full threat category from EN (2) to VU (3) for RLD, as the area of beech forests is continuously increasing (see case study) in recent decades. To show actual tendencies by means of the Red List category, short-term trends function to illustrate modification in the threat situation. Generally, habitats which are characterised by very slow regeneration ability, which have been severely destroyed or deteriorated in historical times (e.g. forest types, raised bogs cf. case studies), can only achieve minor improvements in their 'National Long-term Threat' (nTH) status. Once severely degraded, the period of time required for re-establishing defining features may exceed the reference period of Red List assessments. By using a consecutive assessment scheme, a change in the Red List status is possible even if the long-term threat remains the same. Applying the assessment procedure of previous German Red Lists, a change in the Red List status of a habitat type was only possible if a significant change in the threat situation was achieved with the historical optimum state as a reference.

Overall, the approach of assessing stable trends as (first) successes in nature conservation and therefore with a reduction of the overall threat has proven successful. Otherwise, the current extinction risk for extremely rare but currently stable habitat types that are endangered in the long term would be overestimated when applying the assessment scheme. In nature conservation, the short-term focus should be on the many habitat types that are currently in decline. In the intensively used European cultural landscape, maintaining the same conditions of conservation is also a (small) success. This approach is also laid down in the EU Habitats Directive, whereby a ban on deterioration of the conservation status of habitat types is taken as the minimum objective (Council of the European Communities 1992). However, a long-term goal must also be the improvement of conservation status or rather a “Least Concern” condition. In general, since continuous short-term trends have a long-term effect only after several decades, criterion T functions as a short-time control value. However, the preceding interpretations should be seen with the limitation that the actual impact of the signal effect of RLG through the integration of short-term changes for the necessity of nature conservation action can only be evaluated when regular assessments of the endangerment of Germany’s habitat types are available. By comparing Red List versions, it will be possible in future to map actual developments and thus initiate direct nature conservation measures towards habitats with negative developments. In order to exactly reflect the development in the update cycle of ten years, introduction of two sub criteria of T should be considered, looking ten years into the past on the one hand and ten years into the future on the other (corresponding to the parameter ‘future prospects’ in Habitat Directive assessments of conservation status). As a result, if repeated assessments are available (as in Germany), short time tendencies become particularly relevant for management decisions.

Influence of rarity

Extremely rare habitat types are naturally exposed to a higher risk of severe impairment by individual events (Finck et al. 2017). In the German Red List, the assessment scheme only includes threatened (and near threatened) habitats, so that corresponding conditions were examined before criterion R could have any effect on RLG. Thus, our approach prevents misleading evaluations of naturally extremely rare habitats which are actually not decreasing or degrading (c.f. Gigante et al. 2016). Because rarity has a further aggravating effect in our assessment, extremely rare but currently stable types do not easily lose the focus of nature conservation.

Bland et al. (2017) have the reverse approach, so that thresholds of ‘restricted geographic distribution’ are only decisive if special threat conditions are given (e.g. continuing decline, inferred threatening processes, low number of locations). The criterion of rarity in our approach is so far a yes/no criterion. On the basis of the available data, it should be analysed in more detail as to whether further differentiation of this criterion makes sense in future. A differentiation among various types of habitats would be more appropriate. For example, 500 hectares of Alpine rivers cannot be compared to 500 hectares of beech forests, merely from the surface area point of view.

Future prospects

The assessment system applied to the third edition of the German Red List relies on a full assessment of all criteria and a use of all individual values to determine RLG. In contrast, in the IUCN procedure the highest risk category obtained by any of the assessed criteria represents the overall risk status. Nevertheless, all three criteria of the German methodology indicate spatial changes as symptoms of 'ecosystem collapse' (c.f. Bland et al. 2017; Rowland et al. 2018). The change in quality is also assessed in the case of criterion nTH. However, the effect of individual criteria can deviate greatly from each other (Finck et al. 2017). Therefore, in our opinion, the overall 'risk of loss' can only be assessed by taking all criteria into consideration.

Some recently published European Red List assessments (e.g. Härtel et al. 2009 [CZ]; Essl and Egger 2010 [AT]; Biserkov et al. 2015 [BU]; Finck et al. 2017 [DE]) do not strictly follow the IUCN approach, which has been significantly developed since 2009 (Keith et al. 2009). However, in most of these lists the early draft of the IUCN approach was considered. At least the basic concept of 'ecosystem collapse' has also been applied in the German method. The assessment procedure presented here allows for a clearly defined differentiated assessment of the overall 'risk of loss' (Keith et al. 2009) for individual habitat types under current threat conditions. In principle it has been shown that IUCN criteria for Red List assessment of habitats are applicable within small countries or regions (Bland et al. 2019). Some countries use already an assessment procedure very close to the methodology proposed by the IUCN (e.g. Lindgaard and Henriksen 2011; Delarze et al. 2015, 2016; Gubbay et al. 2016; Janssen et al. 2016; Kontula and Raunio 2019; Chytrý et al. 2019). However, they also had to allow for national or European specifications of the IUCN protocol (e.g. workshop documentation; Finnish Environment Institute 2019). In addition, it is often the case that only some of the IUCN criteria could be evaluated, which can lead to incomplete and sometimes inconclusive risk assessments (see case studies). A detailed comparison of the assessments of these lists is not the subject of this paper. However, in this context the objective of habitat red-listing must be discussed, considering the background of nature conservation goals (see also Delarze et al. 2016; Gigante et al. 2018; Bland et al. 2019; Rowland et al. 2019).

Ultimately, a 'standard criteria system' should offer sufficient flexibility to adapt to national and regional requirements. In this regard, we may need to discuss different thresholds and reference time frames for different habitat groups depending on specific spatial pattern and distribution history. For example, Delarze et al. (2016) have lowered the thresholds for IUCN criteria B1 and B2 in view of the relatively small size of the country. A future prospect will be to integrate useful national approaches to international standards. Since the distribution of ecosystems may extend over different countries (evaluation units), threats to specific ecosystems or habitat types should be determined in a broader spatial scale with the precondition that evaluation systems are comparable. However, this proposal is limited by the actual availability of significant data for the assessed area, which determines the applicability of criteria. Analogous to the improvement of data sources the catalogue of criteria and categories should be adapted and improved. Modifications to apply the IUCN criteria for Red List assessment are a realistic response to the amount of available data for a landscape that is highly diverse, fine-grained and dynamic, as well as strongly affected by cultural influences (cf. Janssen et al. 2016).

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References

Agnoletti M, Rotherham ID (2015) Landscape and biocultural diversity. Biodiversity and Conservation 24(13): 3155–3429. https://doi.org/10.1007/s10531-015-1003-8

Alaniz A, Perez-Quezada JF, Galleguillos M, Vásquez AE, Keith DA (2019) Operationalizing the IUCN Red List of Ecosystems in public policy. Conservation Letters 12(5): 1–11. https://doi.org/10.1111/conl.12665

Beck A, Bettinger A, Boedeker D, Von Drachenfels O, Essl F, Finck P, Gerstner H, Heidecke H, Hinterlang D, Kaiser T, Von der Lancken A, Ludwig G, Petersen W, Raths U, Remy D, Reusch H, Riecken U, Schleyer E, Schlumprecht H, Schröder E, Schuboth J, Schwabe-Kratochwil A, Stellmach M, Störger L, Uhlemann S, Zimmermann F (2013) Bedeutung landesweiter Biotopkartierungen in Deutschland: Ein Eckpunktepapier. Natur und Landschaft 88(3): 101–102. https://doi.org/10.17433/3.2013.50153208.97-102

BfN [Bundesamt für Naturschutz] (2010) Karte der Potentiellen Natürlichen Vegetation Deutschlands. Maßstab 1:500.000. Landwirtschaftsverlag, Münster, 24 pp.

BfN [Bundesamt für Naturschutz] (2017) Determination of the Red List status (RLD), from Finck et al. 2017 (modified). https://www.bfn.de/en/activities/red-list/rl-biotoptypen/aus-marginalspalte/kriterien-und-kategorien.html

BfN [Bundesamt für Naturschutz] (2019) Marine-Monitoring. https://www.bfn.de/en/activities/marine-nature-conservation/marine-monitoring.html

BfN [Bundesamt für Naturschutz] (2018) Ökosystem-Monitoring. https://www.bfn.de/themen/ monitoring/oekosystem-monitoring/

Biró M, Bölöni J, Molnár Z (2017) Use of long-term data to evaluate loss and endangerment status of Natura 2000 habitats and effects of protected areas. Conservation Biology 32(3): 660–671. https://doi.org/10.1111/cobi.13038

Biserkov V, Gussev Ch, Popov V, Hibaum G, Roussakova V, Pandurski I, Uzunov Y, Dimitrov M, Tzonev R, Tsoneva S [Eds] (2015) Red Data Book of the Republic of Bulgaria (Vol. 3). Natural Habitats. Institute of Biodiversity and Ecosystem Research, Sofia, 422 pp.

Blab J (1984) Grundlagen des Biotopschutzes für Tiere:Ein Leitfaden zum praktischen Schutz der Lebensraeume unserer Tiere. Schriftenreihe für Landschaftspflege und Naturschutz 24: 1–205.

Blab J, Riecken U, Ssymank A (1993) Vorschlag eines Kriteriensystems für eine Rote Liste Biotope auf Bundesebene. In: Blab J, Riecken U (Eds) Grundlagen und Probleme einer Roten Liste der gefaehrdeten Biotoptypen Deutschlands: Referate und Ergebnisse. Schriftenreihe für Landschaftspflege und Naturschutz 38: 265–273.

Blab J, Riecken U, Ssymank A (1995) Proposal on a criteria system for a national Red Data Book of Biotopes. Landscape Ecology 10(1): 41–50. https://doi.org/10.1007/BF00158552

Bland LM, Keith DA, Miller RM, Murray N, Rodríguez JP (2017) Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria. Version 1.1. IUCN, Gland. https://doi.org/10.2305/IUCN.CH.2016.RLE.3.en

Bland LM, Rowland J, Regan T, Keith DA, Murray N, Lester R, Linn M, Rodríguez JP, Nicholson E (2018) Developing a standarized definition of ecosystem collapse for risk assessment. Frontiers in Ecology and the Environment 16(1): 29–36. https://doi.org/10.1002/fee.1747

Bland LM, Nicholson E, Miller RM, Andrade A, Etter A, Ferrer-Paris JR, Kontula T, Lindgaard A, Pliscoff P, Skowno A, Zager I, Keith DA (2019) Impacts of the IUCN Red List of Ecosystems on conservation policy and practice. Conservation Letters 12(5): e12666. https://doi.org/10.1111/conl.12666

BMEL (Bundesministerium für Ernährung und Landwirtschaft) (2016) Der Wald in Deutschland. Ausgewählte Ergebnisse der dritten Bundeswaldinventur. 2. korrigierte Auflage. BMEL 2015: 1–52. https://www.bmel.de/SharedDocs/Downloads/EN/Publications/ForestsInGermany-BWI.pdf?__blob=publicationFile

Buder W, Uhlemann S (2010) Biotoptypen: Rote Liste Sachsens (3th edn.). Landesamt für Umwelt, Landwirtschaft und Geologie, Dresden, 140 pp. https://publikationen.sachsen.de/bdb/artikel/11947

Chytrý M, Hájek M, Kočí M, Pešout P, Roleček J, Sádlo J, Šumberová K, Sychra J, Boublík K, Douda J, Grulich V, Härtel H, Hédl R, Lustyk P, Navrátilová J, Novák P, Peterka T, Vydrová A, Chobo K (2019) Red List of Habitats of the Czech Republic. Ecological Indicators 106: 105446. https://doi.org/10.1016/j.ecolind.2019.105446

Council of the European communities (1992) Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. Official Journal L 206: 0007–0050. https://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:L:1992:206:0007:0050:EN:PDF

Delarze R, Gonseth Y, Eggenberg S, Vust M (2015) Lebensräume der Schweiz: Ökologie – Gefährdung – Kennarten (3rd edn.). Ott, Bern, 456 pp.

Delarze R, Eggenberg S, Steiger P, Bergamini A, Fivaz F, Gonseth YG, Hofer G, Sager L, Stucki P (2016) Rote Liste der Lebensräume der Schweiz. Aktualisierte Kurzfassung zum technischen Bericht im Auftrag des Bundesamtes für Umwelt, Bern, 32 pp.

DG Environment (Directorate-General for Environment) (2017) Reporting under Article 17 of the Habitats Directive: Explanatory notes and guidelines for the period 2013–2018. DG Environment, Brussels, 187 pp. http://cdr.eionet.europa.eu/help/habitats_art17

Dimopoulos P, Bergmeier E, Fischer P (2005) Monitoring and conservation status assessment of habitat types in Greece: Fundamentals and exemplary cases. Annali di Botanica nuova serie 5: 7–20. DOI: https://doi.org/10.4462/annbotrm-9207

Doniţă N, Popescu A, Paucă-Comănescu M, Mihăilescu S, Biriş IA (2005) Habitatele din România. Editura Tehnică Silvică, Bucureşti, 496 pp.

Ellwanger G, Raths U, Benz A, Glaser F, Runge S [Eds] (2015) Der nationale Bericht 2013 zur FFH-Richtlinie: Ergebnisse und Bewertung der Erhaltungszustände. Teil 1 – Die Lebensraumtypen des Anhangs I und allgemeine Berichtsangaben auf Grundlage von Daten der Länder und des Bundes. Bundesamt für Naturschutz (Bonn). BfN-Skripten 421(1): 1–417.

Erz W (1971) Landschaftsplanung, Tierökologie und Biotopgestaltung. Natur und Landschaft 46(8): 203–206.

Essl F, Egger G (2010) Lebensraumvielfalt in Österreich – Gefährdung und Handlungsbedarf: Zusammenschau der Roten Liste gefährdeter Biotoptypen Österreichs. Sonderpublikation. Naturwissenschaftlicher Verein für Kärnten, Klagenfurt, 111 pp.

Essl F, Rabitsch W (2013) Biodiversität und Klimawandel. Auswirkungen und Handlungsoptionen für den Naturschutz in Mitteleuropa. Springer Spektrum, Berlin/Heidelberg, 458 pp. https://doi.org/10.1007/978-3-642-29692-5

Essl F, Egger G, Ellmauer T (2002) Rote Liste gefährdeter Biotoptypen Österreichs. Konzept. Umweltbundesamt (Wien). Monographien Band 155: 1–40.

Finck P, Heinze S, Raths U, Riecken U, Ssymank A (2017) Rote Liste der gefährdeten Biotoptypen Deutschlands. Landwirtschaftsverlag (Münster). Naturschutz und Biologische Vielfalt 156: 1–637. https://doi.org/10.19213/973156

Finnish Environment Institute (2019) Workshop-documentation website. Workshop in Helsinki 29–30 October 2019: Red Lists of Ecosystems (RLE) in Europe – similarities and differences in approaches. https://www.ymparisto.fi/en-US/Nature/Natural_habitats/Assessment_of_threatened_habitat_types_in_Finland

Gigante D, Foggi B, Venanzoni R, Viciani D, Buffa G (2016) Habitats on the grid: The spatial dimension does matter for red-listing. Journal for Nature Conservation 3: 1–9. https://doi.org/10.1016/j.jnc.2016.03.007

Gigante D, Acost A, Agrillo E, Armiraglio S, Assini S, Attorre F, Bagella S, Bufa G, Casella L, Giancola C, Giusso GP, Marcenò C, Pezzi G, Prisco I, Venanzoni R, Viciani D (2018) Habitat conservation in Italy: The state of the art in the light of the first European Red List of Terrestrial and Freshwater Habitats. Rendiconti Lincei. Scienze Fisiche e Naturali 29(2): 251–265. https://doi.org/10.1007/s12210-018-0688-5

Gubbay S, Sanders N, Haynes T, Janssen JAM, Rodwell JR, Nieto A, Calix M, Sanders N, Saunders G, Micu D, Kennedy M (2016) European Red List of Habitats. Part 1: Marine habitats. European Commission, Brussels, 46 pp. https://doi.org/10.2779/032638

Härtel H, Lončáková J, Hošek M (2009) Mapování biotopů v České republice: Východiska, výsledky, perspektivy. Agentura ochrany přírody a krajiny ČR (AOPK), Praha. http://www.cetpo.upol.cz/knihovna/biologie/ekologie/

Heinze S, Finck P, Raths U, Riecken U, Ssymank A (2019) Analysis of major threat factors for habitat types in Germany. Natur und Landschaft 94(11): 453–462. https://doi.org/10.17433/11.2019.50153745.453-462

HELCOM (Helsinki-Kommission) (2013) Red List of Baltic Sea underwater biotopes, habitats and biotope complexes. Baltic Sea Environmental Proceedings 138 pp. https://www.helcom.fi/wp-content/uploads/2019/10/BSEP138.pdf

IUCN (2019) Recourses: Background. https://iucnrle.org/resources/background/

IUCN (International Union for Conservation of Nature) (2016) An Introduction to the IUCN Red List of Ecosystems: The Categories and Criteria for Assessing Risks to Ecosystems. IUCN, Gland. https://doi.org/10.2305/IUCN.CH.2016.RLE.2.en

Janssen JAM, Rodwell JS, García Criado M, Gubbay S, Haynes T, Nieto A, Sanders N, Landucci F, Loidi J, Ssymank A, Tahvanainen T, Valderrabano M, Acosta A, Aronsson M, Arts G, Attorre F, Bergmeier E, Bijlsma RJ, Bioret F, Biţă-Nicolae C, Biurrun I, Calix M, Capelo J, Čarni A, Chytrý M, Dengler J, Dimopoulos P, Essl F, Gardfjell H, Gigante D, Giusso del Galdo G, Hájek M, Jansen F, Jansen J, Kapfer J, Mickolajczak A, Molina JA, Molnár Z, Paternoster D, Piernik A, Poulin B, Renaux B, Schaminée JHJ, Šumberová K, Toivonen H, Tonteri T, Tsiripidis I, Tzonev R, Valachovič M (2016) European Red List of Habitats. Part 2. Terrestrial and freshwater habitats. European Commission, Brussels, 1–40. https://doi.org/10.2779/091372

Kaiser T, Schlumprecht H, Finck P, Riecken U (2013) Habitat mapping in the federal states of Germany – status and comparison of methodologies. Natur und Landschaft 88(3): 97–102. https://doi.org/10.17433/3.2013.50153208.97-102

Kaule G (1986) Arten- und Biotopschutz. Grosse Reihe. E. Ulmer, Stuttgart, 461 pp. https://doi.org/10.1007/BF02854827

Keith DA, Orscheg C, Simpson CC, Clarke PJ, Hughes L, Kennelly SJ, Major RE, Soderquist TR, Wilson AL, Bedward M (2009) A new approach and case study for estimating extent and rates of habitat loss for ecological communities. Biological Conservation 142(7): 1469–1479. https://doi.org/10.1016/j.biocon.2009.02.015

Keith DA, Rodríguez JP, Rodríguez-Clark KM, Aapala K, Alonso A, Asmussen M, Bachman S, Basset A, Barrow EG, Benson JS, Bishop MJ, Bonifacio R, Brooks TM, Burgman MA, Comer P, Comı'n FA, Essl F, Faber-Langendoen D, Fairweather PG, Holdaway RJ, Jennings M, Kingsford RT, Lester RE, Nally RM, McCarthy MA, Moat J, Oliveira-Miranda MA, Pisanu P, Poulin B, Regan TJ, Riecken U, Spalding MD, Zambrano-Martı'nez S (2013) Scientific foundations for an IUCN Red List of Ecosystems. PLoS ONE 8(5): e62111. https://doi.org/10.1371/journal.pone.0062111

Keith DA, Rodríguez JP, Brooks TM, Burgman MA, Barrow EG, Bland L, Comer PJ, Franklin J, Link J, McCarthy MA, Miller RM, Murray NJ, Nel J, Nicholson E, Oliveira-Miranda MA, Regan TJ, Rodr'ıguez-Clark KM, Rouget M, Spalding MD (2015) The IUCN Red List of Ecosystems: Motivations, challenges and applications. Conservation Letters 8(3): 214–226. https://doi.org/10.1111/conl.12167

Kontula T, Raunio A (2009) New method and criteria for national assessments of threatened habitat types. Biodiversity and Conservation 18(14): 3861–3876. https://doi.org/10.1007/s10531-009-9684-5

Kontula T, Raunio A [Eds] (2019) Threatened Habitat Types in Finland 2018. Red List of Habitats – Results and Basis for Assessment. Finnish Environment Institute and Ministry of the Environment (Helsinki). The Finnish Environment 2/2019: 1–254. http://hdl.handle.net/10138/308426

Lindgaard A, Henriksen S [Eds] (2011) The 2011 Norwegian Red List for Ecosystems and Habitat Types. Norwegian Biodiversity Information Centre, Trondheim. https://www.artsdatabanken.no/Pages/201624/Norwegian_Red_List_for_Ecosystems

LLUR (Landesamt für Landwirtschaft, Umwelt und ländliche Räume des Landes Schleswig-Holstein) (2012) Potentiale und Ziele zum Moor- und Klimaschutz. Gemeinsame Erklärung der Naturschutzbehörden. LLUR (Flintbek). Schriftenreihe: LLUR SH – Natur 20: 1–37.

Ludwig G, Haupt H, Gruttke H, Binot-Hafke M (2009) Methodik der Gefährdungsanalyse für Rote Listen. In: Haupt H, Ludwig G, Gruttke H, Binot-Hafke M, Otto C, Pauly A [Eds] Rote Liste gefährdeter Tiere, Pflanzen und Pilze Deutschlands. Band 1: Wirbeltiere. Landwirtschaftsverlag (Münster). Naturschutz und Biologische Vielfalt 70(1): 23–71.

Meynen E, Schmithüsen J [Eds] (1953–62) Handbuch der naturräumlichen Gliederung Deutschlands. Selbstverlag, Remagen, 1339 pp.

Panek N (2016) Deutschland deine Buchenwälder. Daten-Fakten-Analysen. Ambaum Verlag, Vöhl-Basdorf, 208 pp.

Petersen B, Ellwanger G, Biewald G, Hauke U, Ludwig G, Pretscher P, Schröder E, Ssymank A [Eds] (2003) Das europäische Schutzgebietssystem Natura 2000. Ökologie und Verbreitung von Arten der FFH-Richtlinie in Deutschland. Band 1: Pflanzen und Wirbellose. Landwirtschaftsverlag (Münster). Schriftenreihe für Landschaftspflege und Naturschutz 69(1): 1–743.

Petrella S, Bulgarini F, Cerfolli F, Polito M, Teofili C [Eds] (2005) Libro rosso degli habitat d'Italia della Rete Natura 2000. Schede, Roma. http://www.parchilazio.it/biblioteca-458-libro_rosso_degli_habitat_d_italia_della_rete_natura_2000

Rath A, Buchwald R (2010) Nutzung von Hochmoorgrünland in Nordwestdeutschland. Naturschutz und Landschaftsplanung 42(4): 108–114.

Raunio A, Schulman A, Kontula T [Eds] (2008) The assessment of threatened habitat types in Finland. Suomen ympäristö, 8/2008 Osa (1): 253–264.

Riecken U, Ries U, Ssymank A (1994) Rote Liste der gefährdeten Biotoptypen der Bundesrepublik Deutschland. Kilda-Verlag (Greven). Schriftenreihe für Landschaftspflege und Naturschutz 41: 1–184. https://doi.org/10.1002/9783527678471.hbnl2003006

Riecken U, Finck P, Raths U, Schröder E, Ssymank A (2003) Standard-Biotoptypenliste für Deutschland. 2. Fassung. Landwirtschaftsverlag (Münster). Schriftenreihe für Landschaftspflege und Naturschutz 75: 1–65.

Riecken U, Finck P, Raths U, Schröder E, Ssymank A (2006) Rote Liste der gefährdeten Biotoptypen Deutschlands. Zweite fortgeschriebene Fassung. Landwirtschaftsverlag (Münster). Naturschutz und Biologische Vielfalt 34: 1–318. https://doi.org/10.1002/9783527678471.hbnl2003006

Riecken U, Finck P, Raths U (2013) Raised bogs of Germany (National Assessment). In: Keith DA, Rodríguez JP, Rodríguez-Clark KM, Nicholson E, Aapala K, Alonso A, Asmussen M, Bachman S, Basset A, Barrow EG, Benson JS, Bishop MJ, Bonifacio R, Brooks TM, Burgman MA, Comer P, Comín FA, Essl F, Faber-Langendoen D, Fairweather PG, Holdaway RJ, Jennings M, Kingsford RT, Lester RE, Nally RM, McCarthy MA, Moat J, Oliveira-Miranda MA, Pisanu P, Poulin B, Regan TJ, Riecken U, Spalding MD, Zambrano-Martínez S (2013) Scientific foundations for an IUCN Red List of Ecosystems. Supplementary material. PLoS ONE 8(5): e62111. https://doi.org/10.1371/journal.pone.0062111

Rodríguez JP, Rodríguez-Clark KM, Baillie JEM, Ash N, Benson J, Boucher T, Brown C, Burgess ND, Collen B, Jennings M, Keith DA, Nicholson E, Revenga C, Reyers B, Rouget M, Smith T, Spalding M, Taber A, Walpole M, Zager I, Zamin T (2011) Establishing IUCN

Red List Criteria for Threatened Ecosystems. Conservation Biology 25(1): 21–29. https://doi.org/10.1111/j.1523-1739.2010.01598.x

Rodríguez JP, Keith DA, Rodríguez-Clark KM, Murray NJ, Nicholson E, Regan TJ, Miller RM, Barrow EG, Bland LM, Boe K, Brooks TM, Oliveira-Miranda MA, Spalding M, Wit P (2015) A practical guide to the application of the IUCN Red List of Ecosystems criteria. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences 370(1662): 20140003. https://doi.org/10.1098/rstb.2014.0003

Rodwell JS, Janssen JAM, Gubbay S, Schaminée JHJ (2013) Red List Assessment of European Habitat Types. A feasibility study. Report for the European Commission, DG Environment, Contract No. 070307/2012/624047/SER/B3, Brussels.

Rowland JA, Nicholson E, Murray NJ, Keith DA, Lester RE, Bland LM (2018) Selecting and applying indicators of ecosystem collapse for risk assessments. Conservation Biology (32): 233–245. https://doi.org/10.1111/cobi.13107

Rowland JA, Bland LM, Keith DA, Juffe-Bignoli D, Burgman MA, Etter A, Ferrer-Paris JR, Miller RM, Skowno AL, Nicholson E (2019) Ecosystem indices to support global biodiversity conservation. Conservation Letters 2019: e12680. https://doi.org/10.1111/conl.12680

Savio L, Gaudillat V (2015) Synthèse des expériences européennes et françaises de Listes Rouges écosystémiques. Version 2. Rapport SPN 2015/35. MNHN-DIREV-SPN, Paris, 78 pp.

Simoncini R, Ring I, Sandström C, Albert C, Kasymov U, Arlettaz R (2019) Constraints and opportunities for mainstreaming biodiversity and ecosystem services in the EU's Common Agricultural Policy: Insights from the IPBES assessment for Europe and Central Asia. Land Use Policy 88: 104099. https://doi.org/10.1016/j.landusepol.2019.104099

Ssymank A (1994) Neue Anforderungen im europäischen Naturschutz: Das Schutzgebietssystem Natura 2000 und die FFH-Richtlinie der EU. Natur und Landschaft 69(9): 395–406.

Ssymank A, Hauke U, Rückriem C, Schröder E, Messer D (1998) Das europäische Schutzgebietssystem NATURA 2000 – BfN-Handbuch zur Umsetzung der Fauna-Flora-Habitat-Richtlinie (92/43/EWG) und der Vogelschutzrichtlinie (79/409/EWG). Landwirtschaftsverlag, Münster. Schriftenreihe für Landschaftspflege und Naturschutz 53: 1–560.

Von Drachenfels O (2012) Einstufung der Biotoptypen in Niedersachsen – Regenerationsfähigkeit, Wertstufen, Grundwasserabhängigkeit, Nährstoffempfindlichkeit, Gefährdung. Informationsdienst Naturschutz Niedersachsen 32(1): 1–59.

Von Hengel U, Westhus W (2011) Rote Liste der Biotoptypen Thüringens. 3. Fassung, Stand 12/2010. Naturschutzreport 26: 526–541. https://tlubn.thueringen.de/naturschutz/biotopschutz/quellenverz/

Williams RJ, Wahren CH, Stott KAJ, Camac JS, White M, Burns E, Harris S, Nash M, Morgan JW, Venn S, Papst WA, Hoffmann A (2015) An International Union for the Conservation of Nature Red List ecosystems risk assessment for alpine snow patch herbfields, South-Eastern Australia. Austral Ecology 40(4): 433–443. https://doi.org/10.1111/aec.12266

Zimmermann F, Düvel M, Herrmann A (2011) Biotoptypenkartierung Brandenburg. Liste der Biotoptypen mit Angaben zum gesetzlichen Schutz (§ 32 BbgNatSchG), zur Gefährdung und zur Regenerierbarkeit. Landesamt für Umwelt, Gesundheit und Verbraucherschutz, Potsdam.

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REVIEW ARTICLE

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Betting the farm: A review of Ball Python and other reptile trade from Togo, West Africa

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Abstract

Our review of the CITES trade database confirmed that the ball python is the most exported species by Togo; with 1,657,814 live individuals – comprising 60% of all live reptiles – reported by importing countries since 1978 (almost 55,000 annually since 1992). In total, 99% of the ball pythons legally exported from Togo under CITES were intended for commercial use, presumably as exotic pets. Since the turn of the century, wild-sourced snakes exported from Togo have been largely replaced with ranched snakes, to the extent that in the last 10 years 95% of these live exports were recorded using CITES source code "R" with the majority destined for the USA. We found discrepancies in the CITES trade database that suggest ball python exports were consistently underestimated by Togo and that both ranched and wild-sourced ball python annual quotas have been exceeded on multiple occasions including as recently as 2017. Furthermore, our field visits to seven of these "python farms" revealed that they are also involved in the commercial trade in at least 46 other reptile species, including eight that are already involved in formal CITES trade reviews due to concerns regarding their sustainability and legality. Ranching operations in West Africa were once thought to provide a degree of protection for the ball python; however, in light of recent research, there is growing concern that ranching may not confer any significant net conservation benefits. Further scrutiny and research are required to ensure the long-term survival of wild ball python and other reptile species populations in Togo.

Keywords
CITES, conservation, *Python regius,* ranching, wildlife trade

Introduction

The exotic pet trade is an enormous global enterprise (Bush et al. 2014) involving international trade in millions of individuals of thousands of species, only some of which are regulated [under the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)] (Can et al. 2019). For some species, captive breeding in destination countries (e.g., the USA and some European countries) also takes place, however others are obtained directly from source countries, commonly from the subtropical and tropical region (e.g., Bush et al. 2014; Harrington 2015; Jensen et al. 2018), which, in some cases, can provide an important income source for local communities (e.g., D'Cruze et al. 2020a).

To assess the long-term sustainability of such trades, an understanding of trade patterns is crucial. With this remit, trade in ball pythons (Python regius) (a popular pet in many countries, particularly the USA), exported by Togo (one of the species' Range States, and one of the main source countries involved in international export), is an informative case study – in part because it illustrates an almost complete shift from wild-captured individuals to the use of ranching. Ranching (defined below) is a production system, intended as a potential solution to the unsustainable harvest of wild animals, dependent on "farms" that do not function as farms in the traditional agricultural sense, rather they are continually dependent on a wild source that is by definition "surplus" to the wild population [i.e. that portion of the population that would likely suffer naturally high mortality rates in the wild (CITES glossary 2019, https://www.cites.org/eng/resources/terms/glossary.php)]. This paper explores changing trade patterns (numbers, source, and markets) in ball pythons from Togo, carried out as part of a broader study of the relationships between in-situ and ex-situ harvesting with respect to sustainable wildlife use. Here we focus on the farms involved in ball python trade (the extent and changing nature of their trade, including the markets that they supply, and other reptiles collected for export), elsewhere we address ball python supply (D'Cruze et al. 2020a) and trade links with neighbouring Range States (Harrington et al. in prep.).

Togo is a relatively small West African country (56,790 km^2) bordered by Ghana to the west, Benin to the east, Burkina Faso to the north and the Gulf of Guinea to the south. It comprises a long strip of land located between a latitude of 6°–11°N and a longitude of 0°–2°E, stretches over 660 km from north to south and has a coastline of 50 km, east-west (Segniagbeto and Van Waerebeek 2010). Its maximal width is 120 km between 7 and 8°N. The wet season is pronounced in the south with two rainfall periods, between April and July, and September and November; the dry season is introduced by the Hamattan desert winds between November and March (Segniagbeto et al. 2011). The landscape is largely a gently undulating plain, with the exception of the Atakora range ("chaîne de l'Atakora"), which crosses the country in a northeast-southwest direction (Segniagbeto and Van Waerebeek 2010). As a consequence of its location, the Togolese landscape consists, from south to north, of a succession of various ecosystems ranging from coastal grasslands to equatorial and wet tropical forests, and ending in Sudan savannahs in the North that is traditionally divided into five distinct ecological zones (Ern 1979; Novinyo et al. 2015). The diversity in these ecosystems is considered as being highly favourable to herpetofaunal diversity, especially snake species of which the majority are confined to specific biotopes (Segniagbeto et al. 2011).

Research focused on reptile diversity in Togo date back to the time of German colonisation and herpetologists such as Sternfeld who published the first inventory of Togolese snakes that included a total of 75 species (Sternfeld 1908). Further research followed in subsequent decades [e.g., Loveridge 1939, 1944, 1958; Hulselmans et al. 1970; Roman 1984; and Anonymous 2002 (a monograph of the national survey of the biological diversity in Togo)]. In 2011, Segniagbeto et al. produced an annotated list of 91 snake species currently recorded from Togo. Although these authors recognised that some taxonomic uncertainties require further scrutiny this remains the most recent review of snake diversity in Togo carried out to date. With regards to their conservation status, 30 (33%) of these snake species, representing 12 different families, have been assessed according to the IUCN Red List of Threatened Species (IUCN 2019). This resulted in 29 species being classified as Least Concern and one species, the lined centipede-eater (*Aparallactus lineatus*) being classified as Near Threatened (IUCN 2019). With regards to their population status, seven of these 30 species have populations considered to be stable and the remaining 23 species have populations of unknown status (IUCN 2019).

Togo is recognised to be one of the main reptile exporters of sub-Saharan western Africa with several species currently harvested at significant levels for the international "exotic" pet trade (Affre et al. 2005; Jensen et al. 2018). In particular the ball python (*Python regius*), a species endemic to parts of Central and West Africa, is being exported in relatively large numbers, in fact it is the single most traded CITES listed live animal legally exported from Africa (Auliya et al. 2020; D'Cruze et al. 2020a). To date, the ball python has been subject to some biological and ecological studies focused on specimens in the wild [e.g., dietary preferences (Luiselli and Angelici 1998), and ectoparasite comparisons (Luiselli 2006) between sexes]. However, these studies notwithstanding, currently there remains a near complete lack of information on ball python home range sizes and activity patterns (in terms of both sex, age class, and season) throughout its range. This lack of information impedes the effective management of commercial trade and the assessment of its (and other anthropogenic-induced) impacts on wild ball python populations (cf. Auliya et al. 2020).

Much of the international ball python trade can be traced back to a number of registered reptile "farms" that are in operation across West Africa, most notably Benin, Ghana and Togo (Robinson et al. 2015). Although some of these farms initially became involved with the international commercial trade of ball pythons in the 1960s (de Buffrénil 1995; Ineich 2006), since 1997, these farms have also officially been engaged in "ranching" (UNEP 2019) which refers to rearing, in a controlled environment, snakes taken as eggs or juveniles from the wild that would "otherwise have had a low probability of surviving to adulthood" (CITES glossary 2019), and releasing a proportion back into the wild (Ineich 2006). Additionally, wild gravid females are also collected, and after laying their eggs in captivity are released back into the wild (Ineich 2006; Luiselli et al. 2007). However, local hunters also collect wild ball python specimens (source code "W") including adult males that are not released (D'Cruze et al. 2020a). In addition to commercially lucrative ball python, several other CITES and non-CITES listed reptile species are also collected for direct export (Ineich 2006). A number of missions have been carried out to assess ball python production methods at these farms (e.g., de Buffrénil 1995; Jenkins 1998; Affo 2001; Harwood 2003) including the most recent by Ineich (2006), who concluded that the practice of ranching being carried out by seven different farms in Togo was being done in "relatively healthy conditions".

In terms of international trade regulation, the family Boidae (including all species taxonomically assigned to the Pythonidae) has been listed on the Appendices of CITES since 1977 (except *Boa constrictor* that was listed in 1975). Togo joined CITES in 1978, and entered into force in 1979 (CITES 2019), and since that time ball python exports have operated under a CITES Appendix II listing. Between 1992 and 2006 there were a number of CITES interventions to ball python trade in Togo. In 1992, the first two commercial reptile farms were reported for Togo (de Buffrénil 1995), and a CITES review of significant trade took place when Togo failed to provide relevant information as previously requested by the CITES Secretariat (Ineich 2006). Subsequently, the CITES Standing Committee recommended a temporary suspension of imports from this country (see Ineich 2006). Between 1993 and 1995 a suspension request was submitted, confirmed and ultimately cancelled while Togo implemented the recommendations from the CITES Animals Committee to better control exports (see Resolution Conf. 2.12). The management authority of Togo accepted the implementations in 1995 on the grounds that specimens should be described as "ranched", not "captive-bred", following an extensive review of practices on reptile farms in the country. CITES quotas were first introduced for wild-taken and ranched ball python specimens exported from Togo in 1990 (Affo 2001), with annual quotas set at 1,500 individuals for wild-sourced snakes, and ranging between 40,000 and 62,500 individuals for ranched snakes since then until 2019 (UNEP 2019). In 1997, the European Union (EU) listed the ball python on Annex B of Council Regulation No 338/97 (EU no 2017/160) which generally equates to CITES Appendix II. In 2015, the EU provided a positive opinion for importing ball pythons exported by Togo that were sourced from the wild, ranched or born in captivity (codes W, R and F respectively) (SRG 73 Soc).

Despite the relatively long-standing history of ball python ranching in Togo and recent / on-going endorsement from major importers such as the EU, the last detailed examination of ball python production systems in West Africa was carried out almost 15 years ago (Ineich 2006). To help provide a more recent update, herein, we present detailed data on the ranching activities of farms currently involved in the ranching and export of ball pythons in Togo. The aim of our study was to:

(1) Assess the extent and characteristics (source, purpose and destination) of ball python trade originating from Togo.

(2) Quantify changes over time in the Togolese ball python trade, specifically with respect to the role of ranching and conformity with national annual trade quotas.

(3) Provide a preliminary assessment of the wider activities of reptile farms in Togo with respect to the other species involved.

The overall objective of this study was to gain insights into potential impacts that this type of wildlife trade activity has on ball pythons and other reptiles in Togo. Ultimately, we hope our findings, and other recently published research focused on the reptile trade in Togo, will inform future interventions to aid conservation initiatives for this important site of herpetological biodiversity.

Methods

Desktop data collection

To determine the number of ball pythons exported from Togo, trade data were obtained from the CITES database. Countries exporting or importing species recognised by CITES are responsible for recording each trade transaction; a central database of all trade is publicly available at https://trade.cites.org/. To obtain numbers of ball pythons traded from Togo, all trade records pertaining to ball pythons exported from Togo, for all purposes, all source codes (outlined in Notification 2002/022) and all trade terms, were downloaded as a comparative tabulation from the CITES trade database. Further analysis of the trade data was limited to records of exports reported as "live" and for "Commercial" (T), "Breeding" (B), "Zoological" (Z) or "Personal" (P) use. All importing countries were included in the search criteria. Trade between the years 1978 (the year that Togo joined CITES) and 2018 were considered, and both exporter- and importer-reported quantities were used, and compared. Each "live animal" reported was presumed to represent an individual animal. The same data were obtained from the CITES trade database for all reptiles exported from Togo to enable assessment of the relative importance of ball pythons in the Togolese reptile trade.

Information detailing the annual trade quotas implemented by Togo (1997–2017) was obtained from the Species+ website (UNEP 2019). Quotas for both ranched and wild specimens were included. The quota data enabled a comparison with trade data obtained from the CITES database, to calculate to what extent Togo complies with trade restrictions for ball pythons.

For the USA specifically, the number of ball pythons imported from Togo between the years 2000 and 2017, was obtained from the U.S. Fish and Wildlife Service Law

Enforcement Management Information System (LEMIS) via a Freedom of Information Act (FOIA) request submitted to the Fish and Wildlife Service, Office of Law Enforcement which was received on 08.05.18 (control number FWS-2018-00788). This provided an independent source of trade data for one of the main importing countries for ball pythons, as well as additional information on individual Togolese exporters.

Field data collection

Research teams (composed of five different individuals) visited and collected data from eight different reptile farms across Togo during three field trips in 2018 (February, June and October) and two field trips in 2019 (February and April) lasting on average between 10 and 21 days. All official visits were organised under the guidance and permission of the CITES Scientific and Management Authorities in Togo. Photos of reptile species observed were taken with the consent of the farm owners; however, our aim was not to carry out a full inventory of the farm owner's stock. Rather, the images were taken to aid subsequent taxonomic identification. For all species, binomial nomenclature and information regarding their conservation, population status, and distribution was gathered from the IUCN Red List of Threatened species (hereafter the IUCN Red List, IUCN 2019). Conservation status was recorded in accordance with the 2001 IUCN Red List categories and criteria system (version 3.1) as Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC) or Data Deficient (DD). Species Not Evaluated by the IUCN Red List (NE) were excluded from the analyses (please see Appendix I). Population status was recorded as Decreasing (D), Increasing (I). Stable (S), Not Evaluated (NE), and Unknown (U). Distribution was recorded for each species as Native (NA), RE (Regional), or W (Widespread) (please see Appendix I). To test whether the proportions of conservation or population status categories for these additional reptile species differed from expectation (relative to all reptile species recorded from Togo, i.e. whether farms were selecting particular categories of wild species), comparable data were collated for all reptile species in Togo from the IUCN Red List (IUCN 2019).

Statistical analysis

To test for trends over time in ball python exports from Togo, as recorded in CITES trade records, we summarised the records by year and used the tslm function in the "forecast" package (Hyndman 2017) in R to fit linear models to the resulting annual time series data, and to quantify and test the significance of trends. This analysis was carried out for all data (total exports) and regional-level data (where regional importers included Asia, Europe and North America [USA and Canada]). We used chi-squared tests to test for changes in the proportion of ranched and wild-sourced ball pythons among time periods and Pearson's correlation coefficient and t-tests to compare exporter- and importer-reported numbers. For USA imports of ball pythons, we used t-tests to compare numbers imported as reported by CITES and LEMIS. Finally, we used chi-squared tests to compare proportions of conservation and population status categories among reptiles observed on farms with those present in the wild; note that this provides only a partial test for farm selection because many Togolese reptile species are not yet listed on the IUCN Red List and thus have no formally assigned conservation or population status. For chi-square tests, we obtained simulated p values (based on 2000 replicates) for tests with low expected values. Statistical analyses were carried out in R (version 3.5.1, R Core Team 2018).

Results

Ball Python and other reptile trade originating from Togo

The CITES trade database (https://trade.cites.org) contained 4,863 records of (live and dead) reptile exports from Togo between 1978 and 2017 (i.e. exports under all trade terms, including live animals, bodies and body parts), involving at least 51 species of 28 genera. 15% (*n* = 723) of all reptile trade records involved ball pythons, and 98% (*n* = 710) of those involved live snakes. Overall, 94% [*n* = 4,564] of all reptile trade records involved live animals. In total, ball python trade records, documenting exports from Togo between 1978 and 2017, represented the export of between 963,344 and 1,657,814 live individuals (according to exporter- and importer-reported quantities, respectively) – these figures comprised 53% of all live reptiles reported by Togo as exports during this period (Fig. 1; or 60% of all live reptiles reported as exports from Togo by importing countries). Other notable species (those involving a total of 100,000 or more individuals reported by Togo as exports over the same time period) were the Nile monitor lizard (*Varanus niloticus*), the savannah monitor lizard (*V. exanthematicus*), and the Senegal chameleon (*Chamaeleo senegalensis*).

According to both exporter- and importer-reported quantities, live ball python exports from Togo increased from < 14,000 per year in the late 1970s/early 1980s [following which there was a statistically significant increase of one to two thousand per year (1978 – 1987: exporter-reported trend = 1,974 per year, $F_{1, 8} = 35.8$, $P < 0.001$, $R^2 = 0.82$; importer-reported trend = 1,490 per year, $F_{1, 8} = 50.6$, $P < 0.001$, $R^2 = 0.86$)] to approximately 60,000 in 1991 (a four-fold increase over the 4 years between 1988 and 1991), reaching a peak of 74,751 (as reported by importing countries) in 1994 (Fig. 2A). Since the early 1990s, numbers reported by Togo (as exporter) suggest an overall increase (1992 – 2016: exporter-reported trend = 1,597 per year, $F_{1, 23} = 11.20$, $P = 0.003$, $R^2 = 0.33$) but this was due to zero exports declared in 1997 and 2000, which importer-reported quantities suggest were incorrect (see Fig. 2A). Importer-reported quantities suggest that numbers of ball pythons exported from Togo since 1992 have actually remained relatively stable (at an annual average of 54,754), albeit with considerable fluctuation among years (SD = 17,105; importer-reported trend = 8.8,

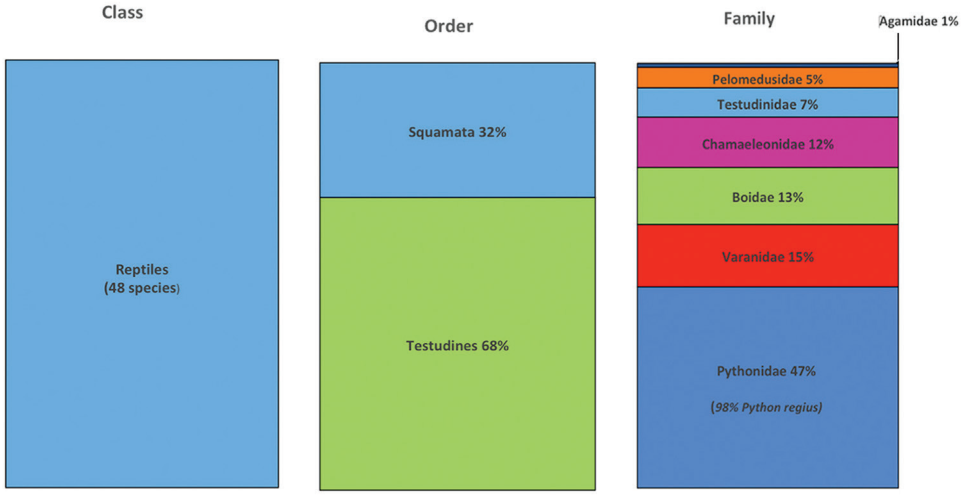

Figure 1. The proportion of all reptile species reported as live exports by Togo (1978–2017) in each taxonomic order and family. Source: CITES trade database, https://trade.cites.org

$F_{1,24} < 0.001$, $P = 0.985$, Fig. 2A). The highest annual export was of 88,959 individuals (reported by importing countries) in 2002 (the lowest reported by importing countries, since 1992, was 21,136 in 1995).

Discrepancies between annual exporter- and importer-reported quantities (Fig. 2B) suggest that, although exporter- and importer-reported annual totals were correlated ($r = 0.54$, $P < 0.001$), exports were consistently (in 31 of 39 years), and significantly, underestimated by Togo (paired t-test: $t = 4.44$, $df = 38$, $P < 0.001$). The maximum discrepancy (underestimate) recorded was 72,747. There were 8 years in which Togo over-estimated numbers exported (as compared to importer-reported quantities), notably, including the most recent four years of the study – in this case, the maximum discrepancy (overestimate) was 31,621. The mean absolute difference between annual exporter- and importer-reported quantities was 20,389.

In total, and according to both exporter- and importer-reported quantities, 99% (n = 950,829 and 1,647,639 respectively) of individual live ball pythons exported from Togo were intended for commercial use (presumably as exotic pets). Since the years 1999/2000 wild-sourced snakes exported from Togo were largely replaced with ranched snakes (Fig. 3). Over the most recent 10–15 years of trade records analysed, at least 95% of total importer-reported exports (83% of exporter-reported exports) were ranched, and although high numbers (ca. 40,000) of wild-sourced animals were reportedly exported in 2001 and 2002 (by importers), wild-sourced animals since 2003 comprised < 5% of total exports reported by importing countries [Fig. 3; although slightly higher figures (< 6% and up to 17 and 20% on two occasions) were reported by Togo]. The change in proportion of animals ranched versus of wild origin over time (time period combined as three decades – 1990s, 2000s, and 2010s) was statistically significant (animal source - time association: $\chi^2 = 663760$, $df = 2$, $P < 0.001$).

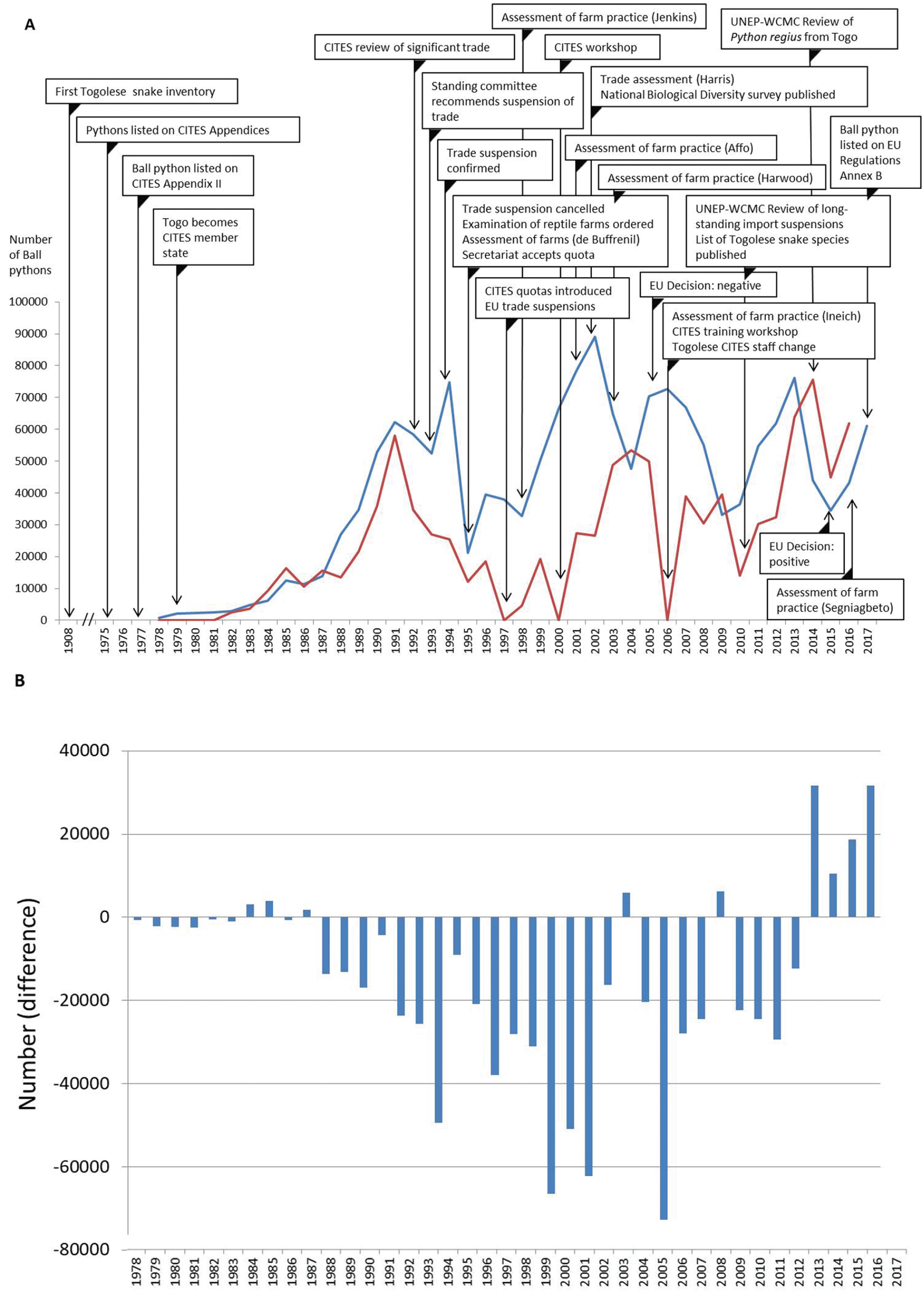

Figure 2. A Number of individual live ball pythons exported from Togo as reported by CITES, 1978-2017, showing exporter- (red) and importer-reported (blue) quantities against timeline of events associated with the regulation of ball python trade. Note that exporter-reported quantities were not available for 2017 at the time of analysis. **B** Annual discrepancies between exporter- (Togo) and importer-reported quantities. Source: CITES trade database, https://trade.cites.org

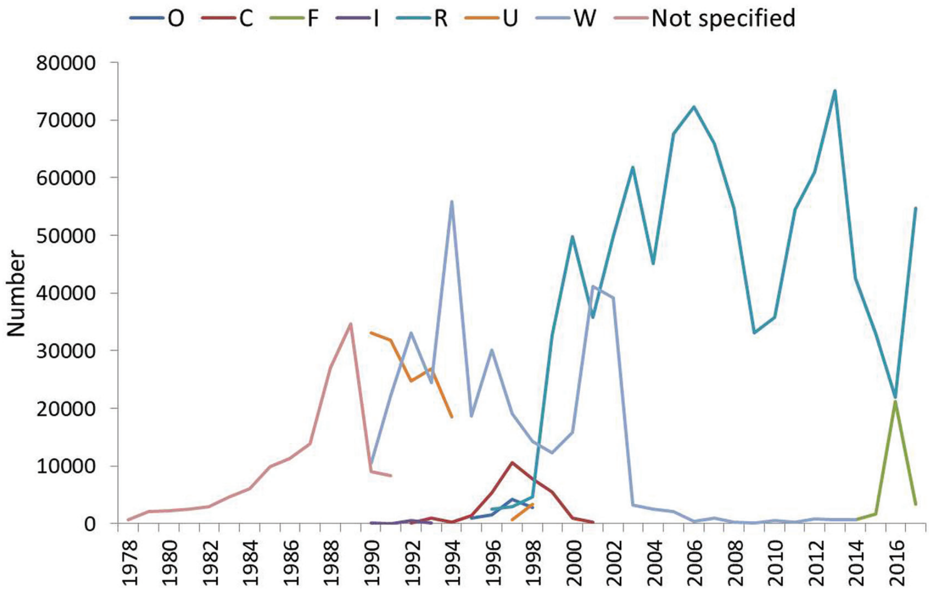

Figure 3. Reported source of ball pythons exported from Togo, based on importer-reported quantities obtained from CITES trade records. Note that the CITES trade database does not contain source information for most records prior to 1991 (unless the specimen was specifically declared as captive-bred, CITES 2013). O=pre-convention specimens, C=captive-bred, F=born in captivity, I=confiscated or seized, R=ranched, U=unknown, W=wild. Source: CITES trade database, https://trade.cites.org

CITES trade records reported exports of ball pythons from Togo to 58 different countries between 1978 and 2017, in total quantities for the period that varied between 1 and over 1 million ball pythons per country; note, however, that minima and maxima represent extreme cases, and that, for most (71.4%) countries, reported exports ranged between 100 and 9,000 ball pythons. Both exporter- and importer-reported quantities showed the USA to be the largest importer, responsible for between 60 and 77% of exports (for exporter- and importer-reported quantities respectively). Six other countries reported importing >1% of the total over this period: France (7.5%), Germany (4.3%), Italy (2.7%), Spain (1.3%), Belgium (1.2%), and Japan (1.1%). Exporter-reported quantities suggested that three additional countries – the Netherlands, Hong Kong and Ghana – also imported >1% of the total (1.6, 1.3 and 1.7, respectively). At a regional level, Europe was second to North America (USA and Canada; there were no exports reported to Mexico) as importer of Togolese ball pythons, responsible for between 30 and 20% of exports (for exporter- and importer-reported quantities respectively) (Fig. 4A, B).

Since the 1990s (following an increase in ball python exports from Togo in the latter half of the 1980s; see Fig. 2), North American imports of ball pythons from Togo have fluctuated considerably through the years. The highest annual imports into North America were of 72,000 ball pythons in both 1994 and 2002 – annual imports since 2002 were at least 13,000 lower than these figures suggesting some level of a decline in

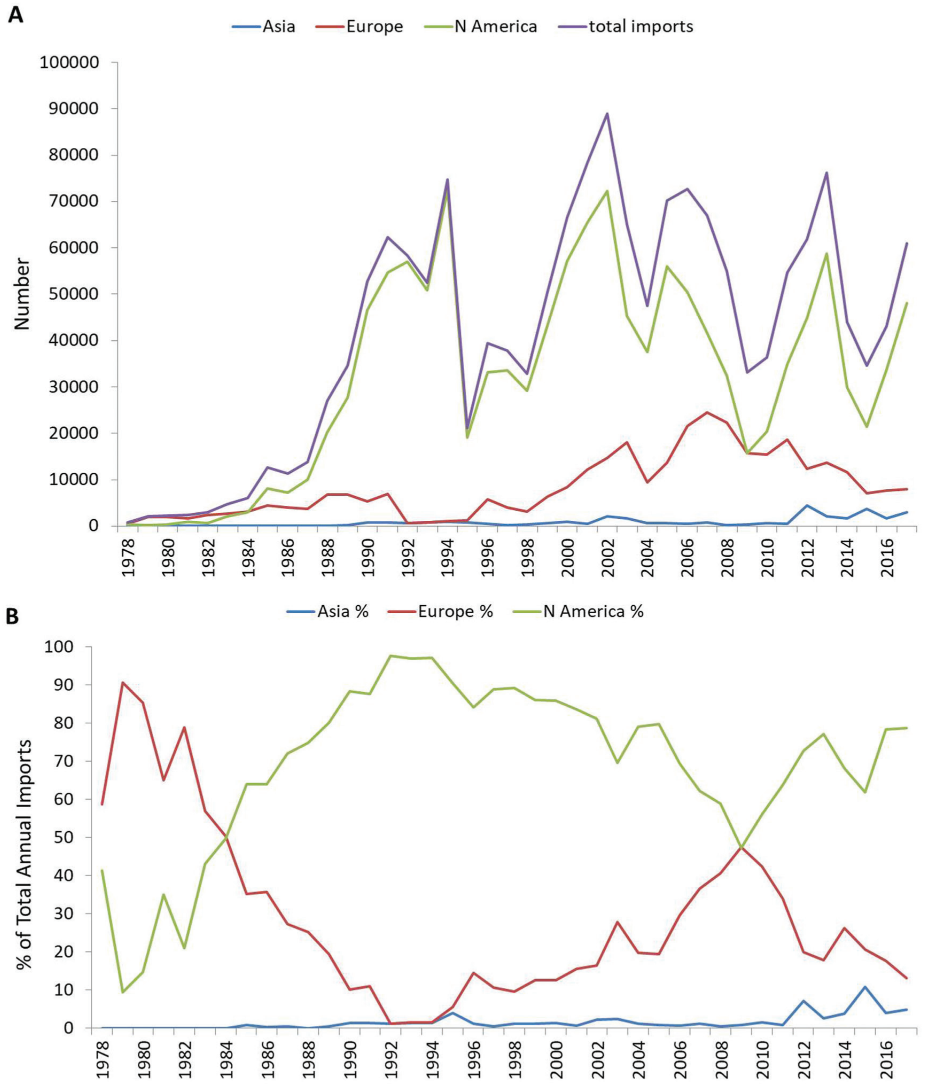

Figure 4. The annual number (**A**) and proportion (**B**) of ball pythons from Togo imported by three main world regions (Asia, Europe and North America), 1978–2017. Proportion shown as % total ball python imports from Togo. Data derived from Togo export records on CITES trade database, and based on importer-reported quantities. Source: CITES trade database, https://trade.cites.org

North American imports (cf. Luiselli et al. 2012), but longer-term trends (1992–2017) were not statistically significant (annual average imports into North America, 1992 – 2017 = 42,507, SD = 15,911, trend = -611, $F_{1, 24}$ = 2.26, P = 0.146, Fig. 4A). More recently, over the last ten years, a decrease in the numbers imported by Europe (from 22,377 in 2008 to 8,026 in 2017, trend = -1499 per year, $F_{1, 8}$ = 39.04, P < 0.001), which was not mirrored by numbers imported into North America (trend = 1645 per year, $F_{1, 8}$ = 1.27, P = 0.292; i.e. there was no evidence that the apparent decline in

North American imports had continued, Fig. 4A) means that the proportion of ball python exports from Togo being imported into North America has increased (from 59% of total imports in 2008 to 79% of total imports in 2017; Fig. 4B). Numbers imported into Asia have also increased over the last ten years (from 226 in 2008 to 2,980 in 2017, trend = 313 per year, $F_{1, 8} = 5.44$, $P = 0.048$) but the low numbers involved mean that in 2017 (the most recent year in the dataset) Asia was responsible for < 5% of all imports (Fig. 4B).

In the 21 years between 1997 and 2017, importer-reported quantities suggest that ranched ball python annual quotas have been exceeded on six different occasions (by an average of 10,421, maximum 19,787), most recently in 2013 (by 12,626 ball pythons, Fig. 5A). However, according to export records provided by Togo, the ranched ball python quota has only been exceeded on three occasions (once, in 2014, by a total of 10,712 ball pythons, and twice more, in 2004 and 2013, by 174 and 23, respectively, Fig. 5A). With regards to ball pythons reported as wild sourced, according to importing countries the CITES quota has been exceeded on ten different occasions (by an average of 13,730; maximum 39,644 in 2001) most recently in 2017 (by 1,450 ball pythons, Fig. 5B), whilst exporter-reported quantities suggest that the quota was exceeded on six occasions (by between 250 and 4,000), most recently in 2014 (by 290 ball pythons, Fig. 5B).

The LEMIS trade database documents the import of a total of 764,527 live ball pythons from Togo into the USA since the year 2000. Ball pythons came from 11 independent Togolese exporters, two of which (Togamin and Pajar Sarl) were responsible for 88% of all ball pythons imported over this period (55%; $n = 422,867$ and 33%; $n = 251,969$, respectively, Fig. 6), five others were responsible for 1–5% imports (Fig. 6). Annual imports documented by LEMIS did not differ significantly from importer-reported quantities in the CITES trade database ($t = -0.01$, $df = 17$, $P = 0.988$, mean = 42,474 vs. 42,495, respectively; Fig. 6).

Current trade: species observed at reptile farms

In total, (including only those identified to species level, plus those of ambiguous taxonomic status indicated by "cf.") 46 reptile species were observed during visits to the eight different farms in Togo (including seven python farms and one venom farm) between February 2018 and April 2019, including 1 Crocodylia (2%), 10 Sauria (21%;), 24 Serpentes (52%;), and 11 Testudines (24%) (please see Appendix I). Observed species diversity at the farms ranged between three and 23.

With regards to conservation and population status of the 46 species observed, over half (59%, $n = 27$) had not yet been evaluated on the IUCN Red List. Of the 19 observed species that were included on the IUCN Red List most (68%, $n = 13$) were categorised as Least Concern but for most of these ($n = 12$, 63% of all Red List species) conservation status was unknown (please see Appendix I, and Appendix II). Four species observed on farms [the African spurred tortoise (*Centrochelys sulcata*), Senegal flapshell turtle (*Cyclanorbis senegalensis*), Home's hinge-back tortoise (*Kinixys homeana*),

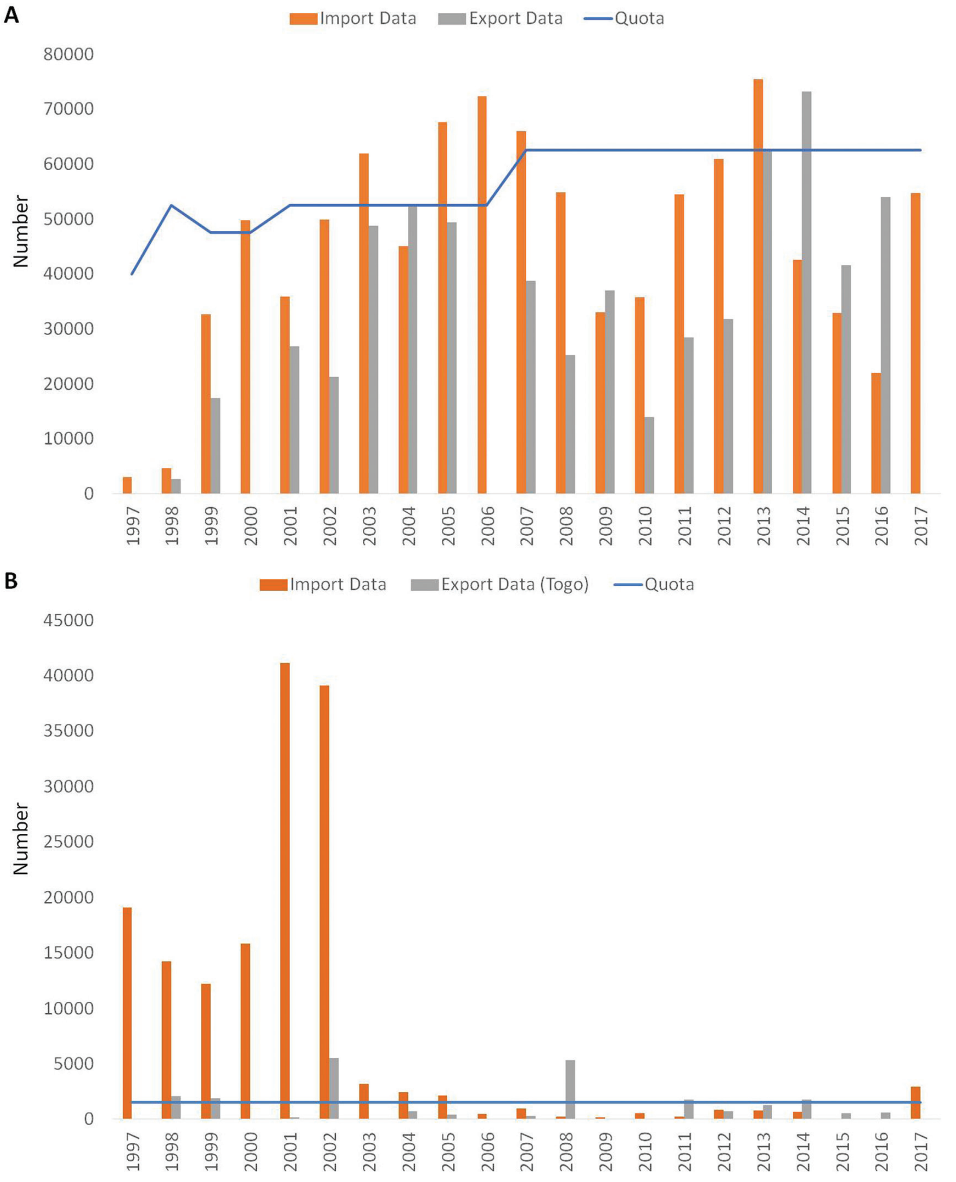

Figure 5. The number of ranched (**A**) and wild (**B**) specimens of ball pythons exported annually since 1997, according to export data (from Togo) and import data (from all importing countries) recorded on CITES trade database, in comparison with the annual quota set for ball python exports from Togo. Source: CITES trade database, https://trade.cites.org

and African softshell turtle (*Trionyx triunguis*)] were categorised on the IUCN Red List as Vulnerable, and two [the West African black turtle (*Pelusios niger*) and Geyr's spiny-tailed lizard (*Uromastyx geyri*)] as Near Threatened - wild populations of all but one of these species were reported to be decreasing. There was no evidence that the propor-

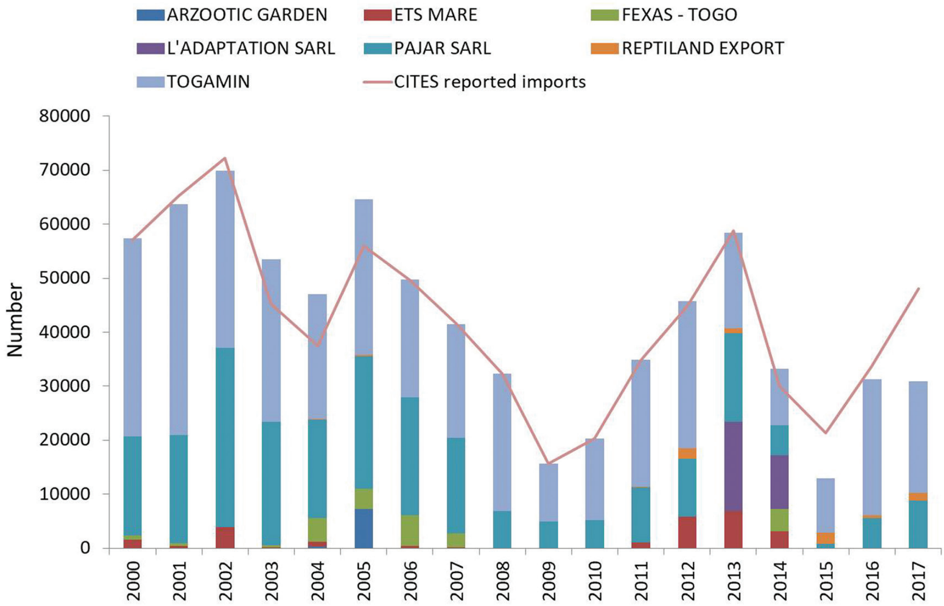

Figure 6. The number of ball pythons exported by the different reptile farms in Togo into the USA annually since 2000, as recorded on LEMIS trade database, shown against CITES-reported USA imports (Source: CITES trade database, https://trade.cites.org).

tion of threatened or declining species observed at farms differed from expectations as compared to all Togolese reptiles listed on the IUCN Red List (conservation status: $\chi^2 = 3.52$, df = 5, P = 0.707; population status: $\chi^2 = 0.427$, df = 2, P = 0.808).

With regards to the geographic distribution of these 46 species, a total of 10 species (22%) are considered as regionally endemic to West Africa, and 32 species (70%) are considered as widespread (extending to regions outside western Africa) according to information provided by the IUCN Red List (IUCN 2019), (please see Appendix I, and Appendix II). A total of 36 (78%) of these species (including three taxa in species complexes) are native to Togo. A total of nine species are considered endemic to other countries and / or regions in Africa than Togo [i.e. *Centrochelys sulcata,* Bell's hinge-back tortoise (*Kinixys belliana*), leopard tortoise (*Stigmochelys pardalis*), Peter's banded skink *(Scincopus fasciatus*), *Uromastyx geyri*, Gaboon viper (*Bitis gabonica*), green mamba (*Dendroaspis angusticeps*; this species is confined to central-eastern and southern Africa), black mamba (*D. polylepis*; a published distribution record of this species for Togo is pending) and the Northeast African carpet viper (*Echis pyramidum*) that is a regional endemic outside of West Africa]. A total of five taxa (11%) have unknown distributions as they were not identified to species level and 26 species (57%) are definitely or likely also supplied by another country or range state, and exported by Togo (please see Appendix I, and Appendix II). A total of 19 observed species (41%) are listed on CITES Appendix II, except the West African crocodile (*Crocodylus suchus*) that is listed on CITES Appendix I [Note: the West African populations of the Nile crocodile are still listed as *Crocodylus niloticus*, UNEP 2019].

Discussion

Togo remains a substantial source of live reptiles, both native and non-native, with at least 19 CITES-listed species currently held at reptile farms. In terms of the number of animals traded, the most notable species exported by Togo since 1978 under CITES is the ball python; with 1,657,814 live individuals - comprising 60% of all live reptiles – reported by importing countries. In total, 99% of the ball pythons legally exported from Togo under CITES have been intended for commercial use as exotic pets. Since the turn of the century, wild-sourced snakes exported from Togo have been largely replaced with specimens declared as "ranched (R)", to the extent that in the last 10 years 95% of all live exports were recorded using CITES source code "R". Ball pythons have been exported from Togo to 58 different countries since 1978. With regards to the global trade trend, CITES importer-reported quantities suggest that numbers of ball pythons exported from Togo since 1992 until 2019 (following a rapid increase in reported trade levels in the late 1980s/early 1990s) have overall remained relatively stable (at an annual average of 54,754 live animals), albeit with considerable fluctuation among years (Fig. 3). Importer-reported quantities by CITES showed the USA to be the largest importing country, responsible for 77% of exports since 1978 (also see Luiselli et al. 2012). At a regional level, over the last ten years, a decrease in the numbers imported by Europe, which was not mirrored by numbers imported into North America (USA and Canada), has meant that the proportion of ball python exports from Togo being imported into North America has effectively increased during this period.

Ball python ranching: boom or bust?

The provision of captive breeding and ranching operations as a replacement for the potentially unsustainable sourcing of wildlife from their natural habitats, as observed for ball python CITES exports reported from Togo, is not new (e.g., Rosen and Smith 2010; Harfoot et al. 2018). Indeed, such initiatives have been in operation for several decades, and in certain circumstances (e.g., crocodilians) deemed a successful conservation tactic overall (Nogueira and Nogueira-Filho 2011), albeit not one without associated animal welfare challenges (Dutton et al. 2013). However, in recent years researchers have also drawn attention to the fact that the actual numbers of species that receive overall net conservation benefits from this type of production system may be relatively few and far between (Tensen 2016). Specifically, in many cases data are lacking as to whether farm produced "products" represent a true substitute for wild sourced counterparts (e.g., bear bile; Dutton et al. 2011), whether farmed wild animals are cost efficient enough to combat poaching and black market prices (e.g., sea turtle meat; D'Cruze et al. 2015), and whether they are being effectively managed well enough to demonstrably disallow laundering [e.g., Tokay geckos (*Gekko gecko*); Nijman and Shepherd 2015; green tree pythons (*Morelia viridis*); Lyons and Natusch 2011].

In the context of ball pythons, recent scientific studies have raised conservation concerns regarding current production methods being implemented in West Africa to supply the international exotic pet trade. In Benin, Toudonou (2015) described a link between the legal hunting / ranching of ball pythons and the illegal trade of ball pythons as bush meat. Toudonou (2015) also stated "farmers and ball python collectors unanimously reported that this species is under severe threat in Benin" via reduced ball python hunting success rates, increased hunting localities, hunting effort and associated costs compared to 20 years previous. Similarly, recent research in Togo has identified a link between the legal hunting / ranching of ball pythons and the illegal trade of ball pythons as traditional medicine (D'Cruze et al. 2020b) and has also revealed that local hunters report a reduction in the availability of wild ball pythons between 2014 and 2018 (D'Cruze et al. 2020a). Furthermore, a recent genetic analysis of wild ball pythons has indicated that the largely unregulated wild release component of the python production process in Togo is resulting in "genetic pollution" that may be having a long-term negative impact on the conservation of wild populations (Auliya et al. 2020) and welfare of individual snakes (D'Cruze et al. 2020a). Our study now also draws attention to the fact that the number of ball pythons exported by Togo is likely to be consistently higher than officially reported to CITES by Togo (predominantly for ranched individuals, although some improvement has been seen in the last three to four years in terms of under-reporting) and that ball python quotas (ranched and wild-sourced) are still frequently exceeded. Indeed, in 2017, exporter-reporter quantities suggest that the wild-source quota was exceeded by over 1,000 individuals.

Conservation and animal welfare implications

The ball python has a relatively large distribution, fast reproductive rate, and occurs in a wide range of habitats including some areas with formal protected status and some areas inhabited by local communities who consider the species to be sacred (Toudonou 2015). Consequently, with regards to its conservation status, this species is currently classified as "Least Concern" with a population trend considered as "Unknown" according to the IUCN Red List of Threatened Species (Auliya and Schmitz 2010). However, this last conservation assessment was made almost ten years ago and is in need of updating. Given the conservation concerns (listed above) associated with current production systems there are questions regarding whether a higher conservation status may be more appropriate (Toudonou 2015), especially given that this species is also traded legally in relatively large volumes that specifically target the most vulnerable life stages (i.e., gravid females) (Toudonou 2015; D'Cruze et al. 2020a), it currently faces a variety of other threats to its survival in the wild [e.g., expanding agricultural mechanisation, pesticide use (Toudonou, 2015) and local subsistence use (Auliya and Schmitz 2010)], and detailed studies on population status and distribution are currently lacking (Auliya and Schmitz 2010). In addition, the causes of regional declines in several snake species distributed across the world still remain unknown (Reading et al. 2010).

Animal welfare impacts are associated with every step of a wildlife trade chain including capture, restraint, transport and subsequent captivity irrespective of a species' legal status, but they rarely feature in the relevant published scientific literature (Baker et al. 2013). In the context of ball python production systems in West Africa, D'Cruze et al. (2020a) have already drawn attention to the animal welfare concerns associated with removal of ball pythons from their burrows, transport and sub-standard care provided at "holding facilities" prior to their arrival at python farms. However, observations made during our visits to python farms also raise additional animal welfare concerns that have not been referred to by previous assessments of these types of facility in Togo (e.g., de Buffrénil 1995; Jenkins 1998; Ineich 2006). In particular, cases of high stocking density (including overt and crypto-over-crowding, Warwick et al. 2013), a lack of space and shelter, poor food, water, hygiene and substrate availability were some of the issues observed at the facilities visited during our fieldwork (Fig. 7). Detailed physical examinations and behavioural observations were not made during this study and data on duration, morbidity and mortality rates were not gathered. However, physical injuries and stress associated with sub-optimal captive conditions at python farms are likely resulting in some deaths and disease.

Other reptile species

Our study provides the most complete account of species diversity present at snake breeding farms in Togo that has been published to date. Many of these species, like ball pythons, are not currently considered threatened according to the IUCN Red List assessment, but although there was no evidence that farms were specifically selecting threatened species, they were not avoiding them either. And, for species that have been categorised as non-threatened, trade is being carried out in a manner that means that a non-threatened status is not necessarily still the case or will be in future. This is reflected by the fact that a number of species observed during our visits to ball python breeding farms are already currently involved in the CITES Review of Significant Trade, a procedure [defined in Res. Conf. 12.8 (Rev. CoP18)] designed to identify species that may be subject to unsustainable levels of international trade, and to identify problems and solutions concerning effective implementation of the Convention (CITES 2019). Specifically, the graceful chameleon (*Chamaeleo gracilis*) and *Kinixys homeana* from Togo have been included in this process since 2010, as have the ornate Nile monitor (*Varanus ornatus*) since 2013, and *Uromastyx geyri* since 2017, respectively due to international concerns relating to sourcing, high volumes and sharp increases in trade (AC30 Doc. 12.2; https://cites.org/sites/default/files/eng/com/ac/30/E-AC30-12-02.pdf).

Figure 7. Images of species encountered during visits to the eight snake farms: **A** juvenile *Python regius* **B** example housing conditions **C** juvenile *Varanus ornatus*; and **D** juvenile *Varanus exanthematicus*. Images show typical barren enclosures with overt overcrowding, no shelter and species exhibiting interactions with transparent boundaries. Photos: **A, B** (Neil D'Cruze) **C, D** (M. Auliya).

Similarly, a number of species observed during our visits to ball python farms are already currently involved in the CITES Review of Trade in Animal Specimens Reported as Produced in Captivity, a procedure (defined in Resolution Conf. 17.7 and Decision 17.105) designed to help prevent the inadvertent misuse of CITES source codes, and deliberate fraudulent claims that wild-caught specimens were captive bred. Specifically, *Varanus exanthematicus* and *Centrochelys sulcata* from Togo have been included in this process due to recent high volumes of trade, shifts, and international concerns that these species are not being "ranched" in conformity with CITES requirements [as stated in Res. Conf.11.16 (Rev. CoP15)]. In the case of *C. sulcata* there are also specific concerns regarding the questionable use of source code "F" (born in captivity) and "R" (ranched) given that this species is not native to Togo.

Our field visits now draw attention to a longer list of species for potential consideration in future CITES review procedures. Togo has a well-established infrastructure for intercontinental shipments and has been identified as one of the major hubs for the export of live African reptiles (Jensen et al. 2018), thus species not native to Togo (please see Appendix I) are additionally very likely all sourced from range States, and in both cases shipped abroad via Togo's capital, Lomé. In particular, the trade of species that are also not native to Togo and not listed on the CITES Appendices, i.e. *Echis pyramidum* and *Scincopus fasciatus*, those already considered as Threatened, native to Togo and listed on CITES i.e. *Cyclanorbis senegalensis* and *Trionyx triunguis*, are arguably of most immediate interest in this regard. Those that have triggered a constant international demand e.g., mud turtles (Pelomedusidae spp.), hinge back tortoises (*Kinxys* spp.; CITES Appendix II), the Fat-tail gecko (*Hemitheconyx caudicinctus*), rough-scaled plated lizard (*Broadleysaurus* [*Gerrhosaurus*] *major*), fire skink (*Mochlus* [*Lepidophyris*] *fernandi*, Mueller's sand boa (*Eryx muelleri*; CITES Appendix II) also likely warrant increased attention from an international trade policy perspective.

Limitations

CITES trade records are known to be incomplete and error-prone (e.g., Phelps et al. 2010) and by definition only reflect legal trade. We have not attempted here to quantify any illegal export of ball pythons out of Togo. However, as regards legal trade, whilst most studies based on CITES trade records tend to use either importer- or exporter-reported quantities we analysed both quantities separately, which provided a more detailed and nuanced insight into trade patterns and trends. The two quantities may differ for a number of reasons including, for example, the use of different codes by exporter and importer and/or exports being recorded as imports the year after they were exported (CITES 2013). There is, therefore, a significant risk that restricting analyses to one quantity under- or over-estimates trade and to misses pertinent details (such as an undeclared destination country) – the limitation associated with comparative analyses of both quantities is that it is not possible from the CITES trade database to verify which is more accurate.

Our time at the eight snake facilities in Togo was limited and intermittent. It is therefore very likely that species lists compiled are incomplete and thus should be treated as an initial conservative list only. The assumption, that these species lists are incomplete, is also supported by species groups such as the Egyptian cobra (*Naja haje*) (Trape et al. 2009) and the forest cobra (*Naja melanoleuca*) (Wüster et al. 2018) that have recently been shown to include additional new species. There are also taxonomic uncertainties brought about by superficial "look-alike" species and or those that may have been introduced through trade activities (e.g., Pelomedusidae spp.) (Vargas-

Ramírez et al. 2010; Wong et al. 2010). However, as these are the species that were presented by owners when asked and aware of our visit in advance – it likely shows a good representation and is ultimately the only existing data available. Ideally, we would have compared the species observed on farms with those that occur in western Africa (particularly with respect to their conservation and their national legislative status) in order to better understand which species are selected by python breeding farms. But this analysis was limited because the majority of Togolese, and indeed West African reptiles, has not yet been assessed for inclusion on the IUCN Red List. Should a national red list assessment be initiated for Togo, we would recommend that the following species should be made a priority given the commercial trade activity observed during this study: chameleons (*Chamaeleo* spp.), monitor lizards (*Varanus* spp.), Calabar ground boa (*Calabaria reinhardtii*), *Eryx muelleri*, Northern African rock python (*Python sebae*), *Kinxys* spp. and non-CITES listed species, e.g., *Mochlus fernandi*, *Hemitheconyx caudicinctus*, *Broadleysaurus major*, Jameson's green mamba (*Dendroaspis jamesoni*), bush vipers (*Atheris* spp.), rhinoceros viper (*Bitis nasicornis*) and the West African gaboon viper (*B. rhinoceros*) (cf. Segniagbeto et al. 2011, 2015).

Recommendations

In light of recent concerns regarding the hunting and release practices that underpin python farms in West Africa (Toudonou, 2015; Auliya et al. 2020; D'Cruze et al. 2020a,b), and concerns regarding the sustainability and compliance of trade in other reptile species produced via these farms (AC30 Doc. 12.2; AC30 Doc. 13.1), it is recommended that ball pythons, and other reptile species, exported using source code "R" (ranched) and "W" (wild) in Togo (but also Benin and Ghana) should be reconsidered for inclusion in future CITES procedures (within the CITES e.g., Reviews of Significant Trade procedures or Trade in Animal Specimens Reported as Produced in Captivity; see https://cites.org/eng/imp/sigtradereview, https://cites.org/sites/default/files/document/E-Res-17-07.pdf). However, given that such a process can be lengthy and time consuming, in the short term we recommend that the Togo export quotas for ball pythons (currently set at 62,500 ranched snakes) be urgently revised (cf. Auliya et al. 2020; D'Cruze at al. 2020a). In addition, given that recent conservation assessments (and biological field data required to underpin them) are currently lacking for reptile species in Togo, it is recommended that Togo should consider composing a National Red List for its reptiles. Such information would be vital to help inform and define any future common legal tools in a tripartite agreement between the three main ball python range states in West Africa that are predominantly involved in their commercial export (Benin, Ghana and Togo). Furthermore, we recommend studies focused on morbidity and mortality rates of species during collection and transport from the point of harvest (within Togo and other range States) to the exporter's premises, prior to export. This information is lacking and vital to understand and potentially revise current trade practices in this regard.

Conclusions

The aim of this paper was to provide information specific to the ball python trade in Togo to inform current management practice. Additional information is needed – notably, on the understanding, attitudes and behaviour of consumers, as well as on the population dynamics and status of ball pythons. More broadly, this system is of interest to those concerned with sustainable use, substitutability, and the links between the various forms of "captive" and wild populations, and as such has parallels with a number of other traded wildlife species including ranched crocodilians (Jenkins et al. 2004), and captive bred lions (Coals et al. 2019). We envisage that the wider relevance of this large-scale reptile ranching system will be addressed fully elsewhere.

Legal and sustainably managed commercial wildlife trade has been proposed as a vital conservation tool, that in some cases is necessary to ensure the long-term survival of wild populations (Dutton et al. 2013). Conversely, when not properly managed, and / or required baseline information is lacking, wildlife trade (both legal and illegal) can impede conservation efforts (Bush et al. 2014). The severe threat of direct exploitation of organisms, such as harvesting animals for trade, was clearly indicated by Maxwell et al. (2016), and was calculated as the 2nd largest driver of change to nature in the UN global assessment report on biodiversity and ecosystem services, closely following changes in land and sea use (IPBES 2019).

The ball python is the most commercially traded live wild animal under CITES from Africa over the past five years which have ostensibly involved ranching operations (Auliya et al. 2020; D'Cruze et al. 2020a). The shift away from wild-sourced ball pythons in West Africa was once considered to provide a degree of protection for the ball python; however, in light of recent research, there is growing concern that the trade in this species is improperly managed and may not confer any significant net conservation benefits. In conclusion, we note that Togo's ranching operation per se is inappropriately managed and lacks overall monitoring (cf. Auliya et al. 2020; D'Cruze et al. 2020a), that impedes a legal, sustainable and traceable trade of the ball python. We recommend further scrutiny and research is required in this regard to ensure the long-term survival of wild ball python populations.

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References

Affo AAB (2001) Commerce international des Reptiles élevés en captivité au Togo: cas des pythons, tortues et caméléons. Unpublished training report, School for the training of wildlife specialist, Garoua, Cameroon, 1–41.

Affre A, Ineich I, Ringuet S (2005) West Africa, Madagascar, Central and South America: Main origins of the CITES-listed lizard pet market in France. Herpetological Review 36: 133–137.

Auliya M, Schmitz A (2010) *Python regius*. The IUCN Red List of Threatened Species 2010: e.T177562A7457411.

Auliya M, Hofmann S, Segniagbeto GH, Assou D, Ronfot D, Astrin JJ, Forat S, Ketoh GKK, D'Cruze N (2020) The first genetic assessment of wild and farmed Ball pythons (Reptilia: Serpentes: Pythonidae) in southern Togo. Nature Conservation 38: 37–59. https://doi.org/10.3897/natureconservation.38.49478

Baker SE, Cain R, van Kesteren F, Zommers ZA, D'Cruze N, MacDonald DW (2013) Rough Trade: Animal Welfare in the Global Wildlife Trade. Bioscience 63(12): 928–938. https://doi.org/10.1525/bio.2013.63.12.6

Bush ER, Baker SE, Macdonald DW (2014) Global Trade in Exotic Pets 2006–2012. Conservation Biology 28(3): 663–667. https://doi.org/10.1111/cobi.12240

Can OE, D'Cruze N, Macdonald DW (2019) Dealing in deadly pathogens: taking stock of the legal trade in live wildlife and potential risks to human health. Global Ecology and Conservation 17: e00515. https://doi.org/10.1016/j.gecco.2018.e00515

CITES (2013) A guide to using the CITES Trade Database. https://trade.cites.org/cites_trade_guidelines/en-CITES_Trade_Database_Guide.pdf

CITES (2019) List of Contracting Parties. https://www.cites.org/eng/disc/parties/chronolo.php

Coals P, Burnham D, Loveridge A, Macdonald DW, t'Sas-Rolfes M, Williams VL, Vucetich JA (2019) The Ethics of Human–Animal Relationships and Public Discourse: A Case Study of Lions Bred for Their Bones. Animals (Basel) 9: 52. https://doi.org/10.3390/ani9020052

D'Cruze N, Alcock R, Donnelly M (2015) The Cayman Turtle Farm: Why we can't have our green turtle and eat it too. Journal of Agricultural & Environmental Ethics 28(1): 57–66. https://doi.org/10.1007/s10806-014-9519-6

D'Cruze N, Harrington L, Assou D, Ronfot D, Macdonald DW, Segniagbeto GH, Auliya M (2020a) Searching for Snakes: A Socio-economic Survey of Ball Python Hunting in Togo, West Africa. Nature Conservation 58: 13–36. https://doi.org/10.3897/natureconservation.38.47864

D'Cruze N, Délagnon P, Coulthard E, Norrey J, Megson D, Macdonald DW, Harrington L, Ronfot D, Segniagbeto GH, Auliya M (2020b) (in press) Snake Oil and Pangolin Scales: Insights into Wild Animal Use at "Marché des Fétiches" Traditional Medicine Market, Togo. Nature Conservation 39: 45–71. https://doi.org/10.3897/natureconservation.39.47879

de Buffrénil V (1995) Les élevages de Reptiles du Bénin, du Togo et du Ghana. Rapport d'étude réalisée pour le Secrétariat de la CITES, Juin 1995, 1–23.

Dutton AJ, Hepburn C, Macdonald DW (2011) A stated preference investigation into the Chinese demand for farmed vs. wild bear bile. PLoS One 6(7): e21243. https://doi.org/10.1371/journal.pone.0021243

Dutton AJ, Gratwicke B, Hepburn C, Herrera EA, Macdonald DW (2013) Tackling unsustainable wildlife trade. Key topics in conservation biology 2, Wiley-Blackwell, Oxford, 74–92. https://doi.org/10.1002/9781118520178.ch5

Ern H (1979) Die Vegetation Togos. Gliederung, Gefährdung, Erhaltung. Willdenowia 9: 295–312.

Harfoot M, Glaser SAM, Tittensor DP, Britten GL, McLardy C, Malsch K, Burgess ND (2018) Unveiling the patterns and trends in 40 years of global trade in CITES-listed wildlife. Biological Conservation 223: 47–57. https://doi.org/10.1016/j.biocon.2018.04.017

Harrington LA (2015) International commercial trade in live carnivores and primates 2006–2012: Response to Bush et al. 2014. Conservation Biology 29(1): 293–296. https://doi.org/10.1111/cobi.12448

Harwood J (2003) West African Reptiles: species status and management guidelines for reptiles in international trade from Benin and Togo. A Report to the European Commission. Prepared for the European Commission. Directorate General E - Environment ENV E.3 – Development and Environment, UNEP-WCMC, 1–51.

Hulselmans JLJ, De Roo A, De Vree F (1970) Contribution à l'herpétologie de la République du Togo: Liste préliminaire des serpents récoltés par la première mission zoologique belge au Togo. Revue de Zoologie et de Botanique Africaines 81: 193–196.

Hyndman RJ (2017) CRAN task view: Time series analysis. https://cran.r-project.org/web/views/TimeSeries.html

Ineich I (2006) Les élevages de reptiles et de scorpions au Bénin, Togo et Ghana, plus particulièrement la gestion des quotas d'exportation et la définition des codes 'source' des spécimens exportés. Rapport d'étude réalisée pour le Secrétariat de la CITES. Projet CITES A-251: 1–113.

IPBES (2019) Global assessment report on biodiversity and ecosystem services of the Intergovernmental Science- Policy Platform on Biodiversity and Ecosystem Services. IPBES Secretariat, Bonn, Germany.

IUCN (2019) The IUCN Red List of Threatened Species. Version 2019-2. https://www.iucnredlist.org

Jenkins RWG (1998) Management and use of *Python regius* in Benin and Togo. Report prepared for Directorate General XI The Commission of the European Union, 11.

Jenkins RWG, Jelden D, Webb GJW, Manolis SC (2004) Review of Crocodile Ranching Programs. Conducted for CITES by the Crocodile Specialist Group of IUCN/SSC. January – April 2004. IUCN-SSC Crocodile Specialist Group. https://www.iucncsg.org/365_docs/attachments/protarea/CSG_-2b73a2ea.pdf

Jensen TJ, Auliya M, Burgess ND, Aust PW, Pertoldi C, Strand J (2018) Exploring the international trade in African snakes not listed on CITES: Highlighting the role of the internet and social media. Biodiversity and Conservation. https://doi.org/10.1007/s10531-018-1632-9

Loveridge A (1939) Revision of the African snakes of the genera *Mehelya* and *Gonionotophis*. Bulletin of the Museum of Comparative Zoology 86: 131–162.

Loveridge A (1944) Further revisions of African snake genera. Bulletin of the Museum of Comparative Zoology 95: 121–247.

Loveridge A (1958) Revision of five snake genera. Bulletin of the Museum of Comparative Zoology 119: 1–198.

Luiselli L (2006) Why do males and females of *Python regius* differ in ectoparasite load? Amphibia-Reptilia 27(3): 469–471. https://doi.org/10.1163/156853806778190105

Luiselli L, Angelici FM (1998) Sexual size dimorphism and natural history traits are correlated with intersexual dietary divergence in royal pythons (*Python regius*) from the rainforests of southeastern Nigeria. The Italian Journal of Zoology 65(2): 183–185. https://doi.org/10.1080/11250009809386744

Luiselli L, Akani GC, Eniang EA, Politano E (2007) Comparative ecology and ecological modeling of sympatric pythons, *Python regius* and *P. sebae*. In: Henderson RW, Powell R (Eds) Biology of the boas and pythons. CPG/Biological Science Press, New York, 88–100.

Luiselli L, Bonnet X, Rocco M, Amori G (2012) Conservation Implications of Rapid Shifts in the Trade of Wild African and Asian Pythons. Biotropica 44(4): 569–573. https://doi.org/10.1111/j.1744-7429.2011.00842.x

Lyons JA, Natusch DJD (2011) Wildlife laundering through breeding farms: Illegal harvest, population declines and a means of regulating the trade of green pythons (*Morelia viridis*) from Indonesia. Biological Conservation 144(12): 3073–3081. https://doi.org/10.1016/j.biocon.2011.10.002

Maxwell SL, Fuller RA, Brooks TM, Watson JEM (2016) The ravages of guns, nets and bulldozers. Nature 536(7615): 143–145. https://doi.org/10.1038/536143a

Nijman V, Shepherd CR (2015) Adding up the numbers: An investigation into commercial breeding of Tokay Geckos in Indonesia. TRAFFIC. Petaling Jaya, Selangor, Malaysia.Nijman V, Shepherd CR, Sanders KL (2012) Over-exploitation and illegal trade of reptiles in Indonesia. The Herpetological Journal 22: 83–89.

Nogueira SS, Nogueira-Filho SL (2011) Wildlife farming: An alternative to unsustainable hunting and deforestation in Neotropical forests? Biodiversity and Conservation 20(7): 1385–1397. https://doi.org/10.1007/s10531-011-0047-7

Novinyo SK, Kossi A, Habou R, Raoufou RA, Dzifa KA, André BB, Ali M, Sokpon N, Kouami K (2015) Spatial Distribution of *Pterocarpus erinaceus* Poir. (Fabaceae) Natural Stands in the Sudanian and Sudano-Guinean Zones of West Africa: Gradient Distribution and Productivity Variation across the Five Ecological Zones of Togo. Annual Research & Review in Biology 6(2): 89–102. https://doi.org/10.9734/ARRB/2015/14771

Phelps J, Carrasco LR, Webb E, Koh L, Pascual U (2010) Boosting CITES. Science 330(6012): 1752–1753. https://doi.org/10.1126/science.1195558

R Core Team (2018) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna. https://www.R-project.org

Reading CJ, Luiselli LM, Akani GC, Bonnet X, Amori G, Ballouard JM, Filippi E, Naulleau G, Pearson D, Rugiero L (2010) Are snake populations in widespread decline? Biology Letters 6(6): 777–780. https://doi.org/10.1098/rsbl.2010.0373

Robinson JE, Griffiths RA, St. John FAV, Roberts DL (2015) Dynamics of the global trade in live reptiles: shifting trends in production and consequences for sustainability. Biological Conservation 184: 42–50. https://doi.org/10.1016/j.biocon.2014.12.019

Roman B (1984) Serpents des Pays de l'Entente. C.N.R.S.T., Ouagadougou, 45 pp.

Rosen GE, Smith KF (2010) Summarizing the Evidence on the International Trade in Illegal Wildlife. EcoHealth 7: 24–32. https://doi.org/10.1007/s10393-010-0317-y

Schmitz A, Mausfeld P, Hekkala E, Shine T, Nickel H, Amato G, Böhme W (2003) Molecular evidence for species level divergence in African Nile Crocodiles *Crocodylus niloticus* (Laurenti, 1786). Comptes Rendus. Palévol 2(8): 703–712. https://doi.org/10.1016/j.crpv.2003.07.002

Segniagbeto G, Van Waerebeek K (2010) A note on the occurrence and status of cetaceans in Togo. Scientific Committee document SC/62/SM11, International Whaling Commission, Agadir, Morocco.

Segniagbeto GH, Trape JF, David P, Glitho IA (2011) The snake fauna of Togo: Systematics, distribution and biogeography, with remarks on selected taxonomic problems. Zoosystema 33(3): 325–360. https://doi.org/10.5252/z2011n3a4

Segniagbeto GH, Trape JF, Afiademanyo KM, Rödel MO, Ohler A, Dubois A, David P, Meirte D, Glitho IA, Petrozzi F, Luiselli L (2015) Checklist of the lizards of Togo (West Africa), with comments on systematics, distribution, ecology, and conservation. Zoosystema 37(2): 381–402. https://doi.org/10.5252/z2015n2a7

Sternfeld R (1908) Neue und ungenügend bekannte afrikanische Schlangen. Sitzungsberichte der Gesellschaft naturforschender Freunde zu Berlin. R. Friedländer & Sohn, Berlin 4: 92–95. https://doi.org/10.5962/bhl.part.12852

Tensen L (2016) Under what circumstances can wildlife farming benefit species conservation? Global Ecology and Conservation 6: 286–298. https://doi.org/10.1016/j.gecco.2016.03.007

Toudonou ASC (2015) Ball Python *Python regius*. Laboratory of Applied Ecology, University of Abomey-Calavi, Cotonou, Benin. https://cites.org/sites/default/files/eng/com/ac/28/Inf/E-AC28-Inf-04.pdf

Trape JF, Chirio L, Broadley DG, Wuester W (2009) Phylogeography and systematic revision of the Egyptian cobra (Serpentes: Elapidae: *Naja haje*) species complex, with the description of a new species from West Africa. Zootaxa 2236(1): 1–25. https://doi.org/10.11646/zootaxa.2236.1.1

UNEP (2019) The Species+ Website. Nairobi, Kenya. Compiled by UNEP-WCMC, Cambridge, UK. www.speciesplus.net [Accessed August 2019]

Vargas-Ramírez M, Vences M, Branch WR, Daniels SR, Glaw F, Hofmeyr MD, Kuchling G, Maran J, Papenfuss TJ, Široký P, Vieites DR, Fritz U (2010) Deep genealogical lineages in the widely distributed African helmeted terrapin: evidence from mitochondrial and nuclear DNA (Testudines: Pelomedusidae: *Pelomedusa subrufa*). Molecular Phylogenetics and Evolution 56(1): 428–440. https://doi.org/10.1016/j.ympev.2010.03.019

Warwick C, Arena P, Lindley S, Jessop M, Steedman C (2013) Assessing reptile welfare using behavioural criteria. In Practice 35(3): 123–131. https://doi.org/10.1136/inp.f1197

Wong RA, Fong JJ, Papenfuss TJ (2010) Phylogeography of the African Helmeted Terrapin, *Pelomedusa subrufa*: Genetic Structure, Dispersal, and Human Introduction. Proceedings of the California Academy of Sciences 61: 575–585.

Wüster W, Chirio L, Trape JF, Ineich I, Jackson K, Greenbaum E, Barron C, Kusamba C, Nagy ZT, Storey R, Hall C, Wüster CE, Barlow A, Broadley DG (2018) Integration of nuclear and mitochondrial gene sequences and morphology reveals unexpected diversity in the forest cobra (*Naja melanoleuca*) species complex in Central and West Africa (Serpentes: Elapidae). Zootaxa 4455: 068–098. https://doi.org/10.11646/zootaxa.4455.1.3

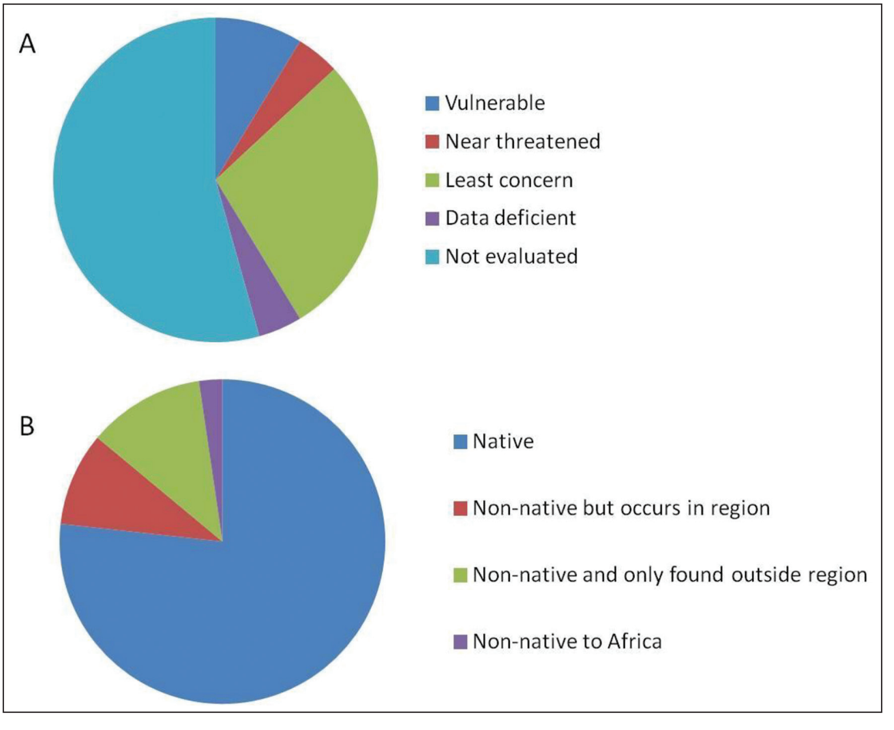
Appendix I

Reptile species observed across seven reptile farms in Togo (2018–2019), with associated IUCN Red List and CITES Appendices classifications. "Distribution" refers to the geographical distribution of the species and allocated to: "RE" = Restricted to western Africa i.e. Benin, Burkina Faso, Cape Verde, The Gambia, Ghana, Guinea, Guinea-Bissau, Ivory Coast, Liberia, Mali, Mauritania, the Niger, Nigeria, Senegal, Sierra Leone and Togo); "W" = widespread species also occurring in Africa outside of this region; when "RE" or either "W" are attached with a "1" means that the species is native to Togo; when "RE" is attached with a "2" implies that western African species extend beyond this region, and when "W" or either "RE" is attached with a "3" that the species is definitely, or likely also supplied by another country or range State; exotic species not distributed on the African continent were classified with "Other"; *): resurrected from *Crocodylus niloticus* (Schmitz et al. 2003); **): refer to species complexes (see text); NA = Not Applicable.

Family	Species	IUCN Conservation Status	Date Assessed (IUCN)	Population Status (IUCN)	CITES Status	Distribution Status
Crocodylidae	*Crocodylus suchus**	NE		NA	I	RE[2,3]
Agamidae	*Agama agama*	NE		NE	NL	W[1]
Agamidae	*Agama* sp.	NA		NA	NA	NA
Agamidae	*Uromastyx geyri*	NT	July 2012	Decreasing	II	RE[3]
Agamidae	*Uromastyx* sp.	NA		NA	NA	NA
Chamaeleonidae	*Chamaeleo senegalensis*	VU	July 2012	Unknown	II	RE[1]
Eublepharidae	*Hemitheconyx caudicinctus*	LC	July 2012	Unknown	NL	RE[2,3]
Gerrhosauridae	*Broadleysaurus major*	NE		NE	NL	W[1, 3]
Scincidae	*Mochlus fernandi*	NE		NE	NL	W[1,3]
Scincidae	*Scincopus fasciatus*	DD	June 2009	Unknown	NL	W[3]
Varanidae	*Varanus exanthematicus*	LC	June 2009	Unknown	II	W[1,3]
Varanidae	*Varanus niloticus*	NE		NE	II	W[1]
Varanidae	*Varanus ornatus*	NE		NE	II	RE[2,3]
Boidae	*Calabaria reinhardtii*	NE		NE	II	W[1,3]
Boidae	*Eryx muelleri*	NE		NE	II	W[1,3]
Colubridae	*Dasypeltis* cf. *gansi*	NE		NE	NL	W[1]
Colubridae	*Dasypeltis* sp.	NA		NA	NA	NA
Colubridae	*Dasypeltis confusa*	NE		NE	NL	W[1]
Colubridae	*Dispholidus typus*	NE		NE	NL	W[1]
Colubridae	*Philothamnus* cf. *irregularis*	LC	June 2009	Unknown	NL	W[1]
Elapidae	*Dendroaspis angusticeps*	NE		NE	NL	W
Elapidae	*Dendroaspis jamesoni*	NE		NE	NL	W[1,3]
Elapidae	*Dendroaspis polylepis*	LC	June 2009	Stable	NL	W[3]
Elapidae	*Dendroaspis viridis*	LC	July 2012	Stable	NL	W[1, 3]
Elapidae	*Naja melanoleuca***	NE		NE	NL	W[1]
Elapidae	*Naja nigricollis*	NE		NE	NL	W[1]
Elapidae	*Naja* sp.	NA		NA	NA	NA
Lamprophiidae	*Mehelya poensis*	NE		NE	NL	W[1]
Psammophiidae	*Psammophis* cf. *sibilans*	NE		NE	NL	W[1]
Psammophiidae	*Rhamphiophis oxyrhynchus*	NE		NE	NL	W[1]
Pythonidae	*Morelia viridis*	LC	June 2017	Stable	II	other

Family	Species	IUCN Conservation Status	Date Assessed (IUCN)	Population Status (IUCN)	CITES Status	Distribution Status
Pythonidae	*Python regius*	LC	June 2009	Unknown	II	W[1,3]
Pythonidae	*Python sebae*	NE		NE	II	W[1]
Viperidae	*Atheris chloroechis*	LC	July 2012	Unknown	NL	RE[1]
Viperidae	*Bitis arietans*	NE		NE	NL	W[1]
Viperidae	*Bitis gabonica*	NE		NE	NL	W[3]
Viperidae	*Bitis nasicornis*	NE		Unknown	NL	W[1, 3]
Viperidae	*Echis ocellatus*	NE		NE	NL	W[1]
Viperidae	*Echis pyramidum*	LC	June 2009	Unknown	NL	RE[3]
Viperidae	*Echis* sp.	NA		NA	NA	NA
Pelomedusidae	*Pelomedusa subrufa***	NE		NE	NL	W[1,3]
Pelomedusidae	*Pelusios c. castaneus*	NE		NE	NL	RE[1,3]
Pelomedusidae	*Pelusios niger*	NT	May 2018	Decreasing	NL	RE[1,3]
Testudinidae	*Centrochelys sulcata*	VU	Aug 96	Unspecified	II	W[3]
Testudinidae	*Kinixys belliana*	NE	Aug 96	NE	II	W[3]
Testudinidae	*Kinixys erosa*	DD	Aug 96	Unspecified	II	W[1,3]
Testudinidae	*Kinixys homeana*	VU	January 2006	Decreasing	II	RE[1,3]
Testudinidae	*Kinixys nogueyi*	NE		NE	II	RE[1,3]
Testudinidae	*Stigmochelys pardalis*	LC	Aug 14	Unknown	II	W[3]
Trionychidae	*Cyclanorbis senegalensis*	VU	May 2016	Decreasing	II	W[1]
Trionychidae	*Trionyx triunguis*	VU	June 2016	Decreasing	II	W[1]

Appendix II



Conservation (**A**) and distribution (**B**) status of reptile species (n = 46 and n = 43, respectively) observed on Togolese farms. Conservation status as categorised by the IUCN Red List (IUCN 2019). Distribution status omitted for three taxa included in species complexes, thus taxonomic status here is considered uncertain.

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Evaluation and sensitivity analysis of the ecosystem service functions of haze absorption by green space based on its quality in China

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Abstract

Evaluation of the ecosystem service functions of haze absorption by green space is important for controlling haze. In this study, the ecosystem service functions of haze absorption by green space in China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 are analyzed based on green space quality and sensitivity using a geographic information system (GIS) and Moderate Resolution Imaging Spectroradiometer (MODIS) imagery. The results showed that the benchmark ecosystem service functions of haze absorption by green space when considering only the area of green space showed a trend that increases first and then decreases in 2001–2018, with 9000458.55 million Kg, 9145110.75 million Kg and 7734526.75 million Kg in 2001, 2013 and 2018, respectively. However, the corrected functions based on green space quality were 7724215.34 million Kg, 8320301.79 million Kg and 6510132.55 million Kg in the corresponding years. This indicated large differences between ecosystem service functions of haze absorption based on the quality and area of green space; only considering the area of green space to evaluate ecosystem service functions will result in overestimation. In terms of the spatial distribution of the ecosystem service functions of haze absorption by green space, there were greater differences in the benchmark and corrected functions, and the spatial distributions of the maximum, intermediate and minimum ecosystem service functions were notably different. However, the benchmark and corrected functions all showed a consistent trend in the rank of their contribution rates and ecosystem service functions as well as consistent distribution trends: the spatial distribution of ecosystem service functions of haze absorption by green space was very different in the same year, but there was little difference among different years. The change coefficients for the ecosystem service functions of haze absorption by arable land and grass land remained stable, whereas the coefficient of sensitivity for forest cover was elastic. Patch density (PD) and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services showed a significant negative correlation, with correlation coefficients of -0.407, -0.511, -.330 and -0.332, respectively. In contrast, the area-weighted mean shape index (SAPE_AM) and ecosystem service functions exhibited significant positive relationships with correlation coefficients of 0.650, 0.634, 0.568 and 0.570, respectively. The results provide an improved method for evaluating the ecosystem service functions of haze absorption by green space as well as a reference for the prevention and control of haze and the coordinated development of regional societies, the economy and the environment.

Keywords

ecosystem service functions, haze absorption, quality of green space, sensitivity analysis

Introduction

In recent years, the frequent occurrence of haze in China has seriously threatened human health and environmental safety, becoming a major livelihood and environmental problem that cannot be ignored and needs to be solved. Exploring haze absorption from the perspective of ecosystem services is of great practical significance for scientific formulation of effective haze control policies (Hong et al. 2013; Song et al. 2019).

Haze, a kind of disastrous weather occurring in the near-ground atmospheric layer, is the result of interaction between specific climatic conditions and human activities (Chuai et al. 2019). Haze is composed of dust, sulfuric acid (H_2SO_4), nitric acid (HNO_3), organic hydrocarbons and other particles in the air, and of these materials, SO_2, NO_X and respirable particulate matter are the main components; the first two are gaseous pollutants, while particulate matter is the main hazardous component (Yu et al. 2018). Sulfur dioxide and NO_X are the main gaseous components of haze. Atmospheric SO_2 is mainly derived from the combustion of sulfur-containing fuel, which is harmful to the human respiratory tract, and high levels of SO_2 can damage leaf tissue. Furthermore, SO_2 is involved in the formation of H_2SO_4 fumes and acid rain, which is very harmful to human health. Nitrogen oxides are mainly derived from emissions from automotive exhaust and stationary combustion sources, and they can weaken the ability of blood to transport oxygen, seriously endanger human health, and contribute to atmospheric photochemical pollution (Sun et al. 2018). Areas with high densities of economic and social activities will inevitably discharge a large amount of fine particles ($PM_{2.5}$), and once the discharge exceeds the atmospheric circulation capacity and bearing capacity, fine particulates will accumulate, contributing to a wide range of haze events (Green and Xu 2007; Waters et al. 1998). There are two main aspects of haze production. The first includes human factors such as automobile exhaust, coal waste gas, industrial emissions, construction and road traffic dust, climate change, waste incineration, and even volcanic eruptions (Hansen et al. 2019). The role of different sources of pollution varies in different haze regions. In addition, haze is affected by meteorological factors such as weather that is not conducive to the spread of pollutants, and when pollutants accumulate under static weather conditions, haze is readily formed. Secondly, meteorological factors, the static wind in the horizontal direction and the inverse temperature in the vertical direction cause pollutants to gather and lead to the formation of haze weather (Li et al. 2019; Wu et al. 2016).

The hazards of haze include the following aspects: on the one hand, haze reduces visibility, increases the frequency of traffic accidents, and has an important impact on highways, railways, aviation, shipping, and power supply systems (Xue et al. 2018). On the other hand, haze also causes a decline in air quality, threatens human health, and increases the incidence and mortality of diseases in the respiratory tract, cardiovascular and reproductive systems (Ramakreshnan et al. 2018). Furthermore, haze can result in a weakening of near-surface ultraviolet light, resulting in an increase in the infectious bacteria in the air. Due to the reduced sunshine during haze weather, the ultraviolet radiation received by children is insufficient and not conducive to growth. Additionally, haze weather will reduce crop yields and quality but can also impact the atmospheric radiation budget, thereby impacting the climate system of the earth (Thach et al. 2010). In 2017, China implemented new air quality standards and monitored 338 cities, of which only 99 meet the annual average air quality standards and 239 exceed them. The frequent occurrence of haze weather affects the physical and mental health of the public and the sustainable development of the ecological environment (Zhang et al. 2016).

Haze is affected by pollution sources (Wang et al. 2015), meteorological conditions (Bei et al. 2016) and vegetation coverage (Ye et al. 2016; Zhang 2019). Reducing pollutant emissions is accomplished by actions such as reducing vehicle pollution and dust, controlling industrial pollution and the emission of NH_3 in agricultural areas, reducing the unorganized combustion of biomass and concentration of air pollutants, and coping with haze pollution (An et al. 2019; Yang et al. 2016). Some researchers have also explored the response mechanism of haze weather to meteorological conditions; rainfall through wet sedimentation and wind speed can accelerate the diffusion of pollutants to alleviate haze (Gao et al. 2015; Zhang et al. 2015a). However, meteorological conditions are external causes and are uncontrollable, while pollution sources are internal causes and are closely related to human activities, although pollution source treatment methods are not yet complete and immature. Previous studies have shown that vegetation leaf area (Gómez-Moreno et al. 2019; Setälä et al. 2013), vegetation coverage (De Carvalho and Szlafsztein 2019; Zhang 2019) and plant community structure (Pandey et al. 2014; Selmi et al. 2016) can absorb and block air pollutants, and vegetation coverage is relatively stable, which can effectively alleviate haze pollution. Therefore, there is still important significance for research on haze absorption by green space. Green space can effectively reduce haze, and it not only has a very important dust retention function but can absorb and convert toxic substances, be used to reduce the concentration of atmospheric particulate matter, and keep the air fresh through photosynthesis (Freer-Smith et al. 2004; Liu and Shen 2014).

Green space is an important part of social, economic and natural systems (Rysgaard et al. 1999). These spaces are completely undeveloped or basically undeveloped natural areas where the natural landscape is restored or where the land is reserved to offset urban construction. They primarily include arable land, forest cover and grass land and provide important ecological service functions such as air purification, water source conservation, climate regulation and biodiversity maintenance (Green et al. 2016). With rapid urbanization to meet the needs of the expanding population on limited land, vegetation is gradually being replaced by buildings (Cuffney et al. 2010). Therefore, green space is constantly being reduced and destroyed, and ecosystem service functioning is being severely diminished or is disappearing, thus weakening the maintenance and regulation of the urban environment. Thus, increasing the footprint of the urban environment results in more serious air pollution and increased haze in cities, and many countries are seeking sustainable social, economic and environmental development to maintain the various types of natural resources and simultaneously achieve both economic and ecological benefits. Urban green space (UGS) can, to a certain extent, alleviate the adverse effects of urbanization, produce urban cooling effects and increase moisture availability, and ease urban heat island effects as well as reduce surface runoff and maintain high evaporation rates and surface permeability. A reasonable amount of green space can control the unlimited expansion of a city and improve the urban environment. Therefore, green space is the core of the healthy development of urban ecosystems (Margaritis and Kang 2016; Park et al. 2017). Green vegetation plays a key role in UGS ecosystems and air purification. First, green vegetation has the unique physiological function of performing photosynthesis, relying on leaf pores to convert gas pollutants, such as sulfur dioxide (SO_2) and nitrogen oxides (NO_X), into non-toxic substances through redox processes; these products are then accumulated in plant organs or excreted by the root system. Second, foliage secretes bactericides and mucus that can absorb particles and retain dust. Third, vegetation can reduce wind speeds, reducing sedimentation. Finally, vegetation blocks and inhibits dust, thereby reducing particulate levels. Haze is mainly composed of SO_2, NO_X, and respirable particulate matter, and green space can purify the air of these materials (Han and Zhou 2015).

Previous studies on green space have focused on the impacts of heat island mitigation (Alavipanah et al. 2015; Heusinkveld et al. 2014), climate regulation (Maimaitiyiming et al. 2014), and ecosystem services monitoring and evaluation (Kopperoinen et al. 2014). Kuttler and Strassburger (Kuttler and Strassburger 1999) investigated the influence of traffic-induced pollutants (e.g., carbon monoxide (CO), nitrogen oxide (NO), nitrogen dioxide (NO_2) and ozone (O_3)) on the air quality of urban green areas in the city of Essen, North Rhine-Westphalia (NRW), Germany. Zoulia et al. (Zoulia et al. 2009) monitored the effect of urban green areas on the heat island in Athens, Greece. Hamada and Ohta (Hamada and Ohta 2010) measured air temperatures in an urban green area that includes forest and grass land as well as the surrounding urban area for a full year in Nagoya, central Japan to elucidate seasonal variations in the differences in air temperature between urban and green areas. Mahmoud and El-Sayed (Mahmoud and El-Sayed 2011) studied sustainable urban green areas in Egypt, and the results revealed that greenways could play a more significant role in bringing nature into the city. Saphores and Li (Saphores and Li 2012) used a hedonic pricing analysis of the single-family housing market to estimate the functions of urban green areas in Los Angeles, California, USA. Larondelle and Haase (Larondelle and Haase 2013) evaluated the climate regulation, cooling and entertainment features of urban ecosystems in Europe, and the results showed that the core of the city does not necessarily provide fewer ecosystem services. Chen et al. (Chen et al. 2015) investigated the impact of reclaimed water irrigation on soil health in urban green areas. Ozimec et al. (Ozimec et al. 2016) monitored air pollution by using lichens in the green space of the university campus in Osijek, Croatia, and the results showed that the air is moderately polluted. Selmi et al. (Selmi et al. 2016) employed the i-Tree Eco model to estimate air pollution removal by urban trees in Strasbourg, France, and the model showed that public trees managed by the city removed approximately 88 t of pollutants during a one-year period (from July 2012 to June 2013).

The dust retention and atmospheric pollutant absorption effects of green space have mostly explored the functional effects of different plant species based on individual differences in the levels of green space and have been limited to small scales (Devuyst et al. 2001). Beckett et al. demonstrated that trees can act as biological filters, removing large amounts of airborne particles, thus improving the air quality in polluted environments due to their large leaf areas relative to the ground on which they stand and the physical properties of their surfaces (Beckett et al. 1998). Davies and Unam monitored and analyzed the relationship between smoke-haze from the 1997 Indonesian forest fires and three tree species (Davies and Unam 1999). McDonald et al. estimated the potential of urban tree planting to mitigate urban PM_{10} using an atmospheric transport model to simulate particulate transport and deposition across two UK conurbations, and the results indicated that increasing the total tree cover in West Midlands from 3.7% to 16.5% removed 110 t of primary PM_{10} from the atmosphere per year (McDonald et al. 2007). However, few studies have been carried out on the national scale, and the ecosystem service functions of haze absorption by green space based on its quality have not been explored. On this basis, the correlation between absorbing haze and the landscape pattern of green space and its sensitivity to the change of ecosystem service functions have been analyzed.

The objectives of this study were: 1) comparison and analysis of the spatial and temporal patterns of the benchmark and corrected values of ecosystem service functions of haze absorption based on the quality of green space; 2) sensitivity analysis of changes in ecosystem service functions; 3) determination of the relationship between the landscape pattern and the ecosystem service functions of haze absorption by green space, providing a scientific basis for the quantitative evaluation of air pollution regulation using service functions, green space planning and urban ecological construction of green space in China.

Materials and methods

Ecosystem service functions of haze absorption by green space per unit area

The main components of haze are SO_2, NO_X and particulate matter. The uptake of haze material by types of green space per unit area (Jin et al. 2005; Ye et al. 1998) (Table 1) and green space area were used to calculate the ecosystem service functions of haze absorption by green space (Han and Zhou 2015; Kuttler and Strassburger 1999).

Calculation of the ecosystem service functions of haze absorption by green space

The ecosystem service functions of haze absorption by green space include the absorption of SO_2, NO_X and respirable particulate matter. According to the various types of green space and the ecosystem service functions of haze absorption by each type of green space per unit area, the total ecosystem service functions of haze absorption by green space in China can be calculated from formula (1) (Han and Zhou 2015; Kuttler and Strassburger 1999).

$$ESF = \sum_{i=1}^{3} \sum_{j=1}^{3} A_i F_{ij} \tag{1}$$

where *ESF* is the total ecosystem service functions of haze absorption by green space; A_i is the area of green space type *i*; F_{ij} is the ecosystem service of absorbing haze component *j* by green space *i* per unit area; *i* is the green space type including forest cover, grass land and arable land; and *j* is the haze component including SO_2, NO_X and particulate matter.

Ecosystem service functions correction based on green space quality

Both the ecosystem itself and its spatial heterogeneity affect ecosystem service functions. Considering the ecological system, the quality of green space plays an important role in its function, and the vegetation coverage (normalized difference vegetation

Table 1. The uptake of haze components by green space per unit area ($kg \cdot ha^{-1} \cdot yr^{-1}$).

Ecosystem service	Green space types			
	Arable land	Forest cover	Grass land	Total
Absorption of sulfur dioxide	45.00	152.13	279.03	476.16
Absorption of nitrogen oxides	33.50	6.00	6.00	45.50
Dust retention	0.95	21655.00	1.20	21657.15
Total	79.45	21813.13	286.23	22178.81

Note: "yr" refers to annum.

index (NDVI)) and net primary productivity (NPP) affect the corresponding service functions. The above ecosystem service functions calculation is only based on the land use area, without considering the impact of green space quality, so the results cannot reflect the true ecosystem service functions of haze absorption by green space. Using NDVI and NPP as evaluation indicators of green space quality and the correction coefficient to adjust the ecosystem service functions, the formula for the calculation is as follows (Gao et al. 2012):

$$f_i = \frac{NDV_i - NDVI_{\text{min}}}{NDVI_{\text{max}} - NDVI_{\text{min}}} \tag{2}$$

$$Q_i = \left[\frac{NPP_i}{NPP_{\text{mean}}} + \frac{f_i}{f_{\text{mean}}} \right] / 2 \tag{3}$$

$$ESF' = ESF \times Q_i \tag{4}$$

where f_i and NPP_i are the NDVI and NPP of grid *I*, respectively; NPP_{mean} and f_{mean} are the mean NPP and NDVI values of various ecosystems in the study region, respectively; $NDVI_{\text{max}}$ and $NDVI_{min}$ are the maximum and minimum NDVI values for the entire growing season; Q_i is the green space quality coefficient; *ESF* is the ecosystem service functions before the green space quality correction; and *ESF`* is the ecosystem service functions after the green space quality correction.

Sensitivity analysis

To reflect the dependence of ecosystem service functions on the ecological functions index over time, the economic elasticity coefficient is selected to calculate the coefficient of sensitivity (formula (6)) (Kreuter et al. 2001).

$$CS = \left| \frac{\left(ESF_j - ESF_i \right) / ESF_i}{\left(F_{jk} - F_{ik} \right) / F_{ik}} \right| \tag{5}$$

where *ESF* is the total ecosystem service functions; *F* is the functions coefficient; *i* and *j* are the initial and adjusted functions coefficients, respectively; *k* is the green space type; and *CS* is the coefficient of sensitivity. If *CS* > 1, the *ESF* for *F* is flexible, indicating that the total ecosystem service functions increase faster than the functions coefficient and that the proportion of the total ecosystem service functions and the functions coefficient are increasing. However, if *CS* < 1, the *ESF* for *F* is inelastic. *CS* = 1 represents complete elasticity; *CS* = 0 represents complete inelasticity. A higher ratio indicates that the elasticity of the ecosystem service functions index is more important.

Landscape pattern indices

Landscape pattern indices are used to describe the spatial organization of a landscape and provide a quantitative measure of the composition and spatial configuration of landscape structure. The interaction between landscape patterns and ecological processes as well as green space impacts haze absorption to different degrees. Based on previous research (Fang et al. 2014), we selected the landscape-level indices of patch density (PD), the interspersion and juxtaposition index (IJI), the area-weighted mean shape index (SHAPE_AM), and Shannon's diversity index (SHDI) to study the relationship between landscape patterns and the ecosystem service functions of haze absorption by green space in China. Among these indices, SHAPE_AM was calculated by the formula from reference (Fang et al. 2014), and PD, IJI and SHDI were calculated by the following formulas.

$$PD = \frac{1}{A}\sum_{i=1}^{n} N_i \tag{6}$$

where PD is patch density; A is the total area of the landscape; N_i is the number of patches in landscape i; i is the landscape element; and n is the total number of patches in the landscape.

$$IJI = \frac{-\sum_{i=1}^{m}\sum_{k=1+1}^{m}\left[\left(\frac{e_{ik}}{E}\right)\cdot \ln\left(\frac{e_{ik}}{E}\right)\right]}{\ln(0.5[m-(m-1)])}\times 100 \tag{7}$$

$$SHDI = -\sum_{i=1}^{m}\left(P_i \cdot \ln P_i\right) \tag{8}$$

where m is the total number of landscape types; i and k are the numbers of patches of types i and k, respectively; e_{ik} is the total boundary length of the patch types between patch types i and k; E is the total boundary length of the landscape, including the background; and p_i is the perimeter of patch type i.

Correlation analysis

The ecosystem service functions of haze absorption by green space, including measures of the absorption of SO_2 and NO_X, dust retention and the total ecosystem service functions, were calculated for different provinces in China using a geographic information system (GIS). The calculations of landscape pattern indexes including PD, IJI, SHAPE_AM, and SHDI for provinces of China were performed in FRAGSTATS. Correlations between landscape patterns and the absorption of SO_2 and NO_X, dust retention and total ecosystem service functions were calculated as Pearson correlation coefficients as follows:

$$\rho_{x,y} = \frac{\mathrm{cov}(X,Y)}{\sigma_X \sigma_Y} \tag{9}$$

where *cov (X, Y)* represents the covariance between two variables, and σ_X and σ_Y refer to the variance of the two variables. The Pearson correlation coefficient is used to measure the correlation between two variables. The value of this coefficient falls between 1 and -1: 1 represents a full positive correlation of the variables; 0 indicates that the variables are independent; and -1 indicates a completely negative correlation.

Research data

A MODIS land cover classification product (mod12q1) was used for the land use data for China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018. The spatial resolution of this product is 500 m, and land use is divided into arable land, forest cover, grass land, construction land, unused land and water bodies. Because the ecosystem service functions of haze absorption by water bodies are relatively small (Han and Zhou 2015), and few studies have been conducted on haze absorption by water bodies, it is difficult to obtain ecosystem service functions for the absorption of SO_2 and NO_X and dust retention by this land use type per unit area (Liu and Yu 2016), so the functions were not included as green space in this study. Therefore, green space in this study includes arable land, forest cover and grass land (Han and Zhou 2015; Kuttler and Strassburger 1999). Both NDVI and NPP are MODIS data products for China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 with a spatial resolution of 500 m, and in addition, the NPP data of NTSG (Numerical Terra-dynamic Simulation Group) was used as a supplement; the resolution of the data was 1 km × 1 km, and the annual NPP of the terrestrial ecosystem was obtained by using the NPP estimation model established by Biome-BGC and light energy utilization model. A dataset of the boundaries of the provinces in China was also included in this study.

Results and analysis

Analysis of the ecosystem service functions of haze absorption by green space in China

As shown in Table 2, the total ecosystem service functions (benchmark values) of haze absorption by green space in China were 9000458.55 million Kg in 2001, 8784710.32 million Kg in 2004, 8900539.79 million Kg in 2007, 9179977.89 million Kg in 2010, 9145110.75 million Kg in 2013, 7761608.74 million Kg in 2016 and 7734526.75 million Kg in 2018. The ecosystem service functions of haze absorption by green space in China showed a trend of first increasing and then decreasing in 2001–2018, exhibiting an upward trend from 2001–2013 and increasing by 144652.20 million Kg (1.61%), primarily because the Chinese government invested 179 billion Yuan in a series of ecological restoration programs (Wang et al. 2007), including the Three North Shelterbelt Development Program, the Conversion from Cropland to Forest Program and the Natural Forest Protection Program, to restore degraded ecological environments and to foster stable and sustainable development. The area of forest cover increased by 6886085.77 ha (1.69%) between 2001 and 2013, but the ecosystem service functions fell by 1410584.00 million Kg from 2013–2018, a decrease of 15.42%, primarily because of adjustment of ecological land structure, the reduction of forest cover with high haze absorption ecological service function, the increase of grass land with low haze absorption function, and the reduction of arable land caused by the expansion of construction land. The ecosystem service functions decreased by 215748.23 million Kg from 2001–2004, a decrease of 2.40%, and the main reasons are a reduction in the area of forest cover and grass land. In contrast, the ecosystem service functions increased by 115,829.48 million Kg from 2004–2007, an increase of 1.32%, mainly due to the increase in forest cover and grass land and the decrease in arable land that is largely attributed to the Program for Conversion from Cropland to Forest and Grass Land in China. Additionally, the ecosystem service functions increased by 279438.10 million Kg from 2007–2010, an increase of 3.14%, primarily due to an increase in forest cover and a reduction in arable land and grass land. From 2010–2013, the ecosystem service functions decreased by 34,867.15 million Kg, a reduction of 0.38%, mainly due to the decrease in forest cover and grass land and the increase in arable land. In comparison, the ecosystem service functions fell by 1,383,502.01 million Kg from 2013–2016, a decrease of 15.13%, primarily because of the increase in grass land and decrease in arable land and forest cover. The ecosystem service functions decreased by 27081.99 million Kg from 2016–2018, a decrease of 0.35%, mainly attributable to a reduction in the area of forest cover and arable land.

The contributions to haze absorption by green spaces indicated that the types are very different (Table 2). The overall contribution of forest cover was the largest, and these proportions were 98.68%, 98.67%, 98.68%, 98.75%, 98.77%, 98.17% and 98.16% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Grass land had the next largest contribution, accounting for 1.15%, 1.13%, 1.14%, 1.08%, 1.05%, 1.67% and 1.67% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The total contribution of arable land was less than 1% from 2001–2018, mainly due to the large area of forest cover combined with the higher per-unit functions of respirable particulate matter and SO_2 in the haze, both of which resulted in a higher contribution to ecosystem service functions from the other types. In contrast, the relatively lower contributions from grass land and arable land were primarily due to the smaller per-unit functions of haze absorption.

The primary haze absorption ecological function by green space was primarily dust retention (Table 2), the functions of which accounted for 97.98%, 97.97%, 97.97%, 98.04%, 98.06%, 97.47% and 97.46% of the total functions in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Sulfur dioxide followed, accounting for 1.90%, 1.89%, 1.90%, 1.83%, 1.82%, 2.40% and 2.41% of the total functions in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The functions of NO_X absorption from 2001–2018 were less than 1% of the total, primarily due to the lower

Table 2. Ecosystem service functions (benchmark values) of haze absorption by green space in China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 (10^6 Kg).

Green space types	Ecosystem service	2001	2004	2007	2010	2013	2016	2018
Arable land	Absorption of sulfur dioxide	8591.57	9622.49	9006.41	8835.75	9329.70	6805.30	6864.45
	Absorption of nitrogen oxides	6395.95	7163.41	6704.77	6577.72	6945.45	5066.17	5110.20
	Dust retention	181.38	203.14	190.14	186.53	196.96	143.67	144.92
	Total	15168.90	16989.04	15901.31	15600.00	16472.11	12015.14	12119.58
	Percentage (%)	0.17	0.19	0.18	0.17	0.18	0.15	0.16
Forest cover	Absorption of sulfur dioxide	61945.79	60457.74	61255.74	63225.21	62993.37	53141.91	52954.45
	Absorption of nitrogen oxides	2443.14	2384.45	2415.92	2493.60	2484.46	2095.91	2088.52
	Dust retention	8817696.60	8605879.12	8719471.22	8999815.52	8966814.78	7564504.47	7537820.05
	Total	8882085.53	8668721.31	8783142.89	9065534.33	9032292.61	7619742.29	7592863.02
	Percentage (%)	98.68	98.67	98.68	98.75	98.77	98.17	98.16
Grass land	Absorption of sulfur dioxide	100608.06	96509.66	98942.51	96357.20	93922.48	126584.95	126285.53
	Absorption of nitrogen oxides	2163.38	2075.25	2127.57	2071.97	2019.62	2721.96	2715.53
	Dust retention	432.68	415.05	425.51	414.39	403.92	544.39	543.11
	Total	103204.12	98999.97	101495.59	98843.57	96346.03	129851.31	129544.16
	Percentage (%)	1.15	1.13	1.14	1.08	1.05	1.67	1.67
Total	Absorption of sulfur dioxide	171145.43	166589.89	169204.66	168418.15	166245.56	186532.16	186104.43
	Absorption of nitrogen oxides	11002.47	11623.11	11248.26	11143.30	11449.52	9884.05	9914.25
	Dust retention	8818310.65	8606497.31	8720086.87	9000416.45	8967415.67	7565192.53	7538508.07
	Total	9000458.55	8784710.32	8900539.79	9179977.89	9145110.75	7761608.74	7734526.75
	Percentage (%)	100.00	100.00	100.00	100.00	100.00	100.00	100
Percentage (%)	Absorption of sulfur dioxide	1.90	1.89	1.90	1.83	1.82	2.40	2.41
	Absorption of nitrogen oxides	0.12	0.13	0.13	0.12	0.13	0.13	0.13
	Dust retention	97.98	97.97	97.97	98.04	98.06	97.47	97.46
	Total	100.00	100.00	100.00	100.00	100.00	100.00	100
	Percentage (%)	100.00	100.00	100.00	100.00	100.00	100.00	100

per-unit area function of the absorption of SO_2 and NO_X by various types of green space. However, the effect of dust retention was clear: the function of respirable particulate matter absorption by forest cover was especially high, indicating that green space plays an important role in dust removal and retention. Respirable particulate matter is the most important component of haze, so planning a reasonable amount of green space is conducive to reducing haze.

The ecosystem service functions (corrected value) of haze absorption by green space in China increased by 596086.46 million Kg (7.72%) from 2001–2013 (Table 3), while decreasing by 1810169.25 million Kg (21.76%) from 2013–2018: 7724215.34 million Kg

Table 3. Ecosystem service functions (corrected values) of haze absorption by green space in China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 (10^6 Kg).

Green space types	Ecosystem service	2001	2004	2007	2010	2013	2016	2018
Arable land	Absorption of sulfur dioxide	10623.08	12214.32	11709.68	11281.32	11648.15	3912.29	3921.82
	Absorption of nitrogen oxides	7908.29	9092.88	8717.21	8398.32	8671.40	2699.42	2691.12
	Dust retention	224.27	257.86	247.20	238.16	245.91	1672.12	1884.59
	Total	18755.64	21565.06	20674.09	19917.80	20565.45	8283.83	8497.53
	Percentage (%)	0.24	0.29	0.27	0.24	0.25	0.13	0.13
Forest cover	Absorption of sulfur dioxide	53221.67	52114.40	53757.20	56257.75	57464.17	45259.90	45242.86
	Absorption of nitrogen oxides	2099.06	2055.39	2120.18	2218.80	2266.38	1776.64	1775.52
	Dust retention	7575858.33	7418243.69	7652088.58	8008029.84	8179758.25	6357175.05	6347113.37
	Total	7631179.06	7472413.48	7707965.96	8066506.39	8239488.80	6404211.47	6394131.50
	Percentage (%)	98.80	98.86	98.89	98.97	99.03	98.29	98.22
Grass land	Absorption of sulfur dioxide	72412.13	63058.21	63850.41	62667.81	58732.04	56177.81	56741.54
	Absorption of nitrogen oxides	1557.08	1355.94	1372.98	1347.55	1262.92	1246.38	1260.91
	Dust retention	311.42	271.19	274.60	269.51	252.58	45770.85	49501.07
	Total	74280.63	64685.34	65497.99	64284.87	60247.54	103195.05	107503.52
	Percentage (%)	0.96	0.86	0.84	0.79	0.72	1.58	1.65
Total	Absorption of sulfur dioxide	136256.89	127386.94	129317.30	130206.88	127844.36	105350.00	105906.22
	Absorption of nitrogen oxides	11564.44	12504.22	12210.37	11964.67	12200.70	5722.45	5727.55
	Dust retention	7576394.02	7418772.74	7652610.38	8008537.51	8180256.73	6404618.02	6398499.03
	Total	7724215.34	7558663.89	7794138.04	8150709.06	8320301.79	6515690.35	6510132.55
	Percentage (%)	100.00	100.00	100.00	100.00	100.00	100.00	100.00
Percentage (%)	Absorption of sulfur dioxide	1.76	1.69	1.66	1.60	1.54	1.62	1.63
	Absorption of nitrogen oxides	0.15	0.17	0.16	0.15	0.15	0.09	0.09
	Dust retention	98.09	98.15	98.18	98.26	98.32	98.30	98.29
	Total	100.00	100.00	100.00	100.00	100.00	100.00	100.00
	Percentage (%)	100.00	100.00	100.00	100.00	100.00	100.00	100.00

in 2001, 7558663.89 million Kg in 2004, 7794138.04 million Kg in 2007, 8150709.06 million Kg in 2010, 8320301.79 million Kg in 2013, 6515690.35 million Kg in 2016 and 6510132.55 million Kg in 2018. In addition to the decrease of 165551.45 million Kg (2.14%) in 2001–2004, decrease of 1804611.45 million Kg (21.69%) in 2013–2016 and decrease of 5557.80 million Kg (0.09%) in 2016–2018, the functions from 2004–2007, 2007–2010 and 2010–2013 increased by 235474.15 million Kg (3.12%), 356571.02 million Kg (4.57%) and 169592.74 million Kg (2.08%), respectively.

The contribution rate of the various types of green space to haze absorption varied greatly. The contribution rate of forest cover was the largest, accounting for 98.80%, 98.86%, 98.89%, 98.97%, 99.03%, 98.29% and 98.22% of the total in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The second was grass land, which accounted for 0.96%, 0.86%, 0.84%, 0.79%, 0.72%, 1.58% and 1.65% of the total in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Arable land made the smallest contribution, accounting for 0.24%, 0.29%, 0.27%, 0.24%, 0.25%, 0.13% and 0.13% of the total in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

The haze absorption by green space was dominated by dust retention, the functions of which accounted for 98.09%, 98.15%, 98.18%, 98.26%, 98.32%, 98.30% and 98.29% of the total in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The function of SO_2 absorption accounted for 1.76%, 1.69%, 1.66%, 1.60%, 1.54%, 1.62% and 1.63% of the total, and the function of NO_X absorption accounted for 0.15%, 0.17%, 0.16%, 0.15%, 0.15%, 0.09% and 0.09% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

As can be seen from Figure 1, the correction based on green space quality reduced the functions by 1276243.21 million Kg (14.18%), 1226046.43 million Kg (13.96%), 1106401.75 million Kg (12.43%), 1029268.84 million Kg (11.21%), 824808.96 million Kg (9.02%), 1245918.40 million Kg (16.05%) and 1224394.20 million Kg (15.83%) in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively, compared with the benchmark values of the ecosystem service of haze absorption by green space. However, in terms of general trends, the benchmark and corrected values of ecosystem services of haze absorption by green space in China increased by 144652.20 million Kg

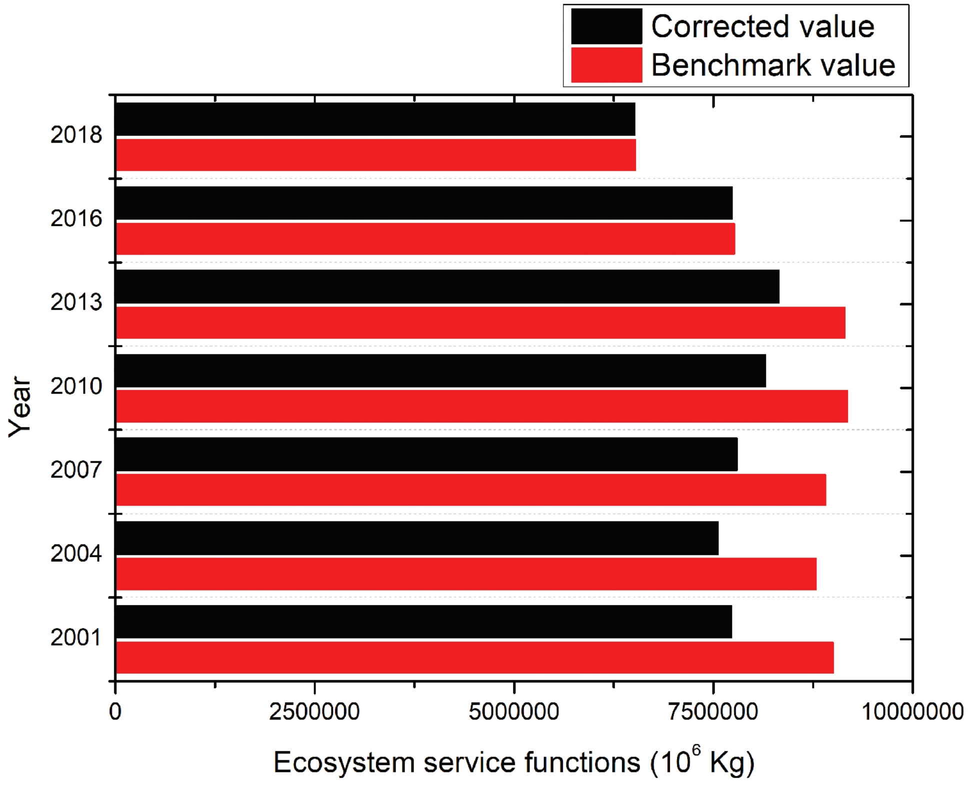

Figure 1. Comparison of benchmark and corrected ecosystem service functions of haze absorption by green space in China in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 based on the quality of green space (10^6 Kg).

(1.61%) and 596086.46 million Kg (7.72%), respectively, from 2001–2013, while decreasing by 1410584.00 million Kg (15.42%) and 1810169.25 million Kg (21.76%), respectively, from 2013–2018, indicating that the ecosystem service functions based on green space quality differ greatly from the functions considering only the area green space. If only the green space area, and not the quality, is considered when evaluating the ecosystem service functions, the evaluation results will be too high. Nonetheless, the overall trends in the benchmark and corrected ecosystem service functions of haze absorption by green space in China are consistent during 2001–2013 and 2013–2018, showing first an increase and then a decreasing trend, indicating that the ecological restoration and conservation projects of the Chinese government have enhanced the ecosystem service functions of haze absorption by green space in 2001–2013, but the adjustment of ecological land structure and the expansion of construction land have led to a reduction in the ecosystem service functions of haze absorption in 2013–2018. The government should strengthen the restoration of forest vegetation with high haze absorption capacity and regulate the speed of urban expansion to improve the ability of haze absorption by ecological land.

Compared with the benchmark values, the contribution rates of the corrected value of haze absorption by forest cover increased by 0.11%, 0.19%, 0.21%, 0.21%, 0.26%, 0.12% and 0.06% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively, while the contribution rates of the corrected value of haze absorption by grass land decreased by 0.18%, 0.27%, 0.30%, 0.29%, 0.33%, 0.09% and 0.02%, whereas those of arable land increased by 0.07%, 0.09%, 0.09%, 0.07% and 0.07% in 2001, 2004, 2007, 2010 and 2013 but decreased by 0.03% and 0.03% in 2016 and 2018, respectively.

The analysis of the ecosystem services functions of haze absorption by green space revealed that the corrected value of dust retention increased by 0.11%, 0.18%, 0.21%, 0.21%, 0.26%, 0.83% and 0.83% compared with the benchmark value in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The corrected value of SO_2 absorption decreased by 0.14%, 0.20%, 0.24%, 0.24%, 0.28%, 0.79% and 0.78% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The corrected value of NO_X absorption in 2001, 2004, 2007, 2010 and 2013 increased by 0.03%, 0.03%, 0.03%, 0.03% and 0.02%, respectively, although the value decreased by 0.04% and 0.04% in 2016 and 2018, respectively. These results indicated that the benchmark and corrected values of the contribution rates of haze absorption by different types of green space and thus the ecosystem service functions are different, but all the functions exhibited a consistent trend. The contribution rates were ranked as forest cover, grass land and arable land, and the order of ecosystem service function was dust retention, SO_2 absorption, and NO_X absorption.

Spatial distribution of the ecosystem service functions of haze absorption by green space in China

Figures 2–8 (benchmark values) show that the ecosystem service functions of haze absorption by green space had different spatial distributions in China from 2001–2018.

In general, different ecosystem service functions had very different spatial distributions within the same year, while the spatial distribution of ecosystem service functions exhibited little difference between different years.

The maximum ecosystem service functions for the absorption of SO_2 (Fig. 2a), dust retention (Fig. 2c) and the total ecosystem services (Fig. 2d) for green space were 80459.35–100608.06 million Kg, 440884.21–8817697.00 million Kg and 69663.42–8882085.53 million Kg, respectively, in 2001. These services were primarily distributed in the northwestern, central-southern and northeastern regions, which is consistent with the spatial distributions of the different ecosystem service functions presented in Figures 3–8 (a, c, d) for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. In contrast, the minimum ecosystem service functions for the absorption of NO_X (Fig. 2b) by green space were 0–964.80 million Kg in 2001, and high values for this service occurred mainly in the eastern and northeastern zones, which is inconsistent with the spatial distribution of NO_X absorption in Figures 3–8b for 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

However, compared with the maximum and minimum values, intermediate ecosystem service functions for the absorption of SO_2 (Fig. 2a) by green space were 7719.35–27868.06 million Kg and 27868.06–80459.35 million Kg in 2001, and these functions were mainly distributed in the northwestern, southwestern, central, northeastern and eastern regions, which is consistent with the spatial distribution of the absorption of

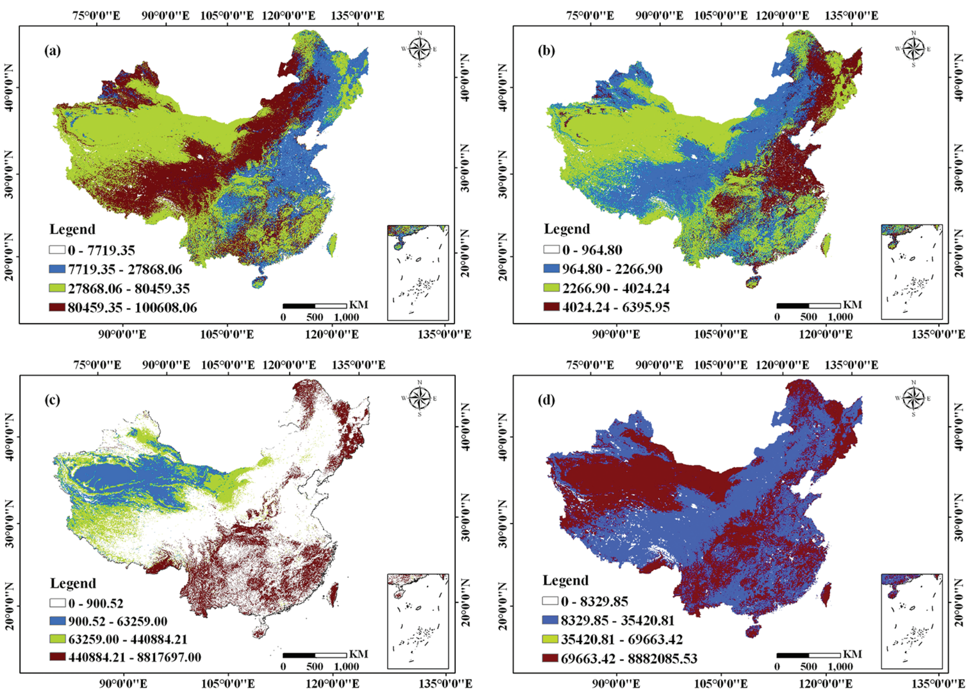

Figure 2. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2001 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

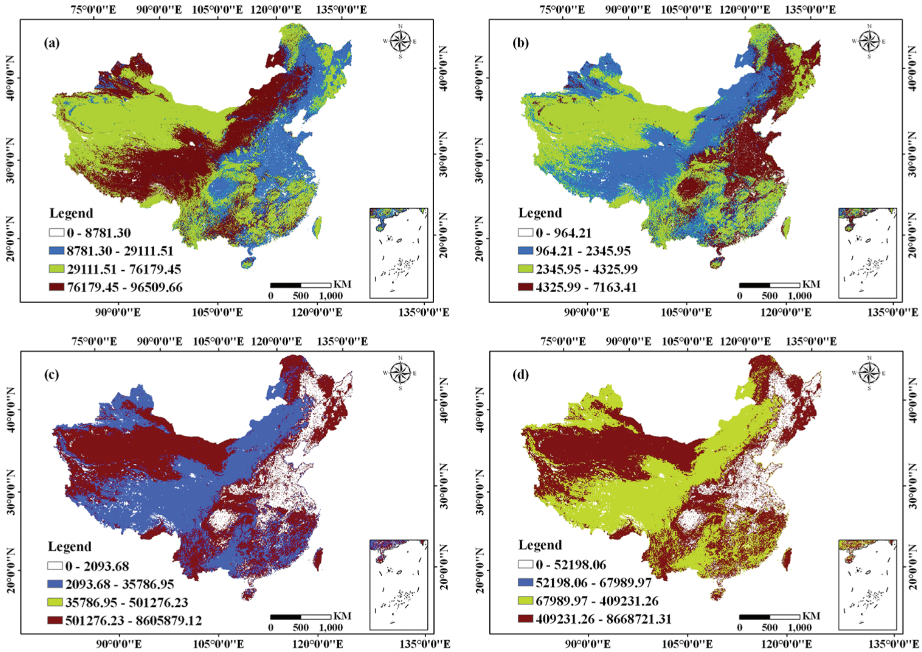

Figure 3. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2004 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

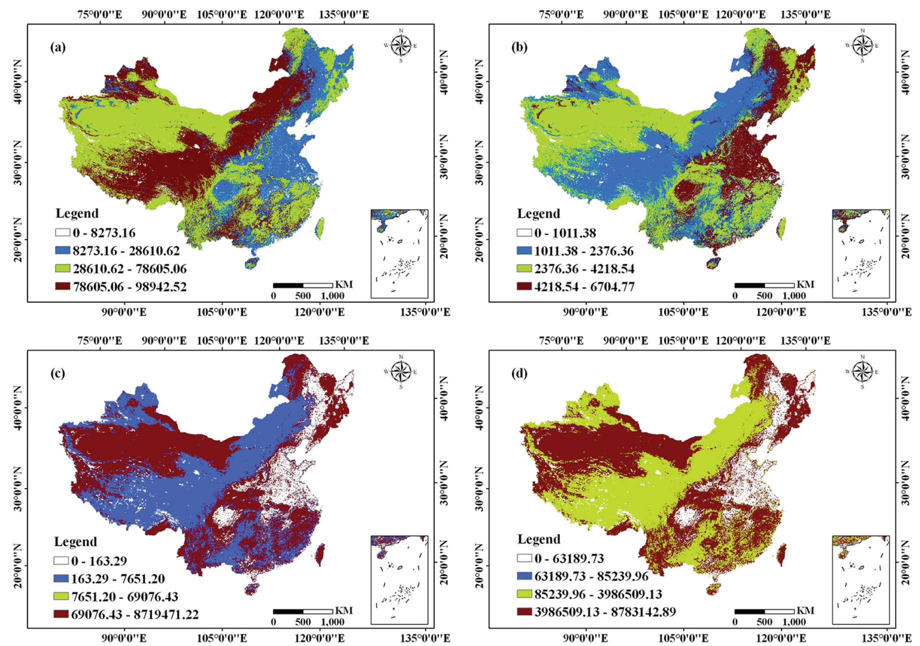

Figure 4. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2007 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

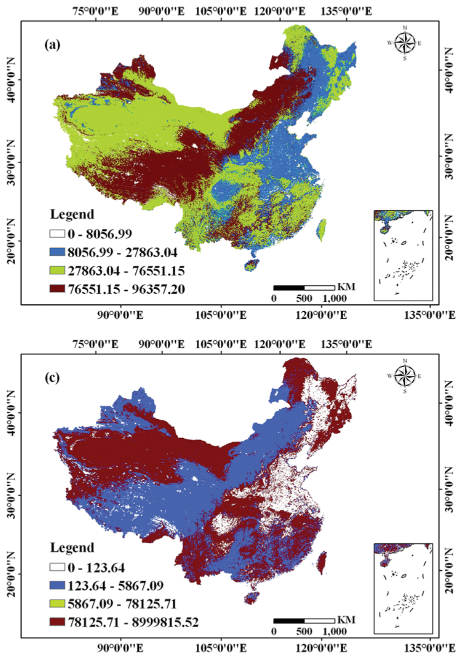

Figure 5. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2010 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

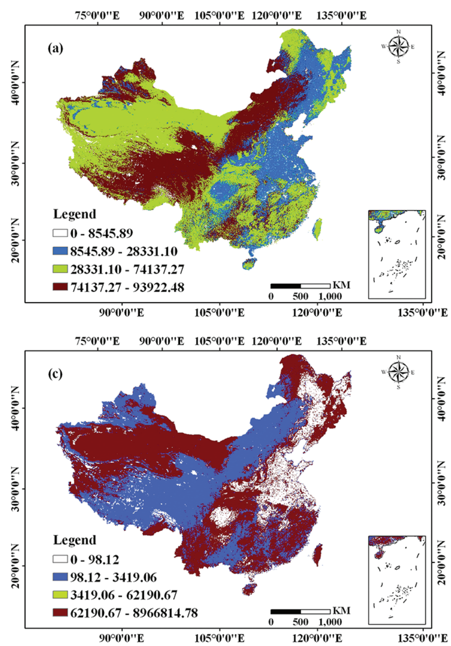

Figure 6. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2013 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

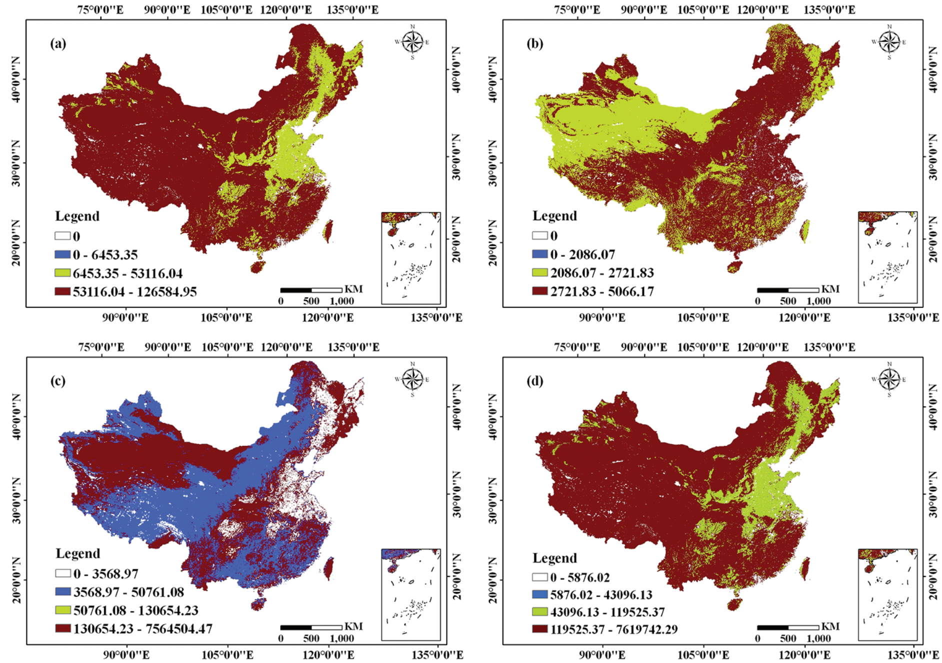

Figure 7. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2016 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

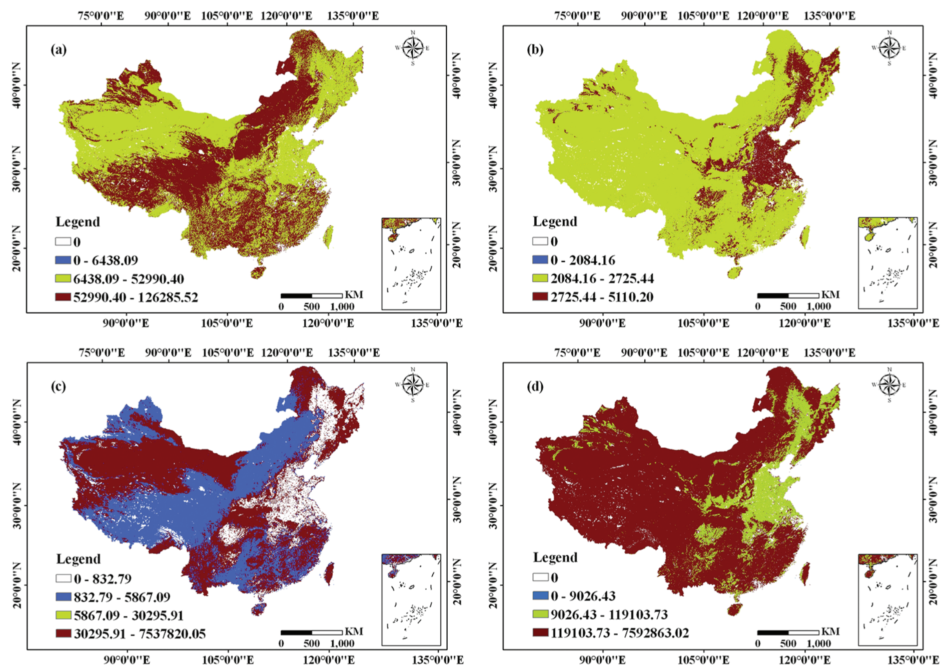

Figure 8. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2018 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

SO_2 in Figures 3–8a for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. In addition, intermediate values for NO_X absorption (Fig. 2b) were 964.80–2266.90 million Kg and 2266.90–4024.24 million Kg in 2001, and these functions were mainly in the western, central-northern, central-southern, southern and southeastern regions, which is in accordance with the spatial distribution of the absorption of NO_X in Figures 3–8b for 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

Intermediate ecosystem service functions for dust retention (Fig. 2c) by green space were 900.52–63259.00 million Kg and 63259.00–440884.21 million Kg in 2001, and these functions were mainly distributed in the northwestern, southwestern, central-northern and northeastern regions, which is consistent with the spatial distribution of dust retention in Figures 3–8 (c) for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Furthermore, intermediate values for total ecosystem services (Fig. 2d) were 8329.85–35420.81 million Kg and 35420.81–69663.42 million Kg in 2001, and these values were mainly in the northwestern, southwestern, southern, eastern and northeastern regions, which is in accordance with the spatial distribution of total ecosystem services in Figures 5–7d for 2010, 2013 and 2016, respectively, but is inconsistent with Figures 3–4 and 8d for 2004, 2007 and 2018, during which these functions were mainly distributed in the northwestern, southwestern, and central regions.

As shown in Figures 9–15 (corrected values), the maximum ecosystem service functions of SO_2 absorption (Fig. 9a), dust retention (Fig. 9c) and total ecosystem services (Fig. 9d) by green space were 53386.20–72412.13 million Kg, 5681893.88–7575858.50 million Kg and 5723384.25–7631179.00 million Kg in 2001, respectively, and were mainly in the southeastern, central-southern and southwestern areas, which is in accordance with the spatial distributions of the different ecosystem service functions presented in Figures 10–15 (a, c, d) for 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

In contrast, the intermediate ecosystem service functions of SO_2 absorption (Fig. 9a) by green space were 0–10506.86 million Kg and 10506.86–53386.20 million Kg in 2001 and were mainly distributed in the southwestern, central-northern and northeastern regions, which is consistent with the spatial distribution of SO_2 absorption in Figures 10–15a for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Moreover, the intermediate ecosystem service functions of dust retention (Fig. 9c) by green space were 1893964.63–3787929.25 million Kg and 5681893.88–7575858.50 million Kg in 2001 and occurred mainly in the northeastern, central and southeastern areas, which is in accordance with the spatial distribution of dust retention in Figures 9–15c for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Furthermore, the minimum values of the total ecosystem service functions (Fig. 9d) by green space were 0–1907794.75 million Kg in 2001 and occurred mainly in the western, central-northern and southwestern zones, respectively, which is in accordance with the spatial distribution of the total ecosystem service functions (Figures 10–15d) in 2004, 2007, 2010, 2013, 2016 and 2018.

The results show that there was a great difference in the spatial distributions of the benchmark and corrected values of haze absorption by green space, and the spatial distributions of the maximum, intermediate and minimum ecosystem

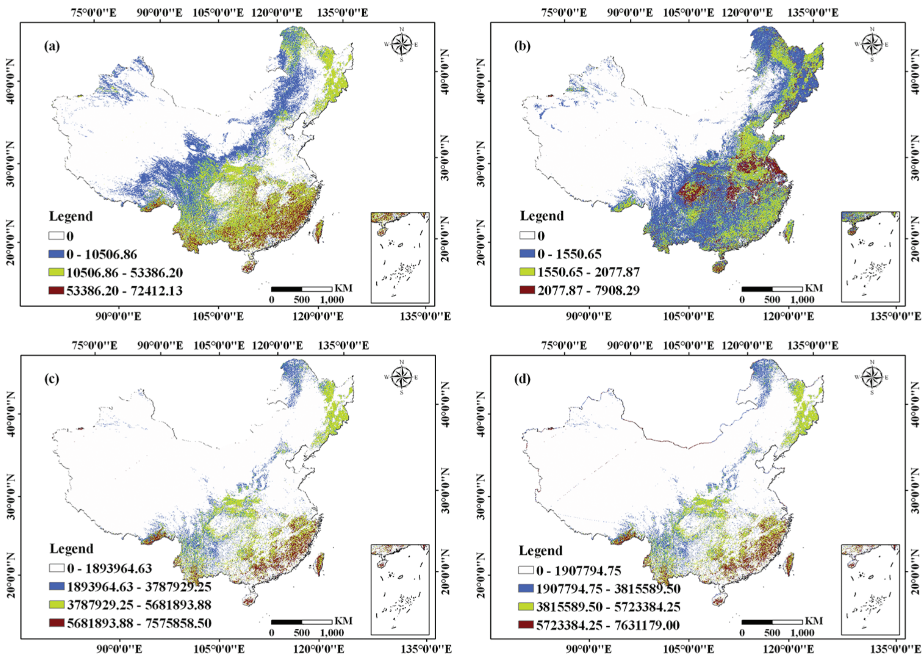

Figure 9. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2001 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

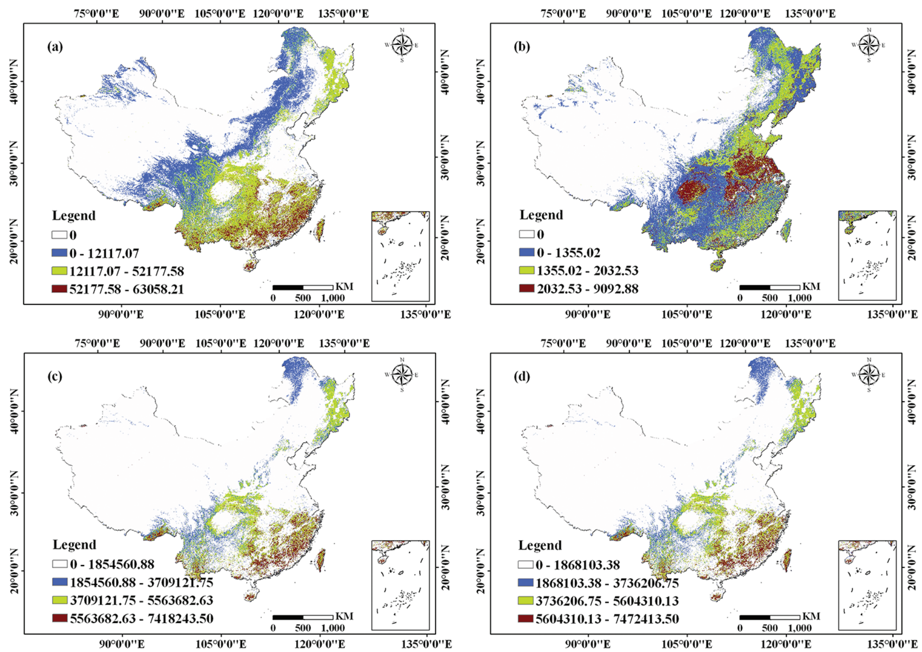

Figure 10. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2004 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

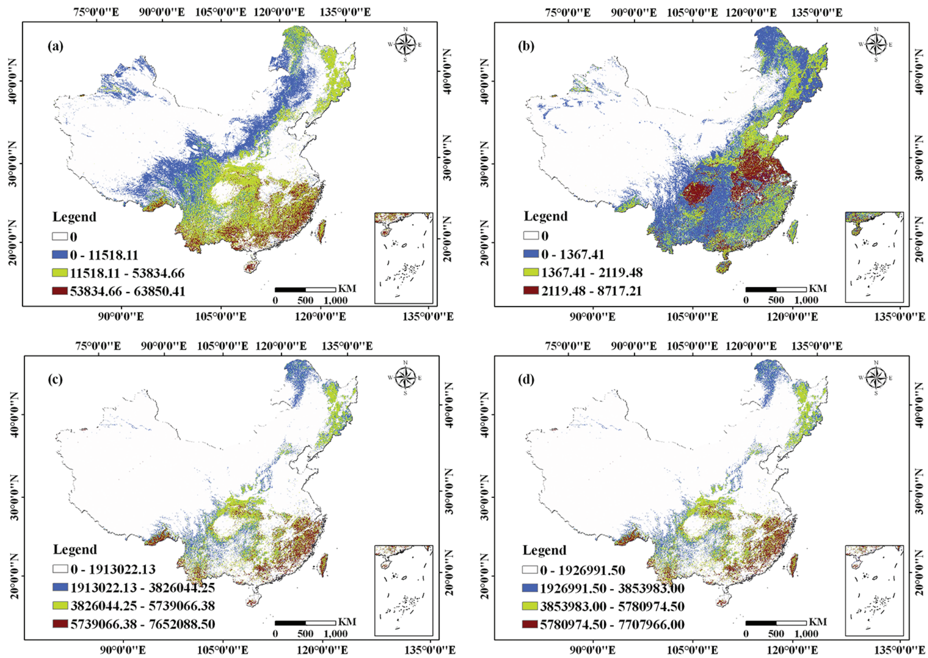

Figure 11. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2007 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

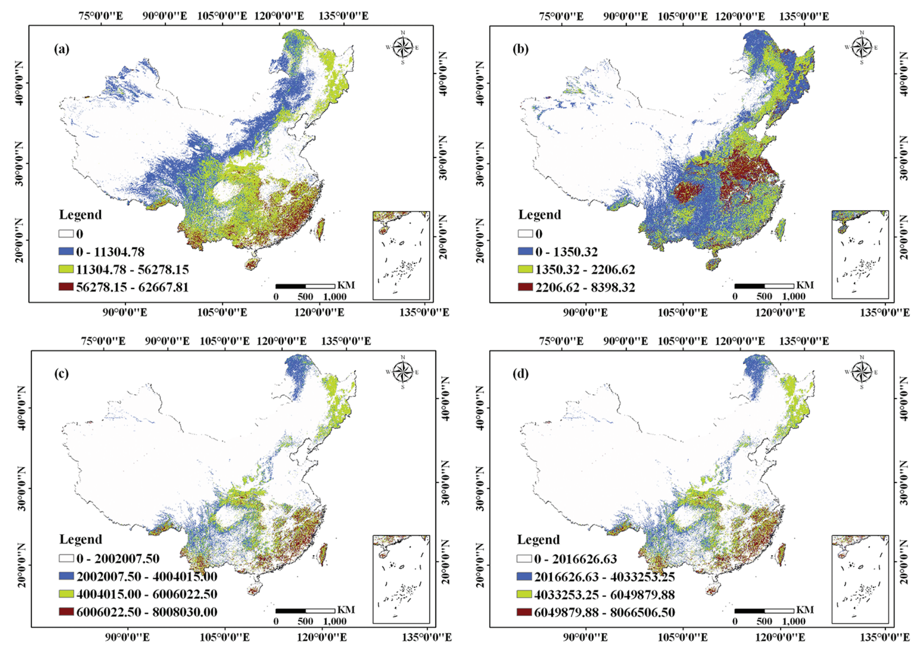

Figure 12. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2010 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

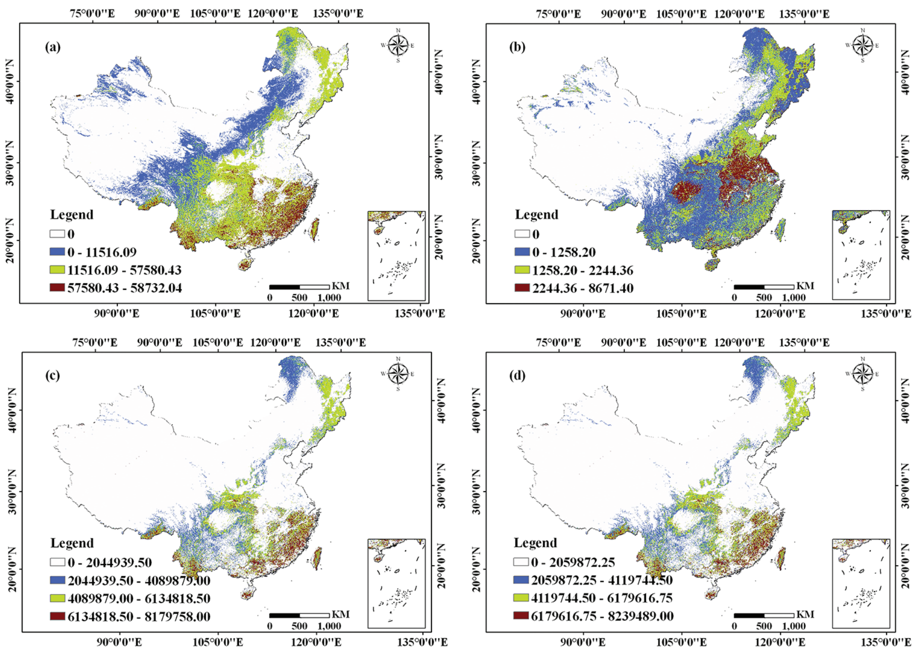

Figure 13. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2013 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

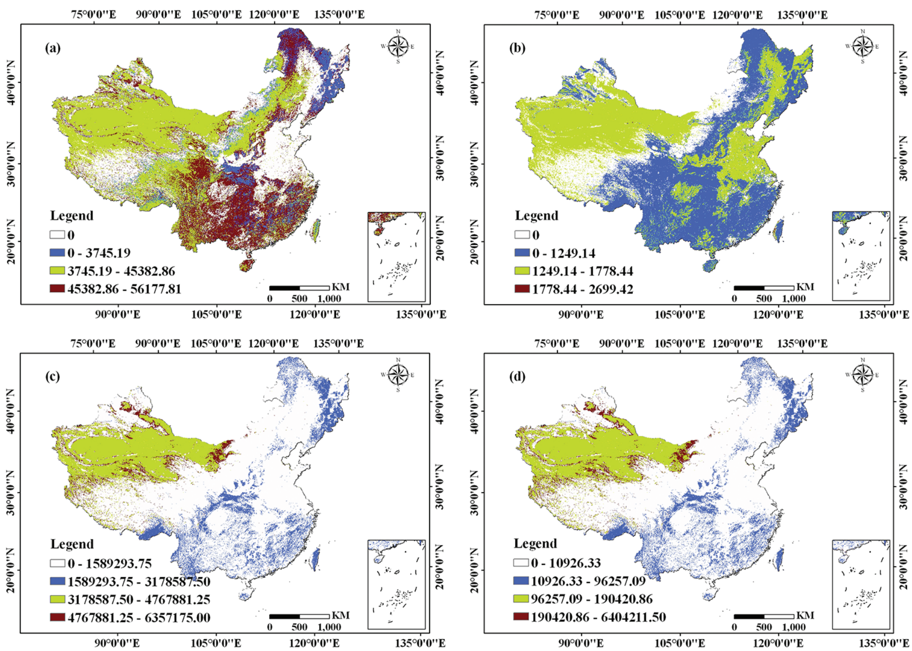

Figure 14. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2016 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

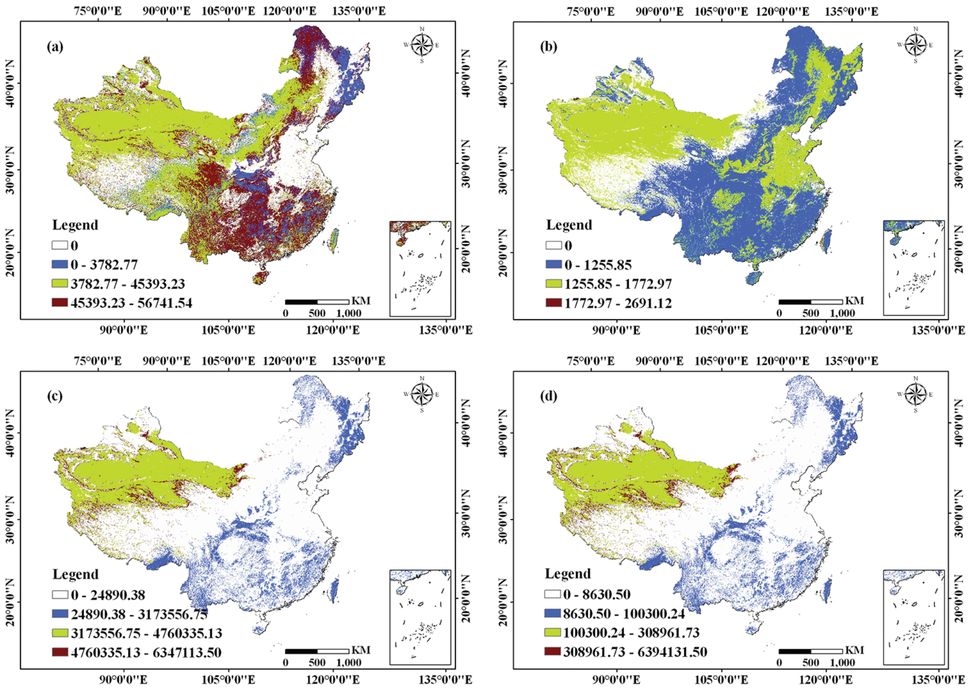

Figure 15. Spatial distribution of the ecosystem service functions of haze absorption by green space in China in 2018 (10^6 Kg) (**a, b, c** and **d** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, and the total ecosystem service functions, respectively).

service function values were obviously different. However, the spatial distributions of the benchmark and corrected values also exhibited the same trend. In the same year, the spatial distribution of the ecosystem service functions of haze absorption by green space was very different, but in different years, the difference in the spatial distribution of the ecosystem service functions of haze absorption by green space exhibited little difference.

Comparison of the ecosystem service functions of haze absorption by green space in different zones

Figures 16–22 (benchmark values) show that the spatial distribution of ecosystem service functions and the proportion of haze absorption by green space differed in different provinces in China. Overall, different ecosystem service functions exhibited different spatial distributions in the same year or between different years. Some spatial distributions were quite different; others were more similar.

The ecosystem service functions of the absorption of SO_2 (Fig. 16a) and NO_X (Fig. 16b) by green space were 2.74–30586.00 million Kg and 0.16–1207.44 million Kg in 2001, respectively. The maximum and minimum values were primarily distributed in Xinjiang and in Shanghai, accounting for 17.88%, 0.02%, 10.98% and

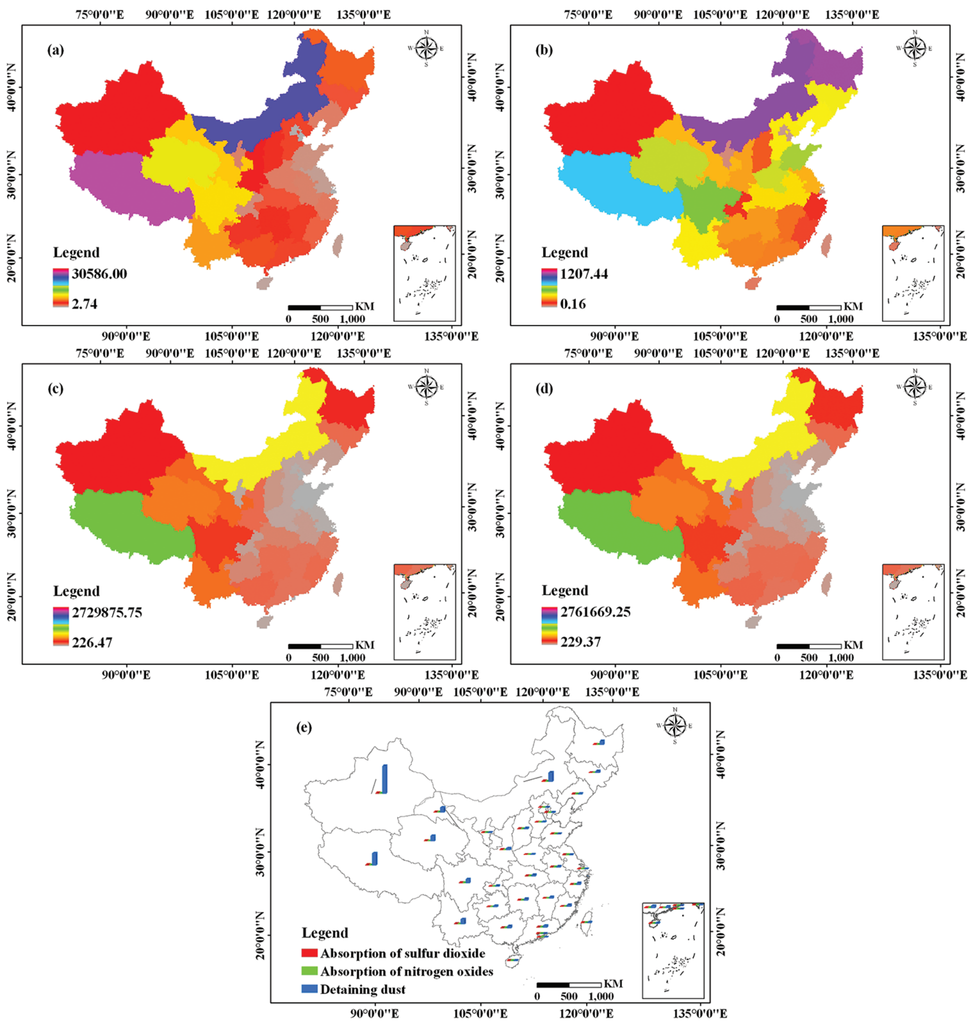

Figure 16. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2001 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

0.14% of the total regional functions, respectively, The spatial distribution of these functions in 2001 was consistent with the absorption of SO_2 and NO_X in Fig. 17a and 17b but different from those shown in Figures 18–22a and 18–22b, which represent values for 2007, 2010, 2013, 2016 and 2018, respectively. Additionally, the functions of dust retention and total ecosystem services (Fig. 16c, d) for green space were 226.47–2729875.75 million Kg and 229.37–2761669.25 million Kg in 2001, respectively, and the maximum and minimum values for these services were primarily

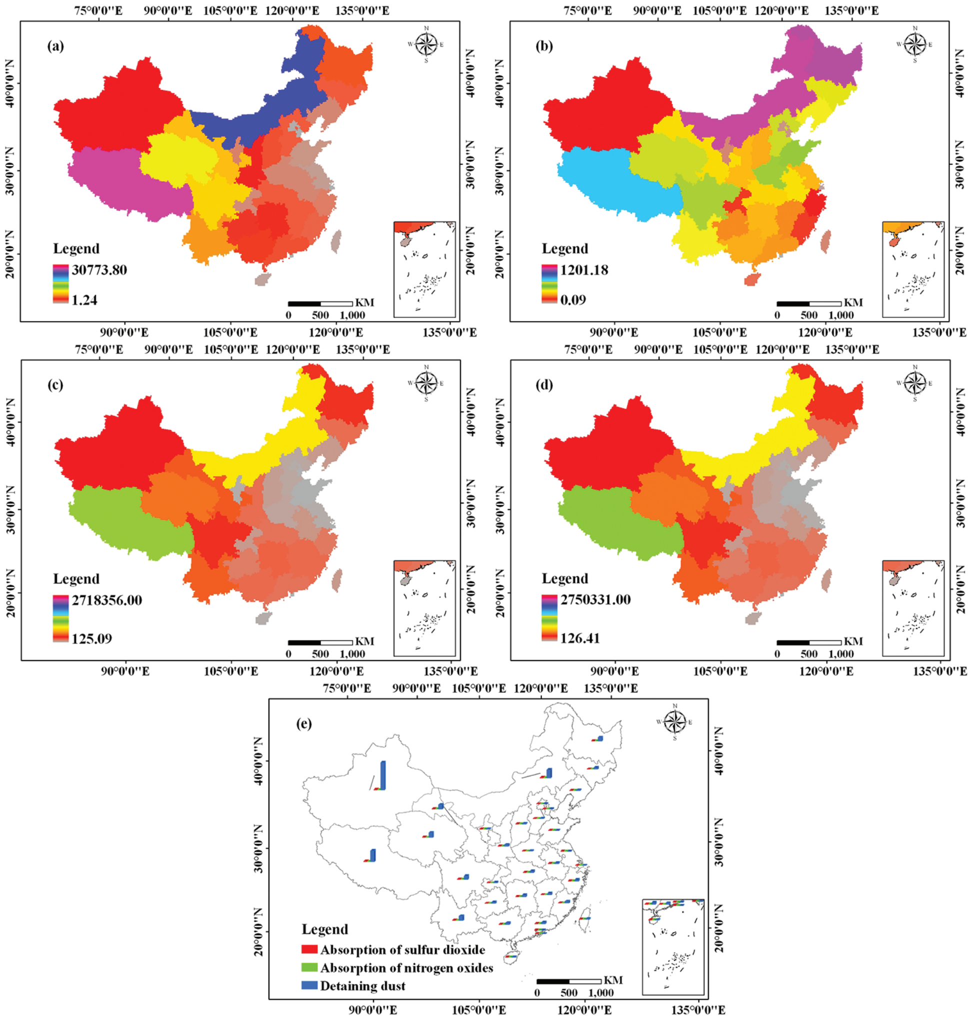

Figure 17. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2004 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

in Xinjiang and Shanghai, accounting for 30.97%, 0.01%, 30.70% and 0.01% of the regional totals, which is consistent with the spatial distribution of dust retention and total ecosystem services in Figures 17–22c and Figures 17–22d in 2004, 2007, 2010, 2013, 2016 and 2018, respectively. Most of the ecosystem service functions of haze absorption by green space were primarily from dust retention, which accounted for approximately 96% of the total. The functions for SO_2 absorption were the next highest, accounting for approximately 3% of the total, while NO_X accounted for approximately

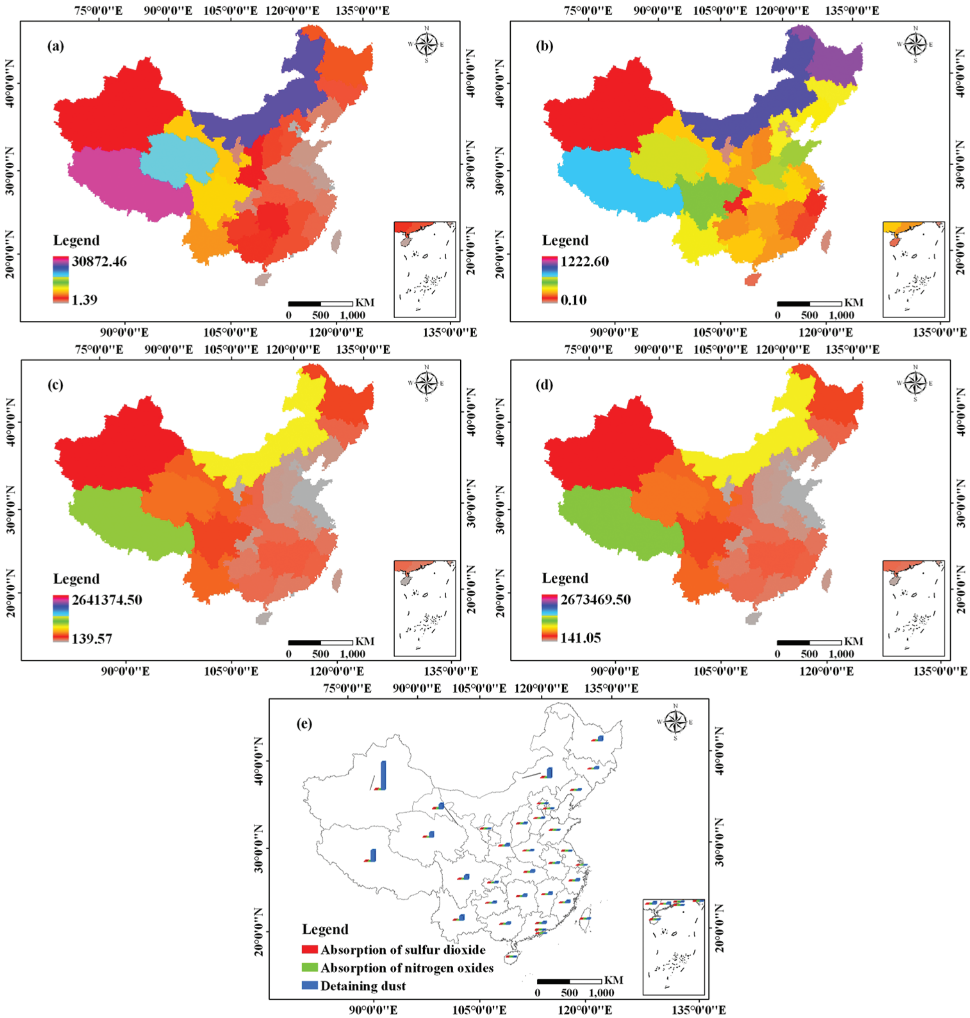

Figure 18. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2007 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

1% (Fig. 16e) in 2001, which is consistent with the percentages of the ecosystem service functions of haze absorption by green space in Figures 17–22e in 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

The ecosystem service functions of the absorption of SO_2 (Fig. 18a) by green space were 1.39–30872.46 million Kg in 2007, and the maximum and minimum values were mainly distributed in Xinjiang and Shanghai, accounting for 18.25% and 0.02% of the regional totals, respectively. This distribution is in accordance with the spatial

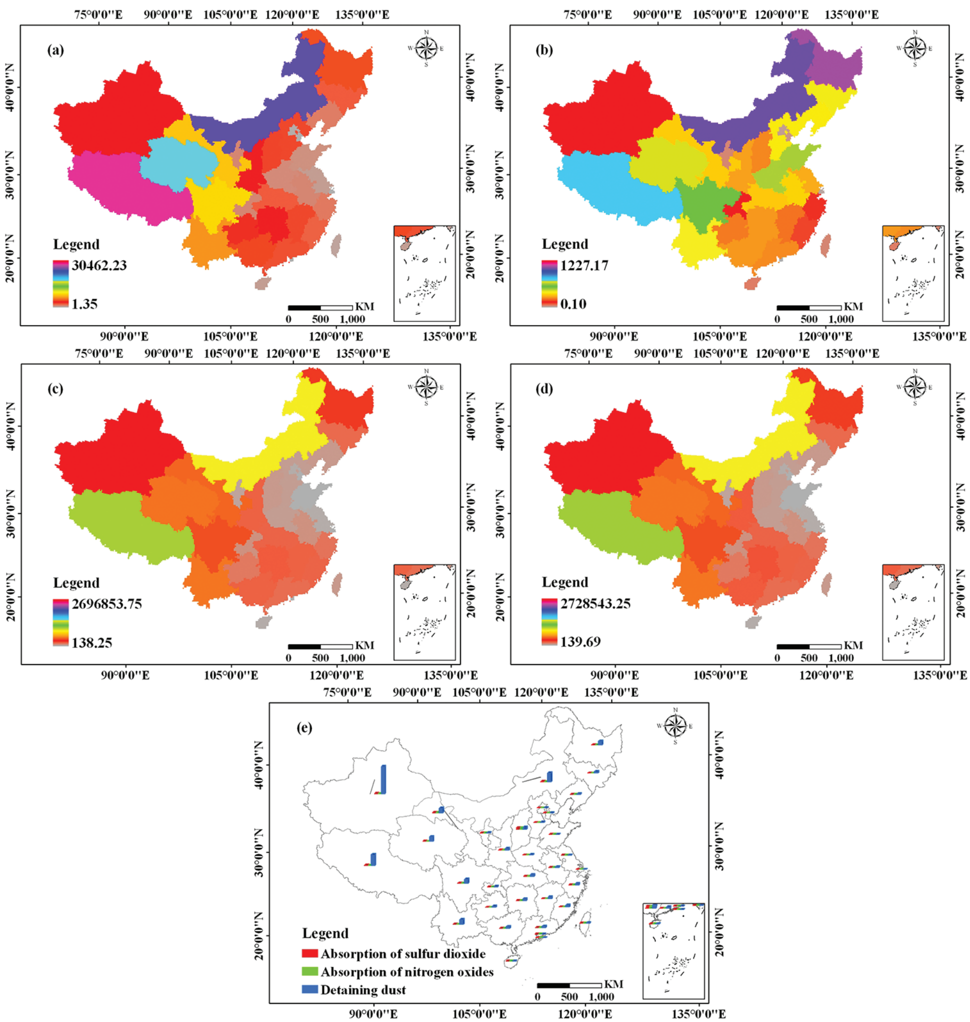

Figure 19. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2010 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

distribution of the absorption of SO_2 in Figures 19–20a in 2010 and 2013, respectively. The ecosystem service functions of the absorption of NO_X (Fig. 18b) by green space was 0.10–1222.60 million Kg in 2007, and the maximum and minimum values were mainly in Xinjiang and Shanghai, accounting for 10.87% and 0.13% of the regional totals, respectively. This distribution is consistent with the spatial distribution of the absorption of NO_X in Figures 16b, 19b and Figures 21–22b in 2001, 2010, 2016 and 2018, respectively.

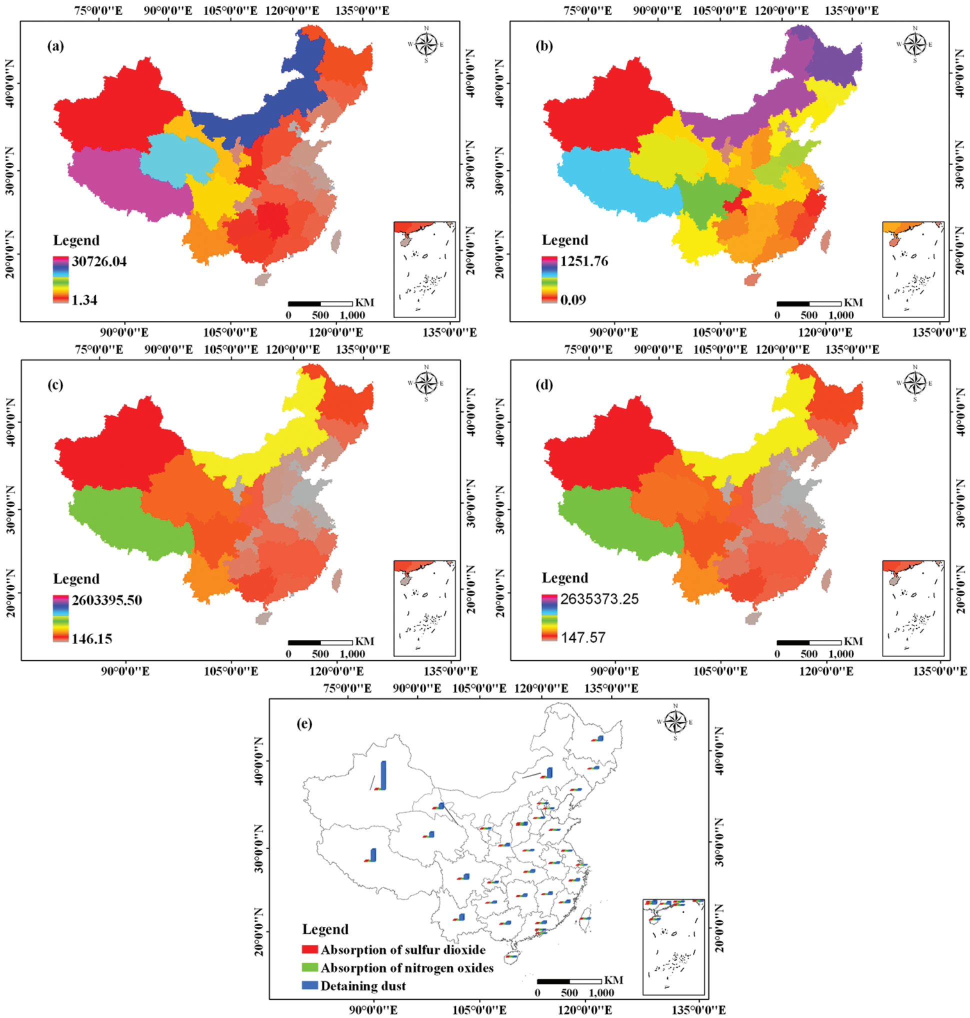

Figure 20. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2013 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

As shown in Figures 23–29 (corrected values), the ecosystem service functions of the absorption of SO_2 (Fig. 23a) by green space were 0.73–11,739.97 million Kg in 2001. The maximum and minimum values were primarily distributed in Yunnan and Shanghai, accounting for 8.62% and 0.03% of the total regional values, respectively. The spatial distribution of this value in 2001 was inconsistent with the absorption of SO_2 in Figures 24–29a, which represent values for 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The ecosystem service functions of the absorption of NO_X

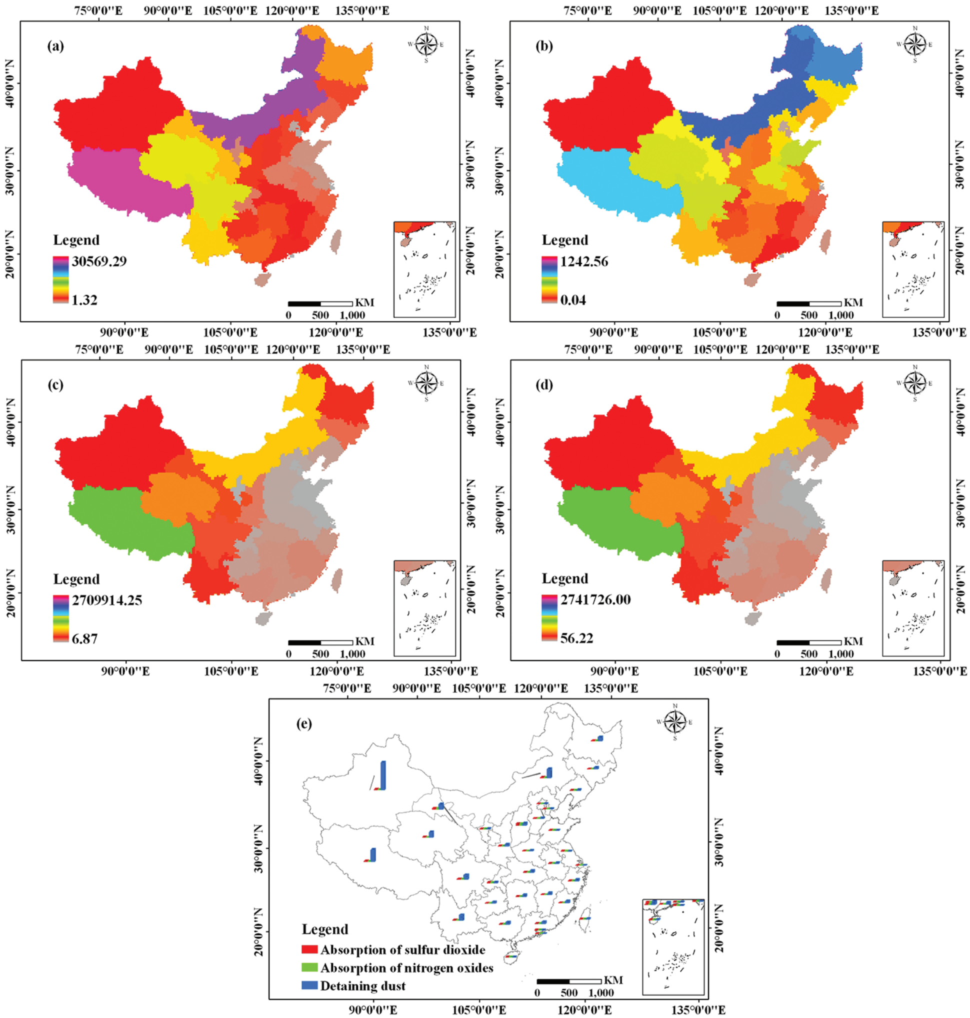

Figure 21. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2016 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

(Fig. 23b) by green space were 0.04–1206.36 million Kg in 2001, and the maximum and minimum values were mainly distributed in Heilongjiang and Ningxia, accounting for 10.43% and 0.11% of the regional totals, respectively, which is consistent with the spatial distribution of absorption of NO_X in Figures 24–27b in 2004, 2007, 2010 and 2013, respectively, but inconsistent with Figures 28–29b in 2016 and 2018, respectively. The ecosystem service functions of the absorption of dust retention (Fig. 23c) and total ecosystem services (Fig. 23d) by green space were 58.62–930,837.56 million

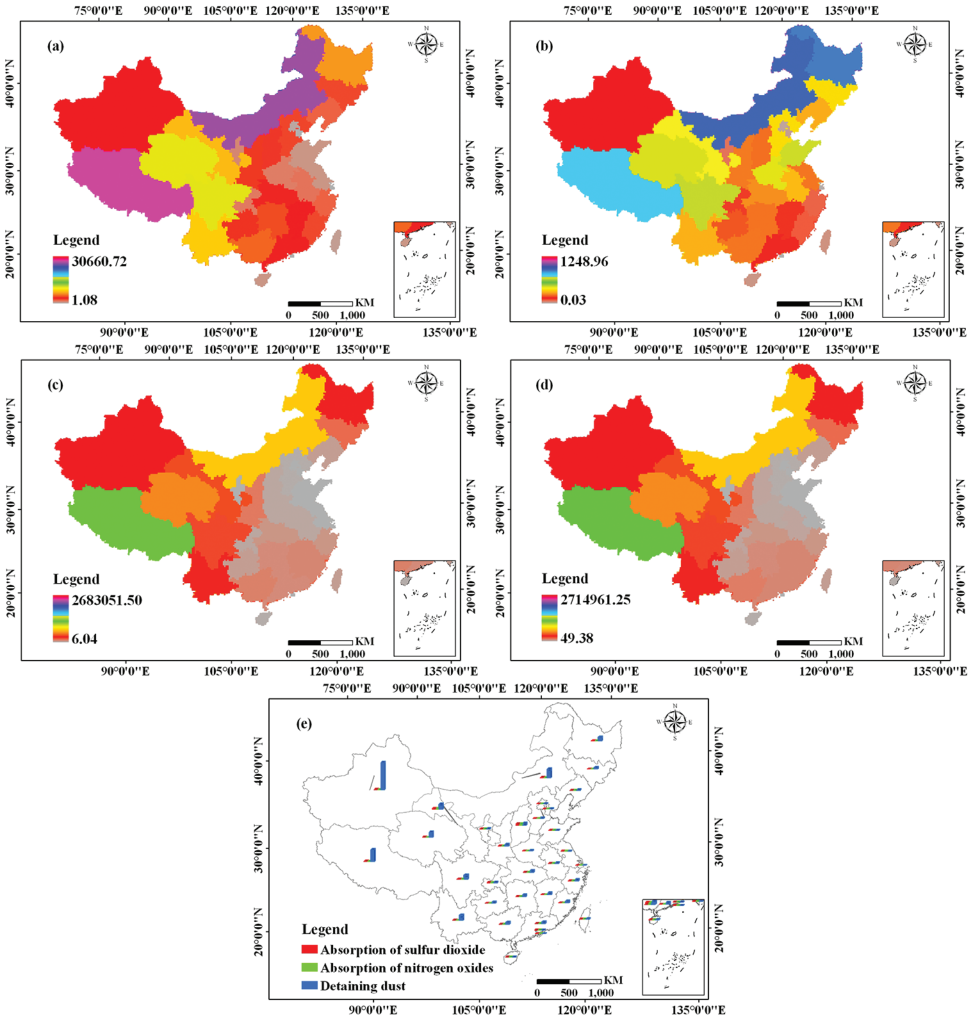

Figure 22. Spatial distribution of the ecosystem service functions of haze absorption by green space in different regions of China in 2018 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

Kg and 59.39–943211.69 million Kg in 2001. The maximum and minimum values were primarily distributed in Yunnan and Shanghai, accounting for 12.29%, 0.01%, 12.21% and 0.01% of the total regional functions, respectively. The spatial distribution of this function in 2001 was consistent with the absorption of dust retention and total ecosystem services in Figures 24–27c and 24–27d, which represent functions for 2004, 2007, 2010 and 2013, respectively, but inconsistent with the Figures 28–29c and 28–29d in 2016 and 2018, respectively.

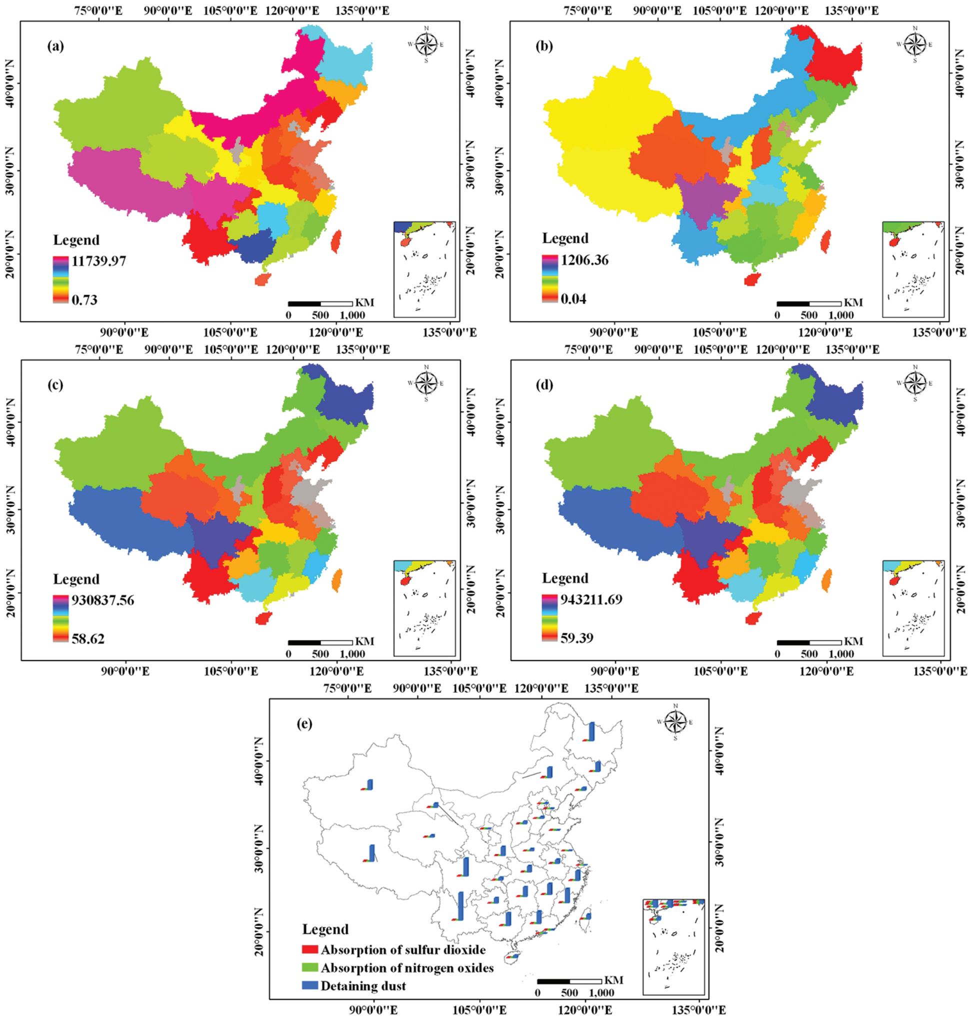

Figure 23. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2001 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

The results show that there was a great difference in the spatial distributions of the benchmark and corrected values of haze absorption by green space in different provinces in China, and the maximum and minimum of ecosystem service functions were obviously different. However, the spatial distributions of the benchmark and corrected values also exhibited the same trend. In the same year, the spatial distribution of the ecosystem service functions of haze absorption by green space was very different in different province, but in different years, the difference in the spatial distribution of the

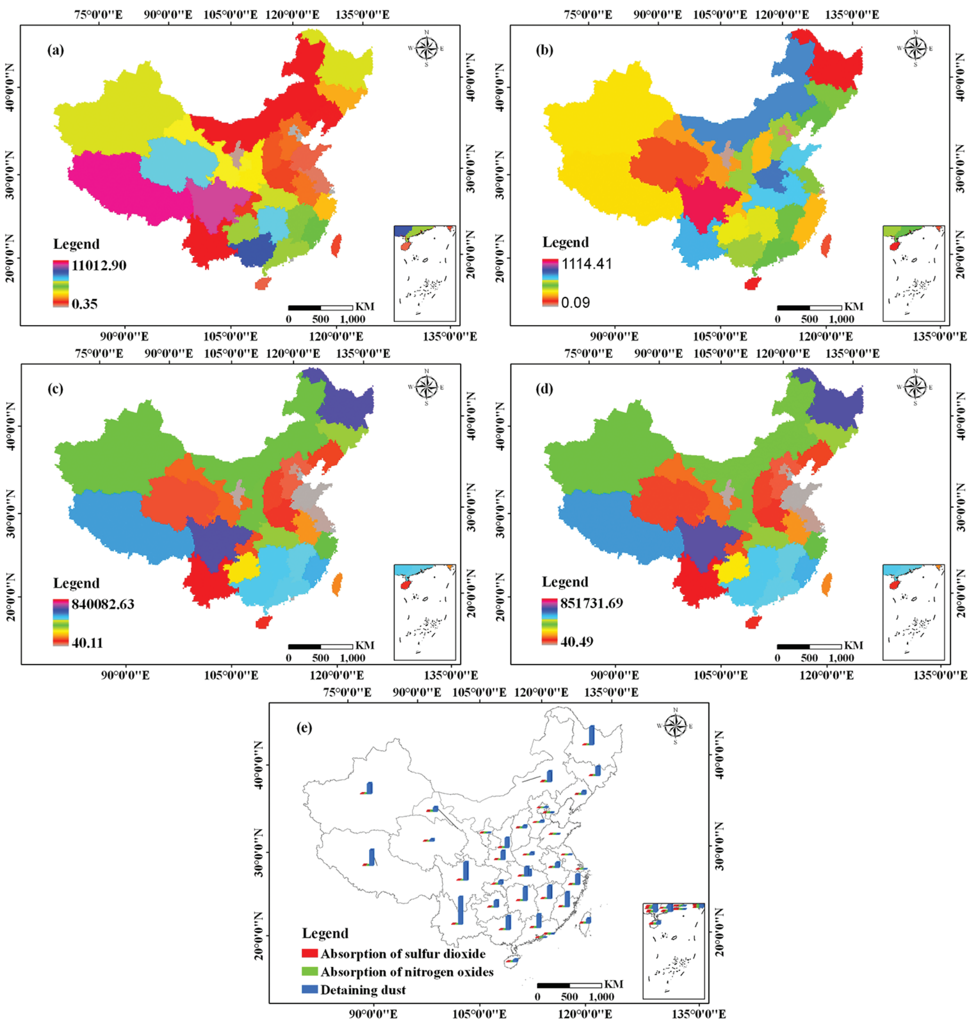

Figure 24. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2004 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

ecosystem service functions of haze absorption by green space exhibited little difference in different provinces.

Sensitivity analysis of the ecosystem service function coefficients for haze absorption by green space

The coefficient of sensitivity of the ecosystem service functions for different green space types was generally quite different from 2001–2018 (Table 4). The sensitivity coef-

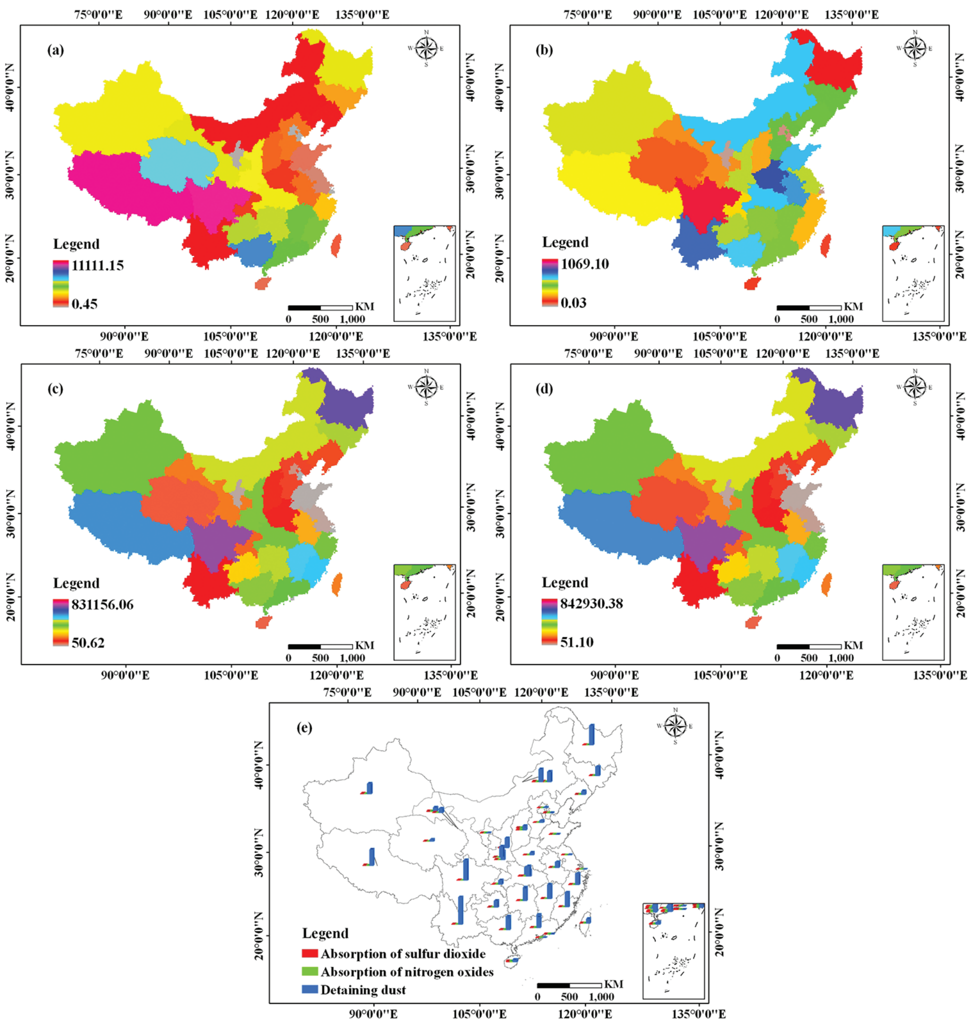

Figure 25. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2007 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

ficients for forest cover were elastic, while those of arable land and grass land were inelastic. The coefficients of sensitivity for forest cover were highest due to the large area of this cover type and the high ecosystem service functions coefficient for haze absorption by green space. The coefficients of sensitivity were 0.9868 in 2001, 2004 and 2007, 0.9875 in 2010, 0.9877 in 2013, 0.9817 in 2016 and 2018, respectively, and the change rates were ± 49.3424%, ± 49.3398%, ± 49.3405%, ± 49.3767%, ± 49.3832%, ± 49.0861% and ± 49.0842%, respectively. The coefficients of sensitivity for grass land were relatively small, with values of 0.0115 in 2001, 0.0113 in 2004, 0.0114 in 2007, 0.0108 in 2010, 0.0105 in 2013, 0.0167 in 2016 and 2018, and the

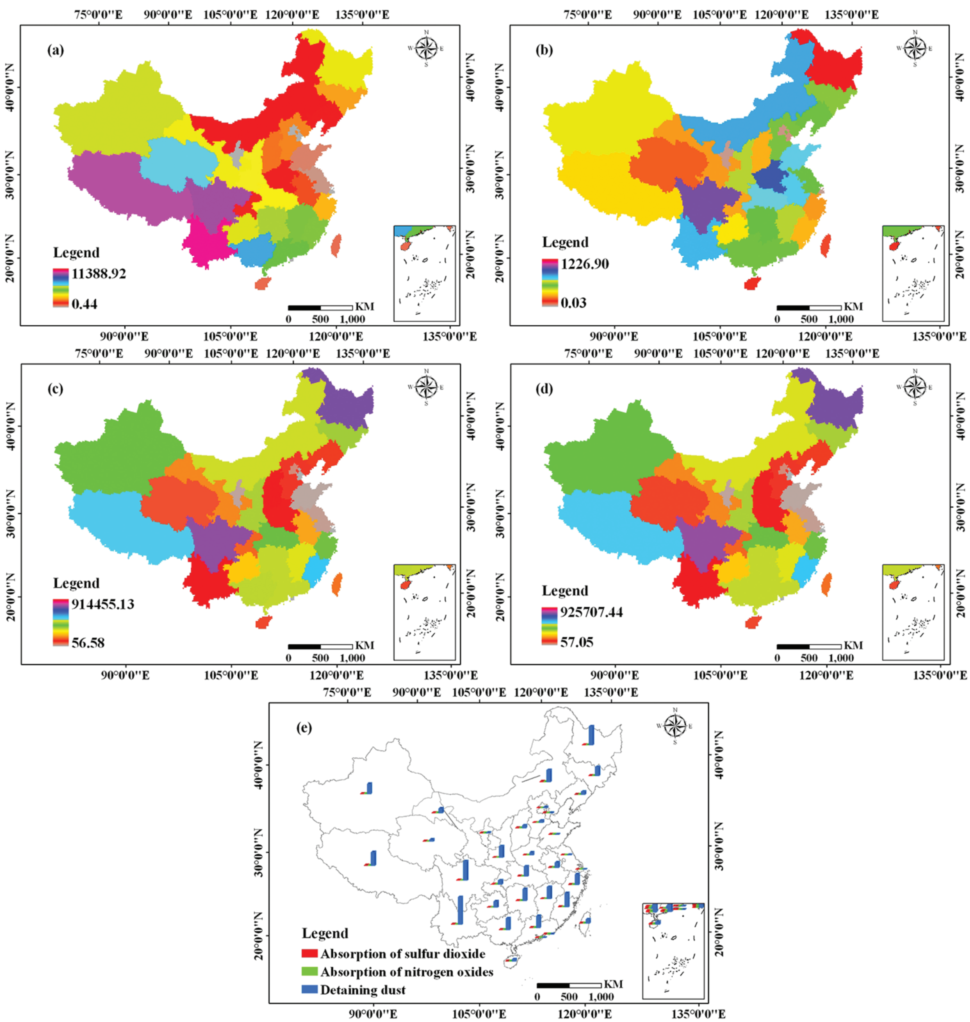

Figure 26. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2010 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

change rates for grass land were ± 0.5733%, ± 0.5635%, ± 0.5702%, ± 0.5384%, ± 0.5268, ± 0.8365 and ± 0.8374%, respectively. The coefficients of sensitivity for arable land were the smallest due to the low ecosystem service functions coefficient of haze absorption by green space. The values of this coefficient were 0.0017 in 2001, 0.0019 in 2004, 0.0018 in 2007, 0.0017 in 2010, 0.0018 in 2013, 0.0015 in 2016, and 0.0016 in 2018, and the change rates were ± 0.0843%, ± 0.0967%, ± 0.0893%, ± 0.0850%, ± 0.0901%, ± 0.0774% and ± 0.0783%, respectively.

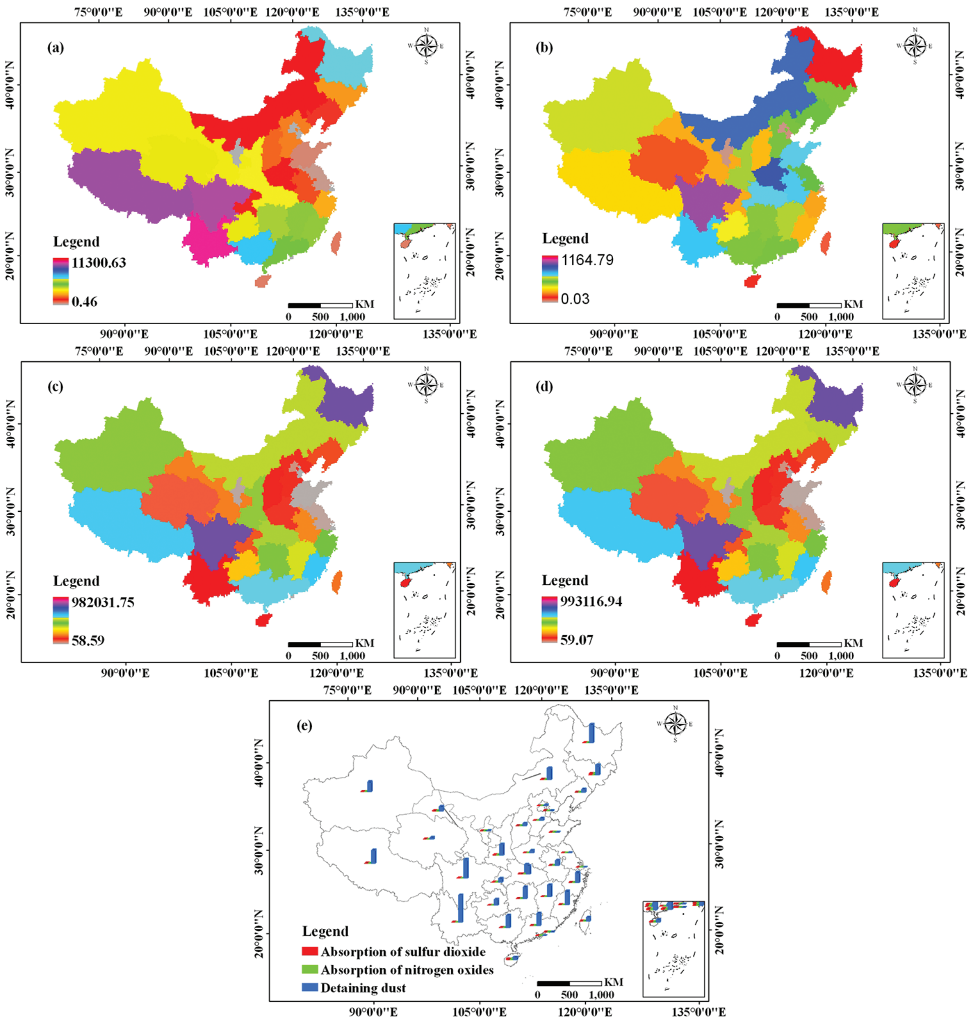

Figure 27. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2013 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

Relationship between landscape patterns and the ecosystem service functions of haze absorption by green space

To quantitatively understand the relationship between land use patterns and ecosystem service functions, a correlation analysis was conducted (Table 5). There were significant correlations between many landscape pattern metrics and ecosystem service functions, which indicated that landscape patterns significantly affected ecosystem service func-

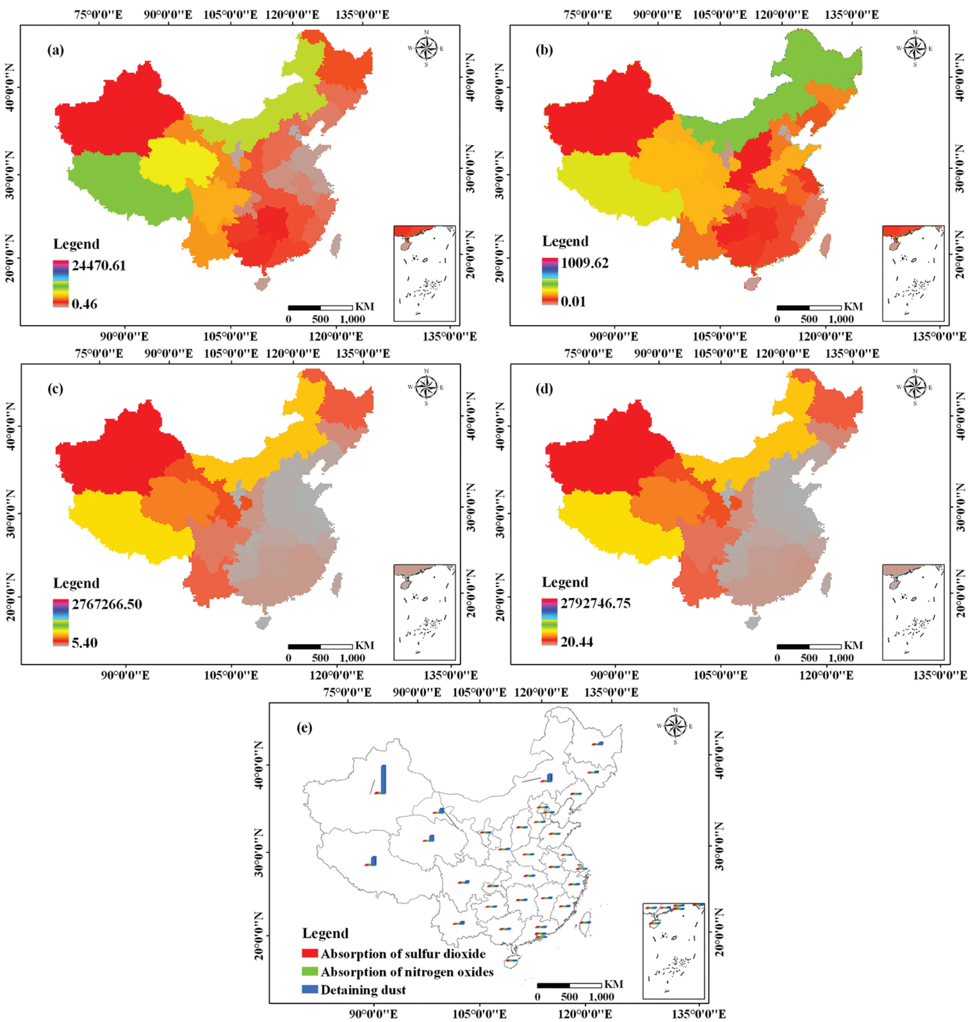

Figure 28. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2016 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

tions. The correlation coefficients between PD and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services exhibited significant negative relationships with correlation coefficients of -0.407, -0.511, -0.330 and -0.332, respectively. In contrast, the correlation coefficients between SHAPE_AM and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services exhibited significant positive relationships with correlation coefficients of 0.650, 0.634, 0.568 and 0.570, respectively. These results indicate

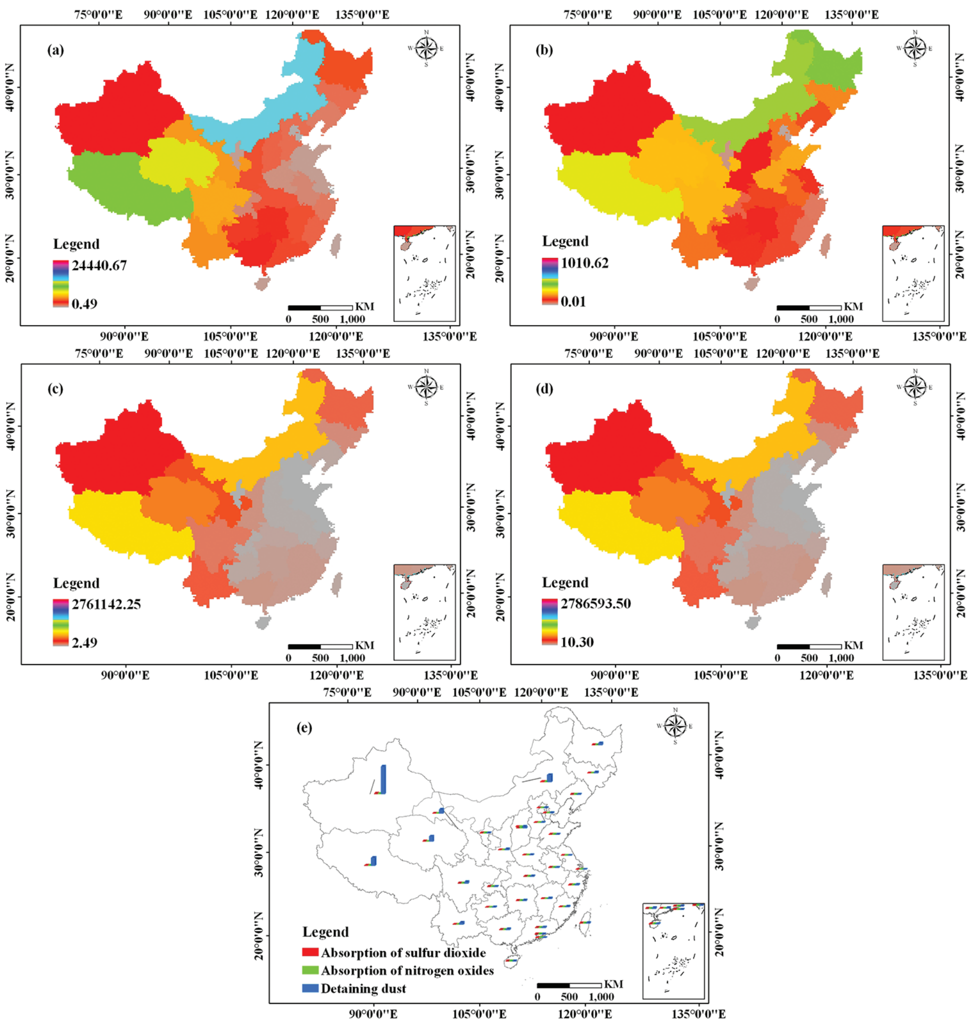

Figure 29. Spatial distributions of the ecosystem service functions of haze absorption by green space in different regions of China in 2018 (10^6 Kg) (**a, b, c, d** and **e** are the absorption of sulfur dioxide and nitrogen oxides, dust retention, the total ecosystem service functions and the percent contribution of different ecosystem service functions to haze absorption in different zones, respectively).

that PD and SHAPE_AM have important effects on different ecosystem service functions. In general, the larger the PD, the smaller the ecosystem service functions; the larger the value of SHAPE_AM, the greater the ecosystem service functions.

The correlation coefficients between IJI and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services exhibited significant negative relationships with correlation coefficients of -0.606, -0.507, -0.449 and -0.452, respectively. The correlation coefficients between SHDI and the ecosystem

Table 4. Sensitivity analysis of the ecosystem service functions of haze absorption by green space in China from 2001–2018.

	Coefficient of sensitivity	Green space types					
		Arable land		Forest cover		Grass land	
		FC+50%	FC-50%	FC+50%	FC-50%	FC+50%	FC-50%
2001	%	0.0843	-0.0843	49.3424	-49.3424	0.5733	-0.5733
	CS	0.0017	–	0.9868	–	0.0115	–
2004	%	0.0967	-0.0967	49.3398	-49.3398	0.5635	-0.5635
	CS	0.0019	–	0.9868	–	0.0113	–
2007	%	0.0893	-0.0893	49.3405	-49.3405	0.5702	-0.5702
	CS	0.0018	–	0.9868	–	0.0114	–
2010	%	0.0850	-0.0850	49.3767	-49.3767	0.5384	-0.5384
	CS	0.0017	–	0.9875	–	0.0108	–
2013	%	0.0901	-0.0901	49.3832	-49.3832	0.5268	-0.5268
	CS	0.0018	–	0.9877	–	0.0105	–
2016	%	0.0774	-0.0774	49.0861	-49.0861	0.8365	-0.8365
	CS	0.0015	–	0.9817	–	0.0167	–
2018	%	0.0783	-0.0783	49.0842	-49.0842	0.8374	-0.8374
	CS	0.0016	–	0.9817	–	0.0167	–

Note: The coefficients of the ecosystem service functions of different land use types were adjusted up and down by 50% to analyze the coefficients of sensitivity for the three land use types and evaluate the changes in the ecosystem service functions caused by changes in the coefficients [1]. CS refers to the coefficients of sensitivity, and FC refers to functional coefficients.

Table 5. Correlation coefficients between landscape pattern metrics and different ecosystem service functions of haze absorption by green space in China.

	SO_2	NO_x	DUST	ALL	PD	SHAPE_AM	IJI	SHDI
SO_2	1.000	0.772**	0.887**	0.891**	-0.407**	0.650**	-0.606**	-0.242**
NO_X	0.772**	1.000	0.750**	0.752**	-0.511**	0.634**	-0.507**	-0.316**
DUST	0.887**	0.750**	1.000	0.999**	-0.330**	0.568**	-0.449**	-0.202**
ALL	0.891**	0.752**	1.000**	1.000	-0.332**	0.570**	-0.452**	-0.203**
PD	-0.407**	-0.511**	-0.330**	-0.332**	1.000	-0.342**	0.564**	0.642**
SHAPE_AM	0.650**	0.634**	0.568**	0.570**	-0.342**	1.000	-0.783**	-0.149
IJI	-0.606**	-0.507**	-0.449**	-0.452**	0.564**	-0.783**	1.000	0.227**
SHDI	-0.242**	-0.316**	-0.202**	-0.203**	0.642**	-0.149	0.227**	1.000

** Significance at the 0.01 probability level. * Significance at the 0.05 probability level
Note: SO_2, NO_X, DUST, and ALL refer to ecosystem service functions of SO_2, the absorption of NO_X, dust retention, and total ecosystem service functions, respectively. PD, SHAPE_AM, IJI and SHDI refer to patch density, the area-weighted mean shape index, the interspersion and juxtaposition index, and Shannon's diversity index.

service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services also exhibited significant negative relationships with correlation coefficients of -0.242, -0.316, -0.202 and -0.203, respectively. These results indicate that IJI and SHDI have important effects on different ecosystem service values. In general, the smaller the IJI and SHDI, the larger the ecosystem service functions.

Discussion

In this paper, the quality of green space is used to modify the ecosystem service functions of haze absorption, making the quantitative assessment results of haze absorption by green space more scientific and reasonable. However, the results revealed that the ecosystem service function of haze absorption by green space in China from 2001 to 2018 shows a trend of first increasing and then decreasing, suggesting that the forest area with high haze absorbing capacity should be increased when adjusting the structure of ecological land use, and the occupation of cultivated land due to the rapid expansion of construction land should be regulated to improve the ability of green space to alleviate haze.

Previous literatures explored the responses of ecosystem service functions to land use change, mainly through analyses of water yield (Li et al. 2018), soil conservation (Zhu et al. 2018), habitat quality (Dai et al. 2019), biodiversity protection (Reiss and Chifflard 2018), and climate regulation (Yang and Wang 2019). However, there are few studies on the haze absorption by green space. Moreover, previous studies conducted assessments of ecological quality. Munné et al. (Munné et al. 2003) evaluated riparian habitat quality using an index combining total riparian vegetation cover, cover structure, cover quality and channel alterations that is easy to calculate and can be used with any other index of water quality to assess the ecological status of streams and rivers. The macroalgal species richness and composition of intertidal rocky seashores has been used by researchers in the assessment of ecological quality under the European Water Framework Directive (Wells et al. 2007). Using GIS and remote-sensing and factor-analysis techniques, some scholars analyzed UGS landscape patterns in the compact city of Hong Kong to determine the landscape-ecological quality of different land uses and districts (Tian et al. 2014). Some experts have analyzed the scale, quality and diversity of green infrastructure through remote-sensing techniques and NDVI combined with fieldwork verification at two scales, the local and regional (Calderón-Contreras and Quiroz-Rosas 2017), and others have conducted research combining ecological quality with ecosystem services. Paetzold et al. (Paetzold et al. 2010) assessed the relationship between ecosystem quality and ecosystem quality, and Yan et al. (Yan et al. 2016) established the assessment framework including V (vigor: NPP), O (organization: proportion of natural ecosystem area, SHDI, and the contagion index [CONTAG]), and R (resilience: ecological elasticity) to analyze the ecosystem services of soil and water conservation based on ecosystem quality. Finally, Sauvage et al. simulated the role of riverbed compartments in the regulation of water quality as an ecological service (Sauvage et al. 2018). Nevertheless, there have been few studies on the quality of green space, so there has been little research on the ecosystem service functions of haze absorption by green space based on its quality. Therefore, this paper analyzed the ecosystem service functions of haze absorption by green space based on its quality, improving the assessment method of previous studies that only considered the area of green space and providing an improved method for evaluating this ecosystem services, and also providing a reference for the prevention and control of haze and the coordinated development of regional societies, the economy and the environment.

There is a correlation between landscape patterns and ecosystem service functions (Garcia et al. 2014; Gong et al. 2019). This paper considers China as the research area and analyzes the relationship between landscape patterns and the ecosystem service functions of haze absorption by green space, landscape diversity (SHDI), fragmentation (PD and SHAPE_AM) and connectivity (IJI) at the national scale, and the correlation coefficients between SHDI, PD, and the ecosystem service functions of the absorption of SO_2 and NO_X, dust retention and total ecosystem services exhibited significant negative relationships. These results are essentially identical to those of Lu et al. (Lu et al. 2018) and Wu et al. (Wu et al. 2015) but differ from those of Zou et al. (Zou et al. 2016).

Uncertainty in ecosystem service assessments has been demonstrated and analyzed by previous studies (Bei et al. 2017; Hou et al. 2013), and haze is affected by industrial pollution sources, meteorological conditions and plant coverage, and these factors affect each other. Therefore, only considering the influencing factor of green space will lead to uncertainty in the study of haze absorption (Snell et al. 2018). Furthermore, the accuracy of input data, model structure, and parameter settings all lead to uncertainty in ecosystem service research (Baustert et al. 2018; Stritih et al. 2019). This study demonstrated uncertainty in the estimation of ecosystem service functions, mainly because ecosystem service functions of haze absorption were estimated by multiplying the area of each land use type by the corresponding functions coefficients.

This paper also has some limitations. First of all, there are many factors affecting haze, including natural factors such as vegetation coverage (Zhang 2019), social and economic factors are comprised of population density, industrial structure and industrial emissions (Li et al. 2016), and meteorological factors consisting of wind speed and rainfall (Bei et al. 2016). This paper only considered the haze absorption by green space, which has some shortcomings. In the future, it should be combined with meteorological conditions, pollution sources and socio-economic factors. Secondly, we must combine field observation data to obtain per-unit functions for the absorption of SO_2 and NO_X and dust retention of different green space types, thus making the results more accurate, and future research should also collect more detailed data on green space and select appropriate parameters to improve the accuracy of the calculations. This paper utilizes the functions coefficient method to evaluate the ecosystem service functions of haze absorption by green space and preliminarily explored the ecosystem service functions of SO_2 and NO_X absorption and dust retention by green space for 2001–2018 in China. A mechanistic model that includes haze diffusion, haze absorption by green space, an assessment of ecosystem service function modules, and the ecosystem service functions of haze absorption by green space should be established to produce more accurate and objective results, and to explore more reasonable methods for future studies (Wang et al. 2016). The application of a national-scale analysis of the ecosystem service functions of haze absorption by green space would ameliorate the shortcomings of the small-scale analyses in previous studies and would enrich research into the effect of scale on the ecosystem service functions of haze absorption by green space. The acquisition of large-scale and high-precision remote-sensing data is still an important direction for future research.

Conclusions

This paper analyzes the temporal and spatial distributions and sensitivities of the ecosystem service functions of haze absorption by green space based on its quality in 2001, 2004, 2007, 2010, 2013, 2016 and 2018 in China. The main conclusions of this work are as follows:

(1) In general, the ecosystem service functions of haze absorption by green space exhibited first an increasing and then decreasing trend from 2001–2018 in China, increasing by 144652.20 million Kg (1.61%) in 2001–2013 primarily due to the implementation of the Three North Shelterbelt Development Program, the Conversion from Cropland to Forest Program and the Natural Forest Protection Program by the Chinese government. However, the ecosystem service functions decreased by 1410584.00 million Kg from 2013–2018, a decrease of 15.42%, primarily because of adjustment of ecological land structure and the reduction of arable land caused by the expansion of construction land. The contributions of forest cover to the ecosystem service values of haze absorption by green space were the largest, with values of 98.68%, 98.67%, 98.68%, 98.75%, 98.77%, 98.17% and 98.16% in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively. The primary ecological function of haze absorption by green space was mainly dust retention, which accounted for 98.09%, 98.15%, 98.18%, 98.26%, 98.32%, 98.30% and 98.29% of the total in 2001, 2004, 2007, 2010, 2013, 2016 and 2018, respectively.

(2) Different ecosystem service functions exhibited great differences in spatial distribution within the same year but small differences between years. In conclusion, the results show that the ecosystem service functions and spatial distribution of haze absorption by green space based on its quality differ greatly from the value considering only the area. Furthermore, the benchmark and corrected values of the contribution rates of haze absorption by different types of green space and ecosystem service functions are different, but the values show a consistent trend. The contribution rates are ranked from largest to smallest as forest cover, grass land and arable land, and the order of ecosystem service function is dust retention, absorption of SO_2, and absorption of NO_X. Moreover, the spatial distributions of the benchmark and corrected values also exhibit the same distribution trend. In the same year, the spatial distribution of the ecosystem service values of haze absorption by green space is very different, but there is little difference among the different years.

(3) The coefficients of sensitivity for the ecosystem service functions for forest cover are elastic with values of 0.9868 in 2001, 2004 and 2007, 0.9875 in 2010, 0.9877 in 2013, 0.9817 in 2016 and 2018, respectively, and the change rates were ± 49.3424%, ± 49.3398%, ± 49.3405%, ± 49.3767%, ± 49.3832%, ± 49.0861% and ± 49.0842%, respectively. The coefficients of sensitivity for arable land and grass land were inelastic. There was a significant negative relationship between

PD and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services, with the correlation coefficients of -0.407, -0.511, -0.330 and -0.332, respectively. Nevertheless, the correlation coefficients between SHAPE_AM and the ecosystem service functions of SO_2 absorption, NO_x absorption, dust retention and total ecosystem services exhibited significant positive relationships with correlation coefficients of 0.650, 0.634, 0.568 and 0.570, respectively. The green space landscape pattern, which exhibited a uniform patch distribution, has an important effect on the absorption of polluted gases, dust retention and air purification. A higher density of green space patches is accompanied by lower levels of fragmentation and higher levels of air purification.

(4) This paper analyzes and evaluates ecosystem service functions and the spatial distributions thereof, based on the quality of green space, providing a basis for further improving the method for calculating haze absorption by green space and revealing the relationship between ecosystem service functions and landscape patterns. This work is important for the rational planning and improvement of green space ecosystems and for improving the city environment.

(5) This paper analyzes the ecosystem service functions of haze absorption by green space in China, and further research should focus on two approaches. The first is the development of a mechanistic model of the ecosystem service functions of haze absorption by green space that should consist of three modules including a haze diffusion module, a module for haze absorption by green space, and a module that evaluates ecosystem service functions. By including rainfall, wind speed, pollution sources, land use and vegetation types, the function coefficients for haze absorption and other data can be collected in a database. After the model is calibrated and validated, the ecosystem service functions dynamics of haze absorption by green space can be analyzed under different green space and climate change scenarios to predict future changes. The second approach includes a first-tier classification of green space to evaluate the ecosystem service functions of haze absorption in this paper, but second-tier classifications can reflect the differences between different green space types, thus providing more objective and reasonable results. Compared to a first-tier classification of forest cover, second-tier classifications, such as trees and shrubs, have different impacts on the ecosystem service functions of haze absorption. Therefore, further research should provide in-depth explorations of second-tier classifications of green space.

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References

Alavipanah S, Wegmann M, Qureshi S, Weng Q, Koellner T (2015) The role of vegetation in mitigating urban land surface temperatures: A case study of Munich, Germany during the warm season. Sustainability 7(4): 4689–4706. https://doi.org/10.3390/su7044689

An Z, Huang R-J, Zhang R, Tie X, Li G, Cao J, Zhou W, Shi Z, Han Y, Gu Z, Ji Y (2019) Severe haze in Northern China: A synergy of anthropogenic emissions and atmospheric processes. Proceedings of the National Academy of Sciences of the United States of America 116(18): 8657–8666. https://doi.org/10.1073/pnas.1900125116

Baustert P, Othoniel B, Rugani B, Leopold U (2018) Uncertainty analysis in integrated environmental models for ecosystem service assessments: Frameworks, challenges and gaps. Ecosystem Services 33(2): 110–123. https://doi.org/10.1016/j.ecoser.2018.08.007

Beckett KP, Freer-Smith PH, Taylor G (1998) Urban woodlands: Their role in reducing the effects of particulate pollution. Environmental Pollution 99(3): 347–360. https://doi.org/10.1016/S0269-7491(98)00016-5

Bei N, Xiao B, Meng N, Feng T (2016) Critical role of meteorological conditions in a persistent haze episode in the Guanzhong basin, China. The Science of the Total Environment 550(1): 273–284. https://doi.org/10.1016/j.scitotenv.2015.12.159

Bei N, Wu J, Elser M, Tian F, Cao J, El-Haddad I, Li X, Huang R, Li Z, Long X (2017) Impacts of meteorological uncertainties on the haze formation in Beijing–Tianjin–Hebei (BTH) during wintertime: A case study. Atmospheric Chemistry and Physics 17(23): 14579–14591. https://doi.org/10.5194/acp-17-14579-2017

Calderón-Contreras R, Quiroz-Rosas LE (2017) Analysing scale, quality and diversity of green infrastructure and the provision of Urban Ecosystem Services: A case from Mexico City. Ecosystem Services 23(1): 127–137. https://doi.org/10.1016/j.ecoser.2016.12.004

Chen W, Lu S, Pan N, Wang Y, Wu L (2015) Impact of reclaimed water irrigation on soil health in urban green areas. Chemosphere 119(1): 654–661. https://doi.org/10.1016/j.chemosphere.2014.07.035

Chuai X, Wang J, Zhou H, Fan C, Han Y, Gao J (2019) A review of causes, influencing factors and countermeasures of haze pollution in China. Meteorological and Environmental Research 10(3): 25–28.

Cuffney TF, Brightbill RA, May JT, Waite IR (2010) Responses of benthic macroinvertebrates to environmental changes associated with urbanization in nine metropolitan areas. Ecological Applications 20(5): 1384–1401. https://doi.org/10.1890/08-1311.1

Dai L, Li S, Lewis BJ, Wu J, Yu D, Zhou W, Zhou L, Wu S (2019) The influence of land use change on the spatial-temporal variability of habitat quality between 1990 and 2010 in Northeast China. Journal of Forestry Research 30(6): 2227–2236. https://doi.org/10.1007/s11676-018-0771-x

Davies SJ, Unam L (1999) Smoke-haze from the 1997 Indonesian forest fires: Effects on pollution levels, local climate, atmospheric CO2 concentrations, and tree photosynthesis. Forest Ecology and Management 124(2–3): 137–144. https://doi.org/10.1016/S0378-1127(99)00060-2

De Carvalho RM, Szlafsztein CF (2019) Urban vegetation loss and ecosystem services: The influence on climate regulation and noise and air pollution. Environmental Pollution 245(1): 844–852. https://doi.org/10.1016/j.envpol.2018.10.114

Devuyst D, Hens L, De Lannoy W (2001) How green is the city? sustainability assessment and the management of urban environments. Columbia University Press: 1–458. https://doi.org/10.7312/devu11802-intro

Fang X, Tang G, Li B, Han R (2014) Spatial and temporal variations of ecosystem service values in relation to land use pattern in the Loess Plateau of China at town scale. PLoS One 9(10): e110745. https://doi.org/10.1371/journal.pone.0110745

Freer-Smith PH, El-Khatib AA, Taylor G (2004) Capture of particulate pollution by trees: A comparison of species typical of semi-arid areas (Ficus nitida and Eucalyptus globulus) with European and North American species. Water, Air, and Soil Pollution 155(1): 173–187. https://doi.org/10.1023/B:WATE.0000026521.99552.fd

Gao L, Zhao Z, Zhang H, Xuebin G, Xiao M (2012) Adjustment of Haikou city ecosystem services value based on habitat quality and ecological location. Beijing Da Xue Xue Bao. Zi Ran Ke Xue Bao 48(5): 833–840.

Gao Y, Zhang M, Liu Z, Wang L, Wang P, Xia X, Tao M, Zhu L (2015) Modeling the feedback between aerosol and meteorological variables in the atmospheric boundary layer during a severe fog–haze event over the North China Plain. Atmospheric Chemistry and Physics 15(8): 4279–4295. https://doi.org/10.5194/acp-15-4279-2015

Garcia X, Llausas A, Ribas A (2014) Landscaping patterns and sociodemographic profiles in suburban areas: Implications for water conservation along the Mediterranean coast. Urban Water Journal 11(1): 31–41. https://doi.org/10.1080/1573062X.2012.758296

Gómez-Moreno FJ, Artíñano B, Ramiro ED, Barreiro M, Núñez L, Coz E, Dimitroulopoulou C, Vardoulakis S, Yagüe C, Maqueda G, Sastre M, Román-Cascón C, Santamaría JM, Borge R (2019) Urban vegetation and particle air pollution: Experimental campaigns in a traffic hotspot. Environmental Pollution 247(1): 195–205. https://doi.org/10.1016/j.envpol.2019.01.016

Gong J, Xie Y, Cao E, Huang Q, Li H (2019) Integration of InVEST-habitat quality model with landscape pattern indexes to assess mountain plant biodiversity change: A case study of Bailongjiang watershed in Gansu Province. Journal of Geographical Sciences 29(7): 1193–1210. https://doi.org/10.1007/s11442-019-1653-7

Green M, Xu J (2007) Causes of haze in the Columbia River Gorge. Journal of the Air & Waste Management Association 57(8): 947–958. https://doi.org/10.3155/1047-3289.57.8.947

Green OO, Garmestani AS, Albro S, Ban NC, Berland A, Burkman CE, Gardiner MM, Gunderson L, Hopton ME, Schoon ML, Shuster WD (2016) Adaptive governance to promote ecosystem services in urban green spaces. Urban Ecosystems 19(1): 77–93. https://doi.org/10.1007/s11252-015-0476-2

Hamada S, Ohta T (2010) Seasonal variations in the cooling effect of urban green areas on surrounding urban areas. Urban Forestry & Urban Greening 9(1): 15–24. https://doi.org/10.1016/j.ufug.2009.10.002

Han Y, Zhou Z (2015) Evaluation on ecosystem services in haze absorption by urban green land and its spatial pattern analysis in Xi'an. Geographical Research 7(34): 1247–1258.

Hansen AB, Witham CS, Chong WM, Kendall E, Chew BN, Gan C, Hort MC, Lee S-Y (2019) Haze in Singapore–source attribution of biomass burning PM 10 from Southeast

Asia. Atmospheric Chemistry and Physics 19(8): 5363–5385. https://doi.org/10.5194/acp-19-5363-2019

Heusinkveld BG, Steeneveld G, van Hove LWA, Jacobs CMJ, Holtslag AAM (2014) Spatial variability of the Rotterdam urban heat island as influenced by urban land use. Journal of Geophysical Research, D, Atmospheres 119(2): 677–692. https://doi.org/10.1002/2012JD019399

Hong H, Chenxing W, Yuesi W, Zifa W, Jianguo L, Yunfa C (2013) Formation mechanism and control strategies of haze in China. Bulletin of Chinese Academy of Sciences 28(3): 344–352.

Hou Y, Burkhard B, Müller F (2013) Uncertainties in landscape analysis and ecosystem service assessment. Journal of Environmental Management 127(1): 117–131. https://doi.org/10.1016/j.jenvman.2012.12.002

Jin F, Zhang Z, Yu X, Rao L, Niu J, Lu S, Xie Y (2005) Value evaluation of forest ecosystem services of Qilian Mountain in Gansu Province. Science of Soil and Water Conservation 3(1): 53–57.

Kopperoinen L, Itkonen P, Niemelä J (2014) Using expert knowledge in combining green infrastructure and ecosystem services in land use planning: An insight into a new place-based methodology. Landscape Ecology 29(8): 1361–1375. https://doi.org/10.1007/s10980-014-0014-2

Kreuter UP, Harris HG, Matlock MD, Lacey RE (2001) Change in ecosystem service values in the San Antonio area, Texas. Ecological Economics 39(3): 333–346. https://doi.org/10.1016/S0921-8009(01)00250-6

Kuttler W, Strassburger A (1999) Air quality measurements in urban green areas – a case study. Atmospheric Environment 33(24–25): 4101–4108. https://doi.org/10.1016/S1352-2310(99)00151-X

Larondelle N, Haase D (2013) Urban ecosystem services assessment along a rural–urban gradient: A cross-analysis of European cities. Ecological Indicators 29(1): 179–190. https://doi.org/10.1016/j.ecolind.2012.12.022

Li Q, Cheng K, Yang X (2016) Economic and social analysis of haze reduction dilemma in China. Environmental and Earth Sciences Research Journal 3(1): 14–22. https://doi.org/10.18280/eesrj.030103

Li S, Yang H, Lacayo M, Liu J, Lei G (2018) Impacts of land-use and land-cover changes on water yield: A case study in Jing-Jin-Ji, China. Sustainability 10(4): 960. https://doi.org/10.3390/su10040960

Li X, Gao Z, Li Y, Gao CY, Ren J, Zhang X (2019) Meteorological conditions for severe foggy haze episodes over north China in 2016–2017 winter. Atmospheric Environment 199(1): 284–298. https://doi.org/10.1016/j.atmosenv.2018.11.042

Liu H-L, Shen Y-S (2014) The impact of green space changes on air pollution and microclimates: A case study of the taipei metropolitan area. Sustainability 6(12): 8827–8855. https://doi.org/10.3390/su6128827

Liu WP, Yu ZR (2016) Simulation on PM2.5 detention service of green space in Haidian District, Beijing, China. Ying Yong Sheng Tai Xue Bao 27(8): 2580–2586.

Lu DB, Mao WL, Yang DY, Zhao JN, Xu JH (2018) Effects of land use and landscape pattern on PM2.5 in Yangtze River Delta, China. Atmospheric Pollution Research 9(4): 705–713. https://doi.org/10.1016/j.apr.2018.01.012

Mahmoud AHA, El-Sayed MA (2011) Development of sustainable urban green areas in Egyptian new cities: The case of El-Sadat City. Landscape and Urban Planning 101(2): 157–170. https://doi.org/10.1016/j.landurbplan.2011.02.008

Maimaitiyiming M, Ghulam A, Tiyip T, Pla F, Latorre-Carmona P, Halik Ü, Sawut M, Caetano M (2014) Effects of green space spatial pattern on land surface temperature: Implications for sustainable urban planning and climate change adaptation. ISPRS Journal of Photogrammetry and Remote Sensing 89(1): 59–66. https://doi.org/10.1016/j.isprsjprs.2013.12.010

Margaritis E, Kang J (2016) Relationship between urban green spaces and other features of urban morphology with traffic noise distribution. Urban Forestry & Urban Greening 15(1): 174–185. https://doi.org/10.1016/j.ufug.2015.12.009

McDonald AG, Bealey WJ, Fowler D, Dragosits U, Skiba U, Smith RI, Donovan RG, Brett HE, Hewitt CN, Nemitz E (2007) Quantifying the effect of urban tree planting on concentrations and depositions of PM10 in two UK conurbations. Atmospheric Environment 41(38): 8455–8467. https://doi.org/10.1016/j.atmosenv.2007.07.025

Munné A, Prat N, Solà C, Bonada N, Rieradevall M (2003) A simple field method for assessing the ecological quality of riparian habitat in rivers and streams: QBR index. Aquatic Conservation 13(2): 147–163. https://doi.org/10.1002/aqc.529

Ozimec S, Florijancic T, Boskovic I (2016) Biomonitoring urban air pollution by using lichens in the green space of the university campus in osijek (croatia). Journal of Environmental Protection and Ecology 17(4): 1269–1275.

Paetzold A, Warren PH, Maltby LL (2010) A framework for assessing ecological quality based on ecosystem services. Ecological Complexity 7(3): 273–281. https://doi.org/10.1016/j.ecocom.2009.11.003

Pandey B, Agrawal M, Singh S (2014) Coal mining activities change plant community structure due to air pollution and soil degradation. Ecotoxicology (London, England) 23(8): 1474–1483. https://doi.org/10.1007/s10646-014-1289-4

Park J, Kim JH, Lee DK, Park CY, Jeong SG (2017) The influence of small green space type and structure at the street level on urban heat island mitigation. Urban Forestry & Urban Greening 21(1): 203–212. https://doi.org/10.1016/j.ufug.2016.12.005

Ramakreshnan L, Aghamohammadi N, Fong CS, Bulgiba A, Zaki RA, Wong LP, Sulaiman NM (2018) Haze and health impacts in ASEAN countries: A systematic review. Environmental Science and Pollution Research International 25(3): 2096–2111. https://doi.org/10.1007/s11356-017-0860-y

Reiss M, Chifflard P (2018) Different forest cover and its impact on eco-hydrological traits, invertebrate fauna and biodiversity of spring habitats. Nature Conservation 27: 85–99. https://doi.org/10.3897/natureconservation.27.26024

Rysgaard S, Nielsen TG, Hansen BW (1999) Seasonal variation innutrients, pelagic primary production and grazing in a high-Arctic marine ecosystem, Young Sound, Northeast Greenland. Marine Ecology Progress Series 179(3): 13–25. https://doi.org/10.3354/meps179013

Saphores J-D, Li W (2012) Estimating the value of urban green areas: A hedonic pricing analysis of the single family housing market in Los Angeles, CA. Landscape and Urban Planning 104(3): 373–387. https://doi.org/10.1016/j.landurbplan.2011.11.012

Sauvage S, Vervier P, Naiman RJ, Alexandre H, Bernard-Jannin L, Boulêtreau S, Delmotte S, Julien F, Peyrard D (2018) Modelling the role of riverbed compartments in the regulation of water quality as an ecological service. Ecological Engineering 118(1): 19–30. https://doi.org/10.1016/j.ecoleng.2018.02.018

Selmi W, Weber C, Rivière E, Blond N, Mehdi L, Nowak D (2016) Air pollution removal by trees in public green spaces in Strasbourg city, France. Urban Forestry & Urban Greening 17: 192–201. https://doi.org/10.1016/j.ufug.2016.04.010

Setälä H, Viippola V, Rantalainen A-L, Pennanen A, Yli-Pelkonen V (2013) Does urban vegetation mitigate air pollution in northern conditions? Environmental Pollution 183(1): 104–112. https://doi.org/10.1016/j.envpol.2012.11.010

Snell RS, Elkin C, Kotlarski S, Bugmann H (2018) Importance of climate uncertainty for projections of forest ecosystem services. Regional Environmental Change 18(7): 2145–2159. https://doi.org/10.1007/s10113-018-1337-3

Song S, Gao M, Xu W, Sun Y, Worsnop DR, Jayne JT, Zhang Y, Zhu L, Mei L, Zhen Z (2019) Possible heterogeneous chemistry of hydroxymethanesulfonate (HMS) in northern China winter haze. Atmospheric Chemistry and Physics 19(2): 1357–1371. https://doi.org/10.5194/acp-19-1357-2019

Stritih A, Bebi P, Gret-Regamey A (2019) Quantifying uncertainties in earth observation-based ecosystem service assessments. Environmental Modelling & Software 111(1): 300–310. https://doi.org/10.1016/j.envsoft.2018.09.005

Sun W, Shao M, Granier C, Liu Y, Ye C, Zheng J (2018) Long-term trends of anthropogenic SO2, NOx, CO, and NMVOCs emissions in China. Earth's Future 6(8): 1112–1133. https://doi.org/10.1029/2018EF000822

Thach T-Q, Wong C-M, Chan K-P, Chau Y-K, Chung Y-N, Ou C-Q, Yang L, Hedley AJ (2010) Daily visibility and mortality: Assessment of health benefits from improved visibility in Hong Kong. Environmental Research 110(6): 617–623. https://doi.org/10.1016/j.envres.2010.05.005

Tian Y, Jim CY, Wang H (2014) Assessing the landscape and ecological quality of urban green spaces in a compact city. Landscape and Urban Planning 121(121): 97–108. https://doi.org/10.1016/j.landurbplan.2013.10.001

Wang G, Innes JL, Lei J, Dai S, Wu SW (2007) China's forestry reforms. Science 318(5856): 1556–1557. https://doi.org/10.1126/science.1147247

Wang Q, Zhuang G, Huang K, Liu T, Deng C, Xu J, Lin Y, Guo Z, Chen Y, Fu Q, Fu JS, Chen J (2015) Probing the severe haze pollution in three typical regions of China: Characteristics, sources and regional impacts. Atmospheric Environment 120(1): 76–88. https://doi.org/10.1016/j.atmosenv.2015.08.076

Wang R, Li R, Sun H (2016) Haze removal based on multiple scattering model with superpixel algorithm. Signal Processing 127(3): 24–36. https://doi.org/10.1016/j.sigpro.2016.02.003

Waters EJ, Hayasaka Y, Tattersall DB, Adams KS, Williams PJ (1998) Sequence analysis of grape (Vitis vinifera) berry chitinases that cause haze formation in wines. Journal of Agricultural and Food Chemistry 46(12): 4950–4957. https://doi.org/10.1021/jf980421o

Wells E, Wilkinson M, Wood P, Scanlan C (2007) The use of macroalgal species richness and composition on intertidal rocky seashores in the assessment of ecological quality under the European Water Framework Directive. Marine Pollution Bulletin 55(1): 151–161. https://doi.org/10.1016/j.marpolbul.2006.08.031

Wu J, Xie W, Li W, Li J (2015) Effects of urban landscape pattern on PM2.5 pollution-a Beijing case study. PLoS One 10(11): 0142449. https://doi.org/10.1371/journal.pone.0142449

Wu JN, Zhang P, Yi HT, Qin Z (2016) What causes haze pollution? an empirical study of PM2.5 concentrations in Chinese cities. Sustainability 8(2): 131–144. https://doi.org/10.3390/su8020132

Xue Q, Lan Z, Lian S, Chen Y, Cao K, Zhao Z, Ma X (2018) Analysis of atmospheric visibility degradation in early haze based on the nucleation clustering model. Atmospheric Environment 193(1): 205–213. https://doi.org/10.1016/j.atmosenv.2018.09.019

Yan Y, Zhao C, Wang C, Shan P, Zhang Y, Wu G (2016) Ecosystem health assessment of the Liao River Basin upstream region based on ecosystem services. Acta Ecologica Sinica 36(4): 294–300. https://doi.org/10.1016/j.chnaes.2016.06.005

Yang Y, Wang K (2019) The effects of different land use patterns on the microclimate and ecosystem services in the agro-pastoral ecotone of Northern China. Ecological Indicators 106(1): UNSP 105522. https://doi.org/10.1016/j.ecolind.2019.105522

Yang Y, Liao H, Lou S (2016) Increase in winter haze over eastern China in recent decades: Roles of variations in meteorological parameters and anthropogenic emissions. Journal of Geophysical Research, D, Atmospheres 121(21): 13050–13065. https://doi.org/10.1002/2016JD025136

Ye WH, Wei B, Tong C (1998) Measurement and application of urban ecological compensation. Zhongguo Huanjing Kexue 8(4): 298–301.

Ye LP, Fang LC, Tan WF, Wang YQ, Huang Y (2016) Exploring the effects of landscape structure on aerosol optical depth (AOD) patterns using GIS and HJ-1B images. Environmental Science. Processes & Impacts 18(2): 265–276. https://doi.org/10.1039/C5EM00538H

Yu S, Li P, Wang L, Wu Y, Wang S, Liu K, Zhu T, Zhang Y, Hu M, Zeng L, Zhang X, Cao J, Alapaty K, Wong DC, Pleim J, Mathur R, Rosenfeld D, Seinfeld JH (2018) Mitigation of severe urban haze pollution by a precision air pollution control approach. Scientific Reports 8(1): 8151. https://doi.org/10.1038/s41598-018-26344-1

Zhang D (2019) Study on the correlation between the temporal and spatial variation of atmospheric PM2.5 and PM10 and vegetation coverage in Xi'an City. Master Thesis. Northwest University (Xi`an).

Zhang L, Wang T, Lv M, Zhang Q (2015a) On the severe haze in Beijing during January 2013: Unraveling the effects of meteorological anomalies with WRF-Chem. Atmospheric Environment 104(1): 11–21. https://doi.org/10.1016/j.atmosenv.2015.01.001

Zhang P, He L, Fan X, Huo P, Liu Y, Zhang T, Pan Y, Yu Z (2015b) Ecosystem service value assessment and contribution factor analysis of land use change in Miyun County, China. Sustainability 7(6): 7333–7356. https://doi.org/10.3390/su7067333

Zhang Z, Zhang X, Gong D, Kim S-J, Mao R, Zhao X (2016) Possible influence of atmospheric circulations on winter haze pollution in the Beijing–Tianjin–Hebei region, northern China. Atmospheric Chemistry and Physics 16(2): 561–571. https://doi.org/10.5194/acp-16-561-2016

Zhu X, Liu W, Jiang XJ, Wang P, Li W (2018) Effects of land-use changes on runoff and sediment yield: Implications for soil conservation and forest management in Xishuangbanna, Southwest China. Land Degradation & Development 29(9): 2962–2974. https://doi.org/10.1002/ldr.3068

Zou B, Xu S, Sternberg T, Fang X (2016) Effect of land use and cover change on air quality in urban sprawl. Sustainability 8(7): 677. https://doi.org/10.3390/su8070677

Zoulia I, Santamouris M, Dimoudi A (2009) Monitoring the effect of urban green areas on the heat island in Athens. Environmental Monitoring and Assessment 156(1–4): 275–292. https://doi.org/10.1007/s10661-008-0483-3