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Floristic composition and plant diversity in distribution areas of native species congeneric with *Betula halophila* in Xinjiang, northwest China

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Abstract

Betula halophila, a wild plant with extremely small populations, is endemic to Xinjiang, northwest China. Its wild populations have declined severely in the field. Understanding the patterns of floristic composition where congeneric species of *B. halophila* are distributed and their determinants is a necessary step to restore the wild populations. Based on literature records, specimen information, shared public data and field survey data, the patterns of floristic composition, diversity and environmental conditions of seed flora within the distribution areas of five native species (i.e. *B. tianschanica*, *B. microphylla*, *B. pendula*, *B. rotundifolia* and *B. humilis*), congeneric with *B. halophila*, were examined. The results are as follows. (1) There were 3013 species, 693 genera and 108 families of seed plants in the distribution area of these congeneric species of *B. halophila*, which accounted for 86.16%, 94.54% and 93.91% of the total seed plants in Xinjiang, respectively. (2) The family composition of seed flora in the distribution area of these congeneric species of *B. halophila* was mainly cosmopolitan; the genus composition of seed flora was dominated by temperate, mainly of northern temperate and Mediterranean components. (3) There are no significant differences existing in plant richness amongst the areas where each of the five congeneric species (*B. tianschanica*, *B. microphylla*, *B. pendula*, *B. rotundifolia* and *B. humilis*) are distributed. (4) The influence of climate factors on species richness is significant across the whole distribution areas of the *Betula* genus, while the main environmental factors determining species richness are different amongst distribution areas of different species. Climate factors impacted significantly on species richness in distribution areas of tree species, but not in distribution areas of shrub species. This study provides a preliminary guideline for the conservation of *B. halophila*, a wild plant with extremely small populations in the field.

Keywords

Betula, congeneric species, geographic elements, richness, wild plant with extremely small populations

Introduction

Betula halophila, a deciduous shrub species in the genus *Betula* of the family Betulaceae, is an endangered and salt-tolerant plant, which is distributed only in Xinjiang in northwest China. In 1999, the species is firstly listed as one of the key protected wild plants of China with a second-class protection level (State Forestry Administration and Ministry of Agriculture 1999). In 2011, the species was listed as one of the 120 wild plants with extremely small populations in China (State Forestry Administration 2011). The species was firstly found in Yanchi Town, Altay City, Xinjiang in 1956 (Xinjiang Altay Forestry Institute, Xinjiang Forestry Academy 2010). No one has paid attention to it since then. Until 1996, the outstanding salt-tolerance character of this species has attracted researchers and then it aroused great interest of researchers again. In 2000 and 2001, experts carried out ex situ conservation on some survival individuals of this species in the field and transplanted them into the plant garden of the Academy of Forestry in Xinjiang (Xinjiang Altay Forestry Institute, Xinjiang Forestry Academy 2010). In 2003, some other wild individuals of this species suffered serious flooding. After that, the relevant researchers found no trace of this species in field.

Betula genus, belonging to Betulaceae, plays an important role in the flora and vegetation composition of Xinjiang. *Betula* is an essential part of the typical mountain broad-leaved forest in Xinjiang. There is a close relationship between the small leaf broad-leaved forest and the typical mountain coniferous forest in Xinjiang. *Betula* genera in Xinjiang include six species (*B. tianschanica*, *B. microphylla B. pendula*, *B. rotundifolia*, *B. humilis* and *B. halophila*) (Editorial Committee of Flora Xinjiangensis 1992). These six species are mainly distributed in the mountains of northern Xinjiang, including Altay, Tianshan and Western Junggar. Amongst them, *B. tianshanensis* is mainly distributed in Tianshan Mountain and the other five species are mainly distributed in the Altai mountain area. Only *B. halophila* is restricted in Xinjiang and occurred in Yanchi Town of Altai mountain area. *B. pendula* and *B. tianschanica* are the pioneer species of typical mountain forest spruce in Xinjiang. They play an important role in the natural regeneration of spruce community. In addition, all species of *Betula* within Xinjiang mostly distribute in Tianshan Mountain and Altai Mountain. The diversity hotspots of seed plants of Xinjiang are mainly located in both of these two mountains (Huang et al. 2018). Therefore, the main distribution area of *Betula* in Xinjiang is consistent with the diversity distribution centre of plant species in Xinjiang.

Researchers have done a lot of research on all species of *Betula* native in Xinjiang. These works are mainly related to taxonomy, phylogeny and physiology. The systematic classification of *Betula* in Xinjiang has always been controversial (Wang et al. 2016). *B. halophila* often forms salt crystals on the surface of the body to reduce the content of Ca^{2+} ion in the body and cope with the salt-stress environment (Zhang et al. 2009). The in-depth research has also focused on the salt-tolerance mechanism of the species (Shao et al. 2018). This species usually reduces the effect of stress by regulating its own osmotic substances, but this ability is very limited (Li et al. 2009). In addition, some research has been carried out on the intraspecific biomass allocation (Zhang et al. 2017), tree ring response to climate (Zhang et al. 2015) and intraspecific ISSR analysis (You et al. 2017) about its congeneric species. These studies provide an important basis for further understanding of the biological characteristics of *Betula* in Xinjiang, but the references for the ecological characteristics of these species are very limited.

The formation of flora is a comprehensive reflection of the evolution and temporal and spatial distribution of the flora in a certain natural historical environment. The flora of a specific region not only reflects the causal relationship between the total plants and the environment in the region, but also reflects the evolution of the flora in the geological history (Takhtajan1969; Wu et al. 2011). At the same time, flora is the main component of the biosphere, the first producer and the most active factor of capacity exchange and material cycle in ecosystem. Flora is the entity of vegetation, which is the reflection of natural geographical environment and the verification of environmental change (Wang 1992). Nowadays, biodiversity conservation is one of the top topics focused on by ecologists and biogeographers. It is closely related to flora (Chen et al. 2014; Weigelt et al. 2020; Wu 2004). In the study and application of basic theories, such as formation mechanism (Mienna et al. 2019) and change prediction of regional species diversity (Bruelheide et al. 2020), evaluation and protection of rare and endangered species (Phillips et al. 2011; Zhang et al. 2014), biodiversity evaluation, construction and planning of natural reserves, the floristic regionalisation and flora characteristics have played important roles, both in theory and practice (Wu 2004).

Arid regions account for 38–41% of the global land area. Compared with other regions and ecosystems, arid ecosystems are very sensitive to climate change and human activities (Reynolds et al. 2007; Maestre et al. 2015). Xinjiang, located in the hinterland of Eurasia, is a typical arid climate area with mountains and also breeds rich and unique biodiversity resources. The arid climate conditions lead to the salinisation of the soil and make it increasingly problematic (Wei and Xu 2005). Furthermore, with the rapid development of modern agriculture in oases, the degree of salinisation is still increasing. Therefore, Xinjiang has become a large area where soil salinisation is widely distributed (Tian et al. 2000). This is a great potential threat to the natural vegetation and species diversity of Xinjiang. Therefore, it is very important to strengthen the study of the adaptability of natural vegetation to salinisation. Research on the protection of salt-tolerant and alkali-resistant plants plays an important role in the protection of biodiversity in Xinjiang. Many plants in the birch family have strong salt tolerance and adapt to the special environment with a harsh environment (Yang et al. 2006). Amongst them, the most salt-tolerant is *B. halophila*, whose salt resistance is three times that of common birch species and ten times that of non-birch species (Wang 2003).

B. halophila is a typical wild plant with extremely small populations, which can provide important genetic resources for the cultivation of salt resistant varieties. However, the wild population of this species has not been found for more than ten years. In order to rescue and restore the populations of this species, the most important is to ensure the preservation of this genetic resource as early as possible. In order to expand the existing population size of this species as much as possible, we hope to expand the population number by adopting the way of near natural re-introduction of seedlings, so as to provide a rational basis for fast restoration of the populations. Based on the distribution data of congeneric species of *B. halophila*, we compared the floristic and diversity characteristics of floras within distribution areas of these species and assessed the main factors affecting the diversity patterns of their floras of these congeneric species of *B. halophila* in Xinjiang. We hope our study can provide a scientific guideline for the conservation of *B. halophila*, a wild plant with extremely small populations.

Materials and methods

Study area

Xinjiang is located in the hinterland of Eurasia (34°25'~ 49°10'N, 73°40'~ 96°18'E), with a total area of about 1.6×10^6 km^2, accounting for about one sixth of China's total land area and is the largest provincial administrative unit in China. Xinjiang has a complex terrain with a typical geomorphic pattern of "three mountains and two basins". From north to south, these are Altai Mountains, Junggar Basin, Tianshan Mountains, Tarim Basin and Kunlun Mountains (Mansuer 2012). Xinjiang has a typical continental climate with an annual average temperature of 4–14 °C and an annual average precipitation of about 150 mm. It is characterised by drought, little rain, severe winter and large temperature difference between day and night (Yao et al. 2015). Due to the complex topography and remarkable climate change, Xinjiang has developed a unique floristic composition (Xinjiang Investigation Group of Chinese Academy of Sciences 1978). Although the number of seed plant species in Xinjiang is far less than that in many provinces or regions of China, the proportion of endemic plants in Xinjiang is relatively higher than those in other provinces and regions. Amongst all the provinces in China, Taiwan has the highest proportion of endemic plants of 78%, followed by Xinjiang, with that of 73% (Huang 2010). The flora of Xinjiang mainly belongs to the Central Asia flora, which is quite different from other provinces or regions in China (Takhtajan 1969). Birch plants are essential components of small leaf broad-leaved forest in the mountainous areas of Xinjiang (Xinjiang Investigation Group of Chinese Academy of Sciences 1978). It is also an important pioneer tree species in the succession of mountain forest vegetation (Chen 1999). Birch plants in Xinjiang are mainly distributed in the Altai Mountains, Junggar Basin, western Junggar Mountains and the northern and southern slopes of Tianshan Mountains.

Data collection

Based on a large number of herbarium specimens (http://www.cvh.org.cn/), a list of Chinese seed plant species and distribution information at provincial level (Wu et al. 1994–2012) and other literature sources that mainly included *Flora Xinjiangensis* (Editorial Committee of Flora Xinjiangensis 1992–2011), A Primer of *Flora Xinjiangensis* (Editorial Committee of Flora Xinjiangensis 2014), *Sylva Xinjiangensis* (Yang 2012), *Desert Plants in China* (Lu et al. 2012) and *Rare Endangered Endemic Higher Plants in Xinjiang of China* (Yin 2006), we identified five congeneric species of *B. halophila* occurring in Xinjiang. These five species are *B. tianschanica*, *B. microphylla B. pendula*, *B. rotundifolia* and *B. humilis*. Combining the altitude range with the geographic distribution of all seed plant species occurring in Xinjiang and integrating data of field investigation in the past 10 years (2008–2018), we established a dataset of seed plants distribution in Xinjiang with a spatial unit of 50 km × 50 km (Huang et al. 2018). Based on this dataset, we extracted a subset of distribution data of Xinjiang seed plants, known as the distribution dataset of *Betula* plants in Xinjiang, according to distribution of these five congeneric species of *B. halophila* with a spatial unit of 50 km × 50 km (Fig. 1a–f). The flora of *Betula* in Xinjiang is distributed within four floristic regions (Fig. 1g), i.e. Altai region (IA2), Tianshan region (IA3), Junggar region (IIC5) and Kashgar region (IIC6) by referring to the Floristic Geography of Seed Plants in China (Wu et al. 2011). All species of *Betula* native in Xinjiang are distributed mostly in north Xinjiang, mainly in Tianshan Mountains, Altai Mountains and Junggar Basin (Fig. 1h). Geographic elements of families and genera are collected by referring to Areal-Types of Seed Plants and Their Origin and Differentiation (Wu et al. 2006). Amongst the five species, *B. tianschanica*, *B. microphylla* and *B. pendula*, are tall trees and *B. rotundifolia* and *B. humilis* are shrubs. Climate data are obtained from the data sharing platform of Worldclim Database (http://www.worldclim.org/). The mean annual evapotranspiration data are obtained from the atlas of the biosphere data sharing platform (https://library.McMaster.Ca/maps/geospatial?Location=89).

Data analysis

In this study, species richness (*S*) was used to indicate species diversity. In order to compare the species composition of different flora and different species distribution areas, the Sørensen similarity coefficient of species composition in different flora areas was calculated (Magurran 1988). The formula of this coefficient is as follows: $SI = 2c / (a + b)$, where SI is the Sørensen index, a and b, respectively, the number of species in two species distribution areas, only in one species distribution area and c represents the number of species shared by two species distribution areas. The proportions of geographical elements of family and genus are directly calculated by the number of families and genera per type in total families and genera in species distribution areas. Base on the spatial unit of 50 km × 50 km, the comparative analysis of plant richness in different species distribution areas and the relationship between species richness and

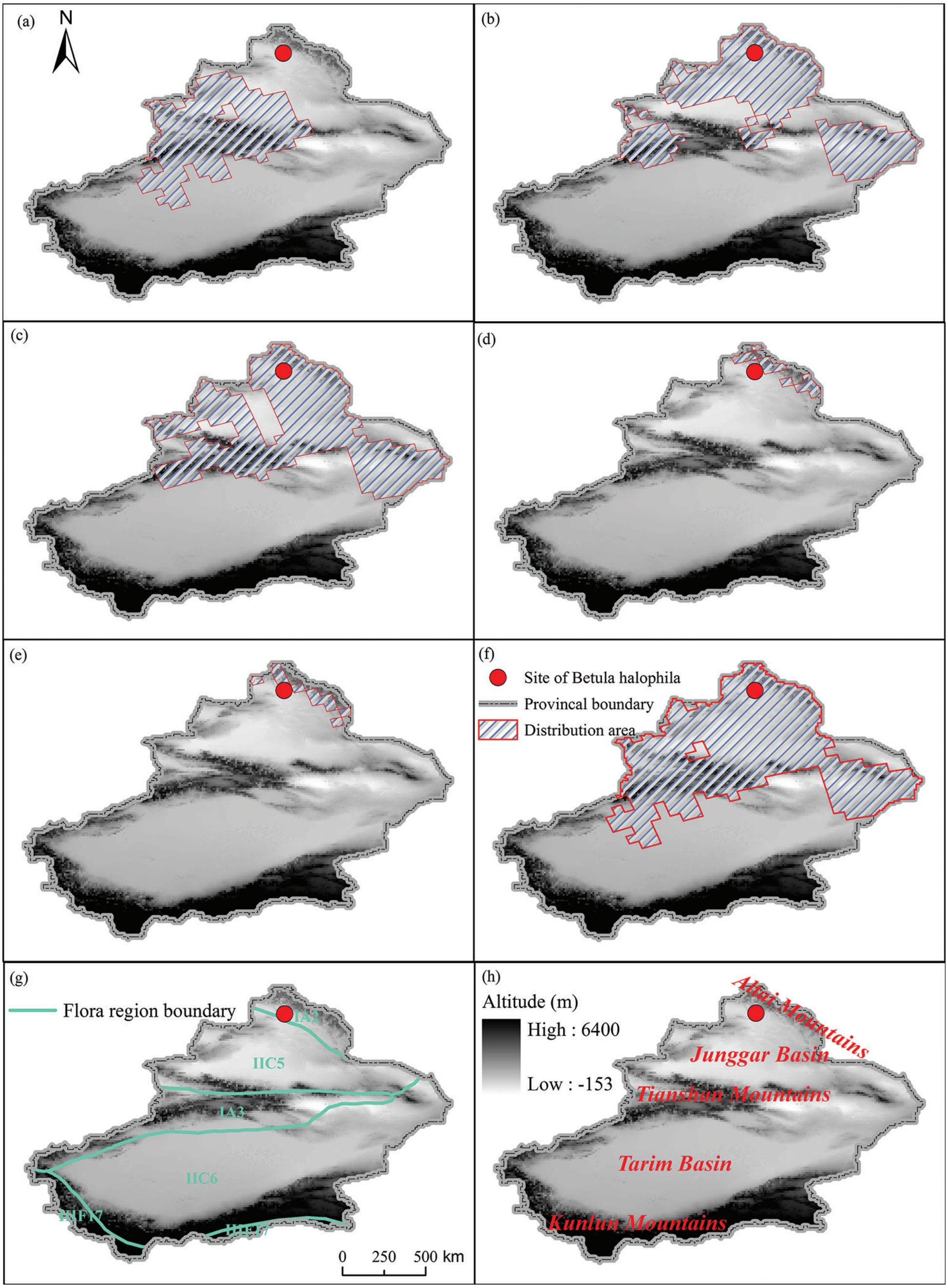

Figure 1. Geographic distribution of congeneric species with *Betula halophila*, flora regions and main geomorphology in Xinjiang, northwest China **a** geographic distribution of *B. tianschanica* **b** geographic distribution of *B. microphylla* **c** geographic distribution of *B. pendula* **d** geographic distribution of *B. rotundifolia* **e** geographic distribution of *B. humilis* **f** geographic distribution of five species of *Betula* genus in Xinjiang **g** geographic distribution of flora regions, IA2 Altai Region, IA3 Tianshan Region, IIC5 Junggar Region, IIC6 Kashgar Region, IIIF17 Tibet, Pamirs and Kunlun Region **h** geographic distribution of main geomorphology.

the main environmental and spatial factors were carried out by ANOVA and Scheffe multiple comparative analyses (Scheffe 1959) and by linear regression analysis, respectively. The above analysis are calculated, in turn, by *sørenson*, *aov*, *scheffe.test* and *lm* functions of *fossil*, *stats*, *agricola* and *vegan* packages in R software (R Core Team 2019). Spatial analysis tools are used to extract the corresponding environmental data values which include longitude, latitude, altitude, mean annual temperature, mean annual precipitation and mean annual evapotranspiration. Longitude and latitude are the longitude and latitude of the central coordinates of each 50 km × 50 km grid. Altitude, mean annual temperature, mean annual precipitation and mean annual evapotranspiration are average altitude, mean annual temperature, mean annual precipitation and mean annual evapotranspiration of each 50 km × 50 km grid, respectively. These four variables are obtained by extracting DEM, mean annual temperature, mean annual precipitation and mean annual evapotranspiration basic data layers of the study area in ArcGIS 9.3 (ESRI 2008). All spatial distribution maps are drawn in ArcGIS.

Results

Floristic composition across distribution areas of the five species congeneric with *B. halophila*

Across all the distribution areas of the five congeneric species of *B. halophila*, there are 3013 species of seed plants, belonging to 693 genera and 108 families. There are 33 species of gymnosperms and 2980 species of angiosperms. Amongst the 108 families, there are 368 species of Asteraceae, 283 species of Poaceae, 253 species of Fabaceae, 152 species of Rosaceae, 147 species of Ranunculaceae, 134 species of Lamiaceae, 128 species of Brassicaceae, 127 species of Chenopodiaceae, 114 species of Caryophyllaceae, 107 species of Apiaceae and 102 species of Liliaceae, all of which contain more than 100 species. Amongst the other families, 63 families contain less than 10 species. There are 22 families with only one species. Amongst the 693 genera, *Astragulus* has the most species, including 92 species, followed by *Carex* (60 species), *Allium* (56 species), *Artemisia* (53 species), *Oxytropis* (41 species), *Ranunculus* (41 species) and *Saussurea* (40 species). In addition, there is only one species in 307 genera.

There are 108 families of seed plants in the distribution area of congeneric species of *B. halophila*. The proportions of the cosmopolitan, temperate and tropical (Qian 2001) are 46.30%, 31.48% and 24.00%, respectively, which can be divided into eight types of geographical elements (Table 1). There are 693 genera of seed plants in the distribution area of the congeneric species of *B. halophila*. The proportions of the cosmopolitan, temperate and tropical are 10.39%, 79.80% and 9.81%, respectively, which can be classified into 15 types of geographical elements (Table 1). Amongst them, the proportions of the three types of geographical elements, Mediterranean region, West Asia and Central Asia, are the highest (Table 1).

The proportions of the cosmopolitan, temperate and tropical families are similar within all five species distribution areas and the proportion of the cosmopolitan fam-

Table 1. Statistics of geographic elements for families and genera in distribution areas for congeneric species with *Betula halophila* in Xinjiang, northwest China.

Geographic elements		Number	Percentage of	Number	Percentage of
Major types	Types	of families	total families (%)	of genera	total genera (%)
Cosmopolitan	Cosmopolitan	50	46.30	72	10.39
Tropical	Tropical Asia- Australia and Tropical America	21	19.44	42	6.06
	Tropical and Subtropical East Asia & (South) Tropical America disjuncted	3	2.78	5	0.72
	Old World Tropics	/	/	8	1.15
	Tropical Asia to Tropical Australia	/	/	4	0.58
	Tropical Asia to Tropical Africa	/	/	3	0.43
	Tropical Asia	/	/	6	0.87
Temperate	North Temperate	26	24.07	228	32.90
	East Asia & North America disjuncted	1	0.93	17	2.45
	Old World Temperate	4	3.70	120	17.32
	Temperate Asia	/	/	25	3.61
	Mediterranean, western Asia to central Asia	2	1.85	77	11.11
	Central Asia	1	0.93	62	8.95
	East Asia	/	/	15	2.16
	Endemic to China	/	/	9	1.30

Note: '/' indicates there is no corresponding geographical element type.

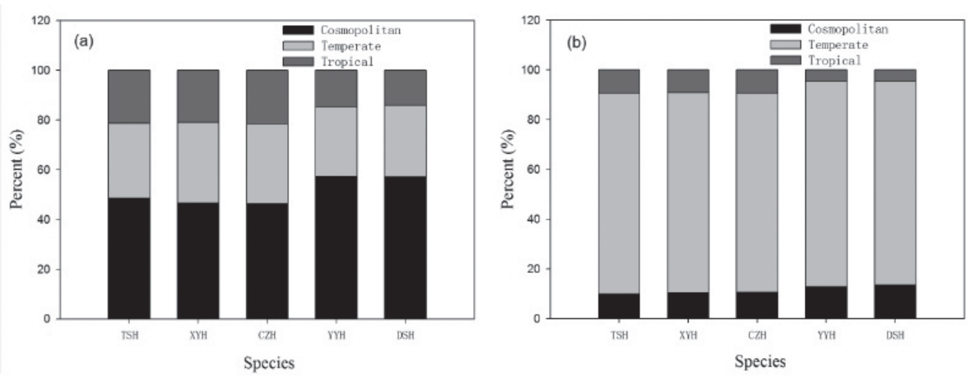

Figure 2. Percentage of geographic elements of compositions of different distribution areas for congeneric species with *Betula halophila* **a** family in different distribution areas for congeneric species with *B. halophila* **b** genus in different distribution areas for congeneric species with *B. halophila* (TSH: *B. tianschanica*; XYH: *B. microphylla*; CZH: *B. pendula*; YYH: *B. rotundifolia*; DSH: *B. humilis*) in Xinjiang, northwest China.

ily is dominant (Fig. 2a). The proportion of the cosmopolitan, temperate and tropical genera within all five species distribution areas are also similar, but the proportion of the temperate genus is absolutely dominant (Fig. 2b). The proportions of the cosmopolitan family and genus within species distribution areas of three shrub congeneric species of *B. halophila* are higher than those within species distribution areas of two tree congeneric species of *B. halophila* (Fig. 2a, b). It can be seen that the cosmopolitan families and temperate genera are dominant in the geographical components of seed flora in the distribution areas of congeneric species of *B. halophila* in Xinjiang.

Richness and similarity of plants amongst different species distribution areas of the five species congeneric with *B. halophila*

There was no significant difference in plant richness amongst the distributional areas of each of the five species congeneric with *B. halophila*, but there was a trend (Fig. 3a–c). The average family richness in distribution area of *B. tianschanica*, *B. microphylla*, *B. pendula*, *B. rotundifolia* and *B. humilis* are 49, 47, 48, 55 and 56, respectively. Family richness in all distribution areas of shrub species is higher than that in distribution areas of tree species (Fig. 3a). The average genus richness in distribution areas of the five species is 193, 197, 202, 235 and 250, respectively. The average genus richness in the distribution area of shrub species is higher than that in the distribution area of three tree species (Fig. 3b). The average species richness in the distribution area of the five species is 347, 358, 366, 473 and 520, respectively. The species richness in the distribution area of shrub species is higher than that in the distribution area of tree species, which is consistent with that of families and genera (Fig. 3c). The analysis of species similarity in different species distribution areas shows that the species composition in the distribution areas of *B. pendula* and *B. rotundifolia* is the most similar (Table 2) and that in the distribution areas of *B. pendula* and *B. microphylla* is the lowest (Table 2).

Changes of species richness with geographic and environmental factors

The species richness in the distribution areas of the five congeneric species of *B. halophila* varied not significantly with the increase in longitude ($R^2 = 0.00$, $P > 0.05$), but significantly increased with the increase in latitude ($R^2 = 0.27$, $P < 0.01$) (Fig. 4a, b). With the increase in altitude, the species richness also has no significant change (Fig. 4c). The species richness decreased with the increase in annual average temperature (Fig. 4d) and increased significantly with the increase in annual average precipitation and actual evapotranspiration (Fig. 4e, f). In all distribution areas of five species, there is a significantly negative correlation between altitude and mean annual temperature (Suppl. material 1: Fig. S1). However, the species richness in the distribution area of any species was not related to altitude (Table 3). In the distribution areas of different species, the species richness in areas of three tree species (*B. tianschanica*, *B. microphylla* and *B. pendula*) increased significantly with the increase of longitude, latitude, annual average temperature, annual precipitation and annual potential evapotranspiration

Table 2. Sørenson similarity coefficients of compositions in distribution areas of different species for congeneric species with *Betula halophila* in Xinjiang, northwest China.

	B. tianschanica	*B. microphylla*	*B. pendula*	*B. rotundifolia*	*B. humilis*
B. tianschanica	/	/	/	/	/
B. microphylla	1.04	/	/	/	/
B. pendula	1.06	1.02	/	/	/
B. rotundifolia	1.35	1.39	1.40	/	/
B. humilis	1.32	1.36	1.37	1.03	/

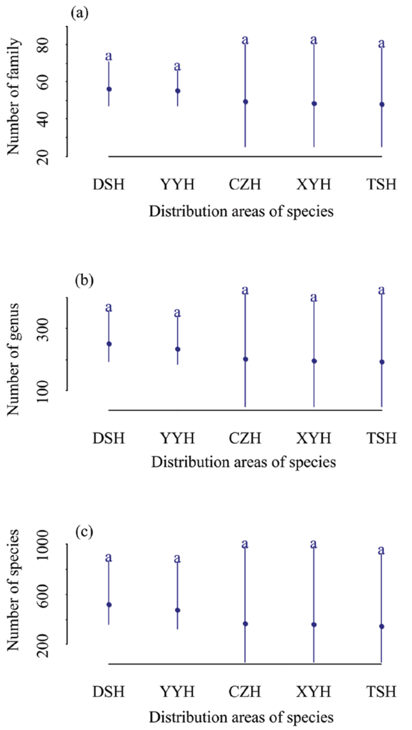

Figure 3. Comparisons of mean plant richness of different distribution areas for congeneric species of *Betula halophila* based on the spatial unit of 50 km × 50 km **a** number of families in different distribution areas for congeneric species with *B. halophila* **b** number of genera in different distribution areas for congeneric species with *B. halophila* **c** number of species in different distribution areas for congeneric species with *B. halophila*)(TSH: *B. tianschanica*; XYH: *B. microphylla*; CZH: *B. pendul*a; YYH: *B. rotundifolia*; DSH: *B. humilis*) in Xinjiang, northwest China.

Table 3. Correlations (R^2) between the species richness in species distribution areas of congeneric species with *Betula halophila* and their main environmental factors in Xinjiang, northwest China.

Species	Longitude	Latitude	Altitude	Mean annual temperature	Mean annual precipitation	Mean annual evapotranspiration
B. tianschanica	0.26*	0.23*	-0.01	0.03*	0.11*	0.14*
B. microphylla	0.11*	0.22*	-0.01	0.04*	0.14*	0.16*
B. pendula	0.07*	0.19*	-0.00	0.01*	0.09*	0.12*
B. rotundifolia	0.23*	0.01	0.07	0.00	-0.11	-0.11
B. humilis	-0.00	0.02	-0.08	-0.07	-0.08	-0.11

Note: The minus sign before the correlation coefficient denotes negative correlation and the * in the upper right corner denotes significant at P < 0.1.

(Table 3); the species richness in the distribution areas of *B. rotundifolia* increased only with the increase in longitude ($R^2 = 0.23$, $P < 0.1$). The temperature, precipitation and evapotranspiration of *Betula halophila* in the field were almost consistent with the most frequent occurrence of *Betula* species (Suppl. material 2: Fig. S2).

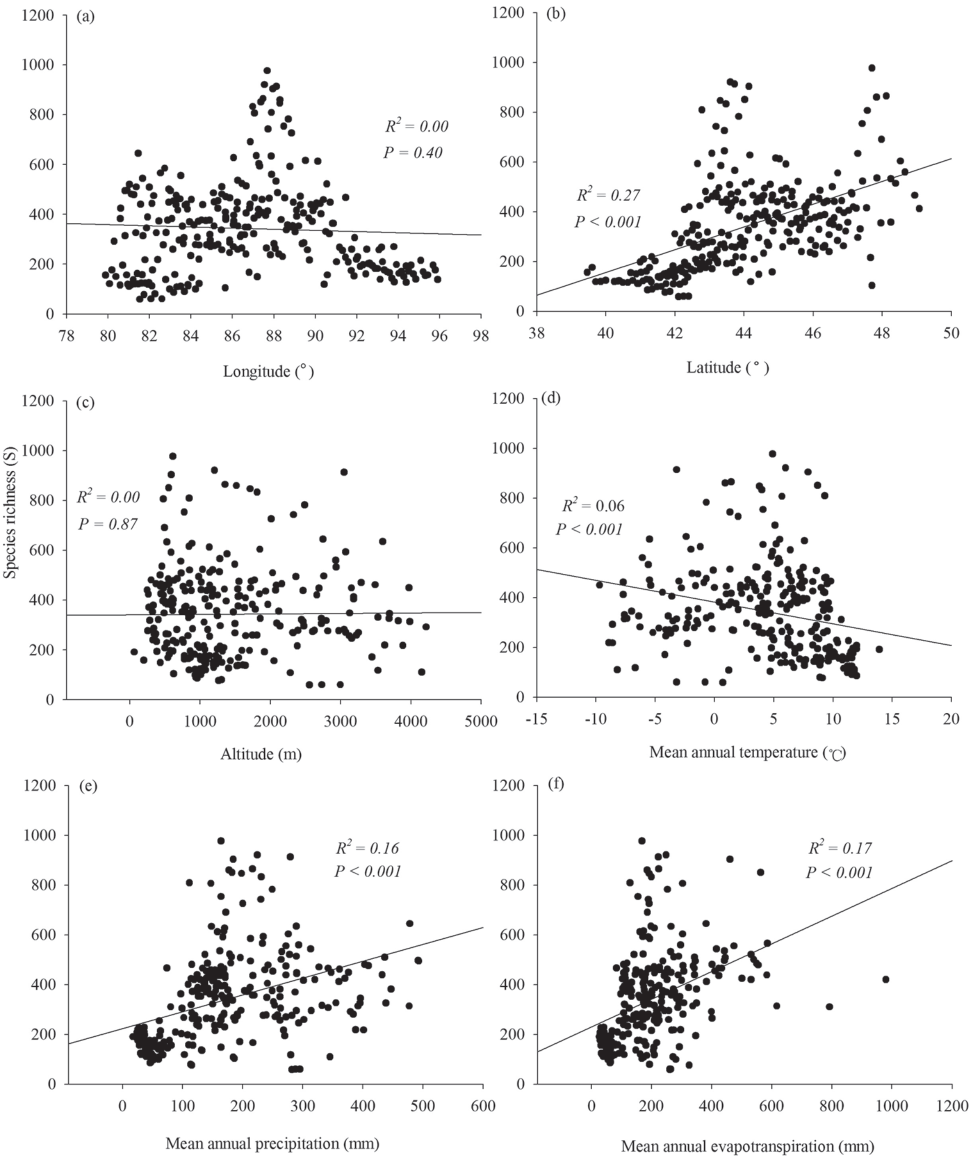

Figure 4. Relationships between species richness of distribution areas for congeneric species of *Betula halophila* and geographical and environmental factors in Xinjiang, northwest China.

Discussion

In the distribution area with *Betula* species, there are 3013 species, 693 genera and 108 families of seed plants, accounting for 86.16%, 94.54% and 93.91%, respectively, of the total seed plants in Xinjiang (Pan 1995). It can be seen that seed plant flora distributed in the distribution area of congeneric species of *B. halophila* is the majority of seed plant flora in Xinjiang. The seed plant flora is mainly composed of cosmopolitan and temperate distribution. The vegetation in the Altai area is gradually formed after the invasion of holarctic and ancient Mediterranean elements (Chen and Yang 2000). The Holarctic and the ancient Mediterranean elements are mainly cosmopolitan and temperate distribution. The main families of the seed plant flora, distributed in the distribution area of congeneric species of *B. halophila*, are Asteraceae, Poaceae, Fabaceae, Rosaceae, Ranunculaceae and Lamiaceae, all of which are cosmopolitan elements. Amongst of these families, the first three are super families of seed plants containing more than ten thousand species in the world (Wu and Wang 1983). Generally, the climatic conditions in the plant distribution areas, dominated by the cosmopolitan elements, are relatively harsh. Therefore, the seed plant flora, distributed in distribution areas of congeneric species of *B. halophila*, are mainly composed of the cosmopolitan plants, which, to a certain extent, indicate that congeneric species of *B. halophila* in Xinjiang have certain adaptability to the harsh climate (Pan 1995). In Xinjiang, there are a small number of tropical families or genera in the distribution area of congeneric species of *B. halophila*, which may be a legacy of flora under a hot climate in the early Tertiary in Xinjiang (Pan 1995). To some extent, it also reflects the uniqueness of the seed plant flora distributed in the distribution area of congeneric species of *B. halophila*, just like the uniqueness of seed plant flora in Xinjiang (Huang et al. 2011; Huang et al. 2018). Nearly 94% of the families of Xinjiang are found in the distribution area of congeneric species of *B. halophila*, which indicates the distribution areas of these species cover the diversity distribution centre of seed plants in Xinjiang. It seems to indicate that the seed plant flora, distributed in the distribution area of congeneric species of *B. halophila*, is of great importance, even equivalent to seed plant flora in Xinjiang. It can be seen that the distribution of the genus *Betula* is closely related to the distribution of seed plant diversity in Xinjiang.

According to the records of *Flora Xinjiangensis* (Editorial Committee of *Flora Xinjiangensis* 1992–2011), all species of congeneric species of *B. halophila* are mainly distributed in the Altai mountain area in Xinjiang, except for *B. tianschanica* which is mainly located in Tianshan Mountains. Amongst those four species mainly distributed in Altai, *B. pendula* and *B. microphylla* are also partly present in the western mountain areas of Junggar Basin. In Xinjiang, mountain coniferous forest is the main forest vegetation type, followed by mountain broad-leaved forest. The latter is closely related to the former. Amongst the mountain broad-leaved forests in Xinjiang, the small-leaved forest, composed of poplar and birch, is especially typical forest (Editorial Committee of Xinjiang Forestry 1989; Xinjiang Investigation Group of Chinese Academy of Sciences 1978). Some studies have shown that birch is an important component of Eurasian boreal coniferous forest (Taiga) which is cold resistant in the upper part of Altai Mountain (Chen and Yang 2000). In the Altai Mountains, birch-dominated broad-leaved forest (Chen and Yang 2000) widely occurs on many exposed hillsides, which reflects that birch forest is an important pioneer tree species in the vegetation development of the region (Chen 1999). Compared with other broad-leaved tree species, the small-leaved tree species is more resistant to severe cold and adapts to the northern mountains with a strong continental climate. The small-leaved tree species can be found almost in any areas distributed with coniferous forest. The small-leaved tree species is a light demander, not strict with soil, grows rapidly in young trees and provides strong competition to weeds. It is a typical pioneer tree species, creating a suitable environment for the regeneration and development of mountain coniferous forest of mountain forest in Xinjiang.

There are significant differences in plant richness amongst seed plant floras in the distribution area of congeneric species of *B. halophila*. The plant abundances of different life forms of these congeneric species are obviously different, while those of common life forms of these species are not significantly different. Latitude and climate have a significant influence on species richness in the distribution areas of congeneric species of *B. halophila*. It has been thought that latitude is actually a comprehensive factor. Therefore, we tend to think that climate is the key factor impacting on the distribution pattern of species diversity in the distribution areas of congeneric species of *B. halophila*, which also conforms to the view that climate plays a key role in species diversity in Xinjiang (Li et al. 2013). The species richness in the distribution areas of all tree congeneric species of *B. halophila* are influenced by climate factors, while the species richness in the distribution areas of all shrub congeneric species of *B. halophila* are not. There is no significant change of species richness with altitude, which may further indicate that there is a regional difference in patterns of species richness with altitude gradients in the distribution areas of all shrub congeneric species of *B. halophila* (Li et al. 2011).

Wild populations of *B. halophila* have not been found in the field since 2003. The biological characteristics of the species need to be further understood in order to realise its effective protection and return in the wild. Due to the lack of wild population, the understanding of its ecological characteristics is very limited. Our study will help to strengthen the understanding of this species through the analysis of species composition, floristic characteristics and species diversity of the native congeneric of this species. From the point of view of phylogeny, it is necessary to carry out an in-depth study combined with the phylogeny of this species. From the current research, the phylogenetic relationship between this species and other related species is not clear. The clear phylogenetic relationship of this species is helpful to reveal the origin of this species and provide an important traceability basis for its protection. In terms of floristic composition, the cosmopolitan families and temperate genera are dominant in the distribution area of *B. halophila*. The climate niche of the species in the wild is consistent with that of *Betula* genus. It seems that niche simulation can be used to simulate the potentially suitable distribution area of *B. halophila*, which provides an important basis for the selection of the wild return to a great extent. Therefore, the current optimum distribution area of *Betula* genus in Xinjiang is the preferred geographical region for wild return of *B. halophila* when the primary wild habitat of *B. halophila* is not suitable for its survival at local scale any more. From this study, we also found that there are obvious differences in the factors affecting the species distribution at the local scale and in different distribution areas of related species. Therefore, more factors should be considered in the follow-up and the community level investigation work should be carried out in order to provide an important ecological reference for the wild return of *B. halophila*.

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References

Bruelheide H, Jansen F, Jandt U, Bernhardt-Römermann M, Bonn A, Bowler D, Dengler J, Eichenberg D, Grescho V, Harter D, Jugelt M, Kellner S, Ludwig M, Wesche K, Lütt S (2020) Using incomplete floristic monitoring data from habitat mapping programmes to detect species trends. Diversity & Distributions 26(7): 782–794. https://doi.org/10.1111/ddi.13058

Chen ZD (1999) Origin and Dispersion of Betulacance. The Geography of Spermatohytic Families and Genera, 236–258.

Chen WL, Yang CY (2000) A floristic study on the seed plant in Mts. Altay of China. Yunnan Zhi Wu Yan Jiu 22: 371–378. https://doi.org/10.3969/j.issn.2095-0845.2000.04.001

Chen LZ, Sun H, Guo K (2014) Flora and vegetational geography of China. Science Press, Beijing.

Editorial Committee of Flora Xinjiangensis (2014) Primer of Flora Xinjiangensis. Xinjiang Science, Technology and Sanitation Press, Urumqi.

Editorial Committee of Flora Xinjiangensis (1992) Flora Xinjiangensis (Vol. 1). Xinjiang Science & Technology & Hygiene Publishing House, Urumqi.

Editorial Committee of Flora Xinjiangensis (1992–2011) Flora Xinjiangensis (Vol. 1–6). Xinjiang Science & Technology & Hygiene Publishing House, Urumqi.

Editorial Committee of Xinjiang Forestry (1989) Xinjiang Forestry. Xinjiang People's Publishing House/China Forestry Publishing House, Urumqi/Beijing.

ESRI (2008) ArcGIS 9.3. ESRI Press, New York.

Li H, Deng JY, Zhang H, Miao C, Bao YL, Cheng P (2009) Physiological characteristics of *Betula halophila* seedlings under NaCl stress. Xibei Zhiwu Xuebao 29(11): 2281–2287.

Huang JH (2010) Study on features and distribution patterns of Chinese endemic seed plant species. PhD Dissertation, Institute of Botany, Chinese Academy of Sciences, Beijing.

Huang JH, Chen JH, Ying JS, Ma KP (2011) Features and distribution patterns of Chinese endemic seed plant species. Journal of Systematics and Evolution 49(2): 81–94. https://doi.org/10.1111/j.1759-6831.2011.00119.x

Huang JH, Liu CR, Guo ZJ, Ma KP, Zang RG, Ding Y, Lu XH, Wang JP, Yu RY (2018) Seed plant features, distribution patterns, diversity hotspots, and conservation gaps in Xinjiang, China. Nature Conservation 27: 1–15. https://doi.org/10.3897/natureconservation.27.23728

Li LP, Mohammat A, Guo ZD, Hai Y, Tang ZY (2011) Study on plant species composition and richness of the mountain coniferous forests in Xinjiang. Ganhanqu Yanjiu 28(1): 40–46. https://doi.org/10.3724/SP.J.1148.2011.00040

Li LP, Wang ZH, Zerbe S, Abdusalih N, Tang ZY, Ma M, Yin LK, Mohammat A, Han WX, Fang JY (2013) Species richness patterns and water-energy dynamics in the drylands of northwest China. PLoS ONE 8(6): e66450. https://doi.org/10.1371/journal.pone.0066450

Lu Q, Wang J, Chu J (2012) Desert Plants in China. China Forestry Publishing House, Beijing.

Maestre FT, Delgado-Baquerizo M, Jeffries TC, Eldridge DJ, Ochoa V, Gozalo B, Quero JL, García-Gómez M, Gallardo A, Ulrich W, Bowker MA, Arredondo T, Barraza-Zepeda C, Bran D, Florentino A, Gaitán J, Gutiérrez JR, Huber-Sannwald E, Jankju M, Mau RL, Miriti M, Naseri K, Ospina A, Stavi I, Wang D, Woods NN, Yuan X, Zaady E, Singh BK (2015) Increasing aridity reduces soil microbial diversity and abundance in global drylands. Proceedings of the National Academy of Sciences of the United States of America 112(51): 15684–15689. https://doi.org/10.1073/pnas.1516684112

Magurran AE (1988) Ecological Diversity and its Measurement. New Jersey Princeton University Press, Princeton. https://doi.org/10.1007/978-94-015-7358-0

Mansuer S (2012) Xinjiang Geography. Beijing Normal University Press. Beijing.

Mienna IM, Speed JDM, Mika B, Thornhill AH, Mishler BD, Martin MD (2019) Differential patterns of floristic phylogenetic diversity across a post-glacial landscape. Journal of Biogeography. https://doi.org/10.1111/jbi.13789

Pan XL (1995) A study on the flora of seed plants in Xinjiang. PhD Dissertation, Sun Yat-sen University, Guangzhou.

Phillips RD, Brown AP, Dixon KW, Hopper SD (2011) Orchid biogeography and factors associated with rarity in a biodiversity hotspot, the Southwest Australian Floristic Region. Journal of Biogeography 38(3): 487–501. https://doi.org/10.1111/j.1365-2699.2010.02413.x

Qian H (2001) Floristic analysis of vascular plant genera of North America north of Mexico: Spatial patterning of phytogeography. Journal of Biogeography 28(4): 525–534. https://doi.org/10.1046/j.1365-2699.2001.00560.x

R Core Team (2019) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna.

Reynolds JF, Smith DMS, Lambin EF, Turner B, Mortimore M, Batterbury SP, Downing TE, Dowlatabadi H, Fernández RJ, Herrick JE, Huber-Sannwald E, Jiang H, Leemans R, Lynam T, Maestre FT, Ayarza M, Walker B (2007) Global desertification: Building a science for dryland development. Science 316(5826): 847–851. https://doi.org/10.1126/science.1131634

Scheffe H (1959) The Analysis of Variance. Wiley. New York.

Shao FJ, Zhang LS, Wilson IW, Qiu DY (2018) Transcriptomic analysis of *Betula halophila* in response to salt stress. International Journal of Molecular Sciences 19(11): 1–3412. https://doi.org/10.3390/ijms19113412

State Forestry Administration (2011) National planning of rescue and conservation project for the wild plants with the extremely small populations (2011–2015), Beijing.

State Forestry Administration, Ministry of Agriculture (1999) List of National Key Protected Wild Plants (I), Beijing.

Takhtajan AL (1969) Flowering Plants: origin and dispersal. Oliver & Boyd, Edinburgh.

Tian CY, Zhou HF, Liu GQ (2000) The proposal on control of soil salinizing and agricultural sustaining development in 21th century in Xinjiang. Arid Land Geography 23: 177–181.

Wang HS (1992) Floristic Plant Geography. Science Press, Beijing.

Wang J (2003) Discovery of salt resistant birch in Xinjiang. Plants 6: 12–13. https://doi.org/10.1049/em:20030312

Wang N, Mcallister HA, Bartlett PR, Buggs RJA (2016) Molecular phylogeny and genome size evolution of the genus Betula (Betulaceae). Annals of Botany 117(6): 1023–1035. https://doi.org/10.1093/aob/mcw048

Wei YJ, Xu M (2005) Study on soil salinization and countermeasures of its prevention in Xinjiang. Earth and Environment 33: 593–597.

Weigelt P, König C, Kreft H (2020) GIFT – A Global Inventory of Floras and Traits for macroecology and biogeography. Journal of Biogeography 47(1): 16–43. https://doi.org/10.1111/jbi.13623

Wu JH (2004) Phytogeography. Higher Education Press. Beijing.

Wu ZY, Wang HS (1983) China's Physical Geography: Phytogeography (Vol. 1). Science Press, Beijing.

Wu ZY, Zhou ZK, Sun H, Li DZ, Peng H (2006) Areal-Types of Seed Plants and Their Origin and Differentiation. Yunnan Science and Technology Press, Kunming.

Wu ZY, Sun H, Zhou ZK, Li DZ, Peng H (2011) Floristics of Seed Plants from China. Science Press, Beijing.

Wu ZY, Raven PH, Hong DY (1994–2012) Flora of China. Science Press of Missouri Botanical Garden Press, Beijing or St. Louis. http://www.iplant.cn/foc/

Xinjiang Altay Forestry Institute, Xinjiang Forestry Academy (2010) New Species – Research and breeding application of *Betula microphylla* in Marshes, Urumqi.

Xinjiang Investigation Group of Chinese Academy of Sciences (1978) The Vegetation of Xinjiang and its Application. Science Press, Beijing.

Yang CY (2012) Sylva Xinjiangensis. China Forestry Publishing House, Beijing.

Yang CY, Wang J, Li WH (2006) New taxa of *Betula* L. from Xinjiang, China. Bulletin of Botanical Research 26: 648–655.

Yao YF, Wang X, Xie G, Li JF, Wang YF, Li CS (2015) Holocene vegetation succession and climate-environment change in Xinjiang Region. Chinese Science Bulletin 28: 183–187.

Yin L (2006) Rare endangered Endemic Higher Plants in Xinjiang of China. Xinjiang Science & Technology Publishing House, Urumqi.

You XL, Cui XH, Zhu Y, He XL, Wang J (2017) ISSR analysis of *Betula microphylla* Bunge and new varieties in Xinjiang. Shanghai Jiaotong Daxue Xuebao. Nongye Kexueban 35: 89–94.

Zhang HB, Zeng YL, Lan HY, Zhang FC (2009) Physiological responses of *Betula halophila* (Betulaceae) to salt stress. Plant Diversity and Resources 31: 260–264. https://doi.org/10.3724/SP.J.1143.2009.08221

Zhang YB, Wang YZ, Zhang MG, Ma KP (2014) Climate change threats to protected plants of China: An evaluation based on species distribution modeling. Chinese Science Bulletin 59(34): 4652–4659. https://doi.org/10.1007/s11434-014-0642-6

Zhang TW, Yuan YJ, Wei WT, Zhang RB, Yu SL, Chen F, Shang HM, Li Q (2015) Development of the tree-ring width chronology of *Betula tianschanica* for the Kaiduhe River watershed and its climate response. Acta Ecologica Sinica 35: 3034–3042. https://doi.org/10.5846/stxb201306111650

Zhang HF, Dilixiati B, Zhu YL, Gao YQ, Ding CF (2017) Biomass allocation patterns and allometric models of *Betula pendula* and *Populus tremula* in Xinjiang. Journal of Northwest Forestry University 32: 33–38.

Supplementary material 1

Figure S1. The correlation between environmental variables in distribution areas of five congeneric species with *Betula halophila*

Authors: Jihong Huang, Zhongjun Guo, Suying Tang, Wei Ren, Guangming Chu, Liping Wang, Ling Zhao, Ruoyun Yu, Yue Xu, Yi Ding, Runguo Zang

Data type: Generating graph based on measurement and occurrence data.

Explanation note: The correlation between longitude (Long), latitude (Lati), altitude (Alti), mean annual temperature (Tem), mean annual precipitation (Pre) and evapotranspiration (Et) in distribution areas of five congeneric species with *Betula halophila*. (a) TSH: *B. tianschanica*; (b) XYH: *B. microphylla*; (c) CZH: *B. pendula*; (d) YYH: *B. rotundifolia*; (e) DSH: *B. humilis*).

Link: https://doi.org/10.3897/natureconservation.42.54735.suppl1

Supplementary material 2

Figure S2. The distribution frequency of *Betula* species varies with the environment gradients

Authors: Jihong Huang1, Zhongjun Guo, Suying Tang, Wei Ren, Guangming Chu, Liping Wang, Ling Zhao, Ruoyun Yu, Yue Xu, Yi Ding, Runguo Zang

Data type: Generating graph based on measurement and occurrence data.

Explanation note: The distribution frequency of *Betula* species varies with the environment gradients and the environmental variable values of *Betula halophila* were once distributed in the field.

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RESEARCH ARTICLE

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Mapping Rocky Mountain ridged mussel beds with preliminary identification of overlapping Eurasian watermilfoil within the Canadian range

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Abstract

The Rocky Mountain ridged mussel (*Gonidea angulata*) is a bivalve species whose Canadian range is limited to the Okanagan Valley, British Columbia. In 2019, conflicts between habitat protection for the mussel and potential habitat alteration to control the invasive Eurasian watermilfoil (*Myriophyllum spicatum*) (milfoil), led to a decision to maintain the status of the mussels as Special Concern under Canada's Species at Risk Act (SARA) rather than classify it as Endangered. Milfoil control can cause direct mortality and/or burial of the mussels, but there had been no systematic study of the impacts of milfoil control on mussel beds. The purpose of this study was to address knowledge gaps by delineating known mussel beds and potential overlap with milfoil to provide information for management decisions that balance the needs of native species protection and invasive species control. Rocky Mountain ridged mussels in three reference locations were enumerated using snorkel surveys. The presence and distribution of milfoil was documented in relation to five sites within these three locations. Milfoil was encroaching on one site, causing some changes to the substrate. At other sites, the differences in the depth and distribution of the mussel and the milfoil could allow milfoil control without damaging the mussel beds. It is recommended that, before milfoil removal near known mussel beds be undertaken, a detailed site evaluation be conducted to determine potential impacts. This study suggests presumed impediments to co-managing the mussels and controlling an invasive species should not preclude classifying the mussels as Endangered and affording protections under SARA.

Keywords
Eurasian watermilfoil, western ridged mussel, conservation management, habitat, invasive species, Okanagan Valley

Introduction

Freshwater mussels (unionids) are one of the most endangered groups of animals in North America (Williams et al. 1993; DFO 2017). The Rocky Mountain ridged mussel (*Gonidea angulata*) is the only extant member of the genus Gonidea (Blevins et al. 2016). It is a bivalve mussel species whose Canadian range is limited to the Okanagan Valley, British Columbia (BC). This area marks the northernmost limit of its patchy distribution which extends south to Napa County, California and from western Oregon and Washington, east to the Snake River Basin in Idaho and south to the Humboldt basin in Nevada (Blevins et al. 2016). The distribution once included southern California, but it is now believed to be extirpated from this area (Howard 2010; Howard et al. 2015). Some Washington, Oregon and Idaho populations are believed to be in decline (Blevins et al. 2016). Comparison of the extent of occurrence (EOO) and area of watershed occupancy of these United States populations from occurrence prior to 1990 and from 1990–2015 indicates a decline in EOO by 28% and watershed area by 43% (Blevins et al. 2016).

The Rocky Mountain ridged mussel has specific habitat preferences. It can be found in both lotic and lentic streams, rivers and lakes (COSEWIC 2003; Stanton et al. 2012; Snook et al. 2019). It is a burrowing species that is typically found partially (most commonly adults) to fully (juveniles) burrowed in substrates which vary from gravel to firm mud including sand, silt or clay (COSEWIC 2003; Stanton et al. 2012). Rocky Mountain ridged mussels are normally found burrowed to at least half their length in fine substrate in water at depths less than 3 m (COSEWIC 2003). However, individuals have been reported in depths of approximately 7.5 m in Vaseux Lake (Stanton et al. 2012).

In 2003, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) assessed this species as Special Concern largely due to its limited distribution (COSEWIC 2003). Damming of the Kootenay, Columbia and Okanagan rivers isolated the Canadian population from those further south. The Canadian population was believed to be in decline as early as 2003 (COSEWIC 2003). In 2010, COSEWIC re-assessed the species as Endangered (COSEWIC 2010). This was due to the potential threat of invasive mussel species (e.g. zebra mussel *Dreissena polymorpha* and quagga mussel *D. rostriformis bugensis*), dredging and channelling of the Okanagan River, foreshore and riparian development and other introduced invasive species, especially Eurasian watermilfoil (*Myriophyllum spicatum*). Eurasian watermilfoil is an aggressive and pervasive plant which can readily displace native vegetation. It can be found in water 1 to 10 m deep (Aiken et al. 1979); in Okanagan Lake, some plants may grow up to 8 m tall (Dunbar 2009). Both the presence of Eurasian watermilfoil and the methods used to remove it can negatively affect the Rocky Mountain ridged mussel (COSEWIC 2010; DFO 2010; Mageroy et al. 2017).

Eurasian watermilfoil can inhibit water flow (Chambers et al. 1999) and increase siltation (Snook 2015). With increased siltation, mussels can become buried or their habitat can become unsuitable. Buried mussels cannot survive (Krueger et al. 2007). Two methods are currently used to control Eurasian watermilfoil in Okanagan Lake, rototilling in the late autumn/winter and harvesting in the summer (Dunbar 2009). Rototilling removes the roots of the plants from the substrate. The machinery can be operated in water up to 4.5 m deep. As re-growth occurs in rototilled areas, this process must be repeated for eradication. Harvesting typically occurs in areas where rototilling is not possible or where re-growth due to rototilling reaches unacceptable levels (Dunbar 2009). Plants are cut at a depth of up to 2 m below the surface when harvested. Although harvesting is reported as a faster control method, it requires the removal of the plant material and can interfere with recreational activities (Dunbar 2009). It has been shown experimentally that rototilling of Rocky Mountain ridged mussel substrate causes mortality both directly by crushing animals and indirectly by burying (Mageroy et al. 2017). Harvesting, as it is currently done, should have no direct effect on Rocky Mountain ridged mussels.

A 2019 decision by the Canadian Federal Fisheries Minister maintained the species as Special Concern rather than reclassify it as Endangered under the Species at Risk Act (SARA). The potential socio-economic impacts of limiting Eurasian watermilfoil control, which could occur if classified as Endangered under the SARA, was cited as one of the reasons for this decision (http://gazette.gc.ca/rp-pr/p2/2019/2019-08-21/html/sor-dors287-eng.html). The article also stated that additional science has become available since the COSEWIC assessment in 2010, which would challenge a classification of Endangered. The Okanagan Basin Water Board felt there was not enough evidence that the mussel could be harmed by rototilling and noted that studies on the socio-economic impacts of terminating Eurasian watermilfoil control had not been completed (Thom 2019). A local member of Parliament stated that there were "too many unanswered questions and lack of recent data to risk the unintended impact of the reclassification of the Rocky Mountain ridged mussel" (Stephen Fuhr, MP Kelowna-Lake Country in Thom 2019).

Okanagan Lake provides many benefits to local tourism and the real estate market. The goal of the Okanagan Eurasian watermilfoil control programme is to minimise environmental impacts while enhancing public enjoyment of Okanagan lakes with a cost-effective programme. No scientific evidence could be found which documented the outcomes of this programme. There has been no systematic study of the impacts of Eurasian watermilfoil control on mussel beds or on the overlap of critical mussel habitat and Eurasian watermilfoil distribution. Mageroy et al. (2017) has demonstrated experimentally that mussels can be negatively impacted (e.g. crushed and/or buried) by rototilling. Evidence-based advice regarding the interactions between this mussel species and Eurasian watermilfoil is needed to support management decisions that balance the needs of native species protection, invasive species control and economic development.

The purpose of this study was twofold. First, to map the extent of Rocky Mountain ridged mussel beds and calculate the mussel bed density at three reference locations in Okanagan Lake, British Columbia in 2019. Second, to relate mussel bed locations to a qualitative description of the presence/absence of invasive Eurasian watermilfoil in and around these reference locations.

Methods

Index sites

Rocky Mountain ridged mussel surveys have been conducted annually by Fisheries and Oceans Canada (DFO) since 2011. DFO selected three reference locations for ongoing population monitoring and included six index sites in Okanagan Lake within the locations. We used five of these index sites in this survey namely Dog Beach 1 and 2, Kinsmen Beach 1 and 2 and 3 Mile (Figure 1 and Table 1). This survey took place in August 2019.

Mussel enumeration

The enumeration methods used in this study are the same as those used since 2017 in the DFO surveys. DFO's methods have not yet been published, but one author (JW) has participated in these surveys for four years and replicated DFO's methods for this study. A surveyor's tape was placed as the baseline at or near the shore of each site, parallel to the shoreline. A leadline transect marker was pulled from the shore to depth, perpendicular to the baseline every 3 m beginning at the 0 m mark. The length of each transect varied depending on water depth and habitat suitability. Transects ended when the substrate became unsuitable for Rocky Mountain ridged mussels (e.g. excess mud) or exceeded 1.5 m depth. To confirm that transects were long enough to span the entire bed, each transect was snorkelled at least 3 m past the end of the transect to ensure no mussels were present. Transects were snorkelled from the maximum depth to as close to the baseline as possible remaining covered in water. Each transect was

Table 1. Geolocation of five index sites at three reference sites used to map and enumerate Rocky Mountain ridged mussels (*Gonidea angulata*) in Okanagan Lake, British Columbia in August 2019.

Reference Site	Index sites	Latitude / Longitude
Peach Orchard Dog Park, Summerland	Dog Beach 1	49.606999; -119.64972
	Dog Beach 2	49.607729; -119.65067
Kinsmen Beach, Summerland	Kinsmen Beach 1	49.598930; -119.65078
	Kinsmen Beach 2	49.59941; -119.65096
3 Mile Park, Penticton	3 Mile	49.538110; -119.57644

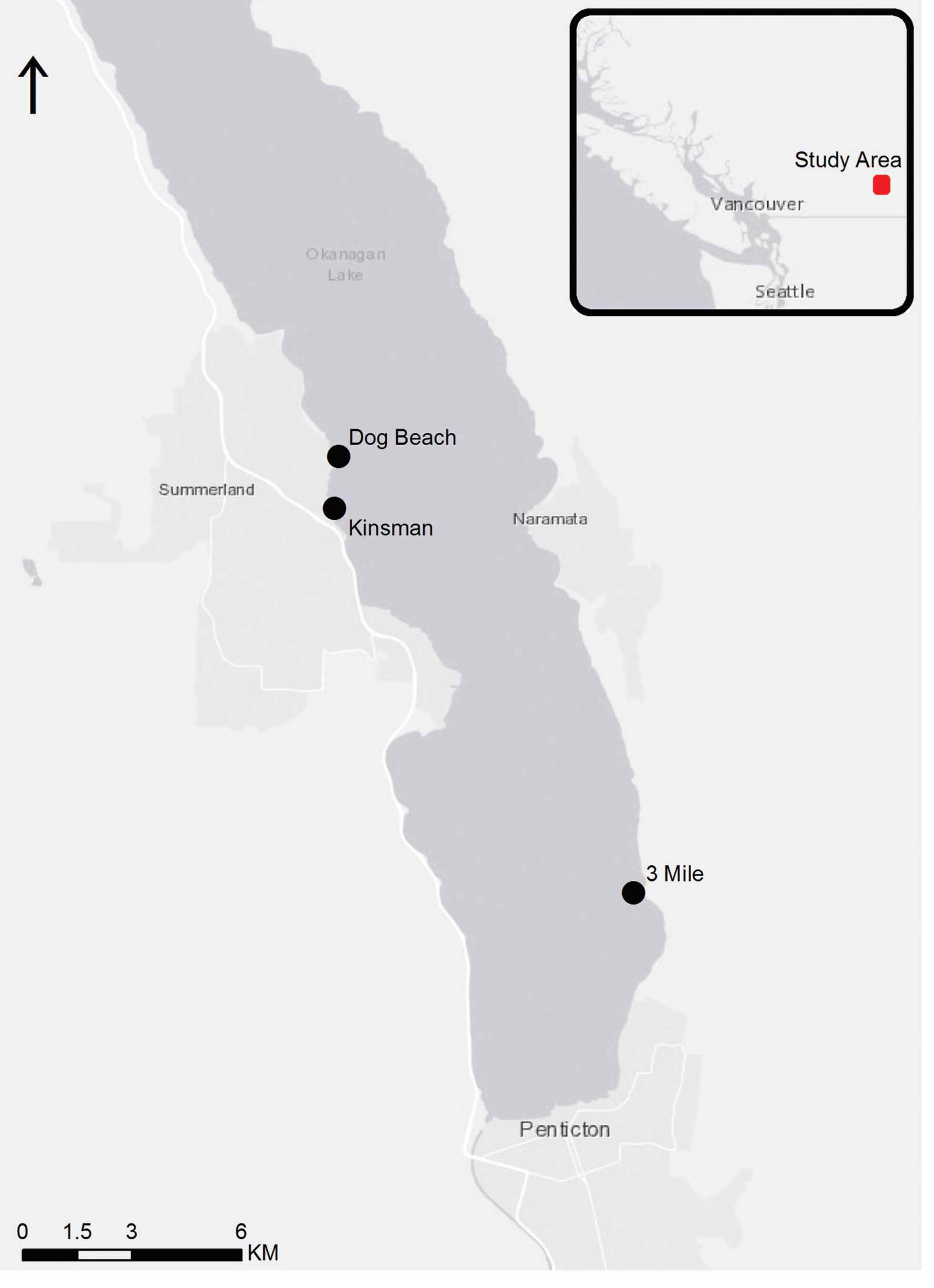

Figure 1. Location of study area and reference sites (Peach Orchard Dog Park (Dog Beach) and Kinsmen Beach in Summerland and 3 Mile near Penticton) used for Rocky Mountain ridged mussel (*Gonidea angulata*) surveys in Okanagan Lake, British Columbia in August 2019.

snorkelled holding a metre stick beside the transect line to delineate the counting area. Live mussels were counted along the transect to a width of 1 m on the right side of the transect line.

The Rocky Mountain ridged mussel burrows in the sediment with its siphon protruding. Only mussels which protruded from the substrate were counted; the substrate was not disturbed to look for mussels. No mussels were handled during any survey activities. Silt or vegetation was gently brushed aside to aid in counting. As a result of these non-invasive methods, very small mussels or those covered in heavy silt, mud, sand or gravel may not have been counted and, therefore, the abundance values are likely to be an underestimate.

Mapping

Rocky Mountain ridged mussels

For each transect, the first and last mussels counted were marked with a numbered float tied to a weight. After each index site was enumerated, the location of each marker was georeferenced using a Trimble GeoXH GPS receiver (+/- 30 cm positional accuracy) and water depth was recorded to the nearest cm. By marking the first and last mussel on the transect, an accurate delineation of the mussel bed could be mapped.

Eurasian watermilfoil (Myriophyllum spicatum)

Visual examination for the presence or absence of Eurasian watermilfoil was undertaken by snorkel surveys at all five index sites. When beds of Eurasian watermilfoil were found, the perimeter was georeferenced using the GPS to delineate the overlap with the mussel bed. When individual plants or small clusters were found, a presence/absence was recorded without quantification. Where it was not possible to map the entire extent of the plant bed because of water depth and safety concerns due to risk of entangled in Eurasian watermilfoil stems, only the perimeter closest to the mussel bed was mapped.

Analysis

Mussel bed mapping

To determine the total mussel bed area, positions of the start and end points of each baseline were determined from the GPS data. With the aid of high resolution satellite imagery, the baseline along the shore was digitised and transect starts were inferred from the length and interval spacing of 1 m. Individual transects were derived from the starting point on the baseline, the GPS-derived positions of the floats and the recorded transect length. Transect end points were then used to digitise the outer boundary of the mussel bed. Bed area was calculated from the area of the polygon formed from digitising the position of the first float on each transect, the outer boundary and the first and last transects (buffered on one side).

Mussel bed density was calculated using the R package "survey", version 3.36 (Lumley 2004) which is a software package with functions for analysis of complex survey data. Densities were estimated separately using the function 'svyratio' (which does ratio estimation and estimates of totals, based or complex survey samples) for each index site and used to determine overall abundance from the mussel bed area (determined as above).

Results

Mussel bed density

Visibility was excellent at all index sites with the exception of the first three transects (0, 3, 6 m at the baseline) of Kinsmen Beach 1. The substrate at these three transects was too muddy and had too much Eurasian watermilfoil to accurately count the mussels. It is normally possible in areas of muddy substrate to wait for the sediment to settle before snorkelling the transect. However, at Kinsmen Beach 1, the Eurasian watermilfoil was too dense to allow visualisation of the substrate in these three transects. Half of the Kinsmen Beach 1 site has a muddy substrate, but mussel visualisation was possible after disturbed mud settled. Rocky Mountain ridged mussel bed areas and densities are presented in Table 2 and bed areas have been delineated in Figure 2.

Mussels were found in water ranging in depth from 38.1 cm to 175.3 cm (Table 2). The largest mussel bed area was delineated at the Dog Beach 1 index site (1,766.78 m^2) with the second smallest mussel bed density of 0.22 mussels/m^2. The greatest mussel bed density was found at 3 Mile (1.23 mussels/m^2). Kinsmen Beach 1 and 2 sites had similar mussel bed areas (738.84 and 619.92 m^2, respectively) as well as densities (0.86 and 0.79 mussels/m^2, respectively). Total population estimates (Table 2) were estimated to range from a low of 128.88 at Dog Beach 2 to a high of 1,668.28 at 3 Mile.

Table 2. Results of snorkel surveys to map and enumerate Rocky Mountain ridged mussels (*Gonidea angulata*) at five index sites in Okanagan Lake, British Columbia in August 2019 (s.e.= standard error).

Index site	# transects surveyed	Range of number of mussels/ transect	Depth (cm) of mussel found		Bed area (m^2)	Bed density (mussels/ m^2)	Population Estimate (s.e.)
			Nearest to shore	Furthest from shore			
Dog Beach 1	21	1–12	53.3	175.3	1,766.78	0.22	383.00 (44.35)
Dog Beach 2	18	0–8	55.9	127.0	971.44	0.13	128.88 (25.43)
Kinsmen Beach 1	16	0–37	38.1	134.6	738.84	0.86	637.90 (134.08)
Kinsmen Beach 2	12	0–27	40.6	124.5	619.92	0.79	490.80 (107.41)
3 Mile	25	2–35	43.2	175.3	1,355.15	1.23	1,668.28 (261.65)

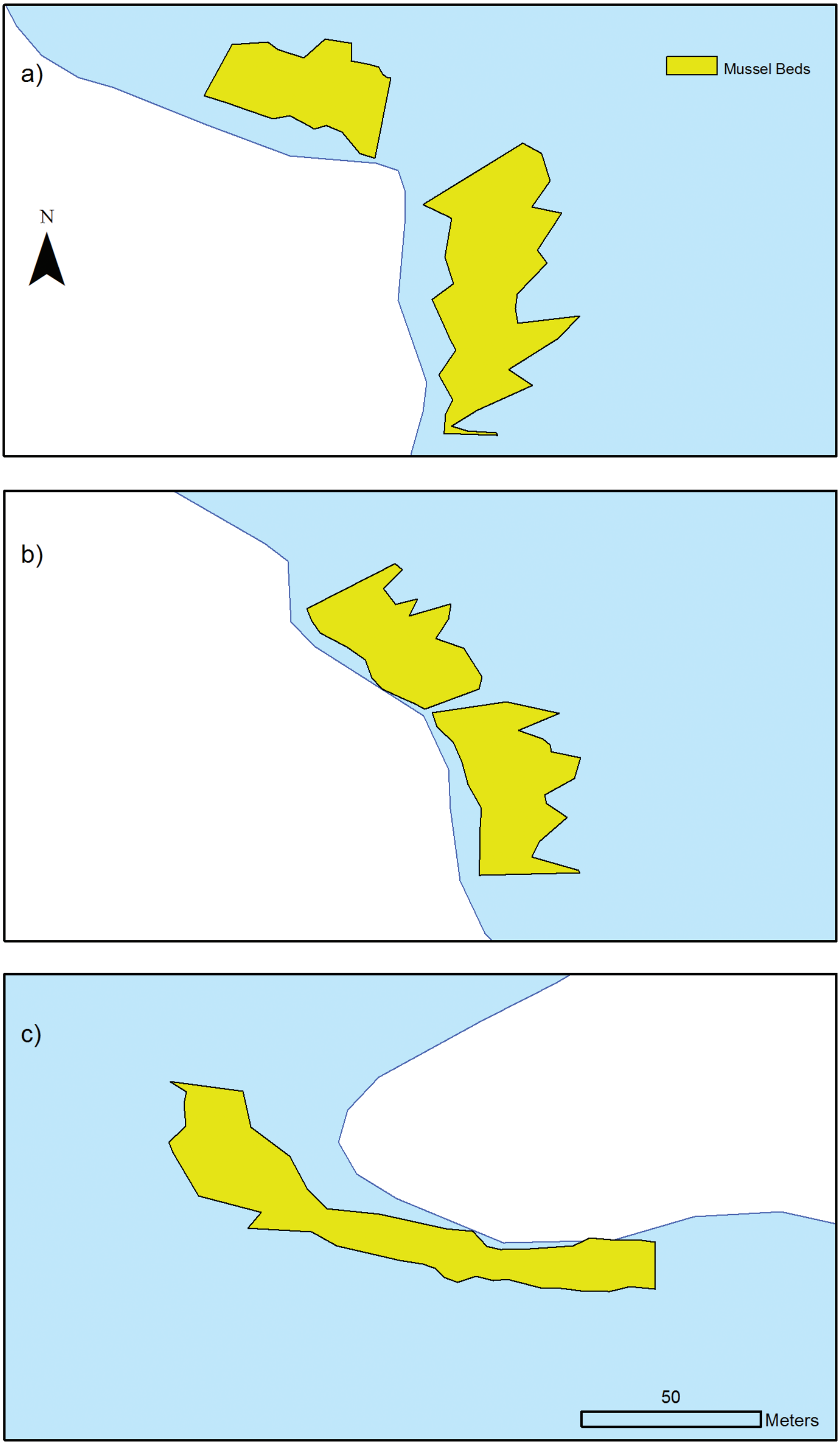

Figure 2. Delineated Rocky Mountain ridged mussel beds at index sites: a) Dog Beach 1 (southern bed) and 2 (northern bed) b) Kinsmen Beach 1 (southern bed) and 2 (northern bed) c) 3 Mile.

Occurrence of Eurasian watermilfoil

Eurasian watermilfoil could be seen within the boundaries of the mussel beds at all three reference sites and four of the five index sites (Table 3). Individual plants occurred sporadically at three index sites (Dog Beach 1, Kinsmen Beach 1 and 2). Two Eurasian watermilfoil beds were seen near or within part of the mussel beds at 3 Mile and Kinsmen Beach 1. Both plant beds were extensive in that the substrate was covered by mats of plants. At 3 Mile, the plants were not seen on the mussel bed, but remained 2–5 m away in deep water. The plant bed was located where water depth increased beyond the depth where mussels are typically found. At Kinsmen Beach 1, the Eurasian watermilfoil had encroached on to the southern end of the mussel bed (first 9 m of the baseline) and the substrate had become muddy (Figure 3). This represented only the

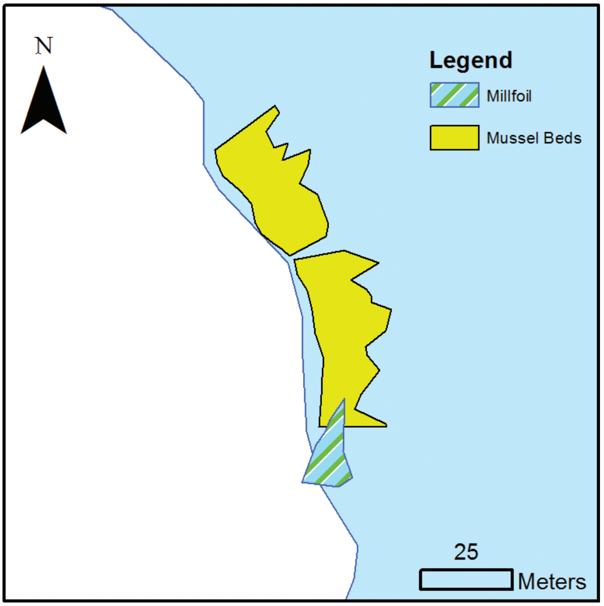

Figure 3. Location of Rocky Mountain ridged mussel (*Gonidea angulata*) beds surveyed at Kinsmen Beach 1 (southern bed) and 2 (northern bed) index sites with encroaching Eurasian watermilfoil (hatched green/blue) in Okanagan Lake, British Columbia in August 2019.

Table 3. Qualitative description of the co-occurrence of Eurasian watermilfoil (*Myriophyllum spicatum*) in Rocky Mountain ridged mussel (*Gonidea angulata*) index sites in Okanagan Lake, British Columbia in August 2019.

Index site	Description
Dog Beach 1	Individual plants found sporadically in the bed.
Dog Beach 2	No plants present.
Kinsmen Beach 1	Individual plants found throughout the mussel bed. Extensive dense plant bed overlapping a portion of the mussel bed.
Kinsmen Beach 2	Individual plants found sporadically in the mussel bed.
3 Mile	Extensive, dense plant bed within 2 m of the mussel bed

leading edge of an extensive Eurasian watermilfoil bed. The Dog Beach 2 index site is a segregated dog swimming area where the substrate is regularly disturbed. No plants were seen at this index site.

Discussion

The proposed management regime in Okanagan Lake would dictate a 100-metre buffer zone between the Rocky Mountain ridged mussel beds and Eurasian watermilfoil control activities (Lirette 2019). The size of this buffer zone created concerns that restrictions on the spatial extent of Eurasian watermilfoil control due to the presence of Rocky Mountain ridged mussels would impede control of the invasive plant, threatening economic and recreational use and safety on the Lake (Duncan 2019). This conflict was highlighted to the federal Minister of Fisheries and Oceans when considering the change in the conservation status of the species. Our results show that the perceived conflict between Rocky Mountain ridged mussel management and Eurasian watermilfoil control did not occur at all index sites and that standard methods of Eurasian watermilfoil control could be conducted within distances of less than 100 m from a mussel bed. Separation of mussel beds and the Eurasian watermilfoil, either due to a small number of plants or plants existing deeper than the preferred mussel habitat, suggests that Eurasian watermilfoil control activities could be conducted near some mussel beds and that early intervention may prevent more extensive overlap of these two species.

Rototilling can remove the root of the plant in water up to 4.5 m deep, whereas cutting can remove it in water as shallow as 2 m. The surveyed mussel beds were in water as shallow as 38.1 cm and as deep as 175.3 cm. The less invasive cutting method would be suitable throughout the depth of these beds.

Our findings demonstrate that the five index sites varied in mussel bed density (0.13–1.23 mussels/m^2) and corresponding population estimates (128.88 to 1,668.28), as did the extent of incursion of milfoil into the beds. These findings support recommendations for site-specific management actions, as opposed to generic, Lake-wide recommendations to balance the need for invasive species control and endangered species management.

For example, removal of the invasive plant at 3 Mile could occur using rototilling without disturbing the existing mussel bed. If plants are not removed from the Kinsmen Beach 1 index site, they will likely continue to take over the mussel bed. We subjectively noted an increase in the extent of mud and siltation at this site compared to surveys in previous years. Cutting of plants overlapping the mussel bed would be recommended for this area. The rest of the plant bed, further away from the mussel bed, could be removed by rototilling without impacting the mussel bed. This recommendation is for this site only and should not be applied to other locations without investigation.

Managing one ecosystem risk, such as an invasive species, can have unintended negative impacts on other goals, such as endemic species conservation. Ecosystem managers have three options when faced with conflicting invasive species and conservation goals; (1) manage the impacts of the invader and accept the collateral damage;

(2) abandon management of the invader and accept its impacts or (3) seek a compromise strategy that allows both goals to be attained (Buckley and Han 2014). The value of Okanagan Lake to local tourism, recreation and real estate preclude option 2. Societal expectations and legislative obligations to protect rare and endangered species preclude option 1. A compromise approach is needed.

There is community, regulatory and scientific information to support the belief that the presence of Eurasian watermilfoil in Okanagan Lake is deleterious to the health of the Rocky Mountain ridged mussel, as well as to the economic and recreational use of the Lake by people. Lampert et al. (2014) concluded that, in general, optimal management of multiple invasive species and conservation goals simultaneously require less-intensive investment over extended periods to fit with the timescale of natural processes. The tremendous data gaps complicate selection of management actions that meet this species evolved needs, especially in the face of concurrent threats from invasive species, climate change and foreshore habitat damage. Conservation success is increasingly seen not just as removal of the threat of extinction, but rather the development of self-sustaining, healthy, resilient species (Stephen and Wade 2018).

Conclusion

We suggest that Eurasian watermilfoil control is one strategy to protect critical resources needed for Rocky Mountain ridged mussel resilience and, therefore, Eurasian watermilfoil control is a shared conservation and invasive species goal. Expanding this work to other areas can further delineate the nature of the overlaps of mussels and Eurasian watermilfoil. Such site-specific information may allow for an integrated strategy to address conservation goals without compromising invasive species management goals by setting evidence-based buffer zones and/or tailoring Eurasian watermilfoil control activities to the nature of site-specific mussel-watermilfoil overlaps.

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References

Aiken SA, Newroth PR, Wile I (1979) The biology of Canadian weeds. 34. *Myriophyllum spicatum* L. Canadian Journal of Plant Science 59(1): 201–215. https://doi.org/10.4141/cjps79-028

Blevins E, Jepsen S, Brim Box J, Nez D (2016) *Gonidea angulata*. The IUCN Red List of Threatened Species 2016: e.T173073A114146359. https://doi.org/10.2305/IUCN.UK.2016-3.RLTS.T173073A62905403.en

Buckley YM, Han Y (2014) Managing the side effects of invasion control. Science 344(6187): 975–976. https://doi.org/10.1126/science.1254662

Chambers PA, DeWreede RE, Irlandi EA, Vandermeulen H (1999) Management issues in aquatic macrophyte ecology: A Canadian perspective. Canadian Journal of Botany 77(4): 471–487. https://doi.org/10.1139/b99-092

COSEWIC (2003) COSEWIC Assessment and Status Report on the Rocky Mountain Ridged Mussel *Gonidea angulata* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, 29 pp. www.sararegistry.gc.ca/status/status_e.cfm

COSEWIC (2010) COSEWIC Assessment and Status Report on the Rocky Mountain Ridged Mussel *Gonidea angulata* in Canada. Committee on the Status of Endangered Wildlife in Canada. Ottawa, 56pp. www.sararegistry.gc.ca/status/status_e.cfm

DFO (2010) Management Plan for the Rocky Mountain Ridged Mussel (*Gonidea angulata*) in Canada [Final]. Species at Risk Act Management Plan Series. Fisheries and Oceans Canada, Vancouver, 52 pp.

DFO (2017) Information needed in support of the identification of the habitat necessary for the survival and recovery of Rocky Mountain Ridged Mussel (*Gonidea angulata*). DFO Canadian Science Advisory Secretariat Research Document 2017/018.

Dunbar G (2009) Management Plan for Eurasian Watermilfoil (*Myriophyllum spicatum*) in the Okanagan, British Columbia. Okanagan Basin Water Board.

Duncan J (2019) Endangered mussel prevents milfoil removal in certain parts of Okanagan Lake. Kelowna Now. https://www.kelownanow.com/watercooler/news/news/Okanagan/Endangered_mussel_prevents_milfoil_removal_in_certain_parts_of_Okanagan_Lake#fs_81410 [Last accessed 11 February 2020]

Howard JK (2010) Sensitive Freshwater Mussel Surveys in the Pacific Southwest Region: Assessment of Conservation Status. Prepared for USDA Forest Service, Pacific Southwest Region.

Howard JK, Furnish JL, Brim Box J, Jepsen S (2015) The decline of native freshwater mussels (Bivalvia: Unionoida) in California as determined from historical and current surveys. California Fish and Game 101: 8–23.

Krueger K, Chapman P, Hallock M, Quinn T (2007) Some effects of suction dredge placer mining on the short-term survival of freshwater mussels in Washington. Northwest Science 81(4): 323–332. https://doi.org/10.3955/0029-344X-81.4.323

Lampert A, Hastings A, Grosholz ED, Jardine SL, Sanchirico JN (2014) Optimal approaches for balancing invasive species eradication and endangered species management. Science 344(6187): 1028–1031. https://doi.org/10.1126/science.1250763

Lirette D (2019) Some Okanagan beaches left in the weeds because of protected mussels. https://www.cbc.ca/news/canada/british-columbia/okanagan-beaches-weeds-threatened-mussels-1.5080107 [Last accessed 31 Jan 2020]

Lumley T (2004) Analysis of Complex Survey Samples. Journal of Statistical Software 9(1): 1–19. https://doi.org/10.18637/jss.v009.i08

Mageroy JH, Snook RM, Nield LM, Walker IR (2017) The impact of control methods for Eurasian Watermilfoil (*Myriophyllum spicatum*, L.) on Western Ridged Mussel (*Gonidea angulata*, Lea): Does rotovation harm the mussel? Northwest Science 91(2): 186–197. https://doi.org/10.3955/046.091.0208

Snook R (2015) Modeling Habitat Suitability for the Rocky Mountain ridged mussel (*Gonidea angulata*), in Okanagan Lake, British Columbia, Canada. MSc. Thesis.

Snook MR, Mageroy J, Pither J, Curtis J, Nield L, Walker IR (2019) Modeling Habitat Suitability for the Western Ridged Mussel (*Gonidea angulata* Lea) in Okanagan Lake, British Columbia, Canada.

Stanton L, Lauzier R, MacConnachie S, Nield L, Pollard S, Heron J, Davies S (2012) Exploratory Surveys and Directed Studies of Rocky Mountain Ridged Mussel (*Gonidea angulata* Lea, 1839) in British Columbia. Canadian Manuscript Report of Fisheries and Aquatic Sciences 3003: viii + 63.

Stephen C, Wade J (2018) Wildlife population welfare as coherence between adapted capacities and environmental realities: A case study of threatened lamprey on Vancouver Island. Frontiers in Veterinary Science 5(227): 1–5. https://doi.org/10.3389/fvets.2018.00227

Thom S (2019) Okanagan Basin Water Board Fights restrictions on milfoil control at popular lakes. Global News. https://globalnews.ca/news/5245005/okanagan-basin-water-board-milfoil-restrictions/ [Last accessed 11 February 2020]

Williams JD, Warren Jr ML, Cummings KS, Harris JL, Neves RJ (1993) Conservation status of freshwater mussels of the United States and Canada. Fisheries (Bethesda, Md.) 18(9): 6–22. https://doi.org/10.1577/1548-8446(1993)018<0006:CSOFMO>2.0.CO;2

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Does public information about wolf (*Canis lupus*) movements decrease wolf attacks on hunting dogs (*C. familiaris*)?

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Abstract

The threat that wolves (*Canis lupus*) pose to hunting dogs is one reason why Finnish hunters have negative attitudes towards wolves and one of the potential motivations for the illegal killing of wolves. During 2010–2017, wolves killed an average of 38 dogs (range 24–50) per year in Finland. Most of the attacks (91%) were directed at hunting dogs during the hunting season. To decrease the risk of attacks, the last seven positions (one position per hour) of GPS-collared wolves were accessible to the public with a 5 × 5 km resolution during the hunting seasons (from August 20th to February 28th) of 2013/2014 (from September 2nd onwards), 2015/2016, 2016/2017 and 2017/2018. The link was visited more than 1 million times in 3 of the 4 seasons. Fatal attacks on dogs occurred on 17% of the days during the hunting seasons of our study (n = 760 days). Both the attacks and visits peaked in September–November, which is the primary hunting season in Finland. According to the general linear model, the number of daily visits to the website was higher on days when fatal attacks occurred than on other days. Additionally, season and the number of days passed from the first day of the season were significantly related to the daily visits. Visits were temporally auto-correlated, and the parameter values in the model where the dependent variable was the number of visits on the next day were only slightly different from those in the first model. A two-way interaction between season and attack existed, and the least squares means were significantly different in 2017/2018. The change in daily visits between consecutive days was related only to the number of days from the beginning of the season. We examined whether this kind of service decreased dog attacks by wolves. Wolf attacks were recorded in 32% of the wolf territories, where at least one wolf had been collared (n = 22). However, within the territories without any GPS-collared wolves, the proportion of territories with wolf attack(s) was significantly higher than those elsewhere (50%, n = 48). Although public information decreased the risk of attacks, it did not completely protect dogs from wolf attacks and may in some cases increase the risk of illegally killing wolves. The most remarkable benefit of this kind of service to the conservation of the wolf population might be the message to the public that management is not overlooking hunters' concerns about wolf attacks on their dogs.

Keywords

Attacks, *Canis lupus*, *Canis familiaris*, GPS, hunting dogs, risk, territory

Introduction

Large carnivores and humans are often in conflict due to carnivore damage to livestock (Ciucci and Boitani 1998; Dahle et al. 1998; Kaczensky 1999; Madhusudan and Karanth 2002; Treves et al. 2002; Madhusudan 2003; Musiani et al. 2003; Polisar and Eisenberg 2003; Gunther et al. 2004; Frank et al. 2006; Iliopoulos et al. 2009; Inskip and Zimmermann 2009; Olson et al. 2015a; Montalvo et al. 2016). Wolves (*Canis lupus*) also kill domestic dogs (*Canis familiaris*) (Ciucci and Boitani 1998; Bangs and Shivik 2001; Kojola and Kuittinen 2002; Fritts et al. 2003).

Damage from large carnivores often generates displeasure and frustration (Bisi and Kurki 2008) and may fuel the illegal killing of carnivores (Liberg et al. 2012; Pohja-Mykrä and Kurki 2014; Olson et al. 2015b; Suutarinen and Kojola 2017). Illegal killing has been very influential, e.g., on Nordic wolf populations (Jansson et al. 2012; Liberg et al. 2012; Suutarinen and Kojola 2017).

Conflict between humans and large carnivores was absent in Europe and North America when large carnivores were almost extinct in the nineteenth and early to mid-twentieth centuries because of intense persecution, prey extermination and habitat conversion (Breitenmoser 1998; Linnell et al. 2009). With increased environmental awareness, protection, including better hunting management, reintroductions and habitat recovery after abandonment, wolves, bears and Eurasian lynxes experienced a continent-wide recovery in the first half of the twentieth century, which has led to conflicts in many countries in Europe and North America (Breitenmoser 1998; Chapron et al. 2014).

The present-day hunting culture in Scandinavia and Finland is highly dependent on dogs (Bisi et al. 2010). In Finland, hunters hunt small game such as small deer (*Cervidae* spp.) and hare (*Lepus timidus*), game birds such as grouse, waterfowl, and big game animals, mostly moose (*Alces alces*) and brown bear (*Ursus arctos*). The most popular big game animal is moose. In 2019, half of the Finnish 200 000 hunters participated in moose hunting (Natural Resource Institute Finland 2020).

Following the return of wolves in the 1980s, wolf attacks on hunting dogs (*Canis familiaris*) have led to conflicts between wolves and hunters, especially in Sweden and Finland (Kojola and Kuittinen 2002; Kojola et al. 2004b; Backeryd 2007; Bisi et al. 2007; Peltola and Heikkilä 2015), where the current dog-assisted hunting culture developed during an era without wolves (Bisi et al. 2007). In Finland, wolves kill an average of 38 dogs (range 24–50) per year. Over 90% of the attacks are on hunting dogs during the hunting season. Additionally, in the Great Lakes area in the USA and Scandinavia, dogs used for hunting are killed more often than are pet dogs (Backeryd 2007; Ruid et al. 2009; Edge et al. 2011; Olson et al. 2015a).

Specifically in Finland, moose hunting with dogs is popular, and its significance has made wolf depredation of hunting dogs even more serious than elsewhere in the world (Bisi et al. 2010). Wolves also compete with humans by preying on moose, which increases conflict (Wikenros 2011). One aspect of conflict is that the status of wolves is regulated by the EU Habitats Directive, which requires member states to establish a system of strict protection for wolves. After Finland joined the EU in 1995, the management of the wolf population has not been in the hands of the local people, and this scenario has also increased conflict (Bisi et al. 2007). Before EU membership, Finland could independently define its wolf policy. The wolf is still listed as a game species, and the population has been controlled by license-based hunting.

Although dog owners receive compensation from the Ministry of Agriculture and Forestry of Finland for wolf-killed dogs, the risk of losing a well-trained, valuable hunting dog to a wolf generates frustration among hunters and might even provoke some hunters to illegally kill wolves (Bisi et al. 2010; Liberg et al. 2012; Pohja-Mykrä and Kurki 2014; von Essen et al. 2014). In 2019, the Ministry of Agriculture and Forestry of Finland paid 160 000 euros in compensation for dogs attacked by wolves. The highest sums compensated for one dog have been approximately 10 000 euros (Ministry of Agriculture and Forestry of Finland 2020). Even if the loss of a dog is compensated for, compensation programs alone are not effective in reducing conflict or preventing poaching (Ciucci and Boitani 1998).

Wolf attacks on dogs also influence public opinion about wolves, which affects wolf conservation in Finland. Hunting dog conflicts, in particular, have been considered one of the most difficult issues in Finnish wolf policy, and resolving the conflict would contribute positively to the wolf policy (Peltola and Heikkilä 2015). Public opinion can influence the management policies of large carnivores (Wolch et al. 1997; Bisi et al. 2010; Olson et al. 2015a). Public opinion can become more positive towards wolves following quick responses to wolf conflicts (Ruid et al. 2009). In Finland, negative attitudes towards wolves were one of the reasons why the Finnish authorities allowed regulated hunting of wolves in 2015 and 2016 (The Finnish Wildlife Agency 2015). Hunters influence the moose population in Finland, which makes the management of both moose and wolves even more complicated.

Preventing wolf attacks on hunting dogs may lead to an increased acceptance of wolves, which is a key factor in protecting the population of Finnish wolves (ca. 200 wolves in 2019; Ministry of Agriculture and Forestry of Finland 2019). The number of wolves has fluctuated between 120 and 245 wolves since 2013. The wolf population has not reached the level of 25 breeding couples, the definition of a viable population (Ministry of Agriculture and Forestry of Finland 2019). On the IUCN Red list, the wolf is globally considered a 'least concern' species, but nationally, in Finland, it is classified as endangered (Hyvärinen et al. 2019).

Hunters' attitudes matter in the conservation of wolves in Finland (Bisi et al. 2007; Bisi et al. 2010). Effective and hunter-accepted ways to protect hunting dogs from wolves do not exist. Among most of the popular hunting methods used for terrestrial games, such as the mountain hare and especially moose, the dog is typically isolated and relatively far from the hunter when chasing game. This scenario makes it very challenging for the hunter to intervene during a wolf attack (Olson et al. 2015a). Ruid et al. (2009) found that attacks on hunting dogs generally occurred while hunters were ≥200 m away. In Finland, hunters use global positioning system (GPS) tracking for hunting dogs, which allows tracking over long distances (Paldanius et al. 2011). This approach may also increase the response time of hunters to a conflict, allowing them to determine that a problem has arisen and respond to that problem more quickly than with traditional VHF tracking systems. However, hunting dogs are exposed to many other risks when hunting. For example, vehicle collisions and shooting accidents kill or injure more hunting dogs than wolves in Sweden (Agria 2019).

Hunters can avoid releasing a dog if they find fresh wolf tracks on their hunting grounds, but this precaution can usually only be implemented when the ground is covered by snow. Harnesses and other means that protect dogs from wolf bites can be functional (Fedderwitz 2010), but they do not prevent dogs from being attacked. Even if the attacked dog survives, it might refuse to participate in hunting anymore (Tallavaara 2007). Therefore, it is important to develop methods that prevent wolf attacks on hunting dogs.

In this study, we examined the seasonal use of publicly accessible, online wolf location data with a 5×5 km resolution and evaluated the effectiveness of this kind of service in preventing wolf attacks on hunting dogs. We hypothesized that as the number of attacks increases, the number of visits to the publicly accessible website increases accordingly and that as the number of collared wolves available for the public to monitor increases, the website will be visited more. Furthermore, we predicted that wolf attacks on dogs will be fewer in territories with one or more collared wolf.

Methods

Study areas

We analysed daily visits to the public website (http://riistahavainnot.fi/suurpedot/havaintokartta; see below) as a daily sum from Finland because it is forbidden by law to reveal the locations of the visitors on the website. A territory-specific study of wolf attacks was carried out in eastern Central Finland (Fig. 1). This area contains three counties: Kainuu, Northern Savo and North Carelia. The area covers a total of 66 401 km^2 (National land Survey of Finland 2016). The population is approximately 484 000. The density of the population varies from 3.58 citizens/km in Kainuu to 10 citizens/km in North Carelia and 15 citizens/km in Northern Savo (Statistics Finland 2019). In every county, the population lives in large cities, and in rural areas, the population is sparse.

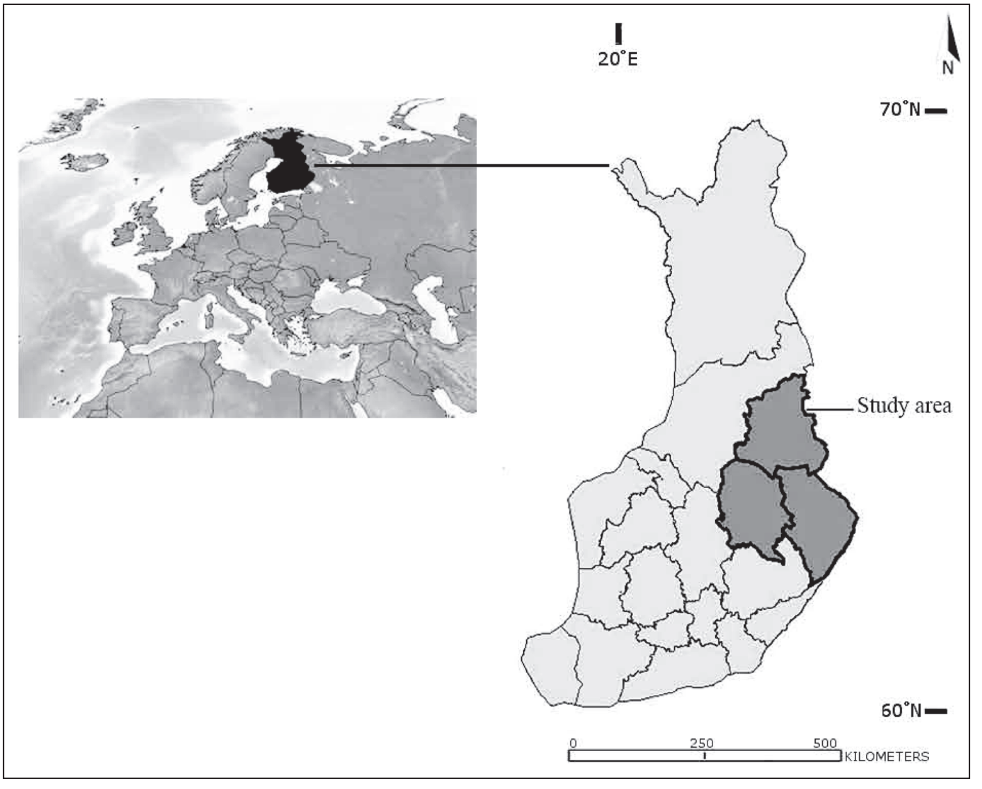

Figure 1. The study area encompassed three provinces: Kainuu, Northern Savo and North Carelia, in eastern Finland.

The study area contains mainly coniferous boreal forest, and the predominant tree species are Scotch pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) mixed with birches (*Betula pendula* and *B. pubescens*) and some other deciduous trees. The forests are commercially exploited, and therefore, young stands and clear cuts are common. The area is covered by a dense network of timber roads that are mostly accessible to everyone and driveable by car, except in the winter. Permanent winter snow usually appears in November and melts in late April or early May.

Moose is the primary prey species of the wolves in our study area (Gade-Jorgensen and Stagegaard 2000; Kojola et al. 2004a). Moose may constitute >90% of the biomass ingested by wolves (Gade-Jorgensen and Stagegaard 2000). The density of moose is 0.2–0.4 animals/km^2 (Natural Resources Institute Finland 2019a). There is a small population (approximately 700) of wild forest reindeer (*Rangifer tarandus fennicus*) in a 10 000 km^2 area in the Kainuu region in eastern Finland, but these reindeer constitute far less of the wolves' diet than moose in that region (20–50%; Kojola et al. 2004a). In addition to wild ungulates, wolves prey on smaller mammals (e.g., hare), which constitute 20–30% of the diets of the wolves (Kojola et al. 2004a).

Between 2013 and 2017, the Finnish wolf population was concentrated in eastern Finland (~50–60% of the population); however, in 2018, the population started to spread to southern Finland, and only ~30% lived in eastern Finland (Natural Resources Institute Finland 2019c). The population has fluctuated over the study years. In 2015 and 2016, the population decreased due to the regulated hunting of wolves, which resulted in the killing of 70 individuals, with a combined known mortality of 120 wolves in 2015 and 2016 (Ministry of Agriculture and Forestry of Finland 2020).

Collared wolves

In our study period, 2013 and 2015–2017, wolves were captured by the Natural Resources Institute of Finland in February–April, mostly by darting them from a helicopter (see Kojola et al. 2016). A few wolves were captured using an armed lasso from a snowmobile. The detailed protocols of the capture and immobilization procedures are given elsewhere (Kojola et al. 2006; Wabakken et al. 2007). Between 2013 and 2017, there were 33 collared wolves at the beginning of the hunting season (August 20th). No wolves were collared in 2014, and therefore, the hunting season of 2014/2015 was excluded from the analyses.

The wolves were collared with transmitters containing a GPS and a global system for mobile communications (GSM) to obtain their locations (Vectronic Aerospace, Berlin, Germany). Capturing, handling, and anaesthetizing the wolves were performed according to the guidelines issued by the Animal Care and Use Committee at the University of Oulu and the permit provided by the National Animal Experiment Board. Collars must receive a signal from at least three satellites to obtain an exact location. The interval between subsequent attempts was one hour, and the collar sent a bundle of locations after it had stored seven locations.

We used the data from only one collared wolf per pack for the following analyses. The movements of one wolf were representative of the movements of the whole pack because each wolf pack moves mostly as one unit in the autumn and winter (Okarma et al. 1998; Mech and Boitani 2003).

The mean 100% kernel territory size for the GPS-collared wolves (n = 22) during the autumn and winter hunting season (from August 20th to February 28th) in eastern Central Finland (Fig. 1) was 1 137 km^2 (range 457 km^2 –1 700 km^2, SE 117, 35), which we used as an approximation of the territory size for the uncollared wolves. Because the shape of the territories without collared wolves was not known for 2013 and 2015, we used circular wolf territories with a radius of 19 km, which is the radius of the 1 137 km^2 home range.

The midpoints of such territories were estimated using point observations provided by a network of large carnivore contact persons (ca. 2 000 people; Kojola et al. 2018) who record wolf observations and input the data into a digital large carnivore observation system, specifying the observation type (sighting, track, prey kill site, or livestock depredation), date, geographic location, age, status, number of animals, and front paw-print dimensions. Contact persons are nominated by local game management associations and are educated about the biology, ecology and movement behaviour of wolves and footprint identification (Kojola et al. 2018). Core wolf territory areas were estimated based on the location of the point observations (Kojola et al. 2018).

For the 2016/2017 and 2017/2018 seasons, the polygons for the territories of wolves without collars were evaluated using volunteer-provided point observations and genetic monitoring by professionals (Kojola et al. 2018). In total, there were 48 territories within the study area in eastern Finland (Fig. 1) occupied by uncollared wolves.

In our study area in eastern Finland, we considered only the attacks that took place during GPS tracking or those when we could be sure that the territory was occupied by collared wolves when the attack occurred. For example, if there was a gap in the GPS signal, then we assumed that the territory was still occupied by the same wolves as those before the gap. With the uncollared wolves, we used the whole hunting season as a reference period. If we could not be sure whether the attack had taken place in a territory occupied by collared or uncollared wolves, then we did not include the case in the analyses.

Wolf attacks

The Ministry of Agriculture and Forestry of Finland and officials of municipalities provided data on wolf attacks on hunting dogs. Not all attacks are reported to the ministry because dog owners did not apply for compensation. Each attack was counted once, and overlaps were deleted. In total, 55 attacks took place within the territories in the study area in eastern Finland.

Public website

The website link (http://riistahavainnot.fi/suurpedot/havaintokartta) was active throughout the main hunting season (from August 20th to February 28th), showing the last locations of the collared wolves with a 5×5 km accuracy. The website is maintained by the Natural Resources Institute of Finland, which also collars the wolves with funding from the Ministry of Agriculture and Forestry of Finland. The website also includes information about the wolf territories in Finland, dispersing wolves, and wolf observations that citizens have recorded.

In Finland, allocation information on collared wolves has been released to the public during hunting seasons since 2013. Some wolves (n = 5) were collared more than once, but we only considered the last year of monitoring when considering how many of them survived through the hunting season.

Statistical analyses

We performed a general linear model (GLM) analysis to test whether the number of daily visits to the webpage was related to the occurrence of fatal attacks on (1) the previous day or (2) the same day and the number of GPS collared wolves, season, and the days passed from the beginning of the season. Because visitors to the webpage may become more reactive to the known attacks, we tested two-way interactions between season and the occurrence of attacks on the number of daily attacks. Then, we examined whether attacks increased with the number of visits via a GLM, where the change in visits between consecutive days was related to the number of attacks, season, and days passed from the beginning of the season. To evaluate the effects of the public webpage on the risk of attack, we performed a non-parametric Chi-square test to compare the distribution of territories of collared and uncollared wolves, with and without attacks. The analyses were conducted with SPSS for Windows version 24.0, using a significance value of 0.05.

Results

The website providing information about the locations of the GPS collared wolves was first published for the 2013/2014 hunting season when the site was visited almost two million times. The visits were not that frequent in 2015/2016 (<million), which followed the season when no wolves had an active transmitter, but in the following hunting seasons of 2016/2017 and 2017/2018, the number of visitors was over a million.

The number of visits to the webpage was high during the day when the link was opened, probably owing to the media release announcing the opening of the service. The overall pattern of daily visits in 2013/2014 and 2015/2016 was a gradual decrease across the season, while in 2016/2017 and 2017/2018, a peak in visits occurred between the 50th and 100th days from the opening of the service (Fig. 2), which was temporally consistent with the first weeks of moose hunting.

At least one fatal wolf attack on dogs was recorded on 17% of the days (n = 760) we monitored during this study. In 4% of the cases, there was also an attack on the next day. The number of daily visits was positively related to the recent occurrence of fatal attacks by wolves on dogs during the hunting season, the number of wolves with an active collar, and the number of days passed from the day when the link was opened (Table 1). The daily visits were temporally auto-correlated, and therefore, having the number of visits in the day following the attack as the dependent variable provided results that were only slightly different from those of the first model (Table 1). This model also included the cases (27 of 129) where at least one attack had taken place on both the previous day and the same day. In a model where only these particular cases were denoted positive and the others negative, statistically significant differences in the number of visits did not exist.

The two-way interaction term between hunting season and the number of daily visits was significant (Table 1), providing evidence that the relationship between attacks and visits varied by season. Least squares means differed only for the last hunting season (2017/2018).

The mean number of days that wolf-specific public information was available was 168 days (range 38–193). The large variation was due to wolf mortalities and technical flaws in the transmitters. The locations of 22 collared wolves were still publicly available after the hunting season ended (February 28th). Of the remaining 11 wolves, six wolves were killed as a result of permitted hunting or other killing, two wolves were

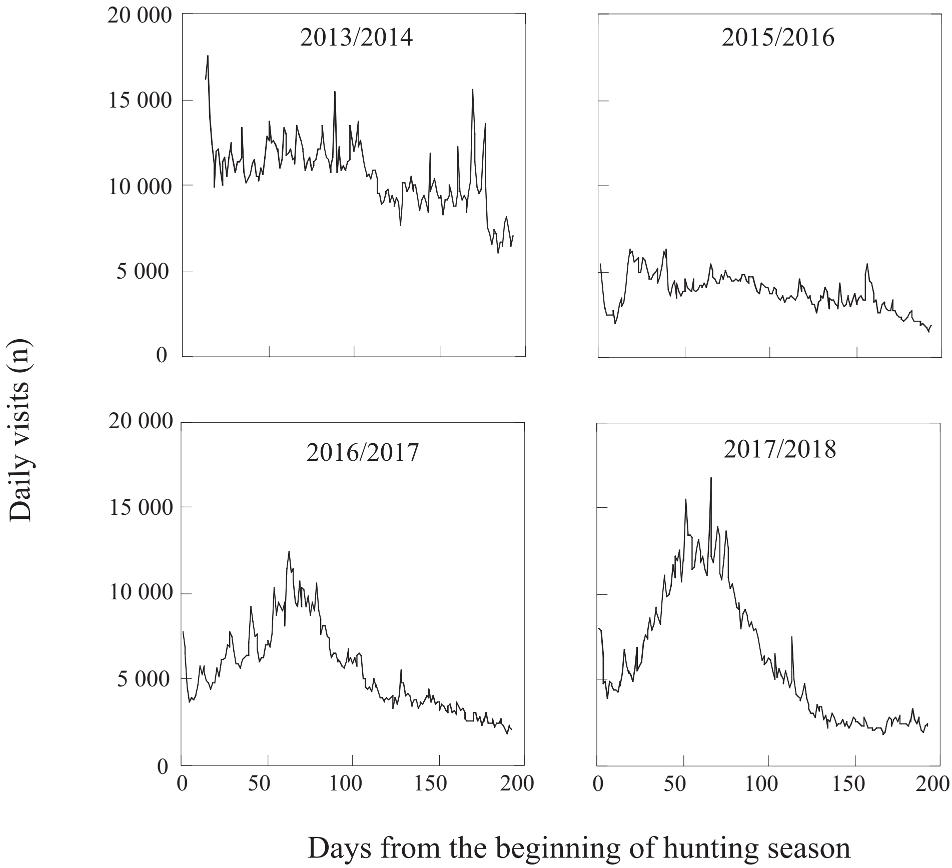

Figure 2. Daily visits to the publicly accessible website showing wolf location information during the hunting seasons (from August 20th to February 28th, in 2013 the website was opened on September 2nd).

found dead (reasons for death unknown), and three wolves were lost during monitoring because the collars stopped working. Considering poaching, it is possible that the wolves that were lost during monitoring or were found dead (5 of 33) were illegally killed. Suutarinen and Kojola (2017) reported that illegal killing was the main cause of mortality in collared wolves (40%); however, the number of possibly illegally killed wolves in this study was relatively low (15%). The time of disappearance or possible death was between October 23th and January 31th. None of the wolves disappeared or were found dead after a dog attack.

In eastern Finland, the proportion of the territories where the attacks took place was higher in the territories without any GPS-collared wolves (50%, n = 48) than that in the territories with at least one collared wolf (32%, n = 22, chi-square for difference = 7.86; p = 0.005; n = 48, Fig. 3). There were 0.59 attacks per territory with at least one collared wolf and 0.88 attacks per territory without any collared wolves.

Table 1. General linear model for the number of daily visits to the public webpage providing information about positions of GPS collared wolves at a 5×5 km resolution during four hunting seasons in Finland. Wolf attack (no or yes) and hunting season (2013, 2015, 2016, or 2017) were treated as categorical variables.

Dependent variable	Independent variable	Level	Coefficient	F	P	Adj. R^2
Visits in the same day	**Constant**		-1 668.51			
	Wolf attack	0	-223.44	5.44	0.020	
	Hunting season	2013	-1 114.33	58.13	< 0.001	
		2015	-1 186.20			
		2016	2 618.60			
	Number of collared wolves		1 806.28	56.37	< 0.001	
	Days from the season's 1st day		-8.44	10.85	0.001	
	Hunting season*attack	0*2013	248.08	9.10	< 0.001	
		0*2015	301.29			
		0*2016	268.15			
						0.709
Visits on the next day	**Constant**		-2 664.20			
	Wolf attack	0	-252.44	6.89	0.009	
	Hunting season	2013	-1 535.99	59.10	< 0.001	
		2015	-1 016.72			
		2016	2 947.91			
	Number of collared wolves		1 183.34	66.61	< 0.001	
	Days from the season's 1st day		-6.92	7.28	0.007	
	Season*attack	0*2013	175.83	7.92	< 0.001	
		0*2015	277.53			
		0*2016	299.13			
						0.710

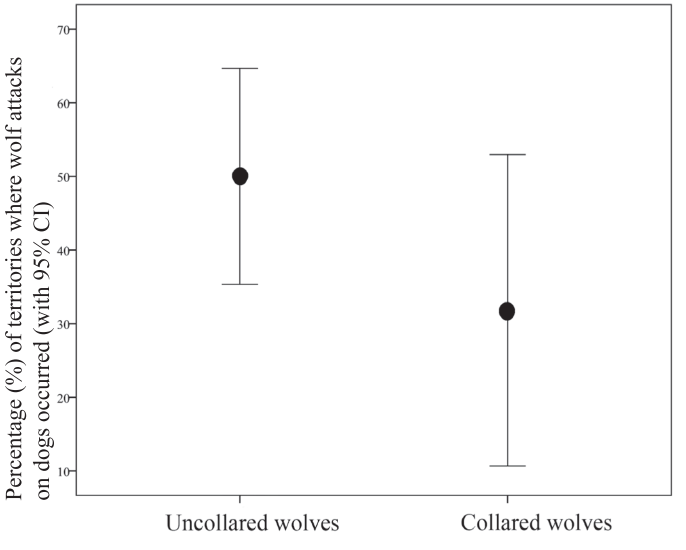

Figure 3. Percentages (%) of the territories in eastern Finland during the hunting seasons of 2013/2014 and 2015/2016–2017/2018 where wolf attacks on dogs occurred, including the mean numbers and 95% confidence intervals of the territories with and without collared wolves that had wolf attacks. As we predicted, the proportion of territories where attacks occurred was higher in territories where none of the wolves have a collar than in other territories (chi-square for difference = 7.86; p = 0.005; n = 48) (n = 22 for collared wolves, n = 48 for uncollared wolves).

Discussion

Public wolf location information was very popular. In the first hunting season when the service was available (2013/2014), the number of visits to the website was 2 million. Our results provided evidence that temporary and recent wolf location information might decrease the risk of attacks on hunting dogs. However, this protective measure has many limitations. The measure is expensive and cannot provide full protection from wolf attacks. In some cases, a collared wolf was perhaps illegally killed because of the website showing its location publicly.

The compensation the Ministry of Agriculture and Forestry of Finland provides for a highly certified dog used in hunting, such as for moose or brown bears, may be even higher than the expenses of collaring a wolf (ca. 10 000 euros, I. Kojola unpublished data). When costs and effects are considered, the website does not prevent dog attacks; it appears likely that the website did protect some dogs or increase the prevention of some attacks. It is notable that people usually form strong emotional bonds to their dogs (including both pet and hunting dogs), and dogs are often regarded as members of the family. Wolf attacks on dogs can result in emotional trauma (Ratamäki 2009; Lescureux and Linnell 2014; Niemi et al. 2014). Therefore, the monetary value of a dog based on different estimates, e.g., dog health and success in dog show and hunting tests (Finlex data bank 2020), does not indicate much about a dog's real value to its owners. Furthermore, losing a dog to wolves can erode the fragile tolerance of hunters for wolves (Lescureux and Linnell 2014).

Delivering information about wolf locations to the public to prevent wolf attacks on hunting dogs is a rare practice, even though wolf depredation on dogs is a well-known phenomenon everywhere unleashed hunting dogs are used within the range of wolves, e.g., Wisconsin (Olson et al. 2015a). In the USA, black bears are hunted with dogs in many states (Bump et al. 2013), and wolf depredation on dogs is a problem, as in Finland and Scandinavia. Solutions include sharing information with hunters about wolf caution areas (Wisconsin Department of Natural Resources 2020).

Showing wolf locations to hunters on publicly accessible websites has only been used in Finland, Sweden and Norway. In Finland, the mean proportion of wolf territories with collared wolves is approximately 25%. Wolf attacks on hunting dogs are much less frequent in western Finland than in our study area in eastern Finland (Kojola et al. unpublished data), where approximately half of the Finnish territories are located (Kojola et al. 2018). One reason for the higher risk might be lower ungulate biomass in eastern territories than in the other territories (Kojola et al. unpublished data). The likely reason almost all territories with collared wolves are situated in eastern Finland is land ownership. Collaring requires that a landowner provide a permit, and only land areas owned by the state or forest companies are large enough for capturing wolves. Such areas are highly concentrated in eastern Finland (Natural Resources Institute Finland 2019b). Negative attitudes towards wolves and wolf research are not rare (Bisi et al. 2010), and if even a few private landowners within wolf territory do not accept collaring, then it is often impossible to conduct.

Technical improvements to collar functions are desired by hunters. The requirement of having seven consecutive locations before the locations are downloadable to the webpage means that even when all consecutive attempts to locate a wolf are successful, the time passed since the last location can be seven hours. Online connections to wolves might help hunters protect their dogs more efficiently but might also encourage some people to try to illegally kill the wolves. The 5×5 km accuracy of the currently available data on the public website is coarse enough to maintain the risk at a relatively low rate, at least during the snow-free season (normally starting in November). Although Suutarinen and Kojola (2017) reported that illegal killing was the main cause of mortality of collared wolves (40%), the number of possibly illegally killed wolves in this study was relatively low (15%). Collars did not appear to affect poaching risk because estimated poaching rates based on the fates of collared wolves were highly correlated with fluctuations in the Finnish wolf population (Suutarinen and Kojola 2017).

There is little evidence that wolves actively seek dogs, and the attacks appear to be more opportunistic in nature (Paquet 1991); however, it is possible to predict the probability of an attack based on non-wolf-related factors such as landscape and the severity of the previous year winter and wolf-dependent factors such as pack size (Edge et al. 2011; Olson et al. 2014). More research is needed because attacks are still not well documented in the scientific literature (Butler et al. 2015).

Although the number of attacks in Finland is not high, from 2010–2017, wolves killed an average of 38 dogs per year, and it is important to note that a much higher number of hunting dogs are potential targets of a wolf attack, especially within the wolf territories. On some occasions, the risk of wolf attacks on dogs has led hunters to stop hunting entirely (Bisi et al. 2007). Hunters may sometimes skip the training and testing of hunting dogs owing to the risk. Training hunting dogs is important for hunters who selectively breed hunting dogs. On the other hand, Finland has to protect its endangered wolf population, and the authorities have an ethical and professional responsibility to manage wildlife populations as best as they can and carefully consider the best long-term solutions, e.g., changing people's attitudes (Bisi et al. 2010), to accomplish this (Wallach et al. 2015).

Conclusions

Our study reveals that knowing where wolves occur decreases the risk of attacks on hunting dogs. Although the measure is expensive and there are many reservations, the website was useful for hunting dog owners and might mitigate the conflict between humans and wolves. Although public information would decrease the risk of attacks, it does not provide full protection for dogs and may in some cases increase the risk of illegally killing of wolves. The most remarkable benefit of this kind of service to conservation of the wolf population might be its message to the public, a demonstration that management is not overlooking hunters' concern about wolf attacks on their dogs.

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References

Agria (2019) Agria. https://www.agria.se/pressrum/statistik-om-djur-djurvard-och-djurhalsa/vad-sager-statistiken-om-skador-vid-jakt/ [Accessed on 6 April 2020]

Backeryd J (2007) Wolf Attacks on Dogs in Scandinavia 1995–2005: Will Wolves in Scandinavia go Extinct if Dog Owners are Allowed to Kill a Wolf Attacking a Dog? PhD Thesis, Swedish University of Agricultural Sciences, Sweden, 22 pp. https://www.slu.se/globalassets/ew/org/inst/ekol/forskning/projekt/skandulv/publikationer/studentarbeten/backeryd-2007-wolf-attacks-on-dogs-in-scandinavia-1995-2005.pdf

Bangs E, Shivik JA (2001) Managing Wolf Conflict with Livestock in the Northwestern United States. USDA National Wildlife Research Center staff Publications, 550 pp. https://digitalcommons.unl.edu/icwdm_usdanwrc/550

Bisi J, Kurki S (2008) The Wolf Debate in Finland. University of Helsinki, Ruralia Institute. http://128.214.67.123/ruralia/julkaisut/pdf/Publications12.pdf

Bisi J, Kurki S, Svensberg M, Liukkonen T (2007) Human dimensions of wolf (*Canis lupus*) conflicts in Finland. European Journal of Wildlife Research 53(4): 304–314. https://doi.org/10.1007/s10344-007-0092-4

Bisi J, Liukkonen T, Mykrä S, Pohja-Mykrä M, Kurki S (2010) The good bad wolf-wolf evaluation reveals the roots of the Finnish wolf conflict. European Journal of Wildlife Research 56(5): 771–779. https://doi.org/10.1007/s10344-010-0374-0

Breitenmoser U (1998) Large predators in the Alps: The fall and rise of man's competitors. Biological Conservation 83(3): 279–289. https://doi.org/10.1016/S0006-3207(97)00084-0

Bump JK, Murawski CM, Kartano LM, Beyer Jr DE, Roell BJ (2013) Bear-baiting may exacerbate wolf-hunting dog conflict. PLoS ONE 8(4): e61708. https://doi.org/10.1371/journal.pone.0061708

Butler JRA, Linnell JDC, Morrant J, Athrey V, Lescureux N, McKeown A (2015) Dog eat dog, cat eat dog: social-ecological dimensions of dog predation by wild carnivores. In: Gompper ME (Ed.) Freeranging Dogs and Wildlife Conservation. Oxford University Press, 117–143. https://doi.org/10.1093/acprof:osobl/9780199663217.003.0005

Chapron G, Kaczensky P, Linnel JDC, von Arx M, Huber D, Andrén H, López-Bao JV, Adamec M, Álvares F, Anders O, Balčiauskas L, Balys V, Bedő P, Bego F, Blanco JC, Breitenmoser U, Brøseth H, Bufka L, Bunikyte R, Ciucci P, Dutsov A, Engleder T, Fuxjäger C, Groff C, Holmala K, Hoxha B, Iliopoulos Y, Ionescu O, Jeremić J, Jerina K, Kluth G, Knauer F, Kojola I, Kos I, Krofel M, Kubala J, Kunovac S, Kusak J, Kutal M, Liberg O, Majić A, Männil P, Manz R, Marboutin E, Marucco F, Melovski D, Mersini K, Mertzanis Y, Mysłajek RW, Nowak S, Odden J, Ozolins J, Palomero G, Paunović M, Persson J, Potočnik H, Quenette P-Y, Rauer G, Reinhardt I, Rigg R, Ryser A, Salvatori V, Skrbinšek T, Stojanov A, Swenson JE, Szemethy L, Trajçe A, Tsingarska-Sedefcheva E, Váňa M, Veeroja R, Wabakken P, Wölfl M, Wölfl S, Zimmermann F, Zlatanova D, Boitani L (2014) Recovery of large carnivores in Europe's modern human-dominated landscapes. Science 346(6216): 1517–1519. https://doi.org/10.1126/science.1257553

Ciucci P, Boitani L (1998) Wolf and dog depredation on livestock in central Italy. Wildlife Society Bulletin 26: 504–514. www.jstor.org/stable/3783763

Dahle B, Sorensen OJ, Wedul EH, Swenson JE, Sandegren F (1998) The diet of brown bears *Ursus arctos* in central Scandinavia: Effect of access to free-ranging domestic sheep *Ovis aries*. Wildlife Biology 4(2): 147–158. https://doi.org/10.2981/wlb.1998.017

Edge JL, Beyer Jr DE, Belant JL, Jordan MJ, Roell BJ (2011) Livestock and domestic dog predations by wolves in Michigan. Human-wildlife interactions 5(1): 66–78. https://doi.org/10.26077/x37j-ph76

Fedderwitz F (2010) Protecting Dogs Against Attacks by Wolves (*Canis lupus*), with Comparison to African wild Dogs (*Lycaon pictus*) and Dholes (*Cuon alpinus*). Masters Thesis. University of Linköping, Sverige, 32 pp. http://www.diva-portal.org/smash/get/diva2:322325/fulltext01.pdf

Finlex data bank (2020) Finlex data bank. [Accessed on 6 April 2020] https://www.finlex.fi/fi/laki/alkup/2018/20180834

Frank L, Hemson G, Kushnir H, Packer C (2006) Lions, conflict and conservation in Eastern and Southern Africa. Eastern and Southern African lion conservation workshop in Johannesburg, south Africa 11–13th of January 2006, background paper. https://www.researchgate.net/publication/242139181_Lions_Conflict_and_Conservation_in_Eastern_and_Southern_Africa

Fritts S, Stephenson R, Hayes R, Boitani L (2003) Wolves and humans. In: Mech D, Boitani L (Eds) Wolves: Behavior, Ecology, and Conservation. University Chicago Press, Chicago, IL, 289–316. https://digitalcommons.unl.edu/usgsnpwrc/317/

Gade-Jorgensen I, Stagegaard R (2000) Diet composition of wolves *Canis lupus* in east-central Finland. Acta Theriologica 45: 537–547. https://doi.org/10.4098/AT.arch.00-52

Gunther KA, Haroldson MA, Frey K, Cain SL, Copeland J, Schwartz CC (2004) Grizzly bear-human conflicts in the Greater Yellowstone ecosystem, 1992–2000. Ursus 15(1): 10–22. https://doi.org/10.2192/1537-6176(2004)015<0010:GBCITG>2.0.CO;2

Hyvärinen E, Juslén A, Kemppainen E, Uddström A, Liukko U [Eds] (2019) The 2019 Red List of Finnish Species. http://hdl.handle.net/10138/299501

Iliopoulos Y, Sgardelis S, Koutis V, Savaris D (2009) Wolf depredation on livestock in central Greece. Mammal Research 54(1): 11–22. https://doi.org/10.1007/BF03193133

Inskip C, Zimmermann A (2009) Human-felid conflict: A review of patterns and priorities worldwide. Oryx 43(01): 18–34. https://doi.org/10.1017/S003060530899030X

Jansson E, Ruokonen M, Kojola I, Aspi J (2012) Rise and fall of a wolf population: Genetic diversity and structure during recovery, rapid expansion and drastic decline. Molecular Ecology 21(21): 5178–5193. https://doi.org/10.1111/mec.12010

Kaczensky P (1999) Large carnivore depredation on livestock in Europe. Ursus 11: 59–72. https://www.jstor.org/stable/3872986

Kojola I, Kuittinen J (2002) Wolf attacks on dogs in Finland. Wildlife Society Bulletin 30: 498–501.

Kojola I, Huitu O, Toppinen K, Heikura K, Heikkinen S, Ronkainen S (2004a) Predation on European wild forest reindeer (*Rangifer tarandus*) by wolves (*Canis lupus*) in Finland. Journal of Zoology 263(3): 229–235. https://doi.org/10.1017/S0952836904005084

Kojola I, Ronkainen S, Hakala A, Heikkinen S, Kokko S (2004b) Interactions between wolves *Canis lupus* and dogs *C. familiaris* in Finland. Wildlife Biology 10(1): 101–105. https://doi.org/10.2981/wlb.2004.014

Kojola I, Aspi J, Hakala A, Heikkinen S, Ilmoni C, Ronkainen S (2006) Dispersal in an expanding wolf population in Finland. Journal of Mammalogy 87(2): 281–286. https://doi.org/10.1644/05-MAMM-A-061R2.1

Kojola I, Hallikainen V, Mikkola K, Gurarie E, Heikkinen S, Kaartinen S, Nikula A, Nivala V (2016) Wolf visitations close to human residences in Finland: The role of age, residence density, and time of day. Biological Conservation 198: 9–14. https://doi.org/10.1016/j.biocon.2016.03.025

Kojola I, Heikkinen S, Holmala K (2018) Balancing costs and confidence: Volunteer-provided point observations, GPS telemetry and the genetic monitoring of Finland's wolves. Mammal Research 63(4): 415–423. https://doi.org/10.1007/s13364-018-0371-3

Lescureux N, Linnell JDC (2014) Warring brothers: The complex interactions between wolves (*Canis lupus*) and dogs (*C. Familiaris*) in a conservation context. Biological Conservation 171: 232–245. https://doi.org/10.1016/j.biocon.2014.01.032

Liberg O, Chapron G, Wabakken P, Pedersen HC, Hobbs NT, Sand H (2012) Shoot, shovel and shut up: Cryptic poaching slows restoration of a large carnivore in Europe. Proceedings. Biological Sciences 279(1730): 910–915. https://doi.org/10.1098/rspb.2011.1275

Linnell JDC, Breitenmoser U, Breitenmoser C, Odden J, von Arx M (2009) Recovery of Eurasian lynx in Europe: What part has reintroduction played? In: Hayward MW, Somers MJ (Eds) Reintroduction of Top-Order Predators. Blackwell Publishing, 72–91. https://doi.org/10.1002/9781444312034.ch4

Madhusudan MD (2003) Crop depredation by large mammals in the interior villages of Bhadra Tiger Reserve, South India. Environmental Management 31: 0466–0475. https://doi.org/10.1007/s00267-002-2790-8

Madhusudan MD, Karanth U (2002) Local hunting and the conservation of large mammals in India. Ambio 31(1): 49–54. https://doi.org/10.1579/0044-7447-31.1.49

Mech D, Boitani L [Eds] (2003) Wolves: Behavior, Ecology, and Conservation. University Chicago Press, Chicago, IL, 472 pp. https://doi.org/10.14430/arctic540

Ministry of Agriculture and Forestry of Finland (2016) Ministry of Agriculture and Forestry of Finland. [Accessed on 1 April 2020] https://mmm.fi/artikkeli/-/asset_publisher/susien-kannanhoidollisen-metsastyksen-arviointi-valmistunut-metsastys-jatkuu-muutoksin#880ac13e

Ministry of Agriculture and Forestry of Finland (2019) Management plan for the wolf population in Finland. Publications of the Ministry of Agriculture and Forestry. http://urn.fi/URN:ISBN:978-952-366-016-8

Ministry of Agriculture and Forestry of Finland (2020) Ministry of Agriculture and Forestry of Finland. [Accessed on 1 April 2020] https://riistavahinko.mmm.fi/login.html

Montalvo V, Alfaro L, Saenz C, Cruz J, Fuller TK, Carrillo E (2016) Factors affecting jaguar and puma predation on livestock in Costa Rica. Wildlife Biology in Practise 12(1): 32–42. https://doi.org/10.2461/wbp.2016.12.3

Musiani M, Mamo C, Boitani L, Callaghan C, Gates CC, Mattei L, Visalberghi E, Breck S, Volpi G (2003) Wolf depredation trends and the use of fladry barriers to protect livestock in Western North America. Conservation Biology 17(6): 1538–1547. https://doi.org/10.1111/j.1523-1739.2003.00063.x

National land Survey of Finland (2016) National land Survey of Finland. [Accessed on 15 March 2017. Retrieved from Finnish surface area by municipalities 1.1.2016]

Natural Resources Institute Finland (2019a) Natural Resources Institute Finland. http://riistahavainnot.fi/sorkkaelaimet/hirvitiheys [Accessed on 27 August 2019]

Natural Resources Institute Finland (2019b) Natural Resources Institute Finland. https://www.luke.fi/tietoa-luonnonvaroista/riista/susi/suden-kanta-arvioinnin-menetelmat/susien-merkintapyynnit-ja-niihin-tarvittavat-luvat/ [Accessed on 27 August 2019]

Natural Resources Institute Finland (2019c) Natural Resources Institute Finland. http://riistahavainnot.fi/suurpedot/kannanarviointi/lausunnot?lang=fi [Accessed on 5 April 2020]

Natural Resources Institute Finland (2020) Natural Resources Institute Finland. https://www.luke.fi/en/natural-resources/game-and-hunting/hunting/ [Accessed on 22 March 2020]

Niemi M, Pellikka J, Hiedanpää J (2014) Metsästyskoirien suojaaminen susilta (Protecting the hunting dogs from wolves). Natural Resources Institute Finland report 49/2014. http://docplayer.fi/5913678-Metsastyskoirien-suojaaminen-susilta.html

Okarma H, Jedrzejewski W, Schmidt K, Sniezko S, Bunevich AN, Jedrzejewska B, Jedrzejewski W, Sniezko S, Jedrzejewska B (1998) Home ranges of wolves in Bialowieza Primeval Forest, Poland, compared with other Eurasian populations. Journal of Mammalogy 79(3): 842–852. https://doi.org/10.2307/1383092

Olson E, Treves B, Wydeven AP, Ventura SJ (2014) Landscape predictors of wolf attacks on bear-hunting dogs in Wisconsin, USA. Wildlife Research 41(7): 584–597. https://doi.org/10.1071/WR14043

Olson E, Deelen T, Wydeven A, Ventura S, Macfarland M (2015a) Characterizing wolf-human conflicts in Wisconsin, USA. Wildlife Society Bulletin 39(4): 676–688. https://doi.org/10.1002/wsb.606

Olson E, Stenglein JL, Shelley V, Rissman AR, Browne-Nuñes C, Voyles Z, Wydeven AP, Van Deelen T (2015b) Pendulum swings in wolf management led to conflict, illegal kills, and a legislated wolf hunt. Conservation Letters 8(5): 351–360. https://doi.org/10.1111/conl.12141

Paldanius M, Kärkkäinen T, Väänänen-Vainio-Mattila K, Juhlin O, Häkkilä J (2011) Communication technology for human-dog interaction: exploration of dog owners' experiences and expectations. CHI 2011: Proceedings of the SIGCHI Conference on Human Factors in Computing Systems, 2641–2650. https://doi.org/10.1145/1978942.1979329

Paquet PC (1991) Winter Spatial Relationships of Wolves and Coyotes in Riding Mountain National Park, Manitoba. Journal of Mammalogy 72(2): 397–401. https://doi.org/10.2307/1382113

Peltola T, Heikkilä J (2015) Response-ability in wolf-dog conflicts. European Journal of Wildlife Research 61(5): 711–721. https://doi.org/10.1007/s10344-015-0946-0

Pohja-Mykrä M, Kurki S (2014) Strong community support for illegal killing challenges wolf management. European Journal of Wildlife Research 60(5): 759–770. https://doi.org/10.1007/s10344-014-0845-9

Polisar J, Eisenberg JF (2003) Jaguars, pumas, their prey base, and catlle ranching: Ecological interpretations of a management problem. Biological Conservation 109(2): 297–310. https://doi.org/10.1016/S0006-3207(02)00157-X

Ratamäki O (2009) Societal sustainability and governance in Finland's wolf policy. Dissertation. University of Joensuu publication no. 94. https://doi.org/10.1177/1070496508320251

Ruid DB, Paul WJ, Roell BJ, Wydeven AP, Willging RC, Jurewicz RL, Lonsway DH (2009) Wolf-human conflicts and management in Minnesota, Wisconsin, and Michigan. In: Wydeven AP, Van Deelen TR, Heske EJ (Eds) Recovery of gray wolves in the Great Lakes Region of the United States: an endangered species success story. Springer New York, 279–295. https://doi.org/10.1007/978-0-387-85952-1_18

Statistics Finland (2019) Statistics Finland. https://pxnet2.stat.fi/PXWeb/pxweb/en/StatFin/StatFin__vrm__vaerak/statfin_vaerak_pxt_11ra.px/ [Accessed on 5 April 2020]

Suutarinen J, Kojola I (2017) Poaching regulates the legally hunted wolf population in Finland. Biological Conservation 215: 11–18. https://doi.org/10.1016/j.biocon.2017.08.031

Tallavaara R (2007) Joku järki pitäs petopolitiikkaan saaha: modernin metsästäjän susikuva postmodernissa tilanteessa [There should be some point in large carnivore policy: how does the modern hunter think about wolves in postmodern situation]. MsD Thesis, University of Jyväskylä, Finland, 91 pp. https://jyx.jyu.fi/bitstream/handle/123456789/38360/URN:NBN:fi:jyu-201208232227.pdf?sequence=1

The Finnish Wildlife Agency (2015) The Finnish Wildlife Agency. https://riista.fi/suden-kannanhoidollinen-metsastys-kaynnistyy-tammikuussa/ [Accessed on 22 August 2019]

Treves A, Jurewicz RR, Naughton-Treves L, Rose RA, Willging RC, Wydeven A (2002) Wolf Depredation on Domestic Animals in Wisconsin, 1976–2000. Wildlife Society Bulletin 30(1): 231–241. https://www.jstor.org/stable/3784658

von Essen E, Hansen HP, Källström HN, Peterson MN, Peterson TR (2014) Deconstructing the poaching phenomenon a review of typologies for understanding illegal hunting. British Journal of Criminology 29: 1481–1484. https://doi.org/10.1093/bjc/azu022

Wabakken P, Sand H, Kojola I, Zimmermann B, Arnemo JM, Pedersen HC, Liberg O (2007) Multistage, long-range natal dispersal by a global positioning system-collared Scandinavian wolf. The Journal of Wildlife Management 71(5): 1631–1634. https://doi.org/10.2193/2006-222

Wallach AD, Bekoff M, Nelson MP, Ramp D (2015) Promoting predators and compassionate conservation. Conservation Biology 29(5): 1481–1484. https://doi.org/10.1111/cobi.12525

Wikenros C (2011) The return of wolf. Effects on prey, competitors and scavengers. PhD Thesis, Svedish University of Agricultural Sciences, Uppsala, 50 pp. https://pub.epsilon.slu.se/8372/1/wikenros_c_111017.pdf

Wisconsin Department of Natural Resources (2020) Wisconsin Department of Natural Resources. https://dnr.wi.gov/topic/wildlifehabitat/wolf/dogdeps.html [Accessed on 6 April]

Wolch JR, Gullo A, Lassiter U (1997) Changing Attitudes toward California's Cougars. Society & Animals 5(2): 95–116. https://doi.org/10.1163/156853097X00015

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RESEARCH ARTICLE

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Illegal capture and internal trade of wild Asian elephants (*Elephas maximus*) in Sri Lanka

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Abstract
The illegal wildlife trade is considered one of the major threats to global biodiversity. Asian elephants (*Elephas maximus*) have been highly valued by various cultures for use in religious and spiritual contexts, as a draft animal, and more recently, as a tourist attraction. Thus, the demand for captive elephants is high. Wild Asian elephants are taken from the wild, often illegally, to maintain these captive populations due to the unviability of captive breeding programs. For the first time, we documented the extent to which wild elephants are being illegally captured and traded in Sri Lanka between January 2008 and December 2018. We collected data from case records maintained by the Sri Lanka court system where the suspects of illegal elephant trade were prosecuted in addition to information gathered by archives and interviews with various stakeholders. We documented 55 cases where elephants were illegally traded. This is probably an underestimate due to the mortality rate of elephants during capture operations, and challenges in collecting data on this highly organized illicit trade. Nearly equal numbers of male and female elephants were traded and more than 50% of them were juveniles, aged ≤5 years. Significantly more elephants were found to be seized in 2014–2015 than in the other time periods combined. We found evidence of the illegal capture of wild elephants from wildlife protected areas and state forests. More importantly, we

identified evidence of corruption of wildlife officers, involvement of politicians and other high-ranking personnel in the illegal wildlife trade, and lack of active enforcement of wildlife law as major challenges to overcome if the illegal capture and domestic trade of wild elephants in Sri Lanka are to be halted. Based on our study, we make a series of recommendations that should result in implementing policy to reduce the trafficking of Asian elephants in Sri Lanka and improve the conservation management of the species.

Keywords
Asian elephant, endangered species, illegal trade, national parks, wildlife crimes, wildlife trafficking

Introduction

The wildlife trade is one of the most profitable multi-billion dollar enterprises, involving direct exploitation of wild plants, animals, other organisms and their derivatives (Rosen and Smith 2010; South and Wyatt 2011; Sas-Rolfes et al. 2019). Wildlife trade operates at a global level, both legally with proper governmental and international licensure or through illicit means, and fundamentally involves trading animal and/or plant material for monetary gain or for exchange of goods and services (Wyatt 2013). Although large volumes of wildlife trade take place across international borders, a substantial amount, and perhaps most, of this trade happens within nations (TRAFFIC 2008; Zhang et al. 2008; Nijman 2010). Wildlife is traded for a variety of uses such as food, medicine, luxury goods, exotic pets, entertainment, spiritual, and laboratory use (Rosen and Smith 2010; Wyatt 2013; NRDC 2020). Possession of certain wildlife products may display social status, wealth, and affluence; the rarest species may incur great demand, high prices and high profit margins (Das 1990; Nijman and Shepherd 2010; Gunasekara 2011; South and Wyatt 2011; Tella and Hiraldo 2014). Thus, species susceptible to harvesting from the wild are pushed further towards the edge of extinction particularly when population growth cannot replenish the rate of harvests (Wilson 1988; Courchamp et al. 2006; Tella and Hiraldo 2014).

Wildlife trade is considered a major threat to global biodiversity (Nijman 2010). For some species, harvesting from the wild for trade is the primary cause of population decline and local extirpation (Rabemananjara et al. 2007; Nijman and Shepherd 2010; Gunasekara 2011; Tella and Hiraldo 2014) whereas in several other species, trade-based overexploitation has greatly stressed the predicament of habitat loss (Sutherland et al. 2009; WWF 2017). Additionally, the wildlife trade acts as a potential source for the spread of invasive species and disease-causing agents (Karesh et al. 2005). Much of the illegal trade in wildlife tends to be associated with a number of charismatic and/or high-profile species and elephants fall into the top-end of this category (Smith et al. 2009; Nijman 2010).

Throughout their centuries-long history across Asia, Asian elephants (*Elephas maximus*) have been revered and closely connected to, and highly valued by, various cultures for use in religious and spiritual contexts, as draft animals and, more recently, as a tourist attraction (De Silva and De Silva 2007; Fernando et al. 2011). As a keystone species, they also play an important role in maintaining the regional forest structure (Ishwaran 1993; Sukumar 2003; De Silva and De Silva 2007). The IUCN Asian Elephant Specialist Group (2019) estimated the global population size of Asian elephants to number 45,671–49,028 individuals. After India, the largest remaining population is found on the island of Sri Lanka; in 2019 this was estimated at ~5,900 individuals or about 13% of the global total (Fernando et al. 2019). At a global level, Asian elephants are listed as 'Endangered' in the IUCN Red List (Choudhury et al. 2008) and it receives this same listing in Sri Lanka (Ministry of Environment 2012).

While habitat loss and fragmentation have been historically considered the key driving forces of population decline of Asian elephants, in recent decades, human-wildlife conflict has intensified (Ishwaran 1993; Bandara and Tisdell 2003; Fernando et al. 2019). Human-elephant conflict is accentuated by a growing human population that is encroaching, degrading and fragmenting natural elephant habitats, and forcing elephants into closer contact with people (Fernando et al. 2005, 2019; Choudhury et al. 2008). A recent survey showed that elephants occupy over 60% of land and people are resident in 69% of the elephant range in Sri Lanka (Fernando et al. 2019). Furthermore, the human-elephant conflict in the country has increased markedly in intensity and its geographic extent, claiming 263 elephants per annum (the global highest annual elephant death rate) in 2010– 2019 (Prakash et al. 2020). Meanwhile, the live capture of wild elephants, previously used for labor and now increasingly for tourism, has played a key role in the decline in wild populations and is now considered a potentially significant threat to wild Asian elephants (Shepherd and Nijman 2008a; Nijman 2014; Schmidt-Burbach et al. 2015).

Throughout their natural range, the numbers of captive elephants are decreasing along with their role as draft animals, increasingly being replaced by machinery. However, their use in tourism is on the increase and may sustain the demand for captive elephants. Efforts to breed Asian elephants in captivity appear to be lagging behind with the noteworthy exception of the Pinnawala Elephant Orphanage in Sri Lanka (Fernando et al. 2011). A long gestation period, low birth rates in captivity, the purported abundance of wild populations, and the previous widespread availability of Asian elephants in the wild have hindered interest in captive breeding, hence the capture of wild individuals to maintain captive populations has long been preferred over breeding in captivity (Lair 1997; Leimgruber et al. 2011). Furthermore, the authorities' actions to stop the smuggling have been limited and ineffectual, so sustaining the illegal wild elephant trade in Sri Lanka (personal observations). As a result, captive populations are still largely maintained by wild captures, both legal and illegal, which is extremely detrimental to the conservation of this endangered species (Fernando and Pastorini 2011; Baker et al. 2013).

The illicit trade of wild-caught Asian elephants is prominent in several nations in Asia, particularly in Thailand, Myanmar, Laos, India, and Sri Lanka (Baskaran et al. 2011; Nijman 2013, 2014; Prakash 2014; Hankinson et al. 2020). Nijman (2014) reported that 79–81 wild elephants were illegally captured from the wild, mostly in Myanmar, and traded in Thailand over a two-year time period (April 2011–March 2013). In addition, in the 1990s, about 50–100 wild elephants were smuggled from

Myanmar every year (Lair 1997). Even though statistics are not available, the illegal captures of wild elephant calves to maintain captive populations have been reported from India (Baskaran et al. 2011).

Although appreciable numbers of studies have been conducted elsewhere in Asia, no comprehensive studies have been done on the illegal live wild elephant trade in Sri Lanka. In this study, we document the extent to which elephants were being illegally captured in the wild between January 2008 and December 2018, and present information on biometrics of smuggled elephants, an analysis of legal documentation process (and the violations) together with information on capturing and trafficking methods, source areas, trade routes, stakeholders involved, and the market value of live Asian elephants in Sri Lanka using best-available data. We expect this information to be used in implementing policy to reduce the trafficking of Asian elephants and conservation management of the species.

Methods

This study was conducted using both quantitative and qualitative data covering the entire country of Sri Lanka. The data for this study was mainly generated through case records maintained by the Sri Lanka courts system where the suspects of illegal elephant trade were prosecuted. Thirty-nine criminal proceedings filed before 15 Magistrate Courts by the Department of Wildlife Conservation of Sri Lanka (DWC) and Crime Investigation Division (CID) of Police Department of Sri Lanka were utilized in this study. A variety of other reliable sources were also used that documented the elephant trade in Sri Lanka. These include reports of three committees appointed by the line ministry of wildlife conservation between 2014 and 2018 to investigate the illegal live wild elephant trade, queries of information for clarification made by Auditor Generals' Department of Sri Lanka, and investigative reports and newspaper articles by environment activists and journalists. Interviews with 23 stakeholders (i.e. field DWC officers, investigative journalists, environmental activists, animal welfare activists, environmental lawyers, key informants, elephant owners, mahouts, and suspects) were also employed in data collection, specifically on major source areas for elephants, methods of live capture, transportation of captured elephants and trade routes.

We have deemed the elephants to be suspected smuggled elephants when legal proceedings were instituted or conducted by the authorized institutions against the respective suspects under the provisions of the Fauna and Flora Protection Ordinance No. 2 of 1937 and/or the Public Property Act No. 12 of 1982 with seizing the elephants (N = 39; Dissanayake 2016). Additionally, we included information on suspected smuggled elephants based on CID investigations, but the elephants were not yet seized (N = 5), and elephants disclosed by the DWC and environmental activists from urban areas (N = 3) and wilderness areas (N = 8; Dissanayake 2016). The elephants seized from wilderness areas were under restraints by the smuggling rackets at the time of seizure (see Table 1 and Fig. 1).

Table 1. Summary of cases utilized for this study.

Category	Total number of elephants	Remarks (N = number of cases)
The case is being heard	39	Smuggled elephant according to the Auditor General's Department observations (N = 3)
		Irregularities have been found in registration documents during the CID investigations (N = 10)
		Smuggled elephant according to the Auditor General's Department observations. Irregularities have been found in registration documents during the CID investigations (N = 11)
		No such information is available (N = 15)
Data on legal proceedings not available	11	Disclosed by the DWC from urban areas (N = 3) Seized by the DWC from wildernesses (N = 8)
CID investigations without seizing the elephant	5	Irregularities have been found in registration documents during the CID investigations (N = 1)
		Handed over to the Pinnawala Elephant Orphanage (N = 1)
		No such information is available (N = 3)
Total	55	

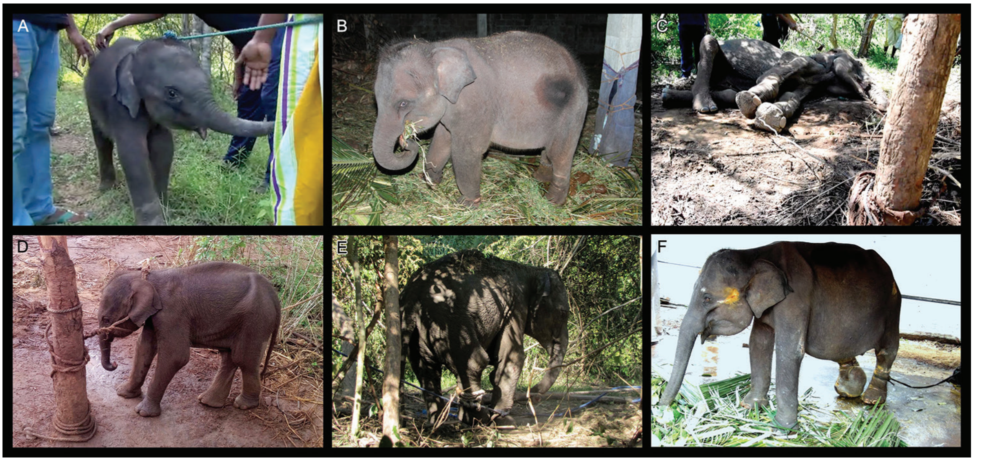

Figure 1. Some illegally captured Asian elephants (*Elephas maximus maximus*) in Sri Lanka: **A** a male, 2–3 years old juvenile (Gōnaganāra 2) seized in 2017 from Monaragala District **B** a male, 3–4 years old juvenile (Hamu) seized in 2014 from Gampaha District Nadika Hapuarachchi **C** a sub-adult, 6–7 years old (Gōnaganāra 1) seized in 2016 from Monaragala District **D** a female, less than 1 year old calf (Hambegamuwa) seized in 2016 from Monaragala District **E** unsexed, and unaged individual seized in 2014 from Kurunegala District **F** a male, 6–7 years old sub-adult (Sahayoga) seized in 2014 from Colombo District.

We used the average of estimated age range for analysis (e.g. estimated age of 3 or 4 years was calculated as 3.5 years). The life stages of elephants were categorized according to the following criteria: calves (≤1 year old), juveniles (2–5 years old), sub-adults (6–10 years old), and adults (≥11 years old). We analyzed temporal patterns in the trade in 2-year periods, beginning in 2008, using χ^2 tests, and differences between reported ages in the registration documents and estimated ages by veterinarians with a Mann-Whitney U test. We accepted significance when $P < 0.05$ in a two-tailed test (Zar 2010).

Results

Age and sex of suspected smuggled elephants

We found records of 55 cases of suspected smuggled elephants. Forty-six (83.6%) of those elephants had an identity reported with a name. Twenty-four (43.6%) elephants were females and 23 (41.8%) were males, while the sex of eight elephants (14.5%) was not reported. Two elephants (3.6%) were identified as calves (≤1 years old), 14 (25.4%) as juveniles (2–5 years old), 29 (52.7%) as sub-adults (6–10 years old), four (7.2%) as adults (≥11 years old) and six (10.9%) were not aged (Fig. 1). However, we identified a major gap between the reported age in the registration documents and the estimated age of elephants by the veterinarians of the DWC (Fig. 2). The average reported age in the registration documents of elephants (12.3 ± SE 1.0 yrs) according to licenses was higher than the estimated age by the veterinarians (6.9 ± SE 0.8 yrs); the difference is statistically significant ($z = 5.543$, $P < 0.0001$).

Legal analysis of suspected smuggled elephants

Only 33 (60.0%) elephants were registered while 17 (30.9%) elephants were not registered, three (5.4%) were under government letters of patent, and no information was available for two elephants (3.6%).

DWC and CID instituted 39 (70.9%) criminal proceedings before 15 Magistrate Courts against the suspects who were involved in the illegal live elephant trade in Sri Lanka. These suspected smuggled elephants were seized by the authorized institutions and kept at the Pinnawala Elephant Orphanage or Elephant Transit Home, Udawalawe. Three (7.7%) were considered smuggled elephants according to the Auditor General's department observation, 10 (25.6%) were considered smuggled elephants as irregularities were found in the registration documents during the CID investigations, and 11 (28.2%) were smuggled elephants according to the Auditor General's Department observation (with additional irregularities in registration documents as revealed by the CID investigations). Furthermore, CID was investigating five other cases, but elephants were not yet seized. Of these five, one is a smuggled elephant as irregularities have been found in registration documents during the CID investigations. One of these suspected smuggled elephants was voluntarily handed over to the Pinnawala Elephant Orphanage by the owner during the investigation (Table 1).

Three elephants were disclosed by the DWC and environmental activists from urban areas in Sri Lanka during the early phase of this study (Prakash 2014). These elephants were suspected as smuggled from the wild; however adequate information was not available. The remaining eight elephants were seized by the DWC from wilderness areas and found restrained by the smuggling rackets at the time of seizure (Fig. 1). These eight elephants were reported from Ruhuna (Yala) National Park (N = 2), Managed Elephant Range – Hambantota (N = 2), Galgamuwa, Maho, Weerawila, and Katagamuwa sanctuary (one in each location; Fig. 3). Moreover, six licenses were found

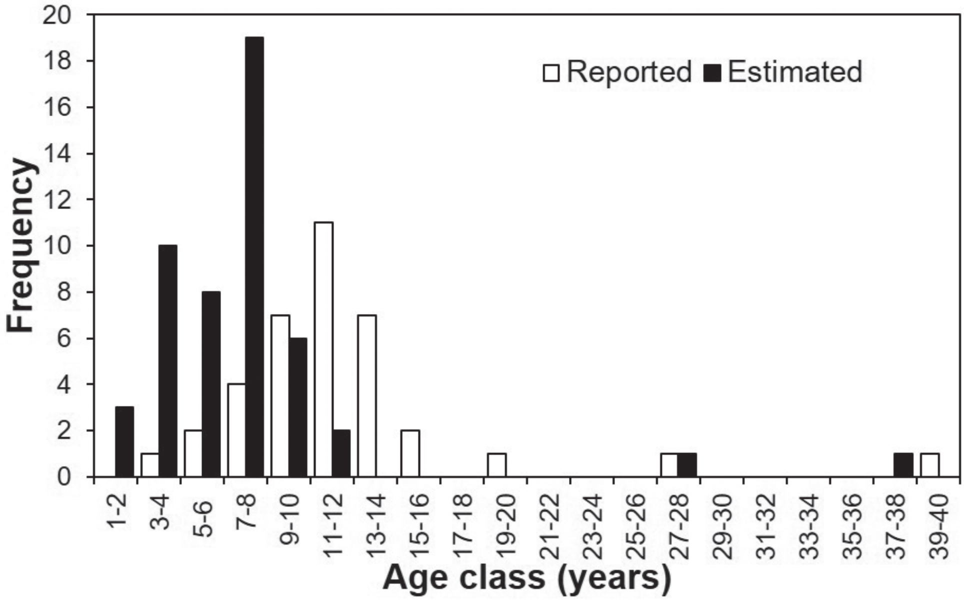

Figure 2. The distribution of reported age in the registration documents and estimated age by the veterinarians of the DWC of elephants studied.

without the existence of live elephants which may indicate plans for future smuggling activities. Therefore, it is clear that a minimum of 55 and possibly more elephants have been illegally captured from the wild in Sri Lanka during the period between January 2008 and December 2018 (Table 1, Figs 1, 3).

Perera (2015) reported that the last elephant birth in captivity was recorded in 1994. However, the report submitted to the Magistrate Court by the Director General of DWC on 09 July 2015 (DWC 2015) stated that 37 applications have been submitted for registration of elephant calves born in captivity during the period of 2000–2015. This raises a serious suspicion about the origin of these 37 elephant calves. Further, this report mentioned that DWC have registered 68 elephants under private ownership after the year 2009 (DWC 2015). Therefore, we find that at least 31 (i.e. 68 minus 37), and up to 68 elephants have been illegally captured from the wild. According to the same report, no elephant conception, birth, miscarriage or stillbirth in captivity was reported since 2009 in accordance with the 2009 amendment of the Fauna & Flora Protection Ordinance. Therefore, the last elephant registration in December 2014 as per the same report is also problematic because the average gestation period of an Asian elephant is 22 months (Lueders et al. 2012).

The discrepancies between the reported age in the registration documents and the estimated age of elephants by the veterinarians of the DWC suggest some irregularities in the elephant registration process. We found information about potential corruption at DWC from reports of committees appointed by the line ministry of wildlife conservation between 2014 and 2018 and queries of information for clarification made by

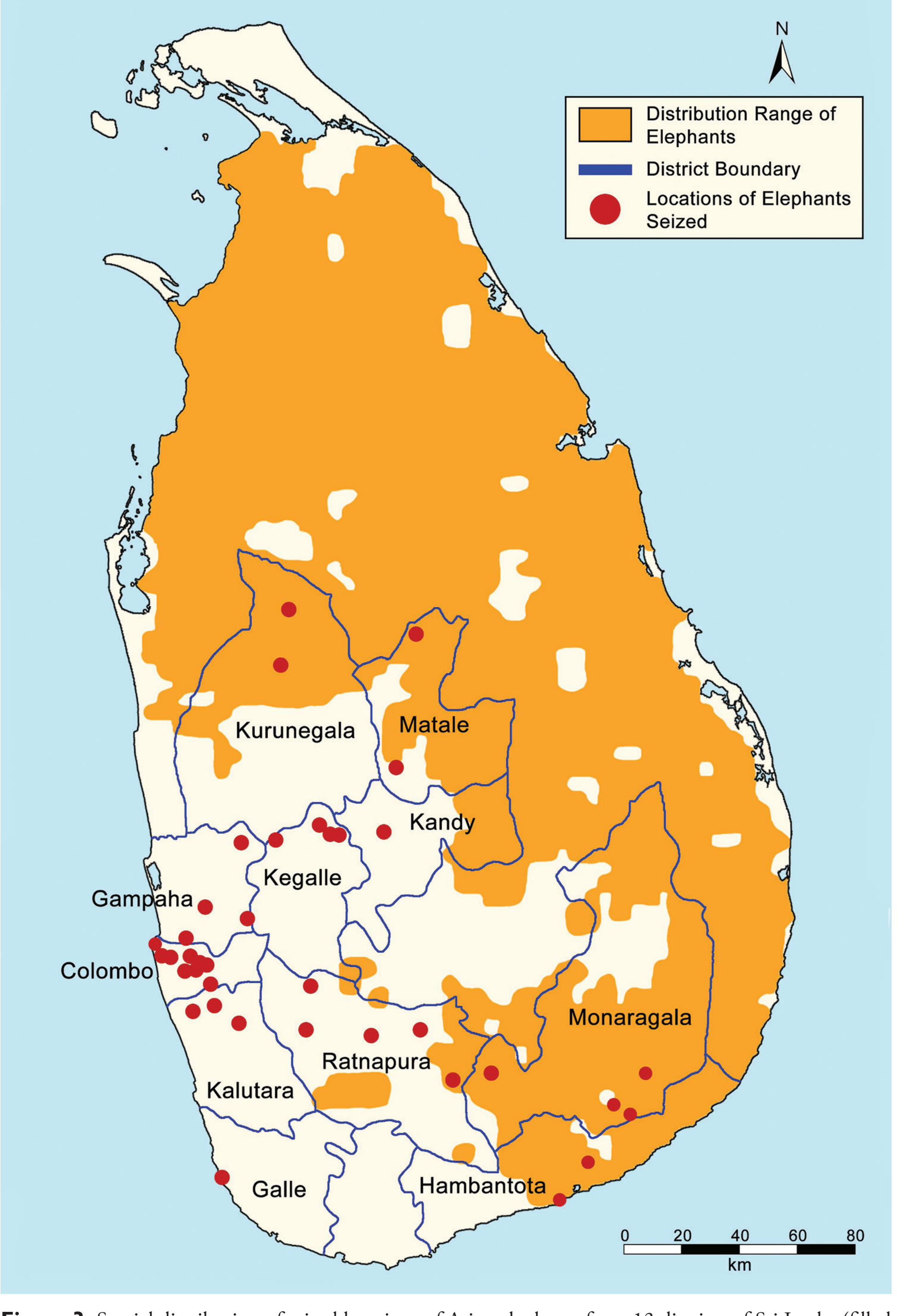

Figure 3. Spatial distribution of seized locations of Asian elephants from 13 districts of Sri Lanka (filled circles) overlaid on the distribution of wild Asian elephants in Sri Lanka (orange shade; after Fernando et al. 2019).

Auditor Generals' Department of Sri Lanka to investigate the illegal live wild elephant trade (see Auditor General's Department 2014; Dissanayake 2016). This included the fraudulent registration of elephants by tendering false and forged documents regarding the birth of elephant calves and matters in respect to the mother elephants, entering false minutes in files and altering entries and replacing photographs in older files. Original entries were erased and new entries were typed over these erasures. Photographs of elephants in the older files have been removed and these were replaced by new photographs of other elephants. To facilitate the fraudulent registrations and issues of licensing, the corrupted officers at DWC have maintained files without elephants and files where elephants have been reported dead without closing or revoking the said files.

Temporal and spatial trends of illegal capture of wild elephants

Elephant seizures were reported from eleven administrative districts of Sri Lanka with the highest number of seizures being reported from Colombo district (16 cases; 31.4%; Fig. 3). The highest number of seizures during the study period occurred in 2015 (20 cases; 36.4%), followed by 2016 (14 cases; 25.4%) and 2014 (11 cases; 20.0%). The number of cases we recorded were not homogenously distributed over the six 2-year time windows ($\chi^2 = 94.2$, df = 5, $P < 0.0001$). Significantly more elephants were seized in 2014–2015 than in the other time periods combined ($\chi^2 = 75.3$, df = 2, $P < 0.0001$). Conversely, the periods 2010–2011 and 2012–2013, for which we found no evidence of elephant seizures, contained significantly fewer cases than in the other time periods combined ($\chi^2 = 7.88$, df = 1, $P = 0.005$).

Through interviews with field DWC officers and investigative journalists, we found direct photographic evidence (Fig. 1) for the illegal capture of wild elephants from wildlife protected areas and state forests including Ruhuna (Yala) National Park, Udawalawe National Park, and Katagamuwa Sanctuary and state forests in Managed Elephant Range – Hambantota, Galgamuwa, Maho, and Weeravila. We also suspect that wild elephants were illegally captured from Minneriya National Park, Kaudulla National Park, Ritigala Strict Nature Reserve, Sigiriya Sanctuary and Hurulu Eco Park, based on interviews conducted with the field DWC officers, investigative journalists, and environmental activists. Additionally, our interviews revealed that culprits have used various methods to capture live elephants from the wild, including capturing young elephants by either killing or sedating the maternal elephant by shooting or using tranquilizing guns, injecting tranquilizers into young elephants, pit-fall trapping, and noosing. It is also suspected that culprits have traded elephants in rehabilitation under the authority of the DWC at Elephant Transit Home (ETH) in Udawalawe, with or without the knowledge of the resident Wildlife Conservation Officers. Furthermore, interviews with the suspects revealed that young elephants orphaned as a result of human-elephant conflicts have also been subjected to trade. Usually, when there is an orphaned young elephant, the general public informs the DWC about it and ETH rescues and rehabilitates these young elephants. However, in some incidences, some criminal enterprises may have intercepted the information flow and abducted the young elephants before DWC and/or ETH reached the orphaned elephant.

Reports of committees appointed by the line ministry of wildlife conservation between 2014 and 2018 to investigate the illegal live wild elephant trade (see Dissanayake 2016) and interviews conducted with field DWC officers, environmental activists, and local journalists provided evidence of the transportation of elephant from the wild by jeeps and vans with tinted windows. This somewhat mimics a motorcade of political elites, so preventing scrutiny by authorized officers and also distracting public attention. Container trucks were also used for calves and juveniles. The trade routes from source areas to detentions could even be public roads as smugglers get off scot-free through these aforementioned methods. Famous people in society, including businessmen, politicians, Buddhist monks, high-ranking government officers, magistrates, tourism entrepreneurs and armed forces personnel, have been suspected participants in the live elephant trade in Sri Lanka (see Auditor General's Department 2013; Dissanayake 2016). This is a very lucrative business and a single elephant can be sold for between 7.5 and 12.5 million Sri Lanka Rupees in 2018 (~USD 40,500–67,500; findings during this study).

Discussion

Our study has shown that the illegal wild elephant trade is a major challenge for the conservation and management of endangered Asian elephants in Sri Lanka. We found at least 55 cases of illegally captured elephants from the wild in Sri Lanka during the period of January 2008–December 2018. Although it is still only about 0.1% of total Sri Lankan wild population / year (55 cases over 10 years equals 5.5 cases per year), we want to stress that our number may represent an underestimate due to two major reasons; the secretive operations of this illegal trade in Sri Lanka, and the unreported mortality rate of elephants during the capturing process, transport and in captivity. Although our data was largely based on anecdotal reports (i.e. court reports, investigation reports, media reports, and stakeholder interviews), there is a consistency between sources that gives our claims a measure of legitimacy. More importantly, we have used the best available information from multiple sources to dissect the illegal live elephant trade in Sri Lanka.

The percentage of calves and juveniles in illegal trade (29.0%) is higher than in wild populations in Sri Lanka (17.8%; Perera 2015), and similar to that found by Nijman (2014) for illegally traded elephants in Thailand (i.e. 17/79, 21.5%). Unlike Thailand, we found an equal number of males and females, whereas in Thailand 80% were female (Nijman 2014). Without an established captive breeding program in Sri Lanka, the higher percentage of calves and juveniles in captivity suggest an input from wild populations providing further evidence of elephant smuggling.

The actual number of captures of elephants from the wild could be higher due to the high mortality rate in captivity under illicit trade. Even at ETH, the mortality rate of arrivals is around 40% where an intensive care facility with close monitoring by resident veterinary surgeons and trained staff of DWC is maintained for orphaned elephants until they are fit enough to be released back into the wild (De Silva and De Silva 2007). Wildlife traffickers are unlikely to provide any veterinary care for illegally captured elephants. Additionally, the licenses maintained by the suspects without elephants show a range of ages from 1 year to 20 years. We suspect that they may have held these licenses with the expectation of smuggling elephants from the wild in the future. Additionally, the reported age of elephants based on registration documents is significantly higher than the estimated age by veterinarians, suggesting misconduct in registrations.

Smith et al. (2015) argued that corruption is one of the significant challenges in the conservation of elephants, while Milman (2013) identified the role that park staff, enforcement officers and politicians have been playing in the illegal trade of elephants. This chain of corruption is operated through a system of bribery, which may weaken efforts to combat the illegal elephant trade, and enforce wildlife conservation law (Barnes et al. 1995). Our findings suggest a similar chain of corrupt personnel behind the illegal wild elephant trade in Sri Lanka. The registration of wild-caught elephants falsely claiming that they had been born in captivity is very challenging due to the amended 2009 Act No. 22 of the Fauna & Flora Protection Ordinance. According to the amended act, in the event of a pregnancy of a captive female elephant, the owner of such an elephant has to report this to the Director General of the DWC including information such as the details of the sire, an uneventful death of the mother elephant during birth etc. However, elephant smugglers and corrupt officers have not been deterred in their activities by the above amendment. They had adopted devious methods like falsely backdating their applications and tendering fraudulent documents and affidavits containing false material to obtain registrations. Further, the minutes of the files had also been altered. The applications made in 2012, 2013, and 2014 were backdated to dates in 2008 with the involvement of corrupt officers of the DWC. This may explain the reason behind the significant difference between the reported age of elephants based on registration documents and the age estimated by veterinarians.

Wild elephants are often caught in Myanmar using pit fall traps where they are corralled into pits with the aid of captive elephants (Nijman 2014). Researchers have reported – although this has not been necessarily verified – that automatic weapons are increasingly being used to kill protective members of the herds in Myanmar and Thailand (Nijman 2014). In Sri Lanka, smugglers have used much more sophisticated methods compared to Myanmar and Thailand to capture elephants such as sedating the maternal elephants using tranquilizing guns and injecting tranquilizers into the young elephants. Meanwhile, automatic weapons are also used to kill protective members of the herds. As live young elephants are prized higher than adults in Myanmar and Thailand (Nijman 2014), the same market trend can be anticipated in Sri Lanka.

Legal protection for wild Asian elephants in Sri Lanka

Elephants are mainly protected by three acts of legislation and two other acts of legislation can also be utilized in combating the illegal live wild elephant trade in Sri Lanka, the authority for which has been delegated to two government Institutions (Table 2). There is no legal provision given to any private entity to capture elephants from the wild for any reason in any circumstances without prior permission granted by the Director General of DWC. The constitution of the Democratic Socialist Republic of Sri Lanka itself provides for protection for nature. According to Article 27(14) of the constitution, the state shall protect, preserve, and improve the environment for the benefit of the community. Wild elephants are a part of the nature; therefore, the state bears the responsibility of protecting them. Other than that, Prevention of Cruelty to Animals Ordinance No.13 of 1907 and Fauna and Flora Protection Ordinance No. 2 of 1937 (FFPO) provide more legislative provisions for combating illegal live wild elephant smuggling in the country.

The most relevant law is the FFPO as amended by Act No. 49 of 1993 and Act No. 22 of 2009 which deals with both wild and domestic elephants. DWC was established and charged with management and implementation of the Fauna and Flora Protection (Amendment) Act, No.22 of 2009. In view of provisions in Section 22A (12) of the last amendment in 2009 any elephant which has not been registered under the provision of the FFPO shall be presumed to have been taken or removed from the wild without lawful authority or approval and such elephants shall be deemed to be public property. The provisions of the Offences Against Public Property Act, No. 12 of 1982 shall accordingly apply in respect of such elephants. Furthermore, any offence committed under the Act involving an elephant shall be a non-bailable offense and the provisions of the Bail Act, No. 30 of 1997 and the Code of Criminal Procedure Act, No. 15 of 1979 shall apply in respect of such an offense (DWC 2017).

Prevention of Cruelty to Animals Ordinance No.13 of 1907 make better provision for the prevention of cruelty to all animals and these provisions can be utilized to protect elephants found in captivity suffering pain by reason of starvation, mutilation, or other ill-treatment. The Penal Code (Amendment) Act No. 29 of 1998 is also important as there is increasing evidence of false documents being used to smuggle elephants and get licenses.

In addition to national protection, international protection against live elephant trade is afforded by The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) which is a multinational agreement to which countries voluntarily join. The aim of CITES is to ensure that legal international trade does not endanger the survival of wild plants or animals. A permit system is the primary mechanism by which wildlife trade is regulated through CITES. Asian Elephants have been listed in Appendix I of CITES since its inception in 1975, generally prohibiting international trade in wild individuals and their derivatives except in exceptional circumstances. Article III.3.c of CITES allows trade in an Appendix I species if "a Management Authority of the State of import is satisfied that the specimen is not to be used for primarily commercial purposes". Therefore, the international trade of elephants does occur on a small but regular basis in the zoo trade. Sri Lanka has been a member of CITES since 1979 (CITES 2014). However, transactions of live wild elephant trade in Sri Lanka appear not to have included cross border trade and therefore CITES is unable to be used as an effective mechanism to curb this type of trade.

Table 2. Legislations and authorized government institutions related to Asian elephant conservation in Sri Lanka.

Legislation	Authorized Institutions/Persons
The constitution of the Democratic Socialist Republic of Sri Lanka	General Public (under fundamental right jurisdiction)
Prevention of Cruelty to Animals Ordinance No.13 of 1907	Department of Police
Fauna and Flora Protection Ordinance No. 2 of 1937	DWC Department of Police General Public (under section 60E)
Offences committed against Public property Act No. 12 of 1982	Department of Police
Penal Code (Amendment) Act No. 29 of 1998	Department of Police

The corruption behind the illegal wild elephant trade in Sri Lanka

Government officials responsible for elephant conservation in Sri Lanka, under the influence of political pressure or influenced by bribery, either avoid making an honest attempt to combat this illegal trade of wild elephants or offer only limited resistance. On several occasions, attempts were made to release the suspected smuggled elephants to the offenders with only minor penalties. For example, the Director General of DWC was pressurized by high ranking politicians to release the elephants using loopholes in the legislation and even cabinet memoranda were presented (i.e. Cabinet memorandum number 16/2204/708/017-1 and dated 04th July 2016 forwarded by the Minister of Sustainable Development and Wildlife and cabinet memorandum number PMO/CM/44/2019 dated 10th September 2019 forwarded by the Prime Minister and two Ministers), to release the elephants to the offenders.

The climax of these organized wildlife crimes in Sri Lanka can arguably be considered to have been the August 2013 misplacement of the register of captive elephants archived at the head office of DWC. As a result, the DWC and the Police Department of Sri Lanka have launched several investigations into elephant smuggling since 2014. This has also led to protests by the general public and environmental activists against this illegal trade.

The past and the future of captive elephants in Sri Lanka

Historically, the captive elephant population in Sri Lanka mainly depended on wild captures and noosing and kraaling were widely used for capturing free ranging elephants (Elapatha 1997). However, the emotional outcry following an unfortunate incident of an elephant being killed in Panamure Kraal in Sabaragamuwa Province of Sri Lanka led to the ban on elephant captures by private individuals in the country in 1950 (Elapatha 1997; Katugaha 2008). However, private captures were again allowed in 1972–1974 and a few elephants that were captured by DWC were transferred to various parties (e.g. high-profile personnel, clergy etc.) until the 1980s. Again, this action was prohibited due to the rapidly declining number of elephants in the wild (Fernando et al. 2011). A national policy for the conservation and management of wild elephants was developed in 2006, which proposed to domesticate problematic elephants to satisfy demands for captive elephants, but this policy has not yet been implemented effectively. Furthermore, fearing post-reproduction wearing of females, captive elephant owners do not permit captive breeding. Besides, it takes well over 10 years for newborns to serve as draft animals; thus calves do not help to generate any income for the elephant owners. Therefore, continuing demands for captive elephants as draft animals are fulfilled by illegally capturing from the wild.

Meanwhile surplus demand for elephants has been created since the end of the civil war in Sri Lanka in 2009. The emergence of peace has enabled the public to engage more in religious and cultural activities. Elephants hold a central position in the country's two main religions, Buddhism and Hinduism (Wisumperuma 2004). Therefore, demand increased to domesticate elephants to provide for religious and cultural festivals. The demand for live elephants had also increased tremendously among the emerging class of new rich businessmen because elephants are symbolic statements of physical and mental strength, intelligence, responsibility, and prosperity (Fernando et al. 2011).

Corruption, ineffective laws, weak judicial systems, lack of enforcement of wildlife law and light sentences allow criminal networks to keep plundering wildlife with little regard for the consequences (De Silva and De Silva 2007; Shepherd and Nijman 2008b; Nijman 2010). These factors make the illegal wildlife trade a low-risk business with high returns. The masterminds behind illicit wildlife trade operations are not penalized by the law enforcement. Instead, local poachers are usually the only ones caught, leaving the real culprits operational and capable of striking again (WWF 2017), and the situation is more or less similar in Sri Lanka. At the same time, smugglers use Buddhist culture as a Trojan horse and the influence of certain religious leaders is tapped to manipulate investigations into the live elephant trade and related policy decisions in Sri Lanka (Prakash 2014).

Domestic and wild elephants in Sri Lanka are treated under the same legislation, the Fauna and Flora Protection Ordinance No. 2 of 1937 (FFPO), and this is an undoubted advantage for combating the illegal live elephant trade in the country. The population of wild Asian elephants in Thailand is estimated at between 3126–3341, and about 3783 individuals belong to the Thai domestic elephant population (Asian Elephant Specialist Group 2019). Compared to Thailand, the percentage of elephant population under captivity in Sri Lanka is very low and has historically been declining (Fernando et. al. 2011), making it relatively easy to control such illegal trade in Sri Lanka.

In terms of impact, habitat loss associated with smuggling leads to extirpation of elephants from certain home ranges. The Sri Lankan illegal live elephant trade specifically targeted males. This can skew the sex ratio of a population toward a female bias and reduce genetic variability, fecundity and recruitment (Sukumar 2003). Several studies have also found that such targeted extractions interfere with the herd's complex social structure and can cause long-lasting psychological effects on individuals (Bradshaw et al. 2005; Ishengoma et al. 2008; Archie and Chiyo 2012) as they have strong social networks (see Perera 2015).

Recommendations for improved conservation and management

Based on our study, we make a series of recommendations that should result in implementing policy to reduce the trafficking of Asian elephants in Sri Lanka and to improve the conservation management of the species.

In the short term, we urge the relevant authorities and government bodies to speed up the judiciary process against suspects and penalize the offenders who smuggled the elephants from the wild for trade purposes. It goes without saying that this should happen in a fair and just manner, irrespective of the suspect's social status, political affiliation or role in society.

If officers of the DWC are either directly or indirectly involved in the trade, including by assisting the smuggling rackets, immediate legal and/or disciplinary action should be taken. Any measures taken should be made public in order to deter those tempted by this illegal act in the future.

In the intermediate term, we urge that funds, expertise and time be made available to assist the Elephant Transit Home in the Udawalawe National Park and Pinnawala Elephant Orphanage with the smuggled elephants under their care. This assessment will help to determine whether these elephants are fit enough to be released back to the wild.

Still in the intermediate term, we urge the enactment of a national policy on captive elephants which introduces a scientific and transparent process regarding the registration and renewal of licenses to hold captive elephants. This should lead to a limit on the use of captive elephants for cultural, religious, and tourism purposes. As part of this, we urge the authorities responsible for the welfare and conservation of Asian elephants in Sri Lanka to adopt the standardized captive elephant registration protocols and best practices proposed by the Seventeenth Conference of Parties of CITES in 2016 and the second Asian Elephant Range States' meeting in 2017. These guidelines include DNA registration, monitoring protocols for captive populations, guidelines for the management and welfare of captive elephants, disease management including zoonotic diseases, training and capacity building of staff and mahouts, and specific national policy to manage the captive elephant population to avoid illicit live elephant trade (Sakamoto 2017). These new protocols may discourage the malpractices associated with the illegal trade of wild Asian elephants and secure the welfare of captive Asian elephants in Sri Lanka.

Conclusion

Our study, for the first time, has provided the best available information regarding the extent, mechanisms and the potential impacts of live wild Asian elephant smuggling in Sri Lanka. Although the numbers of smuggled elephants are relatively low compared to neighboring counties, it is very clear that smugglers have been using sophisticated methods and operate under strong networks involving corrupt wildlife officers, politicians, clergymen and even military personnel. Despite the availability of sufficient local legislation to stop these illicit activities and protect endangered Asian elephants from smuggling, the lack of active enforcement of wildlife law is hindering the progress of conservation of wild elephants in Sri Lanka.

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Reference

Archie EA, Chiyo PI (2012) Elephant behaviour and conservation: Social relationships, the effects of poaching, and genetic tools for management. Molecular Ecology 21(3): 765–778. https://doi.org/10.1111/j.1365-294X.2011.05237.x

Asian Elephant Specialist Group (2019) The Asian Elephant. https://www.asesg.org [Last accessed 25 September 2020]

Auditor General's Department (2013) Inspection of illegally captured baby elephants from the wild (Text in Sinhala Language). (TM/EL/WL/A.B/2013/V:V:02). Central Environment Authority of Sri Lanka.

Auditor General's Department (2014) Inspection of illegally captured baby elephants from the wild (Text in Sinhala Language). (TM/EL/WL/A.B/2013/V:V:02). Auditor General's Department of Sri Lanka.

Baker SE, Cain R, van Kesteren F, Zommers ZA, D'Cruze N, Mac Donald 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

Bandara R, Tisdell C (2003) Comparison of rural and urban attitudes to the conservation of Asian elephants in Sri Lanka: Empirical evidence. Biological Conservation 110(3): 327–342. https://doi.org/10.1016/S0006-3207(02)00241-0

Barnes RFW, Blom A, Alers MPT (1995) A review of the status of forest elephants *Loxodonta africana* in Central Africa. Biological Conservation 71(2): 125–132. https://doi.org/10.1016/0006-3207(94)00014-H

Baskaran N, Varma S, Sar CK, Sukumar R (2011) Current status of Asian elephants in India. Gajah 35: 47–54.

Bradshaw GA, Schore AN, Brown JL, Poole JH, Moss CJ (2005) Elephant breakdown. Nature 433(7028): 807–807. https://doi.org/10.1038/433807a

Choudhury A, Lahiri Choudhury DK, Desai A, Duckworth JW, Easa PS, Johnsingh AJT, Fernando P, Hedges S, Gunawardena M, Kurt F, Karanth U, Lister A, Menon V, Riddle H, Rübel A, Wikramanayake E (2008) *Elephas maximus*. The IUCN Red List of Threatened Species 2008. https://www.iucnredlist.org/species/7140/12828813 [Last accessed 01 June 2018]

CITES (2014) Sri Lanka. https://www.cites.org [Last accessed 08 September 2020]

Courchamp F, Angulo E, Rivalan P, Hall RJ, Signoret L, Bull L, Meinard Y (2006) Rarity Value and Species Extinction: The Anthropogenic Allee Effect. PLoS Biology 4(12): e415. https://doi.org/10.1371/journal.pbio.0040415

Das I (1990) The trade in freshwater turtles from Bangladesh. Oryx 24(3): 163–166. https://doi.org/10.1017/S0030605300033895

De Silva M, De Silva PK (2007) The Sri Lankan Elephant: Its Evolution, Ecology and Conservation. WHT Publications, Colombo, 278 pp.

Dissanayake NE (2016) The report of the one-member committee appointed to investigate into whether any act of corruption or fraud had been committed by the officers of the Department of Wildlife Conservation in registering and issue of annual licenses contrary to the provisions of the flora and fauna ordinance and further to investigate into whether any irregularities had been committed in the issue of "CITES permits" (Text in Sinhala Language). Ministry of Sustainable Development and Wildlife Conservation, Battaramulla, 133 pp.

DWC (2015) Report forwarded under Sections 115(1), (2), (3), (4), 120(1), (3) of chapter number XI of the Code Of Criminal Procedure Act, No. 15 of 1979 (Text in Sinhala Language). (2968/15). Department of Wildlife Conservation of Sri Lanka.

DWC (2017) Fauna and Flora Protection Ordinance. http://www.dwc.gov.lk [Last accessed 08 September 2020]

Elapatha Jr S (1997) Elephant kraals at Panamure – A historical record. Loris 21: 118–125.

Fernando P, Pastorini J (2011) Range-wide status of Asian elephants. Gajah 35: 15–20.

Fernando P, Wickramanayake E, Weerakoon D, Jayasinghe LKA, Gunawardene M, Janaka HK (2005) Perceptions and patterns in human-elephant conflict in old and new settlements in Sri Lanka: Insights for mitigation and management. Biodiversity and Conservation 14(10): 2465–2481. https://doi.org/10.1007/s10531-004-0216-z

Fernando P, Jayewardene J, Prasad T, Hendavitharana W, Pastorini J (2011) Current status of Asian elephants in Sri Lanka. Gajah 35: 93–103.

Fernando P, De Silva MKCR, Jayasinghe LKA, Janaka HK, Pastorini J (2019) First country-wide survey of the Endangered Asian elephant: Towards better conservation and management in Sri Lanka. Oryx: 1–10. https://doi.org/10.1017/S0030605318001254

Gunasekara S (2011) Export trade of indigenous freshwater fishes of Sri Lanka. Samantha Gunasekara, Colombo, 126 pp.

Hankinson E, Nijman V, Abdullah (2020) Asian elephants: 15 years of research and conservation. Journal of Physics: Conference Series 1460: 1–5. https://doi.org/10.1088/1742-6596/1460/1/012055

Ishengoma DRS, Shedlock M, Foley CH, Foley LJ, Wasser SK, Balthazary ST, Mutayoba BM (2008) Effects of poaching on bull mating success in a free ranging African elephant (*Loxodonta africana*) population in Tarangire National Park, Tanzania. Conservation Genetics 9(2): 247–255. https://doi.org/10.1007/s10592-007-9332-0

Ishwaran N (1993) Ecology of the Asian elephant in lowland dry zone habitats of the Mahaweli River Basin, Sri Lanka. Journal of Tropical Ecology 9(2): 169–182. https://doi.org/10.1017/S0266467400007148

Karesh WB, Cook RA, Bennett EL, Newcomb J (2005) Wildlife trade and global disease emergence. Emerging Infectious Diseases 11(7): 1000–1002. https://doi.org/10.3201/eid1107.050194

Katugaha HIE (2008) The Last Kraal in Sri Lanka. Gajah 29: 5–10.

Lair R (1997) Gone Astray: The Care and Management of the Asian Elephant in Domesticity. http://www.fao.org/3/ac774e/ac774e00.htm [Last accessed 28 June 2019]

Leimgruber P, Oo ZM, Aung M, Kelly DS, Wemmer C, Senior B (2011) Current status of Asian Elephants in Myanmar. Gajah 35: 76–86.

Lueders I, Niemuller C, Rich P, Gray C, Hermes R, Goeritz F, Hildebrandt TB (2012) Gestating for 22 months: Luteal development and pregnancy maintenance in elephants. Proceedings. Biological Sciences 279(1743): 3687–3696. https://doi.org/10.1098/rspb.2012.1038

Milman O (2013) Ranger corruption 'impeding global fight against poaching'. The Guardian 27 March. http://www.theguardian.com/environment/2013/mar/27/ranger-corruption-impeding-fight-poaching [Last accessed 28 July 2019]

Ministry of Environment (2012) The National Red List 2012 of Sri Lanka; Conservation Status of the Fauna and Flora. Ministry of Environment, Colombo, 452 pp.

Nijman V (2010) An overview of international wildlife trade from Southeast Asia. Biodiversity and Conservation 19(4): 1101–1114.

Nijman V (2013) Asian elephant loans from Lao PDR to Japan. Traffic Bulletin 25(2): 46–46. https://www.traffic.org/site/assets/files/3002/traffic_pub_bulletin_25_2_elephant_loans.pdf

Nijman V (2014) An assessment of the live elephant trade in Thailand. TRAFFIC International, Cambridge, 38 pp.

Nijman V, Shepherd CR (2010) The role of Asia in the global trade in CITES II-listed poison arrow frogs: Hopping from Kazakhstan to Lebanon to Thailand and beyond. Biodiversity and Conservation 19(7): 1963–1970. https://doi.org/10.1007/s10531-010-9814-0

NRDC (2020) Wildlife trade. https://www.nrdc.org [Last accessed 25 September 2020]

Perera BV (2015) Living with Wild Elephants: Experiences of a wildlife Veterinarian in Sri Lanka. BV Perera, Panadura, 253 pp.

Prakash TGSL (2014) Life with Elephants (Text in Sinhala Language). Center for Conservation and Research, Rajagiriya, 240 pp.

Prakash TGSL, Wijeratne AW, Fernando P (2020) Human-Elephant Conflict in Sri Lanka: Patterns and Extent. Gajah 51: 16–25.

Rosen GE, Smith KF (2010) Summarizing the Evidence on the International Trade in Illegal Wildlife. EcoHealth 7(1): 24–32. https://doi.org/10.1007/s10393-010-0317-y

Sakamoto M (2017) Recent topics on CITES related to Asian Elephants in particular. Gajah 47: 42–44.

Sas-Rolfes M, Challender DW, Hinsley A, Veríssimo D, Milner-Gulland EJ (2019) Illegal wildlife trade: Scale, processes, and governance. Annual Review of Environment and Resources 44(1): 201–228. https://doi.org/10.1146/annurev-environ-101718-033253

Schmidt-Burbach J, Ronfot D, Srisangiam R (2015) Asian elephant (*Elephas maximus*), pig-tailed macaque (*Macaca nemestrina*) and tiger (*Panthera tigris*) populations at tourism venues in Thailand and aspects of their welfare. PLoS One 10(9): e0139092. https://doi.org/10.1371/journal.pone.0139092

Shepherd CR, Nijman V (2008a) Elephant and Ivory Trade in Myanmar. TRAFFIC Southeast Asia, Selangor, 32 pp.

Shepherd CR, Nijman V (2008b) An overview of the regulation of the freshwater turtle and tortoise pet trade in Jakarta, Indonesia. TRAFFIC Southeast Asia, Petaling Jaya, 24 pp.

Smith KF, Behrens M, Schloegel LM, Marano N, Burgiel S, Daszak P (2009) Reducing the risks of the wildlife trade. Science 324(5927): 594–595. https://doi.org/10.1126/science.1174460

Smith RJ, Biggs D, St John FAV, 't Sas-Rolfes M, Barrington R (2015) Elephant conservation and corruption beyond the ivory trade. Conservation Biology 29(3): 953–956. https://doi.org/10.1111/cobi.12488

South N, Wyatt T (2011) Comparing illicit trades in wildlife and drugs: An exploratory study. Deviant Behavior 32(6): 538–561. https://doi.org/10.1080/01639625.2010.483162

Sukumar R (2003) The Living Elephants: Evolutionary Ecology, Behavior, and Conservation. Oxford University Press, Oxford, 478 pp.

Sutherland WJ, Adams WM, Aronson RB, Aveling R, Blackburn TM, Broad S, Ceballos G, Côté IM, Cowling RM, Da Fonseca GAB, Dinerstein E, Ferraro PJ, Fleishman E, Gascon C, Hunter Jr M, Hutton J, Kareiva P, Kuria A, Macdonald DW, Mackinnon K, Madgwick FJ, Mascia MB, Mcneely J, Milner-Gulland EJ, Moon S, Morley CG, Nelson S, Osborn D, Pai M, Parsons ECM, Peck LS, Possingham H, Prior SV, Pullin AS, Rands MRW, Ranganathan J, Redford KH, Rodriguez JP, Seymour F, Sobel J, Sodhi NS, Stott A, Vance-Borland K, Watkinson AR (2009) One hundred questions of importance to the conservation of global biological diversity. Conservation Biology 23(3): 557–567. https://doi.org/10.1111/j.1523-1739.2009.01212.x

Tella JL, Hiraldo F (2014) Illegal and Legal Parrot Trade Shows a Long-Term, Cross-Cultural Preference for the Most Attractive Species Increasing Their Risk of Extinction. PLoS One 9(9): e107546. https://doi.org/10.1371/journal.pone.0107546

TRAFFIC (2008) Wildlife trade: What is it? http://www.traffic.org [Last accessed 05 July 2019]

Wilson EO (1988) The current state of biological diversity. In: Wilson EO, Peter FM (Eds) Biodiversity. National Academy Press, Washington, 3–18.

Wisumperuma D (2004) Human-elephant relationships in Sri Lanka: An historical and archaeological perspective. In: Endangered Elephants, Past Present and Future. Biodiversity and Elephant Conservation Trust, Rajagiriya, 6–12.

WWF (2017) Illegal wildlife trade. http://wwf.panda.org [Last accessed 05 March 2018]

Wyatt T (2013) Wildlife trafficking: A deconstruction of the crime, the victims, and the offenders. Palgrave Macmillan, London, 215 pp. https://doi.org/10.1057/9781137269249

Zar JH (2010) Biostatistical Analysis (5th edn). Prentice-Hall/Pearson, Upper Saddle River, 944 pp.

Zhang L, Hua N, Sun S (2008) Wildlife trade, consumption and conservation awareness in southwest China. Biodiversity and Conservation 17(6): 1493–1516. https://doi.org/10.1007/s10531-008-9358-8

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RESEARCH ARTICLE

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Number and distribution of large old ginkgos in east China: Implications for regional conservation

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Abstract

Large old ginkgos (LOGs), having important ecological, cultural and historical values, are widely distributed in China. However, little is known regarding their quantity and tree-habitat quality in the mesoscale distribution. Here, the quantity, spatial distribution and conservation status of *Ginkgo biloba* L. older than 100 years in Jiangsu Province, east China were examined using ArcGIS software and detrended correspondence analysis (DCA). Based on our collated data, Jiangsu Province included 2,123 LOG individuals and 237 LOG groves and both mostly occurred in southern and central Jiangsu. Most LOGs grew well and were distributed in villages, temples and government institutions. Ginkgos' growth status was largely associated with tree-habitat types. LOGs performed worse in commercial areas, roadsides and residential districts than in other tree-habitat types. To protect these ginkgos, dynamic monitoring and strengthening of scientific management are required, especially for tree-habitats in the process of urban planning and construction. It is also necessary to improve the relationship between religious culture and conservation measures. This is the first study examining LOGs in Jiangsu Province using a unified standard and our findings provide a baseline for future studies and insights into the regional conservation of LOGs.

Keywords

Abundance, ancient tree, *Ginkgo biloba*, spatial distribution, tree-habitat conservation, urbanisation impact

Introduction

Large old trees play significant roles in the ecosystem and biodiversity. They provide shelters for animals, store a large quantity of carbon and create microhabitats for organisms (Lindenmayer et al. 2012; Nolan et al. 2020). At present, research on the distribution, survival and conservation of large old trees has been carried out in many regions around the world (Aerts 2013; Mahmoud et al. 2015; Lindenmayer and Laurance 2016; Mölder et al. 2020).

Ginkgo biloba L. is one of the most common ancient trees across China (Fu et al. 2014; Zhu et al. 2019; Liu et al. 2020). *Ginkgo biloba*, endemic to China, is a long-lived deciduous tree (Tredici 2007), whose seeds have been used as food and medicine for thousands of years. Ginkgos have been cultivated in China since 1273 (i.e. Song Dynasty) (Crane 2016). *Ginkgo biloba* is cultivated in 32 Provinces across China, ranging from 50°10'N to 22°51'N, from 127°00'E to 121°30'E. Nowadays, a large number of old trees of this species are scattered in 23 Provinces across China, with an age over 100 years old (Xing 2013). Most large old ginkgos (LOGs) are distributed in the eastern and central subtropical China (Cao 2007).

Current studies on LOGs mainly focus on its origin, evolution and genetic diversity. Both ecological and molecular evidence indicate that Tianmu Mountains, Zhejiang Province, hold wild populations of ginkgos (Tredici et al. 1992; Li et al. 2011). Zhao et al. (2019) identified three refugia in China by re-sequencing *Ginkgo* genomes across the world and pointed out that anthropogenic introductions resulted in the spread of ginkgos from eastern China to other countries. A multi-feature analysis of ginkgo vascular cambial cells found that old ginkgos include a high expression of disease resistance-related genes, which may lead to its longevity (Wang et al. 2020). However, little is known about the population and conservation ecology of LOGs (Chi et al. 2020). Most current studies have been confined to a small area and researchers only made a simple description of ginkgos including their number, the diameter at breast height (DBH) and tree height due to a lack of uniform criteria or survey standards. Accordingly, it is difficult to compare these survey data retrieved from different regions.

Here, we selected Jiangsu Province in east China, as a representative sampling site to examine its spatial distribution and appraise its conservation status. With a history of more than 2,500 years, Jiangsu Province has two of the six counties called “Ginkgo Village” around the country (i.e. Taixing and Pixian). Moreover, the seed production of *Ginkgo biloba* in Jiangsu Province ranks first nationwide for the past few decades. For example, in Taixing, annually 4,000 tonnes of ginkgo nuts are produced, accounting for approximately one-third of the country’s total output (personal communication, data from Jiangsu Forestry Bureau).

In this study, we compiled a dataset of LOGs, including the number, distribution and growth status for the first time at the provincial level in China. Moreover, we analysed the relationship between tree-habitat types and growth status and then explained the effect of local culture and history on the growth of LOGs. Each sample of ginkgos was more than 100 years old. Specifically, the three objectives of this study are:

1. to map the spatial distribution of LOGs across Jiangsu Province (i.e. southern, central and northern Jiangsu) and their distribution in three protection categories;
2. to analyse DBH size class distributions of LOGs in the three Regions of Jiangsu Province;
3. to characterize the growth status of these LOGs and to determine underlying factors affecting their growth and conservation.

The purpose of this study is to address the quantity, distribution and growth characteristics of LOGs in east China and to provide implications for regional conservation and management.

Methods

Study area

The study area, Jiangsu Province (30°45'N–35°07'N, 116°22'E–121°55'E), has a long cultivation history of *Ginkgo biloba*. It is located in the centre of the eastern coast of mainland China. This Province covers 107,200 km^2 and has a coastline of over 1,000 km along the Yellow Sea. Geographically, Jiangsu Province can be divided into southern (S. Jiangsu), central (C. Jiangsu) and northern (N. Jiangsu).

The climate of Jiangsu Province ranges from temperate to subtropical. N. Jiangsu has a warm temperate humid monsoon climate, while S. Jiangsu and C. Jiangsu have a subtropical humid monsoon climate. The annual mean temperature in the Province is 13–16 °C. Jiangsu accommodates many rivers and lakes and the mean annual precipitation of Jiangsu is 998.5 mm. The terrain is mainly flat and the highest altitude is 624.4 m. There are several soil types from northern to southern Jiangsu which are, namely, brown soil, leached cinnamon soil, yellow-brown soil and red-yellow soil (Jiangsu Forestry Bureau 2017).

Data collection

An inventory of Forest Genetic Resources, including ancient trees, was carried out from 2016 to 2020 in Jiangsu Province. The standard of large old trees followed "*Technical Guidelines for Document Establishment of General Survey of National Ancient-Famous Trees*" of 2001 in China. In this guideline, ancient trees are divided into three protection categories: 100 years ≤ tier 3 ≤ 299 years, 300 ≤ tier 2 ≤ 499 years and tier 1 ≥ 500 years. The information of each ancient tree in the territory, including global positioning system (GPS) coordinates, DBH, tree age, growth status and photographs were transferred to the Cloud Platform for Forest Genetic Resources Information of Jiangsu Province (with restricted access).

In this study, all records of LOGs in Jiangsu Province were downloaded from the platform and processed by the following standards: (1) *Ginkgo biloba* with DBH < 50 cm was

Table 1. The nine tree-habitat types which accommodate LOGs in Jiangsu Province, east China.

Abbreviation	Tree-habitat types	Paraphrase
VF	Villages and farmlands	Including rural areas and farmlands where houses are low and far from the city centre with small population density, usually surrounded by croplands.
RC	Religious sites and cemeteries	Including ancestral temples, Taoist temples, nunneries, memorial parks which are mostly associated with the sacrificial activities of ancestors and gods.
GC	Government, institutional units and community grounds	Including schools, hospitals, libraries, museums, retirement homes and military regions which are usually non-profit institutional units directly managed by the State.
EC	Enterprises and commerce places	Including factories, supermarkets and business towers which are usually places for commercial activities to generate revenue.
RD	Residential districts	Including residential quarters and apartments where buildings are compact and close to the city centre with large population density, usually surrounded by shops and entertainment venues.
PG	Parks and gardens	Including public parks, forest parks, street gardens, botanical gardens and historical sites which often aim to provide a good ecological environment.
WP	Wooded areas and plant nurseries	Including firewood farms, arboretums and fruit ranches that are dedicated to the cultivation and tending of trees.
RS	Roadsides	Including pedestrian lanes, isolation belts and traffic circles where trees are often near the main roads directly monitored by the city virescence management office.
OT	Others	Including all sites that are not well-described or fail to fall into the other eight categories, such as ferry stations, golf courses and well fields.

eliminated (Zhu et al. 2019). (2) Incorrect or questionable data (e.g. DBH > 7,000 cm) was re-investigated and corrected. (3) The growth status of ginkgos was classified as good, fair, poor and dying. The good category implies that the ginkgo is vigorous and without interference. The fair category means the ginkgo has an average growth performance, as well as minor damage and interference. The poor category means the ginkgo is weak, growing slowly and suffering serious damage. The dying category indicates the ginkgo is moribund and whose branches are mostly withered (Zhang et al. 2017). (4) Nine tree-habitat types were identified from field records and tree owners (Table 1).

Data analysis

Based on GPS coordinates, the distribution map of LOGs in Jiangsu Province was prepared using ArcGIS 10.1 (Fig. 1). Differences amongst the three Regions of Jiangsu can be directly shown in this distribution map.

Size classes of DBH were used in this study, which is similar to age classes (Silvertown and Charlesworth 2003), because size classes of DBH and age classes of a tree can consistently respond to the environment (Frost and Rydin 2000; Li and Zhang 2015). LOGs in Jiangsu Province mainly exist in the form of individuals rather than groves. Most individuals sporadically occur in Jiangsu Province and are less likely to be managed and protected, thus making them exist in a semi-natural state. Accordingly, these LOG individuals over hundreds of years old are commonly different from trees in a plantation.

Here, we divided all LOG individuals in each Region into various groups by size classes of DBH in order to analyse their distribution. The metrics 50–70 cm stand for the first class (I) and the size class increases by one with each 20 cm increase in DBH

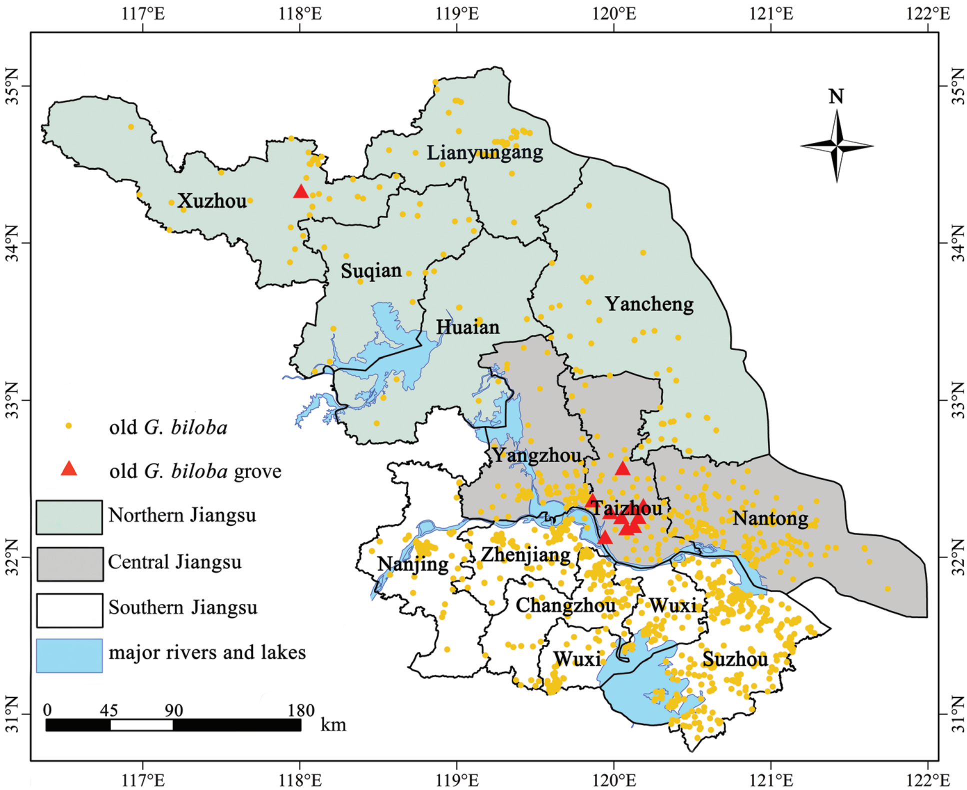

Figure 1. Geographical distribution of LOGs in Jiangsu Province, east China.

(II–XV). In general, a certain range of DBH represents a corresponding range of tree ages. Hence, we also divided these LOGs into the following three categories: 50 cm ≤ tier 3 < 90 cm, 90 cm ≤ tier 2 < 130 cm and tier 1 ≥ 130 cm. By doing so, they can correspond to the three protection categories of national ancient-famous trees in China.

To better understand the conservation status of LOGs across different tree-habitat types, a three-dimension graph of ginkgo performance in nine habitat types in Jiangsu Province was drawn using Origin 2019 and detrended correspondence analysis (DCA) was performed using Canoco v5.02 to further reveal the relationship between the growth status and tree-habitat types (Zhang et al. 2017; Huang et al. 2020).

Results

Spatial geographical distribution

We compiled a dataset of 2,123 LOG individuals and 237 groves across 13 prefecture-level cities in Jiangsu Province, China (Fig. 1, Suppl. material 1). Regarding LOG individuals, their distribution was uneven and mostly occurred on both sides of the Lower Yangtze River Basin in Jiangsu Province. Moreover, there were more ginkgos in S. Jiangsu (48.42%) and C. Jiangsu (38.48%) than in N. Jiangsu (13.10%), although N. Jiangsu has the largest administrative area. In addition, amongst these 13 cities, LOG groves were only found in Taizhou and Xuzhou (mainly in Taizhou). Accordingly, most LOGs occurred in the form of individuals rather than groves in Jiangsu Province.

DBH size class distribution

We divided 2,123 LOG individuals into 15 size classes, amongst which the first four (I–IV) size classes contained most of the individuals (Fig. 2). We found that the number of size classes in C. Jiangsu and N. Jiangsu was smaller than that in S. Jiangsu (Fig. 2a–c). In addition, there were more super-aged senior LOGs in S. Jiangsu than in C. or N. Jiangsu.

More than 60% of LOGs were classified as third-class (tier 3) ginkgos in Jiangsu Province (Table 2). Amongst these ginkgos, 43.51% were in C. Jiangsu and 41.71% in S. Jiangsu. Regarding the second-class (tier 2) ginkgos, more than half occurred in S. Jiangsu. Finally, the first-class (tier 1) ginkgos were always the fewest in each Region, amongst which S. Jiangsu included 218 ancient trees accounting for 74.15% of the total first-class LOGs.

Growth status

We also explored the relationship between tree-habitat types and the growth status of LOG individuals in Jiangsu Province (Fig. 3). Fig. 3a illustrates the relative quantitative differences, while Fig. 3b explores the relationship between tree-habitat types and the growth status of those ginkgos.

More than 85% of the LOGs grew well (Suppl. material 1). Amongst them, 50.31% of the growth status was fair with the largest proportion, followed by the good status accounting for 34.95%. This result indicated that most ginkgos in Jiangsu Province were well preserved. Regarding the nine tree-habitat types (see Table 1 for the specific definitions), most LOGs were distributed in VF, RC, and GC, accounting for 39.10%, 21.06%, and 17.15%, respectively. Moreover, the percentage of these ginkgos grown in VF was always the largest in each growth status type.

LOGs in a good growth status were close to all these nine tree-habitat types (Fig. 3), indicating that the frequency of their occurrences in each tree-habitat type was high and relatively similar. Ginkgos in a fair growth status were closest to WP, meaning that the frequency of their appearance in WP was higher than those in other tree-habitat types. Dissimilar to ginkgos in better conditions (i.e. good and fair), those in a poor growth status were the closest to RD and RS and the dying ones were the closest to EC. Ginkgos in worse conditions gained a higher probability to grow in EC, RD or RS, implying that these three tree-habitat types were less suitable for the survival of LOGs in Jiangsu Province, China.

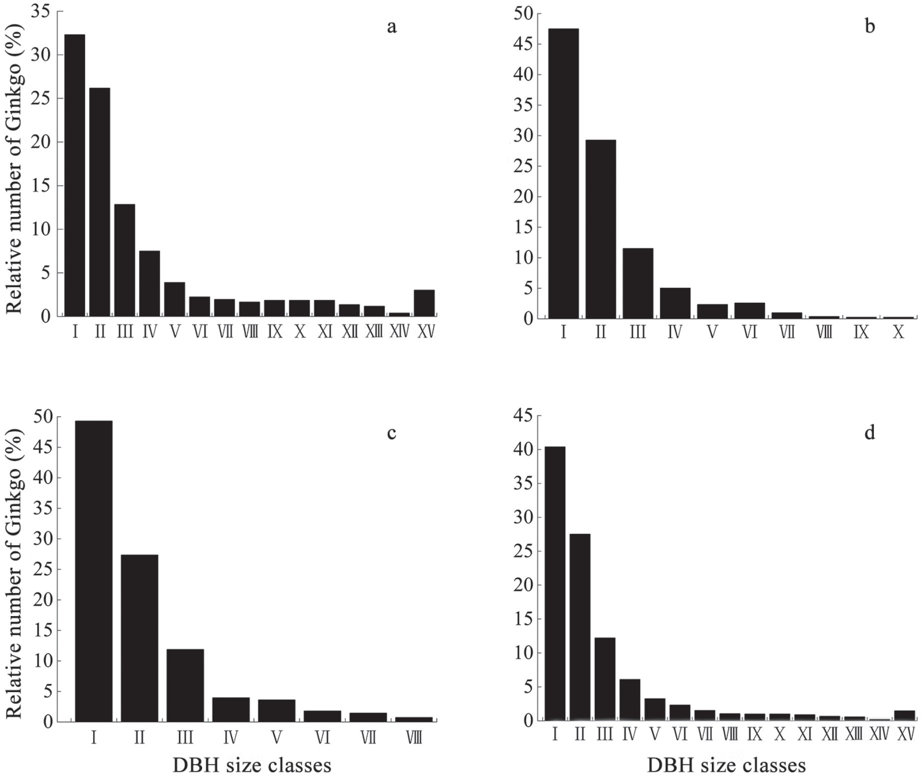

Figure 2. DBH size class distributions of LOGs in Jiangsu Province **a** S. Jiangsu **b** C. Jiangsu **c** N. Jiangsu **d** the whole Province.

Table 2. The number of LOG individuals classified by DBH size classes in three Regions of Jiangsu Province.

Class of ginkgos	DBH (cm)	Size classes	S. Jiangsu		C. Jiangsu		N. Jiangsu		Subtotal	
			Number	%	Number	%	Number	%	Number	%
Third-class	50–90	I–II	601	58.46	627	76.74	213	76.62	1441	67.88
Second-class	90–130	III–IV	209	20.33	135	16.52	44	15.83	388	18.28
First-class	≥130	≥V	218	21.21	55	6.73	21	7.55	294	13.85
Total			1028	100.00	817	100.00	278	100.00	2123	100.00

Discussion

Quantity and distribution of LOGs

Ginkgo biloba, a relic of the Tertiary period, are now rare in the wild, whereas the long cultivation history of ginkgos has made it one of the most common cultivated trees, widely distributed in 32 provinces, autonomous regions and municipalities across China (Guo et al. 2019). The vast majority of LOGs in Jiangsu Province derive from artificial cultivation according to field records and our survey.

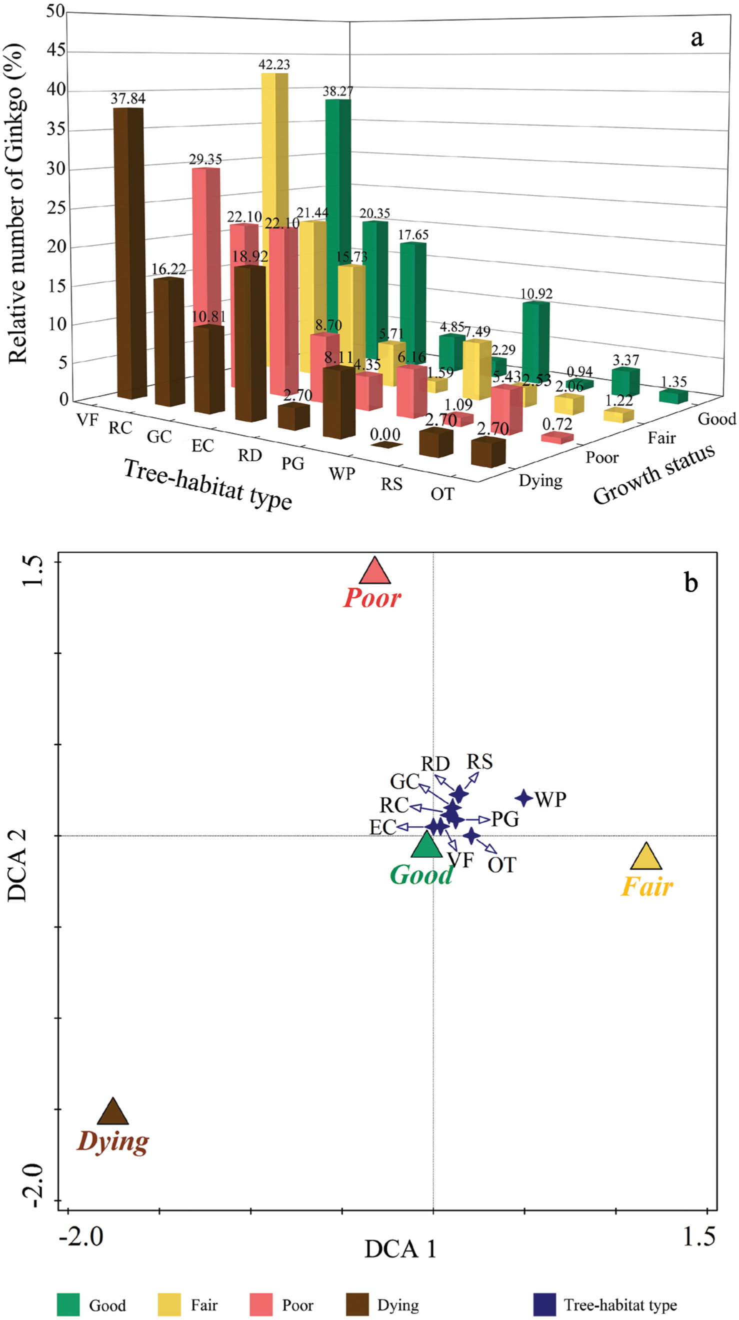

Figure 3. The growth status of LOGs across different tree-habitat types in Jiangsu Province (see Table 1 for the explanation of abbreviated habitat types) **a** three-dimension graph of the number of ginkgos across nine tree-habitat types **b** detrended correspondence analysis of the relationship between tree-habitat types and growth status with the cumulative explained variation as 78.71%. Tree-habitat types and growth status are drawn as stars and triangles, respectively.

Here, we sorted out reliable information of 2,123 LOG individuals and 237 LOG groves in Jiangsu Province for the first time. This is markedly different from previous survey data of LOGs. Only 77 LOGs with tree height and DBH were listed in a retrospective monograph titled "*Ginkgo in China*" (Cao 2007). Nevertheless, Fu et al. (2014) supposed that there were 11,858 LOGs in Jiangsu Province, amongst which only 632 had partial growth indices, such as DBH. Liu et al. (2019a) counted a total of 836 LOGs through reports and documents, while the data sources only involved several prefectural-level cities in Jiangsu Province. Possible reasons for such differences are chiefly as follows: (1) Inconsistent survey methods. For example, we set the starting DBH of 50 cm in our survey (Zhu et al. 2019). However, the survey criteria used in other studies were either different from ours or not stated. (2) Different degrees of investigation. Our survey was based on a census of field investigation from 2016 to 2020, but the actual investigation degree and scope in other studies may be much shorter than ours. Therefore, our study can be used as a baseline for the protection and management of LOGs in Jiangsu Province.

Our findings indicate that there were more LOGs in S. Jiangsu and C. Jiangsu and most of the first-class ginkgos were distributed in S. Jiangsu (Figs 1, 2 and Table 2). Such differences can be mainly attributed to two factors: (1) Climate differences. The mean annual precipitation and temperature in S. and C. Jiangsu are higher than those in N. Jiangsu. Many large old trees are sensitive to drought and persistent to high temperatures (Bennett et al. 2015; Lindenmayer et al. 2016; Venter et al. 2017; Choat et al. 2018). Regarding tall LOGs with strong root stems, their evaporation and water requirement are relatively high and, therefore, the subtropical humid monsoon climate in S. and C. Jiangsu is more suitable for ginkgos to survive than in N. Jiangsu. (2) Soil differences. There are almost no coastal cities in S. and C. Jiangsu (except Nantong) (Fig. 1). However, both Yancheng and Lianyungang in N. Jiangsu have a much longer coastline than Nantong has in C. Jiangsu. In these coastal areas, a narrow saline belt extends from north to south, where the soil has a salt content of 1–4% (Jiangsu Forestry Bureau 2017). When the salt content in soil increases to more than 0.3%, it can pose a fatal threat to the growth of ginkgos (He et al. 1997). This factor can explain that almost no LOGs can be found near coastal areas. In contrast, S. Jiangsu is on the southern bank of the Lower Yangtze River. In alluvial plains along rivers and lakes, humus and soil layers are thick and rich in nutrients such as calcium, magnesium, potassium and phosphorus (He et al. 1995), which are suitable for the growth of LOGs. Liu et al. (2020) found that the first-class ancient trees were mostly distributed in southeast Anhui Province, especially to the south of the Yangtze River. This result is also consistent with the distribution of the first-class ginkgos in Jiangsu Province according to our analysis. Besides, in terms of financial development, S. Jiangsu is the best, followed by C. Jiangsu and N. Jiangsu is the third in the past few decades. Additionally, S. Jiangsu has a longer cultural history than the other parts of Jiangsu Province.

In short, climate and soil conditions, together with history and socioeconomic backgrounds in S. and C. Jiangsu are more suitable for the growth of LOGs. In addition, the distribution of LOGs may be also in connection with phyletic evolution to some extent, which are worthy of further study.

Tree-habitat types and growth status of LOGs

Most LOGs in Jiangsu Province grew well and were distributed in VF, RC and GC (Figs 3, 4a). Ginkgos grow well in a climate with four distinct seasons, in relatively low altitudes (≤ 1000 m), warm annual mean temperature (10–18 °C), moderate annual precipitation (600–1,000 mm) and high relative humidity (70–90%) (He et al. 1997; Tredici 2007; Liu et al. 2019b). Besides, ginkgos prefer fertile, moist and well-drained soil (He et al. 1995). The climate and soil conditions in Jiangsu Province are suitable for the growth of LOGs. As a result of the long cultivation period, ginkgos have been gradually endowed with religious, cultural and historical values (Jim 2004a; Lindenmayer et al. 2014). In Chinese traditional geomantic culture, ginkgos planted in the front of temples, houses and villages cannot be arbitrarily cut down, because these Fengshui woods are treated as protective barriers to keep buildings safe (Lü et al. 2009). These ancient trees connect villagers to previous generations and become part of the heritage in their family (Mahmoud et al. 2015). LOGs in the temple are also considered as “Buddha tree” or “sacred tree” (Ma 2003; Crane 2019; Huang et al. 2020). Therefore, people admire and actively protect these ancient trees.

However, approximately 15% of the LOGs in Jiangsu Province are still under threat due to mismanagement and urbanisation impacts (Figs 3, 4d) (Zapponi et al. 2017; Khapugin et al. 2020). For the sake of beauty and cleaning, building materials, such as cement, sometimes cover the soil around the stems of LOGs in PG and RS (Fig. 4d–6), which affects the root respiration of LOGs (Jim 2004b; Jim and Zhang 2013). Some citizens have deposited their electromotors, wastes and other miscellaneous items under these ancient trees which may degrade or destroy tree-habitats (Fig. 4d–4). The mean DBH value of LOGs in Jiangsu Province was 94.50 cm, the mean crown width was 11.39 m and the mean height was 17.01 m (Suppl. material 1), indicating that the growth of these ginkgos requires more open space. LOGs poorly performed in EC, RD and RS (Fig. 3b). The three tree-habitat types are characterised by high population mobility and rapid urbanisation. This situation causes ginkgos to be threatened with restricted space and insufficient light due to the shade from buildings (Fig. 4b, 4d–5, d–7). In addition, several natural events, such as lightning strikes, storms, insect infestations and epiphytic entwinement (Fig. 4c) also threaten the survival of LOGs (Yu and Gao 2004; Takács et al. 2020; Wang et al. 2020).

Suggestions for LOGs’ conservation

Based on the distribution features and tree-habitat quality of LOGs in Jiangsu Province, we offer three suggestions to conserve these ancient trees:

(1) Strengthening of LOGs’ management and protection

At present, there is a shortage of forestry skills in Jiangsu Province, especially professionals proficient in the protection and management of local ancient trees. The

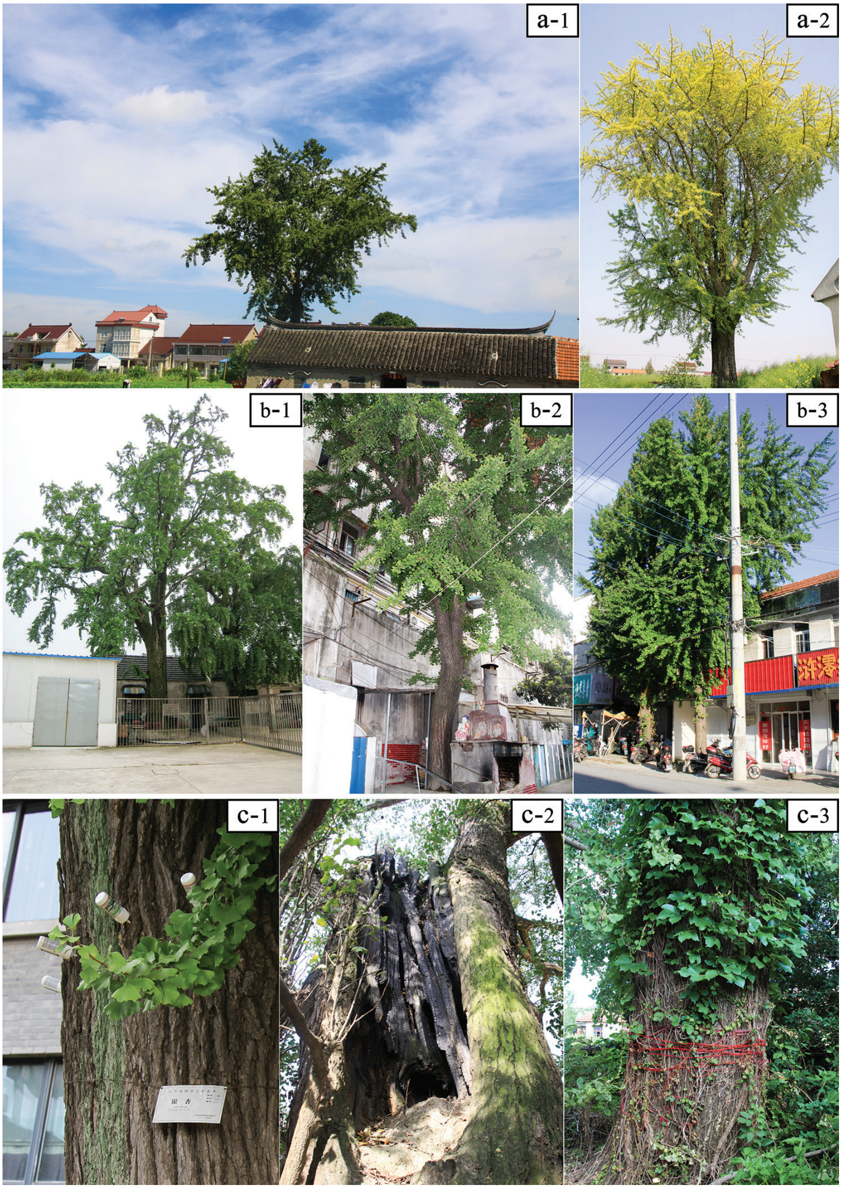

Figure 4. Photographs of LOGs across different tree-habitat types in Jiangsu **a** panoramas of LOGs in good tree-habitat types in villages (VF) with (**–1**) open space and plenty of sunshine and (**–2**) no shade and good soil environment **b** panoramas of LOGs in bad tree-habitat types (**–1**) in the factory (EC) (**–2**) in the old town (RD) (**–3**) near the roadside (RS) **c** details of natural disasters experienced by LOGs in different tree-habitat types, including (**–1**) insect infestations, (**–2**) lightning strikes, and (**–3**) the effect of epiphytes **d** details of the anthropogenic interference experienced by LOGs in different tree-habitat types, including (**–1**) stuffing objects into the stem, (**–2**) incense burning and incense ash piling, (**–3**) a large number of red ropes wound around the branches, (**–4**) stacking sundries under the trees, (**–5**) the sheltered side of houses, (**–6**) cement-sealed ground and (**–7**) construction. The photographs were provided by Zhang G.F.

Figure 4. Continued.

conservation of LOGs needs specialised knowledge. We suggest that provincial forestry authorities regularly provide professional guidance to the front-line management personnel of LOGs. These guidelines include pest control, epiphytes removal and pollarding (Zapponi et al. 2017). For those LOGs often disturbed by residents' activities, large fences (at least as large as the crown width) should be installed around the trees (Mahmoud et al. 2015).

(2) Appropriate planning during urban and rural construction

The survival and protection of LOGs should be considered as an important factor and incorporated into urbanisation or rural reconstruction to ensure enough living space and good light conditions for these ancient trees. If a suitable ecological niche is maintained in the construction design and a large area of cement is replaced by lawn, the growth of LOGs can be guaranteed (Jim and Zhang 2013; Zhang et al. 2017; Lai et al. 2019). Sometimes, when problems, such as restricted space in old towns, are hard to resolve in urban reconstruction, the government can also consider transplanting these LOGs into urban parks or green space, where unenclosed overground space and underground soil can provide good growth conditions for tree stems and roots (Jim 2004b; Zhang et al. 2017; Lai et al. 2019). Moreover, for those LOGs still maintaining a good status, we suggest local forestry administration should recognise the dynamic development of LOGs and conserve their tree-habitats.

(3) Improvement of the relationship between religious culture and conservation measures

Worship of LOGs may sometimes unintentionally cause a major threat to their survival (Fig. 4d–1-3). For example, tourists often burn incense sticks around the stems of these ancient trees or even insert coins into the cracks of stems to pray for good fortune. Therefore, we should coordinate the relationship between LOGs' protection and cultural custom. More specifically, it may be a good choice to build fences around the LOGs and set special places out of the fences for worship. Residents may also be mobilised to make conservation plans and work together for the protection of LOGs.

Conclusions

Ginkgo biloba is a Mesozoic relict species endemically distributed in China. We, for the first time, addressed the number, distribution and growth status of LOGs at the provincial level. The impacts of urbanisation and religious culture on both tree-habitat types and the growth of LOGs should be considered to conserve these ancient trees. Our findings can offer useful advice for the protection of LOGs in east China with a similar natural environment. Meanwhile, this study may provide a helpful reference for the regional conservation of other ancient trees. In addition, the effect of phyletic evolution on LOGs' distribution deserves further study.

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References

Aerts R (2013) Old trees: Extraction, conservation can coexist. Science 339(6122): 904. https://doi.org/10.1126/science.339.6122.904-a

Bennett AC, McDowell NG, Allen CD, Anderson-Teixeira KJ (2015) Larger trees suffer most during drought in forests worldwide. Nature Plants 1(10): e15139. https://doi.org/10.1038/nplants.2015.139

Cao FL (2007) Ginkgo in China. China Forestry Publishing House, Beijing.

Chi XL, Yang G, Sun K, Li XL, Wang TL, Zhang AJ, Li Y, Cheng M, Wang QG (2020) Old ginkgo trees in China: Distribution, determinants and implications for conservation. Global Ecology and Conservation 24: e01304. https://doi.org/10.1016/j.gecco.2020.e01304

Choat B, Brodribb TJ, Brodersen CR, Duursma RA, López R, Medlyn BE (2018) Triggers of tree mortality under drought. Nature 558(7711): 531–539. https://doi.org/10.1038/s41586-018-0240-x

Crane PR (2016) Ginkgo: The tree that time forgot. Higher Education Press, Beijing. [Translated by Hu YH, Zhang QF]

Crane PR (2019) An evolutionary and cultural biography of ginkgo. Plants People Planet 1(1): 32–37. https://doi.org/10.1002/ppp3.7

Frost I, Rydin H (2000) Spatial pattern and size distribution of the animal-dispersed *Quercus rubur* in two spruce-dominated forests. Ecoscience 7: 38–44. https://doi.org/10.1080/11956860.2000.11682569

Fu ZJ, Xing SY, Liu XJ, Liu LJ, Xin H, Wu QK (2014) Investigation on ancient ginkgo resources in Jiangsu Province. China Forestry Science and Technology 28(4): 63–67.

Guo Y, Lu Y, El-Kassaby YA, Feng L, Wang GB, Wang TL (2019) Predicting growth and habitat responses of *Ginkgo biloba* L. to climate change. Annals of Forest Science 76(4): 101.

He F, Wang YQ, Lu FD (1995) Soil investigation in ginkgo distribution area of Hunan Province. Economic Forest Researches 13(3): 1–12. https://doi.org/10.1007/s13595-019-0885-0

He SA, Gu Y, Pang ZJ (1997) Resources and prospects of *Ginkgo biloba* in China. In: Hori T, Ridge RW, Tulecke W, Tredici PD, Trémouillaux-Guiller J, Tobe H (Eds) *Ginkgo biloba* A global treasure: From biology to medicine. Springer-Verlag, Tokyo, 373–383. https://doi.org/10.1007/978-4-431-68416-9_28

Huang L, Tian L, Zhou L, Jin C, Qian SH, Jim CY, Lin DM, Zhao L, Minor J, Coggins C, Yang YC (2020) Local cultural beliefs and practices promote conservation of large old trees in an ethnic minority region in southwestern China. Urban Forestry & Urban Greening 49: 126584. https://doi.org/10.1016/j.ufug.2020.126584

Jiangsu Forestry Bureau (2017) The key protected wild plants in Jiangsu Province. Nanjing Normal University Press, Nanjing.

Jim CY (2004a) Evaluation of heritage trees for conservation and management in Guangzhou City (China). Environmental Management 33(1): 74–86. https://doi.org/10.1007/s00267-003-0169-0

Jim CY (2004b) Spatial differentiation and landscape-ecological assessment of heritage trees in urban Guangzhou (China). Landscape and Urban Planning 69(1): 51–68. https://doi.org/10.1016/j.landurbplan.2003.09.008

Jim CY, Zhang H (2013) Species diversity and spatial differentiation of old-valuable trees in urban Hong Kong. Urban Forestry & Urban Greening 12(2): 171–182. https://doi.org/10.1016/j.ufug.2013.02.001

Khapugin AA, Kuzmin IV, Silaeva TB (2020) Anthropogenic drivers leading to regional extinction of threatened plants: insights from regional Red Data Books of Russia. Biodiversity and Conservation 29: 2765–2777. https://doi.org/10.1007/s10531-020-02000-x

Lai PY, Jim CY, Tang GD, Hong WJ, Zhang H (2019) Spatial differentiation of heritage trees in the rapidly-urbanizing city of Shenzhen, China. Landscape and Urban Planning 181: 148–156. https://doi.org/10.1016/j.landurbplan.2018.09.017

Li W, Zhang GF (2015) Population structure and spatial pattern of the endemic and endangered subtropical tree *Parrotia subaequalis* (Hamamelidaceae). Flora 212: 10–18. https://doi.org/10.1016/j.flora.2015.02.002

Li L, Zhang GF, Wang R, Sun G, Zhao MS (2011) Life table of natural *Ginkgo biloba* population in Tianmu Mountain Nature Reserve. Chinese Journal of Ecology 30(1): 53–58.

Lindenmayer DB, Laurance WF (2016) The ecology, distribution, conservation and management of large old trees. Biological Reviews of the Cambridge Philosophical Society 92(3): 1434–1458. https://doi.org/10.1111/brv.12290

Lindenmayer DB, Laurance WF, Franklin JF (2012) Global decline in large old trees. Science 338(6112): 1305–1306. https://doi.org/10.1126/science.1231070

Lindenmayer DB, Banks SC, Laurance WF, Franklin JF, Likens GE (2014) Broad decline of populations of large old trees. Conservation Letters 7(1): 72–73. https://doi.org/10.1111/conl.12079

Lindenmayer DB, Blanchard W, Blair D, McBurney L, Banks SC (2016) Environmental and human drivers influencing large old tree abundance in Australian wet forests. Forest Ecology and Management 372: 226–235. https://doi.org/10.1016/j.foreco.2016.04.017

Liu JJ, Lindenmayer DB, Yang WJ, Ren Y, Campbell MJ, Wu C, Luo YQ, Zhong L, Yu MJ (2019a) Diversity and density patterns of large old trees in China. Science of the Total Environment 655: 255–262. https://doi.org/10.1016/j.scitotenv.2018.11.147

Liu Z, Hu ST, Li Y, Zhu P, Chu XD, Sun Y (2019b) Ancient *Ginkgo biloba* resources characters and age detection in Changxing County. Journal of Zhejiang Forestry Science and Technology 39(6): 87–91.

Liu DW, Wang YJ, Xie CP, Xu P (2020) Analyses on resource characteristics and impact factors of first-class old trees in Anhui Province. Journal of Plant Resources and Environment 29(1): 59–68.

Lü HR, Liu SS, Zhu JY, Ye YC, Chen HY, Mo LJ (2009) Effects of human disturbance on understory woody species composition and diversity in Fengshui forests. Biodiversity Science 17(5): 458–467. https://doi.org/10.3724/SP.J.1003.2009.09174

Ma WX (2003) Developing situation and the value analyses of *Ginkgo biloba* L. Journal of Biology 20(4): 34–35.

Mahmoud T, Gairola S, El-Keblawy A (2015) Large old trees need more conservation attention: A case of *Tamarix aphylla* in the arid deserts of the United Arab Emirates. Journal of Asia-Pacific Biodiversity 8(2): 183–185. https://doi.org/10.1016/j.japb.2015.04.006

Mölder A, Schmidt M, Plieninger T, Meyer P (2020) Habitat–tree protection concepts over 200 years. Conservation Biology cobi.13511. https://doi.org/10.1111/cobi.13511

Nolan V, Reader T, Gilbert F, Atkinson N (2020) The ancient tree inventory: A summary of the results of a 15 year citizen science project recording ancient, veteran and notable trees across the UK. Biodiversity and Conservation 29: 3103–3129. https://doi.org/10.1007/s10531-020-02033-2

Silvertown JW, Charlesworth D (2003) Introduction to plant population ecology (Fourth edition). Higher Education Press, Beijing. [Translated by Li B, Dong HQ, Lu JZ]

Takács M, Szénási Á, Malatinszky Á (2020) Polypores, *Agrobacterium* and ivy damage on Hungarian ancient trees. Nature Conservation 40: 1–38. https://doi.org/10.3897/natureconservation.40.51633

Tredici PD (2007) The phenology of sexual reproduction in *Ginkgo biloba*: Ecological and evolutionary implications. Botanical Review 73(4): 267–278. https://doi.org/10.1663/0006-8101(2007)73[267:TPOSRI]2.0.CO;2

Tredici PD, Ling H, Yang G (1992) The Ginkgos of Tian Mu Shan. Conservation Biology 6(2): 202–209. https://doi.org/10.1046/j.1523-1739.1992.620202.x

Venter M, Dwyer J, Dieleman W, Ramachandra A, Gillieson D, Laurance S, Cernusak LA, Beehler B, Jensen R, Bird MI (2017) Optimal climate for large trees at high elevations drives patterns of biomass in remote forests of Papua New Guinea. Global Change Biology 23(11): 4873–4883. https://doi.org/10.1111/gcb.13741

Wang L, Cui JW, Jin B, Zhao JG, Xu HM, Lu ZG, Li WX, Li XX, Li LL, Liang EY, Rao XL, Wang SF, Fu CX, Cao FL, Dixon RA, Lin JX (2020) Multifeature analyses of vascular cambial cells reveal longevity mechanisms in old *Ginkgo biloba* trees. Proceedings of the National Academy of Sciences 117(4): 2201–2210. https://doi.org/10.1073/pnas.1916548117

Xing SY (2013) Ginkgo germplasm resources in China. China Forestry Publishing House, Beijing.

Yu GP, Gao BN (2004) Biological characteristics and control of *Pammene ginkgoicola*. Entomological Knowledge 41(5): 475–477.

Zapponi L, Mazza G, Farina A, Fedrigoli L, Mazzocchi F, Roversi PF, Peverieri GS, Mason F (2017) The role of monumental trees for the preservation of saproxylic biodiversity: Re-thinking their management in cultural landscapes. Nature Conservation 19: 231–243. https://doi.org/10.3897/natureconservation.19.12464

Zhang H, Lai PY, Jim CY (2017) Species diversity and spatial pattern of old and precious trees in Macau. Landscape and Urban Planning 162: 56–67. https://doi.org/10.1016/j.landurbplan.2017.02.002

Zhao YP, Fan GY, Yin PP, Sun S, Li N, Hong XN, Hu G, Zhang H, Zhang FM, Han JD, Hao YJ, Xu QW, Yang XW, Xia WJ, Chen WB, Lin HY, Zhang R, Chen J, Zheng XM, Lee SMY, Lee J, Uehara K, Wang J, Yang HM, Fu CX, Liu X, Xu X, Ge S (2019) Resequencing 545 ginkgo genomes across the world reveals the evolutionary history of the living fossil. Nature Communications 10(1): 1–10. https://doi.org/10.1038/s41467-019-12133-5

Zhu W, Zhang GF, Zan ZY (2019) Population structure and dynamics of old *Ginkgo biloba* in Nantong, Jiangsu Province. Chinese Journal of Ecology 38(12): 3609–3616.

Supplementary material 1

The large old ginkgos in Jiangsu Province, east China

Authors: Jie Liu, Ruo-Yan Jiang, Guang-Fu Zhang

Data type: Occurences

Explanation note: The supplementary file includes two parts: the large old ginkgo (LOG) individuals and LOG groves in Jiangsu Province, east China. The first part contains the site, GPS, DBH, protection category, growth status and habitat of each LOG individual. The second part contains the site, GPS, mean DBH, number of ginkgos, growth status and habitat of each LOG grove. Dryad doi: 10.5061/dryad.gtht76hk6

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Suitability of contract-based nature conservation in privately-owned forests in Germany

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Abstract

The successful implementation of contract-based nature conservation in privately-owned forests requires a framework of reasonable operational measures. Our study aimed at developing such a framework by; 1) defining forest conservation objects including structures, processes, and habitat types, 2) assessing their conservation value based on the need for, and worthiness of, protection, 3) reviewing the suitability of contract-based measures for conservation. Overall, we defined 67 conservation objects, with 8 of them used as case studies: deadwood, habitat trees, natural succession after large-scale disturbance, coppice-with-standards, bog and fen woodlands, dry sand pine forests, and beech forests. We considered contract-based conservation suitable if, within the contract period, outcomes of measures resulted in ecological upgrading or avoidance of value loss. We identified contract-based conservation suitable for 42 combinations of objects and measures. Our approach of assessing the potential of contract-based measures for forest conservation is novel with regards to its broad range of objects, defined criteria, and various contract periods. It can help to progress conservation and improve outcomes of measures, especially in privately-owned forests in Germany. Further prerequisites are sufficient financial resources, effective administration, consultancy and the mid- to long-term stability of funding programmes.

Keywords

Forest conservation objects, funding, nature conservation value, need for protection, private forests, suitability assessment, worthiness of preservation

Introduction

In the European Union (EU-28), about 60% of the forested area is privately owned, with huge differences among the member states (Eurostat 2018). Germany, which lies slightly below the EU-28 average with about half of its forest area privately-owned (Polley et al. 2016), may serve as an example to highlight the problems and opportunities for nature conservation in private forests. Implementing conservation measures in private forests may cause additional costs or expenses for forest owners (Sotirov 2017). At present, forest conservation measures in private forests are implemented in Germany primarily through regulations, rather than through subsidies. In stark contrast to agriculture, contractual agreements and funding instruments to compensate for economic losses caused by the implementation of nature conservation measures are rarely used in German forestry (Güthler et al. 2005; Franz et al. 2018b). However, German legislation indicates that the country grants voluntary agreements preference over legal regulations and constitutes in § 3(3) of the German Federal Nature Conservation Act that "… priority shall be given to reviewing whether the intended purpose could also be achieved via contractual agreements". Contract-based agreements are assumed to have a higher acceptance among private forest owners than purely regulatory measures (Franz et al. 2017). The National Strategy on Biological Diversity seeks to "promote contract-based nature conservation in 10% of privately-owned forest land" (BMU 2007), but this target is still far from being achieved, not least because the conditions for contract-based forest conservation have not yet been met (Franz et al. 2018a). Furthermore, overall funding frameworks, for instance for the implementation of Natura 2000, are lacking (Geitzenauer et al. 2017; Sotirov 2017). In contrast to regulations, contract-based nature conservation strives to achieve a consensual, bilateral agreement. In Germany, such voluntary agreements are usually contracted between private forest owners and funding bodies such as the country, federal states, foundations, or private investors. Context-specific conservation measures, referring to specific conservation objects, funding periods and amounts, as well as possible monitoring to verify success, are contractually agreed upon. A broad consensus among different stakeholders in Germany with respect to conservation objectives (Demant et al. 2019) may further promote the implementation of contract-based conservation in private forests.

A prerequisite for the implementation of nature conservation measures in forests is the identification of an operational catalogue of forest conservation objects covering all aspects of forest habitat and biodiversity conservation. An approach using conservation objects accounts for temporal context-specificity and spatial variability, if there is a broad selection of widely accepted and properly defined objects and consensus about suitable preservation measures. At present, the most commonly addressed conservation objects in private forests are habitat trees, deadwood, and historical types of forestry use, such as coppicing or wood pasture (Franz et al. 2018b). However, numerous further objects may be taken into consideration in order to fully tap the potential of private and other forests for the restoration and preservation of biodiversity.

The aim of our study was to develop a comprehensive catalogue of forest conservation objects and measures eligible for contract-based funding. We built on the framework of conservation objectives suggested by Demant et al. (2019) and focussed on forest habitat types, structural elements, and developmental processes as the most relevant conservation objects. We identified the conservation value of the objects by assessing the need for protection (owing to threat, endangerment) and the worthiness of preservation. The guiding questions for our study were:

(1) How can forest conservation objects be assessed in a way that reflects their nature conservation value, particularly in terms of their need for, and worthiness of, preservation?
(2) Which forest conservation objects are suitable for effective contract-based conservation measures and over which contractual periods should measures reasonably be funded?
(3) What consequences for nature conservation practitioners and forest owners can be derived?

Methods

Assessment of the nature conservation value of forest conservation objects

To assess the nature conservation value of a forest conservation object, we considered the initial value (before conservation measures were implemented) and the conservation value achieved after application of a measure over varying time periods. According to Frenz and Müggenborg (2016), worth of preservation alone is not enough for an object to justify a legal priority protection setting – conservation objects must also be (potentially) threatened. Thus, we differentiated between the two components 'worthy' (contributing to the preservation of characteristic species and gene pools in natural or semi-natural landscapes or ecosystems) and 'need' or 'urgency' (degree of threat as a result of adverse effects of land-use and environmental changes) to assess the conservation value of the objects (Fig. 1).

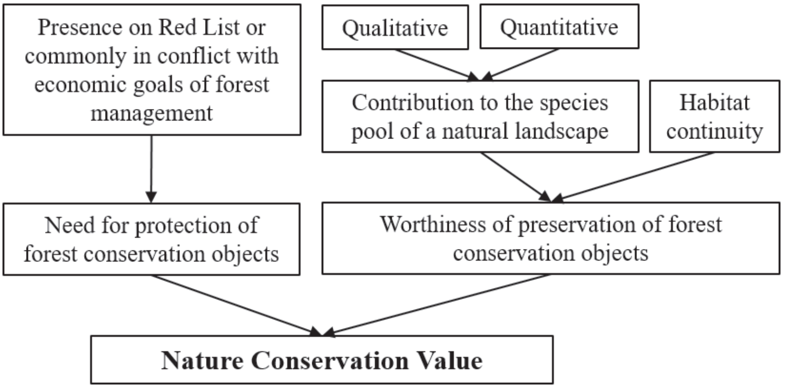

Figure 1. Assessment of the nature conservation value of forest conservation objects.

Table 1. German Red List categories of habitat types and their translation into numerical and verbal reference values.

Red List category	Description	Need for protection	Value
0	Collapsed	Very high	5
1!	Critically endangered (acutely)	Very high	5
1	Critically endangered	Very high	5
1–2	Endangered to critically endangered	High	4
2	Endangered	High	4
2–3	Vulnerable to endangered	Moderate	3
3	Vulnerable	Moderate	3
3-V	Near threatened to vulnerable	Low	2
V	Near threatened	Low	2
*	No current risk of loss trend (least concern)	Very low	1
#	Classification not meaningful, or no risk	No	0

We based the assessment of the need for protection on the national and the European Red List status categories (Janssen et al. 2016; Finck et al. 2017) translated into an ordinal scale (Table 1). The Red List status categories encompass long-term threat (assessed at national and regional levels), the current trend (stable, increasing, decreasing), rarity, and the ability to regenerate (Finck et al. 2017). Threats are "human activities or processes that have impacted, are impacting, or may impact the status of the taxon being assessed" (IUCN 2013).

The forest structures and processes that we assessed have a high urgency for protection. For example, the retention of deadwood and a natural forest development are commonly in conflict with the economic goals of forest management.

Based on an assumption that the maintenance of core ecosystem functions was of high value we selected forest conservation objects, whether they represent structures, processes, or habitat types, as worthy of preservation if they are integral parts of natural self-sustaining, or semi-natural, managed forest ecosystems (Frenz and Müggenborg 2016). We also assumed that higher value would be placed on objects with a greater importance for a region's natural and cultural heritage. The longer the habitat continuity, i.e. the period in which a conservation object has evolved its typical biodiversity, the more important it is to be preserved (Nordén et al. 2014). As the habitat continuity increases, so, too, does the responsibility of preserving the conservation object to meet "the needs of the present without compromising the ability of future generations to meet their own needs" (WCED 1987). Wood-pastures, for example, have a centuries-long habitat continuity (Bergmeier et al. 2010; Plieninger et al. 2015), and are regarded as being part of the European cultural-natural heritage (Leuschner and Ellenberg 2017).

Apart from habitat continuity, other factors determining the worth of a conservation object were the quantitative (absolute number of species) and qualitative (relative to a desired reference state) contribution of a conservation object to the species pool of a natural landscape. For example, intact peat bog woodlands may have a relatively low absolute number of species, but a high qualitative contribution to the typical diversity of the natural landscape. We based our assessment of the worthiness on expert valuations and distinguished six levels in a qualitatively ranked ordinal scale (Table 2).

Table 2. Variables for the evaluation of the worthiness of preservation.

Habitat continuity (HC)		Quantitative contribution (Q1)		Qualitative contribution (Q2)		Worthiness = [HC+ ((Q1+Q2)/2)]/2
Very long	5	Very high	5	Very high	5	5
Long	4	High	4	High	4	4
Medium	3	Moderate	3	Moderate	3	3
Short	2	Low	2	Low	2	2
Very short	1	Very low	1	Very low	1	1
None	0	None	0	None	0	0

Q1 = quantitative (absolute) contribution, Q2 = qualitative contribution to the typical diversity of the natural landscape.

For example, dry oak-hornbeam forests (*Galio-Carpinetum*) have a Red List status of 1–2 (Endangered to Critically endangered; Finck et al. 2017) which means their need for protection was high (4). Furthermore, they have a very long habitat continuity (HC = 5), a high quantitative (Q1 = 4), and a very high qualitative (Q2 = 5) contribution to the diversity of the natural landscape. Their worthiness of preservation resulted in 'very high' ([5+ ((4+5)/2)]/2 = 4.75).

We assert that structures and processes, as essential components of natural forests, are highly worthy insofar as they allow maintenance of key ecosystem functions (Walentowski and Winter 2007). The final nature conservation value resulted from the calculation of the mean values of the two protection criteria, worthiness and need, with the classes 0 = no, 1 = very low, 2 = low, 3 = moderate, 4 = high, and 5 = very high conservation value. In the example above the final conservation value is high ((4.75+4)/2 = 4.375).

Forest conservation objects

The nature conservation value assessment was carried out for eight forest structural elements, four processes, and 55 forest-related habitat types (Finck et al. 2017; see Suppl. material 1: Table S1). In the main body of the present paper, representative assessments for 8 out of 67 forest conservation objects were made, characterised in Table 3.

Suitability assessment scheme

We assessed the suitability of contract-based funding for forest conservation objects by comparison of the initial and final conservation value (Fig. 2). The initial conservation value of the conservation object was scaled between very low (0) and high (5). After projecting the expected development and outcomes over a contract length, we calculated a final conservation value, again scaled between very low and high (Fig. 2). As relevant development periods differ greatly among conservation objects, we considered three potential contractual periods: short-term (< 10 years), mid-term (10–30 years), and long-term (> 30 years).

The assessment of the worthiness of, and need for, protection of forest conservation objects resulted in a single nature conservation value, although each individual variable may have different values (Suppl. material 2: Table S2). Conservation objects

Table 3. Characteristics of 8 case study forest conservation objects.

Conservation object	Characteristics	Possible conservation measure during contract period	References
Deadwood	Key structure in forest ecosystems, variable in terms of amount, decay stages, size classes, wood diameters, microclimatic conditions, and tree species.	Retention of dead trees or logging residues; supply ring-barking, crown cutting, felling or knocking-over of trees.	Harmon et al. 1986; Davies et al. 2008; Lassauce et al. 2011; Lindenmayer et al. 2012; Agnew and Rao 2014; Seibold et al. 2015;
Habitat trees	Characterised by various tree-related microhabitats (e.g., hollows or dead branches), indicating habitat continuity; important for countless species supported by dieback structures of old-growth forest stages.	Protection of existing habitat trees and retention of potential once; creation of structures by breaking-off branches, making bark injuries or bark-removal, constructing cavities, dendrothelms (water-filled tree hollows).	Winter and Möller 2008; Fedrowitz et al. 2014; Kraus et al. 2016; Larrieu et al. 2018; Asbeck et al. 2019; Gustafsson et al. 2019; Mölder et al. 2020
Natural forest development	Characterised by typical regional and local-scale old-growth forest structures and associated biodiversity. With ongoing cessation of forestry interventions, typical developmental and structural features gradually develop over long periods of time.	Continuation of natural forest development initiated several decades ago, recent decommissioning of near-natural commercial forests. Minimum standards as defined by Engel et al. (2016, p. 38) apply.	Meyer and Schmidt 2008; Vandekerkhove et al. 2011; Kraus and Krumm 2013; Paillet et al. 2015
Natural succession after large-scale disturbance	Natural disturbances (e.g., by wildfires, windstorms, or insect infestations; intensity and frequency are expected to increase under climate change) are important drivers of forest dynamics and associated biodiversity. They contribute to maintaining pioneer species and habitats, enhance structural heterogeneity, and make forests more resilient to future disturbances.	Allowing and supporting natural development in early-successional stages.	Runkle 1989; Franklin et al. 2002; Lindenmayer et al. 2008; Swanson et al. 2011; Seidl et al. 2017; Thorn et al. 2018; AK Waldökologie GfÖ 2019; Müller et al. 2019
Coppice-with-standards	Two-layered stands with an overstorey consisting of mature trees (standards) used for timber and fruit setting. Even-aged understorey regrowth (coppice) consists of multi-stemmed trees cut at a 20–30-year rotation cycle. Offer a mosaic of habitats and structures favourable for light-demanding and thermophilic species due to conditions of alternating shade and light. Abandoned coppice-with-standards with all trees left uncut ('overstood', stems having the size of mature forest stands) are commonly converted to high forests (even-aged forest stands).	Continuation and resumption of coppice-with-standard management.	Barnthöl 2003; Groß and Konold 2009; Kirby et al. 2017; Meyer et al. 2018; Unrau et al. 2018
Bog and fen woodlands	Ecosystems of coniferous or broadleaved trees and shrubs on low-productive peaty soils with high water level. When intact, they contribute to climate protection, if drained, they emit greenhouse gases at high rates. Habitats for many specialised, rare and endangered species, and highly threatened by hydrological changes caused by forest management and drainage.	Restoration of degraded bog and fen woodlands by raising the water level, regeneration of the acrotelm, the active peat zone containing living plants, removal of non-native tree species and renouncement of peat extraction.	Moore and Knowles 1989; Joosten 2012; EEA 2013; Joosten et al. 2015; EEA 2019
Dry sand pine forests	Lichen-rich dry pine forests on nutrient-poor, acidic sands with low shrub, herb, and litter cover. Being the result of historical land use (mainly litter raking and sod cutting) they depend on nutrient removal to accommodate typical epigeous (growing on the soil surface) lichen species. They are highly endangered, mainly due to discontinuation of litter raking, and by nitrogen deposition caused by agriculture and traffic emissions and have both high historic-cultural and biodiversity significance.	Protection of extant lichen-rich pine forests and restoration of degraded lichen-poor sand pine forests through litter and topsoil removal.	Heinken 1990; Heinken 2008; Fischer et al. 2009; Fischer et al. 2014; Brackel and Brackel 2016; Stefańska-Krzaczek et al. 2018
Beech forests	Naturally self-sustaining ecosystems dominated by beech (*Fagus sylvatica*), but commonly managed as productive high forests.	Prolonging of rotation cycles beyond conventional harvesting age, thus preserving old-growth-associated biodiversity, and enhancing natural regeneration.	Kroiher and Bolte 2015; Meyer et al. 2015; Winter et al. 2016

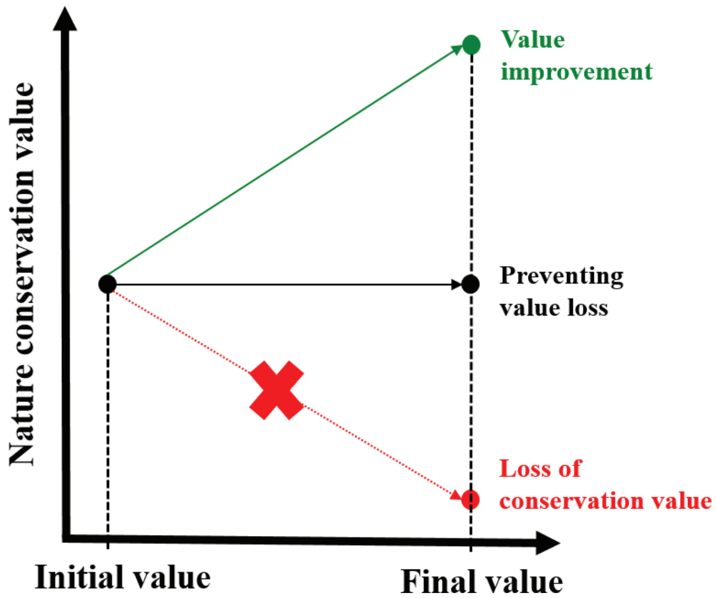

Figure 2. Development pathways of the initial nature conservation value.

Table 4. Description and assignment of the final nature conservation value (NCV) to the suitability assessment of conservation measures and the corresponding colour in Table 7 and Supplement S1.

Final nature conservation value	Description	Suitability of conservation measures	Colour
0	No NCV	Not suitable	Red
1	Very low NCV	Not suitable	Red
> 1 – 2	Low NCV	Not suitable	Red
> 2 – 3	Moderate NCV	Moderately suitable	Yellow
> 3 – 4	High NCV	Suitable	Light green
> 4 – 5	Very high NCV	Very suitable	Dark green

may achieve a high value when preservation measures have been implemented and have produced positive results, when degraded objects have been restored successfully (restoration measures), or when the objects have been newly created. A high conservation value towards the end of a contractual period indicates an improvement of an initially lower conservation value, or the prevention of value loss of an initially high value.

Contract-based funding would be particularly suitable for conservation objects with high initial conservation value that would suffer value loss in the absence of conservation measures, or for objects with rather low initial value but considerable restoration potential to achieve a higher final value. If the conservation value of a newly created conservation object (initial value = 0) was likely to increase over a given contract period, contract-based funding of conservation measures was also considered reasonable. If both initial value and restoration potential were low, contract-based conservation was deemed inappropriate. The suitability assessment is depicted as a four-level colour scheme, reflecting the final value (Table 4).

Results

Initial nature conservation value of forest conservation objects

More than 82% of all conservation objects were assessed as being highly or very highly worthy of preservation. However, only 39% had a high to very high need for protection and these were found exclusively within the group of objects of high to very high worthiness. Thus, some conservation objects can be regarded as very valuable, but are not seriously threatened, such as mesic beech forests or riparian alluvial forests (Suppl. material 1: Table S1). Forest structures and processes made up only a small proportion of all conservation objects. For forest structures, the proportion of low-value and non-threatened objects was higher than that of highly valuable and threatened ones, since many structures are being developed or newly implemented (e.g., the active supply of deadwood, or the designation of potential habitat trees).

One quarter of all forest conservation objects were assessed as having a high to very high initial nature conservation value (Fig. 3). The conservation objects coppice-with-standards, wood pastures, intact bog and fen woodlands, continuation of natural forest development, natural succession after large-scale disturbance, deadwood retention, eyrie tree protection (nesting sites for birds of prey) and protection of habitat trees were assessed as having very high conservation value. About three quarters of all conservation objects were ascribed a moderate to very high initial conservation value. Habitat types, comprising 55 out of the 67 identified forest conservation objects, made a major contribution to high conservation-value objects (initial value higher than 3; Table 5).

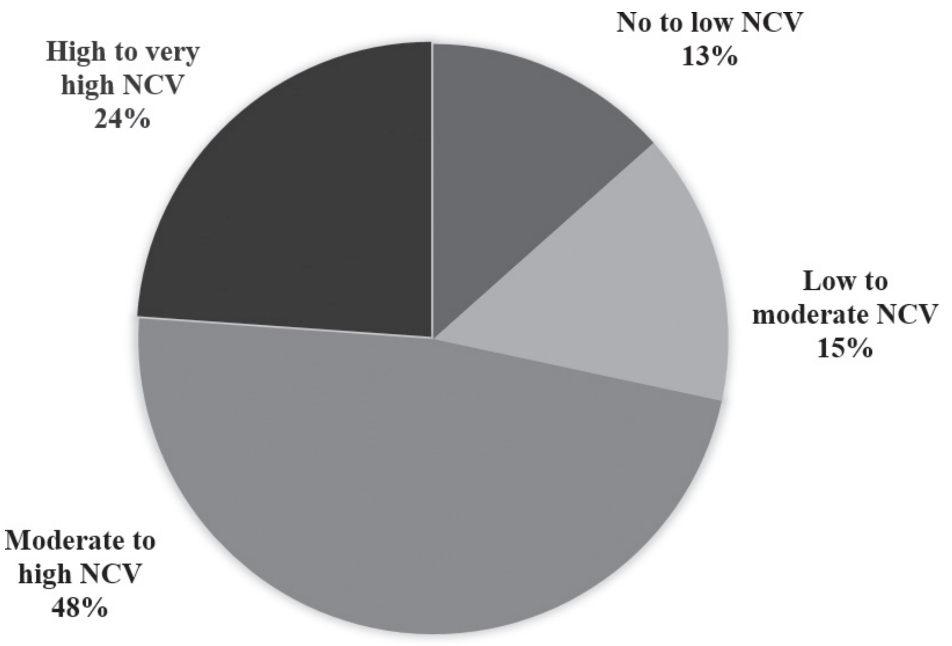

Figure 3. Initial nature conservation value (NCV) of all 67 forest conservation objects analysed.

Table 5. Distribution of the shares of the initial nature conservation value (NCV) classes for all 67 forest conservation objects.

Description	NCV	Habitat types	Structures	Processes
No to low NCV	0 – 2	3	5	1
Low to moderate NCV	> 2 – 3	9	0	1
Moderate to high NCV	> 3 – 4	32	0	0
High to very high NCV	> 4 – 5	11	3	2

Table 6. Suitability assessment proportions of forest conservation objects for different contract terms (years).

Suitability	Contract duration (years)	Forest conservation object group		
		Structures (8)	Processes (4)	Habitat types (55)
Not suitable	<10	1	0	22
	10–30	0	0	23
	>30	0	0	23
Moderately suitable	<10	0	1	3
	10–30	1	0	2
	>30	0	1	2
Suitable	<10	3	0	22
	10–30	0	2	18
	>30	1	0	18
Very suitable	<10	4	3	8
	10–30	7	2	12
	>30	7	3	12
Total proportion [%]		**11.9**	**6.0**	**82.1**

Suitability of contract-based forest conservation

As many as 42 out of 67 forest conservation objects proved suitable for contract-based conservation measures (Suppl. material 1: Table S1). Most of the assessed forest structures and processes were considered suitable or very suitable for contract-based conservation, irrespective of the contract period. For forest habitat types, accounting for the largest part of all assessed conservation objects, the findings are more nuanced. Short-term contracts (<10 years) were found to be very suitable for 15 out of 67 forest conservation objects (3 process-related, 4 structural and 8 habitat types; Table 6 and Suppl. material 1: Table S1). The conversion of forest stands of non-native trees, the continuation of traditional forest management (wood pastures, coppice-with-standards), and the retention of deadwood belong in this category. Mid-term contracts (10–30 years) were found to be very suitable for 31% of all conservation objects, including the resumption and continuation of traditional forest management, the restoration of degraded habitat types, the active creation of habitat trees, micro-habitats, as well as the conservation management of high-valued habitat types (Suppl. material 1: Table S1). Long-term contracts (>30 years) were assessed as being very suitable for about 33% of all conservation objects, mostly the same as for mid-term contractual periods, though with a few exceptions, such as the continuation of a natural forest development, or the retention of potential habitat trees. Contract-based agreements were rated not suitable for 34% of all conservation objects, regardless of the contractual period. This category includes almost exclusively habitat types, chiefly because they are either legally protected habitats (Box 1) or low-valued pioneers.

Table 7. Suitability assessment of representative forest conservation objects and conservation measures for different contract duration periods. For the scaling of the nature conservation value (NCV), based on worthiness of preservation and need for protection see Tables 1, 2 and 4.

		Forest conservation object	Possible conservation measure during contract period	Period (years)	Initial NCV: Worthiness	Initial NCV: Need	Final NCV: Increase in value with contract–based conservation	Final NCV: Loss of value without contract-based conservation	Suitability for contract-based conservation
Structural elements		**Deadwood**	Active deadwood provisioning to ensure continuous supply of a certain amount	< 10	0	0	4	No	s
				10–30			5		vs
				> 30			5		vs
			Retention of naturally supplied or silvicultural routine deadwood	< 10	5	5	5	Yes	vs
				10–30					vs
				> 30					vs
		Habitat trees	Retention of potential habitat trees	< 10	0	0	0–1	No	ns
				10–30			3		ms
				> 30			5		vs
			Initial creation of microhabitats	< 10	0	0	4	No	s
				10–30			5		vs
				> 30			5		vs
			Protection of habitat trees	< 10	5	5	5	Yes	vs
				10–30					vs
				> 30					vs
Processes		**Natural forest development**	Recent near-natural forest set-aside	< 10	3	3	3	Yes	ms
				10–30			4		s
				> 30			5		vs
			Continuation of natural forest development initiated several decades ago	< 10	5	5	5	Yes	s
				10–30					vs
				> 30					vs
		Natural succession after large-scale disturbance	Sites of wind-throws or other disturbances in native forests left to itself	< 10	5	5	5	Yes	vs
				10–30			4		s
				> 30			3		ms
Habitat types		**Coppice-with-standards**	Resumption of traditional coppice-with-standard management	< 10	4	3	3.5	Yes	s
				10–30			5		vs
				> 30			5		vs
			Continuation of coppice-with-standard management	< 10	5	5	5	Yes	vs
				10–30					vs
				> 30					vs
	Bog/fen woodland	**Intact bog and fen woodlands** *	Renouncement of degrading measures	< 10	5	5	not assessable	No	ns
				10–30					ns
				> 30					ns
		Degraded bog and fen woodlands	Restoration (rewetting)	< 10	4	3	4	Yes	s
				10–30			5		vs
				> 30			5		vs
	Dry pine forests	**Intact lichen-rich dry sand pine forest (*Cladino-Pinetum sylvestris*)** *	Conservation- and habitat-adapted management	< 10	5	4	4.5	Yes	vs
				10–30					vs
				> 30					vs
		Degraded (lichen-poor) dry sand pine forest	Restoration through litter and topsoil removal	< 10	3	3	4	No	s
				10–30			5		vs
				> 30			5		vs
	Beech forests	**Dry limestone beech forest (*Carici-Fagetum*)** *		< 10	5	4	4.5	Yes	vs
				10–30					vs
				> 30					vs
		Mesic beech forest on base-rich sites (*Galio odorati-Fagetum, Mercuriali perennis-Fagetum*)	Conservation- and habitat-adapted management	< 10	5	2	3.5	Yes	s
				10–30					s
				> 30					s

NCV, nature conservation value. Colours: red = not suitable (ns), yellow = moderately suitable (ms), light green = suitable (s), dark green = very suitable (vs). * = legally protected habitat (§30 BNatSchG).

For most suitable conservation objects (82%) contract duration was considered of little relevance. Nevertheless, longer funding durations are to be preferred. This would not apply, however, to wind-throws or other large-scale disturbances left to natural succession, because here, the early succession stages are the intended objective.

Forest conservation objects – case studies

Deadwood

Measures to actively supply deadwood were assumed to have a positive short- to long-term effect on the richness of saproxylic (depending on dead or decaying wood) organisms (Table 7). Therefore, short-term contracts were considered suitable. When contracting for mid-term periods, it should be considered that, due to decay, deadwood needs to be replenished to ensure continuous provisioning of different deadwood qualities (see deadwood estimation tool, Meyer et al. 2009). With further contractual period extension, the conservation value is expected to increase, provided that a continuous deadwood supply is guaranteed. Natural deadwood, or silvicultural routine deadwood, has a very high initial conservation value, making even short-term contracts very suitable. Mid- to long-term contracts to secure continuous deadwood supply would result in a very high conservation value.

Habitat trees

We considered trees with trunk diameter far beyond the typical harvest size (DBH > 80 cm for deciduous trees on normal sites, for oaks > 90 cm), and/or the site-specific harvesting age (e.g., beech > 200 a, oak > 300 a), as well as trees rich in microhabitats and/or with very large crowns or low crown bases, to be particularly qualified to become habitat trees (Table 7). As the natural formation of tree microhabitats was assumed to take >50 years at minimum (Larrieu et al. 2012), only long-term contracts qualify. Trees with microhabitats created through management measures have no initial object-specific conservation value (Table 7), but this may increase soon, making even short-term contracts reasonable. Mid- to long-term contracts were considered very suitable to achieve very high conservation value.

Natural forest development

Forests with long habitat continuity, where forestry ceased many decades ago, contribute considerably to the biodiversity of the natural landscape. Therefore, their worthiness was rated very high (Table 7). Due to their low presence in German forests (only 2.8% of the total forest area; Engel et al. 2019), their need for protection is also very high. The continued protection of forests with a long-lasting natural development was recommended for all contractual periods. Semi-mature forests that have been recently decommissioned have a moderate need for protection. Positive effects on biodiversity of such forests may only be measurable after many years or decades. Therefore, contract-based decommissioning of forests was assumed to be suitable for mid- to long-term periods only. Follow-up contracts were recommended for prolonged natural development.

Natural succession after large-scale disturbance

Natural forest succession after major disturbance events requires silviculturists to refrain from salvage logging, deadwood removal and replanting. Untouched early-successional stages are rarely found in privately-owned forests and are thus regarded as highly vulnerable (Table 7). As such pioneer habitats support numerous warmth- and light-dependent species, they are worthy and, consequently, of high initial conservation value. As disturbed areas decrease in object-specific conservation value over time, mid-term contracts were considered particularly suitable. Long-term contracts would only be meaningful if non-disturbed, surrounding stands are simultaneously targeted beyond the given conservation object.

Coppice-with-standards

Traditional coppice-with-standards woodlands can be protected from being transformed into high forests by continuing their specific management. As coppice-with-standards contribute much to the biodiversity of the natural landscape, they were granted a very high worthiness (Table 7). Due to their extreme rarity (less than 0.4% of the forest area in Germany; Albert and Ammer 2012) and susceptibility to management change, they were also assessed as having a very high need for protection and risk of value loss. Therefore, all contract terms were considered suitable, with long-terms preferred.

Abandoned and 'overstood' coppice-with-standards may be restored by resuming the former management. As a moderate loss of habitat continuity and species richness was assumed, their worth of, and need for, protection were given medium ratings (Table 7). Since one rotation cycle usually takes 20–30 years, short-term contracts do little to increase the conservation value of 'overstood' coppice-with-standards. More suitable contract periods are mid- to long-term.

Bog and fen woodlands

As part of the landscape's natural vegetation, intact bog and fen woodlands have a very long habitat continuity and, consequently, very high worthiness. Due to their high level of endangerment, they also have an urgent need for protection (Table 7). Intact bog and fen woodlands have been protected under the Federal Nature Conservation Act. As mere preservation is not compensable (Box 1), contract-based conservation was considered unsuitable, unless combined with additional measures. As remnant or slightly degraded bog and fen woodlands may still contribute to the biodiversity of the natural landscape, they have been assigned medium to high worthiness and medium need for protection (Table 7). Because the restoration of slightly degraded bog and fen woodlands promptly leads to a value increase, even short-term contracts were deemed to be adequate.

Box 1. Legally protected habitat types.

Special case: Legally protected habitat types

Some German forest habitat types are legally protected according to § 30 BNatSchG. These are primarily natural and self-sustaining habitat types that do not require management, and include among others fen and bog woodlands, riparian forests, forests of ravines, slopes and screes, and xerothermic forests and shrub lands. Destruction or actions with significant adverse effects are prohibited by law. Forest owners are obliged to protect and maintain these habitats and to refrain from destruction or considerable impairment. Private land owners cannot be compensated for fulfilling these legal obligations. In contrast, for habitat types that rely on active conservation measures, such as mixed oak forests derived from coppicing, financial compensation appears reasonable.

Likewise, for restoration of degraded habitat types, such as drained swamp forests, financial compensation is possible. The successful restoration of degraded habitats may result in permanent restriction of the forest owner's right of disposal once the status of a legally protected habitat is reached. Franz et al. (2018a) argued that, for reasons of fairness, this permanent use restriction should be permanently compensated.

Dry sand pine forests

The qualitative contribution of lichen-rich dry sand pine forests to the biodiversity of the natural landscape was top-rated and, consequently, their worthiness was also high (Table 7). Being endangered, they have a very high need for protection. However, as a legally protected habitat type, forest owners cannot be compensated for its mere preservation (Box 1). Contract-based maintenance was therefore considered unsuitable unless combined with extra measures, such as rotational litter and topsoil removal.

For degraded forms, if still restorable and credited with medium conservation value, financial compensation for measures to initiate recolonisation of characteristic lichen species was recommended. Short-term contracts were considered suitable, although long-term contracts rendered higher conservation value.

Beech forests

A long habitat continuity and high relevance for the biodiversity of the natural landscape were assumed to result in very high worthiness (Table 7). Our assessment is that financial compensation for preservation-friendly management of dry and mesic beech forest complexes is highly recommendable, whatever the contractual period, if it clearly extends beyond regular forestry practice.

Discussion

Assessing the nature conservation value of forest conservation objects

By means of various indicators or criteria, evaluating conservation objects may be understood as the transfer of factual knowledge to a valuation scheme (Plachter 1991; Schultze et al. 2016). This valuation approach has formed the basis of many studies that have applied scoring techniques (Usher 1994; Gastauer et al. 2013; Capmourteres and Anand 2016; IUCN 2016), and we used it to develop our framework of reasonable and operational measures to assess the nature conservation value of forest conservation objects.

Our conservation valuation comprises different attributes, with single summarised scores, to allow for its country-wide application. With contextual modifications such as other Red List levels to specify the need for protection, the approach may be applicable in yet other regions. By including forest structures, processes, and habitat types, we tried to cover relevant attributes of forest biodiversity. The selected conservation objects are representative for forest conservation management and include those in urgent need of conservation actions. They are particularly relevant in times of climate change, as they encompass short-term objects (e.g., wind-throw sites), climax habitat types (e.g., beech forests), habitats of carbon sink relevance (bog and fen woodlands), habitats with climate-sensitive species (e.g., dry pine forests), and habitats with considerable economic potential for financial risk spreading (coppice-with-standards).

Contract duration to safeguard forest conservation objects

We showed that contractual agreements can be appropriate to support conservation measures in forests. The evaluation of 67 forest conservation objects showed that contract-based conservation agreements prove suitable for 42 objects, albeit with different contract durations. Short-term contracts are less suitable for the retention of habitat trees and for decommissioning semi-mature forests, while long-term contracts are not recommended for funding natural succession after large-scale disturbance. Contract-based conservation is particularly suitable for high-valued objects, such as coppice-with-standards, that depend on active conservation measures to prevent deterioration. Even short-term contracts may be adequate in cases of objects with low to medium initial conservation value if a prompt value increase is to be expected, e.g., newly created habitat trees. In contrast, short-term contracts are less meaningful for conservation objects with low initial conservation value and slow value improvement.

Permanent compensation and long-term agreements would be required for private owners of forests under permanent statutory use restriction (e.g., in bog and fen woodlands). A short contract duration, covering only initial investment expenses but no further maintenance measures, would fail to produce a return on landowner's investment. However, if there is a general willingness of forest owners to accept follow-up contracts, and if suitable funding resources are available, short-term contracts are better than no agreement.

Consequences for nature conservation and forestry practice

As far as forest habitat types are concerned, our conservation objects are in line with the EU Habitats Directive (Natura 2000) and the European Nature Information System (EUNIS) classification (Suppl. material 3: Table S3) and our approach may help to improve the mandatory assessment of the conservation status. In the EU Natura 2000 network, the preservation of diverse forest structures (e.g., deadwood, habitat trees) is a necessary element for a particular forest habitat type to achieve favourable conservation status (Winkel et al. 2015; Alberdi et al. 2019). Since a high proportion of European forest habitat types have been assigned an unfavourable conservation status (European Commission 2015), enhancing these forest structures helps to improve their conservation status.

Our suitability assessment revealed that the conservation or restoration of forest conservation objects may have synergetic effects and simultaneously result in the protection and improvement of other objects. These synergies should be given special consideration (Margules and Pressey 2000; Cimon-Morin et al. 2013). Potential trade-offs and competing objectives across conservation objects should be weighed in the light of the conservation objectives, site conditions and the expected value development. For instance, natural forest development and coppice-with-standards management cannot be implemented in the same site. In general, forest owners cannot meet all possible conservation objectives in a single stand. A contract usually covers a single conservation object and the necessary measures (setting, extent, feasibility, financial framework), but several contracts may be concluded for different objectives in the same forest stand.

Given an underlying value structure that aims to protect typical regional forest biodiversity, the responsibility to protect can only be justified for native species appropriate to the site and location, long-term natural and semi-natural processes and structures, and the cultural development history (Meyer 2013). Consequently, management in privately-owned non-cultural types of forest should be committed to close-to-nature forestry (extension of rotation periods, deadwood provisioning, and tree retention). Since this paradigm shift may cause additional costs for forest owners, suitable compensation structures are needed.

However, financial incentive systems in privately-owned forests are as yet lacking in Germany (Seintsch et al. 2018). Other countries successfully developed their own subsidy programmes, such as the 'English Woodland Grant Scheme' introduced in 2005 (Forestry Commission 2010; Fuentes-Montemayor et al. 2015), replaced in 2015 by 'Countryside Stewardship' grants. Such a country-wide system can lead to more transparency and acceptance among forest owners to support forest biodiversity conservation. Although some German federal states have developed their own incentive instruments, there is substantial variability in requirements and capacity for funding across states. For instance, the Bavarian contract-based forest conservation programme supports the conservation of coppice-with-standards woodlands, the preservation of habitat trees and deadwood. In Hesse, forest conservation measures are funded by the Natura 2000 Foundation, but only within the Natura 2000 network. Additional funding options with differing requirements and payment amounts exist in Germany, yet none of these have nation-wide applicability (BMEL 2019; European Commission 2020). Unfortunately, the operational implementation of these general systems has by no means reached the individual private forest owner. Franz et al. (2018a) pointed out that there is an urgent need for action and to create the prerequisites for contract-based conservation in privately-owned forests, such as a solid foundation of trust, the involvement of committed intermediaries, result-oriented payments, success bonuses, as well as the identification of suitable indicators. Our comprehensive catalogue of forest conservation objects and measures eligible for contract-based funding is valid throughout Germany and in line with the Federal Compensation Directive (BMU 2020) just published. It does not, however, explain the possible trajectories between initial and final conservation values of objects. Forest owners are encouraged to use our catalogue for their conservation intentions. Given that they know the tree species composition and structural characteristics of their forest stands, they can easily identify conservation objects such as potential habitat trees, and choose a reasonable contract duration. The biggest challenge yet for contract-based nature conservation is to find suitable funding options, which vary between the German federal states. Authorities, nature conservation agencies, or NGOs might assist on this point. Therefore, while this paper provides a rationale and an objective-related design for contract-based nature conservation on forests, it can't guide private forest owners towards an operational implementation. Such a guidance, generalised at the level of administrative units or federal states, remains yet to be elaborated.

Conclusions

The nature conservation value assessment of forest conservation objects provided in this paper enables forest owners to assess the conservation value of objects in their forest stands and to consider options for contract-based nature conservation, specifically in privately-owned forests in Germany. We also touch upon the much-discussed topic of conservation responsibility. We believe that the comprehensive catalogue of forest conservation objects and measures may be applicable in a wider Central European context. Furthermore, the nature conservation value assessment can help to improve the conservation status of Natura 2000 forest habitat types. We showed the suitability of many conservation objects to financial incentives and advocate conservation object-dependent variation in contract duration. We noticed a particular need for action in the case of conservation objects susceptible to an imminent loss of value in the absence of conservation measures.

Currently, however, a general framework for successful implementation of contract-based forest conservation, including factors such as legal security, fairness, continuity, and flexibility, is not available. The reference framework presented here and the considerable number of combinations of objects and measures found suitable for contract-based conservation, together with the recommendations for a forest conservation funding system given by WBW and WBBGR (2020), may help to enhance this implementation process. For the sake of diversified nature conservation in forests, politicians and stakeholders at all governmental levels should rethink and revise benefit payment programmes towards mid- to long-term contracts (Gemeinholzer et al. 2019), and thus encourage private forest owners to acknowledge biodiversity-related funding.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Agnew JM, Rao S (2014) The creation of structural diversity and deadwood habitat by ringbarking in a Scots pine *Pinus sylvestris* plantation in the Cairngorms, UK. Conservation Evidence 1: 43–47. https://www.conservationevidence.com/individual-study/5487

AK Waldökologie GfÖ (2019) Wissenschaftler sehen Störungen in Wäldern auch als Chance für biologische Vielfalt und Klimaanpassung. Gesellschaft für Ökologie. [Retrieved August 25, 2020] https://gfoe.org/de/node/69

Alberdi I, Nunes L, Kovac M, Bonheme I, Cañellas I, Castro Rego F, Dias S, Duarte I, Notarangelo M, Rizzo M, Gasparini P (2019) The conservation status assessment of Natura 2000 forest habitats in Europe: Capabilities, potentials and challenges of national forest inventories data. Annals of Forest Science 76(2): 1–34. https://doi.org/10.1007/s13595-019-0820-4

Albert K, Ammer C (2012) Biomasseproduktivität ausgewählter europäischer Mittel- und Niederwaldbestände – Ergebnisse einer vergleichenden Metaanalyse. Allgemeine Forst- und Jagdzeitung 183: 225–237.

Asbeck T, Pyttel P, Frey J, Bauhus J (2019) Predicting abundance and diversity of tree-related microhabitats in Central European montane forests from common forest attributes. Forest Ecology and Management 432: 400–408. https://doi.org/10.1016/j.foreco.2018.09.043

Bärnthol R (2003) Nieder- und Mittelwald in Franken. Waldwirtschaftsformen aus dem Mittelalter. Fränkisches Freilandmuseum, Bad Windsheim.

Bergmeier E, Petermann J, Schröder E (2010) Geobotanical survey of wood-pasture habitats in Europe: Diversity, threats and conservation. Biodiversity and Conservation 19(11): 2995–3014. https://doi.org/10.1007/s10531-010-9872-3

BMEL Bundesministerium für Ernährung und Landwirtschaft (2019) Rahmenplan der Gemeinschaftsaufgabe „Verbesserung der Agrarstruktur und des Küstenschutzes" 2019–2022. Berlin.

BMU Bundesministerium für Umwelt, Naturschutz, und Reaktorsicherheit (2007) Nationale Strategie zur biologischen Vielfalt, Bonn.

BMU Bundesministerium für Umwelt, Naturschutz, und Reaktorsicherheit (2020) Bundeskompensationsverordnung (BKompV). Deutscher Bundestag, Drucksache 19/17344.

Capmourteres V, Anand M (2016) "Conservation value": A review of the concept and its quantification. Ecosphere 7(10): e01476. https://doi.org/10.1002/ecs2.1476

Cimon-Morin J, Darveau M, Poulin M (2013) Fostering synergies between ecosystem services and biodiversity in conservation planning: A review. Biological Conservation 166: 144–154. https://doi.org/10.1016/j.biocon.2013.06.023

Davies ZG, Tyler C, Stewart GB, Pullin AS (2008) Are current management recommendations for saproxylic invertebrates effective? A systematic review. Biodiversity and Conservation 17(1): 209–234. https://doi.org/10.1007/s10531-007-9242-y

Demant L, Meyer P, Sennhenn-Reulen H, Walentowski H, Bergmeier E (2019) Seeking consensus in German forest conservation: An analysis of contemporary concepts. Nature Conservation 35: 1–23. https://doi.org/10.3897/natureconservation.35.35049

EEA (2013) 91D0 Bog woodland – Report under the Article 17 of the Habitats Directive Period 2007–2012. EEA, European Environment Agency.

EEA (2019) Habitat assessments at EU biogeographical level. EEA, European Environment Agency. [Retrieved August 3, 2019] https://bd.eionet.europa.eu/article17/reports2012/habitat/summary/?period=3&subject=91D0

Engel F, Bauhus J, Gärtner S, Kühn A, Meyer P, Reif A, Schmidt M, Schultze J, Späth V, Stübner S, Wildmann S, Spellmann H (2016) Wälder mit natürlicher Entwicklung in Deutschland: Bilanzierung und Bewertung. Naturschutz und Biologische Vielfalt 145: 1–221.

Engel F, Meyer P, Demant L, Spellmann H (2019) Wälder mit natürlicher Entwicklung in Deutschland. AFZ, Der Wald 74: 30–33.

European Commission (2015) Natura 2000 and Forests Part I-II. Technical report 2015, 88. European Commission, DG Environment.

European Commission (2020) Rural development. [Retrieved February 26, 2020] https://ec.europa.eu/info/food-farming-fisheries/key-policies/common-agricultural-policy/rural-development

Eurostat (2018) Agriculture, forestry and fishery statistics. 2018 edition. doi:10.2785/340432

Fedrowitz K, Koricheva J, Baker SC, Lindenmayer DB, Palik B, Rosenvald R, Beese W, Franklin JF, Kouki J, Macdonald E, Messier C, Sverdrup-Thygeson A, Gustafsson L (2014) Can retention forestry help conserve biodiversity? A meta-analysis. Journal of Applied Ecology 51(6): 1669–1679. https://doi.org/10.1111/1365-2664.12289

Finck P, Heinze S, Raths U, Riecken U, Ssymank A (2017) Rote Liste der gefährdeten Biotoptypen Deutschlands – dritte fortgeschriebene Fassung (3rd ed.) Naturschutz und Biologische Vielfalt. Landwirtschaftsverlag Münster, Bonn-Bad Godesberg.

Fischer P, Heinken T, Meyer P, Schmidt M, Waesch G (2009) Zur Abgrenzung und Situation des FFH-Lebensraumtyps "Mitteleuropäische Flechten-Kiefernwälder" (91TO) in Deutschland. Natur und Landschaft 84: 281–287.

Fischer P, Bültmann H, von Drachenfels O, Heinken T, Waesch G (2014) Rückgang der Flechten-Kiefernwälder in Niedersachsen seit 1990. Informationsdienst Naturschutz Niedersachsen 34(1): 54–65.

Forestry Commission (2010) English Woodland Grant Scheme - EWGS Summary. [Retrieved October 1, 2020] http://adlib.everysite.co.uk/adlib/defra/content.aspx?doc=122813&id=122814

Franklin JF, Spies TA, Van Pelt R, Carey AB, Thornburgh DA, Berg DR, Lindenmayer DB, Harmon ME, Keeton WS, Frank DC, Bible K, Chen J (2002) Disturbances and structural development of natural forest ecosystems with slivicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management 155(1–3): 399–423. https://doi.org/10.1016/S0378-1127(01)00575-8

Franz K, Dieter M, Möhring B (2017) Naturschutzförderung neu gedacht. AFZ, Der Wald 72: 44–47.

Franz K, von Blomberg M, Demant L, Dieter M, Lutter C, Meyer P, Möhring B, Paschke M, Seintsch B, Selzer AM, Spellmann H (2018a) Perspektiven für den Vertragsnaturschutz. AFZ, Der Wald 73: 30–33.

Franz K, von Blomberg M, Demant L, Lutter C, Seintsch B, Selzer AM (2018b) Umsetzung von Vertragsnaturschutz im deutschen Wald. AFZ, Der Wald 73: 13–15.

Frenz W, Müggenborg HJ (2016) Bundesnaturschutzgesetz Kommentar (2nd ed.). Erich Schmidt Verlag, Berlin.

Fuentes-Montemayor E, Peredo-Alvarez VM, Watts K, Park KJ (2015) Are woodland creation schemes providing suitable resources for biodiversity? Woodland moths as a case study. Biodiversity and Conservation 24(12): 3049–3070. https://doi.org/10.1007/s10531-015-0997-2

Gastauer M, Trein L, Meira-Neto JAA, Schumacher W (2013) Evaluation of biotope's importance for biotic resource protection by the Bonner Approach. Ecological Indicators 24: 193–200. https://doi.org/10.1016/j.ecolind.2012.06.014

Geitzenauer M, Blondet M, De Koning J, Ferranti F, Sotirov M, Weiss G, Winkel G (2017) The challenge of financing the implementation of Natura 2000-Empirical evidence from six European Union Member States. Forest Policy and Economics 82: 3–13. https://doi.org/10.1016/j.forpol.2017.03.008

Gemeinholzer B, Demant L, Dieterich M, Eser U, Farwig N, Geske C, Feldhaar H, Lauterbach D, Reis M, Weisser W, Werk L (2019) Artenschwund trotz Naturschutz. Biologie in Unserer Zeit 49(6): 444–455. https://doi.org/10.1002/biuz.201910689

Groß P, Konold W (2009) Mittelwald als Agroforstsystem zwischen geordneter Nachhaltigkeit und Gestaltungsvielfalt – Eine historische Studie. Allgemeine Forst- und Jagdzeitung 181: 64–71.

Gustafsson L, Bauhus J, Asbeck T, Augustynczik ALD, Basile M, Frey J, Gutzat F, Hanewinkel M, Helbach J, Jonker M, Knuff A, Messier C, Penner J, Pyttel P, Reif A, Storch F, Winiger N, Winkel G, Yousefpour R, Storch I (2019) Retention as an integrated biodiversity conservation approach for continuous-cover forestry in Europe. Ambio 49(1): 85–97. https://doi.org/10.1007/s13280-019-01190-1

Güthler W, Market R, Häusler A, Dolek M (2005) Vertragsnaturschutz im Wald – Bundesweite Bestandsaufnahme und Auswertung. BfN-Skripten 146: e182.

Harmon ME, Franklin JF, Swanson FJ, Sollins P, Gregory SV, Lattin JD, Anderson NH, Cline SP, Aumen NG, Sedell JR, Lienkaemper GW, Cromack K, Cummins KW (1986) Ecology of coarse woody debris in temperate ecosystems. Advances in Ecological Research. Academic Press 15: 133–302. https://doi.org/10.1016/S0065-2504(08)60121-X

Heinken T (1990) Pflanzensoziologische und ökologische Untersuchungen offener Sandstandorte im östlichen Aller-Flachland (Ost-Niedersachsen). Tuexenia 10: 223–257.

Heinken T (2008) Synopsis der Pflanzengesellschaften Deutschlands Heft 10 Vaccinio-Piceetea (H7) Beerstrauch-Nadelwälder Teil 1: Dicrano-Pinion – Sand- und Silikat-Kiefernwälder. Floristisch-soziologische Arbeitsgemeinschaft. Göttingen.

IUCN (2013) Threats Classification Scheme (Version 3.2). IUCN, International Union for Conservation of Nature. [Retrieved June, 6, 2020] https://www.iucnredlist.org/resources/threat-classification-scheme

IUCN (2016) A global standard for the identification of Key Biodiversity Areas, Version 1.0. 1st. ed. IUCN, International Union for Conservation of Nature, Gland, Switzerland.

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, Gardfjel 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. ISBN 978-92-79-61588-7. doi: 10.2779/091372

Joosten H (2012) Zustand und Perspektiven der Moore weltweit. Natur und Landschaft 87: 50–55.

Joosten H, Berghöfer A, Couwenberg J, Dietrich K, Holsten B, Permien T, Schäfer A, Tanneberger F, Trepel M, Wahren A (2015) Die neuen MoorFutures – Kohlenstoffzertifikate mit ökologischen Zusatzleistungen. Natur und Landschaft 90: 170–175.

Kirby KJ, Buckley GP, Mills J (2017) Biodiversity implications of coppice decline, transformations to high forest and coppice restoration in British woodland. Folia Geobotanica 52(1): 5–13. https://doi.org/10.1007/s12224-016-9252-1

Kraus D, Krumm F (2013) Integrative approaches as an opportunity for the conservation of forest biodiversity. European Forest Institute Freiburg.

Kraus D, Bütler R, Krumm F, Lachat T, Larrieu L, Mergner U, Paillet Y, Rydkvist T, Schuck A, Winter S (2016) Cataloque of tree microhabitats – Reference field list. Integrate + Technical Paper. European Forest Institute Freiburg.

Kroiher F, Bolte A (2015) Naturschutz und Biodiverstität im Spiegel der BWI 2012. AFZ, Der Wald 70: 23–27.

Larrieu L, Cabanettes A, Delarue A (2012) Impact of silviculture on dead wood and on the distribution and frequency of tree microhabitats in montane beech-fir forests of the Pyrenees. European Journal of Forest Research 131(3): 773–786. https://doi.org/10.1007/s10342-011-0551-z

Larrieu L, Paillet Y, Winter S, Bütler R, Kraus D, Krumm F, Lachat T, Michel AK, Regnery B, Vandekerkhove K (2018) Tree related microhabitats in temperate and Mediterranean European forests: A hierarchical typology for inventory standardization. Ecological Indicators 84: 194–207. https://doi.org/10.1016/j.ecolind.2017.08.051

Lassauce A, Paillet Y, Jactel H, Bouget C (2011) Deadwood as a surrogate for forest biodiversity: Meta-analysis of coorelations between deadwood volume and species richness of saproxylic organisms. Ecological Indicators 11(5): 1027–1039. https://doi.org/10.1016/j.ecolind.2011.02.004

Leuschner C, Ellenberg H (2017) Ecology of Central European Forests – Vegetation Ecology of Central Europe Volume I. Springer International Publishing, Cham. https://doi.org/10.1007/978-3-319-43042-3

Lindenmayer DB, Burton PJ, Franklin JF (2008) Salvage Logging and Its Ecological Consequences. Island Press, Washington.

Lindenmayer DB, Laurance WF, Franklin JF (2012) Global Decline in Large Old Trees. Science 338(6112): 1305–1306. https://doi.org/10.1126/science.1231070

Margules CR, Pressey RL (2000) Systematic conservation planning. Nature 405(6783): 243–253. https://doi.org/10.1038/35012251

Meyer P (2013) Biodiversität im Wald – Aus der Sicht der Forstwissenschaft. AFZ, Der Wald 68: 24–25.

Meyer P, Schmidt M (2008) Aspekte der Biodiversität von Buchenwäldern – Konsequenzen für eine naturnahe Bewirtschaftung. Beiträge aus der NW-FVA 3: 159–162.

Meyer P, Menke N, Nagel J, Hansen J, Kawaletz H, Paar U, Evers J (2009) Abschlussbericht des von der Deutschen Bundesstiftung Umwelt geförderten Projekts – Entwicklung eines Managementmoduls für Totholz im Forstbetrieb, Abschlussbericht. Abschlussbericht des von der Deutschen Bundesstiftung Umwelt geförderten Projektes. http://www.nw-fva.de/?id=234

Meyer P, Spellmann H, Nagel RV, Fischer C (2015) „Gräben vertiefen, wo Brücken gebaut werden müssen" – Stellungnahme zum Beitrag ‚Alte Buchenwälder nehmen in Hessen drastisch ab' von Norbert Panek, Agenda zum Schutz deutscher Buchenwälder in Naturschutz und Landschaftsplanung 47(4): 124–125. Naturschutz und Landschaftsplanung 47(7): 225–226.

Meyer P, Schmidt M, Lorenz K, Bedarff U (2018) Vergleich von Artenvielfalt, Vegetation und Waldstruktur des Mittelwaldes „Heißum" und des Hochwaldes „Lewer Berg" im Niedersächsischen Forstamt Liebenburg. NW-FVA, Göttingen. https://www.nw-fva.de/index.php?id=683

Mölder A, Schmidt M, Plieninger T, Meyer P (2020) Habitat-tree protection concepts over 200 years. Conservation Biology 34(6): 1444–1451. https://doi.org/10.1111/cobi.13511

Moore TR, Knowles R (1989) The influence of water table levels on methane and carbon dioxide emissions from peatland soils. Canadian Journal of Soil Science 69(1): 33–38. https://doi.org/10.4141/cjss89-004

Müller J, Noss RF, Thorn S, Bässler C, Leverkus AB, Lindenmayer D (2019) Increasing disturbance demands new policies for intact forests. Conservation Letters 12(1): 1–7. https://doi.org/10.1111/conl.12449

Nordén B, Dahlberg A, Brandrud TE, Fritz Ö, Ejrnaes R, Ovaskainen O (2014) Effects of ecological continuity on species richness and composition in forests and woodlands: A review. Ecoscience 21(1): 34–45. https://doi.org/10.2980/21-1-3667

Paillet Y, Pernot C, Boulanger V, Debaive N, Fuhr M, Gilg O, Gosselin F (2015) Quantifying the recovery of old-growth attributes in forest reserves: A first reference for France. Forest Ecology and Management 346: 51–64. https://doi.org/10.1016/j.foreco.2015.02.037

Plachter H (1991) Naturschutz. UTB, G. Fischer, Stuttgart.

Plieninger T, Hartel T, Martin-Lopez B, Beaufoy G, Bergmeier E, Kirby K, Montero MJ, Moreno G, Oteros-Rozas E, Van Uytvanck J (2015) Wood-pastures of Europe: Geographic coverage, social-ecological values, conservation management, and policy implications. Biological Conservation 190: 70–79. https://doi.org/10.1016/j.biocon.2015.05.014

Polley H, Hennig P, Kroiher F, Marks A, Riedel T, Schmidt U, Schwitzgebel F, Stauber T (2016) Der Wald in Deutschland: Ausgewählte Ergebnisse der dritten Bundeswaldinventur. Federal Ministry of Food and Agriculture, Berlin, Germany.

Runkle JR (1989) Synchrony of Regeneration, Gaps, and Latitudinal Differences in Tree Species Diversity. Ecology 70(3): 546–547. https://doi.org/10.2307/1940199

Schultze J, Reif A, Gärtner S, Bauhus J, Engel F, Späth V (2016) Naturschutzfachliche Bewertung der Wälder mit natürlicher Entwicklung in Deutschland. Naturschutz und Biologische Vielfalt 145: 75–150.

Seibold S, Bässler C, Brandl R, Gossner MM, Thorn S, Ulyshen MD, Müller J (2015) Experimental studies of dead-wood biodiversity – A review identifying global gaps in knowledge. Biological Conservation 191: 139–149. https://doi.org/10.1016/j.biocon.2015.06.006

Seidl R, Thom D, Kautz M, Martin-Benito D, Peltoniemi M, Vacchiano G, Wild J, Ascoli D, Petr M, Honkaniemi J, Lexer MJ, Trotsiuk V, Mairota P, Svoboda M, Fabrika M, Nagel TA, Reyer CPO (2017) Forest disturbances under climate change. Nature Climate Change 7(6): 395–402. https://doi.org/10.1038/nclimate3303

Seintsch B, Franz K, Meyer P, Möhring B, Paschke M (2018) Das WaVerNa-Forschungsprojekt im Überblick. AFZ, Der Wald 73: 10–12.

Sotirov M (2017) Natura 2000 and forests: Assessing the state of implementation and effectiveness. What Science Can Tell Us 7. European Forest Institute, 146 pp.

Stefańska-Krzaczek E, Fałtynowicz W, Szypuła B, Kącki Z (2018) Diversity loss of lichen pine forests in Poland. European Journal of Forest Research 137(4): 419–431. https://doi.org/10.1007/s10342-018-1113-4

Swanson ME, Franklin JF, Beschta RL, Crisafulli CM, DellaSala DA, Hutto RL, Lindenmayer DB, Swanson FJ (2011) The forgotten stage of forest succession: Early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment 9(2): 117–125. https://doi.org/10.1890/090157

Thorn S, Bässler C, Brandl R, Burton PJ, Cahall R, Campbell JL, Castro J, Choi CY, Cobb T, Donato DC, Durska E, Fontaine JB, Gauthier S, Hebert C, Hothorn T, Hutto RL, Lee EJ, Leverkus AB, Lindenmayer DB, Obrist MK, Rost J, Seibold S, Seidl R, Thom D, Waldron K, Wermelinger B, Winter MB, Zmihorski M, Müller J (2018) Impacts of salvage logging on biodiversity: A meta-analysis. Journal of Applied Ecology 55(1): 279–289. https://doi.org/10.1111/1365-2664.12945

Unrau A, Becker G, Spinelli R, Lazdina D, Magagnotti N, Nicolescu VN, Buckley P, Bartlett D, Kofman PD (2018) Coppice forests in Europe. Albert Ludwig University, Freiburg.

Usher MB (1994) Erfassen und Bewerten im Naturschutz. Quelle & Meyer, Heidelberg, Wiesbaden.

Vandekerkhove de K, Keersmaeker L, Walleyn R, Köhler F, Crevecoeur L, Govaere L, Thomaes A, Verheyen K (2011) Reappearance of Old-Growth Elements in Lowland Woodlands in Northern Belgium: Do the Associated Species Follow? Silva Fennica 45(5): 909–935. https://doi.org/10.14214/sf.78

von Brackel W, von Brackel J (2016) Ein Pilotversuch zur Wiederherstellung von Flechten-Kiefernwäldern. ANLiegen Natur 38: 1–9.

Walentowski H, Winter S (2007) Naturnähe im Wirtschaftswald – was ist das? Tuexenia 27: 19–26.

WBW and WBBGR (2020) Wege zu einem effizienten Waldnaturschutz in Deutschland. Stellungnahme. WBW and WBBGR, Wissenschaftlicher Beirat Waldpolitik und Wissenschaftlicher Beirat Biodiversität und Genetische Ressourcen. Berlin, 62 S. [Retrieved April, 21, 2020] https://www.bmel.de/SharedDocs/Downloads/DE/_Ministerium/Beiraete/biodiversitaet/stellungnahme-effizienter-waldnaturschutz.html

WCED (1987) Report of the World Commission on Environment and Development: Our Common Future. WCED, World Commission on Environment and Development. Oxford University Press, Oxford.

Winkel G, Blondet M, Borrass L, Frei T, Geitzenauer M, Gruppe A, Jump A, De Koning J, Sotirov M, Weiss G, Winter S, Turnhout E (2015) The implementation of Natura 2000 in forests: A trans- and interdisciplinary assessment of challenges and choices. Environmental Science & Policy 52: 23–32. https://doi.org/10.1016/j.envsci.2015.04.018

Winter S, Möller GC (2008) Microhabitats in lowland beech forests as monitoring tool for nature conservation. Forest Ecology and Management 255(3–4): 1251–1261. https://doi.org/10.1016/j.foreco.2007.10.029

Winter S, Begehold H, Herrmann M, Lüderitz M, Möller G, Rzanny M, Flade M (2016) Praxishandbuch – Naturschutz im Buchenwald Naturschutzziele und Bewirtschaftungsempfehlungen für reife Buchenwälder Nordostdeutschlands (2nd ed.). Land Brandenburg.

Supplementary material 1

Table S1

Authors: Laura Demant, Erwin Bergmeier, Helge Walentowski, Peter Meyer

Data type: Table

Explanation note: Suitability assessment of representative forest conservation objects and conservation measures for different contract duration periods. For the scaling of the nature conservation value (NCV), based on worthiness of preservation and need for protection see Tables 1, 2 and 4. German Red List Status 1! = critically endangered (acutely), 1 = critically endangered, 1-2 = endangered to critically, 2 = endangered, 2-3 = vulnerable to endangered, 3 = vulnerable, 3-V = near threatened to vulnerable, V = near threatened, * = no current risk of loss trend (least concern). Colors: red = not suitable (ns), yellow = moderately suitable (ms), light green = suitable (s), dark green = very suitable (vs). # = legally protected habitat (§30 BNatSchG).

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Supplementary material 2

Table S2

Authors: Laura Demant, Erwin Bergmeier, Helge Walentowski, Peter Meyer

Data type: Table

Explanation note: Proportions of the worthiness of preservation and need for protection of all forest conservation objects (FCO), and for each group.

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Supplementary material 3

Table S3

Authors: Laura Demant, Erwin Bergmeier, Helge Walentowski, Peter Meyer

Data type: Table

Explanation note: German Red List Status (1! = critically endangered (acutely), 1-2 = endangered to critically endangered, 2-3 = vulnerable to endangered, V = near threatened), Natura 2000 assignment and EUNIS (European Nature Information System) classification of the exemplary FCOs.

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