Nature Conservation 48: 1–29 (2022)
doi: 10.3897/natureconservation.48.76273
https://natureconservation.pensoft.net

RESEARCH ARTICLE

A peer-reviewed open-access journal
Nature Conservation
Launched to accelerate biodiversity conservation

Temporal and spatial changes of biodiversity in Caverns of Heaven and Places of Blessing, Zhejiang Province, China from 1990 to 2020

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Academic editor: Ji-Zhong Wan | Received 8 October 2021 | Accepted 29 January 2022 | Published 1 April 2022

http://zoobank.org/D520C587-0320-4FDE-A66B-8041D6BA8335

Citation: Zhu Y, Jin H, Zhong L (2022) Temporal and spatial changes of biodiversity in Caverns of Heaven and Places of Blessing, Zhejiang Province, China from 1990 to 2020. Nature Conservation 48: 1–29. https://doi.org/10.3897/natureconservation.48.76273

Abstract

Caverns of Heaven and Places of Blessing (CHPB) are the earliest Ecological Reserve in China, but in recent years, due to the accelerated process of urbanization and weak protection, the Chinese traditional ecological reserve represented by CHPB has been damaged to a certain extent. How to accurately measure the dynamic changes of ecological value in existing ecological protection and construct is an initial topic of CHPB protection. To understand the temporal and spatial changes characteristics of biodiversity in CHPB, this paper selects three-time nodes in 1990, 2005, and 2020, and takes CHPB in Zhejiang Province as an example, comprehensive three influencing factors: habitat quality, landscape pattern, and nighttime-light. To provide a relevant theoretical basis for the protection of CHPB, this paper quantitatively analyzes the changes of ecological environment and biodiversity in recent 30 years. The results showed that from 1990 to 2020, the biodiversity of CHPB in Zhejiang Province showed a positive change, the decline in Caverns of Heaven overall area slowed down, and the core area rebounded. The spatial distribution change of biodiversity is highly consistent with the land-use changes. The low value regions of biodiversity are mainly concentrated in the regions with intensive human activities, and the area decreases with the expansion of construction land. The core areas are primary areas with high biodiversity and overlap with nature reserves, natural parks, Scenic and Historic Interest Area, and other protected areas. In a word, CHPB still plays a vital role in ecological and environmental protection. In the future development, we should still pay attention to its biodiversity protection, and give full play to its role in ecological and environmental protection, and realize the contemporary application of CHPB's traditional ecological knowledge.

Keywords
Comprehensive biodiversity, habitat quality, landscape pattern, nighttime-light, traditional Protected Areas

Introduction

Biodiversity refers to all species and organisms on the earth or in a specific ecosystem, which can provide human beings with a large number of and multifaceted well-being (Berry et al. 2018), and is the infrastructure to support all life. However, with the increasingly serious global environmental changes, the loss of biodiversity gradually intensifies under the influence of multiple factors. The rate of species disappearance is 1000 times faster than any period in human history (UNEP 2020), which reduces the elasticity of the ecosystem and the supply of ecosystem service functions. It accelerates the spread of the virus (Lorentzen et al. 2020), global warming, and other hazards that significantly impact human livelihoods. In 2020, the world economic forum listed the loss of biodiversity as one of the five major social risks in the world. It is urgent to strengthen the research on biodiversity and its protection.

Protected Areas are currently the most effective biodiversity conservation measures globally (Geldmann et al. 2018; Hockings et al. 2019; MacKinnon et al. 2020). By September 2021, the total number of protected areas recorded in the World Database of Protected Areas (WDPA) has reached 266658, covering 245 countries and regions. The Protected Areas will carry out special protection and management for natural enrichment, good biodiversity conditions, and special significance. Chinese religions had similar protection ideas in ancient times. They would call the area with an excellent ecological environment, superior natural resources, and rich biodiversity as Caverns of Heaven and Places of Blessing (CHPB) and delimit a specific scope for protection. CHPB is the earliest Protected Areas in China (Lemche 2019), and Caverns of Heaven (CH) has an almost perfect ecological structure. Therefore, from the perspective of biodiversity, CHPB is a hot area with high biodiversity. In addition, CHPB integrates religious, social, cultural, and ecological meanings, significant in biodiversity protection. Strengthening the security of CHPB is China's significant contribution to biodiversity protection worldwide.

Since establishing the CHPB system in the Tang Dynasty, CHPB has been effectively protected under official-led protection measures. However, in recent years, with the excessive tourism development and the acceleration of urbanization, the land-use and the spatial density of human activities in CHPB and its surrounding areas have changed greatly. These two changes are the main driving factors for the reducing biodiversity (Gosselin and Callois 2018). The advantages and disadvantages of this change on the biodiversity of CHPB have not been discussed. Whether the favorable conditions of CHPB can play a specific protective role against these effects has not been quantitatively studied. Many scholars have analyzed species richness (Zhou 2019, 2020, 2021; Wang et al. 2020), community diversity (Ding et al. 2015; Wang et al. 2018) and landscape diversity (Zhang 2015) in the areas overlapping with CHPB in

spatial distribution. However, these studies are not aimed at CHPB, and there is still a lack of systematic analysis of CHPB biodiversity. Therefore, it is of great significance to understand the changes of CHPB in the temporal and spatial sequence of gradual development and intensified interference, the contribution of CHPB to the regional biodiversity protection, and the key factors causing the biodiversity changes of CHPB under the influence of regional urbanization and intensified human activities.

In recent years, with the rapid development of remote sensing technology, there have been more and more studies on landscape biodiversity assessment by building models or based on qualitative scoring systems. Compared with the traditional field measurement and statistical methods (Bai et al. 2020), this evaluation method has the characteristics of low cost and high speed (Compson et al. 2020). It is especially suitable for biodiversity analysis on a large regional scale.

Based on remote sensing data, people often analyze biodiversity changes on a large regional scale from the perspective of landscape biodiversity with habitat quality (Nelson et al. 2009; Berta et al. 2020; Hong et al. 2021), landscape pattern (Plexida et al. 2014; Rastandeh et al. 2018), and nighttime-light intensity (Li and Li 2015; Venter et al. 2016; Shi et al. 2018) as indicators.

Habitat quality is an important indicator of regional ecological security and can reflect the level of regional biodiversity (Bai et al. 2019). Habitat refers to the space that provides resources and conditions for species survival and breeding. Habitat quality refers to the ability of the ecological environment to provide suitable conditions for species survival and reproduction in a certain time and space (Hall et al. 1997), and affects the adaptability of organisms through the changes of resources and environmental conditions (Bernstein et al. 1991; Ah-King 2010). Habitat quality focuses on the overall situation of ecosystem state (Polasky et al. 2011; Czúcz et al. 2014), which can lay the basic level of regional biodiversity to a great extent and play a leading role in the biodiversity of some regions or species (Leira and Sabater 2005; Dures and Cumming 2010). In addition, studies have shown that habitat deterioration is the most prominent factor leading to the reduction of biodiversity (Wilcove et al. 1998; Myers et al. 2000; Horváth et al. 2019), while areas with high habitat quality can contain more organisms (Terrado et al. 2016). Therefore, habitat quality is an important embodiment of regional ecological environment and can be used as an alternative method for biodiversity analysis (Griffen and Drake 2008; Terrado et al. 2016; Sun et al. 2019; Li et al. 2021a). At present, this method has been widely used in biodiversity assessment in mountainous areas, wetlands, protected areas, cities and other regions (Gong et al. 2019; Huang et al. 2020b; Yu et al. 2020; Hong et al. 2021).

Landscape pattern also has a profound impact on biodiversity and its dynamics. Landscape pattern refers to the spatial arrangement and combination of landscape elements with different sizes and shapes, including the type, number, spatial distribution and configuration of landscape components (Turner et al. 2001). It is not only the concrete embodiment of landscape heterogeneity, but also the result of various ecological processes on different scales (Levin 1978; Forman and Godron 1981). Landscape pattern emphasizes the dynamic characteristics of landscape (Walz 2011; Uuemaa et al. 2013; Duarte et al. 2018), which affects biodiversity by affecting ecological processes,

such as the range of activities, migration law, population size and so on (Olff and Ritchie 2002; Correa Ayram et al. 2016; De Oliveira-Junior et al. 2020). Landscape pattern is also often used as an alternative indicator of species richness (Griffiths and Lee 2000; Dauber et al. 2003; Santini et al. 2017). Landscape ecology has developed a large number of landscape pattern indexes, such as landscape diversity index, evenness, landscape fragmentation and connectivity, which can realize the rapid evaluation of regional biodiversity (O'Neill et al. 1988; Sahani and Raghavaswamy 2018).

With the intensification of urbanization, the impact of human activities on biodiversity is expanding (Shochat et al. 2006). Climate change, environmental pollution and alien species invasion caused by human activities will seriously affect the change of local biodiversity (Bowler et al. 2020). Therefore, the intensity of human activities is often used to assess biodiversity changes. Nighttime-Light directly highlights the intensity of human activities (Elvidge et al. 1997; Zhao et al. 2019), which can directly reflect the process of urbanization and evaluate the ecological and environmental problems caused by urbanization (Li et al. 2016). In addition, nighttime-light will interfere with and change the living habits of organisms, especially nocturnal animals, and then affect biodiversity (Koen et al. 2018). At present, many studies have shown the negative effects of nighttime-light on different organisms (Longcore and Rich 2004; Hölker et al. 2010; Rodrigues et al. 2012; Gaston et al. 2013). Therefore, nighttime-light can be used as an indicator to reflect the impact of human activity intensity on biodiversity.

Previous studies mainly evaluated biodiversity changes from a single dimension of habitat quality, landscape pattern, and nighttime-light. However, the changes in biodiversity are not only affected by one factor and often affected by multiple factors simultaneously (De Chazal and Rounsevell 2009; Watson et al. 2014). For example, the change of land-use will change the habitat quality and landscape pattern at the same time: when an urban land is converted to forest land, it will improve the habitat quality of the region, but it may also lead to the fragmentation of the landscape pattern of the area, which is not conducive to most organisms (Hargis et al. 1999; Verga et al. 2017). To make a more scientific and objective quantitative evaluation of biodiversity changes in CHPB, Zhejiang Province, this study will integrate the above three dimensions for biodiversity analysis. Existing studies have shown that combining multi-dimensional indicators is feasible and necessary to evaluate biodiversity (Riedler and Lang 2018; Gong et al. 2019; Li et al. 2021a).

Zhejiang Province is the area with the most concentration of CHPB in China. In addition to the traditional mountain type, its landscape characteristics also include characteristics such as coastal and plain, including the main landscape types of CHPB in China. Therefore, the study of CHPB in Zhejiang Province has guiding significance for CHPB in China. The modern construction of CHPB is mainly reflected in the development of tourism activities, especially in the development of Scenic and Historic Interest Area (Han 2006). The Scenic and Historic Interest Area is essential for China's famous mountains and rivers to carry out ecotourism. Their system originated in 1982 and has developed rapidly since the 1990s. After 2005, the Scenic and Historic Interest Area application speed has decreased significantly (Zhu et al. 2021a). The large-scale tourism development activities in CHPB have dropped considerably, and the tourism development activities

pay more attention to ecological protection. In addition, since 2005, Zhejiang Province has practiced the economic and social green development model of "lucid waters and lush mountains are invaluable assets (Two Mountains)". Urban development also pays more attention to ecological and environmental protection, impacting biodiversity (Yunlong 2020). Therefore, 1990, 2005, and 2020 are three key time nodes that may be closely related to the change of the ecological environment of CHPB: 1990 was the period of rapid development and construction of CHPB in Zhejiang Province, 2005 was a significant turning point when the construction speed slowed down and paid more attention to ecology, and 2020 was the phased achievement under the new development model.

Therefore, combined with the actual situation of the natural conditions, human activity interference degree and the operability of the assessment, three key time nodes that may be closely related to the ecological environment change of CH blessed land in 1990, 2005, and 2020 are selected. The three indicators of habitat quality, landscape pattern index and night light intensity are integrated to reflect the temporal and spatial changes of biodiversity in CHPB, Zhejiang Province, in order to provide reference for the management and protection of CHPB.

Study area and methods

Study area

Zhejiang Province (118°01'–123°10'E, 27°02'–31°11'N), located in the south wing of the Yangtze River Delta along the southeast coast of China, in the transition zone between Eurasia and the Northwest Pacific, belongs to a typical subtropical monsoon climate zone. The monsoon is remarkable, the four seasons are distinct, the annual temperature is moderate, the sunshine is sufficient, the rainfall is abundant, the air is humid, the rain is hot in the same season, the climate resource allocation is diverse. There are many meteorological disasters. Zhejiang Province has a land area of about 101800 km^2. The terrain fluctuates wildly. The landscape tilts from southwest to northeast. There are mountains in the southwest and northwest, hills and basins in the middle and Southeast, and plains in the northeast. The forest area reaches 60591 km^2, and the forest coverage rate reaches 61%, ranking in the forefront of the country.

In Taoism, CHPB refers to a famous mountain resort where immortals live. It has an ideal natural environment and rich biodiversity. The thought of CHPB originated in the Jin Dynasty (265–420 A.D) and matured in the Tang Dynasty (618–907 A.D). There are 10 "Great Caverns of Heaven", 36 "Lesser Caverns of Heaven" and 72 "Places of Blessing". Caverns of Heaven and Places of Blessing (CHPB) refers to the general name of Caverns of Heaven (including Great Caverns of Heaven and Lesser Caverns of Heaven) and Places of Blessing. In Taoism, it is considered that the natural environment of Caverns of Heaven (CH) is better than Places of Blessing (PB). According to the *"Plan of Celestial and Terrestrial Palaces and Residences"* (hereinafter referred to as *"Plan"*) written by Sima Chengzhen of the Tang Dynasty. Zhejiang Province has 30

CHPB, accounting for 25.4% of the total, including 3 Great Caverns of Heaven, 9 Lesser Caverns of Heaven, 18 Places of Blessing. Extant can be verified for 3 Great Caverns of Heaven, 9 Lesser Caverns of Heaven, 14 Places of Blessing. The ancient Chinese determined the central position of CHPB according to the advantages and disadvantages of the ecological environment. When delimiting the protection scope, not only the areas with excellent ecological environment will be included in the protection, but also the surrounding human activity areas will be included in the protection management. *"Plan"* indicates the central position of the mountain where the main caves and palace buildings area in CHPB and also clearly records the overall protection scope of CH. The broad protection scope of CH is "30 Li (a unit of length was used in ancient China) of Zhouhui (i.e., the circumference)" to "10000 Li of Zhouhui", and "1 Li" is about "531 m" of the modern international metric system. However, the overall scope of PB has not been determined.

To sum up, the study on the diversity of CHPB in Zhejiang Province includes two aspects: 1) core area: the area contained in the central outer contour of the mountains where each CHPB is located, which are respectively recorded as CH core area and PB core area, collectively referred to as CHPB core area. 2) Overall area: offset the main outer contour line of the mountain where each CH is located outward to the perimeter equal to the " Zhouhui" length of the CH, which is the scope of the CH. Since PB does not specify "Zhou Hui" in the *"Plan"*, this study does not delimit the overall area of PB. Hence, the overall area study only refers to the overall area of CH, which is recorded as the CH overall area. Considering that the scope of the second Great Caverns of Heaven is too large, some areas exceed the scope of Zhejiang Province, and overlap with most of CHPB in Zhejiang, to simplify the research and data display, this study reduces its scope to "thousands of Li". The specific distribution and scope of CHPB are shown in Fig. 1.

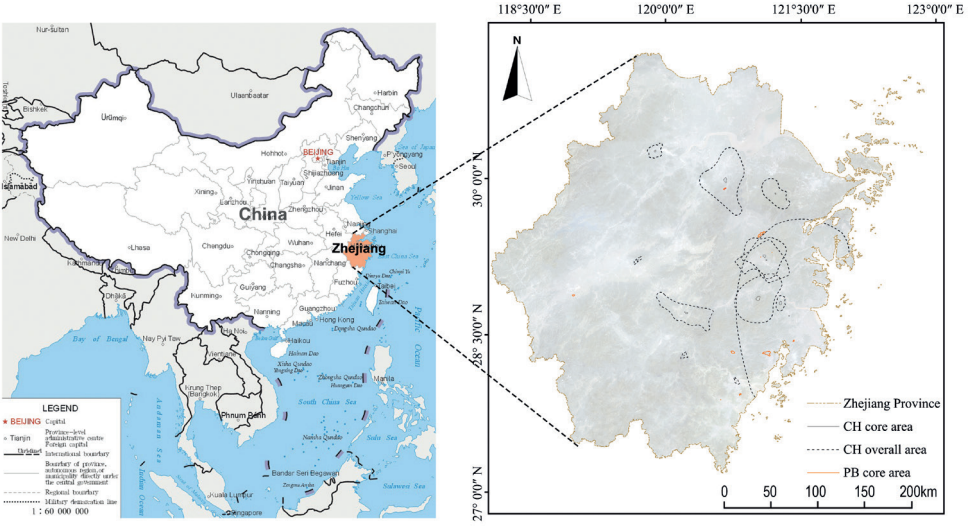

Figure 1. The overall area of Caverns of Heaven (CH) and the core area of Caverns of Heaven and Places of Blessing (CH, PB) in Zhejiang Province

Data sources and processing

The remote sensing data used for habitat quality assessment and landscape pattern index analysis were analyzed through Geospatial Data Cloud (www.gscloud.cn). The data in 1990 and 2005 are from Landsat5 TM, and the data in 2020 are from Landsat8 OLI_TIRS, the image resolution is 30 meters. The nighttime-light data is from the national Qinghai Tibet Plateau scientific data center (http://data.tpdc.ac.cn) (Zhang et al. 2021).

To include all the influencing factors as much as possible and eliminate the interference due to subjective reasons, this study first processes the data of Zhejiang Province. It then extracts the relevant data within the scope of the research object for analysis and discussion. The specific reasons are as follows: first, if only the content of CHPB is used to delimit the processing scope, the shape and size of the patch will be changed. As a result, the landscape pattern used for analysis differs from the actual landscape pattern. Secondly, the ecological environment and biodiversity of a region are often affected by internal and external environmental factors (Mcdonald et al. 2009); that is, the biodiversity of the study region may be affected by noise (Illner 1992; Mockford and Marshall 2009), industrial waste (Chang et al. 2019; Jia et al. 2021; Perlatti et al. 2021), and other factors at a certain distance from the region. If the scope of data analysis is consistent with the scope of the study region, the impact of surrounding towns on biodiversity in the study area is excluded.

Data analysis

Habitat quality

When using satellite remote sensing data to research on a large spatial scale, the Habitat Quality module of InVEST model is often used to evaluate the habitat quality (Huang et al. 2020a), which can better grasp the overall pattern and relatively truly reflect the threat of human activities to habitat and the relationship between ecosystem protection and human economic development. The InVest model considers that the habitat quality map is generated by analyzing land-use and land cover and its threat to biodiversity. Generally speaking, the higher the degree of naturalization, the higher the suitability and the smaller the threat (Sharp et al. 2020; Li et al. 2021b). As shown in Table 1, referring to the relevant studies (Lorenzo et al. 2017; Gong et al. 2019; Berta et al. 2020), combined with the actual situation of CHPB in Zhejiang Province, the parameters of threat factors, the habitat suitability of different habitat types and the sensitivity to threat factors are determined.

Landscape pattern index

FRAGSTATS is the most commonly used landscape pattern index calculation software, which is used to calculate various landscape indexes of classified map patterns and quantify landscape structure (McGarigal and Marks 1995; Zhang et al. 2020). Combined with the existing research (Cheng et al. 2020; Guo et al. 2021; Zhu et al.

Table 1. Parameters of threat factors, habitat suitability of different habitat types, and sensitivity to threat factors.

Land-use type code	Types of land-use	Relative habitat suitability	Threat factor†				
			Residential & industrial/mining land	Arable land	Railways and highways	National/provincial roads	County roads
1	Arable land	0.30	0.20	0.30	0.20	0.20	0.15
2	Forest & grasslands	1.00	0.70	0.50	0.45	0.40	0.30
3	Waters	1.00	0.80	0.70	0.60	0.60	0.50
4	Residential & industrial/mining land	0.00	0.00	0.00	0.00	0.00	0.00
5	Unused land	0.01	0.10	0.10	0.20	0.20	0.10
Maximum impact distance (km)			9	1	1.5	2	0.7
Weight			0.7	0.6	0.5	0.8	0.4
Relevance			exponential	exponential	linear	linear	linear

† the range of sensitivity of all threat factors is [0,1].

2021b) and the specific situation of CHPB in Zhejiang Province, the landscape pattern indexes significantly related to biodiversity were selected for landscape pattern analysis, including the largest patch index (LPI) (Su et al. 2015; Guo et al. 2021), the number of patches (NP) (Sahani and Raghavaswamy 2018; Cheng et al. 2020), patch density (PD) (Rüdisser et al. 2015; Liu et al. 2016), landscape shape index (LSI) (Liu et al. 2016; Guo et al. 2021), splitting index (SPLIT) (Sahani and Raghavaswamy 2018; Cheng et al. 2020; Guo et al. 2021), mean patch fractal dimension (FRAC_Mn) (Schindler et al. 2008; Guo et al. 2021), Shannon diversity index (SHDI) (Schindler et al. 2008; Su et al. 2015; Sahani and Raghavaswamy 2018), and Shannon evenness index (SHEI) (Sahani and Raghavaswamy 2018), and the landscape pattern indexes were weighted equivalently.

Nighttime-light intensity

Nighttime-light data are derived from satellite remote sensing data and based on the ArcGIS 10.2 platform for data processing. By using the Jenks classification method (North 2009; Chen et al. 2013), the nighttime-light data is divided into five levels according to the nighttime-light intensity, and the reverse value is assigned, which is a negative correlation.

Comprehensive biodiversity calculation method

Referring to the comprehensive biodiversity assessment model established by Gong (Gong et al. 2019) and Riedler (Riedler and Lang 2018). The habitat quality, landscape pattern index, and nighttime-light intensity were standardized, and the value range was [0,5]. The comprehensive biodiversity was obtained by superposition according to the weights of 0.45, 0.45, and 0.1. According to the results scored by experts, the weight value is obtained by using Analytic Hierarchy Process on SPSSAU online statistical analysis platform. The comprehensive biodiversity assessment model is as follows:

$$B_x = 0.58Q_x + 0.28P_x + 0.14L_x$$

In the formula, B_x represents the comprehensive biodiversity of CHPB in x year, Q_x is the habitat quality of CHPB in x year, P_x is the landscape pattern index of CHPB in x year, and L_x is the nighttime-light intensity of CHPB in x year.

Data resources

The remote sensing data come from Geospatial Data Cloud (www.gscloud.cn).

The nighttime-light data is from the national Qinghai Tibet Plateau scientific data center (http://data.tpdc.ac.cn).

Results

Analysis of habitat quality change

According to the calculation results of InVest model, the habitat quality change graph (Fig. 2) and habitat quality grade map (Fig. 3) were drawn. According to the figures: 1) the overall habitat quality of Zhejiang Province declined, and the decline rate in 2005–2020 was much higher than that in 1990–2005. In each stage, the habitat quality of CHPB core area was much higher than the average level of the whole province in the same period. The habitat quality in 2005–2020 of the CH overall area is lower than that in the whole province and higher than that in 1990–2005. 2) The habitat quality of the CH core area decreased by 0.039, but the extent was less than the average of the

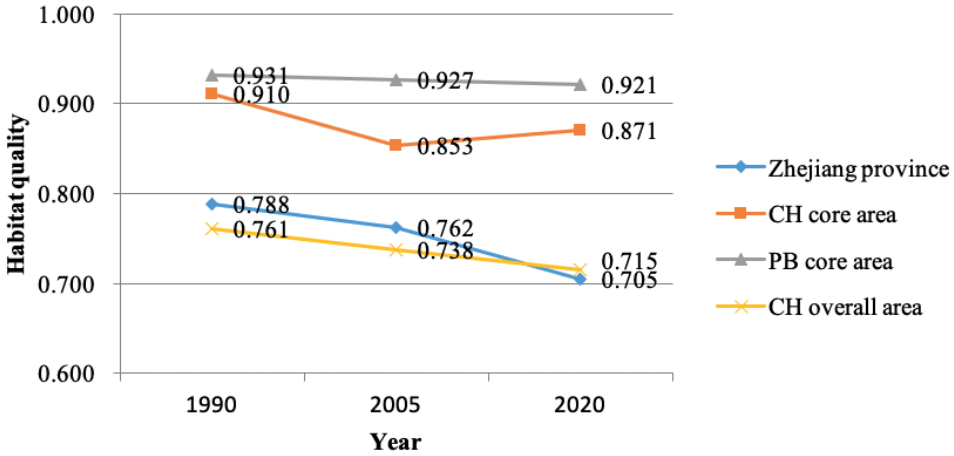

Figure 2. Habitat quality changes of Caverns of Heaven (CH) overall area & Caverns of Heaven and Places of Blessing (CH, PB) core area in Zhejiang Province from 1990 to 2020.

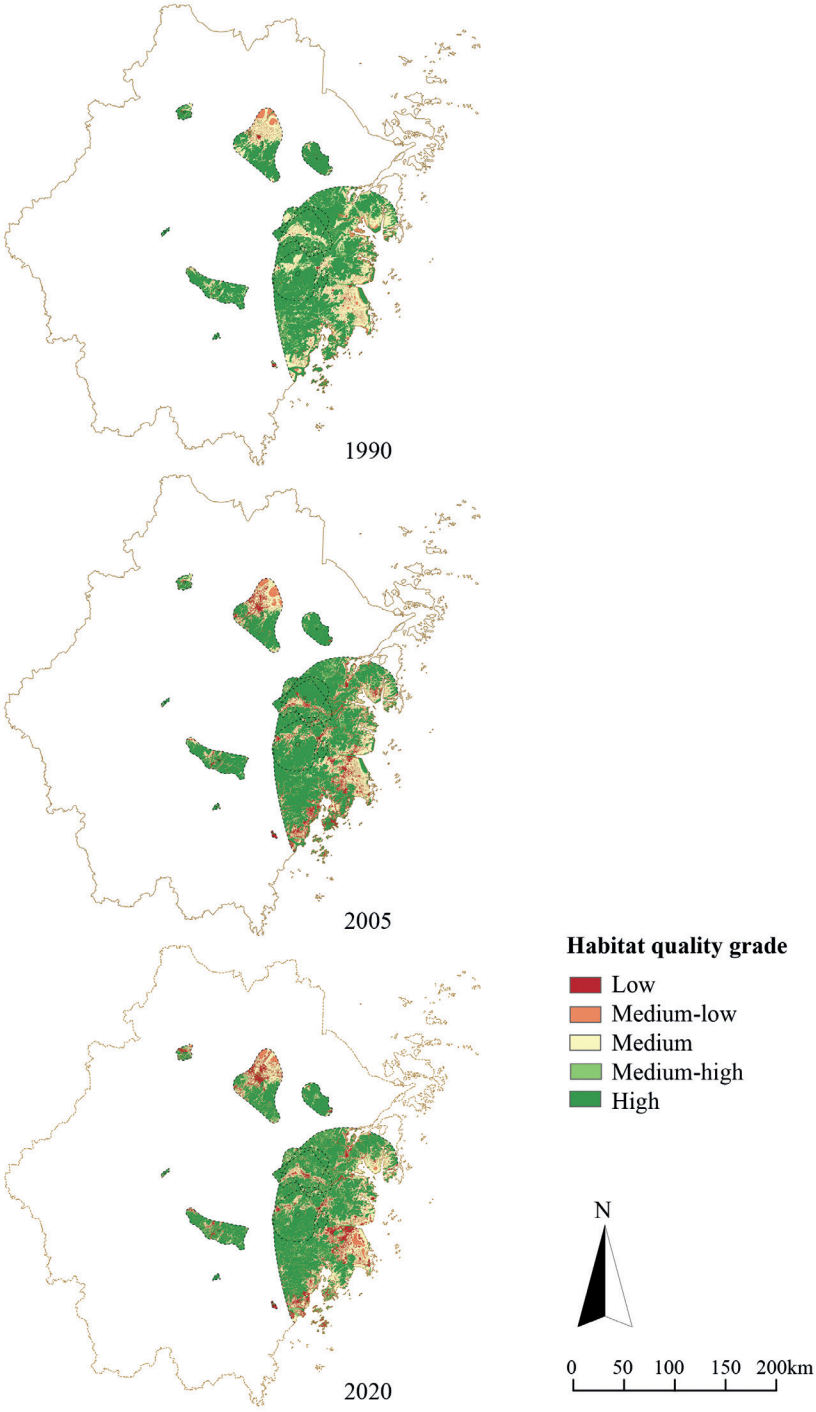

Figure 3. Habitat quality changes of Caverns of Heaven (CH) overall area in Zhejiang Province from 1990 to 2020.

whole province. The habitat quality of the CH core area decreased the fastest from 1990 to 2005, with a decrease of 0.057, which was higher than the average level of the whole province. Although it recovered in 2005–2020, the level was still lower than that in 1990. As to the PB core area, the habitat quality continuously went down, with a slight overall decline, but the decline in 2005–2020 was slightly higher than that in 1990–2005. 3) The overall habitat quality in the CH overall area showed a downward trend. In 1990, the habitat quality was high, and the low value areas were relatively few and concentrated. After that, the low value areas expanded greatly, and showed a trend of dispersion and fragmentation, indicating that the degree of habitat degradation was increasing.

Analysis of landscape pattern index

The results of landscape pattern index calculation showed that: 1) the indexes of CH overall area and CHPB core area have, except for LPI, all values increased, and the scope of landscape pattern index changed gradually expanded (Fig. 4). 2) The overall comprehensive landscape pattern index showed an upward trend. CH core area increased rapidly from 1990 to 2005, with an increase of 0.268, then the growth rate remained unchanged, and the comprehensive landscape pattern index was lower than the average level of the whole province in the second half. The change of PB core area first decreased and then increased, but it was always lower than the average level of the whole province. The growth rate of CH overall area was low in 1990–2005, but it ac-

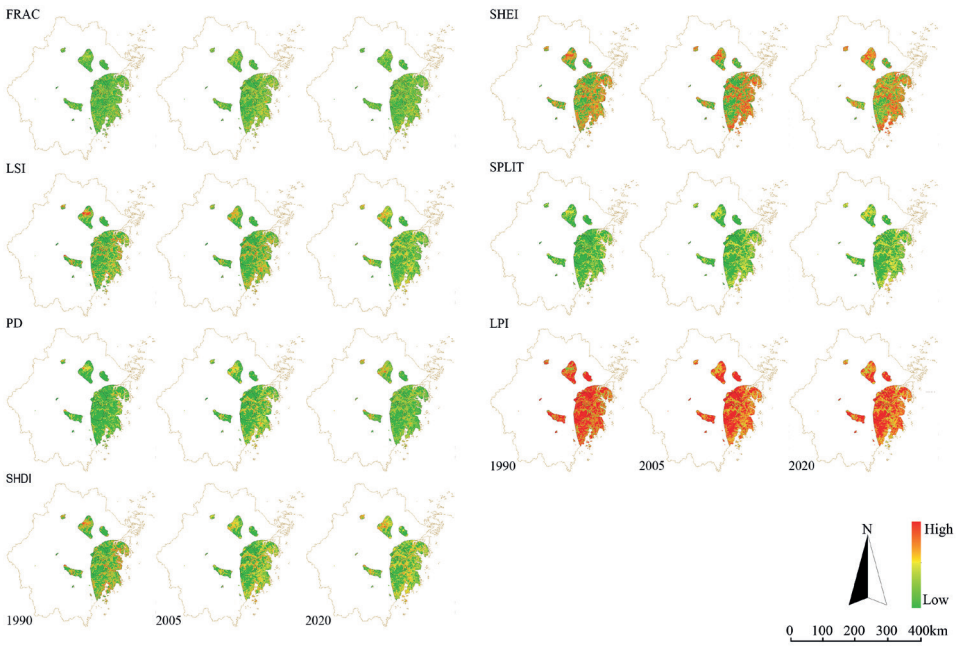

Figure 4. Landscape pattern indices' changes of Caverns of Heaven (CH) overall area in Zhejiang Province from 1990 to 2020.

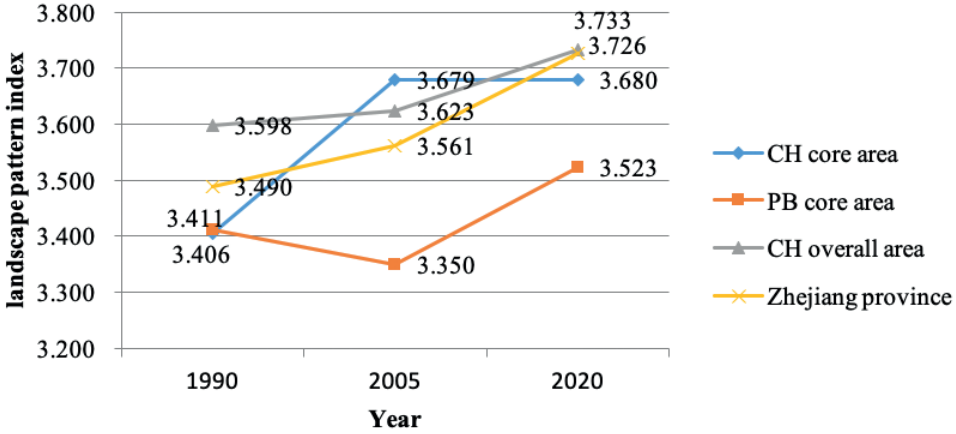

Figure 5. Comprehensive landscape pattern index changes of Caverns of Heaven (CH) overall area & Caverns of Heaven and Places of Blessing (CH, PB) core area in Zhejiang Province from 1990 to 2020.

celerated and increased in 2005–2020, and it would be slightly higher than the average level of the whole province in 2020 (Fig. 5). 3) The comprehensive landscape pattern index of CH overall area is high. The areas with a high comprehensive landscape pattern index have increased spatial distribution, but the overall change is small. (Fig. 6).

Analysis of nighttime-light intensity

The overall increase of nighttime-light intensity in Zhejiang province is large. 1) In 1990, the nighttime-light intensity of CH overall area and CH core area was almost the same level as the average of the whole province. After that, it was higher than the average level of the whole province. Besides, the growth trend of the CH core area was the most prominent, with both intensity level and the growth rate higher than other areas obviously. 2) The nighttime-light intensity of the PB core area was lower than that of other areas at all stages, and the growth rate in 2005–2020 slowed down, 368.553 less than that in 1990–2005. (Fig. 7). 3) During the whole period, the area without nighttime-light in the CH overall area shrunk considerably, and the area with high and low nighttime-light intensity expanded in a large area (Fig. 8)

Comprehensive biodiversity analysis

The habitat quality, landscape pattern index and nighttime-light intensity were standardized, and the comprehensive biodiversity evaluation results were obtained by weighted superposition (Table 2). The results showed that: 1) the comprehensive biodiversity of the whole province obvious linear downward trend. 2) The comprehensive biodiversity of CHPB core area in each period was much higher than the average value of the whole province, showed a trend of first decreasing and then increasing, especially in the PB core area; The comprehensive biodiversity in the CH overall area decreased

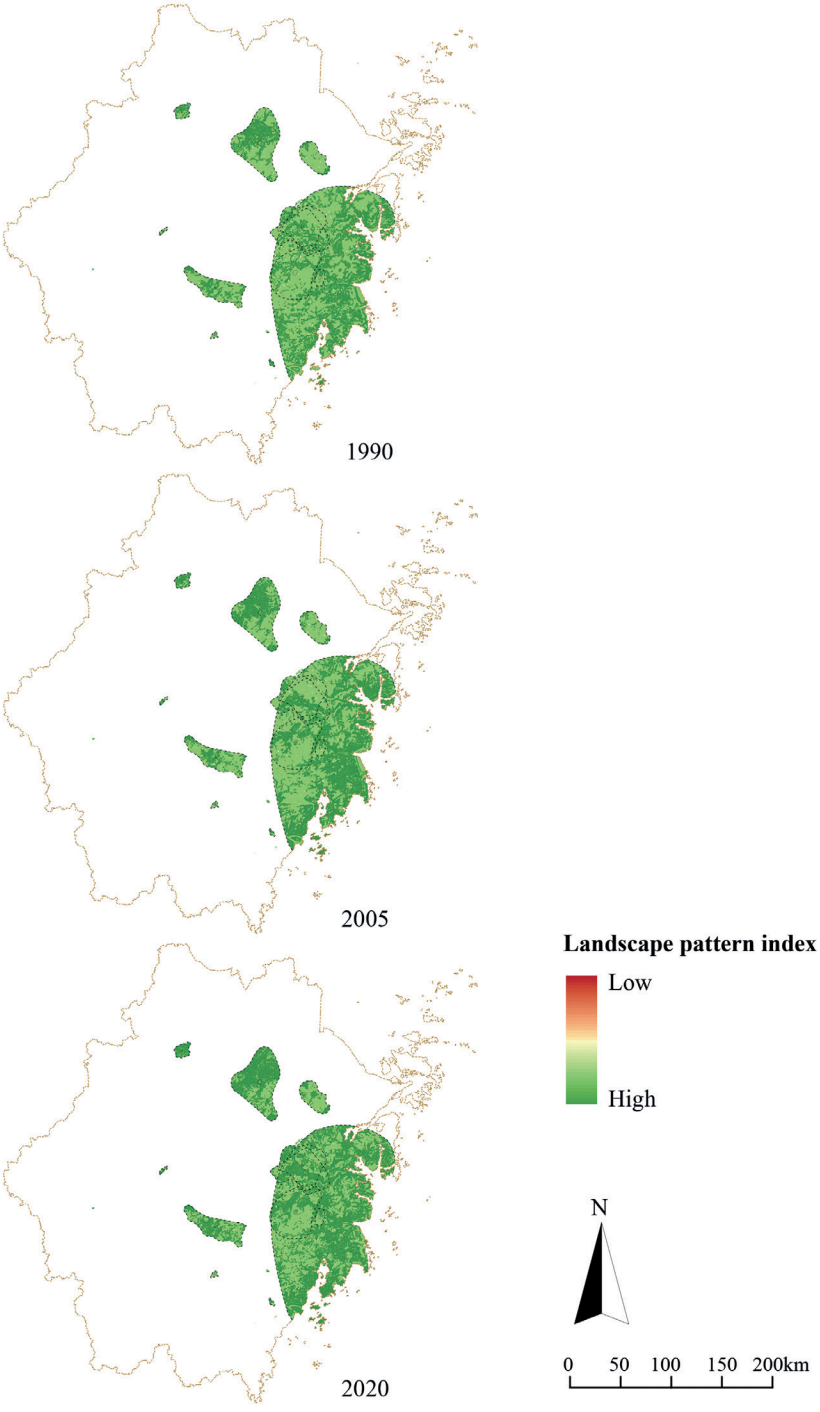

Figure 6. Comprehensive landscape pattern index changes of Caverns of Heaven (CH) overall area in Zhejiang Province from 1990 to 2020.

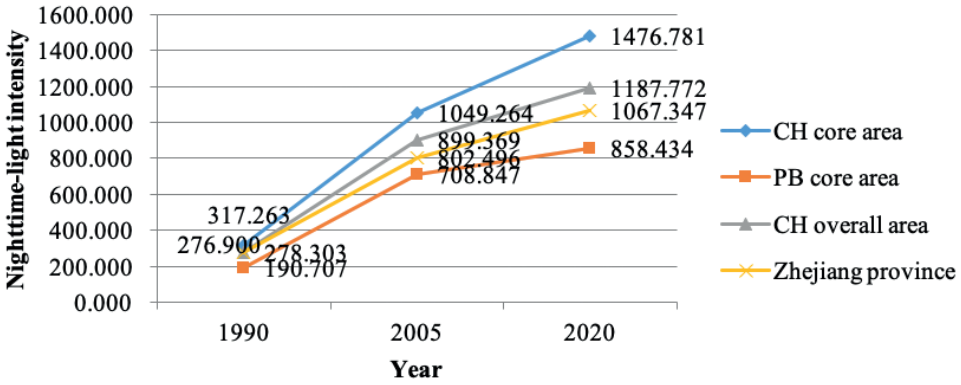

Figure 7. Nighttime-light intensity changes of Caverns of Heaven (CH) overall area & Caverns of Heaven and Places of Blessing (CH, PB) core area in Zhejiang Province from 1990 to 2020.

generally, but the decline speed slowed down in the second half, made it changed from lower to a slightly higher level compared with the average level of the whole province. 3) The standard deviation of comprehensive biodiversity evaluation of CH overall area and CHPB core area gradually increased, and was higher than the average value of the whole province; especially in 2020, the figure had reached 1.170, which indicated that the comprehensive biodiversity difference in CH overall area was evident. 4) The spatial distribution of comprehensive biodiversity was highly consistent with land-use change and the spatial density of human activities. The low value comprehensive biodiversity areas were mainly concentrated in the residential and industrial/mining land areas with intensive human activities, and are positively related to expanding of such land-use (Figs 9, 10).

Habitat quality, landscape pattern, and nighttime-light intensity had different influences on regional biodiversity were distinct in different periods. The CH overall area was mainly affected by the habitat quality and nighttime-light intensity in 1990–2005, and mainly by the habitat quality and landscape pattern in 2005–2020; From 1990 to 2020, the effects of the three factors were evident in the CH core area, and the impact of habitat quality is dominant; the main influencing factors of the PB core area were nighttime-light and landscape pattern in 1990–2005, and was landscape pattern in 2005–2020 (Fig. 11).

Table 2. Average and standard deviation of comprehensive biodiversity in Caverns of Heaven (CH) overall area & Caverns of Heaven and Places of Blessing (CH, PB) core areas in Zhejiang Province from 1990 to 2020.

	PB core area		CH core area		CH overall area		Zhejiang Province	
	Average	Standard deviation	Average	Standard deviation	Average	Standard deviation	Average	Standard deviation
1990	4.309	0.615	4.216	0.710	3.875	1.010	3.922	0.983
2005	4.187	0.667	4.014	0.977	3.728	1.125	3.797	1.098
2020	4.277	0.669	4.049	0.919	3.661	1.170	3.649	1.202

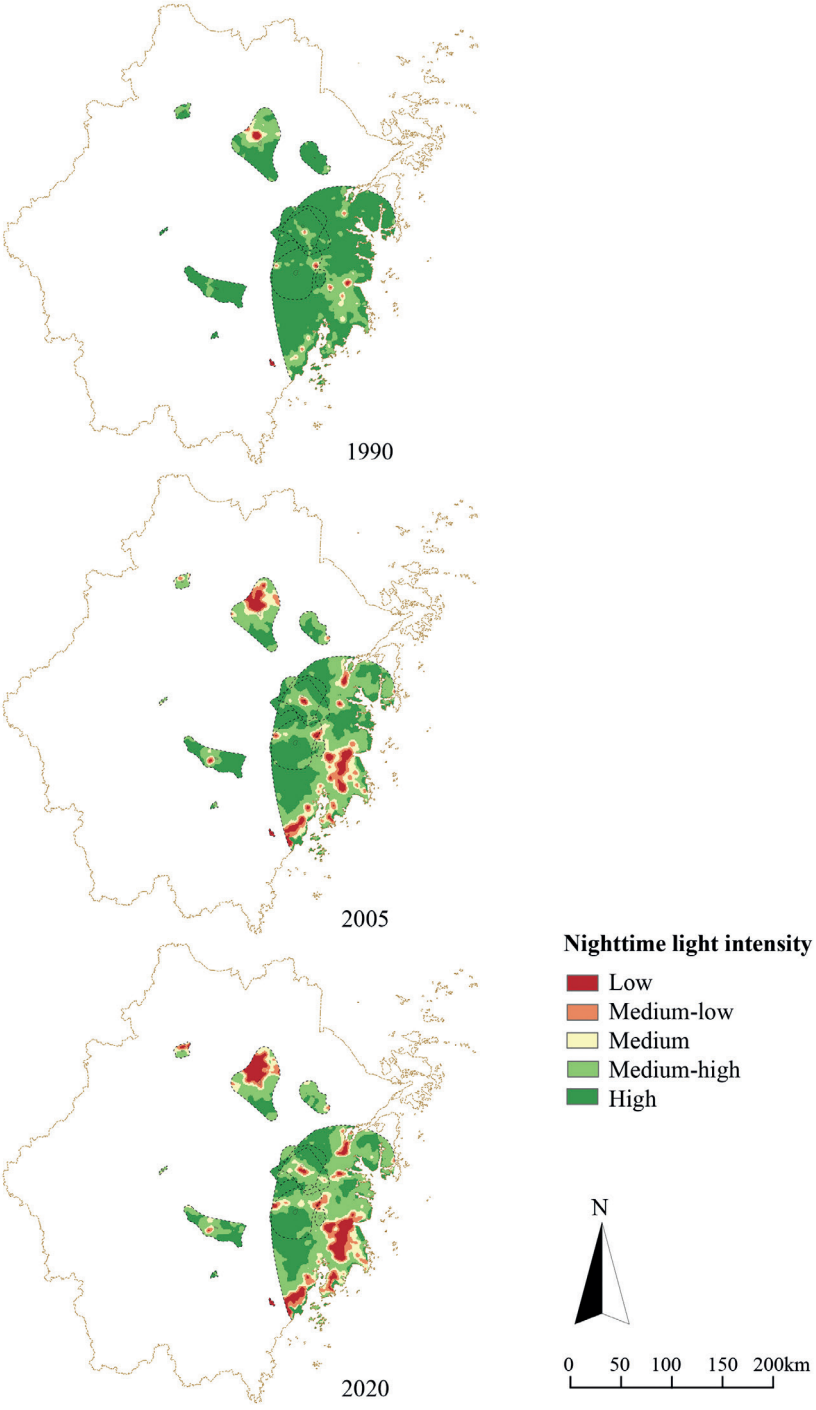

Figure 8. Nighttime-light intensity changes of Caverns of Heaven (CH) overall area in Zhejiang Province from 1990 to 2020.

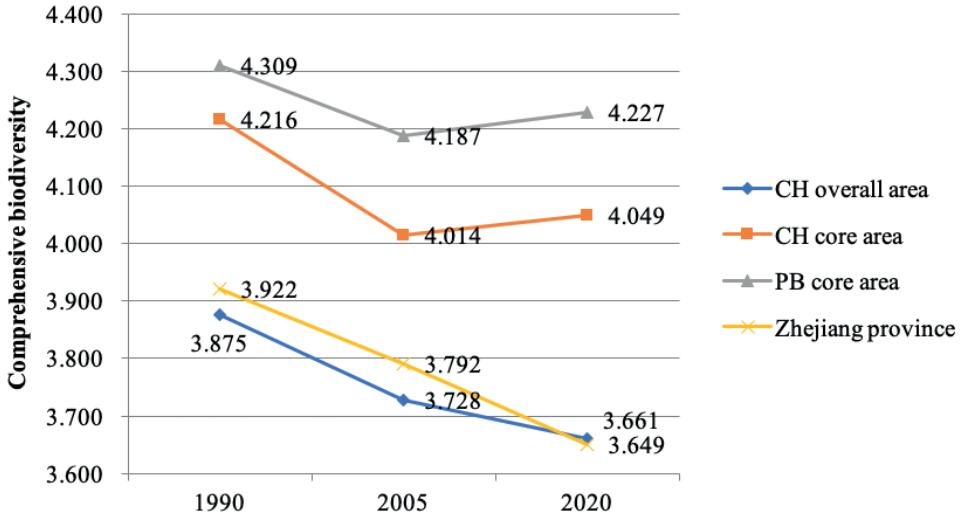

Figure 9. Comprehensive biodiversity changes of Caverns of Heaven (CH) overall area & Caverns of Heaven and Places of Blessing (CH, PB) core area in Zhejiang Province from 1990 to 2020.

Discussion and conclusion

Compared with the biodiversity changes in the whole province, CHPB has played a positive role in biodiversity protection. Under the influence of habitat quality, landscape pattern and nighttime-light, the temporal and spatial differentiation is evident: from 1990 to 2020, the trend of biodiversity change in CHPB in Zhejiang Province showed positive changes,in which the decline rate of CH overall area slowed down, and the CHPB core area rebounded. The spatial distribution change of comprehensive biodiversity is highly consistent with the land-use change. The low value areas of comprehensive biodiversity are mainly concentrated in the areas with intensive human activities, which continue to decrease with construction land expansion. The core areas are primary areas with high comprehensive biodiversity, which are highly overlapped with natural parks, Scenic and Historic Interest Area, and other protected areas.

Although there is no research on the biodiversity of CHPB at present, the research on nature reserves has found that the spatial change of biodiversity at the landscape level is significantly related to land-use changes, and the downward trend has slowed down under effective protection measures, and the biodiversity in the core area is higher than that in other areas. The results of this study are similar to those of other countries (Ren et al. 2015; Gong et al. 2019; Katoh and Matsuba 2021; Yang 2021). Some studies also show that during the period from 1990 to 2005, the forest coverage in Zhejiang Province of China decreased, and the forest fragmentation accelerated (Li et al. 2011), and the construction land increased rapidly (Liu et al. 2008; Ruishan and Suocheng 2013). Especially in small cities in Zhejiang Province, the expansion rate increased of construction land was the fastest before 2000 (Li et al. 2014). From 2005

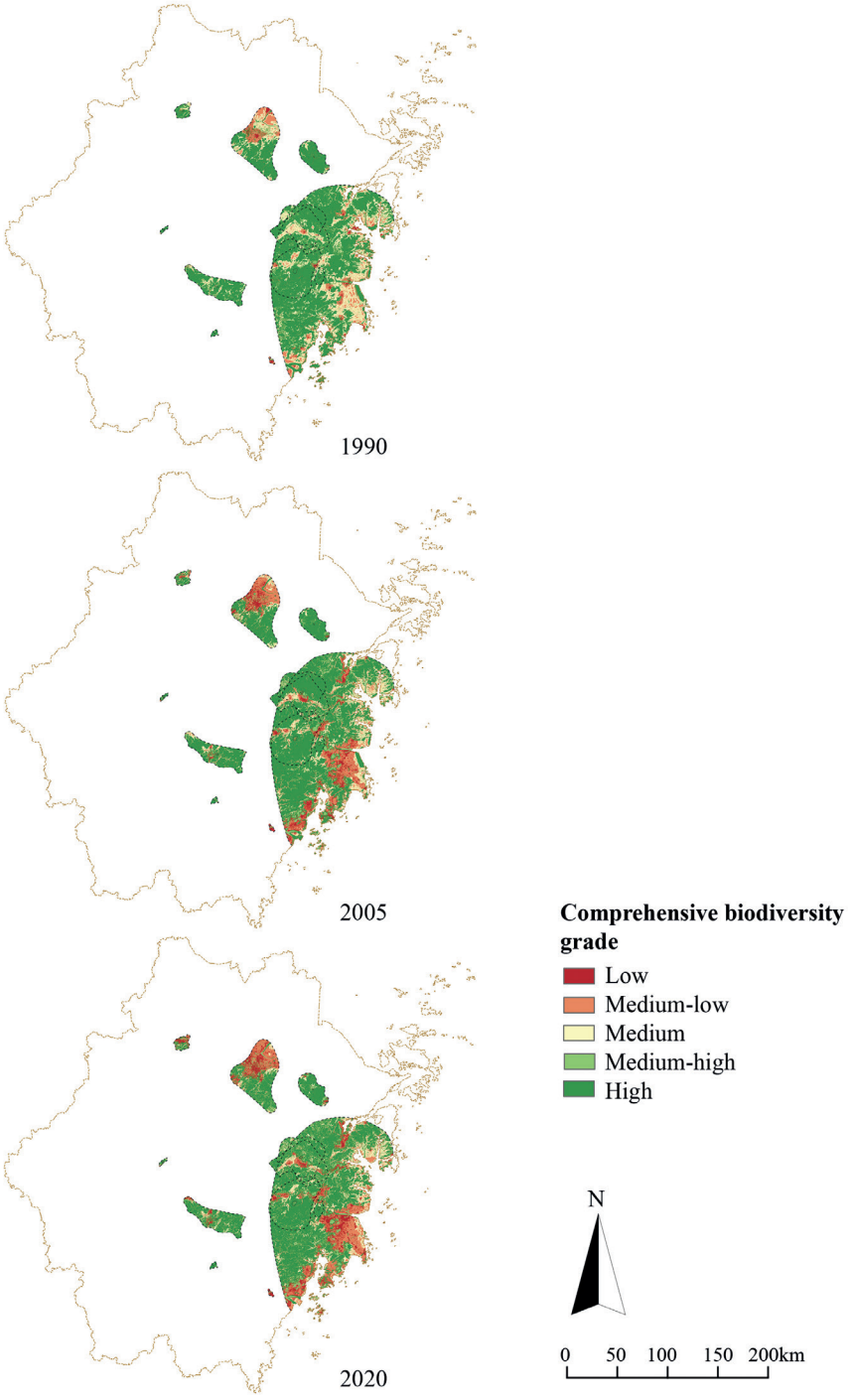

Figure 10. Biodiversity changes of Caverns of Heaven (CH) overall area in Zhejiang Province from 1990 to 2020.

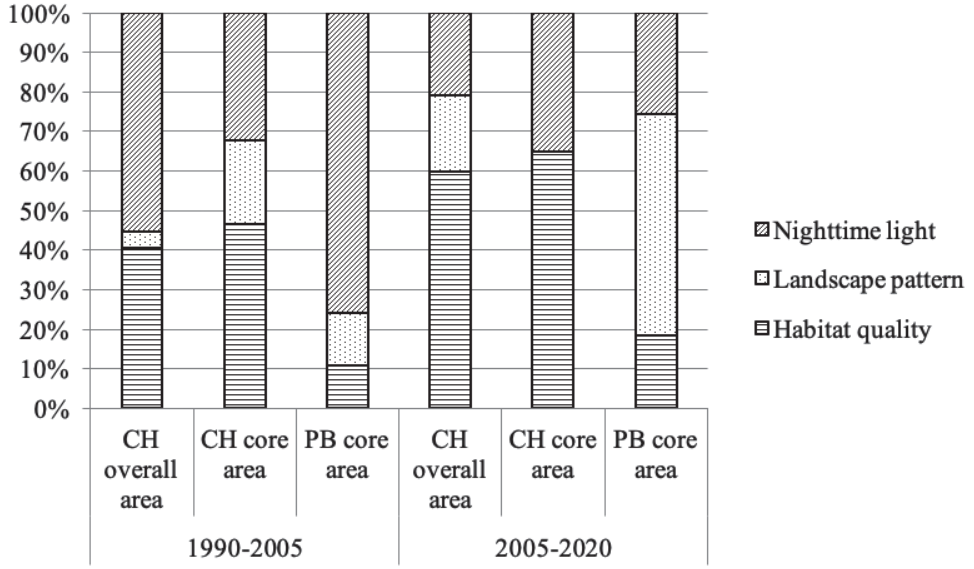

Figure 11. Effects of different factors in different periods on biodiversity in different areas.

to 2020, according to the announcement of the Zhejiang Provincial Bureau of Statistics and the Zhejiang Provincial Department of Ecological Environment, the area of construction land in Zhejiang Province is still increasing. However, the decline rate of forest land area is slowing down, the forest coverage rate still has a small increase, the forest land has high habitat quality, and the general biodiversity will be higher (Sharp et al. 2020; Li et al. 2021b). In addition, the research results of some areas overlapping with CHPB also show that the land-use intensity increased (Cao et al. 2018) and the diversity of some biological species decreased (Yang et al. 2005) from 1990 to 2005, and the net primary productivity of plants increased from 1990 to 2020 (Chen et al. 2017), and the ecosystem pattern in some regions has improved (Zhang 2015). The results of this paper are similar to these.

Combined with the actual situation of CHPB and the differences of habitat quality, landscape pattern and nighttime-light on biodiversity changes in different periods, it is speculated that the reasons for these changes may be as follows: 1) most of the core areas of CHPB are far away from the urban center and close to mountains and forests, mainly relying on natural landscapes such as mountains and lakes, with rich biodiversity, high habitat suitability, and ecological environment. The system is relatively stable and has a stronger recovery ability after being damaged. As for CH overall area, in addition to forest land, grassland, lakes and other areas, it also covers a large number of cultivated land, construction land, and other areas, which is more related to human activities. Biodiversity change is greatly affected by economic and social development concepts (Liu et al. 2008; Yunlong 2020). Rapid economic development, urbanization, industrialization and population growth directly lead to land-use change (Randall and Mulla 2001; Han et al. 2016), affecting habitat quality, landscape pattern, nighttime-light intensity, and then

biodiversity. 2005 is the key turning point of the concept of economic and social development in Zhejiang Province. Before that, the development concept based on economic construction has led to the acceleration of urbanization, the rapid expansion of construction land (Kamal-Chaoui et al. 2009), and the weak awareness of ecological protection. Therefore, the declining trend of biodiversity is obvious. In 2005, China's President Xi Jinping put forward the theory that "lucid waters and lush mountains are invaluable assets (Two Mountains)" for the first time in Zhejiang province. Emphasizing the importance of ecological and environmental protection, the concept of green development has gradually taken root in the hearts of the people. With the implementation of the "Five Water Treatment", "The Renovation of Old Residential Areas,Old Industrial Plant Reformation and the Renovation of Urban-village, and Demolish the Illegal Building" and "Demonstration of Thousands of Villages, Renovation of Thousands of Villages" project (The project won the highest environmental protection honor of the United Nations – "Earth Guardian" Award in 2018), etc, the forest coverage rate, air quality and section water quality in Zhejiang Province have increased significantly. Therefore, from 2005 to 2020, the ecological environment of Zhejiang Province will steadily improve, which is conducive to the growth of biodiversity in CHPB. 2) The biodiversity change in the core area of CHPB may be affected by the development process of Scenic and Historic Interest Area. The temporal variation characteristics of biodiversity change in CH core area, which is highly overlapped with Scenic and Historic Interest Area in spatial distribution, are consistent with the characteristics of the development process of Scenic and Historic Interest Area. The large-scale construction of the CH core area in Zhejiang Province is mainly concentrated in the 1990s (Mao et al. 2002; Han 2006), while the construction of the PB core area is relatively late, mostly in the 21st century. In 2006, "*the Regulations on Scenic and Historic Interest Area"* were issued, which has become the highest legal form for the management of Scenic and Historic Interest Area. Before that, although it was emphasized to protect the natural resources of Scenic and Historic Interest Area, it was not implemented in practice. As the main carrier of ecotourism activities, Scenic and Historic Interest Area also lead to the change of management rights of some Scenic and Historic Interest Area into tourism enterprises (Song and Yan 2020). Therefore, from 1990 to 2005, the tourism of CHPB, which is associated with Scenic and Historic Interest Area, developed vigorously and attracted many citizens. Coupled with the construction of supporting tours and service facilities, the habitat quality was inevitably affected, thus affecting biodiversity. It was not until "*the Regulations on Scenic and Historic Interest Area"* issued in 2006 that the development of the system of Scenic and Historic Interest Area was relatively mature: paying more attention to the protection of the background of scenic resources, clarifying the management of Scenic and Historic Interest Area, and implementing the planning, protection, supervision, and management. Because of the lag of ecological protection measures, they need to accumulate for a certain period of time to be effective (Moglen and Palmer 2014; Watts et al. 2020). The ecological protection measures carried out before 2005 may not see the ecological effect until after 2005. Therefore, the comprehensive biodiversity of the core area of CHPB showed an upward trend from 2005 to 2020. From the time difference of the impact of landscape pattern on biodiversity in the core area, it can also be seen that tourism development and other

behaviors will greatly impact on biodiversity in the core area. Therefore, the development process of Scenic and Historic Interest Area may be an important reason for the change of biodiversity in the core area of CHPB. 3) CH is a Taoist holy mountain, and Taoist traditional ecological protection thought runs through its development process. Taoism advocates nature, pays attention to environmental protection (Ji et al. 2017); traditional ecological protection ideas are contained in the teachings of Taoism. Taoism has also actively advocated ecological protection in recent years and put forward the concept of " Taoist ecological concept". Since 2006, it has successively put forward or issued *"Qinling Mountain Manifesto"*, *"Maoshan Mountain Manifesto of ecological Taoist concept and Outline of the Eight Years (2010–2017) Plan of Chinese Taoism for Environmental Protection"*, and other ecological protection measures have played a vital role in the protection of biodiversity in CHPB, especially in the core area.

Therefore, in the follow-up development planning, we should fully consider the ecological regulation capacity of CHPB, avoid over development and construction, and exceed our ecological balance capacity. We should standardize the tourism development behavior, especially the construe of supporting tourism facilities such as homestay (B&B), rural tourism around the core area, and deal with the relationship between natural ecology, social economy and community residents at the boundary of CHPB. Paying attention to protecting human activity areas is the best way to preserve biodiversity and effectively reduce the pressure on protected areas (Hilborn and Sinclair 2021). Therefore, while covering areas with high biodiversity, it is also necessary to protect sites with rapid biodiversity reduction to avoid their free spread. Protect biodiversity with traditional ecological knowledge (Gavin et al. 2015; Niesenbaum 2019), give full play to the ability of CHPB in biodiversity protection, and actively explore the strategies provided by Taoist traditional ecological knowledge represented by CHPB in biodiversity protection and revitalizing local resource utilization.

In a word, CHPB, which has a history of nearly 2000 years, is the prototype of the protected areas and still has important historical, cultural and ecological value. From 1990 to 2020, based on the site conditions of CHPB in Zhejiang Province, the comprehensive biodiversity reflected by CHPB in Zhejiang Province showed positive changes in habitat quality, landscape pattern and nighttime-light intensity under the joint action of the concept of economic and social development, the construction of Scenic and Historic Interest Area and Taoist ecological protection measures. It plays a vital role in ecological and environmental protection. Understanding the temporal and spatial changes of CHPB biodiversity is of great significance to CHPB protection. In the future development, we should still pay attention to its biodiversity protection. Play its role in ecological and environmental protection and realize the contemporary application of traditional ecological knowledge in CHPB.

Acknowledgements

Mr. Hu Wenhao of Zhejiang agriculture and Forestry University put forward many valuable opinions in the process of revising the paper. I would like to express my sincere thanks.

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Nature Conservation 48: 31–56 (2022)
doi: 10.3897/natureconservation.48.79936
https://natureconservation.pensoft.net

RESEARCH ARTICLE

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Environmental citizen science in Greece: perceptions and attitudes of key actors

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Academic editor: J. Tzanopoulos | Received 29 December 2021 | Accepted 10 March 2022 | Published 7 April 2022

http://zoobank.org/1C1161C5-FBFF-4C80-A8C6-FDB74C46951A

Citation: Galanos C, Vogiatzakis IN (2022) Environmental citizen science in Greece: perceptions and attitudes of key actors. Nature Conservation 48: 31–56. https://doi.org/10.3897/natureconservation.48.79936

Abstract

Citizen Science (CS), the voluntary participation of lay people in scientific work, is well-established in the fields of nature conservation and biodiversity monitoring due to its potential to create large environmental datasets. This study aims to understand the familiarity, perceptions and attitudes towards CS of the key environmental actors in Greece. The target group consisted of employees and/or representatives of Environmental Non-Governmental Organisations (ENGOs), scientists and civil servants related to nature conservation. Quantitative data were collected using an electronic questionnaire, 178 fully completed questionnaires and subsequently eight semi-structured interviews with experts were conducted. Descriptive statistics were used to measure the familiarity and attitude of the actors, as well as the obstacles to the development of CS in Greece. We used Cronbach's test to measure the reliability of the used Likert scale and Kruskal-Wallis non-parametric test to identify significant differences amongst the three groups of actors. Qualitative data were analysed following a Thematic Analysis methodology. The results show that ca. 40% of the key actors are familiar with the terms and CS practice while over 65% with the concept. The general attitude of the actors towards CS is positive although concerns about data quality collected were highlighted. "Lack of cooperation culture", "Ignorance of the existence of the phenomenon" and "Lack of know-how" emerged as the most important obstacles to CS development in Greece. Although CS is present in Greece, it is not visible enough. The main reasons are that relevant projects employ different terms, are under-represented in the formal literature and include limited, if at all, project dissemination. There are significant differences regarding familiarity and the attitude towards CS between actors, but also similarities concerning the main obstacles. The study sets a baseline which can be employed to improve and further expand Environmental Citizen Science (ECS) in Greece.

Keywords
Academia, government, mixed methods, nature conservation, NGOs, survey

Introduction

Public participation and knowledge production can take many forms (Shanley et al. 2019) with Citizen Science being, perhaps, the most widely used and better understood by the public. In fact, the practice of Citizen Science (CS) is not new, since for most of the recorded history, it was the public that advanced science - often by just observing nature (Miller-Rushing et al. 2012). The history of amateurs conducting research and collecting environmental data from the field is very old and expands from Norway to France, Japan to China and US (Miller-Rushing et al. 2012). Undoubtedly, the current concept of CS, that is, the involvement of the public in scientific work, has evolved primarily over the past two decades (Cohn 2008). This is obvious by a number of indicators, such as the steep rise in the biodiversity-orientated CS projects over the last 30 years (Theobald et al. 2015), the expansion of scientific reports and peer-reviewed articles resulting from CS project data (Bonney et al. 2014; Vann-Sander et al. 2016) and the establishment of professional CS organisations around the world, especially in the US, Europe and Australia (Ellwood et al. 2017). Moreover, CS projects cover a breadth of scientific disciplines from biology and biodiversity data collection, to the interpretation of astronomic images, archaeology and chemistry (Dickinson et al. 2010; Conrad and Hilchey 2011; Franzoni and Sauermann 2014; Kullenberg and Kasperowski 2016).

There are certain factors which enabled the proliferation of CS during the last decades. One of the most important drivers is the technological breakthrough and all the new possibilities for data gathering and dissemination of information from the public (Silvertown 2009; Dickinson et al. 2012). These technological innovations include the internet, smartphones, Global Positioning Systems (GPS), web geographic information system applications, the increased availability of domestic internet connections and the reduction of costs for computer storage. On one hand, the increasing realisation amongst professional scientists that the public can provide free labour, skills, computing power and even funding and, on the other hand, the growing demands from large research funders for public engagement led to new and innovative CS projects (Cohn 2008; Silvertown 2009). In addition to technological factors, social factors also played an important role in the expansion of CS. Two of those prevail. The first one is the growth in the population of well-educated individuals who possess the skills to perform scientific tasks, but do not use their knowledge in their daily life. The second factor is the increase in leisure activities as a result of the reduction in working hours in advanced economies (Haklay 2013).

CS is best established in biological sciences and, in particular, in biodiversity and natural resources monitoring (Follett and Strezov 2015; Kullenberg and Kasperowski 2016; Schade and Tsinaraki 2016; Pettibone et al. 2017; Pocock et al. 2017). Nowadays, probably the principal reason for this dominance of biology-related CS projects

is the realisation of scientists that CS is perhaps the only practical way to gather data at large geographical scales, time periods and private lands. Such large datasets are necessary for biologists to understand the processes and to address ecological questions, such as climate change, patterns of migration and spread of diseases (Dickinson et al. 2010).

In recent years, a number of studies reviewed the field of Environmental Citizen Science (ECS), to determine the effective use of the data collected in biodiversity research (Theobald et al. 2015; Chandler et al. 2017), to assess the diversity and evolution of the ECS field (Pocock et al. 2017), to understand the diversity of CS in specific countries (Pettibone et al. 2017) or to assess projects that can be of relevance to environmental European policy (Bio Innovation Service 2018).

Studies about CS as a social phenomenon have dealt mainly with how the citizens react, participate and understand it, with emphasis mainly on motivations (Alender 2016; Domroese and Johnson 2017) or barriers to participation (Martin et al. 2016). For example, information from interviews with representatives of biodiversity-recording organisations shed light on motivations of citizens to participate in biodiversity-monitoring CS projects and demonstrated that people from socio-economically deprived areas are under-represented in those schemes (Hobbs and White 2012). During recent years, emphasis is given in motivation to participate in online CS projects (Raddick et al. 2010; Nov et al. 2014), the use of smartphones (Land-Zandstra et al. 2016b) and gamification strategies (Greenhill et al. 2016) to enhance citizens' participation. What remains limited though, are insights about the main initiators/groups of actors of ECS projects (ENGOs, scientists, governmental organisations) and interaction between those engaged (Rotman et al. 2012; Weng 2015). Motivation and data collection are perceived differently by citizens and groups of actors (Jiang et al. 2018) with scientists more concerned about the quality of data and peers' approval (Riesch and Potter 2014; Burgess et al. 2017) rather than the opinion and needs of citizens when engaging in ECS projects.

In Greece, most of the examples of ECS projects are usually initiated by ENGOs, such as the Hellenic Ornithological Society (HOS) or the Hellenic Marine Environment Protection Association (HELMEPA). The country may benefit from such projects given that: a) its biodiversity is considered to be one of the richest in endemism in Europe and in the Mediterranean (Georghiou and Delipetrou 2010) and that: b) due to the recent economic crisis, the adoption of an austerity programme (Matsaganis 2014) has weakened environmental administration and surveillance mechanisms, adding extra threats to Greek biodiversity (Lekakis and Kousis 2013). Therefore, CS has a distinct role to play as a mechanism to support nature conservation as well as public engagement with nature and science (Devictor et al. 2010). Nevertheless, the specific cultural and socio-economic characteristics of each country play a decisive role in the success or failure of such projects. These might be expressed through past environmental history, collaborative culture, legal frameworks and organisational context. Recommendations exist in literature, but most studies come from relatively rich, industrialised countries and little knowledge exists for the specific challenges of initiating monitoring schemes in different contexts (Danielsen et al. 2003) with few perhaps exceptions (e.g. Loos et al. 2015). Although users' profiles are important factors in shaping citizen

science (Amarasinghe et al. 2021; Aristeidou et al. 2021), this study focuses on three specific sectors (Public bodies, Research/Academic Community, ENGOs) since they are the ones which deal with conservation on a day-to-day basis, can formulate policies and/or influence decision-making.

The aim of the study is to understand the perceptions and attitudes of the main actors engaged in environmental management and nature conservation in Greece towards CS. Three target groups, environmental data collection actors, from ENGOs, research centres and/or higher educational institutions (Universities/Technological Institutes) and government sectors were selected. To achieve this aim, five key questions were addressed:

- to what extent are the three groups of actors familiar with the term, the concept and practice of CS?
- what is the attitude of the actors towards CS?
- which are the main obstacles to the development of CS in Greece?
- which are the main reasons for the reduced visibility of the term in Greece?
- are there significant differences amongst the actors in relation to the abovementioned questions?

Materials and methods

The research approach was based on mixed methods and, more specifically, an explanatory sequential, where a quantitative step was first conducted, followed by a qualitative part to elucidate the most interesting findings. In the first quantitative phase of the study, survey data were collected from a number of respondents using an electronic questionnaire (Appendix 1 Questionnaire). The sample was drawn from a population of professionals within the field of environmental monitoring and nature conservation: employees and members of ENGOs, environmental scientists working on research centres and higher educational institutions (Universities/Technological institutes) and civil servants and government authorities all engaged in the topic of environmental conservation. This represents a convenient sampling technique; thus, our results cannot be considered representative and cannot be generalised. However, for exploratory studies like this one, convenience samples are considered sufficient (Sue and Ritter 2007, 25).

Our sample comprised 644 emails, both personal and organisational. Contact details including e-mail addresses used in this study were publicly available on the internet through the organisations' official website (for researchers and civil servants) and the database for ENGOS maintained and regularly updated by the National Centre for Social Research in Greece (1 EKKE/IAAK. (n.d.). Retrieved 17 February 2019, from http://ekke.gr/estia/eng pages/eng index.htm)

The questionnaire included questions on demographics, familiarity with the term, the concept and the practice of CS, using a Likert-scale with an aim to capture the perceptions and attitudes of the actors towards CS. The statements expanded from the

most cited positive effects of the practice (Devictor et al. 2010; Dickinson et al. 2010; Roy et al. 2012; Science Communication Unit 2013; Pocock et al. 2014) up to the most common reasons for reluctance to accept it, for example, data quality (Rotman et al. 2012, 2014; Hyder et al. 2015; Minkman et al. 2015; Geoghegan et al. 2016; Burgess et al. 2017; Haklay et al. 2017), as well as a question about the main obstacles to the development of CS in Greece. All the questions of the survey were compulsory, except for a question about the name of the organisation and the last field where the respondents were asked to provide their e-mails in case they were also willing to provide an interview. For the compulsory questions, the option "l do not know / l do not answer" or "l do not answer" was also given to the respondents.

We piloted the questionnaire by distributing to six researchers, one representative of an ENGO and a public servant who provided initial feedback. Their answers were not included in the final dataset, while those respondents did not answer the final version of the questionnaire. The survey was initiated on the 23 April 2018 and it was closed on the 17 May 2018. The software used to publish the survey and collect the responses was LimeSurvey Version 2.06, an open-source online survey tool installed on the servers of CIHEAM-MAICh. The questionnaire was designed to take approximately ten minutes.

Those quantitative data were used to answer four out of the five research questions. The quantitative data from the survey were imported and analysed in Rstudio Version 1.1.453 (R version 3.3.3). Ggplot2, reshape2 and sjp.likert functions in R were used to produce the graphs. Descriptive statistics, such as summaries, percentages, means and standard deviations, were also calculated.

After the initial data analysis, the fourth research question was formulated. The qualitative phase was conducted to help answer the fourth research question and explain the most important results of the third research question. For the second phase of the study, semi-structured interviews of eight experts were conducted.

For the purposes of the qualitative part of the research, a smaller sample of 30 possible interviewees was created after an initial analysis of the questionnaire data. The possible interviewees were selected on the basis of two criteria: they were familiar with the term and/or the practice of CS and they had responded positively in the final question of the survey concerning their availability for a telephone interview. First, the participants that met the above-mentioned criteria were identified and then ten for each group of actors were randomly selected. Even if the aim of the semi-structured interviews was to interview representatives from all three groups of actors, unfortunately, we did not receive a positive reply from any representative of the public sector. Thus we interviewed five employees/representatives from ENGOs and three scientists from the research/academic sector.

The interviews were undertaken from 26 June to 17 July 2018. The interview guide was composed of one introductory question, two main topics of discussion and a closing part where the interviewees were asked to write anything additional which they considered important. The introductory question was how the interviewee's work is related to environmental monitoring and nature conservation. One main topic of discussion was about possible reasons for the non-visibility of CS in Greece. Regarding

the other topic, the interviewees were asked to provide their opinion about the first two obstacles to the development of CS in Greece as depicted from the survey.

In particular, the methods used to answer the research questions were as follows: for the first research question, we calculated percentages from questions B3, B4 and B5 of the questionnaire (see Appendix 1). For the second research question, we employed questions C1 and C3 of the questionnaire (Appendix 1). Question C1 was a Likert-scale question of eighteen statements. Alpha function was used to calculate Cronbach's a for reliability analysis. To measure the positive or negative attitude of the respondents towards the Likert statements, we assigned numeric values from 1 up to 4 corresponding to their level of agreement (Not at all (1), A little (2), Significantly (3), A lot (4)). The negative wording statements were reversed. Then the sum for each statement was divided by the number of the respondents minus the "l don't know / l don't answer" cases. Summing the means of all the 18 statements, we concluded with a score between 18 and 72, depicting the attitude of the respondents towards CS. Then this score was converted to a 0–100 scale to facilitate interpretation. The same process was followed for all the respondents and for each group of actors. From question C3, a table of percentages was produced.

For the third research question, at first, we employed question C2 of the questionnaire and percentages were calculated. During the semi-structured interviews (see Appendix 2 for the list of interviewees), some clarification concerning the first two main obstacles was asked by the experts. The answers were analysed using a Thematic Analysis approach (Attride-Stirling 2001).

For the fourth research question, we analysed the transcribed experts' interviews using a thematic network analysis approach (Attride-Stirling 2001) to RQDA software. In total, 125 codes were generated. Out of these codes, fourteen categories, or themes, emerged in relation to the reasons for the non-visibility of the practice in Greece and to interpret better the two main obstacles in the development of the CS in Greece according to the actors.

Finally, for the fifth research question, comparisons amongst the three groups of actors were performed using the Kruskal-Wallis test, followed by post-hoc pairwise comparisons with the Dunn-Bonferroni approach.

Results

Out of the 644 e-mails sent, we received 80 delivery failures with 564 emails reaching their target. The survey resulted in 178 fully completed questionnaires, so a return rate of about 30% which is the usual response rate for web-based surveys (Sue and Ritter 2007, 8). From the survey respondents, 16.9% work or represent an ENGO, 30.9% are researchers and 52.2% are working in public bodies.

Regarding familiarity of the actors with the term (see question B3), 41.6% of the respondents answered "Yes", 57.9% answered "No" and 0.6% choose the option "l don't know/l don't answer". The percentages differed significantly when we examine the results within the three different groups of actors. Researchers were the most

familiar group with 72.7% answering that they know the term CS, followed by the representatives from ENGOs with 53.3% and then the public servants with 19.4%.

To investigate the familiarity of the actors with the concept, we employed question B4, with 34.8% of the respondents answering "None of the above", which means that the remaining 65.2% knew at least one of the terms which relate to CS (Fig. 1).

The researchers are the ones who were most familiar with the concept of CS since they had the lowest percentage (16.4%) for the option "None of the above" when asked if they know the terms. The second familiar group was the NGOs with 23.3% followed by the public sector with 49.5% (Fig. 2).

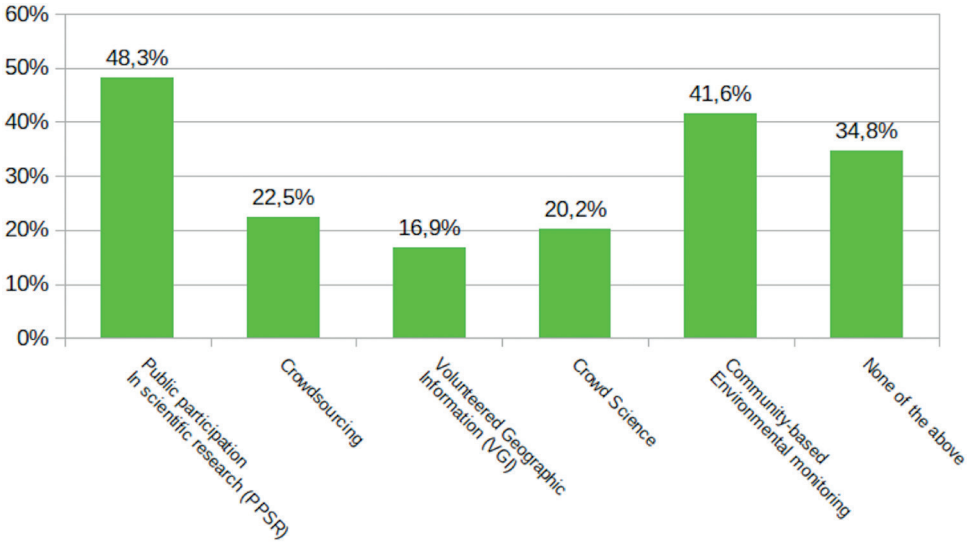

Figure 1. Responses to question B4 "Do you know any of the following terms" (all respondents).

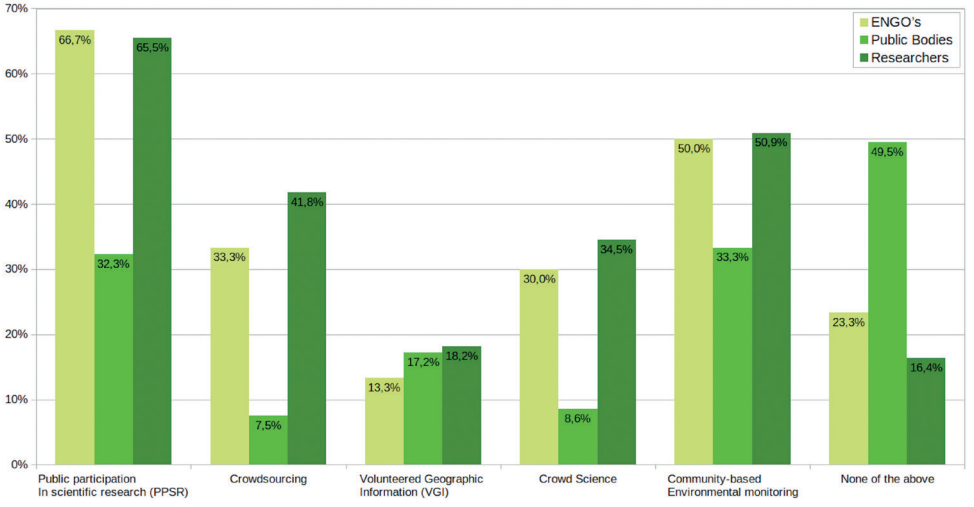

Figure 2. Responses to question B4 "Do you know any of the following terms" (per group of actors).

Finally, regarding the familiarity of the actors with the actual practice of CS, we employed question B5 for which 39.3% of the respondents answered "Yes" to that question, 57.3% answered "No" and 3.4% chose the option "l don't know/l don't answer".

In order to answer the second research question of this study, we analysed the responses to question C1 and C3 of the questionnaire. The Likert scale used in C1 had internal reliability: Cronbach's a = 0.79. As a general rule of thumb, reliabilities with Cronbach's a above 0.7 are considered acceptable (Cortina 1993).

For the analysis of the Likert scale, numeric values were assigned to the levels of agreement of the respondents and then the mean was calculated for each item and a total sum of all the items. Since the range of the numeric values that we chose to assign was from 1 up to 4, the mean of each item falls within this range. According to this rationale, an item with a mean close to the upper (4) or lower (1) value has a clearer depiction (positive or negative) of the attitude of the actors towards the relevant statement. On the other hand, items with a mean value around 2.5 are statements more debatable that divide the respondents. Moreover, the scores of the negative wording statements have been reversed. The sum of the items - for all the respondents and for each group separately - was converted to a 0–100 scale for better interpretation. The scores for all the respondents and for each group of actors are presented in Table 1.

Finally, regarding the attitude of the actors towards CS, question C3, 76.4% of the respondents answered "Yes", 6.7% answered "No" and the remaining 16.9% chose the option "l don't know/l don't answer".

Within the groups of actors, the researchers and the employees/representatives of the NGOs had very similar answers. Around 90% believe that a CS programme would be a positive addition to their activities. The majority of the public sector (63.4%) was also positive; 26.9% answered "l don't know/l don't answer" and 9.7% chose the option "No" (Fig. 3).

To answer the third research question of this study, we used the responses to question C2. The results are presented in Fig. 4 for all the respondents and Fig. 5 for each group of actors. In these two figures, the dark green colour represents the three most important obstacles while light green shows the obstacles above 20%. The three most important obstacles for all the respondents were "Lack of cooperative culture", "Ignorance of the existence of the phenomenon" and "Lack of know-how". The first two obstacles stand out from the rest with 53.9% and 45.5%, respectively. The third obstacle was selected in ca. 30% of the responses, while the rest of the obstacles received under 25%.

Table 2 presents all the obstacles above 20% - for all the respondents and for each group of actors. From this table, it becomes obvious that five obstacles are very important for all the actors while some obstacles are of importance mainly for the public bodies and the researchers.

Following the quantitative analysis, we aimed to further clarify the first two obstacles. Therefore, during the interviews, we asked the experts what they believe the reasons were for those obstacles. Regarding the first obstacle "lack of cooperation culture", one important reason that was mentioned by the majority of the interviewees was the problematic operation of the State. This statement refers to a range of activities, such

as inconsistent policies and official committees that never function, the absence of participatory processes, no continuity in the priorities of the ministries due to political changes and long delays. Some other reasons were: the absence of volunteering mentality in Greece, types of behaviour that make difficult the relationships between the research community and the NGOs, lack of proper information and finally, the lack of a culture of acknowledgment by both the State and the researchers creates problems in possible collaborations with the public and NGOs.

Regarding the second obstacle "Ignorance of the existence of the phenomenon", two main reasons emerged through the interviews. We termed the first one "Lack of external stimulus" to refer to bureaucratic organisations with no motivation for participatory research or practices. The second reason was that the actors (ENGOs, researchers, public bodies) have not communicated the term successfully to the general public, so there is a lack of relevant information.

Table 1. Mean values per statement of the Likert scale and total sum of the means.

Statements	All	NGOs	Public Bodies	Researchers
CS can support the collection of environmental data on a large geographic scale	3.04	3.39	2.83	3.19
Data gathered by citizens is not sufficiently reliable to use for public policy	2.48	3.00	2.34	2.44
CS can help environmental awareness of ordinary citizens	3.46	3.62	3.27	3.69
The quality of environmental data collected by non-professionals is inadequate for scientific research	2.55	3.07	2.41	2.49
CS can contribute to the collection of environmental data in cases of limited resources (Time, Money)	3.14	3.46	3.08	3.07
Increasing the phenomenon of CS may pose a threat to some jobs of professional scientists	3.45	3.79	3.31	3.49
CS can support government agencies in collecting environmental data as a cost-effective alternative	2.61	2.86	2.60	2.51
CS can help democratise science through the involvement of citizens in scientific processes	2.64	3.07	2.47	2.70
The collection of environmental data with low-cost devices such as smartphone sensors is un-acceptable in the context of scientific research	3.15	3.56	2.99	3.21
CS can help create social cohesion through voluntary engagement of citizens, building skills and engaging in problem-solving processes	3.12	3.48	2.97	3.16
Citizens do not have enough incentives to volunteer in scientific research	2.31	2.10	2.29	2.45
Citizens cannot follow the protocols required by the collection of environmental data in the context of scientific work	2.52	2.59	2.50	2.53
CS can help to involve different stakeholders in policy design and management of local ecosystems	2.93	3.24	2.75	3.05
CS can help to create creative activity for people outside the labour market, for example, retirees	3.01	3.28	2.87	3.09
With appropriate training, ordinary citizens can collect environmental data of satisfactory quality	3.15	3.62	2.99	3.18
CS can support local communities to protect the environment	3.27	3.55	3.12	3.36
Collaboration with volunteers from the general public is usually problematic	2.83	3.03	2.67	3.00
The resources required (time, money) for a citizens' science programme are excessive in relation to the results it generates	3.01	3.32	2.85	3.04
Sum (range 18-72)	52.67	58.05	50.30	53.68
Sum (conversion 0-100 range)	**64.21**	**74.16**	**59.82**	**66.08**

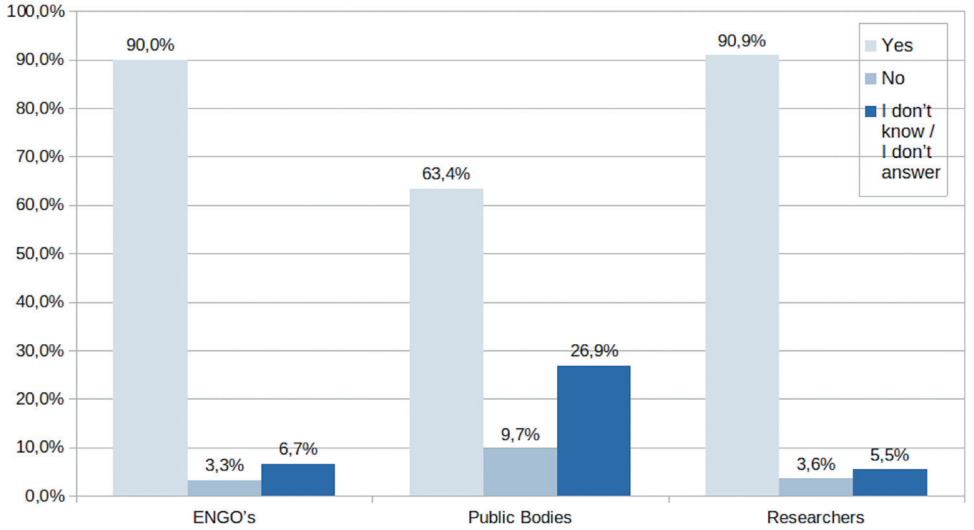

Figure 3. Responses to question C3 of the questionnaire (per groups of actors) "Do you think that a Citizen Science programme would offer something positive to your organisation's activities or to the research you are conducting?"

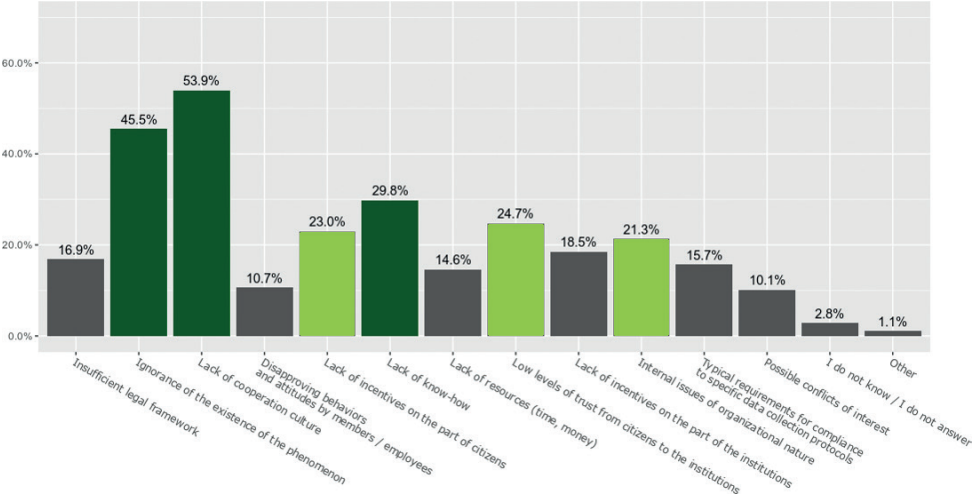

Figure 4. Main obstacles to the development of Citizen Science in Greece (all respondents). Dark green shows the first three obstacles, light green obstacles above 20%.

During the semi-structured interviews, the interviewees were asked their opinion about the main reasons for the reduced visibility of the term in Greece. A reason that was reported is that the researchers who participate in CS projects often do not publish due to data quality issues. Another reason is that CS within the organisations, if it exists, is a side-line activity, amongst others. Thus, it is not easy to be promoted because promotion needs extra time and effort and the resources normally are scarce. In addition, most of the CS projects that are initiated with some form of EU funding (e.g.

Horizon or LIFE+ programmes), do not continue once the funding is over (usually the end of the project). Therefore, there is insufficient time to develop and disseminate their added value in the society. Finally, the majority of the interviewees mentioned that the term is not so visible in Greece because organisations that run relevant projects do not use the term Citizen Science.

The familiarity with the term, the concept and the practice of CS differed significantly amongst the three groups of actors as demonstrated by the Kruskal-Wallis test (Table 3). Post-hoc pairwise comparisons, using the Dunn-Bonferroni test, revealed that respondents from public bodies are significantly less familiar with CS than respondents from ENGOs and the researchers.

From the eighteen statements of the Likert scale, the Kruskal-Wallis test depicted eight statements as statistically significant (Table 4). The most contradictory pair of actors is "ENGOS-public bodes" which is statistically significantly different in seven statements from the eighteen of the scale. Then the pair "researchers-public bodies" is significantly different in three statements and lastly, the "ENGOs-researchers" pair differs significantly on two occasions.

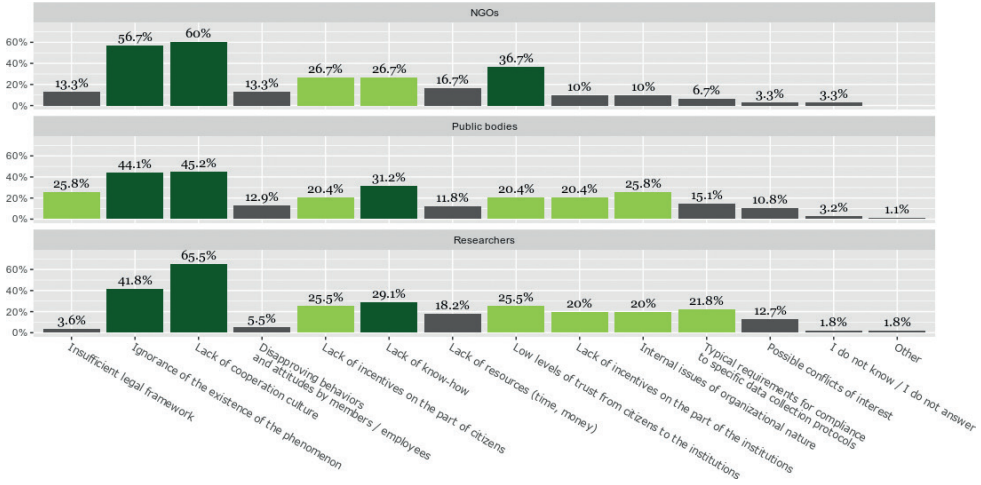

Figure 5. Main obstacles to the development of Citizen Science in Greece (per group of actors). Dark green shows the first three obstacles, light green obstacles above 20%.

Table 2. Obstacles over 20% for all the respondents and per group.

Obstacles	All	ENGOs	Public Bodies	Researchers
Lack of cooperation culture	53.9% (1st)	60% (1st)	45.2% (1st)	65.5% (1st)
Ignorance of the existence of the phenomenon	45.5% (2nd)	56.7% (2nd)	44.1% (2nd)	41.8% (2nd)
Lack of know-how	29.8% (3rd)	26.7% (4th)	31.2% (3rd)	29.1% (3rd)
Low levels of trust from citizens to the institutions	24.7% (4th)	36.7% (3rd)	20.4% (5th)	25.5% (4th)
Lack of incentives on the part of citizens	23.0% (5th)	26.7% (4th)	20.4% (5th)	25.5% (4th)
Internal issues of organisational nature	21.3% (6th)		25.8% (4th)	20.0% (6th)
Lack of incentives on the part of the institutions			20.4% (5th)	20.0% (6th)
Typical requirements for compliance to specific data collection protocols				21.8% (5th)
Insufficient legal framework			25.8% (4th)	

Table 3. Comparisons of familiarity with Citizen Science using the Kruskal-Wallis test followed by pairwise comparisons using the Dunn-Bonferroni test.

Familiarity	Kruskal-Wallis test			Pairwise comparisons		
	H	df	P-Value	Pairs	P-Value	r
With the term	42.6	2	0.000	ENGOs – Public **Bodies**	**0.0014***	3.3
				ENGOs – Researchers	0.1302	-1.71
				Researchers – Public **Bodies**	**0.0000***	-6.36
With the concept	18.68	2	0.000	ENGOs – Public **Bodies**	**0.0138***	-2.6
				ENGOs – Researchers	0.7806	0.64
				Researchers – Public **Bodies**	**0.0001***	4.07
With the practice	17.02	2	0.000	ENGOs – Public **Bodies**	**0.0193***	2.49
				ENGOs – Researchers	0.8105	-0.61
				Researchers – Public **Bodies**	**0.0002***	-3.89

*: Statistically significant difference detected at p ≤ alpha/2, alpha = 0.05 (after Bonferroni adjustment for multiple comparisons).

Table 4. Statistically significant Likert-scale statements using the Kruskal-Wallis test followed by pairwise comparisons using the Dunn-Bonferroni test.

Familiarity	Kruskal-Wallis test			Pairwise comparisons		
	H	df	P-Value	Pairs	P-Value	r
CS can support the collection of environmental data on a large geographic scale	11.52	2	0.003	ENGOs – Public **Bodies**	**0.0072***	-2.82
				ENGOs – Researchers	0.8027	0.62
				Researchers – Public **Bodies**	**0.0120***	-2.65
Data gathered by citizens is not sufficiently reliable to use for public policy	11.24	2	0.004	ENGOs – Public **Bodies**	**0.0021***	-3.2
				ENGOs – Researchers	**0.0047***	-2.95
				Researchers – Public **Bodies**	1	0.01
The quality of environmental data collected by non-professionals is inadequate for scientific research	12.81	2	0.002	ENGOs – Public **Bodies**	**0.0026***	3.14
				ENGOs – Researchers	**0.0009***	-3.43
				Researchers – Public **Bodies**	0.7264	-0.7
CS can help democratise science through the involvement of citizens in scientific processes	6.99	2	0.030	ENGOs – Public **Bodies**	**0.0175***	2.52
				ENGOs – Researchers	0.3469	1.2
				Researchers – Public	0.1942	-1.52
The collection of environmental data with low-cost devices, such as smartphone sensors, is un- acceptable in the context of scientific research	7.89	2	0.019	ENGOs – Public	**0.0091***	-2.74
				ENGOs – Researchers	0.206	-1.49
				Researchers – Public **Bodies**	0.2406	1.4
CS can help create social cohesion through voluntary engagement of citizens, building skills and engaging in problem-solving processes	6.55	2	0.038	ENGOs – Public **Bodies**	**0.0229***	2.43
				ENGOs – Researchers	0.3989	1.11
				Researchers – Public	0.1967	-1.51
CS can help to involve different stakeholders in policy design and management of local ecosystems	15.37	2	0.000	ENGOs – Public	**0.0027***	3.13
				ENGOs – Researchers	0.9311	0.49
				Researchers – Public	**0.0021***	-3.2
Collaboration with volunteers from the general public is usually problematic	10.13	2	0.006	ENGOs – Public	0.0408	-2.21
				ENGOs – Researchers	1	0.09
				Researchers – Public **Bodies**	**0.0066***	2.85

*: Statistically significant difference detected at p ≤ alpha/2, alpha = 0.05 (after Bonferroni adjustment for multiple comparisons).

Table 5. Statistically significant obstacles using the Kruskal-Wallis test, followed by pairwise comparisons using the Dunn-Bonferroni test.

Obstacle	Kruskal-Wallis test			Pairwise comparisons		
	H	df	P-Value	Pairs	P-Value	r
Insufficient legal framework	12.37	2	0.002	ENGOs – Public **Bodies**	0.1703	-1.58
				ENGOs – Researchers	0.3826	1.14
				Researchers – Public **Bodies**	**0.0008***	3.47
Lack of cooperation culture	18.68	2	0.000	ENGOs – Public **Bodies**	0.2361	1.41
				ENGOs – Researchers	0.946	-0.48
				Researchers – Public **Bodies**	0.0255	-2.39

*: Statistically significant difference detected at p ≤ alpha/2, alpha = 0.05 (after Bonferroni adjustment for multiple comparisons)

Lastly, two obstacles were statistically significant amongst the three groups of actors: “Insufficient legal framework” and “Lack of cooperation culture” (Table 5). After the Dunn-Bonferroni post-hoc pairwise comparisons, only the obstacle about the insufficient legal framework revealed a significantly differed pair. This pair was researchers vs public sector employees, with 3.6% of the former and 25.8% of the latter choosing "Insufficient legal framework” as one of the main obstacles to the development of CS in Greece.

Discussion

This study’s aim was to understand the perspectives of the three main groups of environmental actors in Greece towards CS. Similar studies about the main initiators of ECS projects (ENGOs, researchers, governmental organisations) are limited.

To our knowledge, no other comparative studies between the main initiating actors of ECS projects (ENGOS, researchers and relevant public bodies) have been conducted in Greece to date. Our initial hypothesis was that the majority of the actors in Greece would not be familiar with the practice. After the completion of the quantitative part of the study, it became evident that the reality in the field was rather different. Therefore, the findings that the three groups of actors have significant differences amongst them (Tables 3, 4 and 5), with researchers being more familiar amongst the three groups, with the most positive group towards CS coming from ENGOs, can be considered as the first of this kind.

Regarding the familiarity with CS, we are not aware of another study that quantitatively depicts the knowledge of the actors concerning the term, the concept or the practice of CS. Nevertheless, at a survey of public familiarity with the CS term and concept, we read “...we found that less than half of respondents were familiar with the term “citizen science,” but over 70% were familiar with the concept by another name. (Lewandowski et al. 2017). This is very similar to our questionnaire results where 41.6% knew the term, 65.2% knew the concept and 39.3% have participated in a project with CS characteristics.

We found that, in Greece, the term "Public participation in scientific research" (PPSR) is the most familiar amongst the representatives of the actors and this comes in contradiction with the notion expressed in other studies that PPSR has proven to be difficult to use and that the term "citizen science" is already well-established (Eitzel et al. 2017). Regarding the positive attitude of the actors towards CS, our findings are in agreement with previous studies. For example, Minkman et al. (2017) found that water practitioners in The Netherlands are willing to embrace CS while Riesch and Potter (2014) found that, for scientists participating in Open Air Laboratories (OPAL), CS projects have been a very positive experience.

The term crowdsourcing and crowd science were those for which the actors who took part in the study demonstrated less familiarity (Fig. 1). This is despite the increasing use of participatory research during the past 20 years. Although not synonymous, the terms are part of the gradient in participatory research where science meets society. Both CS and crowdsourcing may facilitate the dialogue between researchers and public and increase the influence of citizens on research agenda formulation. However, there have been increased concerns about crowdsourcing taking over citizen science which may result in displacing other forms of participatory research (Eitzel et al. 2017).

The concerns about data quality collected by citizens are well reported in literature (Burgess et al. 2017) and our study is no exception. Moreover, Riesch and Potter (2014), in exploring the perceptions of scientists about CS, found that the concern of scientists for any disapproval by their colleagues in case they use citizens' data might discourage them from doing so. Nevertheless, the actors in Greece believe, similarly to other studies (Danielsen et al. 2005; Bio Innovation Service 2018), that, with appropriate training, citizens can collect quality environmental data.

Although previous studies suggest that, very often, the main motivation for citizen scientists' involvement is to contribute to science (Raddick et al. 2013; Alender 2016; Land-Zandstra et al. 2016a), according to the opinion of the actors in our study "*citizens do not have enough incentives to volunteer/engage in scientific research*". Either the actors are right that the public in Greece have low incentives for engaging in general or the actors are failing to recognise volunteers' prevalent motivations (Rotman et al. 2012). Previous studies support strongly the opinion of the actors regarding the low levels of volunteering trust and social capital in Greece (Lyberaki and Paraskevopoulos 2002). For example, according to the 2008 European Social Survey, the percentage of Greeks who believe that one needs to be cautious of others and thus not trust them was almost double compared to the European average (Clarke et al. 2015, 11). Many studies have emphasized the crucial importance of trust-based relationships and credibility for a sustainable collaborative environment (Rotman et al. 2012; Stone et al. 2014; Vann-Sander et al. 2016). Such an environment is a must for ECS projects to flourish.

Semi-structured interviews revealed two possible causes regarding the first obstacle i.e. "Lack of cooperation culture". The first is the relationships amongst the three groups of actors (issues of trust, recognition, reward) and the second, a luck of culture of volunteering in Greek society. Those are in agreement with previous studies that argue for the low levels of trust, volunteerism and social capital in Greece (Clarke et al.

2015, 10–11). If we will consider that the fourth and fifth obstacle, "Low levels of trust from citizens to the institutions" (24.7%) and "Lack of incentives on the part of citizens" (23%), respectively, are somehow related with the issue of trust, we understand the multi-faceted influence of this factor.

Finally, our findings regarding the basic reasons of the reduced visibility of CS in Greece are in agreement with previous studies. More specifically, the first reason emerging from the semi-structured interviews conducted with the Greek actors was the under-representation of CS in formal literature. This possibility is well documented in the literature and has mainly two causes. The first one is related to the scientists' concerns about data quality (Burgess et al. 2017) and their perceptions that CS data will not be well received by other scientists (Riesch and Potter 2014). The second one is that some CS projects do not have as their objective peer-reviewed publication, for example, NGO projects that mainly aim to educate or affect policy (Burgess et al. 2017). The other reason for reduced visibility of the term in Greece was the usage of different terms in relevant projects. This finding is also in agreement with other studies reporting on a variety of terms used to describe data arising from citizens (Shirk et al. 2012; Kullenberg and Kasperowski 2016; See et al. 2016; Eitzel et al. 2017). The third obstacle about problematic or no promotion of relevant projects, for example, no website, no relevant articles in the press, is mentioned in a report by Roy et al. (2012), but also by other authors who tried to create indexes of CS projects (Chandler et al. 2017; Pettibone et al. 2017; Pocock et al. 2017; Bio Innovation Service 2018).

Implications for Future CS Attempts and Further Work

A key finding of this study was that a significant percentage of actors in Greece, who are professionally involved in the environmental data collection procedure, are familiar with the term, the concept or the practice of CS. This finding can have important implications for future ECS attempts since it makes clear that familiarity with CS is not the main issue amongst the actors. A practical step following this finding could be the organisation of a congress or a forum about ECS in Greece. Such an event would be important in order to build synergies and trust amongst the actors, that is partly, the number one obstacle to the development of CS in Greece according to the actors.

Since we know which terms are the most familiar within the groups of actors, a possible implication is that the concept of CS can be communicated more effectively in order to be understood by the audience. For example, if we want to communicate the idea to a governmental institution, it is advisable to mention also the terms "community - based environmental monitoring" or "public participation in scientific research" since our chances to convey the concept will significantly increase. The same goes for the ENGOs, mainly for the term "public participation in scientific research".

Our findings showed that all the groups of actors (ENGOs, researchers, public bodies) are positive towards CS, although to varying degrees. This information could be important if we would like to follow the advice of Bonney et al. (2009) "*A successful*

CS project requires a development team comprising multiple disciplines. ... Small groups or organisations that do not have internal access to all disciplines can partner with other organisations or adapt national CS projects for use at local or regional scales.". In Greece, the majority of the organisations are small to medium size, so the idea of partnerships probably is the most viable one. Of course, the issue of trust and the consequent lack of cooperation culture in Greece are the main obstacles to the development of CS in the country. However, a practical result of this study is the specific information arising about each group of actors and which can be used for future efforts. For example, we identified the existence of a legal framework as important factor for the public bodies and compliance to specific data collection protocols important for scientists. Moreover, from the analysis of the Likert scale, we can observe differences amongst the groups of actors in greater detail.

In addition to similarities, the study also highlighted significant differences amongst the group of actors who usually initiate ECS projects. These actors are characterised by different levels of familiarity with CS and relevant terms, they prioritise differently the obstacles and they are more positive or negative towards different aspects of CS. Further studies are needed to understand the motivations of scientists and citizens to engage in collaborative projects. For example, Rotman et al. (2012) found that the motivations of participants shift over time and often the scientists are unaware or mistaken about those. Another study showed that the perceptions about CS between experts and citizens can differ and this can lead to conflicts and distrust (Weng 2015) or that the experts and the citizens have different views of formal and informal environmental sensing data (Jiang et al. 2018). These studies can be supportive especially for building synergies and interdisciplinary ECS working groups. More in-depth research in the direction of assessing the perspectives of the actors towards possible collaborations, needs assessment for building ECS projects, mapping the different organisational skills and aligning them with the needs within the framework of synergies could be of much help for the development of the field. Moreover, research on case studies of ECS projects that are collaborations of ENGOs, governmental organisations and research institutes we believe are of high importance. Lessons learned and best practices could be of much support, especially for similar contexts to Greece where organisations are of small size without specialised know-how. Future qualitative research should be conducted regarding civil servants and especially their point of view for the main obstacle to the development of CS in Greece that was "Lack of cooperative culture". Despite our efforts, their point of view is missing in this study. We believe that is of great importance if we want to understand better the relationships amongst the actors.

The factors that influence the application of CS are diverse and include the specific socio-economic characteristics of each country and geographical regions, history and culture of volunteerism, NGO activity, social capital etc. (Burgess et al. 2017). This study was an exploratory one for Greece and we are aware of at least one in a similar context in Romania (Loos et al. 2015). Despite the fact that the interviews and the questionnaire were carried out in 2018 and significant progress in CS has been made

since then, the results set a baseline for further similar studies in Greece as a means to monitor progress in a fast-expanding field.

The daily use of mobile apps, social media and online platforms has increased familiarity of lay persons with similar tools engaged in CS nowadays (Willis et al. 2017; Zotos and Vogiatzakis 2018). This familiarisation became a necessity during the ongoing COVID-19 pandemic and resulted in a fast-paced mass digital transition. The ability of the actors in Greece and elsewhere to take advantage of this transition, so to create CS online projects, is an important parameter that we have not explored in this study.

As CS advances, new challenges emerge, such as the participants' personal data protection in CS projects (Suman and Pierce 2018). In the European Union, the legal context is quite advanced with General Data Protection Regulation (G.D.P.R) being the main legal text since 2016 (European Parliament and the Council of 27, 2016). However, the level of familiarity and compliance of the actors with G.D.P.R obligations is still variable and should be the focus of future research.

There is a need for more regional studies if we want to better understand the specific challenges to the development of CS - and of citizen engagement in biodiversity and conservation science in general - in modern societies.

Conclusions

Contrary to our initial hypothesis, we demonstrated that the actors in Greece are familiar with and have a positive view of CS to a great extent (majority of respondents in Academia and ENGOs and ca. 50% of public servants). We found significant differences amongst the groups of actors regarding aspects of CS - such as familiarity with it - but also similarities, for example, concerning the main obstacles to the development of CS. We argue for the importance of comparative studies amongst the actors in terms of building knowledge to support synergies and interdisciplinary working groups. The main reasons for the reduced visibility of the term CS in Greece became obvious and are in agreement with previous research. The specific historical and socio-economic context which characterises a country or a geographically-defined area underpins the adoption and implementation of CS. We argue that more studies in different socio-economical contexts and possible comparisons between them, would be of scientific interest and of practical use to formulate public policies. For example, a study from Romania exploring the challenges to initiate a new CS monitoring scheme (Loos et al. 2015) would be of more relevance with the Greek context rather than studies from the US or UK. In addition and given that "ignorance of the existence of the phenomenon" was ranked the second most important obstacle by the actors for the development of CS, there is a need for raising public awareness of CS itself. Towards that direction, a web portal would support further the development of CS in the country by disseminating results beyond the end of each project and create a web presence for projects which currently lack one.

Acknowledgements

We would like to thank the eight interviewees for the willingness and time to participate in this study, namely: Giorgos Theodoridis, Stelios Kazantzis, Christos Arvanitidis, Giorgos Kalliampetsos, Christos Georgiadis, Christina Contaxi, Manolia Vougioukalou and Thomi Vlachogianni. In addition, we are grateful to all those who responded to the questionnaire.

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Appendix I

Questionnaire
Demographics
A1(Q1). What is the type of organisation you represent or work with? (List (radio))
• Environmental NGO
• University / Technological institute
• Research Centre
• Protected area management body
• Decentralised administration / local government
• Ministry
• Natural History Museum
Other
A2(Q2). Name of the organisation you represent or work with (Short free text)
A3(Q3). Working area (List with comment)
• Athens
• Thessaloniki
• Other city (10,000 + residents)
• Town (2,000 up to 10,000 residents)
• Village (up to 2,000 residents)
• I don't answer
A4(Q4). Gender (List (radio))
• Man
• Woman
• I don't answer
A5(Q5). Age (List (radio))
• 18–24
• 25–34
• 35–44
• 45–54
• 55–64
• 65 and above
• I don't answer
A6(Q6). Education level (List (radio))
• I didn't go to school at all
• Secondary education graduation
• Private Institute for Vocational Training
• Public Institute for Vocational Training
• University / Technological institute
• Postgraduate / Doctorate
General questions related to the Citizen Science phenomenon
B1(Q7). As an institution (or as a researcher), have you been involved in an environmental data collection activity? (List (radio))
• Yes

• No
• I don't know / I don't answer
B2(Q8). As an institution (or as a researcher), have you ever involved ordinary citizens as volunteers in environmental actions that you have organised? (List (radio))
• Yes
• No
• I don't know / I don't answer
B3(Q9). Did you know the term 'Citizen Science' before completing the questionnaire? (List (radio))
• Yes
• No
• I don't know / I don't answer
B4(Q10). Do you know any of the following terms? (Multiple choice)
• Public participation in scientific research (PPSR)
• Crowdsourcing
• Volunteered Geographic Information (VGI)
• Crowd science
• Community-based environmental monitoring
• None of the above
B5(Q11). As an institution (or as a researcher), have you been involved in a scientific programme for collecting environmental data with Citizen Science features i.e. involving ordinary citizens?
(List (radio))
• Yes
• No
• I don't know / I don't answer
Attitudes, Perceptions, Obstacles
C1(Q12). How much do you agree with the following statements? - A lot, Significantly, a little, not at all, I don't know / I don't answer - (By the term Citizen Science, we mean scientific activities in which ordinary citizens participate voluntarily in the collection of data and/or in the analysis and/or dissemination of a scientific work) (Array)
• Citizen Science can support the collection of environmental data on a large geographic scale
• Data gathered by citizens is not sufficiently reliable to use for public policy
• Citizen Science can help environmental awareness of ordinary citizens
• The quality of environmental data collected by non-professionals (ordinary citizens) is inadequate for scientific research
• Citizen Science can contribute to the collection of environmental data in cases of limited resources (Time, Money)
• Increasing the phenomenon of citizen science may pose a threat to some jobs of professional scientists
• Citizen Science can support government agencies in collecting environmental data as a cost-effective alternative

• Citizen Science can help democratise science through the involvement of citizens in scientific processes
• The collection of environmental data with low-cost devices, such as smartphone sensors, is unacceptable in the context of scientific research
• Citizen Science can help create social cohesion through voluntary engagement of citizens, building skills and engaging in problem-solving processes
• Citizens do not have enough incentives to volunteer in scientific research
• Citizens cannot follow the protocols required by the collection of environmental data in the context of scientific work
• Citizen Science can help to involve different stakeholders in policy design and management of local ecosystems
• Citizen Science can help to create creative activity for people outside the labour market, for example, retirees
• With appropriate training, ordinary citizens can collect environmental data of satisfactory quality
• Citizen Science can support local communities to protect the environment
• Collaboration with volunteers from the general public is usually problematic
• The resources required (time, money) for a citizens' science programme are excessive in relation to the results it generates

C2(Q13). What do you think are the main obstacles to the development of the Citizen Science phenomenon in Greece? Please select no more than 3 replies (Multiple choice)
• Insufficient legal framework
• Ignorance of the existence of the phenomenon
• Lack of cooperation culture
• Disapproving types of behaviour and attitudes by members/employees, for example, negative attitude from the management
• Lack of incentives on the part of citizens
• Lack of know-how (technical issues, volunteer management …)
• Lack of resources (time, money)
• Low levels of trust from citizens to the institutions
• Lack of incentives on the part of the institutions (distrust of the results of such a programme, for example, data quality)
• Internal issues of organisational nature (e.g. rigidity, bureaucracy)
• Typical requirements for compliance to specific data collection protocols
• Possible conflicts of interest, for example, employees who are employed in data collection to treat it as a threat
• I do not know / I do not answer
• Other

C3(Q14). Do you think that a Citizen Science programme would offer something positive to your organisation's activities or to the research you are conducting? (List with comment)
• Yes
• No

• l don't know / l don't answer
Question for interview
D1(Q15). In the case of a telephone interview for the purposes of this diploma thesis (duration of 30' up to 60' minutes) (List with comment)
• I would probably be positive
• Maybe
• In no case
D2. Contact info, email (Short free text)

Appendix 2

Table A1. List of Interviewees.

Interviewee number	Role	Date
Interviewee 1	Employee at the Environmental Organisation for Wildlife and Nature CALLISTO	26 June 2018
Interviewee 2	Member of environmental NGO "Ecological Collaboration"	28 June 2018
Interviewee 3	Programme/Policy Officer at The Mediterranean Information Office for Environment, Culture & Sustainable Development (MIO-ECSDE)	17 June2018
Interviewee 4	Associate researcher Institute of Environmental Physics and Sustainable Development, National Observatory of Athens, Greece	27 June 2018
Interviewee 5	Senior Researcher at Institute of Marine Biology, Biotechnology and Aquaculture (IMBBC), Hellenic Centre for Marine Research (HCMR)	27 June 2018
Interviewee 6	Researcher at Museum of Zoology of the National and Kapodistrian University of Athens	28 June 2018
Interviewee 7	Employee at the environmental NGO Hellenic Ornithological Society	6 July 2018
Interviewee 8	Director of environmental NGO MEDITERRANEAN SOS Network	5 July 2018

Nature Conservation 48: 57–81 (2022)
doi: 10.3897/natureconservation.48.77722
https://natureconservation.pensoft.net

RESEARCH ARTICLE

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Not only range, but quality: human influence and protected areas within the distribution of mammal species subject to use in the Department of Cundinamarca, Colombia

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Academic editor: W. Magnusson | Received 7 November 2021 | Accepted 7 February 2022 | Published 7 April 2022

http://zoobank.org/AD998F0C-A745-4109-BF04-D13BD7F6057B

Citation: Aguirre Sierra L, Zárrate-Charry DA, Lemus-Mejía L, Morales-Perdomo J, González-Maya JF (2022) Not only range, but quality: human influence and protected areas within the distribution of mammal species subject to use in the Department of Cundinamarca, Colombia. Nature Conservation 48: 57–81. https://doi.org/10.3897/natureconservation.48.77722

Abstract

Mammal conservation in transformed landscapes depends heavily on the role of protected areas, especially for species used by local communities both within and around these areas. We evaluated the level of representation and the magnitude of the influence of humans, via human footprint, across the range of mammals used by local communities in the Department of Cundinamarca, Colombia. We emphasised the differences of the human influence at a department scale and inside Protected Areas (PA). The definition of species used by local communities refers to using a resource for its economic, religious and/or traditional value. Specifically, we addressed whether there is a difference between the magnitude of human influence inside and outside the PAs and if the impact is greater on threatened species, species with greater or lesser representation or according to their use. We found 43 species subject to use in our analysis, with low values of representation when compared with global targets ($\bar{X}$ ± CD = 10.69% ± 4.99) and with high values of vulnerability, based on the mean value of the Spatial Human Footprint Index (HSFI) (57 ± 2.74). We found a difference of 10.72 points between the average HSFI of the Department and that of the PAs ($\bar{X}$ ± CD = 10.73 ± 5.98%). This shows that the status of each species' habitats is less impacted by hu-

man activities within PAs and that the conservation areas for all species depend largely on their presence in largely transformed landscapes. Although this seems an expected outcome, the Department of Cundinamarca is one of the less represented on PAs at a national level and has suffered from severe fragmentation; thus, our results highlight the need for improving and expanding the current PA system as most species, especially those subject to use, will depend on their existence for their conservation on the long run.

Keywords
Andes, human footprint, Protected Areas, species modelling, species range

Introduction

Mammals are one of the main groups widely used to assess landscapes and ecosystems' ecological integrity and health in different parts of the world (Rondinini et al. 2011; González-Maya et al. 2015; Di Minin et al. 2016; González-Maya et al. 2016). This largely responds to their role in the functioning and maintenance of ecosystems, which supports its use for conducting conservation status assessments (Aubry et al. 2003; Prugh et al. 2009; Ripple et al. 2014). Mammals stand out for their ability and capacity to disperse seeds, maintain the balance of trophic chains and their role as soil fertilisers and pollinators, amongst many others (Aubry et al. 2003; Noss et al. 2012; Lacher et al. 2019). Likewise, they have been widely used to understand protected areas (PA) status, connectivity and conservation contribution, particularly when analysing at landscape scales (Beier 1993; Cullen et al. 2013; Zárrate-Charry et al. 2018).

Even when mammal species provide a diversity of ecosystem services, they are one of the most threatened taxonomic groups globally due to the loss of their habitat, overexploitation, climate change among others (Schipper et al. 2008). These pressures, that overall affect biodiversity in general, are considered severe for many mammals since most of them have high energy requirements, usually depending on quality habitats and abundant resources (Schipper et al. 2008; Ripple et al. 2014); although some species might tolerate certain levels of intervention, overall, mammals require habitats and resources which in general make them good ecological indicators (Sinclair 2003; Schipper et al. 2008; Pineda-Guerrero et al. 2015; González-Maya et al. 2017). In addition to being one of the most important groups of animals for most ecosystems, mammals are also one of the groups most directly used by human communities (Cortés-Gregorio et al. 2013; Ripple et al. 2014; Van Vliet et al. 2015). From a sociocultural perspective, they are not only a tangible resource, object of appropriation and the basis of various recreational, cultural and subsistence needs, but they are also part of the collective imagination of intangible forms, whether associated with myths, legends, art or folklore, even contributing to the identity of various peoples and communities (Vargas-Clavijo 2008; Vargas-Clavijo 2009).

Along with the many strategies for mammal conservation, various management actions have been designed, ranging from conservation plans (Castaño-Uribe et al. 2013; Ministerio del Ambiente and Wildlife Conservation Society 2014), strategies

for sustainable use of species or landscapes (Fischer et al. 2010; Sims and Alix-Garcia 2017) and strategies for landscape conservation, for which Protected Areas (PAs) continue to be one of the main pillars (Stolton and Dudley 2010; González-Maya et al. 2015; Di Minin et al. 2016; Zárrate-Charry et al. 2022). PAs are an essential tool for ensuring the natural and cultural heritage of a country (Sánchez-Azofeifa et al. 1999; Forero-Medina and Joppa 2010), and these protected landscapes aim at safeguarding both natural and cultural elements that are representative of a particular region (Davey 1998; Loucks et al. 2008; Forero-Medina and Joppa 2010; Roncancio-Duque and Vélez Vanegas 2019). Previous efforts have contributed to recognising that PAs play an important role in maintaining patterns of land use and biodiversity, which contributes to social aspects and in the preservation of various species and cultural characteristics (Olmos Martínez et al. 2013); PAs are critical for the provision of environmental goods and services, while safeguarding critical habitats for the maintenance of species (Armsworth et al. 2007; Luck et al. 2009; Ferraro et al. 2011).

Colombia is considered the sixth country with the highest mammal richness worldwide, with about 530 species (Ramírez-Chaves et al. 2019). Of these, 236 species are confirmed for the Cundinamarca Department (Lemus-Mejía 2021). Despite this large number of mammal species, there is minimal information on integrating these species into the different management plans or conservation strategies applied in the areas set aside for this purpose (Sánchez et al. 2004). PAs and the landscapes in which they are located, have been affected in recent decades by the increase of various stressors that directly affect biodiversity. Some of the main stressors identified for Colombia include deforestation, agriculture, poaching, presence of exotic invasive species, among others (Parques Nacionales Naturales de Colombia 2021); all these have a great effect on biodiversity and are now included as targets and priorities in different plans for their mitigation. At the national level, about 9.6% of the total area of the National System of Protected Areas (**SINAP**) has been transformed (IDEAM et al. 2017). This, coupled with the increasing rate of deforestation (Clerici et al. 2020), seriously affects the ecological processes on which species and ecosystems depend. Added to this, pressures associated with the drivers of global change, such as the unsustainable use of natural resources, the increase in the presence and abundance of invasive species and the challenges imposed by climate change, seriously affect the habitats of most species (Guerra et al. 2019; Clerici et al. 2020; Harfoot et al. 2021; Murillo-Sandoval et al. 2021). Likewise, the direct pressures generated by the increasing use of resources and territory, due to accelerated demographic growth, generate the expansion of more urban areas (Alberti and Marzluff 2004; Etter et al. 2008; Curtis et al. 2018), reflecting on the intensity generated by the anthropogenic impact on terrestrial ecosystems where the human contribution is increasing (Sanderson et al. 2002; Correa Ayram et al. 2020).

This trend, associated with the increase in biodiversity loss stressors, generates an urgency for the conservation and management of species and their habitats, especially those subject to direct use (Bogoni et al. 2020; Green et al. 2020; Nickel et al. 2020). To better design management and conservation strategies for species, it is vital to un-

derstand the potential effect that the transformation of ecosystems and human actions has on the habitats and distribution areas of the species (Bogoni et al. 2020). Correa Ayram et al. (2020), through the multitemporal analysis of the Human Spatial Footprint Index (a measure that assesses the human impact on ecosystems derived from multiple variables), managed to identify that, in the last 45 years, the impact, or the magnitude of the human footprint, has increased by 50% in Colombia, with the Caribbean and the Andes being the regions where this increase has been greater. Likewise, they evaluated the future trend and predicted that, if there is no change in the pattern of use, by the year 2030, the Human Spatial Footprint Index will have increased by 12% more. In addition, approximately 65% of the land has already been subject of transformation processes (Correa Ayram et al. 2020). Specifically for the Andean Region, current values of the HSFI are estimated as high (Fig. 1A) and this has reflected on decreasing wealth of local or regional fauna (Etter and van Wyngaarden 2000; Etter et al. 2006) and triggered concentration of species populations to the small vegetation fragments that remain relatively intact (Armenteras et al. 2003; Cortés-Delgado and Pérez-Torres 2011; Magioli et al. 2021).

Here, we evaluated the representation of mammals in PAs and the magnitude of the human influence over the range of mammal species subject to use in the Cundinamarca Department and whether this influence is less significant within PAs. For this purpose, we developed three specific objectives: i) to analyse the representation of mammal species subject to use within the current PAs system, ii) to evaluate the magnitude of human influence, using the Human Spatial Footprint Index (HSFI) approach, across the range of all mammals subject to use inside and outside PAs and iii) to identify differences in the magnitude of human influence according to representation and different level of threat.

Materials and methods

Study area

The study area comprised the Cundinamarca Department (political division homologous to states), located in central Colombia, in the Andean Region (Fig. 1). Cundinamarca is the most populated Department in Colombia and includes the country's capital city. The Department has an average altitude of 3,341 m a.s.l., a total area of 24,210 km^2 distributed in 116 municipalities with 2,919,060 inhabitants without considering the capital city's population (7,743,955 inhabitants) (DANE 2019). Being one of the most populated regions also presents one of the highest levels of transformation (Fig. 1A). The Department includes 184 protected areas distributed in 73 of the national or regional level and 111 private reserves of the civil society, covering 315,894.45 hectares; many of these areas are very small (mean area of 41.6 ha) and scattered through the Department (RUNAP 2019) (Fig. 1B).

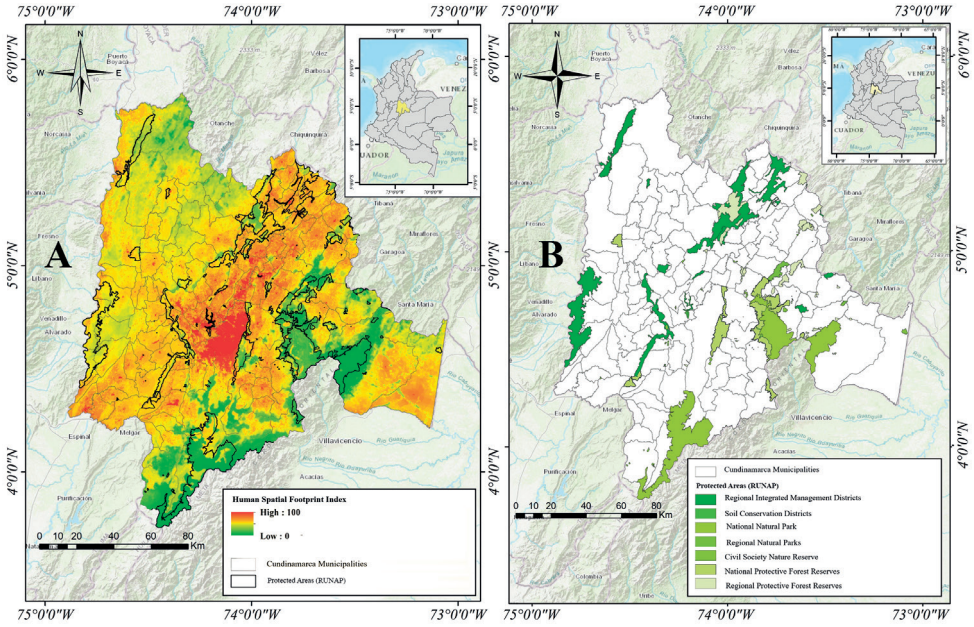

Figure 1. Human Footprint and Protected areas in Cundinamarca **A** representation of the Human Spatial Footprint Index and **B** location of Protected Areas in the Department of Cundinamarca, Colombia. PA categories are based on the national classification.

Species selection

In our study, we analysed the current human influence within the potential distribution range of mammal species used by local communities. The definition of species used by local communities refers to using a resource for economic, religious and traditional values governed by social, cultural and economic trends (Racero-Casarrubia et al. 2008; Cunha-Ribeiro and Schiavetty 2009). These species are used as the study object because they can generate, in the short term, a greater impact on the well-being of local communities; if their abundance or presence is affected, it will have a direct effect on ecosystems and their services, thus affecting human well-being. For all species confirmed in the Department (González-Maya et al. 2021b; Lemus-Mejía 2021), we conducted a bibliographic search for their potential social and cultural uses worldwide (e.g. food, medicine, economy, religion and others). Specifically, we searched for all reported uses on multiple databases, including the IUCN Red List of Threatened Species (IUCN 2018) and complemented with local and regional literature available on multiple databases (i.e. Google Scholar, Web of Science, SCOPUS, among others.). We used Boolean operators (AND, OR and NOT) creating different searching equations that combine the colloquial and scientific name of each species with words, such as “Food”, “Pet”, “Control”, “Subproducts”, “Medicine” and any synonym of these words that may lead to finding information about the possible uses of the species. This first lit-

erature search was a basic approximation to make the first filter on the species that were considered as subject of use. Complementarily, we conducted interviews across the Department to survey potential uses for the entire species list. The interviewed people were from 20 municipalities of the Department, all identified as areas of high vulnerability due to presence of forest remnants and high species richness combined with high levels of human transformation. We conducted a semi-structured interview including a visual guide of species potentially present in the area and the different types of uses (Barbosa Camargo 2020), amongst other questions related to the perception of changes in their abundance, conflict, amongst others. In order to characterise the type of use or value that communities give to each species, we interviewed about the type of relationship that each interviewee considered to have with each of these species. The relationship could vary from conflict events and retaliatory killing to direct consumption, commercialisation, use of it as products, medicine, contemplation or any other cultural relationship (Castaño-Uribe et al. 2013; Ministerio de Ambiente y Desarrollo Sostenible and Fundación Omacha 2016; Tinoco-Sotomayor et al. 2021).

To secure a representative sample size, we defined the number of interviews for each locality according to the extension of each municipality and the human population census for 2018 (DANE 2019). Based on the corresponding number of interviews for each municipality, we identified cells with forest assuming the presence of wildlife and the potential use by the communities; thus, we located core areas where we conducted the interviews. From February to March 2020, we conducted 200 interviews in over 120 villages of 20 prioritised municipalities and to people ranging between 15 and 70 years of age and with at least six months of residence in the area.

We then categorised each mammal species according to four use categories: Food, defined as any direct consumption of a mammal; Pet/Traffic, defined as any report of direct use as a pet or subject to illegal traffic for multiple purposes; Control, defined as those species subject to retaliatory killing, usually due to previous conflict or considered as a "pest" and; Subproducts, defined as those species used whole or their parts for the production of a secondary product (Osbahr and Morales 2012; Castaño-Uribe et al. 2013; Barbosa Camargo 2020). Furthermore, we categorised each species according to its international conservation status, following the IUCN Red List of Threatened Species Categories and Criteria (Critically Endangered (**CR**), Endangered (**EN**), Vulnerable (**VU**), Near Threatened (**NT**), Least Concern (**LC**), Data Deficient (**DD**) or Not Evaluated (**NE**)).

Potential distribution and representation

We estimated representation as the percentage of potential distribution, or range extension, of a species that is currently protected or included by existing PAs (González-Maya et al. 2015; House et al. 2017). In order to have a range for each species, we based our analysis on models for all species (González-Maya et al. 2021b; Lemus-Mejía 2021), based on an ecological niche modelling approach (Peterson et al. 2011). In order to create distribution models for all species, we constructed a database composed

of historical and current records for the 236 mammalian species confirmed for the Department of Cundinamarca (González-Maya et al. 2021b; Lemus-Mejía 2021); records were obtained from different sources, such as biological collections and published and grey scientific literature available in the country (Zárrate-Charry 2018; González-Maya et al. 2021b (i.e. SIB Colombia, VertNet, GBIF). To construct species distributions, we used the occurrence records that passed a quality filter and a spatial filter thinning, focusing our analysis on species with 20 or more records (Lemus-Mejía 2021), thus securing a better distribution hypothesis. We used a set of bioclimatic variables (Fick and Hijmans 2017) with a 1 km^2 resolution: Bio1 (Annual Mean Temperature), Bio2 (Mean Diurnal Range), Bio4 (Temperature Seasonality), Bio12 (Annual Precipitation) and Bio15 (Precipitation Seasonality). The variables Bio13 (Precipitation of the Wettest Month), Bio14 (Precipitation of the Driest Month) and elevation (Instituto Geográfico Agustín Codazzi 2016) were additionally included since the first two could reflect the ENSO (El Niño-Southern Oscillation) phenomenon and the third is considered a proxy for variables, such as radiation and oxygen concentration (Burneo et al. 2009). We developed potential distribution models using RStudio Desktop 1.4.1106 (R Team Development Core 2019) and the Wallace package (Kass et al. 2018). Details on the modelling approach and the specific models constructed for our analyses are also available elsewhere (González-Maya et al. 2021b; Lemus-Mejía 2021). A total of 30 models for each species were created (Lemus-Mejía 2021) using six different values of Regularisation Multipliers (RM; 0.5–3 in intervals of 0.5), five Feature Classes (FC) combinations (linear = **L**, linear-quadratic = **LQ**, hinge = **H**, linear-quadratic-hinge = **LQH** and linear-quadratic-hinge-product = **LQHP**), with 10,000 background points. The best models for each species were selected from the AUC test and the AICc Delta value (Zárrate-Charry et al. 2018). We then compared the areas covered by the binary representation of the species distribution model with the most updated polygon of all PAs in the Department (RUNAP 2019). A representation value was estimated for each species, including the total area of the Department covered by the potential distribution and the percentage of the range included in PAs and, likewise, the average representation value of the total species within the Department. Additionally, we compared representation values for species categorised under any threat category and non-threatened species using a Mann Whitney U test. Furthermore, considering that different types of uses represent differential pressures for each species, according with the type of use, we compared the representation between types of uses and within each type of use using a Kruskal-Wallis non-parametric test.

Human influence on landscapes (Human Spatial Footprint Index-HSFI)

To evaluate the human influence over each mammal species subject to use, we used the most updated Human Spatial Footprint Index (HSFI) for the country with an accuracy of 300 m^2 (Correa Ayram et al. 2020). This Index shows a spatial representation of the cumulative impact that human pressures have on the environment (Venter et al. 2016). The human footprint measures directly, through spatial data, the impact

on demand and consumption that humans have on Earth and human practices that are significantly reducing the resilience or recovery capacity of ecosystems causing irreversible effects on diversity, such as local extinction of species (Correa Ayram et al. 2017). The three dimensions assessed are soil intensity, time of anthropogenic intervention and biophysical vulnerability (Correa Ayram et al. 2020). This Index has been widely used to assess landscape changes and humans' potential impact on both species' habitat and connectivity (Nori et al. 2015; Correa Ayram et al. 2017). We overlapped the Human Spatial Footprint Index (**HSFI**) with the potential distribution areas of all species. This process allowed us to obtain a layer of values associated with the HSFI for each species, calculating a mean value and a standard deviation of the HSFI for each species for the whole Department. We then evaluated the mean value of the HSFI for the orders to identify potential groups that present a greater vulnerability due to the low quality of their distribution areas and due to the high values of human impacts. To evaluate the role that PAs can play for maintaining quality habitats for all species, we performed an evaluation of the mean value of the HSFI within the PA and contrasted these values with the total HSFI value of its distribution area; we used a t-paired test to explore if statistically significant differences existed between HSFI values in and out Protected Areas. This procedure also allowed obtaining a mean value (± SD) to understand the degree of human influence present in species distribution within PAs. The Human Spatial Footprint Index values are presented from 0 to 100, with 0 being the areas considered "natural" and 100 the areas with the maximum value of Human Footprint or anthropogenic impact (Correa Ayram et al. 2020). Finally, we compared the HSFI overall for the Department and within PAs between types of uses and within each type of use, using a Kruskal-Wallis non-parametric test.

All geographic analyses were performed using ArcGIS 10.x (Environmental Systems Research Institute 2016) and all statistical analyses were performed in R language (R Team Development Core 2021).

Results

We identified 43 species subject to use for the Department, mostly associated with direct use, but with some others related to cultural and religious uses (Table 1). Of the total 43 species subject to use in the Department, nine are threatened according to the IUCN Red List of Threatened Species and 10 are threatened according to the country's national legislation. Carnivora was the order with the highest number of species analysed with 16 (37.21%), followed by the order Primates with seven, Rodentia with six and Pilosa with five (16.28%, 13.95% and 11.63%, respectively; Table 1).

Distribution of species richness showed an important concentration of species for the lowlands bordering the Magdalena River, on the western flank of the eastern range of the Andes (Fig. 2). The mean value of representation within PAs for all species was lower than the global representation goals (17%; Aichi targets; Gannon et al. 2019) and, in general, is considered under most national goals (Mean ± SD = 10.69 ± 4.99%; Fig. 3).

Table 1. Mammals subject to use identified for the Department of Cundinamarca, Colombia, including the type of use reported.

Order	Species	Common name	IUCN Cat.	Use reported			
				Food	Pet/ Traffic	Control	Subproducts
Artiodactyla	*Mazama rufina*	Dwarf red brocket	VU	X		X	X
	Pecari tajacu	Collared peccary	LC	X	X	X	X
Carnivora	*Cerdocyon thous*	Crab-eating fox	LC				X
	Eira barbara	Tayra	LC			X	X
	Herpailurus yagouaroundi	Yaguarundi	LC		X	X	
	Leopardus pardalis	Ocelot	LC		X		X
	Leopardus tigrinus	Oncilla	VU		X		X
	Leopardus wiedii	Margay	NT		X		X
	Lontra longicaudis	Neotropical otter	NT			X	X
	Mustela frenata	Long-tailed weasel	LC				X
	Nasua nasua	South American coati	LC	X	X	X	X
	Nasuella olivacea	Western mountain coati	NT	X	X		X
	Panthera onca	Jaguar	NT	X	X	X	X
	Potos flavus	Kinkajou	LC	X	X		X
	Procyon cancrivorus	Crab-eating raccoon	LC		X		X
	Puma concolor	Puma	LC		X	X	X
	Tremarctos ornatus	Spectacled bear	VU	X	X		X
	Urocyon cinereoargenteus	Grey fox	LC		X	X	X
Chiroptera	*Desmodus rotundus*	Vampire bat	LC			X	X
	Myotis nigricans		LC			X	
Cingulata	*Dasypus novemcinctus*	Nine-banded armadillo	LC	X	X		X
	Cabassous centralis	Naked-tailed armadillo	LC	X			X
Didelphimorphia	*Caluromys lanatus*	Brown-eared woolly opossum	LC				X
	Chironectes minimus	Water opossum	LC				X
	Didelphis marsupialis	Common opossum	LC	X		X	X
Pilosa	*Bradypus variegatus*	Three-toed sloth	LC	X	X		X
	Choloepus hoffmanni	Hoffmann´s two-toed sloth	LC	X	X		
	Myrmecophaga tridactyla	Giant anteater	VU	X	X	X	X
	Tamandua mexicana	Northern tamandua	LC	X	X		
	Tamandua tetradactyla	Southern tamandua	LC			X	X
Primates	*Alouatta seniculus*	Colombian red howler monkey	LC	X			X
	Aotus griseimembra	Grey-handed night monkey	VU	X	X		
	Ateles belzebuth	White-bellied spider monkey	EN	X	X		
	Lagothrix lagotricha	Common woolly monkey	VU	X	X	X	X
	Saguinus leucopus	Silvery-brown tamarin	EN		X		
	Saimiri sciureus	Guianan squirrel monkey	LC		X		
	Sapajus apella	Black-capped capuchin	LC	X	X		
Rodentia	*Cavia aperea*	Brazilian guinea pig	LC	X			
	Cuniculus paca	Agouti	LC	X	X		
	Cuniculus taczanowskii	Mountain paca	NT	X		X	
	Dasyprocta fuliginosa	Black agouti	LC	X			
	Dasyprocta punctata	Central American agouti	LC	X			
	Hydrochoerus hydrochaeris	Capybara	LC	X	X	X	X

More than half of the species presented in the analysis have a representation value lower than 10%, with the rest of their ranges being outside PAs (Fig. 3). Orders with the lowest mean representation value are the order Pilosa and Didelphimorphia (Mean ± SD = 7.21 ± 4.44% and 7.626 ± 1.37%, respectively), while Carnivora and Artiodactyla showed

the highest mean representativity (Mean ± SD = 13.61 ± 4.70% and 13.63 ± 9.92%, respectively). The order Pilosa contained the species with the lowest PA representation in the whole study, the anteater (*Tamandua tetradactyla*), a species that, besides being under-represented, has a very small distribution area in the jurisdiction of the Department.

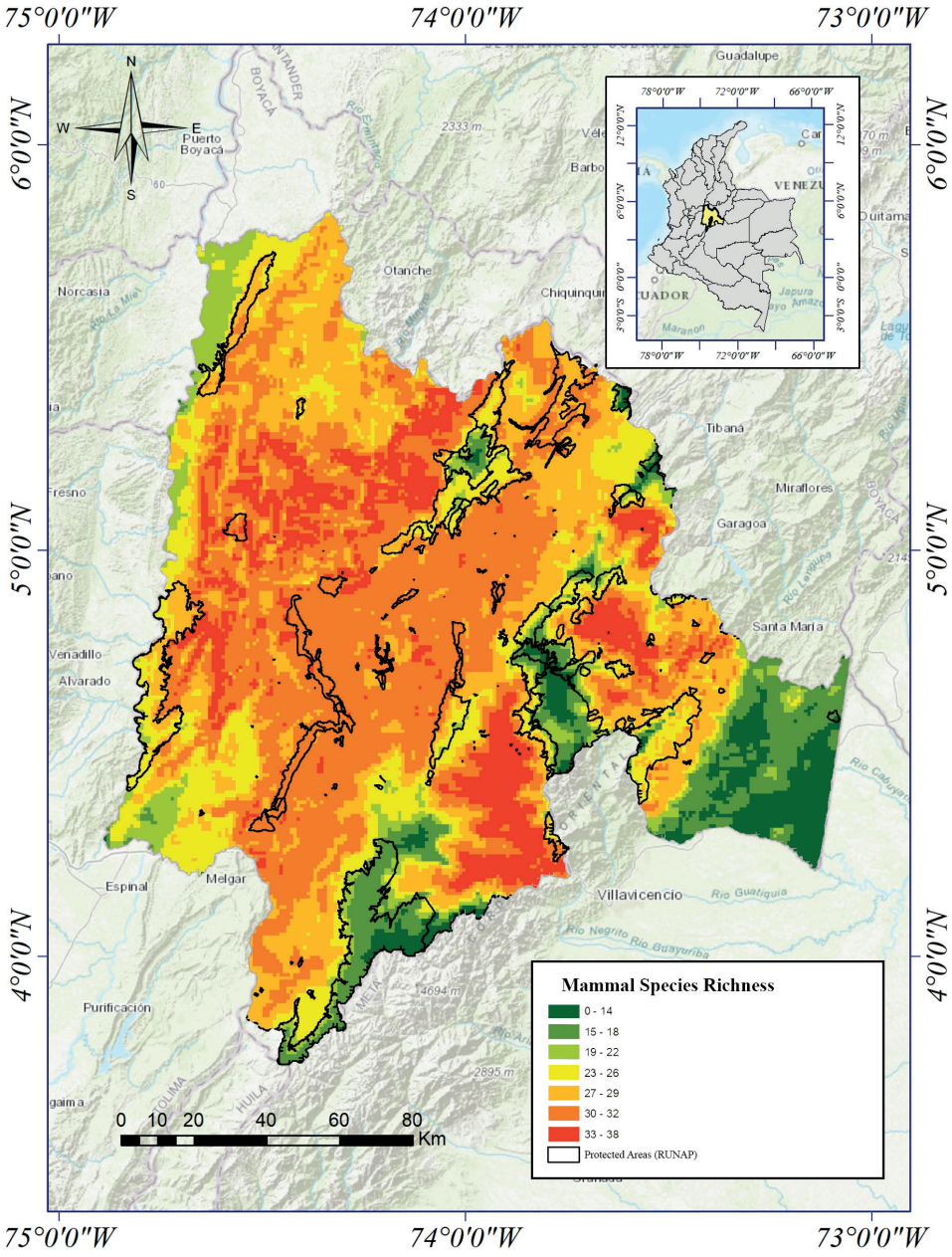

Figure 2. Mammal species richness. Potential distribution of species richness of mammals subject to use in the Department of Cundinamarca, Colombia, with a resolution of 1 km.

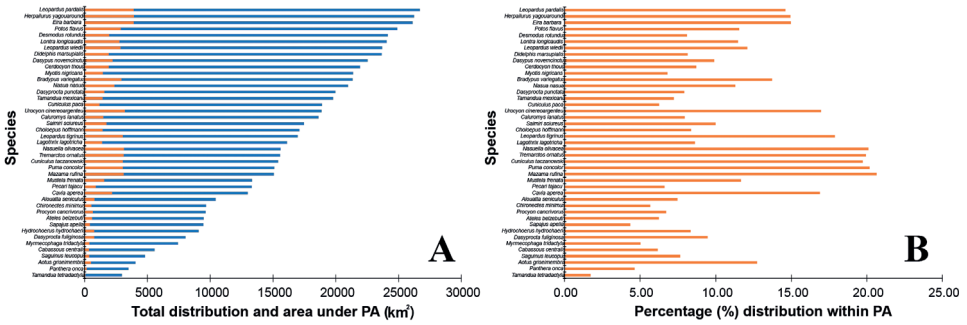

Figure 3. Distribution and representation of mammals in PAs **A** Total distribution and area under Protected Areas values and **B** representation (%) within PAs for all mammal species subject to use in the Department of Cundinamarca, Colombia.

In terms of species under any risk category, mean representation was significantly lower than for the non-threatened species (69.91 vs. 46.19%; W = 105.00, p = 0.026). Of these species, the Jaguar (*Panthera onca*) has the lowest representation for the Department (4.64%). The mean value (± SD) of the Human Spatial Footprint Index (HSFI) for the distribution areas of all species was 57.08 (± 2.74). This is a medium value, but it is very close to values considered high according to the HSFI, which are those over 60. We found very few species with areas with HSFI values lower than 40, which means that there are no areas that could be considered with low footprint values (Fig. 4).

Although for most of the Department of Cundinamarca, HSFI values are high or medium, within the PAs, the values are lower, which is evident in PAs such as

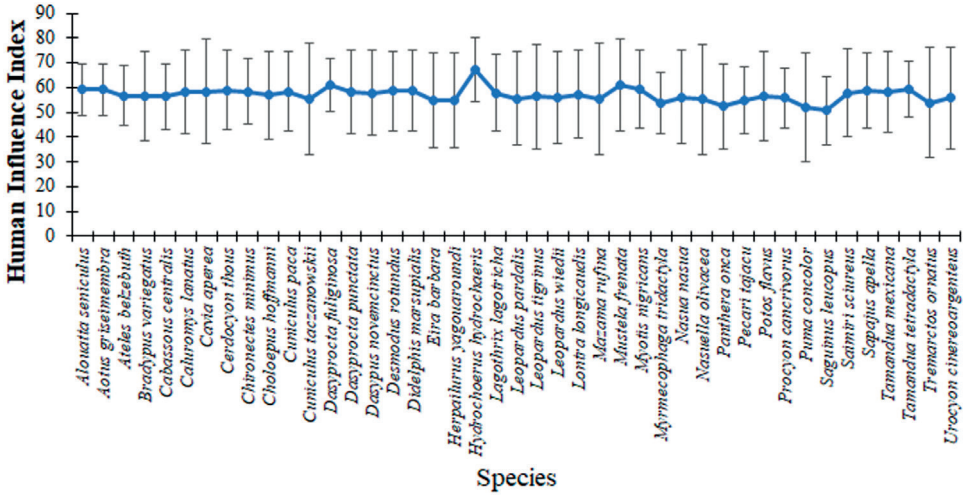

Figure 4. Mean Human Spatial Footprint Index within mammal ranges. Distribution and mean value of the Human Spatial Footprint Index (HSFI) for mammal species subject to use within the Department of Cundinamarca, Colombia.

Chingaza National Natural Park, the Cuchillas Negra and Guanaque and Cuchilla San Cayetano Integrated Management Regional Districts and the multiple protective forest reserves in the areas near Chingaza. In order to demonstrate whether PAs are ensuring habitat quality within the distribution of mammal species subject to use, we compared mean values of the HSFI of the distribution of the species in unprotected areas of the Cundinamarca Department and the mean value within all PAs. We found statistically significant differences ($T = 11.74$, $p < 0.01$) where almost all species have higher HSFI values throughout the Department than in PAs. On average, there is a difference of 10.72 points between the average HSFI of the Department and that of the PAs (Mean ± SD = 10.73 ± 5.98%). This shows that the state of the species' habitats is less impacted by human activities within PAs and that the conservation areas of most species depend, to a large extent, on them (Fig. 5).

Finally, when including the type of use and considering the differential pressure that different types of uses represent for each species, we found slight differences between representation and the level of human intervention on species ranges both in the whole Department and only inside PAs (Fig. 6). When comparing between species with and without use for each type, we only found significant differences for species subject to traffic/use as pets in terms of HSFI in the Department ($H = 3.95$, $p = 0.046$) and within PAs ($H = 5.93$, $p = 0.014$), with those species showing lower levels of intervention (Table 2). No significant differences were found for the rest of uses and for the three variables (Table 2). We found no differences between the types of uses for the three variables for those species subject to use (Table 2).

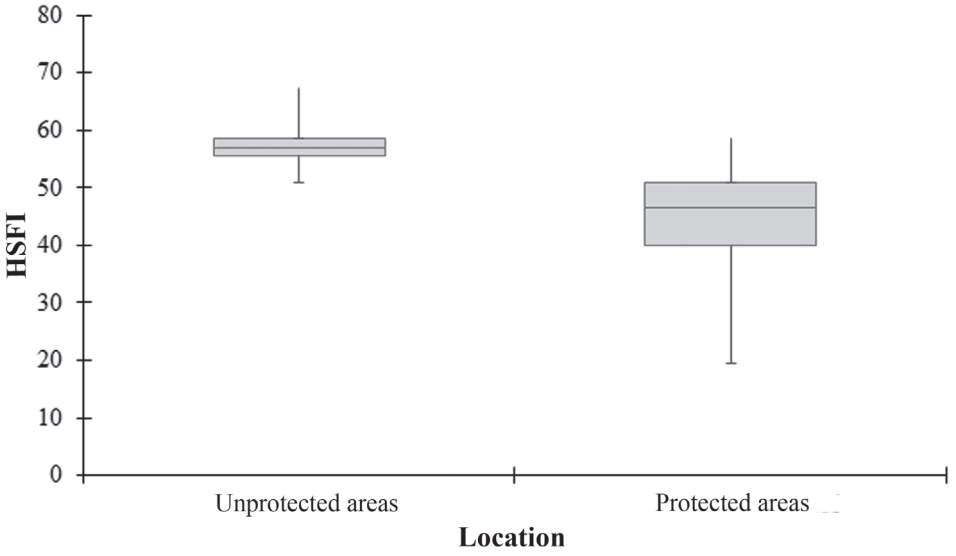

Figure 5. Human Spatial Footprint Index in and out of PAs. Comparison of the mean values of the Human Spatial Footprint Index of the distribution areas of the mammals subject to use inside and outside the PAs of the Department of Cundinamarca, Colombia.

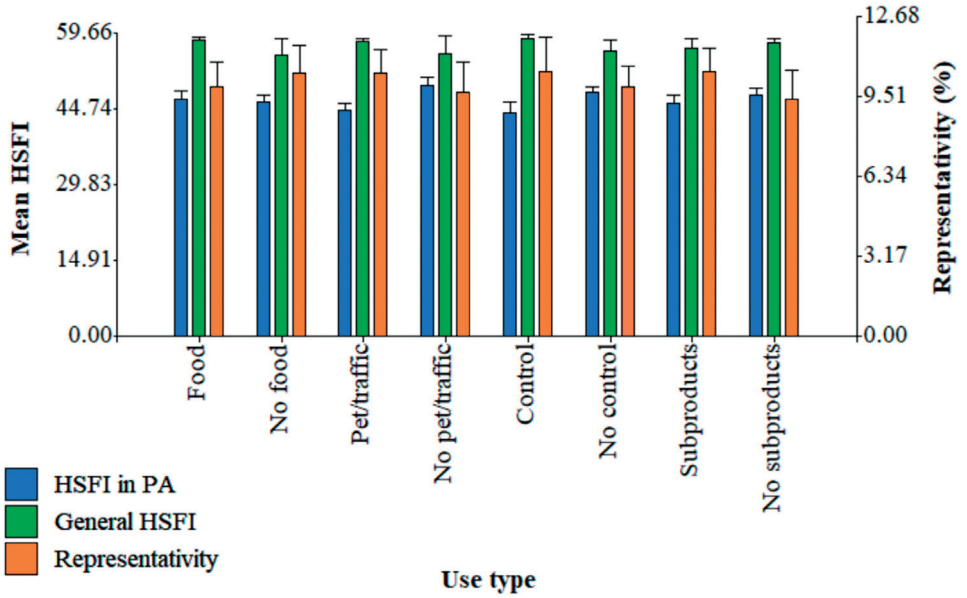

Figure 6. Human Spatial Footprint Index and representation for different species uses. Overall Human Spatial Footprint Index in the Department and within PAs compared with species subject to different types of uses in the Department of Cundinamarca, Colombia. Percentage of representation is on the secondary Y-axis.

Table 2. Comparison between mammal and use and no-use between different use types and for Human Spatial Footprint Index overall for the Department and within protected areas and for representation for the Department of Cundinamarca, Colombia. * Indicates significant differences.

Comparison	Type of use	Variable	H	p
Use / No use	Consumption	Representation	0.555	0.460
		HSFI Department	0.097	0.767
		HSFI in PA	0.059	0.824
	Control	Representation	0.099	0.765
		HSFI Department	1.733	0.188
		HSFI in PA	0.002	0.960
	Pet/traffic	Representation	0.200	0.654
		HSFI Department	3.959	0.040*
		HSFI in PA	5.936	0.014*
	Subproducts	Representation	0.656	0.421
		HSFI Department	0.002	0.9587
		HSFI in PA	0.153	0.697
Between uses	Consumption vs. Control vs. Pet/traffic vs. Subproducts	Representation	0.490	0.921
		HSFI in PA	1.753	0.624
		HSFI Department	1.073	0.783

Discussion

With its wide elevation gradient and mountainous topography, Cundinamarca harbours a wide variety of ecosystems and biodiversity (Conservación Internacional Colombia and Corporación Autónoma Regional de Cundinamarca 2011), even

considered as part of one of the global hotspots: the Tropical Andes (Myers et al. 2000). However, the continuous historic and ongoing land-cover change (Etter and van Wyngaarden 2000; Etter et al. 2006; Correa Ayram et al. 2020), the high impact of human activities and the effects related to global change drivers have affected and will keep affecting biodiversity, in general, and mammals, in particular (Schipper et al. 2008; Correa Ayram et al. 2018; Castillo et al. 2020). Human influence across species ranges is usually conceived as one of the best proxies of species threats and risk, especially for species with a close relationship with humans or those directly affected by their use (Sanderson et al. 2002; Woolmer et al. 2008). Here, we presented one of the first systematic approaches to the effects of human influence on an ecological and culturally important group of species, as a basis for appropriate decision-making and for providing information for conservation.

Most of the species distribution and the correspondent richness values respond to the heterogeneity of the transition zone between the Andes and the lowlands of the inter-Andean valleys of the Magdalena River and the Llanos, both located in the same area of the transformation front and the area where the highest HSFI values are located (Correa Ayram et al. 2020). This is evident in the Department's eastern and western zones in the lower fringe of the mountain range, except for the areas within some PAs, like Chingaza PNN and the various protection figures that surround it (Fig. 3).

More than half of the species presented in the analysis have a representation value lower than 10%, with most of their ranges located outside PAs; therefore, depending to a large extent on actions and management that takes place in private lands, a situation that has been seen before in mammal species, such as jaguar and puma (de la Torre et al. 2017; Zárrate-Charry et al. 2018). Furthermore, no order showed representation values above 15%, which is lower than the global representation targets for elements, such as biomes related to the Aichi targets (Woodley et al. 2012; Bacon et al. 2019). These values are well below other required areas for some groups, such as carnivores, where more ambitious targets such as 30% representation are proposed to ensure their ecological needs (Di Minin et al. 2016). The species with the lowest representation for the Department of Cundinamarca are the southern tamandua (*Tamandua tetradactyla*), the black-capped capuchin (*Sapajus apella*) and the jaguar (*Panthera onca*). Reasons associated with such low representation are likely explained by the restricted distribution of these species in the Department (Alzate-Gaviria et al. 2016; Payán et al. 2016; Olaya-Rodríguez et al. 2020), mostly restricted to the lowland areas, where the distribution of PAs is scarce, as the Department's system of PAs is predominantly montane, located in areas > 2,000 m above sea level (RUNAP 2019). This is especially worrisome for species, such as the black-capped capuchin and the jaguar, since they depend on conserved landscapes or extensive areas with remnant natural habitats and, in both cases, their representation in PAs was below 5%. Remarkably, half of the nine threatened species showed representation values lower than 10%, which represents a significant threat to their survival, especially considering most of their threats are related to habitat loss and degradation which are very high in most of the Department, especially outside

PAs. Overall, the low representation within PAs for most species and the high level of transformation and human impact outside them, indicate the Department retains very unsuitable conditions for most species subject to use and, therefore, the probability of survival for the long term of most species in the Department is likely very low.

Natural cover remnants with good quality and with considerable size in the Department are scarce and poorly represented in PAs, as in most of the Andean Region (Armenteras et al. 2003) and the magnitude of human influence in the entire region is amongst the highest in the country, along with the Caribbean Region (Correa Ayram et al. 2020). Furthermore, given the generalised scarcity of information on the ecology of most species, including their habitat requirements, it is necessary to use other approaches to assess the current status or vulnerability of species through their habitats to increasing pressures related to anthropogenic variables (Collen et al. 2008; Leidig and Teeuw 2015). That is why we evaluated the influence of humans across the range of all mammal species subject to use by means of the Human Spatial Footprint Index (HSFI) and compared the status of species distribution in the Department and inside PAs, in order to evidence whether being inside a PA makes the magnitude of the impact of human pressures lower.

All species subject to use in the Department have a mean value of human influence over their entire distribution higher than 50, which is of particular concern since this value is well over the threshold of high intervention according to the Index (Correa Ayram et al. 2017; Correa Ayram et al. 2018). The two species that showed the mean higher values (> 60) were the long-tailed weasel (*Mustela frenata*) and the capybara (*Hydrochoerus hydrochaeris*). For the long-tailed weasel, for instance, the distribution includes a large part of the central zone of Cundinamarca, including Bogotá, the capital city of Colombia and its surrounding areas, some of the most populated and industrialised regions in the country (Gobernación de Cundinamarca 2020; González-Maya et al. 2021a). In general, the results indicate that human activities overall have fewer impacts over species' habitats within PAs and that the conservation areas for most species depend largely on them (Fig. 4). We found very few species with areas with low (< 30) index values, which means that, for the study area, there are no areas with a low human footprint (Fig. 5). This value is critical for supporting how to design conservation strategies since the existing PAs are not so extensive and have not been designed to ensure the representation of mammal species or, in fact, any other group; this translates in that conservation efforts depend mainly on private and productive areas, but in the Department, these are particularly adverse since they have very high values of human impact. Given that our focal species are already under severe pressure from their close relationship with humans, quality habitat then becomes an even more important aspect to be considered given the synergistic effect of multiple stressors over their populations. Our results should be considered when defining conservation actions or prioritisation processes for restoration or management zones in the Department to ensure at least the best remaining areas within an otherwise significantly transformed landscape.

Conclusions

Human activities have irreparably affected species habitats and the functioning of ecosystems globally, a dynamic that is becoming increasingly pronounced with catastrophic effects for biodiversity (Ceballos et al. 2015; González-Maya et al. 2017). This reality is no different for Cundinamarca, a Department that, like several in the Andean zone, has suffered the greatest impacts related to human activities in the country (Correa Ayram et al. 2018). Based on this reality, the needs and conservation planning for maintaining species is a great challenge, especially when managing species subject to use that possess characteristics that are part of the culture, use and tradition of human communities (Andrade Pérez and Corzo Mora 2011). Protected Areas have historically functioned as the cornerstone of conservation strategies (Stolton and Dudley 2010), being areas where species can exist and survive and, at the same time, function as a source for colonising recovered surrounding areas (Guerra et al. 2019). Currently, in Colombia, the role and importance of PAs are becoming increasingly evident, but the challenges for their maintenance are also increasing and it is urgent to ensure effective management of these areas and the landscapes that contain them (Ospina Moreno et al. 2020).

Our results make evident that the representation of mammal species within the existing PAs is below the globally defined thresholds and well below the requirements that most species may have (Di Minin et al. 2016; Wilson 2016). Likewise, the conservation status of habitats both inside and outside PAs is low and they have been subjected to various human impacts that make the HSFI values very high for the entire Department. Although the impact values associated with the HSFI are lower within PAs, they are not of optimal quality and efforts are required to redirect this trend and achieve landscape conservation and functionality. Currently, the development and planning of a new PA policy (CONPES 450) provide elements to improve the effectiveness and conservation of PAs and, at the same time, ensures that the surrounding landscapes are managed in a way that ensures the functionality of ecological processes and habitat and species connectivity. In the specific case of the mammals of Cundinamarca, this is critical because, to a large extent, their distribution areas and their management are located on private properties outside of conservation zones and only by strengthening management measures, land use plans and defining new conservation strategies can the maintenance of their habitats be warranted.

Acknowledgements

This work is part of the Project “Distribución histórica, actual y futura de mamíferos y sus relaciones e importancia sociocultural en el departamento de Cundinamarca: herramientas de planificación de conservación” funded by the Convocatoria 829-2018 Proyectos de I+D para el desarrollo tecnológico base biológica-Cundinamarca from the Ministerio de Ciencia, Tecnología e Innovación of Colombia, executed by Corpo-

ración Universitaria Minuto de Dios – UNIMINUTO and Proyecto de Conservación de Aguas y Tierras - ProCAT Colombia. We thank the supervisors at MINCIENCIAS and UNIMINUTO and to ProCAT Colombia staff for their support for the development of this project. Thanks to J. Schipper for improving the language. Additionally, we thank the editor and three anonymous reviewers for their comments that significantly improved the manuscript.

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Nature Conservation 48: 83–93 (2022)
doi: 10.3897/natureconservation.48.79902
https://natureconservation.pensoft.net

RESEARCH ARTICLE

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A rare shrub species as flagship for conserving desert steppe in arid Inner Mongolia

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Academic editor: M. Nazre | Received 29 December 2021 | Accepted 12 April 2022 | Published 20 April 2022

http://zoobank.org/3BD079FB-FEB6-4943-9E03-043F13D38B8D

Citation: Yang H, Xu C, Chu J, Chen J, Gan H, Zhou Z (2022) A rare shrub species as flagship for conserving desert steppe in arid Inner Mongolia. Nature Conservation 48: 83–93. https://doi.org/10.3897/natureconservation.48.79902

Abstract

The rare species *Amygdalus pedunculata* Pall. (Rosaceae) in arid northern China is endangered to the point of extinction. Determined to save it, the local government of Inner Mongolia Autonomous Region encouraged the herdsmen to limit grazing activities. Here, we are testing if this species could be considered as a conspicuous flagship for restoring and conserving wind-sensitive arid lands as desert steppe in northern China. We examined statistically the growing states and environmental roles of *A. pedunculata* populations under the comparative conditions of free and limited grazing in winter since the year 2001. This species was observed to play a critical role in preventing wind erosion and stabilising the lands, as was indicated by the formation of micro-dunes under the shrubs. This role can be attributed mainly to the crown diameters or cover from the shrubs. Under the grazing limitation condition, accompanying species and plants around the shrubs increased significantly. Regardless of free or limited grazing conditions, the shrubs were not observed to inhibit the occurrence or growth of other plants. The grazing limitation over a period of 20 years has caused the effective revival of the rare *A. pedunculata* species, with statistically larger and taller *A. pedunculata* individuals than under the free grazing condition, as well as a slightly higher population density and total crown cover. The grazing limitation policy for saving *A. pedunculata* is believed to be effective and the rare *A. pedunculata* shrub is a conspicuous flagship for helping to conserve wind-sensitive desert steppe in terms of ecosystem integrity and authenticity.

* These authors contributed equally to this work.

Keywords
Amygdalus pedunculata Pall. (柄扁桃), desert steppe, ecosystem conservation, flagship species, sand stabilisation, wind erosion

Introduction

Aeolian erosion is a key factor against arid lands, such as desert steppe in northern China (Wang 2014; Wijitkosum 2021; Wu et al. 2021). Plants in desert steppe are normally reduced in size and density; once these plants are damaged, winds can cause intense soil erosion (Meng et al. 2018). Inhabitants basically live by grazing sheep, cattle, horses and camels from time immemorial (Du et al. 2019). Vegetation degradation has occurred in these areas because of overgrazing in the past; subsequent wind erosion then led to land degradation and severe dust storms (Liu et al. 2009; Liu et al. 2017; Du et al. 2019). Dusts spread extensively to pollute the atmosphere on a vast scale (Tian et al. 2021). On the other aspect, a number of native species are becoming extinct with the degradation (Liu et al. 2015; Yang et al. 2017). These emerging crises are detrimental not only to the sustainable livelihood of the local population, but also to public interests in national and even international levels for resource availability (Gholizadeh et al. 2021; Luo et al. 2022). Any rare species with unique genetic resources may potentially benefit all human beings, other than just local people within a limited area (Cardinale et al. 2012; O'Brien et al. 2021).

Amygdalus pedunculata Pall. [柄扁桃, used to be named *Prunus pedunculata* Pall.] is a rare shrub species naturally endemic to the desert steppe in northern China; it produces beautiful flowers in early spring and delicious oily seeds in autumn (Chu et al. 2015, 2017; Gao et al. 2016). However, this species has suffered from overgrazing with undue livestock herbivory and trampling for many years and most of the local populations have disappeared, except at few sites. Considering the rarity and endangered status of this species, the local government of Inner Mongolia Autonomous Region, in 1989, introduced a policy to save it (Government-Bulletin 1989). This species may be a conspicuous flagship for restoring and conserving the desert steppe as land- and biotic-resources (Herrera-Sanchez et al. 2020; Lachowska-Cierlik et al. 2020; Shen et al. 2020). From 2001, some herdsmen whose pastures contain remnant *A. pedunculata* populations are officially encouraged to limit their grazing activities, thereby ensuring that these remnant populations will renew or revive. According to a special policy, the government will provide the participating herdsmen with necessary financial compensation if they agree to cease grazing in lands where *A. pedunculata* grows, except in winter when forage is extremely short.

Even so, no herdsmen were willing to have all their lands used as this policy required because they must live by graze. Thus, only a part of remnant *A. pedunculata* populations are conserved as regulated by the policy, and other parts are still suffering from free grazing. After nearly 20 years, we herein investigated the growing status of *A. pedunculata* populations under the comparative conditions of

free and limited grazing. We tested whether the current conservation policy is effective for saving and restoring this rare species and whether this species is a useful conspicuous flagship for restoring and maintaining the desert steppe as a whole of land- and biotic-resources.

Methods

Sampling and measuring

This study was conducted at a natural distribution site of *A. pedunculata* (42°34'33.93"N, 112°30'58.74"E, Fig. 1a). Elevations range from 900 m to 1260 m. The soil is brown calcic, covered with thin sand. The underground water is more than 20 m below the surface. The mean annual precipitation is about 190 mm and the mean annual evaporation is about 2400 mm. The rainy season is June, July and August, accounting for nearly 70% of the annual precipitation (Shen and Wei 2008). Strong winds (> 17.2 m/s) occur in ca. 60 d each year, and sand storms occur in more than 12 days each year. Natural vegetation is sparse desert steppe (Fig. 1b). Overgrazing in this area was very common before 2000, thus causing vegetation and land degradation (Zhang et al. 2021a; Zhang et al. 2021b). Since 2001, some involved herdsmen have been financed to graze only in winter in some pastures where *A. pedunculata* occurs; even so, many *A. pedunculata* populations are out of the protection.

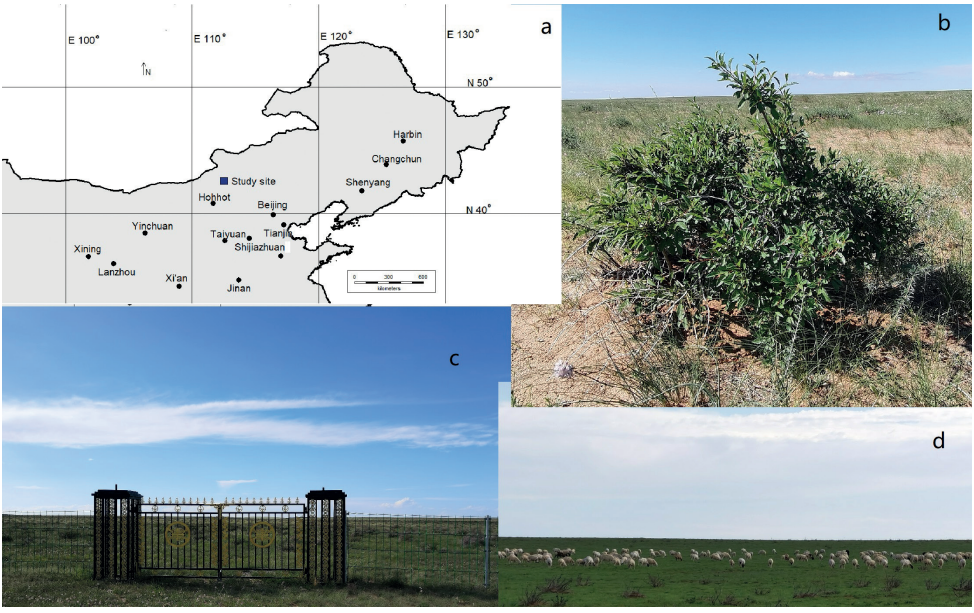

Figure 1. Study site (**a**) and *A. pedunculata* (**b**) in arid desert steppe in northern China. Two scenes were compared: limited grazing (**c**) to only winter and free grazing (**d**) throughout the year. Taken by Hongxiao Yang.

We chose two typical scenes for the study: one is fenced since 2001 and only for winter grazing (Fig. 1c) and the other is open and used for free grazing throughout the year (Fig. 1d). In each scenario, we randomly chose 27 individuals of *A. pedunculata* shrubs. For each shrub, we measured the relative height (cm) from the top to the foot of the micro-dune under the shrub and the crown diameter (cm), which is the mean of the longest and shortest diameters of each shrub crown. We counted the number of branches that emerged from the ground. We also investigated accompanying plants around the shrub with three 1 m parallels at three distance (position) levels (a, just inside the crown projection; b, at the edge of the crown projection; c, 1 m away from the crown projection); this investigation was repeated four times in different directions. In these sampling lines, we counted the number of occurring species and the total number of growing plants. We then established 15 random 10 × 10 m plots in each scene, where we counted all *A. pedunculata* shrubs and measured their heights (cm) and crow diameters (cm) as above.

Data analysis

We calculated Pearson coefficients of the dune height with height, branch number and mean crown diameter of the corresponding shrub to determine which feature is critical for the effect of sand stabilisation. We also established a linear model for demonstrating the relationship. Using two-way ANOVA, we examined the effects of the two grazing modes and the distances away from the shrubs on the numbers of occurring species and plants. We conducted a T-test for comparing the growth states of *A. pedunculata* populations in the two grazing modes by using four indices, namely, population density (shrub number in a 10 × 10 m plot), total crown area (total *A. pedunculata* crown area in a 10 × 10 m plot) and maximum height and crown diameter of *A. pedunculata* individuals in each plot. The total *A. pedunculata* crown area was set as the total crown area of all *A. pedunculata* individuals in a plot and crown area (cover) of an *A. pedunculata* individual was calculated with the formula: $\pi \times (\text{diameter}/2)^2$. All these analyses were completed in R4.1.1 software (www.r-project.org).

Results

The sand stabilisation effect of the shrubs can be attributed mainly to the crown diameters of the shrubs and can be fitted with a linear model (Table 1, Fig. 2). Branch number also contributed to this effect under the condition of free grazing, but had less contribution under the condition of limited grazing.

Table 1. Correlation of dune height with shrub features under the two grazing conditions.

Grazing mode	Crown diameter	Branch number	Shrub height
Free grazing	0.614***	0.508**	0.307
Limited grazing	0.549**	0.280	0.086

*, $r_{(1, 25)0.05} = 0.381$; **, $r_{(1, 25)0.01} = 0.487$; **, $r_{(1, 25)0.001} = 0.597$

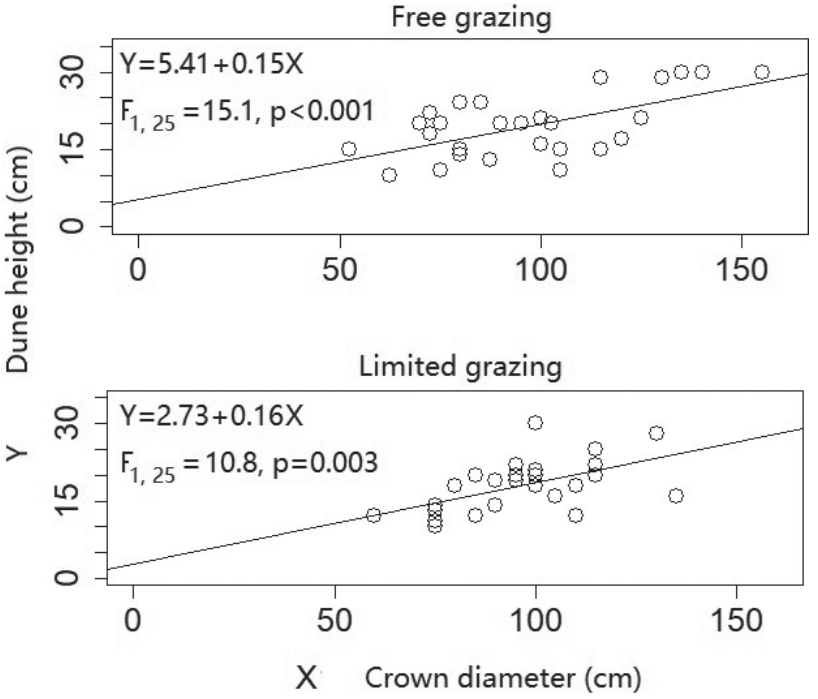

Figure 2. Relationship of height of shrub-caused micro-dune with crown diameter of the pertaining *A. pedunculata* shrub. The upper part is under the condition of free grazing and the lower part is under the condition of limited grazing.

Species and plant numbers were affected by the grazing modes, instead of the shrubs (Fig.s 3, 4). The two-way ANOVA indicated that both plant and species numbers under or near the shrubs were sensitive to the grazing modes, other than the shrubs and their interactive effect with the grazing modes (Plants: grazing modes, $F_{(1, 114)} = 61.73$, $p < 0.001$; distances to the shrubs, $F_{(2, 114)} = 7.1$, $p = 0.47$; interactive effect of the grazing and the distances, $F_{(2,114)} = 4.1$, $p = 0.65$. Species: grazing modes, $F_{(1, 114)} = 78.41$, $p < 0.001$; distances to the shrubs, $F_{(2, 114)} = 3.02$, $p = 0.29$; interactive effect of the grazing and the distances, $F_{(2,114)} = 5.82$, $p = 0.10$). Averages of plant numbers in the sampling unit were < 5 in the free grazing condition, and > 5 in the limited grazing condition (Fig. 3); averages of species numbers were ca. 3 in the free grazing condition, and evidently > 3 in the limited grazing condition (Fig. 4).

The grazing limitation evoked positive changes in *A. pedunculata* populations (Fig. 5). Under the condition of limited grazing, large-crowned *A. pedunculata* shrubs were observed to be more common than those under the condition of free grazing (T-test, $t = 2.62$, $df = 24.717$, $p = 0.015$). Similarly, tall *A. pedunculata* shrubs were significantly more common than those under the condition of free grazing (T-test, $t = 3.40$, $df = 26.11$, $p = 0.002$). The population density and total crown area also increased but not to such a significant extent (Density: T-test, $t = 1.20$, $df = 25.44$, $p = 0.241$. Total crown area: T-test, $t = 1.55$, $df = 25.857$, $p = 0.135$).

Figure 3. Numbers of plants occurring in different distances to the nearest *A. pedunculata* shrub. The upper part is under the condition of free grazing and the lower part is under the condition of limited grazing.

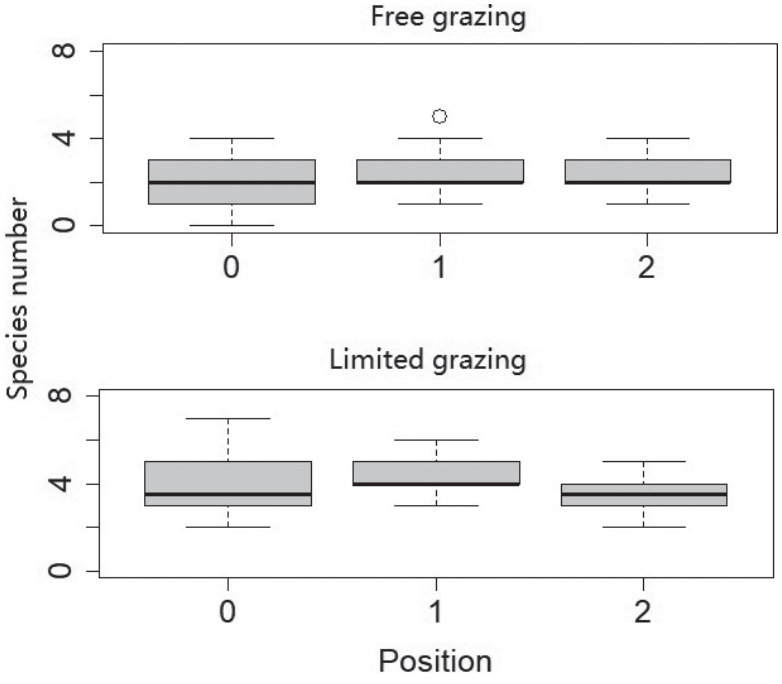

Figure 4. Numbers of species occurring in different distances to the nearest *A. pedunculata* shrub. The upper part is under the condition of free grazing and the lower part is under the condition of limited grazing.

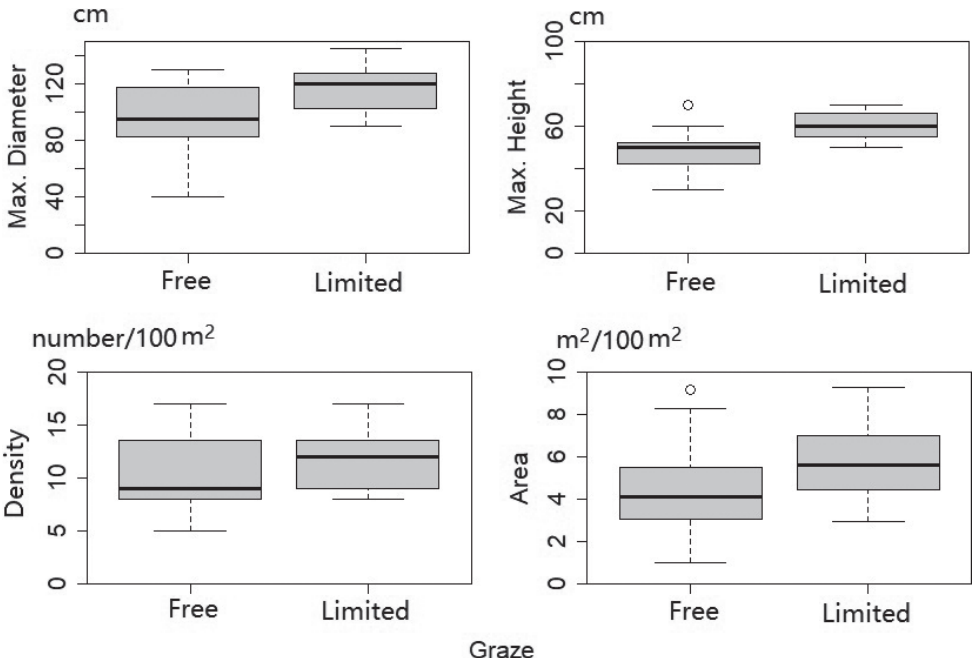

Figure 5. Changes in *A. pedunculata* populations in response to the two grazing modes: the upper part, free grazing; the lower part, limited grazing. The compared indices include maximum crown diameter and height of *A. pedunculata* shrubs in a sampled 10 × 10 m plot, as well as density and total crown area of all *A. pedunculata* shrubs in each plot.

Discussion

Amygdalus pedunculata populations and accompanying plants grew much better under the condition of limited grazing than under the condition of free grazing, presumably because of weakened herbivory and animal trampling. One significant change is that large and tall *A. pedunculata* individuals became more common under the condition of limited grazing. Another significant change is that the number of accompanying plants and species was definitely higher than that under the condition of free grazing. Other changes, such as population density and total crown area of *A. pedunculata* populations, were not so significant, but certainly not lower than those under the condition of free grazing. This evidence demonstrates that the policy of grazing limitation is effective to facilitate the thriving of *A. pedunculata* and accompanying plants. In contrast, free grazing with heavy herbivory and trampling prevented *A. pedunculata* and accompanying plants from renewing and thriving.

Most arid deserts are sensitive to wind erosion (Chi et al. 2019; Fenta et al. 2020). Once they are deprived of vegetation cover for long-term overgrazing, they will seriously suffer from erosive winds or storms so as to output flying dust and rolling sand (Zhang et al. 2021c). Under the condition of free grazing, we found that sand was deposited under *A. pedunculata* crowns, piling as micro-dunes, and accompanying plants were too scarce to protect the land from potential wind erosion. As thus, we state that it is *A. pedunculata* shrubs, instead of accompanying plants, that mainly stop sand drift and protect the lands

(Zhan et al. 2017). This finding may be due to the fact that soft herbs or grasses, as main forage, are often the prior victim in grazing events, whereas woody shrubs as *A. pedunculata*, except their seedlings, can avoid this relatively better for lessened herbivory and trampling (Fan et al. 2016; Zhang et al. 2019; Sun et al. 2020). In addition, the crown diameter of the *A. pedunculata* shrub was observed to be closely correlated with the height of the under-crown fine sand dune, which shows the effect of sand stabilisation.

The rare *A. pedunculata* is urgently endangered. However, involved herdsmen must live by grazing stocks, and they especially concern forage production, i.e., grass growth, other than *A. pedunculata*. This study shows that *A. pedunculata* shrubs do not inhibit accompanying plants for growing, no matter whether under the condition of free grazing or under the condition of limited grazing. For this reason, herdsmen or land owners can be assured that *A. pedunculata* is almost harmless to forage production, and that, after *A. pedunculata* populations are re-established, the lands can be protected better than without the shrubs from potential wind erosion and land degradation. What is more, they can harvest some *A. pedunculata* seeds to be sold in the market as raw materials for horticultural breeding or oil production (Li et al. 2010; Chu et al. 2015; Wang et al. 2020). Thus, monetary values of this species can be realised partly in serving more people.

Conclusions

The rare species *A. pedunculata* is worth conserving, because it can protect lands from wind erosion, while does not inhibit the growth of accompanying plants. The current policy for saving *A. pedunculata* is effective. With the grazing limitation over a period of 20 years, *A. pedunculata* resources have been renewed effectively and the pertaining lands have been restored with more plants and species. The shrub species *A. pedunculata* can be viewed as a conspicuous flagship for comprehensively restoring and conserving the natural desert steppe in northern China in terms of ecosystem integrity and authenticity, that is, all ecosystem components including native plants, animals and their desired habitat can be conserved as a whole because of the in-situ conservation for *A. pedunculata*.

Acknowledgements

We gratefully acknowledge HE Dongxiao in Qingdao Agricultural University. Funding: This study is funded by Fundamental Research Funds of CAF (CAFYBB2020ZB001), Special Foundation for National Science and Technology Basic Research Program of China (2017FY100205) and Qingdao Agricultural University.

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Nature Conservation 48: 95–97 (2022)
doi: 10.3897/natureconservation.48.84287
https://natureconservation.pensoft.net

LETTER TO THE EDITOR

A peer-reviewed open-access journal
Nature Conservation
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Strict laws fail to deter illegal trade of China's largest and most endangered freshwater turtle

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Academic editor: Mark Auliya | Received 24 March 2022 | Accepted 24 March 2022 | Published 21 April 2022

http://zoobank.org/D3420B94-4558-4051-8CC9-B355C81C36A1

Citation: Hu S, Lee TM, Shi H (2022) Strict laws fail to deter illegal trade of China's largest and most endangered freshwater turtle. Nature Conservation 48: 95–97. https://doi.org/10.3897/natureconservation.48.84287

In May 2021, six people were arrested for the illegal wildlife trade (IWT) of one of the most endangered freshwater turtle species in China (Public Security News from Meizhou City Government 2021). An adult female *Pelochelys cantorii* (about 22.6kg and carapace length of 70cm) was poached in Meizhou, Guangdong province, and was sold for over 5,000 RMB (or about 770 USD). The turtle was advertised on Chinese social media, attracting huge public interest, which led to the arrest of the suspects by the Forestry Branch of the Public Security Bureau.

Unfortunately, the seized turtle, which was sent to S. Hu (first author) for expert identification, died soon after examination. According to Chinese national laws and the List of National Key Protected Animals in 2021, IWT offenders for this turtle species may be culpable for criminal charges, as well as estimated heavy fines of up to 1.5 million RMB (a value calculated based on the No. 5 Order of the Ministry of Agriculture and Rural Affairs of the People's Republic of China - Measures for valuation of aquatic wild animals and their products in 2019).

Shockingly, *P. cantorii* has been a Class I protected species (no trade allowed) in China for the last 32 years (Law of the People's Republic of China on the Protection of Wildlife in 1989), yet it was traded publicly on social media after being poached from a protected area. How did we fail in our efforts to prevent the illegal trade of this

threatened species despite it being listed as a protected species for three decades and China having one of the strictest penalties for IWT worldwide?

P. cantorii is one of the world's largest freshwater turtle species, weighing up to about 200kg and a carapace length of up to 130cm. Historically, *P. cantorii* was widely distributed throughout China but due to overexploitation for use in food and medicine, and native aquatic habitat loss, degradation and fragmentation (Lau and Shi 2000; Gu and Ma 2000), its population began to decline rapidly and was evaluated 'Endangered' in the IUCN Red List (IUCN 2000).

For instance, in the late 1990s, the Oujiang River Basin in Zhejiang Province recorded the largest population of about 80 individuals in Mainland China (Gu and Ma 2000). Another *P. cantorii* population at the Youxi River, Fujian Province, was observed to be threatened by freshwater habitat degradation due to pollution from industrial sewage and the building of hydro-electric dams (Ruan et al. 2001). As a result, the species was listed in Appendix II of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and was assessed as a 'Critically Endangered' species in the Chinese Species Red List (CITES 2003; Wang and Xie 2009). In the last 20 years, media reports regarding *P. cantorii* have brought news of illegal hunting and accidental bycatches (Kuang 2018; Jiang 2018).

Despite its high threat status, the fact that it is being illegally hunted, and its rare sighting in the wild, conservation efforts to protect *P. cantorii* have been underwhelming. And this is not to mention the critically low level of public awareness of its status (Hong et al. 2019) and the ineffective network of reserves in protecting Chelonian species (Gong et al. 2017; Gong et al. 2020). Although China has improved enforcement against IWT in the recent decade, this particular offense has highlighted the need to consider urgent conservation actions for *P. cantorii*.

We urge relevant Chinese authorities to reduce the impacts of the hydro-electric dams on freshwater habitats, as well as improve management of existing protected areas. A survey of villagers in the area of the river where there are *P. cantorii* revealed that more than three-quarters of the fishers believed that the dams reduce its population and the quantity of fish. At least 70% of the fishers also believed that the sewage from the factories along the river decrease the fish species richness and abundance, and that river pollution led to habitat destruction for *P. cantorii* (Hong 2020). We recommend that the authorities increase funding and resources in the protected areas, strengthen patrolling and enforcement, and prohibit any illegal fishing operations and habitat destruction activities. In some critical regions, advanced technological applications such as drone aerial photography and infrared camera monitoring could be introduced to deter and reduce poaching and illegal activities.

Authorities should certainly consider increasing the scale and effectiveness of public awareness about the illegality and penalties associated with IWT. When the public's legal and conservation awareness is low and weak, law-breakers, when driven by economic interests, would risk breaking the law to poach even highly protected animals in the reserves, and blatantly trade them on social media publicly. A questionnaire survey among villagers in the areas where the turtle is distributed shows that 63.6% of the coastal villagers do not know the species, and only 15.2% of the fishermen have seen the species, and most of these incidences happened more than a decade ago (Hong 2020).

Current priorities should include increasing the public's awareness regarding the legal consequences of actions, raising the public's awareness of conservation, so that they may be socially engaged, and consciously resist, and actively report relevant illegal acts. Unless China meets the urgent conservation needs of this iconic species, the extinction of this species will likely not be averted (Wang et al. 2021).

Aknowledgements

H.S. receives funding from the National Natural Science Foundation of China (32170532).

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Nature Conservation 48: 99–121 (2022)
doi: 10.3897/natureconservation.48.80988
https://natureconservation.pensoft.net

RESEARCH ARTICLE

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Harvest quotas, free markets and the sustainable trade in pythons

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Academic editor: Mark Auliya | Received 21 January 2022 | Accepted 22 April 2022 | Published 11 May 2022

http://zoobank.org/81F1318F-2BB5-4A9F-9447-6FBC842A5139

Citation: Nijman V (2022) Harvest quotas, free markets and the sustainable trade in pythons. Nature Conservation 48: 99–121. https://doi.org/10.3897/natureconservation.48.80988

Abstract

Assessing the sustainability of the harvest of animals can be done by obtaining data from processing facilities and establishing that vital attributes of the harvested animals (e.g., size, age structure, sex ratio) do not change over time. This model works if the traders operate in a free market without any regulations on what can be harvested, processed or exported, and when harvest methods and harvest areas do not change between assessment periods. Several studies assessed the harvest effects on blood pythons (*Python brongersmai*) in North Sumatra, Indonesia seemingly under a free market scenario, with some concluding that trade was sustainable and the others hinting at an overharvest. Indonesia has established harvest and export quotas and, internationally, trade in blood pythons is regulated through CITES, and the blood python trade clearly does not operate in a free market. Data suggest that the three (or four) slaughterhouses included in these studies processed ~27,000 blood pythons a year against a quota of 18,000. There is a risk that data from traders alone purporting to show that harvest is sustainable will lead to an increase of quotas or an abandonment of quotas altogether. There is no conclusive data to support that the harvest of blood pythons in North Sumatra is sustainable but there is sufficient evidence to suggest that a substantial part of this trade is illegal. Likewise, at a global level there are clear indications of misdeclared, underreported and illegal trade involving 10,000 s of blood pythons. While important biological information can be obtained from harvested animals, to assess whether harvest is sustainable there is no substitute for monitoring wild populations. After decades of international trade in blood pythons from Indonesia, during which at least half a million blood pythons were exported, it is all the more urgent that systematic monitoring of wild populations commences.

Keywords
CITES, conservation, Indonesia, *Python brongersmai*, reptiles, short-tailed python, sustainable management, trade regulations, wildlife trade

Introduction

Several studies have addressed the unsustainable exploitation of reptiles (Gibbons et al. 2000; Schlaepfer et al. 2005; Auliya et al. 2016; Marshall et al. 2020; Janssen 2021; Cox et al. 2022) and according to some the intentional harvest of them is one largest threat to the survival of many reptile species, especially crocodiles and turtles (Böhm et al. 2013; Cox et al. 2022). Reptiles represent one of the most species-rich vertebrate class in international trade (Nijman 2010; Scheffers et al. 2019). Species extinction processes that are causally linked to harvest and trade remain relatively undocumented. However, it is evident that those species that are long-lived with long generation times, those with low fecundity, those that are rare, or those that have a very restricted geographic range or small population size, are particularly vulnerable to over-collection (Reznick et al. 2002; Auliya et al. 2016). As noted by Challender et al. (2021) understanding the impact of trade-driven harvest on wild populations requires data on critical population parameters, including intertemporal harvest rates and their influence on density. Using trade data subjectively to determine that trade is detrimental as a shortcut for these in-depth analyses can be problematic (Challender et al. 2021).

In natural resource management the concept of sustainability is central as it would allow to maintain a long-term yield. From an economic perspective obtaining the largest harvest while maintaining the harvested population at a given size indefinitely (or at least long term) is preferable. This is known as the maximum sustainable yield (Tsikliras and Froese 2019); it remains a very useful concept as it provides an invaluable reference point as an ideal against which current practice can be compared (Sutherland 2001). In theory, with population growth following a sigmoid curve, the greatest growth rate occurs at intermediate population sizes. When harvest is sustainable it should not exceed the maximum sustainable yield under any carrying capacity, and it should not lead to a decrease in population size. But sustainability and maximum sustainable yields are not dependent on population size, and we can sustainably harvest at different population sizes. Fig. 1 shows this for three different population sizes – this can represent the trajectory of a single population, for instance water monitors (*Varanus salvator*) on the Thai-Malay Peninsula, that experiences a series of population declines, but at three different intervals (with large, intermediate and small population sizes) the harvest is sustainable. The sections between A and B, C and D and E and F can also represent three different populations, for instance tokay geckos (*Gekko gecko*) on the islands of Borneo (743,330 km^2), Sumatra (473,481 km^2) and Java (128,297 km^2), that have different population sizes with different likelihoods of extinction.

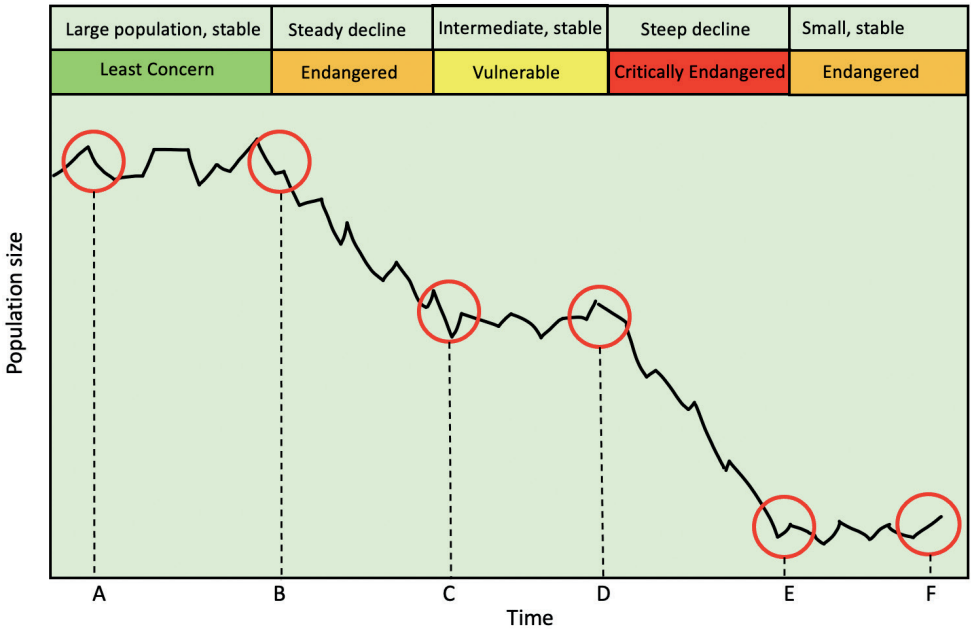

Figure 1. Conceptual framework of the relationship between population size, sustainable harvest and global conservation status. The harvest that took place between A and B, C and D, and E and F, could be considered sustainable, whereas it is unsustainable between B and C and D and E. The global threat assessment based on two of the IUCN threat level criteria (population size and declining populations) are not tightly linked to harvest sustainability (modified after Yamaguchi 2014).

As with all assessments over time the definition of the parameters needs to be the same in both assessments. If a population is initially defined as the number of mature individuals, then it cannot be changed to the total number of individuals later on. Likewise, the area under consideration cannot become increasingly larger over time. From Fig. 1 it is also clear that sustainability and conservation status are not synonymous – on the one end a species can be correctly assessed 'Least Concern' (one of the IUCN Red List categories) and harvested unsustainably and, on the other end, an 'Endangered' species can be harvested sustainably. It is, however, important to note that harvest quotas are normally set at the species level without taking into account genetic diversity (e.g., Auliya et al. 2002; Murray-Dickson et al. 2017), behavioural variants, or possible cryptic diversity (e.g., Rawlings and Donnellan 2003), and that a lack of checks of what is actually traded may provide opportunities for laundering species under incorrect names (including those of look-alike species). This brings us to legality of trade. There is no one-on-one relationship between sustainability of a harvest and its legality. A species can be harvested sustainably in the absence of a permit to do so, or a species can be legally harvested to extinction (Table 1).

Several studies reported on the sustainability of the harvest and trade in blood pythons *Python brongersmai* in Indonesia based on visits to the same processing fa-

Table 1. Relationships between the legality of harvest and trade and its sustainability, with examples from reptiles in Indonesia, based on harvest quotas allocated for 2021 (Anonymous 2021) and seizure data from Shepherd et al. (2020). Only one of these four scenarios, legal and sustainable, is desirable.

	Legal	**Illegal**
Sustainable	Harvest of 750 tokay geckos (*Gekko gecko*) from Sulawesi and 250 from Bali	Harvest of 250 tokay geckos from the province of Central Kalimantan
Unsustainable	Harvest of 1,992,750 tokay geckos from Java	Harvest of 10,000 s pig-nosed turtles (*Carettochelys insculpta*) from Papua

cilities and covering the same geographic harvest area (Shine et al. 1999; Semiadi and Sidik 2011; Siregar 2012; Sianturi 2016; Sianturi et al. 2018; Natusch et al. 2020). Shine et al. (1999) concluded that with the data they had available to them it was not possible to assess sustainability. Semiadi and Sidik (2011) noted a 25–50% decline in the number of blood pythons (and to a lesser degree Sumatran pythons (*P. curtus*)) arriving at facilities over a ten-year period. Siregar (2012) suggested that there was no decline in sizes of blood python that were caught but 7/9 harvesters questioned indicated that fewer were caught over time. Sianturi (2018) and Sianturi et al. (2018) concluded that trade was sustainable and Natusch et al. (2020) hinted at an overharvest.

Auliya (2006) indicated that much more research is required to reliably assess the species' long-term sustainability of the harvest and trade in blood pythons, and he noted that the relatively stable export quotas of recent years indirectly conceal trade dynamics. In the Red List assessment, it was noted that in Indonesia, when the legal harvest quota was reached before the end of the year, harvesting continued and skins were stockpiled and smuggled out of the country (Grismer and Chan-Ard 2012). Stockpiling, whereby skins from harvested animals remain with traders over longer periods of time, circumvents the ability to monitor annual harvests and allows for the harvest to occur above and beyond agreed quotas without this being noticed in trade statistics. It remains uncertain whether this additional off-take contributes to a severe decline of local blood python populations (Grismer and Chan-Ard 2012). According to Saputra, the former chair of IRATA, the Indonesian Reptile and Amphibian Trade Association, another significant problem was the smuggling of skins, especially from Sumatra to Singapore, using old permits still available in Singapore (Erdelen et al. 1997).

Sutherland (2001) gave a series of pointers for assessing the sustainability of wildlife harvests. He recognised that it is usually easier (and often cheaper) to measure changes in the numbers exploited (and presumably other relevant biological characteristics such as size) but noted that this measure combines changes in population size and changes in exploitation methodology. He warned that these changes in methodology may be subtle. This includes harvesters having created new paths through a forest, or improved access to transport or a better exchange of information (e.g., mobile phones) within the harvest and trade chain (including harvesters, middlemen, transporters, processors, and exporters). Harvesters also learn and over time get better at finding

their species of interest. If the number of animals removed per day or their size distribution has remained constant over time this can be because the population is indeed being exploited sustainably. It is also possible that population is decreasing but this is compensated for by increased efficiency in harvesting or locating animals, or by expansion of the harvest area. Finally, the population could be increasing, but because of regulations on what can be harvested or traded the number of animals that enter the trade chain remain constant. Sutherland (2001) furthermore noted that if illicit exploitation is taking place, then this will be excluded from the estimates of exploitation.

Using publicly available information I here establish if there is sufficient data to assess whether blood pythons are indeed exploited sustainably. Specifically, I assess the following:

1. Is there a sufficient time between the first and the second assessment period, and if so (a.) do methods of harvest remain the same (or largely similar) between the two assessment periods; (b.) has access to harvest areas changed over the assessment period; (c.) has the harvest area remained the same; (d.) has the regulatory landscape remained the same
2. Is there evidence for illicit trade in blood pythons and if so (a.) does this happen at the national or provincial level; (b.) does this happen at the international level.

Methods

Study species

The blood python is found in eastern Sumatra and Bangka, and smaller offshore islands of Indonesia and the Thai-Malay Peninsula (with one record in Vietnam near the Cambodian border: Barker et al. 2018). The species' exact range within Sumatra, e.g., to what extent it occurs in the western, northern and southern parts of the island, is unclear (Auliya 2006; Grismer and Chan-Ard 2012). Blood pythons can grow up to ~250 cm with a mass of 15 kg (females can grow larger than males) and are found in a wide range of habitats, including agroforests and palm oil plantations (Abel 1998; Keogh et al. 2001; Grismer and Chan-Ard 2012; Siregar 2012). Most biological data we have of the species is derived from processing facilities or slaughterhouses (examining the animals that are brought in prior to or after slaughter) (Fig. 2). Indonesia is the main supplier of blood python skins and live blood pythons in international trade.

Blood pythons are not included on Indonesia's protected species list, but their harvest and trade, both domestically and internationally, is regulated by a quota system (Anonymous 2008, 2011, 2016, 2020, 2021). The harvest for domestic trade typically constitutes 10% of what is allowed to be exported. The export of blood pythons is regulated through the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), where the species is included on Appendix II (regulating all international commercial trade). The species is listed as Least Concern on the IUCN Red List of Threatened Species (Grismer and Chan-Ard 2012).

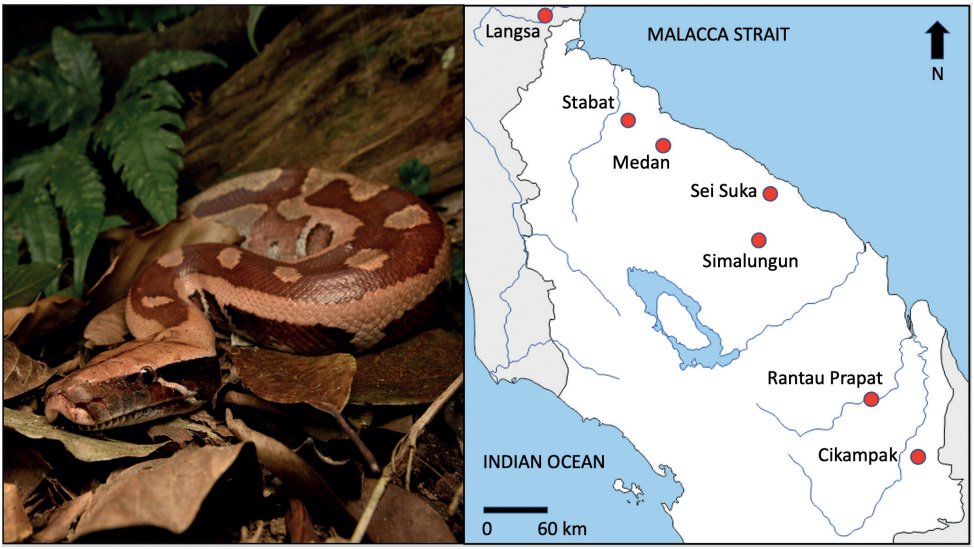

Figure 2. Left: Blood python (*Python brongersmai*) in Kaeng Krachan National Park, Thailand Right: Map of the province of North Sumatra, Indonesia, indicating the locations of slaughterhouses that were visited by various researchers to assess sustainability of the harvest and trade in blood pythons. Photo by Tontan Travel (CC-BY-SA2.0).

Data acquisition and analysis

Based on detailed observations in Indonesia between the mid-1990s to present, and through active participation in quota setting meetings, collaboration with NGOs and government bodies, and visits to numerous reptile traders, slaughterhouses, and processing facilities throughout western Indonesia, I gained an insight into the regulatory landscape of the reptile trade in Indonesia. In addition, for 12 years I was a member of the Dutch CITES Scientific Authority, with the Netherlands being one of Indonesia's main wildlife trading partners (Janssen and Blanken 2016), providing me opportunities to observe the international trade in pythons and python skins.

The facilities where pythons are killed and skinned, have their gall bladders and meat removed, and where skins are cleaned, pegged to dry, and processed, are called slaughterhouses (rumah pemotongan or rumah potong in Indonesian) by most of the researchers (e.g., Shine and Harlow 1998, 1999; Shine et al. 1999; Keogh et al. 2001; Auliya et al. 2002; Natusch and Lyons 2014; Sianturi 2018; Sianturi et al. 2018). They were also referred to as skinning premises (Shine and Harlow 1999) or skinneries (Auliya et al. 2002) but in recent years some researchers now refer to them as processing facilities (e.g., Natusch et al. 2016, 2019a, 2019b, 2020). I refer to them as slaughterhouses, acknowledging that just like slaughterhouses for the meat production the activities in these slaughterhouses are not restricted to just the killing of the animals.

To assess if any illicit trade in blood pythons occurred within Indonesia, I first focussed on the annual harvest quotas for the province of North Sumatra in 2015 and

compare them with data collected at various slaughterhouses (Table 2). Sianturi et al. (2018) and Natusch et al. (2020) both visited python slaughterhouses in two cities, i.e., Rantau Prapat (both), Cikampak (Natusch et al. 2020) and Stabat (Sianturi 2018) (Fig. 2). Natusch et al. (2020) report on two periods, i.e., August 1996–June 1997 and November 2014–September 2015, and Sianturi et al. (2018) report on one period, October–December 2015 (Sianturi 2016). Both Natusch et al. (2020) and Sianturi (2016) reported on the number of blood pythons that were brought into these slaughterhouses on the days they were present, and these were considered by them to be representative for the remainder of the year. In an earlier study, Natusch et al. (2018) report on the number of blood pythons brought into three slaughterhouses in North Sumatra combined, i.e., Rantau Prapat and Cikampak, as in Natusch et al. (2020), as well as Simalungun. Subtracting the number that were brought into Rantau Prapat and Cikampak, as reported in Natusch et al. (2020), should give the number for Simalungun. However, the observation days (24), the number of blood pythons (1,019 or 1,020) and mean number of blood pythons arriving per day for the facilities combined (42.5), are the same for Natusch et al. (2018) and Natusch et al. (2020), suggesting that the 2020 study sampled three facilities rather than two.

Secondly, for comparison, I focussed on the province of South Sumatra. Natusch et al. (2018) report on three one day visits in 2015 to one slaughterhouse in Palembang. This facility was also included in the study by Shine et al. (1999).

In calculating the number of blood pythons that were processed in these slaughterhouses in 2015 I conservatively assumed that they are operational six days a week (from experience it is more likely that they receive snakes every day). I obtained harvest quota and export quota data from the website of the Indonesian Ministry of Forestry for the period 2008 to 2022.

The time between the first and the second assessment period in the blood python studies was taken from the data presented in Sianturi et al. (2016) and Natusch et al. (2020). The same sources, in addition to Shine et al. (1999), Siregar (2012) and Sianturi (2016) were used to assess whether the methods of harvest and the harvest areas have remained the same between the two assessment periods. Additional information on the sustainability of the blood python harvest was taken from Semiadi and Sidik (2011). Semiadi and Sidik (2011) visited 10 slaughterhouses throughout North Sumatra (and

Table 2. Overview of visits to slaughterhouses to assess the sustainability of the harvest and trade in blood pythons (*Python brongersmai*) in the province of North Sumatra, Indonesia. See Fig. 2 for locations.

Year (total days)	Snakes	Rantau Prapat	Cikampak	Sei Suka	Medan	Stabat	Simalungan	References
1996–1997 (≥43)	2,063	22 days	13 days	≥4 days	≥4 days			Shine et al. 1999; Natusch et al. 2020
2007–2008 (<31)	260	not specified			not specified			Semiadi and Sidik 2011
2010 (<90)	not specified	not specified			not specified			Siregar 2012
2014–2015 (24)	1,020	8 days	16 days					Natusch et al. 2020
2014–2015 (24)	1,020	not specified	not specified				not specified	Natusch et al. 2018
2015 (≥6)	541	not specified				not specified		Santiuri 2016; Santiuri et al. 2018

one facility in Langsa just across the border in Aceh), including one or more that were included in the studies mentioned above. They obtained records of the number pythons arriving per month in 2007–2008 and compared them with records from 1998–2000.

To assess any illegal international trade, in January 2022 I obtained data on the trade in blood pythons from the CITES trade database for the period 2004 to 2020. This covers the trade in live blood pythons, skins and leather products. Data from 2021 was not yet available and prior to 2004 Indonesia did not recognise *Python brongersmai* as a species different from *P. curtus* (Keogh et al. 2001). I used data as reported by exporters.

Prices were normally quoted in Indonesian Rupiah; I corrected these for inflation to December 2021 and then converted them to US dollars.

Results

Assessing sustainability in the harvest of blood pythons

In order to assess whether or not harvest is sustainable one needs at least two temporary separated assessment points. Semiadi and Sidik (2011) compared records of the number of blood pythons (and to a lesser degree short-tailed pythons) arriving at slaughterhouses in North Sumatra from 1998–2000 with those from 2007–2008 and concluded there had been a 25 to 50% decline. Siregar (2012) based his assessment on information provided by harvesters who were asked to assess if any current trade had changed compared to three years ago (i.e., comparing 2010 with 2007). While the size of the blood pythons reportedly had remained the same, seven out of nine harvesters indicated that fewer blood pythons were harvested and two stated it had remained the same. Siregar (2012) concluded that there was a decrease in catches per area, but this was compensated by an increase of the number of harvest areas. Sianturi et al. (2018) only visited the slaughterhouses over a relatively short period (October–December 2015) thus precluding assessing whether or not harvest was indeed sustainable. The observation that an equal number of males and females are brought into a facility does not provide support that harvest occurs in a random and non-selective manner. Likewise, the observation that 272 female blood pythons were present in two slaughterhouses during three visits does not support the conclusion made by Sianturi et al. (2018) that blood pythons are still abundant in the wild. Natusch et al. (2020) did assess two time periods, but the length of time between the two (up to 20 years), with no data from the intervening period, makes it likely that changes other than just changes in the population of blood pythons have occurred.

Harvest methods, access and harvest areas

There is clear evidence from North Sumatra that over the last decades there have been marked changes in the way blood pythons are harvested, from opportunistic capture to, at least in part, targeted collection. Likewise, the harvest area has changed as well.

Siregar (2012) found that over a relatively short time period (2007 to 2010) there had been a decrease in the number of blood pythons that could be harvested in any particular area, but this was compensated for by an increase in the area where the species was harvested. Shine et al. (1999: 251) wrote that in 1995–1996 " .. most of the blood pythons came from oil palm plantations near the slaughterhouses, and were often brought in (as single snakes or in small numbers) by the people that had caught them. These snakes had been captured serendipitously, usually in palm oil plantations." Natusch et al. (2020), working with the same slaughterhouse operators, stated that "… > 80% of pythons were captured from oil palm plantations immediately surrounding the processing facilities (< 50 km) …" and "… evidence indicated that local people also targeted this species using a flashlight at night, and could actively identify and capture snakes from shelter sites". Concerning the change in collecting area, Natusch et al. (2020) commented that this "changed little between surveys" but this seems to be at odds with the data provided.

In addition, there have been significant changes in infrastructure in Sumatra with the percentage of road sections that were > 80% paved increasing from 56% in 1995 to 71% in 2005 (Rothenberg 2013), and this has increased in the years since. Hence, in the mid-1990s in North Sumatra, to travel 50 km, over even 40 or 30 km, could be quite an undertaking. "Surrounding the processing facilities (< 50 km)" does not equate with "near the slaughterhouses", and it would not be profitable to travel tens of kilometres to deliver single or small numbers of blood pythons. It is also worth rephrasing Natusch et al. (2020)'s statement – up to 20% of blood pythons that arrived at the two processing facilities were harvested more than 50 km from these facilities– as this does not give the suggestion of a local harvest and a local trade. The later conclusion is supported by data from 2008 when it was reported that the harvest of blood pythons, Sumatran pythons and reticulated pythons was carried out at distances between 2 to 90 km (and up to 120 km) from the slaughterhouses (Semiadi and Sidik 2011). Erdelen et al. (1997) furthermore indicated that the catchment area of reptile dealers in Rantau Prapat reached as far as Sibolga on the western coast of North Sumatra, at a straight-line distance of 130 km, and to Labuan Bilik on the east coast, a distance of 70 km.

Semiadi and Sidik (2011) noted that it was challenging to gain insight into the state of the python populations in North Sumatra based on information from harvesters and traders, as the change in the number of blood pythons that are brought into slaughterhouses was influenced by more than their status in the wild. They list the complexity of the harvesters-middlemen-exporter network, with individuals moving up and down the network, moving to other networks and establishing their own network. They furthermore note that at times when the price of palm oil was low (e.g., 1999–2000; 2004), many oil palm workers switched to catching wild animals, including blood pythons. Semiadi and Sidik's (2011) point of reference is a ten-year period (1998–2008) and covers a significant part of the province of North Sumatra, and thus may not be directly comparable to the experiences of individual slaughterhouse owners, but it does give the impression that much has changed in terms of how blood pythons are, and were, harvested.

Changes in regulation over time

The economic and regulatory landscape of the blood python trade has changed considerably over the last two decades, making it difficult to disentangle whether any change that Natusch et al. (2020) observed between periods is due to changes in the populations of the snakes or due to changes in the decisions consumers, traders and / or harvesters had to make.

Indonesia has an annual quota system in place for the harvest of both CITES listed and non-CITES listed species (Table 3). Exploited species have restrictions on the number of individuals that can be harvested from a specific area (typically a province) and on the number of individuals that can be exported (Amir et al. 1998; Siswomartono 1998; Nijman et al. 2012). Government-approved traders in the reptile skin trade have to be members of IRATA and are allocated part of the quota (expressed as a number of individual snakes) and they cannot trade above and beyond these quotas. If in a given year a trader does not meet its allocated quota it is not allowed to carry the remainder over to the next year. The trade in blood pythons from Indonesia is predominately for international trade. This is regulated through CITES, which oversees a limit on the number of python skins that can be exported (Kasterine et al. 2012). When exports exceed agreed numbers, sanctions can be put in place through CITES. This happened in 2004 when CITES's Scientific Review Group formed a negative opinion on the import of blood pythons from Indonesia and temporarily suspended international trade, and later that year permitted trade subject to the use of species-specific quotas (Kasterine et al. 2012). In recent years it is no longer permitted to harvest gravid female blood pythons (Anonymous 2020), but in the past there was no restriction on this.

Table 3. Harvest quotas for blood pythons in Indonesia (number of individuals for the live pet trade / number of individuals for the skin trade) for four selected years; export quotas are 90% of the harvest quotas. Bangka-Belitung was established as its own province in 2000; up to 2020 its quota was included in South Sumatra.

Province	2008	2015	2020	2021
Aceh	0 / 2,850	0 / 10,000	0 / 10,000	600 / 7,100
North Sumatra	1,500 / 21,090	1,500 / 17,840	1,500 / 18,000	1,500 / 20,000
Riau	0 / 4800	300 / 4,900	0 / 4,900	0 / 4,900
West Sumatra	0 / 0	0 / 0	0 / 0	0 / 0
Jambi	0 / 4,000	0 / 4000	0 / 4,000	0 / 4,000
Bengkulu	0 / 0	0 / 0	0 / 0	0 / 0
South Sumatra	250 / 4,300	300 / 4,300	300 / 4,300	700 / 5,000
Bangka-Belitung	-	-	-	0 / 0
Lampung	750 / 4,000	0 / 0	0 / 0	0 / 0
Total	38,740	43,140	43,000	43,800

Quotas, unregulated and illegal trade

There is considerable evidence of unregulated and illegal trade in blood pythons and that they are harvested in numbers that are above the set quotas. Erdelen et al. (1997)

noted that during their visits to the slaughterhouses of North Sumatra (including those in Rantau Prapat), it could not be determined whether harvesters who delivered their catches were turned away because of the implementation of the quota system. They inferred that the catches that were above the quotas were transferred to other provinces, and an unquantifiable or difficult-to-quantify proportion of this entered the illegal international trade. The province of North Sumatra, where Shine et al. (1999), Semiadi and Sidik (2011), Siregar (2012), Sianturi et al. (2018) and Natusch et al. (2020) conducted their studies, is allocated the largest proportion of Indonesia's blood python quota, and typically, approximately 18,000 individuals are allowed to be harvested each year (Table 3). For the slaughterhouses in Rantau Prapat and Cikampak, Natusch et al. (2020) indicate that 27.0 and 50.2 blood pythons / day are received; Sianturi (2016) suggests this number to be 10 blood pythons / day for Stabat. Combined, the three slaughterhouses receive 87.2 blood pythons/ day; assuming they are operational six days a week this amounts to more than 27,000 blood pythons each year, or around 9,000 over their allocated quota. In order to remain within the quota, on average, these facilities have to be closed for three days of the week, every week. Even if these were the only three slaughterhouses in North Sumatra, the number of snakes processed would be an overharvest of the agreed quotas, similar to that seen for a range of other reptile species (Nijman et al. 2012). However, there are other slaughterhouses that operate in the province (Shine et al. 1999; Semiadi and Sidik 2011; Natusch et al. 2018) (Fig. 2) and Siregar (2012) reported that seven exporters were active in North Sumatra.

Natusch et al. (2018) during three one day visits in 2015 recorded an average of 52 blood pythons per visit in a slaughterhouse in Palembang, South Sumatra. The harvest quota for 2015 for South Sumatra was 4,300; for the facility to remain within its quota it has be operational, or receive blood pythons, just three times every two weeks.

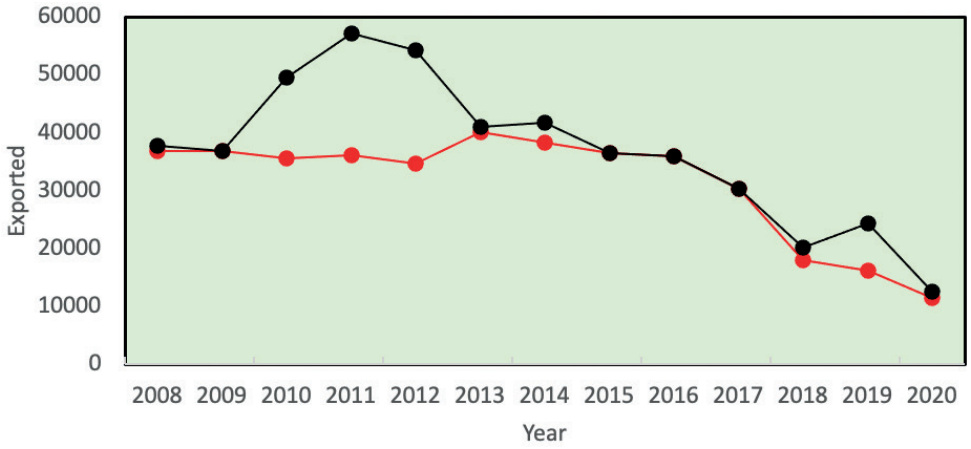

Figure 3. Export of wild-caught blood python skins as reported by exporting range countries (red: Indonesia; black: Indonesia, Singapore, Malaysia, Thailand, Myanmar). Indonesia did not report any export for 2013 or 2014 and for these years data from importing countries was used. In 2009–2012 and 2019–2020 Singapore reported the re-export of large numbers of blood python skins from Indonesia, markedly above what was reported by Indonesia as being exported to Singapore, and this surplus has been added to the total.

The overall trend of the export of whole skins from wild-caught blood pythons over the last thirteen years is downwards (Pearson's r = -0.753, P = 0.003) (Fig. 3). This downward trend is not reflected in the setting of Indonesian quotas (Table 3). Singapore emerges as a major re-exporter (i.e., a country that imports blood python skins from Indonesia and other countries and then exports them to other countries); in many years the number of blood python skins Singapore re-exports from Indonesia is substantially above what Indonesia reports as being exported to Singapore. More curious, and potentially of concern, is the re-export of blood python skins by Singapore that they declare as being imported from Lao PDR. For every year between 2009 and 2013 Singapore reported the import of blood pythons from Lao PDR, for a total of 57,500 individuals. Singapore also reported the re-export of 57,980 blood python skins (and 13 small leather products) originating from Lao PDR to the rest of the world. Lao PDR does not report the export of any blood python skins to Singapore (or any other country for that matter) other than 30,000 blood python 'chips' in 2009. The blood python skins are reported as being derived from captive-bred snakes, viz. second-generation offspring of above (CITES source code 'C'). Lao PDR is not a blood python range country and dating back to at least 2003 no country ever reported the export of live blood pythons (or fertilised eggs) to Lao PDR. Moreover, other than the imports and re-exports from Lao PDR, the 1.13 million blood python skins that were exported or re-exported between 2003 and 2020 are all labelled as wild-caught (source code 'W'). Thus, if the data that Singapore reported to the CITES Secretariat are correct then Lao PDR is the only country in the world that exported such large numbers of commercially captive-bred blood pythons for the skin trade, and did so for a short five-year period only, from stock that never was reported as being exported to Lao PDR. Perhaps a more parsimonious explanation is that blood python skins imported and re-exported by Singapore did not come from Lao PDR, were not from captive-bred snakes, but instead were wild-caught individuals imported very likely sourced in Indonesia (as the main trader in blood python skins) or, less likely, West Malaysia.

The CITES trade database also gives some additional information on the legality of the trade in blood pythons. Various countries, including Singapore and Italy, report the international trade of 14,144 blood python skins that were at one point traded without proper permits (CITES source code 'I'), all but seven of these skins, i.e., 99.9%, originated from Indonesia. The 14,137 illegally traded blood python skins that were intercepted, represent 3.0% of the total trade in blood python skins that was reported by Indonesia over this period. Accepting that only a proportion of the illegally exported blood python skins would be intercepted and then traded internationally, this suggests a significant illegal trade in the species.

There are some peculiarities in the export of live blood pythons from Indonesia over the last 17 years (Fig. 3). Just like the trade in skins, their harvest is subject to a quota system, and a large proportion of this (60–70%) is allocated to the province of North Sumatra (Table 3). Typically, for all of Indonesia, each year between 1,890 and 2,250 live blood pythons can be exported for the pet trade, and indeed Indonesia reports the export of, on average, 2,195 ± 441 live wild-caught individuals a year.

In addition, Indonesia reports the export of live captive-born (F), and some captive-bred (C), blood pythons, but these numbers vary considerably between years (mean 1,112 ± 731 individuals, range 0 to 2,527 individuals). Excluding three years when Indonesia did not report the export of any captive-born blood pythons, there is a positive correlation between the (log-transformed) number of wild-caught and captive-born live blood pythons that are exported (Pearson's R = 0.8254, N = 14, P = 0.0003). In years when many wild-caught blood pythons are exported – this being years that the maximum quota is reached – many more captive-born ones are exported in addition to the wild-caught individuals. In years when fewer wild-caught individuals are exported –and when there is room within the quota to export more—far fewer captive-born ones are exported. This may suggest that in certain years when demand is high, wild-caught blood pythons are exported under the label of captive-born.

Finally, the trade in blood pythons from Indonesia is not restricted to whole skins or live animals. Indonesia is also the exporter of small leather products made from wild-caught blood pythons (it reports the export of 67,950 small leather products over the period 2004–2020, Malaysia reports the export of 2 small leather products over

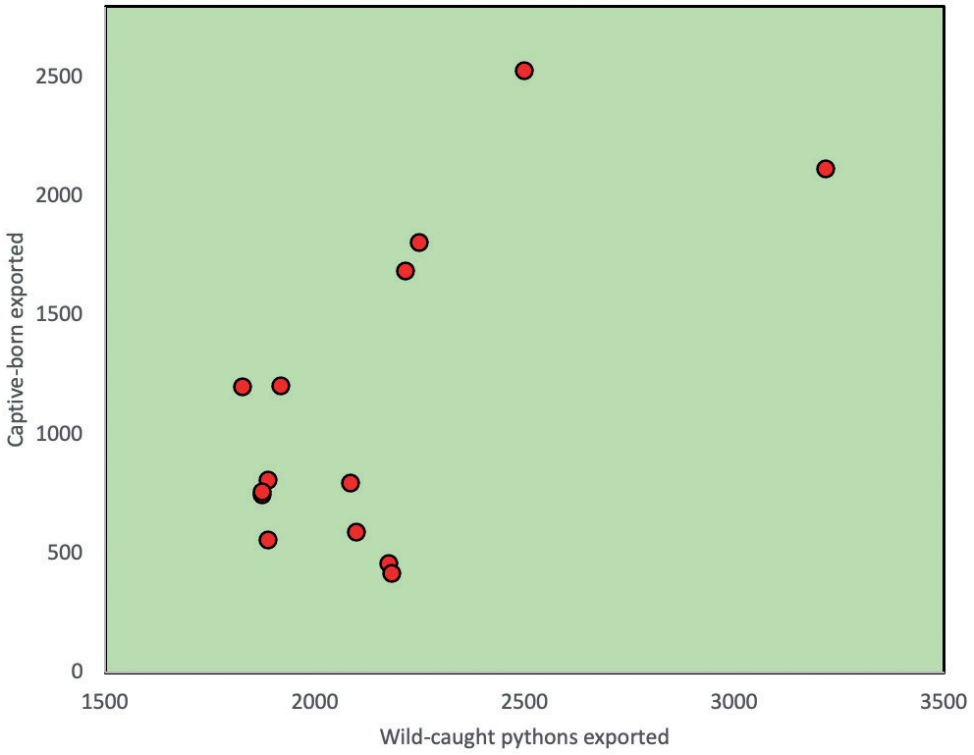

Figure 4. Relationship between the annual number of live wild-caught blood pythons that Indonesia reports as being exported between 2004 and 2020 and the number of captive-born blood pythons it exports in that same year. Wild-caught blood pythons are subject to a harvest and export quota (between 1,890 and 2,250 individuals per year) whereas the export of captive-born blood pythons is not subjected to a quota.

this period). For those years that Indonesia did report these exports (in 2013 and 2014 Indonesia did not report any export), on average 4,850 small leather products are exported. It is difficult to convert this to whole animal equivalents, and the export may occur years after the blood pythons have been harvested, but this adds to the export of whole skins and live blood pythons reported above.

Discussion

Assessing sustainability in the harvest and trade of blood pythons

I here aimed to establish if there was sufficient data to assess whether blood pythons are exploited sustainably based on information that has been presented in the literature. The first question that needed to be answered was whether there was sufficient time between the first and the second assessment period for an assessment to be made. This varied greatly between studies, ranging from a few months to twenty years. The lower end of this is obviously too short to make an assessment of sustainability and the latter makes it unlikely that what is measured reflects just changes (or the absence of changes) in the population of blood pythons. Siregar (2012) and Semiadi and Sidik (2011) assessed sustainability over a three-year and ten-year period, respectively, and based on information provided by traders found that there had been a decline in numbers that were harvested.

For the studies that assessed change over a longer period, it was clear that the methods, and quite likely intensity, of harvest had not remained the same, with more targeted collecting in the latter compared to the former periods. Likewise, there is good support that in North Sumatra access to harvest areas improved over time. Harvest areas had either been enlarged or significant shifts in harvest areas had occurred. As such any changes in the numbers or sizes or condition of the blood pythons that end up in trade may be due to changes in harvest methods or changes in harvest areas. Conversely, any non-change can be due to genuine stability of the population, or it can be due to harvesters moving to other areas or deploying different collecting strategies. The other areas may even include neighbouring provinces as suggested by Natusch et al. (2018).

While Natusch et al. (2020) stressed that the two slaughterhouses they visited in 1996–1997 and 2014–2015 were continually operated by the same owners, in a similar manner, across the survey period, it appears that the former period included data from visits to slaughterhouses in two cities and the latter period, including data from one more city. Shine et al. (1999) reported on the harvest in blood pythons in four cities, i.e., Medan, Sei Suka, Rantau Prapat, and Cikampak, and the latter two were included in Natusch et al. (2020) study. Natusch et al. (2018) however, reports the same data (survey days, number of blood pythons harvested, harvest rate) as Natusch et al. (2020), but this was obtained by visits not to two but three facilities. The third one, Simalungun, is a five-hour drive from Rantau Prapat and a seven-hour drive from Cikampak so it is unlikely that this could have been legitimately pooled with either

city. Simalungun is also not near Medan or Sei Suka so it cannot have been pooled with these cities in the earlier study (Fig. 2). Hence it seems that Natusch et al. (2020) were comparing data from three slaughterhouses visited in 2014 and 2015 with data from two slaughterhouses in 1996–1997. This obviously is problematic in terms of assessing sustainability of the harvest and trade.

The predictions on how blood python populations were expected to change are based largely on the knowledge of the biology of exploited animal populations (Sianturi 2016; Sianturi et al. 2018; Natusch et al. 2020); they are not based on the decisions made by harvesters, traders, exporters, regulators, or consumers. Biologists only expect these changes in the biological traits of blood python populations over time if the actions of the people involved in the trade remain static. The economic and regulatory landscape of the blood python trade has changed considerably over the last two decades, making it difficult to disentangle whether any change between periods is due to changes in the populations of the snakes or due to changes in the decisions consumers, traders and / or harvesters had to make. Especially changes in how the domestic and international trade is regulated can have a marked impact on what will be observed at a trader's premises. Imagine a trader that is free to buy and sell each and every python that is brought to their processing facility and assume that profits are positively correlated to size (i.e., the largest profit can be made on the largest individuals: corrected for inflation to December 2021 values, Siregar (2012) reported that harvesters in North Sumatra fetch a price of US$13.35 for a large blood python but only US$8.00 for a small one, thus supporting that assumption). A logical economic decision for a trader is to buy all those animals that will give a profit, including large, medium, and small ones (but perhaps not the very small ones) especially as the process of inspection and measuring already requires an investment. Over time over-exploitation will lead to harvesters bringing in fewer snakes and proportionally smaller ones and fewer very large ones. These are the assumptions under which Natusch et al. (2020)'s model works, and this was the situation that may have been the case when Shine et al. (1999) collected their data in the late 1990s. But now imagine that a trader is not free to buy whatever he or she wants, and that not all can be sold into a free market (i.e., trade is regulated). In this scenario, the most profitable economic decision is to buy first (or preferably) the ones that will bring the largest profit (in our example, the largest snakes), then the ones that bring in sufficient profit (the medium-sized snakes) but traders will not purchase the smallest ones that bring in only a small profit (using data from Siregar (2012) suggest a loss of potential revenue in the order of 40% if a small rather than a large blood python enters the trade). If the restrictions are time-bound, this may translate into a trader buying only large snakes at the beginning of the year, large and medium-sized snakes in the middle of the year, and large, medium-sized and small snakes at the end of the year to maximize profit. The anthropogenic Allee effect proposes that when prices for wildlife products increase with species rarity, then financial incentives are created to extract the last remaining individuals of a population, despite higher search and harvest costs (Holden and Mcdonald-Madden 2017). Harvesters and others involved in the blood python trade chain will increase their efforts to catch those snakes that

make the most profit, i.e., the largest individuals, resulting in an increase in costs but not necessarily in a change in the number of large snakes entering the trade. In these scenarios, even for heavily exploited populations one would not necessarily expect to see the changes in snakes at the processing facilities predicted by Natusch et al. (2020) and Sianturi et al. (2018). This is the situation in which blood python snake traders and harvesters operate at present.

Misdeclared, underreported and illegal trade in blood pythons

As noted by Sutherland (2001) when assessing the sustainability of exploitation, almost invariably, the illicit part of this is not, and often cannot be, taken into account. While in the present study it was not possible to assess whether individual traders had engaged in any illegal activities or had underreported the real intensities of trade, at the aggregate level it is clear that there are numerous indications that blood pythons are traded illegally, that trade is underreported and/or that trade is misdeclared. At the international level, reported levels of trade in blood pythons do not match, and this takes different forms. For instance, Singapore re-exports many more blood python skins from Indonesia than Indonesia exports to Singapore. Lao PDR for a short period was a significant exporter of blood python skins without it being a range country and without it having imported blood pythons to initiate this trade.

Trade in live blood pythons from Indonesia, shows anomalies, whereby in the years that the maximum number of wild-caught individuals is reached, the number of blood pythons that are declared as captive-born (and not included in the annual harvest quota) that are exported actually increases. At least since 2019 it has not been allowed to harvest gravid female blood pythons from the wild and the captive-born and captive-bred blood pythons must have been derived from dedicated captive breeding facilities. It is questionable, however, that actual captive breeding blood pythons makes economic sense. As noted above, depending on the size, harvesters get paid between US$8–15 for a blood python (Siregar 2012). Housing, feeding, maintaining, and breeding blood pythons in a commercial setting cost more (Anonymous 2011; Siregar 2012). This is supported by research conducted by TRAFFIC (2013) who, for three species of blood python concluded "There is little evidence of captive breeding these species due to the relatively low price paid for skins compared to the larger python species". The close link with the number of wild-caught blood pythons that are exported and the number of captive-born ones, such that the latter is dependent on the former, suggests an economically uncertain market, and makes it less plausible that blood pythons are indeed bred at a commercial scale (for a similar case with wild-caught vs captive-bred Tokay geckos, see Nijman et al. 2018).

The reason the Indonesian government, and in a global arena, CITES, sets limits on harvest and export, is to prevent species such as the blood python from being overharvested. In Indonesia, it is the Indonesian Institute of Sciences that provides the scientific justification for these limits (Soehartono and Mardiastuti 2002). In Indonesia, for many species Non-Detriment Findings (the conclusion by experts that the

export of specimens of a particular species will not impact negatively on the survival of that species in the wild) are rare (Soehartono and Mardiatuti 2002; Auliya et al. 2016). This is partially due to the lack of biological and harvest data, but also due to economic pressure and lack of political will. In practice then, the final decision on quota numbers and what is harvested where is negotiated and decided upon annually in a meeting where members of the IRATA are present in addition to members of the Indonesian Institute of Sciences and the Ministry of Forestry (Amir et al. 1998; Nijman et al. 2012).

It is possible that one of the outcomes of the studies by Shine et al. (1999), Semiadi and Sidik (2011), Siregar (2012), Sianturi (2016), Sianturi et al. (2018) and Natusch et al. (2020) is to argue that despite a substantial harvest above the permitted quotas and evidence of illegal trade, the harvest in blood pythons as conducted so far is sustainable, and therefore harvest quotas can be (and perhaps should be) raised. I would argue that given that the decisions traders make in relation to what they can and cannot buy and sell are not just theirs to make, the conclusion that the best way to assess sustainability of the commercial harvest is to monitor the attributes of harvested pythons, is flawed.

While harvest quotas for different provinces, or indeed for Indonesia as a whole, differ little from year to year when seen over a longer time period some marked geographic changes are apparent (Table 3). For instance, Riau saw a ten-fold increase and Aceh a more than three-fold increase between 2008 and 2015. Harvest of 4,750 blood pythons (12% of the country's total) was allowed from Lampung in 2008, but none were permitted to be harvested in 2015. While these changes may be indicative of changes in the abundance of blood pythons, possibly due to local overharvesting, they most likely simply reflect the requests of Indonesian Reptile and Amphibian Trade Association members and what they wish to export. The harvest of blood pythons from South Sumatra requires scrutiny and continued monitoring. The data presented by Natusch et al. (2018) may suggest that the number of blood pythons arriving at one slaughterhouse already exceeds agreed quotas. It is furthermore noteworthy that Shine et al. (1999) recorded very few blood pythons at the Palembang slaughterhouse (mean of 11.7 / visit), and Natusch et al. (2018) indicate that very few blood pythons arriving at this facility were harvested in the South Sumatra. There seems to be a consensus that the species is not abundant in the province. The blood pythons that were received originated from Bangka Island (Natusch et al. 2018). Bangka (and the neighbouring island of Belitung) administratively separated from South Sumatra in 2000 and thus became Indonesia's 31st province. In the period up to 2019, quota-wise, Bangka was included in South Sumatra, but in 2021 and 2022 it has been given its own quota of zero (Table 3). The quota for South Sumatra has remained stable at 5,000 skins and 700 live blood pythons. Monitoring is required to ensure that blood pythons arriving at the slaughterhouse and at traders' premises in Palembang are indeed harvested in South Sumatra and not in Bangka (as they have been in the past), and conversely, that the harvest of blood pythons on Bangka, a regular occurrence for decades, is indeed completely halted.

Monitoring of wild populations

Contrary to, for instance Natusch et al. (2019b), as argued convincingly by Sutherland (2001), to assess sustainability of harvest (and trade) it is better to monitor the population than the harvest. For this in-depth and objective analysis of critical population parameters such as harvest rates at different points in time (past, present, and potentially future) and how this affects density is crucial (Challender et al. 2021). The fact that different studies, visiting the same slaughterhouses in roughly the same time period, measuring, in part, the same parameters, come to diametrically opposite conclusions, is testament that just visiting processing facilities to determine sustainability needs reconsidering. Determining changes in population size is better for adjusting the exploitation level as it is the population size that really matters (Sutherland 2001). The reason Natusch et al. (2020) gave for not assessing the sustainability of the trade in pythons directly by measuring the effects of the harvest on the populations themselves, as done by Abel (1998), Riquier (1998) and Auliya (2006), amongst others, and as advocated by Sutherland (2001), was due to difficulties in obtaining these data from the field (another argument is that it is presumably cheaper). The studies conducted by Abel (1998), cited by Natusch et al. (2020) as an example where field studies had been unsuccessful in quantifying abundance and demographic attributes of harvested python populations, and Siregar (2012) offers some insight in the feasibility of conducting mark-recapture population estimates as a potential approach for these assessments. Abel (1998), over a 4 to 5-month period, during which two other reptile species were studied, captured 113 blood pythons, recaptured 22 of them twice and three were recaptured three times. Two professional python harvesters interviewed by Siregar (2012) were able to collect 503 blood pythons over a 7 to 8-month period. With a suitable team, including one or more professional python harvesters, collecting relevant data in a standardised manner from the field rather than from just relying on potentially biased data from processing facilities should be feasible. Based on data Indonesia reported to CITES it exported over three million snake skins over the last decade; there is no valid reason why this multi-million-dollar business cannot properly assess the real effects on the wild populations it is dependent on.

Conclusion

Measuring sustainability is often done indirectly by testing predictions regarding what would happen if a harvest was not sustainable. These studies in general cannot demonstrate sustainability; at best all they can do is to demonstrate with a level of certainty that harvest is not not sustainable. Five studies commented or attempted to assess sustainability in the harvest of blood pythons in the province of North Sumatra using data from slaughterhouses to establish if vital attributes of the harvested animals changed over time. Semiadi and Sidik (2011) and Siregar (2012), relying on data provided by the staff of the slaughterhouses, suggested that the number of blood pythons that were harvested had declined and that harvesters had shifted collection areas. Shine et al.

(1999) and Sianturi et al. (2018) conducted their studies over relatively short time intervals, and while the former concluded that it was not possible to assess sustainability, the latter suggested it did without providing any relevant data to support this conclusion. Natusch et al. (2020) included data from Shine et al. (1999) in their assessment, and concluded that the harvest had been too severe. However, the introduction of regulations on harvest and export, including harvest and export quotas, likely changes in the harvest area (from near the processing facilities to further away), differences in the number of processing facilities between years (two vs three) and collection methods (from opportunistic to in part targeted collection) largely invalidates their predictions.

As such, despite decades of commercial trade from North Sumatra, there is insufficient data to suggest whether the harvest and trade of blood pythons out of this province is sustainable. Data on the sustainability from other parts of the blood python's range, including from southern Sumatra (Lampung, South Sumatra, Jambi, Bangka) where over the last decade ~100,000 individuals were allowed to be harvested, are lacking. There is, however, substantial evidence of underreported and illegal international trade in blood pythons. Part of any assessment of sustainability of the harvest and trade in blood pythons must address this as a matter of urgency.

Acknowledgements

I thank the Indonesian Institute of Sciences, the Ministry of Forestry, individual members of IRATA, and TRAFFIC Southeast Asia for their support over the years. Three reviewers and the editor provided very helpful suggestions for improvement, for which I am grateful.

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Nature Conservation 48: 123–147 (2022)
doi: 10.3897/natureconservation.48.81635
https://natureconservation.pensoft.net

RESEARCH ARTICLE

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Potential for informal guardianship in community-based wildlife crime prevention: Insights from Vietnam

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Academic editor: Thomas Ziegler | Received 2 February 2022 | Accepted 22 April 2022 | Published 19 May 2022

http://zoobank.org/C6E53BAB-82A1-49C2-B3A6-F76FA5448953

Citation: Viollaz J, Rizzolo JB, Long B, Trung CT, Kempinski J, Rawson BM, Reynald D, Quang HX, Hien NN, Dung CT, Huyen HT, Thuy Dung NT, Gore ML (2022) Potential for informal guardianship in community-based wildlife crime prevention: Insights from Vietnam. Nature Conservation 48: 123–147. https://doi.org/10.3897/natureconservation.68.81635

Abstract

The notion that indigenous people and local communities can effectively prevent conservation crime rests upon the assumption that they are informal guardians of natural resources. Although informal guardianship is a concept typically applied to "traditional" crimes, urban contexts, and the global North, it has great potential to be combined with formal guardianship (such as ranger patrols) to better protect wildlife, incentivize community participation in conservation, and address the limitations of formal enforcement in the global South. Proactive crime prevention is especially important for illegal snare hunting, a practice that has led to pernicious defaunation and which has proved difficult to control due to its broad scope. This paper uses interview data with community members in protected areas in Viet Nam where illegal snare hunting is commonplace to 1) analyze the conditions for informal guardianship in the study locations; 2) explore how community members can become more effective informal guardians; and 3) examine how formal and informal guardianship mechanisms can be linked to maximize deterrence and limit displacement of illegal snaring. Results indicate that conditions for informal guardianship exist but that re-

spondent willingness to intervene depends upon the location, offender activity, and type of offender (outsider versus community member). While respondents generated numerous strategies for wildlife crime prevention, they also listed crime displacement mechanism offenders used to avoid detection. We discuss how informal guardianship can be integrated with formal guardianship into an overall model of situational crime prevention to protect wildlife and incentivize community-led deterrence of illegal snaring.

Keywords
conservation crime, defaunation, deterrence, local communities, situational crime prevention, snare hunting

Introduction

The diversity, engagement, and inclusion of local stakeholders in community-based conservation is a mainstream practice and proven theory (e.g., Doyle-Capitman et al. 2018). The power of community-based conservation, however, lies in its evolution as a response to the shortcomings of the fortress model of conservation in which governments or other actors created protected areas for wildlife and prohibited their use by indigenous people and local communities, or evicted them (IPLCs) (Turner 2004). These exclusionary forms of conservation often failed to achieve successful preservation of biodiversity due to social resistance or non-cooperation.

Community-based conservation reflects participatory approaches that view IPLC forms of knowledge as requisite for resource management (Berkes 2004). Importantly, the mere inclusion of stakeholder involvement in conservation practice does not guarantee positive outcomes for biodiversity or livelihoods. The specific characteristics of stakeholder participation, and their interaction effects, have consequences for efficacy and sustainability (Young et al. 2013). There are also numerous examples of how inattention to community differentiation and attributes like gender, identity, age, ethnicity, and wealth can limit the effectiveness of community-based conservation (e.g., Little 1994; Leach et al. 1999; Alexander and McGregor 2000). In conservation, the notion of community overwhelmingly refers to a group of people who live in spatial proximity to one another and/or share common interests or social identities (Murphree 1994). However, social bonds, or the level of group affiliation, is another factor that is particularly relevant when community-based conservation involves responses to conservation crime (Rizzolo et al. 2017).

Conservation crime refers to crime that involves the natural world and its inhabitants, such as illegal logging, illegal or unregulated fishing, illegal pollution, and the illegal trade and consumption of wildlife, among others (Gibbs et al. 2010). The notion that IPLCs (and their knowledge, skills, and relationships) can effectively prevent conservation crime rests upon the assumption that they are informal guardians of natural resources. Informal guardianship is a concept developed in and typically studied by scholars in the global North, traditional crime contexts, and urban settings (Reynald 2009, 2011a, b; Jacques and Reynald 2012; Hollis-Peel and Welsh 2014; Moir et al. 2017). Informal guardianship may have unrealized potential to be combined with for-

mal guardianship (such as ranger patrols) to better protect wildlife, incentivize community participation in conservation, and to address the limitations of formal enforcement in the global South (Kahler 2018). To date, the conservation science literature on informal guardianship is underdeveloped. There is a clear opportunity to synergize these approaches to better understand the potential and limitations of community-based crime prevention and informal guardians.

Informal guardianship

As a field devoted to analyzing the human causes of, and solutions to, crime, criminology can enhance existing conservation-led enforcement work (Gibbs et al. 2010). Crime occurs where a motivated offender, a suitable target (in this case, wildlife), and the absence of a capable guardian intersect (Cohen and Felson 1979). A capable guardian is any person or thing that discourages criminal violations from occurring (Cohen and Felson 1979). Contemporary definitions of capable guardianship focus on the importance of human guardianship rather than the protection provided by objects such as CCTV (Hollis-Peel et al. 2013). The mere presence of a capable guardian can prevent crime through their ability to keep an eye on potential targets (Felson 1995). Empirical research shows that guardians can discourage crimes within their communities through their presence, supervision, and intervention when necessary (Reynald 2011b; Felson and Eckert 2016); guardians can be either formal or informal (Table 1).

Thus far, the concept and practical study of informal guardianship has been focused on traditional crimes (e.g., property destruction) in urban environments where population density is high, ownership of property is clear, and offenders are primarily nonlocal (Reynald 2009, 2010, 2011a, b; Jacques and Reynald 2012; Hollis-Peel and Welsh 2014; Moir et al. 2017). This creates a high likelihood that an informal guardian will be present at the same time and place as an offender, can identify an offender, and will intervene. These characteristics are not guaranteed for illegal snare hunting in Viet Nam, where some hunters are outsiders while others are local (Viollaz et al. 2021). That "offenders" can be the neighbors of informal guardians makes it far more complex for them to intervene. There are social costs to policing one's neighbors that

Table 1. Types of capable guardians.

Type*	Definition	Benefits
Formal	Representatives of the state (e.g., police or rangers) with official enforcement powers	Power to enforce legal sanctions Professionally trained in crime detection/prevention
Informal	Non-professionalized (e.g., community-based) protectors of targets such as wildlife	Can serve as force multipliers: can amplify and complement the reach of formal guardians (Carter and Gore 2013) Valuable in rural areas where law enforcement density is lower (Carter and Gore 2013) or where ranger motivation and/or capacity is limited (Spira et al. 2019) Can supervise and protect potential targets during the course of their routine activities (Felson and Boba 2010)

*One actor cannot act both informally and formally at the same time. Conceivably, an off-duty ranger who does not identify herself as such can act as an informal guardian in her community. Then she would be, in role, a formal guardian (because of her job) but, since she wasn't acting as part of her formal duties, would be categorized as an informal guardian in that context.

could play a role in community members' willingness to act as informal guardians. Further, although some people may know that it is illegal to hunt in a protected area, the offense is not necessarily viewed by communities as deviant since wild meat is readily consumed in their circles (Van Song 2008; Ngoc and Wyatt 2013; Lee et al. 2014).

Despite these theoretical and practical complexities, there is the potential and motivation to leverage the concept of informal guardians for more effective conservation practice. Research and field surveys conducted over the past 15 years have documented a rapid decline in the fauna and flora of Viet Nam (Wood et al. 2013), challenging policy makers, scientists, and local communities to explore additional options for tackling Viet Nam's illegal snaring problem (Polet and Ling 2004; Zingerli 2005; Ngoc and Wyatt 2013). The close social ties between neighbors and the contextual familiarity they have within these communities, defined geospatially, provides an excellent opportunity for informal guardianship because neighbors are aware of each other's conduct on a daily basis. Community members are therefore in an exceptional position to know when someone is doing something illegal than in cases with outside offenders or when outside authorities monitor a community's behavior. People in these communities can therefore provide guardianship where formal guardianship is not readily available because of low capacity or lack of resources.

Snare hunting

Snaring is one conservation issue to which informal guardianship may be an especially well-suited solution. Snaring is one of the largest contributors to defaunation and a pervasive threat to biodiversity across continents (Watson et al. 2013; Gray et al. 2018; Belecky and Gray 2020). The detriments of snares are challenging for numerous reasons. Snares are cost-effective to construct, clandestinely placed in remote locations, and yield indiscriminate wildlife injury and mortality (MacMillan and Nguyen 2014; Gray et al. 2018). This makes snares both a significant threat to wildlife species and makes them difficult to detect and their users hard to identify and sanction. Snare detection and removal are essential risk mitigation strategies, and there have been important experimental and practical studies of snare detection techniques meant to optimize enforcement resources (e.g., Watson et al. 2013; O'Kelly et al. 2018). However, due to the sheer volume of snares and the ease with which snares can be replaced, snare removal alone is not sufficient to protect wildlife species (Gray et al. 2018).

Snaring in Southeast Asia is a conservation priority (Belecky and Gray 2020) because of the region's large number of threatened wildlife species, its high rates of forest loss, and pervasive road and other infrastructure encroachment on wildlife habitat, which facilitates snare placement. Rapid economic growth in Viet Nam has also contributed to a robust and growing consumer base for wildlife products, particularly wild meat, both locally and when transported to urban areas (Sandalj et al. 2016; Gray et al. 2018). In Viet Nam, trappers with more access to valuable species (such as pangolins) and to wildlife traders tend to participate in commercial trade (MacMillan and Nguyen 2014). Research by MacMillan and Nguyen (2014) suggests that local communities

in Viet Nam have the knowledge to manage forests in a sustainable manner and prevent professional hunters from entering reserves where land tenure is clear and could potentially be recruited to deter conservation crimes as informal guardians (MacMillan and Nguyen 2014). However, local participation in such deterrence is dependent upon variables such as the presence of nonlocals (and whether nonlocals are prosecuted for conservation crimes), location, and cultural traditions of wildlife utilization (Rizzolo et al. 2017).

At the same time, larger limitations and issues with enforcement highlight the importance of further work on crime prevention strategies, particularly in the context of snaring. Rangers often have a large, protected area to cover and are subject to various occupational stressors that can affect motivation and capacity (Moreto 2016). At times, rangers can be subject to larger forces of corruption and can engage in misconduct that limits their effectiveness (Moreto et al. 2015). In locations where there is weak governance and/or the criminal justice system does not function effectively, the conservation impacts of enforcement may be limited (Nijman 2017), making crime prevention even more important. For example, in Viet Nam's Pu Huong Natural Reserve, Wildlife Management Clubs set up by local communities play an important role in wildlife protection and anti-trafficking efforts.

All of these factors suggest that informal guardianship could be a valuable strategy in the prevention of snaring-related conservation crimes. As members of the community, informal guardians have more flexibility to use community social cohesion and social bonds to leverage changes in behavior or promote compliance (Wilcox et al. 2007). Therefore, it is possible that community members can better discourage crime than formal guardians, especially in situations where there are tensions between communities and rangers or other conservation stakeholders. This paper uses interview data with community members in protected areas in Viet Nam where illegal snare hunting is commonplace to 1) analyze the conditions for informal guardianship in the study locations; 2) explore how community members can become more effective informal guardians; and 3) examine how formal and informal guardianship mechanisms can be linked to maximize deterrence and limit displacement (the transfer of crime from one location to another, Johnson et al. 2014) of illegal snaring from one protected area to another.

Methods

From May to August 2018, the research team from Vinh University in Viet Nam conducted 303 semi-structured interviews with community members (both hunters and non-hunters) in 12 villages that surround three protected areas: Quang Nam Saola Reserve, Thua Thien-Hue Saola Reserve, and Pu Mat National Park in the Central Annamites region of Viet Nam (Fig. 1; Table 2).

About 30% of the sample was composed of hunters, and approximately 60% of those hunters were self-identified "subsistence hunters" whose livelihoods were dependent on agriculture but hunted in their spare time, mainly for personal con-

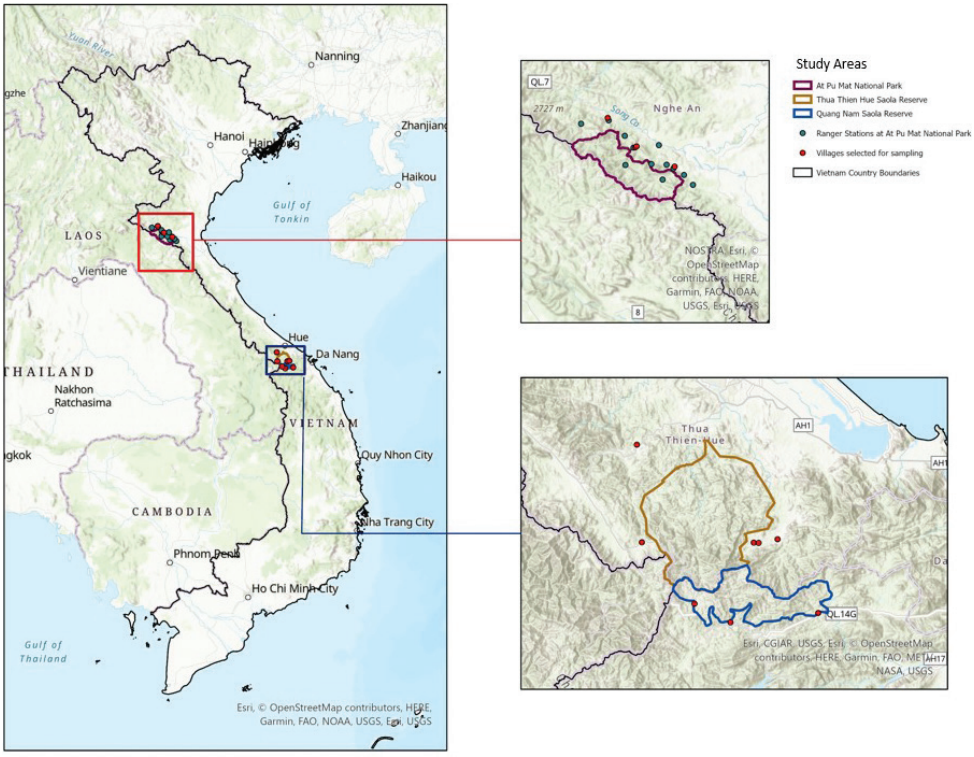

Figure 1. Maps of site locations.

Table 2. Interview locations and number of interviews conducted per site (from Viollaz and Gore 2019).

Quang Nam Saola Reserve				
Arec (A Vuong)		Atep 2 (Bhalee)		But Nga (Song Kon)
25		25		29
Thua Thien Hue Saola Reserve				
A Roang 1 (A Roang)	Village #3 (Thong Quang)	Village #3 (Thuong Long)	Cha Du (Huong Nguyen)	Village #3 (Thuong Nhat)
25	25	25	25	25
Pu Mat				
Xieng (Mon Son)	Na (Chau Khe)	Bu (Chau Khe)	Quang Phuc (Tam Dinh)	
23	25	26	25	

sumption (Table 3) The rest were what the researchers termed "inside professional hunters" because they also belonged to the community (rather than coming from outside the province to hunt) but relied mainly on hunting for income and had specialized knowledge and tools for finding the best game (see Viollaz and Gore 2019 for more details). While an additional category of hunters ("outside professional hunters") was mentioned by respondents, these hunters were not members of the community and were not interviewed for the purpose of this study. An "outside professional hunter" was defined as a hunter who relies mainly on hunting for income, is prolific, has expert knowledge of navigation and best places to hunt using specialized tools and snares but

Table 3. Prevalence and characteristics of hunters and non-hunters in the sample (from Viollaz and Gore 2019).

Pu Mat (N = 99)			Quang Nam Saola Reserve/Thua Thien Hue Saola Reserve (N = 204)		
Hunter (%)					
Yes	No		Yes	No	
26% (N = 26)	74% (N = 73)		30% (N = 61)	70% (N = 143)	
Hunter type (%)					
Subsistence	Inside professional	Outside professional	Subsistence	Inside professional	Outside professional
58%	42%	0%	60%	40%	0%
Sex (%)					
Male	Female		Male	Female	
84%	16%		78%	22%	

who lives primarily in other Provinces and who travels long distances to hunt in the park or reserve. This research was part of a larger project that looked at both community and ranger perspectives towards wildlife crime prevention in Viet Nam (Viollaz et al. 2021; Rizzolo et al. 2021).

Interviews focused on involvement in illegal snare hunting, knowledge of hunting practices, potential hunting deterrents, and community members' willingness to intervene to prevent the behavior (two researchers from Vinh University and Michigan State University also conducted exploratory interviews with rangers to get their perspectives on the potential for informal guardians to work in the context of these protected areas, see Rizzolo et al. 2021). The Human Subjects Protection Program at Michigan State University's Institutional Review Board approved the methods and analysis (IRB #00000372). In the analysis phase, the study sites were divided into two categories: Hue-Quang Nam Reserve (henceforth, "HQN") and Pu Mat National Park (henceforth, "Pu Mat"). The quantitative answers from the community interviews were coded in SPSS v25 (IBM Corp 2017). During analysis, the project translator gave regular input and corresponded with the interviewers to guarantee the cultural context of answers was not lost. Descriptive statistics were run on the data once coded. For those answers that could not be quantified, content analysis techniques were used to pull out patterns and trends in answers.

Results

Conditions for informal guardianship

The first aim of this paper was to examine the conditions for informal guardianship. Results indicated that the three conditions necessary for community-based informal guardianship (availability, knowledge of context, and willingness to intervene) are present in our study sites. A large portion of the 303 interviewees (88% in HQN and 81% in Pu Mat) reported being present in their communities for a minimum of 25 days per month, which indicates sufficient time spent there to be available as informal guardians. In terms of knowledge of context, a majority of interviewees (62% in HQN and

54% in Pu Mat) reported that they knew their neighbors well enough to be aware of their habits and to detect when their behavior varied from the norm. The percentage of respondents who knew specifically when their neighbors engaged in hunting was slightly lower, at 36% for HQN and 47% for Pu Mat.

Overall, respondents reported a strong willingness to intervene. When given the example of a general crime being committed in their community (e.g., the crime of "stealing a buffalo"), 92% of interviewees in HQN (and 96% in Pu Mat) noted that they would be very likely or likely to intervene. However, responses varied on how they would intervene. The largest percentage of respondents (41% in HQN and 51% in Pu Mat) said they would "intervene indirectly," with a smaller portion of interviewees (32% in HQN and 35% in Pu Mat) reporting that they would "intervene directly." Some community members clarified (27% in HQN and 14% in Pu Mat) that their response would depend upon the situation. There was also a significant interest in the protection of wildlife among respondents. Across all sites, 28% of interviewees noted that it was everyone's or the community's responsibility to protect wildlife.

A variety of situational factors impacted respondents' degree of willingness to intervene. These included their gender and role in the community, the context (the type of crime and the perceived effectiveness of intervention), and the type of offender (whether or not that person is a local or non-local resident). There were pervasive gender differences in willingness to intervene. All the women surveyed in Pu Mat reported that they would respond indirectly (e.g., through requesting help). In HQN, most women favored indirect interventions, although a subset of 28% reported that their response would depend upon the situation. However, for male respondents across sites, direct and indirect interventions were evenly endorsed. Further, authority figures in the community were more likely to favor direct interventions such as confrontation of the offender.

Approximately one-fifth of respondents in HQN (and 13% in Pu Mat) said they were very likely to intervene if they witnessed an individual snaring in a protected area. However, perceived effectiveness of this intervention was low. Only 35% of community members in HQN and 31% in Pu Mat believed that they could stop a person from snaring inside the protected area. Willingness to intervene was related to the respondent's belief that the intervention would be successful. Almost all the respondents who reported neutral or negative answers to intervention noted that they felt they could not effect change.

As with other types of crime, whether the offender was perceived as a local or non-local impacted intervention (Table 4). Several interviewees at both sites noted that they would intervene if they saw an outsider laying snares in the protected area. Their rationale was that this land belonged to their community, and thus an outsider should neither be present there nor be allowed to take resources. Several respondents also noted their responsibility to protect their village by intervening. For all three types of transgressions (entering the protected area, laying snares in the protected area, exiting the protected area with bushmeat), a higher percentage of respondents would intervene if the offender was an outsider than if they were a community member. Hunters were

also willing to confront other hunters who laid snares in their own hunting territory, which indicates that the designation of "outsider" is flexible and can occur within a group (a group of hunters) as well as across groups (across geographic communities). Further, respondents' motivation to intervene often hinged on the protection of a fellow community member from detrimental outcomes such as prison time or monetary fines. Several respondents in Pu Mat also stated they would intervene to stop someone from laying snares in the protected area because they feared cattle would accidently be injured in a snare.

Table 4. Respondent willingness to intervene based on location, offender activity, and type of offender.

	HQN		Pu Mat	
	Outsider (% Yes)	**Community member (% Yes)**	**Outsider (% Yes)**	**Community member (% Yes)**
Entering PA	67%	34%	35%	24%
Laying snares in PA	71%	53%	68%	60%
Exiting PA with bushmeat	46%	31%	57%	42%

Enhancement of effective informal guardianship

The second aim of this paper was to examine how community members could be mobilized to become effective informal guardians. In our sample, rationales for non-intervention in the prevention of snaring included "it's not my job," "it's not my duty," and "I don't have the authority to act." Compounding this perceived ineffectiveness of intervention was the fear that confronting a hunter could lead to a) physical injury from an altercation, especially if the guardian was alone and b) social-psychological damage to interpersonal relationships or retaliation (This reason for inaction was confirmed in the interviews we conducted with rangers, see Rizzolo et al. 2021). Further, the condition of the poached animal had an impact; several interviewees mentioned that, when they saw someone exit the protected areas with bushmeat, they were more likely to alert rangers if the animal was alive.

Another dynamic that influenced inaction was that communities overall did not have a sense of ownership over the wildlife in the protected areas; most respondents reported that wildlife belonged to the park/reserve and the rangers who patrolled the protected area. Only 3% of interviewees in HQN (and 5% in Pu Mat) reported that wildlife belonged to community members. A belief that wildlife ownership lay with reserves/rangers rather than communities also appeared in perceptions of responsibility to act. At both locations, approximately 60% of respondents said that the protection of wildlife was the responsibility of the reserves and the rangers. In contrast, about 20% of interviewees noted that the community was responsible for wildlife protection.

When asked for potential solutions to reduce illegal snare hunting, community members had multiple suggestions. The integration of these strategies may be particularly valuable for the enhancement of community ownership over wildlife and wildlife crime prevention since they emerged from the perspective of potential informal guardians themselves. Several strategies mentioned are already well-established in

community-based conservation: these included awareness-raising, building infrastructure for communities, improving enforcement, increasing penalties for non-compliance, and providing resources (such as technical expertise and funding) for alternative livelihoods. However, there were also unique responses that could inform site-specific solutions. These strategies are congruent with principles of situational crime prevention, or SCP (Table 5), which indicates that they could be integrated with informal guardianship into a comprehensive SCP framework.

Table 5. Community-generated wildlife crime prevention strategies by situational crime prevention principle.

Increase incentives for informal guardianship
• Provide community loans to develop non-forest-based economic opportunities
• Provide opportunities for communities to learn forest protection
Increase risks for offenders
• Conduct focused surveillance of suspected hunters
• Establish a "quick reaction" team composed of community members and people from other provinces
• Focus enforcement on middlemen (include investigations of restaurants)
• Install cameras to monitor the forest
Remove excuses for offenders
• Alert conscience: have officials criticize offenders
• Establish village-chosen rules about hunting punishments
• Integrate a hunting prohibition into village conventions
• Make ownership of snares (and/or precursor materials) a strict liability offense
Reduce rewards for offenders
• Deny benefits: refuse identification papers to people caught hunting
• Focus fines on recidivists

Integrating formal and informal guardianship to maximize deterrence and limit displacement

The final aim of this research was to gain information on how formal and informal guardianship can be combined to maximize deterrence of illegal snaring. Most interviewees reported that they would only be deterred by harsher punishments that were likely or very likely to occur within one week of the crime. About half of the respondents (57% in HQN and 45% in Pu Mat) reported that it was likely or very likely that they would be apprehended if they snare hunted in the protected area. However, the likelihood of being caught was seen as having an element of randomness rather than certainty; comments such as "only the unlucky ones get caught" were not uncommon.

When asked what factors would deter them most from snare hunting, participants mentioned both people and punishment. The strongest potential deterrent on snare hunting was rangers (i.e., formal guardians) with 81% of interviewees in HQN (and 96% in Pu Mat) noting that being caught by rangers would deter them most from snare hunting. However, when asked about what currently stops them from snare hunting (in reality rather than theory), responses were different. Although rangers had

a strong deterrent effect, with 49 to 59% of respondents listing them as a current deterrent, these numbers were not as high as the percentage of interviewees who said they "would" be deterred by rangers. In Pu Mat, approximately one-fifth of respondents noted that the Frontier Army currently stops them from snare hunting, but that number was much lower in HQN.

In terms of punishment, legal sanctions, rather than extralegal sanctions, were viewed as most effective. Extralegal sanctions such as social shame (e.g., officials criticizing offenders) and confiscations of hunting tools did not seem to deter respondents much. However, in HQN only, the economic-based sanction of withholding shared village economic benefits, such as funds received from the PFES program (see Viollaz and Gore 2019), was ranked as the most efficient deterrent after prison sentences and fines. In both HQN and Pu Mat, the most persuasive punishments were prison sentences and then fines. In HQN, fines that ranged from 500,000 to 10,000,000 VND were mentioned as strong deterrents (mode: 11,000,000; mean: about 8,900,000). For Pu Mat, the suggested value of these fines was higher and ranged from 1,000,000 to 21,000,000 VND (mode: 4,000,000; mean: about 4,700,000).

Respondents were asked about relationships between rangers/other formal guardians and communities, with the results suggesting that tensions are mild. 20% of interviewees in HQN (27% in Pu Mat) reported tension between community members (including hunters) and rangers. In HQN, 8% of respondents reported tension between communities and forest guards, with no such tensions in Pu Mat. Although there were a few instances where hunters reported resentment of rangers for the confiscation of bushmeat or snares, overall, there was respect for rangers as well as a healthy amount of fear of ranger authority, a good sign for deterrence.

Community members did note strategies of crime displacement in which they avoided rangers through displacing their hunting activities either temporally or geographically (Table 6).

Table 6. Strategies used for displacement of snare hunting.

Displacement strategy	HQN (% responded in affirmative)	Pu Mat (% responded in affirmative)
Avoid ranger stations/send spotters ahead to gauge ranger presence/gain advance information on ranger patrols	22	45
Go to the forest late at night or in the early morning when rangers do not patrol	21	17
Go to the forest secretly and actively hide any trace of their presence	30	5
Use alternative paths not used by rangers	16	23

These displacement strategies differed between sites, with avoidance of ranger stations the most frequent in Pu Mat and hiding traces of one's presence the most common in HQN. In HQN, 42% of respondents (and 39% in Pu Mat) had knowledge of where and/or when rangers patrolled on a regular basis.

Discussion

Conditions for informal guardianship

Crime prevention is essential for addressing the severe defaunation impacts of snares in Viet Nam. Informal guardianship is one underutilized technique to enhance communities' participation in crime prevention and build upon and complement existing formal guardianship. Our results indicate both the detriments and benefits of social bonds in terms of illegal snaring prevention. The informal guardian's role in the community, and whether the offender was a member of the community, both had a strong effect on willingness to intervene. Although social bonds between an informal guardian and offender can serve as an obstacle to intervention, as close community bonds mean that there are social costs to reporting each other, they can also function as an incentive. Respondents' motivation to intervene often hinged on the protection of a fellow community member from detrimental outcomes such as prison time or monetary fines. This indicates a level of care for neighbors (but is not necessarily a sign of homogeneity between villagers, as care for community members can extend across diverse populations within the village). This community-mindedness implies a will to leverage social capital to ensure better outcomes for the village as a whole. Thus, "protection" of the offender from the potential outcomes of his crime could be a mechanism for informal guardianship.

These data also reveal important factors in willingness to intervene in wildlife crime. The presence of marked gender effects in intervention preferences (e.g., women in the sample preferred indirect intervention) indicates how gender might structure willingness to intervene. Women were less likely to say they would intervene directly, which suggests a need for gender-specific training and roles within the sphere of informal guardianship. Further, if intervention did not jeopardize the respondent's safety, the threshold for action seemed to be lower for non-local offenders.

Locus of control, or perceived power to effect change in a situation, also constrained informal guardianship. Almost all the respondents who reported neutral or negative answers to intervention noted that they felt they could not effect change. Thus, these respondents might be willing to intervene if they had a stronger belief in their ability to effect change (i.e., a stronger locus of control over crime prevention). Mechanisms to increase locus of control could include an anonymous village reporting system with regular feedback on actions taken in response to tips, or the requirement to return village development fund money publicly if community members are caught hunting (Viollaz and Gore 2019). Both provide a mechanism for reporting and transparency about the enforcement consequences. There are precedents for anonymous reporting systems such as these (e.g., one run by Education for Nature Vietnam) and, in our study areas, the A Roang Community Conservation Group should help facilitate the implementation of this system. Currently, this group is an informal reporting system, as villagers report to group members when hunters are going hunting so that this information can be passed

on to rangers (therefore, the desire to report already exists). To avoid having communities associate these groups with enforcement, an anonymous reporting system would not require their direct involvement. There could be anonymous mechanisms for rangers to receive this information, such as a number to text that rangers routinely check before designing their patrols,

Locus of control could also be enhanced through increasing the community's sense of "ownership" of wildlife (which was low in the sample) and promoting a sense of effectiveness in the prevention of wildlife crime. There appears to be room to enhance internal motivation to develop ownership as, across all sites, 28% of interviewees indicate that it was everyone's or the community's responsibility to protect wildlife. Therefore, there is a baseline level of interest in wildlife (or stake in wildlife) that could be further explored and fostered. This could occur by ensuring that communities receive benefits from wildlife-based activities and/or through implementing wildlife crime prevention strategies that have been generated by community members themselves (see Table 5).

Enhancement of effective informal guardianship

Informal guardianship can be enhanced by leveraging peoples' sense of empowerment, ownership, and their perception of a responsibility to act. Empowerment relates to the belief that one has the capacity and resources to effect change. A lack of ownership can be an obstacle to empowerment, as people often have less incentive to protect what is not "theirs" and in which they do not have a stake. Our results indicate several challenges for enhancement of informal guardianship; several respondents noted reasons for non-intervention such as "it's not my job," "it's not my duty," or "I don't have the authority to act." These answers are congruent with a larger focus in Vietnamese society on collective rather than individual action (Van Dao 2020). However, this also seemed related to a "laissez-faire" orientation towards non-compliant community members. If a community member attended awareness-raising meetings about not hunting protected wildlife and yet continued to hunt, there was little that other members of the community felt they could do to change this behavior. Further, while approximately 60% of respondents said that the protection of wildlife was the responsibility of the reserves and the rangers, a lower percentage (20%) of interviewees noted that the community was responsible for wildlife protection. If there is a widespread belief that responsibility for wildlife protection only lies with formal guardians, this might diffuse the responsibility that residents feel and could lead to the assumption that informal guardianship is unnecessary. However, while 20% is lower than the percentage who cited reserves/rangers as responsible, it represents the potential for growth of informal guardianship.

Since the results indicated that respondents were more likely to intervene when the offender was non-local, it is important to consider the process of labeling someone as an outsider or "othering" (in the cultural-political context of Viet Nam, a Communist country, this may occur through viewing others as acting contrary to the good of Viet Nam, see Van Dao 2020). While, ethically, this process cannot be introduced externally (by conservation organizations, etc.), it is important to be

aware of this social process. The designation of "other" is flexible and people within the community can also be "othered." For example, the results indicate "othering" between hunters. This could potentially be leveraged to defend against outside hunters or even hunters perceived as not needing the income, as several respondents indicated they would be more lenient to hunters they felt lived off the forest and therefore "needed" to hunt compared to others that were not "poor." However, this may only be accomplished if community members buy in to the notion that illegal snare hunting is harmful to the community. Regulations generated by the community may help accomplish this, as these can be more influential than state-based laws on beliefs about the human consequences of poaching (Rizzolo et al. 2017). Although stigmatization can be harmful in certain contexts, it can also have a protective function that promotes conservation and community norms (Rizzolo 2020). In cases where the offender is a community member, there is potential to build on the strong social bonds within the villages to encourage community members to report hunters "for their own good," particularly if the penalties were mild for first offenses but incrementally stronger for repeat offenders.

The enhancement of informal guardianship can also be achieved through the social leverage model suggested in Viollaz and Gore (2019). In this model (Fig. 2), communities include hunting regulations and sanctions in village conventions, including specific actions individuals need to take to stop hunters when they see them. This links to enforcement because fines doled out to hunters are paid from the village development fund like those under PFES. This "stick" (the use of financial fines) has to affect all community members as equally as possible so that an individual's transgressions have consequences for the entire group. Since the consequences affect the group, this can produce social pressure and leverage community social cohesion to force hunters to stop snare hunting, since offenders will lose face in front of other members. The more hunters are sanctioned from the community (and the more that are caught), the more depleted those common development funds become. Community members noted that more development aid and improvements to their standard of living (e.g., vocational training, technical farming assistance, infrastructure like roads and irrigation systems, etc.) were a priority. The aim would be to link a reduction in snaring to better standards of living and increased snaring to visible and practical communal losses.

This "stick" could be combined with a "carrot" or soft approach such as the work of the A Roang Community Conservation Group near Thua Tien Hue Saola Reserve. This group's youth identified hunters who were not dissuaded by general awareness raising efforts and enforcement, then built relationships with them by helping with key needs like seed acquisition and planting or fixing up their houses. During these regular interactions with hunters and their families, the youths talked about their conservation work and beliefs. After four to six months of regular interaction, the hunters felt uncomfortable continuing to hunt whilst receiving help from group members. They tended to gradually stop hunting, with group members continuing to check in on them after they had done so (Viollaz and Gore 2019).

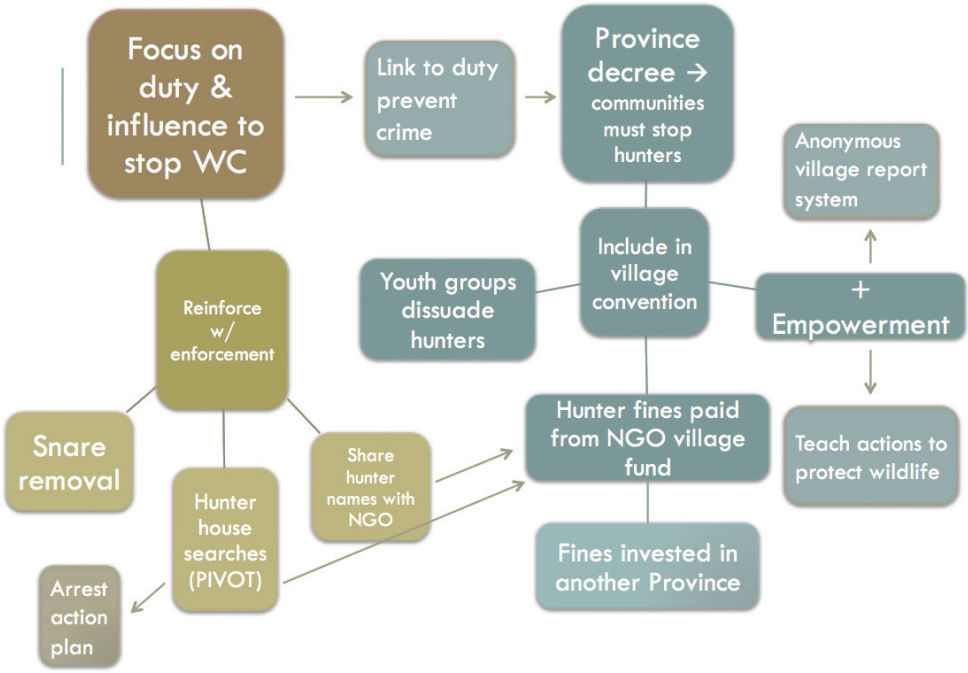

Figure 2. Model of how the formal guardianship (left/yellow) and informal guardianship (right/green) techniques discussed complement one another.

Integrating formal and informal guardianship to maximize deterrence and limit displacement

This paper has also illustrated the challenges and opportunities for integrating formal and informal guardianship. Formal patrols are conducted by conservation organizations near HQN and Pu Mat. However, for punishment to function as a crime deterrent, it must be swift, certain, and severe (Moreto and Gau 2017). Since prison sentences in these areas tend not be certain or swift (Young 2017), immediate fines are perhaps the more effective on-the-ground deterrent.

There appears to be an "opening" for informal guardians to add to formal guardianship, as the number of respondents who listed rangers as a current deterrent was not as high as the percentage of interviewees who said they "would" be deterred by rangers. It may be that the perceived threat of rangers is larger than their actual effect, if, for example, rangers regularly let offenders off with a warning instead of arrest. While there are laws that indicate when an arrest should be made, some rangers use a large degree of discretion on when to apply the rules. This tends to be due to the numerous obstacles that rangers experience in their vocation, which can lead to low motivation and/or capacity to enforce regulations (see Rizzolo et al. 2021). In these situations, informal guardians could step in and deter possible offenders in alternative ways to complement rangers' efforts or both types of guardians could work together to target specific

threats, like middlemen or wildlife traffickers/traders. This makes sense given rangers tended not to blame communities for snaring but rather found middlemen culpable for provoking tensions between them and communities (Rizzolo et al. 2021) and focusing enforcement on middlemen was also a community-generated crime prevention strategy (Table 5). Such integration would also counterbalance conservation's focus on formal guardianship (e.g., state/ranger "ownership" of wildlife and the responsibilities of rangers to protect wildlife), which, for some respondents, served as an impediment to the notion that they should protect wildlife themselves as informal guardians. Further, our results indicate that displacement is possible to achieve with minimal effort, particularly if community members share this knowledge with one another openly and hunters use this information to facilitate illicit behavior. The prevalence of displacement strategies suggests the utility of complementing formal guardianship (e.g., ranger patrols) with informal guardianship, as the latter is particularly effective at combatting displacement (Hollis-Peel et al. 2011).

One of the challenges of formal guardianship is lack of ranger motivation. For formal and informal guardianship to be well-integrated, both "prongs" must function effectively (see Fig. 2). One way to strengthen formal guardianship in low-motivation environments is to target key crime facilitation locations that are easier to access and more geographically circumscribed than the forest itself, a technique known as Place Network Investigations (PNI) in criminology (see Madensen et al. 2017; Hammer 2020, and for its application to wildlife crime Viollaz and Gore 2019). For example, rangers could visit key storage locations like hunters' homes to catch them as they return from hunting or could regularly visit "bia hois" and other corrupting spots that encourage illegal behavior like bushmeat consumption. Through making rangers' work less resource-intensive by focusing their efforts on specific locations, yet more visible and efficient, you encourage willingness to intervene to prevent wildlife crime on the part of communities (Viollaz and Gore 2019). Demonstrations that wildlife crime is taken seriously by formal guardians can enhance informal guardianship because community members often want to protect their neighbors from serious consequences of poaching such as fines and prison sentences (Viollaz and Gore 2019).

Despite the contributions in this paper, it has inevitable limitations that leave open avenues for future research. The generalizability of the results is constrained by the relatively small sample size and geographical similarities between the study sites. As mentioned, research on informal guardianship in Southeast Asia is nascent, and future research could expand this work to other sites in this geographic region. Further, in studies of illegal behavior, there is always the risk of bias in answers due to social desirability bias or other factors (Rizzolo 2020). In this work, there was the additional hurdle of cultural barriers, such as the need to translate interviewees' responses into English and the potential presence of specialized language (or argot) among snare hunters. While it is important to acknowledge these factors as potential limitations, the researchers attempted to minimize the effects of these obstacles through the creation of a cross-cultural research team, the use of best practice survey research methods, and the triangulation of community interview data with other sources of data within the larger project (such as ranger interviews, see Rizzolo et al. 2021).

Conclusion

The potential of informal guardianship is still under-utilized in the context of wildlife crime (UNODC 2020) but presents an opportunity for addressing the issue of illegal snaring. Snares are a pervasive contributor to defaunation yet the volume of snares, the often-anonymous nature of the offense, and the ease by which snares can be replaced means that the effects of formal detection and enforcement are often limited. The number of snares in protected areas in South-East Asia is staggering (Belecky and Gray 2020) and their removal occurs at high cost and with limited results. Combined with the additional challenges of ranger resources and motivation, there is a lack of capacity to address snaring.

Informal guardianship is part of a larger focus on wildlife crime prevention. This is particularly valuable in the context of snares because, once the animal has been killed or maimed, there are negative conservation ramifications regardless of whether or not the offender is apprehended (Gray et al. 2018). Our results confirmed the importance of intervention prior to when wildlife is killed. If the animal had already been killed, there was the perception that there was less rationale to intervene (e.g., the hunter may as well keep the animal) and that, since the animal could not be salvaged, the witness might as well ask for some meat and share in the benefits.

This emphasizes the need for frameworks, such as situational crime prevention or SCP (Clarke 1997), that discourage the decision to offend through escalation of the costs and reduction of the benefits of crime. SCP fits within the larger approach of problem-oriented policing, which is crime and place specific, encourages prevention, and promotes combinations of strategies (Lemieux and Pickles 2020). The wildlife crime prevention strategies generated by the interviewees corresponded to principles of SCP, which demonstrates how an integrated approach that utilizes informal guardianship and various deterrence strategies could be successful within Viet Nam's cultural and geographic context.

While informal guardianship is an important tool for increasing risks to offenders, and preventing crime before it occurs, research on this approach has focused on traditional crime (e.g., property theft) in an urban and Western context (Hollis-Peel et al. 2013). This paper has applied informal guardianship to a new context: illegal wildlife snaring in Viet Nam. Our results indicate that the conditions for informal guardianship exist in our study sites. Further, this work has helped map the potential and the obstacles for the use of informal guardianship and the integration of formal and informal guardianship. Such information is essential for both the protection of wildlife and the enhancement of community involvement in the prevention of illegal snare hunting and conservation initiatives in Viet Nam.

Acknowledgements

The Ministry of Agriculture and Rural Development, The Viet Nam Forest Department, Forest Protection Department of Nghe An, Hue and Quang Nam provinces kindly facilitated and issued permission (Permit numbers 46/SNgV-LS Quang Nam,

132/SNgV-LS.NVNONN Hue, 01-LDDTBXH-QXN). The views expressed herein are those of the author(s) and do not necessarily reflect the views of the United Nations. The authors would like to thank all those individuals who worked on the research planning phases of the broader project, in equal order: Nguyen Thi Bich Lien, Ho Thi Le Nhu, Ho Van Po, Tran Thi Thuy Dung, Nguyen Thi Yen, Le Thanh Phuc, Le Thi Hang, Le Thi Chien, Le Thi Kim Dung, and Do Van Thoai. This work was supported by Ministry of Education and Training Grant B2018-TDV-11, Global Wildlife Conservation – Michigan State University Grant No GWC 5118-0115. MG was supported by National Science Foundation Grant CMMI-1935451 and IIS-2039951.

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Appendix A: Interview questionnaire

Date:	**Interviewee #:**
Interviewer #:	Others present:
Field site #:	Verbal consent granted: Yes No
Age:	Gender: Male Female
Ethnic group:	Hunter: Yes No
Religion:	
Hunter type:	Confirmation of hunter type by:
Subsistence	1. Village elder or headman: Yes No
Inside Professional	2. Forest ranger or NGO personnel: Yes No
Outside Professional	3. Community member: Yes No
	4. Interviewer's opinion: Yes No

1. What do you do for a living?
2. How long have you lived in this village? (Answer in years; use village event timeline to pinpoint specific time period)
3. Do you take part in any work in the village? (Check all that apply)
   Communist party
   - □ Village management board
   - □ Elder
   - □ Union (specify union)
   - □ None
4. Where have you snare hunted in the last hunting cycle (1 year) (Show map of use area without protected area boundary and ask respondent to place beans where they went to hunt, then take picture of map)
   4a. INTERVIEWER QUESTION, DO NOT ASK THE INTERVIEWEE. Based on the locations where the interviewee has placed beans do they hunt in the protected area?
   - □ Yes
   - □ No

The remaining questions of this questionnaire only refer to snare hunting in the protected area. QUESTIONS 5–8 should only be asked if the person is a hunter hunting in the protected area (if you answered yes to question 4a.).

5. How many animals have you caught in the last hunting cycle (1 year)?
6. What are the reasons for you to hunt with snares (if non-hunter "what are the reasons for people to hunt with snares?")? (Only prompt interviewee if they cannot come up with reasons on their own: for personal consumption/to sell/to relax/other.)

Reason for hunting	Ranking (1 – most common, 2 – others)

7. Would you like your child to learn to hunt if other alternative jobs were possible?
   - □ Yes
   - □ No

   7a. Why?
8. Can you describe the process when someone goes hunting? How do they prepare, what do they do once in the forest, once they've caught an animal, and to return home with their catch?
   (Steps: Prep – Entering PA – Staying in PA – Set snare – Exit PA – Prep return pickup – check snare – Exit PA with meat – Sell or consume meat)
9. At present, how many people hunt with snares in your community? Of those how many come from the outside?
   Inside hunters:

Outside hunters:

10. What stops you from snare hunting?
    10a. What do you do to avoid these difficulties?
11. If you saw someone stealing a buffalo from your community, how would you react?
    11a. How likely would you be to intervene to prevent the theft?
        □ Very likely
        □ Likely
        □ Neutral (50/50 chance)
        □ Unlikely
        □ Very unlikely
    11b. How would you intervene?
        □ Directly
        □ Indirectly
        □ It depends on the situation

    Please explain why you would intervene that way.
12. How likely is it for someone to be caught hunting with snares in the protected area?
    □ Very likely
    □ Likely
    □ Neutral (50/50 chance)
    □ Unlikely
    □ Very unlikely
13. Here are some scenarios of people catching a hunter snaring. What would happen in each instance, and would that hunter stop hunting as a result of being punished? Note: Write down types of punishment and use the respective number as shown below:

Scenario	Punishment
1. An outsider/community member is found snaring inside PA by a government ranger.	1
	2
	3
2. An outsider/community member is found snaring inside PA by an international organization (WWF or FFI) forest guard.	1
	2
	3
3. An outsider/community member is found snaring inside PA by a community elder or headman.	1
	2
	3

How quickly would this punishment occur?

(0) Immediately (1) Within 1 week (2) Within 1 month
(3) Within 6 months (4) Within 1 year (5) Never

How severe do you think this punishment is?

(1) Not severe enough (2) Adequate (3) Severe

How likely is this person to stop hunting as a result of being punished?

(1) Very likely (2) Likely (3) Neutral (50/50 chance)
(4) Unlikely (5) Very unlikely

14. Which set of characteristics would deter you the most from snare hunting? (Pick 1 option in each category)

Person catching
□ Government ranger
□ Elder/headman
□ International organization (WWF/FFI)
□ Forest guard
Punishment
□ Confiscation of meat
□ Confiscation of snares
□ Prison sentence
□ Village ban
□ Shaming by community
□ Shaming by elder/headman
□ Refusal to give household certificate
□ Withholding of shared village economic benefits
□ Large fine (if picked ask how much?)
□ Withholding of political favors (entrance to army, educational opportunities)
Time to punishment
□ Immediately
□ Within one week
□ Within one month
□ Within 6 months
□ Within 1 year
Certainty of punishment
□ Very likely
□ Likely
□ Neutral (50/50 chance)
□ Unlikely
□ Very unlikely

15. Do you feel you know your village and your neighbors well enough to know if something is not right in your community (including the protected area)?
    - □ Yes
    - □ No

    15a. If someone in your village goes hunting illegally, do you know that?
    - □ Yes
    - □ No

16. Generally, how many days are you NOT present in your village per month?
17. Who does the wildlife belong to?

    17a. Whose responsibility is it to protect the wildlife?

    17b. Are there traditional customs and actions that contribute to protecting wildlife?
18. How willing are you to intervene if you see someone hunting with snares inside the protected area?
    - □ Very likely
    - □ Likely
    - □ Neutral (50/50 chance)

□ Unlikely
□ Very unlikely

19. Do you think you could stop the person from snare hunting inside the protected area? Why or why not?
□ Yes
□ No
19a. Why do you think that?
20. Here are some scenarios about times when you might see someone doing different types of things. What would you do in each situation?

Scenario	**How would you intervene to stop the person?** (Include reason for no intervention)	**Why would you intervene that way?**
1.a. If you saw an outsider entering the PA, would you take any action?		
1.b. What if this was a community member?		
2.a. If you saw an outsider laying a snare inside the park, would you take any action?		
2.b. If it was a community member?		
3.a. If you saw an outsider exiting the park with bushmeat, would you take any action?		
3.b. If it was a community member?		

21. What is the relationship like between government wildlife officials and the community in your area? Why?
21a. Do you know the area where they patrol and when? How do you know this?
22. What is the relationship like between international organization (WWF/FFI) forest guards and the community since they started working as forest guards? Why?
22a. Do you know the area where they patrol and when? How do you know this?
23. Is hunting in the protected area legal or illegal? (This question must be asked at the end of the interview!)
□ Legal
□ Illegal
23a. If they break this law, do they feel shame (i.e., lose self-respect)?
□ Yes
□ No
24. If it was your responsibility to stop hunting with snares in the protected area, what would you do and how? (Make sure not to ask only about changes to the law)

Nature Conservation 48: 149–160 (2022)
doi: 10.3897/natureconservation.48.68243
https://natureconservation.pensoft.net

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What shapes the mammal species poaching in protected areas: biophysical or anthropogenic factors? A case study in Pendjari Biosphere Reserve

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Academic editor: Enrico Di Minin | Received 4 May 2021 | Accepted 16 December 2021 | Published 25 May 2022

http://zoobank.org/CBB1A5CF-9926-4E5F-89FA-2E0061F4D7CE

Citation: Degbelo FG, Djagoun CAMS, Toyi SSM, Padonou EA, Kouton M, Gichohi N, Muruthi P, Sinsin B (2022) What shapes the mammal species poaching in protected areas: biophysical or anthropogenic factors? A case study in Pendjari Biosphere Reserve. Nature Conservation 48: 149–160. https://doi.org/10.3897/natureconservation.48.68243

Abstract

Understanding what shapes the mammal species poaching in protected areas is critical to developing targeted management strategies for reducing poaching. We collected the data for poaching incidents on the GPS coordinates from 2011 to 2017 to map poaching incidents in the Pendjari Biosphere Reserve. Poaching incidents were then related to environmental and anthropogenic variables using regression analyses. The study shows that poaching is more concentrated along the main river in the Pendjari National Park. Only nearest distance to the main river significantly predicted the location of high poaching incidents. These results could be used as the starting point by the park managers when planning the anti-poaching activities.

Keywords

Benin, GIS layers, Pendjari Biosphere Reserve, poaching incidents, wildlife

Introduction

The major driver of large mammal species population decreasing in Africa is poaching (Bouche et al. 2011; Maisels et al. 2013; Chase et al. 2016). A recent work on large mammal species population declines in Africa's protected areas (PAs) by Craigie et al.

(2010) has provided a first continent-wide assessment, warning on the decline between 1970 and 2005 of about 59% in population abundance of large mammal species, with a collapse of 85% when considering only western Africa. The authors have targeted the limitation in financial resources and personnel to protect the species as the major drivers increasing the poaching in many PAs despite the conservation role deserved (Thouless et al. 2008).

Some decades ago, an investigation on a continent-wide scale about elephant anti-poaching efficacy recommended a range of USD 50–200 per km^2 annually to protect them in their natural ranges in Africa (Jachmann and Billiouw 1997). Regarding the personnel, a minimum of one park ranger for every 24 km^2 of PAs is recommended if effective patrolling and policing is to be realised (Jachmann and Billiouw 1997). Again, several PAs including Pendjari in West Africa cannot meet this staffing level obligation and is, thus, unlikely to allocate more funds towards wildlife conservation (Lindsey et al. 2014; Tranquilli et al. 2014). Therefore, it is important to explore strategies that involve more efficient use of the limited available resources. By assessing what shapes the wildlife poaching in PAs, important insights about the characteristics of particular PAs where wildlife is more vulnerable to human-induced death can be generated, which can help guide effective deployment of anti-poaching patrols. Anti-poaching patrols in Western Africa PAs are challenging because of limited resources and the expansive area of the parks that limits the effectiveness of patrols by park rangers. It is in the interests of conservation to investigate some research questions such as: (i) what are the spatial patterns of large mammal species poaching in the Pendjari Biosphere Reserve (PBR); (ii) what mammal species are of concern and (iii) What are the relationships between observed patterns of poaching and anthropogenic, biophysical variables? Answering those questions will be useful to facilitate the development of effective and optimally targeted management strategies to reduce poaching in a critical poaching hotspot, such as in protected area landscape under the conditions of resource limitation.

Efforts to assess the drivers of large mammal species poaching in PAs have highlighted several factors. It shown that areas with extensive forest cover, with more challenging patrolling and enforcement than in open savannah, show, for example, a top level of poaching in elephant (Burn et al. 2011). Distance to water is targeted as the primary environmental factor influencing the density of large mammal species population in PAs (Redfern et al. 2003; Djagoun et al. 2014) and driving the poacher's interest on those sites.

This study aims to describe the most wildlife species poached between 2011 and 2017, together with the spatial distribution of large mammal species poaching incidents in the Pendjari's landscape and to identify the biophysical and human factors that determine the distribution of poaching incidents. We use an explicit spatial modelling approach to quantify the relative contribution of multiple potential factors described in literature as a priori to explain the poaching incidents. We hypothesised that poacher sites would be associated with: (1) water availability, (2) accessibility (roads and topography) and (3) proximity of human settlements and land uses.

Methods

Study site

The Pendjari Biosphere Reserve successively classified as a National Forest, a partial Wildlife Reserve of the Pendjari loop and a National Park, acquired the labels of Biosphere Reserve in 1986, RAMSAR site in 2007 and now considered as a UNESCO World Heritage Site. The PBR is in the Atakora Province, north-western Benin. It is situated at 10°30' to 11°30'N; 0°50' to 2°00'E (Fig. 1). It was declared as a Game Reserve in 1954 and upgraded to National Park in 1961. It is nowadays composed of a strictly-protected core area (the Pendjari National Park with 2,660 km^2) and three hunting zones in west and south sides (1,971 km^2). In the north and east parts, the River Pendjari forms a natural border of the PBR. Its northern boundary is also the country's border with Burkina Faso (Delvingt et al. 1989; MAB-UNESCO 1990). Climate is tropical, between late October and early April, there is a seven-month dry period. The PBR is located in the Sudanian zone with a single dry season from November until March and one wet season from April/May to October.

Annual mean precipitation is 1000 mm, with 60% falling between July and September (Sinsin et al. 2002).The mean annual temperature is 27 °C (Verschuren 1988). The mean daily temperatures for the period 1979 to 2010 peaked from March to

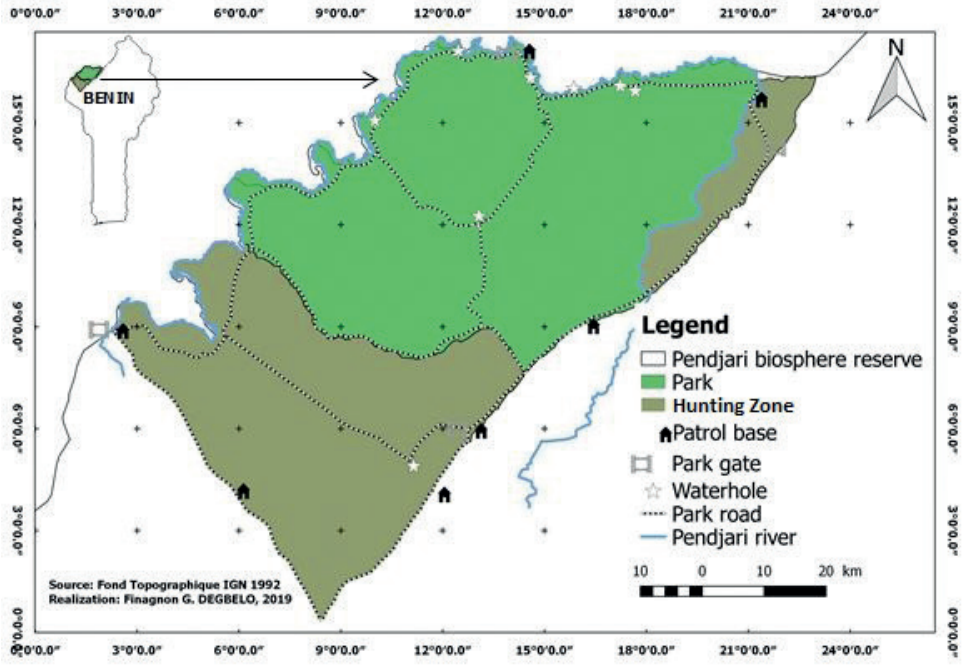

Figure 1. Map of Pendjari Biosphere Reserve.

April and reached a low value during November to January. Mean daily minimum and maximum temperatures were 19.2 °C and 30.9 °C for the cold month (December), 24 °C and 40 °C for the hottest month (March). The impact of the rainfall regime is very important because the rate of wildlife sighting through line transects is negatively correlated with the green cover in the top level in the rainy season (Strahler 1998). The rainy season allows the filling of many small and large waterholes in the centre of the National Park, namely Tiabiga, Fogou, Mondri, Diwouni, Yangouali and Bali. During the dry season, many waterholes attract a variety of animals, with large mammal species searching for water. The dominant vegetation type is savannah interspersed by some patches of dry forests with deciduous trees (Sokpon et al. 2001). Savannah vegetation is burnt every year for management to provide fresh pasture to herbivores that dominate the Reserve, provide visibility to wildlife tourists and hunters who visit mostly during the dry season and to avoid uncontrolled mid- or late dry season fires that are often started by poachers to camouflage illegal activities or that originate from surrounding villages.

Data collection

Mammal species poaching data from 2011 to 2017 was obtained from the PBR anti-poaching database, which has been developed over the years during routine daily patrols by rangers. The anti-poaching patrols are randomly distributed and the poaching incidents observed during these surveillance patrols are geo-referenced. Data on poached mammal species were then entered into an EXCEL spreadsheet. Each record had the following fields: X and Y coordinates (using Universal Transverse Mercator), date of registration and name of the place where the poaching incident occurred. A total of 279 poaching points were recorded by the guards for the period. Of these locations, 228 points fell inside the PBR. A total 303 mammal individuals were poached in the PBR from 2011 to 2017.

The locations of ranger patrol bases and park gates were obtained by visiting the sites and recording their locations using a Global Positioning System (GPS). The geographic coordinates of the Park boundaries, roads, rivers and waterholes were obtained from an ecological biomonitoring service (Fig. 2).

Poaching incidents locations, as well as locations of ranger patrol bases and park gates, have been projected on to the Pendjari Biosphere Reserve map. Then, with the ARCGIS 9.2 software, the biophysical variables, such as the closest distance to waterholes (NDis_Wh); the distance closest to the main river (NDis_Rv) and anthropogenic variables, such as the closest distance to the park gate (NDis_Pg); closest distance to the patrol base (NDis_PaB); distance closest to the park road (NDis_Pr) and the distance closest to the park boundary (NDis_Pb) were measured for each of the identified poaching sites.

The measured values of each of the variables, cited above, were used to model the distribution of poached species within the RBP.

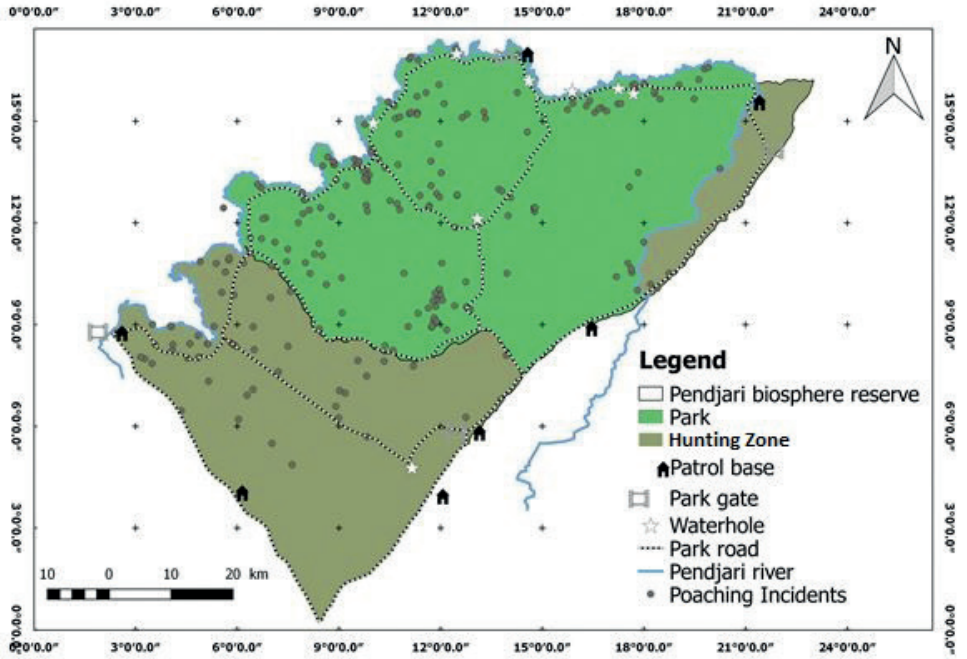

Figure 2. GIS layers generated showing locations of poaching incidents in Pendjari Biosphere Reserve.

Data analysis

Estimating the large game poached species according to the management zones

We performed all analyses in the statistical programme R v. 3.5.2 (R Core Team 2018). The percentage of most poached species in Pendjari Biosphere Reserve and the most threats to the wildlife were calculated and presented as bar plots. A Student's t-test was performed to assess the difference in the number of poached species per management zone.

Biophysical and human factors predicting the poaching areas

To assess multi-collinearity into the variables, variance inflation factors (VIF) were examined. This parameter estimates how much the variance of a coefficient is increased due to a linear relationship with other predictors (Barnier et al. 2019). A VIF value less than or equal to 4 indicates that there is no multi-collinearity between variables (Rakotomalala 2015). The package "car" (Fox et al. 2012) was used and applied on the linear model to calculate the variance inflation factors (VIF) with the live function. After analysis, we found for the variables that waterholes (VIF = 1.173), park_road (VIF = 1.074), main_river (VIF = 2.185), park_boundary (VIF = 2.110) are considered for the model because their VIF is less than 4. For the other two variables, namely

park_gate (VIF = 8.741) and patrol_base (VIF = 9.049773), the VIFs were higher than 4, therefore are not used in the model. The model's odds ratios were calculated using "questionr" R package.

Spatial distribution pattern of poaching incidents around waterholes

We also estimated spatial distribution patterns of poaching sites in the PBR and around waterholes from 2011 to 2017. We used functions from the spatstat package to calculate K statistics to model Monte Carlo envelopes (999 simulations) to test the complete spatial randomness (CSR) hypothesis (Baddeley and Turner 2004) in R software (R Core Team 2018). Ripley's K distribution above the upper confidence interval indicates clustering, between confidence intervals indicates a random spatial pattern and below the lower confidence interval indicates a regularly distributed pattern.

Results

Estimating the large game poached species according to the management zones

Figure 3 shows that 14 ungulate species was regularly poached in the PBR. We notice a statistical difference in the poaching average of individuals across the difference zones (t = - 9.4525, P-value < 0.05) with a much more poaching activities in the Park. Only red flanked duiker (*Cephalophus rufilatus*) were poached exclusively in the Park. Amongst these species, *Loxodonta africana*, *Ourebia Ourebi*, *Papio anubis* and *Hippopotamus amphibious* were mostly poached in the Park, respectively in the proportion of 88%, 85%, 76% and 70%.

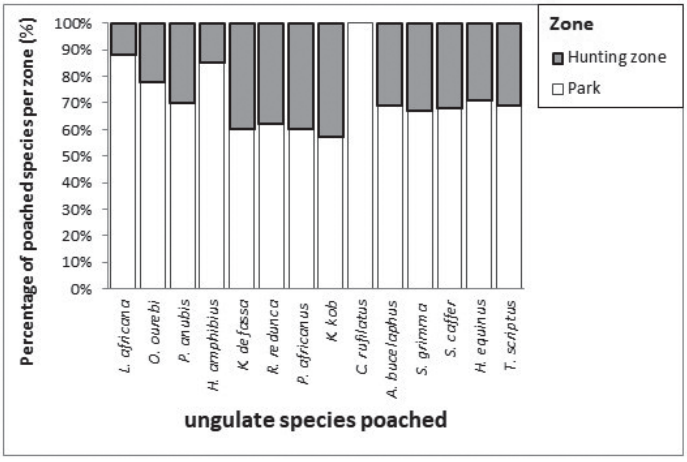

Figure 3. Hunted ungulate's species percentage per zone.

Biophysical and human factors predicting the poaching areas

Only the variable "nearest distance to the main river" (p-value = 0.041) contributed significantly in explaining the poaching incidents (Table 1). When the nearest distance to the main river increases by one kilometre, the probability of having a high poaching site increases by 0.8%. Other variables, such as nearest distance to the Patrol Base; nearest distance to the Waterholes; nearest distance to the Park Road, have not contributed to explain the poaching incidents.

Spatial distribution pattern of poaching incidents around waterholes

Figure 4 shows the Ripley's K-function analysis performed on the mammal species poaching site in the PBR. This showed significant random patterns up to 26 km and clusters beyond that distance (Fig. 4A). However clustered patterns of the mammal species poaching sites are revealed extensively in the PBR when considering the waterholes (Fig. 4B).

Discussion

Our study helped to assess the biophysical or anthropogenic factors predicting the mammal species poaching areas in the western African PAs using poaching data over

Table 1. Results of the generalised linear model between poaching incidents locations and predictors.

Parameters	Signs	Coef.	Odds ratio	Robust SE	Pr(> |z|)
(Intercept)	-	1.53945	0.19178	2.212365	0.68366
NDis_Wh	-	0.97483	0.92650	0.025397	0.29987
NDis_Rv	-	0.89485	0.79059	0.06097	0.0410*
NDis_Pr	-	0.98852	0.89355	0.026652	0.63059
NDis_PaB	+	1.01985	0.95631	0.037449	0.55261

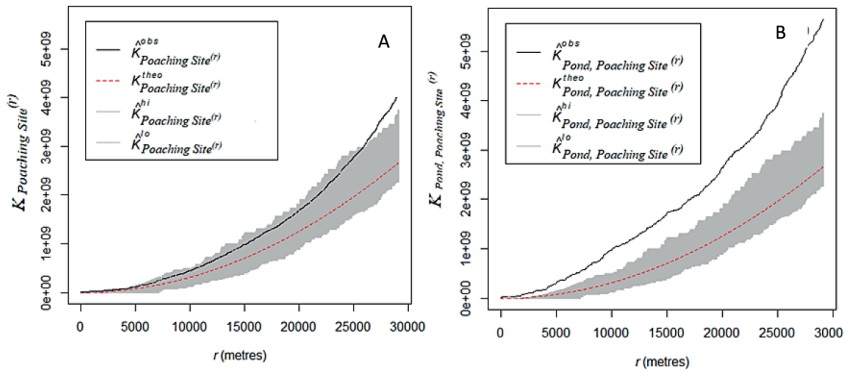

Figure 4. Ripley's K-function analysis performed on the mammal species poaching site in the PBR: **A** all poaching sites **B** poaching site vs. waterholes.

seven years (2011–2017). In total, we found 14 ungulate species to be mostly poached in the PBR with more poaching occurring in the Park. The study further highlighted the nearest distance to the river as the main driver of the poaching incidents in the PBR. Ripley's K-function analysis, performed on the mammal species poaching site in the PBR, showed significant random patterns up to 26 km and clusters beyond that. However clustered patterns of the mammal species poaching sites are found extensively in the PBR when considering the waterholes.

The results about the most species poached confirm the work of many authors, notably, Van Schuylenbergh (2009) who affirms that poaching appears even more profitable in areas where animal species are protected, such as National Parks which fuel a profitable trade in trophies. The species poached in the Ebo Forest Reserve at Cameroon are dominated by ungulates (Fuashi et al. 2019). In addition, commercial animal poaching, such as for ungulates, occurs in the areas with greatest densities (Maingi et al. 2012). Our results also confirm those that are reported by the Wildlife Census (PAPE 2013), which revealed that, in the protected area, poaching is the most common form of human pressure. However, the pressure varies according to the zoning within the protected areas. Knowledge of the spatial distribution of poaching activities is very important for managers. It will allow them to bring together all the resources suited to the areas of concentration (Treves et al. 2011).

The poaching incidents mapping within Pendjari Biosphere Reserve shows that the high poaching areas are near to the main river and far from the park road, waterholes and patrol base. The high incidence of poaching along the river leads them to conduct repeated main patrols in the area. These results support previous research, such as Sibanda et al. (2016) research, who reported an increased activity of poachers near waterholes in the mid-Zambezi Valley, as did the Maingi et al. (2012) work on spatio-temporal models of elephant poaching in south-eastern Kenya which shows that hotspots of poaching were identified in areas with higher densities of waterholes, rivers and streams. The same information was reported in the "Emergency Action Plan Against Poaching (PAULAB)" by UEMOA (2014), which indicates that poaching occurs mainly along the main river. The proximity to water is the most important factor leading to poaching (Kuiper et al. 2020). This observation is explained because populations of wildlife species are mainly concentrated at the waterholes (Djagoun et al. 2014; Rich et al. 2019). Non-commercial animal poaching was associated with high-wetness areas and near rivers, possibly because there is a need for a certain amount of woody vegetation to conceal snares and create funnels for wildlife to move into the snare. Roads, waterholes and patrol bases play an important role in the fight against poaching. The history of wildlife conservation has shown that the presence of agents in living areas would reduce poaching. This study showed a negative relationship between the presence of patrol officers and illegal activities, as increasing the distance from living bases, roads and waterholes increases the likelihood of poaching. However, part of our results contradict the work of Sibanda et al. (2016) and Maingi et al. (2012) which also showed that poaching activities are observed close to roads. The proximity to major residential areas and roadways has a strong influence on poaching incidents

(Nieman et al. 2019). This statement is confirmed by Subedi and Subedi (2017) who showed that poaching incidents are located near the roads and also agree with us by specifying that these zones are located far from the bases of life. The claim that poaching areas are close to roads can be explained by the fact that, in the natural resource management scenario, roads facilitate the movement of people to previously-inaccessible areas. If the area is easier to reach, then the poacher can go to the area in a short time to poach (Toxopeus 1996; Subedi and Subedi 2017). The road infrastructures in these areas, therefore, facilitate poachers' access. However, in the context of the PBR, these roads could reduce the presence of poaching acts, but can allow the transport of game according to the fact that those roads are often used by rangers for patrols and, on the other hand, could be useful for the poacher to escape while patrols were not present in the area at the time of poaching. The poaching is concentrated in the main river because the rangers have not sufficient means to intensify the patrols in the area. Despite the efforts of the rangers, this plague continues to increase day by day.

Knowledge of the spatial distribution of poaching activities is very important for managers. It will allow them to bring together all the resources suited to the areas of concentration (Treves et al. 2011).

Our work has the particularity of having used spatial analysis methods to understand poacher's behaviour according to the biophysical or anthropogenic factors in the PAs. This study finding is important for ranger deployment and demonstrates the value of a full spatio-temporal analysis. This study could, therefore, form the basis for the formulation of future hypotheses which test the effect of poaching on the wildlife conservation in PAs. Future studies exploring similar hypotheses should include seasonality to understand the temporal patterns of poaching. This will allow better generalisations regarding the incidence of poaching according to the different seasons. Additionally, our findings represent a baseline for any further evaluation of the new management system put in place in PBR by the African Park Network since 2017.

Acknowledgements

The authors thank the African Wildlife Foundation for funding this study, all the staff members of the Pendjari National Park management office and the Laboratory of Applied Ecology (LEA) for providing help and assistance during the fieldwork.

The author(s) disclosed receipt of the following financial support for the research, authorship and/or publication of this article: this study received financial support from the African Wildlife Foundation.

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Nature Conservation 48: 161–184 (2022)
doi: 10.3897/natureconservation.48.77899
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RESEARCH ARTICLE

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Natura 2000-sites: Legal requirements for agricultural and forestry land-use

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Academic editor: Jukka Similä | Received 11 November 2021 | Accepted 21 February 2022 | Published 9 June 2022

http://zoobank.org/DF854E81-D248-4FB1-B26E-13C700899C55

Citation: Möckel S (2022) Natura 2000-sites: Legal requirements for agricultural and forestry land-use. Nature Conservation 48: 161–184. https://doi.org/10.3897/natureconservation.48.77899

Abstract

The agricultural and forestry use of land does not only mark large parts of the European Union, but also a significant share of land within the European ecological network Natura 2000. Member States, therefore, try to exempt as far as possible these land uses from the protection regime of Natura 2000-sites. However, at the same time, Member States latest reports on habitats and wild species of Community importance indicate that, in particular, the intensification of agriculture and forestry in recent decades has made it more difficult to improve conservation statuses or even worsened them. Hence, the aim of this article is to examine in detail the extent to which the protection regime of Article 6(2 and 3) Habitats Directive is applicable to land-use in agriculture and forestry. In this context, of particular relevance is the question of when the use of land in agriculture and forestry or individual management measures within and near Natura 2000-sites are projects for which an appropriate assessment is necessary before implementation; and which, in the case that significant adverse effects to a site cannot be ruled out, are permitted only under the reasons for exemption given in Article 6(4) Habitats Directive. The analysis includes the case law of the European Court of Justice, as well as decisions of the German Federal Constitutional Court and the German Federal Administrative Court.

Keywords

Agriculture, appropriate assessment, Birds Directive, forestry, Habitats Directive, Natura 2000

1. Introduction

The Habitats Directive (HD) 1992/43/EEC and the Birds Directive 2009/147/EC (formerly Directive 79/409/EEC) are the most important EU instruments[1] in achieving the biodiversity objectives of the Convention on Biodiversity and other international treaties on nature conservation[2] and the Natura 2000-network is the centrepiece of the EU's biodiversity strategy.[3] Under Article 2(2) HD and the 9th recital of the Birds Directive, Member States are to maintain or restore favourable conservation statuses for habitat types and species of Community interest in all of the EU's biogeographical regions.

According to the Member States status reports for the Habitats Directive reporting period 2013 to 2018, out of the 1,389 species in the EU's biogeographical regions, only 27 percent display a good conservation status, while three quarters of the 233 habitat types display a poor or bad status.[4] Compared to the previous reporting period,[5] the status improved for six percent of the species and nine percent of the habitat types. In more than a third of habitat types, a further deterioration was recorded. In the case of European bird species and migratory bird species, less than half of all species have a good population status, while almost 40 percent display a poor or bad status.[6]

The main cause for the often recorded unfavourable conservation statuses is the unsustainable use of land in agriculture and forestry and, in particular, the intensification of agriculture and the loss of high nature value farmlands via the conversion of natural grasslands and pastures into arable land, the greater use of fertilisers and pesticides, the removal of small landscape features and the drainage of areas, as well as the reduction of old-growth forests, clear-cutting and the removal of dead or old trees.[7] Nutrients and pesticides are also brought into Natura 2000-sites from the outside via the air or water and the critical loads for eutrophication have been exceeded widely in many regions of Europe.[8] Overall, the use of land in agriculture and forestry is of particular relevance for achieving favourable conservation statuses for habitats and wild species within and outside of Natura 2000-sites.[9]

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1 European Commission 2016, p. 10 et sqq.; Milieu, IEEP and ICF 2016, p. 14 et sqq.

2 Especially the Ramsar Convention on Wetlands of International Importance (1971), the Bonn Convention on the Conservation of Migratory Species of Wild Animals (1979) and the Bern Convention on the Conservation of European Wildlife and Natural Habitats (1979).

3 European Commission 2020, p. 4 sqq.; ECA 2017, p. 12.

4 EEA 2020, p. 35.

5 European Commission 2015b.

6 EEA 2020, p. 14.

7 Cf. EEA 2020, p. 70 et sqq.; Anderson/Mammides Ambio 2020, 1963 et sqq.

8 Cf. EEA 2017; Kattwinkel et al. Ecological Applications 2011 et sqq.; BVL 2020; Hofmann et al. 2020; Brühl et al. Scientific Reports 2021.

9 EEA 2020, p. 14, 34, 69–87; Tucker et al. 2019, p. 73 et sqq. Cf. Beckmann et al. Global Change Biology 2019 et sqq.; Bowler et al. Long-term declines of European insectivorous bird populations and potential causes 2019; Hallmann et al. PLOS One 2017 et sqq.

'About 40% of the total land area of the EU-28 is agricultural land (Eurostat 2020a). Results show that current agricultural practices are by far the most dominant driver affecting habitats and species (...). However, the richness and abundance of biodiversity associated with agricultural habitats is strongly correlated with the degree of modification (e.g. draining, ploughing) and the intensification of management (e.g. use of fertilisers, irrigation and pesticides). Extensive agricultural management creates and maintains semi-natural habitats with a diverse fauna and flora. Since the 1950s, however, the intensification and specialisation of the agricultural sector has increasingly contributed to ongoing biodiversity loss. Changes in agricultural management are, thus, the most frequently reported type of pressure'.[10]

In addition to the EU-wide provisions on species protection, Natura 2000-sites are the most important instrument for safeguarding favourable conservation statuses. In 2021, the European Commission registered a total of 26,935 Natura 2000-sites in the EU27 (excluding the United Kingdom), which together comprise around 17.5 percent of Europe's land area (764,222 km²) and 450,752 km² expanses of water.[11] The terrestrial share is particularly high in south-eastern European countries (Slovenia 38 percent, Croatia 37 percent, Bulgaria 35 percent, Slovakia 30 percent, Cyprus 30 percent and Greece 27 percent) and in Spain (27 percent). Overall, this is the world's largest ecological network of protected sites.

The conservation status of habitat types and species of Community interest in the Natura 2000-sites is, on average, significantly better than outside of the network.[12] There is also less intensification or land use change in Natura 2000-sites.[13] Nonetheless, there is still considerable room for improvement and, in particular, the impact on entire biogeographical regions is still too small.[14] The dominant change in land-use within the Natura 2000-network was the conversion of natural grassland, transitional woodland-shrub and mosaic farmland into arable land, forests or pastures.[15] Many of the land use changes and intensifications are possible because the protection regime of Art. 6(2 and 3) HD is insufficiently applied in the Member States, especially for agricultural and forestry land.[16] The type and intensity of land-use in agriculture and forestry within the Natura 2000-sites are, therefore, also of great relevance for the protection of Natura 2000-sites and the conservation objectives, as these are predominantly not wilderness areas. Managed forests and grassland, as well as arable land, make up over 60 percent of the terrestrial area of the Natura 2000-network.[17] Furthermore,

10 EEA 2020, p. 73.

11 European Commission 2021b.

12 EEA 2020, p. 118 et sqq.

13 Hermoso/Morán-Ordóñez/Brotons Landscape Ecology 2018 1454 et sqq.; Anderson/Mammides Ambio 2020, 1965 et seq.

14 Cf. EEA 2020, p. 122 et sqq.; Pellissier et al. Conservation Biology 2020; Rada et al. Diversity and Distributions 2019; WWF 2017.

15 Hermoso/Morán-Ordóñez/Brotons Landscape Ecology 2018.

16 Cf. ECA 2017, p. 33 et sqq.; Milieu, IEEP and ICF 2016, p. 102 et sqq.; Sundseth/Roth 2013.

17 EEA 2020, p. 112 et sqq.

Natura 2000-sites are also affected by surrounding land use and related emissions (e.g. nitrogen and pesticide emissions).[18]

The following article examines the extent to which the protection regime of Article 6 (2 and 3) HD is applicable to land-use in agriculture and forestry within or in the vicinity of Natura 2000-sites. For this purpose, section 2 gives an overview of the legal protection regime of the Natura 2000-network. Then, in section 3, it is discussed whether and when the use of land in agriculture and forestry or individual management measures are projects that require assessment and which requirements apply to land-use that does not require assessment. A brief overview of the requirements for an appropriate assessment is given in section 4. Section 5 then looks into the question of when, in the event of an established incompatibility, land-use or individual management measures would be permitted as an exception.

2. The legal protection regime of Natura 2000

Based on the Special Protection Areas (SPA) under Article 4 Birds Directive and Sites of Community Importance (SCI) under Article 3 and 4 HD, the European Union (EU) and its Member States created the ecological network Natura 2000. The Network serves to protect 231 habitat types and 450 wild species, which have been identified as conservation priorities.[19] According to the European Court of Justice (ECJ),[20] it protects European natural heritage, which is why it requires a high level of protection, a high level of accuracy, clarity and certainty in implementing the two Directives, as well as a high level of necessary monitoring and control in the respective Member States.[21] This concerns not only the protection regime for Natura 2000-sites, but also the obligations for species protection according to Article 12 HD and Article 5 Birds Directive, which apply in and outside Natura 2000-sites.[22]

With their regulations, the Member States must ensure that the requirements of the HD and Birds Directive are fully observed, when it comes to official decisions.[23]

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18 Kelleghan et al. Atmospheric Environment 2021; Liess et al. Water Research 2021; Buijs/Mantingh 2020; Hofmann et al. 2020; Tegner Anker et al. JEEPL 2019; Sánchez-Bayo/Wyckhuys Biological Conservation 2019; Möckel JEEPL 2019.

19 European Commission 2015b, p. 3.

20 All ECJ decisions can be located based on their case number and can be freely accessed under: http://curia.europa.eu/juris/recherche.jsf?language=en.

21 ECJ, adjudication of 20.10.2005 – C-6/04, margin number 21, 25 et seq. and Ls. 1; adjudication of 10.1.2006 – C-98/03, margin number 59; adjudication of 10.5.2007 – C-508/04, margin number 58 et sqq., 73, 79, 98.

22 Cf. the recent jurisdiction of ECJ, adjudication of 28.10.2021 – C-357/20; adjudication of 4.3.2021 – C-473/19 and C-474/19; adjudication of 2.7.2020 – C-477/19; adjudication of 11.6.2020 – C-88/19; adjudication of 10.10.2019 – C-674/17.

23 Cf. ECJ, adjudication of 12.7.2007 – C-507/04, margin number 137, 162, 280 et seq., 287.

No legal uncertainties must remain[24] and a Directive-compliant implementation, as well as an effective and timely control by the competent authorities, must be ensured.[25] Imprecise, indeterminate or ambiguous regulations that allow leeway for an interpretation contradicting European law and/or can only be brought into line with the two Directives by means of a Directive-conform interpretation are, therefore, not sufficient.[26] According to the ECJ and contrary to general ECJ case law,[27] inadequate or even contradicting legal regulations are inapplicable and not merely to be interpreted in conformity with the Directives.[28] In the absence of (correct) national regulations, the competent authorities and courts have to apply directly the relevant regulations of the Directives, provided that they are sufficiently specific and unconditional.[29] This does neither require the subjective rights of individuals nor does an indirect burden on third parties preclude direct application, since the latter follows solely from a State's obligation to implement under Article 288(3) TFEU.[30]

For Natura 2000-sites, in 1992, the Member States of the European Union agreed an ambitious protection regime in Article 6 HD,[31] which has, however, not yet been fully implemented in the Member States.[32] Under Article 4 HD and Article 4 Birds Directive, the SCI and SPA listed are legally binding and have to be established with the necessary protection regulations and specific conservation objectives for the respective protected habitat types, as well as animal and plant species, including those species characteristic of the habitat types concerned. In addition, under Article 6(1) HD, management plans are to be drawn up for each Natura 2000-site and the necessary maintenance and development measures have to be determined and implemented.

Furthermore, under Article 6(2) HD, the Member States are obliged to avert deterioration and disruptions with the help of preventative protective measures. In addition, under Article 6(3) HD, all projects and plans that, individually or in combination with other plans and projects, could significantly adversely affect a Natura 2000-site, must be checked for their compatibility with the conservation objectives of the Natura

24 ECJ, adjudication of 20.10. 2005 – C-6/04, margin number 37; adjudication of 10.5.2007 – C-508/04, margin number 79.

25 Cf. ECJ, adjudication of 26.1.2012 – C-192/11, margin number 46.

26 Cf. ECJ, adjudication of 10.5.2007 – C-508/04, margin number 79; adjudication of 20.10.2005 – C-6/04, margin number 103 et seq.; adjudication of 10.1.2006 – C-98/03, margin number 77 et seq.

27 E.g. ECJ, adjudication of 26.9.2000 – C-262/97, margin number 40; adjudication of 04.2.1988 – 157/86, margin number 11.

28 ECJ, adjudication of 10.5.2007 – C-508/04, margin number 79 et seq.

29 ECJ, adjudication of 22.6.1989 – 103/88, margin number 29 et seqq.; adjudication of 11.08.1995 – C-431/92, margin number 24 et seq., 37 et seqq.; adjudication of 7.1.2004 – C-201/02, margin number 64 et seq.

30 ECJ, adjudication of 11.8.1995 – C-431/92, margin number 24 et seq., 37 et sqq.; adjudication of 7.1.2004 – C-201/02, margin number 64 et seq.; adjudication of 10.1.2006 – C-98/03, margin number 40 et seq.

31 In detail European Commission 2018a.

32 ECA 2017; European Commission 2016; Milieu, IEEP and ICF 2016; Sundseth/Roth 2013.

2000-site before they are approved and implemented.[33] If, on the basis of objective circumstances and the best available scientific knowledge, a significant adverse effect to a Natura 2000-site cannot be ruled out with certainty,[34] the proposed projects and plans must be prohibited by the respective Member States.[35] Under European law, these projects come, therefore, with a reservation of assessment and prohibition. In practice, however, even after two decades, there are considerable problems in carrying out full appropriate assessments for all relevant projects and plans in the member states.[36]

3. The necessity of Natura 2000-appropriate assessments for land-use in agriculture and forestry

'Any plan or project not directly connected with or necessary to the management of the site, but likely to have a significant effect thereon, either individually or in combination with other plans or projects, shall be subject to appropriate assessment of its implications for the site in view of the site's conservation objectives' (Article 6(3)(1) HD). For the use of land in agriculture and forestry, it has, therefore, to be elucidated when measures are part of the conservation and restoration management within the meaning of Article 6(1) HD or projects that are subject to review.[37] This distinction is particularly difficult for semi-natural habitats like forest or grassland.[38]

3.1. Exempted conservation and restoration measures

As an exception, the scope of such site management measures is to be interpreted narrowly and only given, if they are intended to promote the respective conservation objectives in the area.[39] After all, only in these cases can significant adverse effects be generally excluded. The measure must be carried out by the site administration or on their behalf. Other

33 ECJ, adjudication of 14.10.2010 – C-226/08, Rn 48 et seq.; adjudication of 24.11.2011 – C-404/09, margin number 125, 174; adjudication of 17.4.2018 – C-441/17, margin number 148; adjudication of 29.7.2019 – C-411/17, margin number 122–145. In detail European Commission 2021a; European Commission 2018a; Garcia-Ureta Journal of Property, Planning and Environmental Law 2018 and Möckel Nature Conservation 2017c.

34 On the difficulties of certainty Balias Journal of International Wildlife Law and Policy 2018; Sobotta Journal for Nature Conservation 2018, p. 263.

35 Settled case law ECJ, adjudication of 29.7.2019 – C-411/17, margin number 134; adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 59–73; adjudication of 17.4.2018 – C-441/17, margin number 117–120, 179; adjudication of 14.1.2016 – C-399/14, margin number 43 et seq., 48 et seq.; adjudication of 11.9.2012 – C-43/10, margin number 111 et sqq.; adjudication of 7.9.2004 – C-127/02, margin number 41–49, 56–59.

36 ECA 2017, p. 33 et sqq.; Milieu, IEEP and ICF 2016, p. 102 et sqq.; Sundseth/Roth 2013. cf. also Balias Journal of International Wildlife Law and Policy 2018.

37 Cf. ECJ, adjudication of 17.4.2018 – C-441/17, margin number 122–127.

38 In detail Sobotta Journal for Nature Conservation 2018.

39 ECJ, adjudication of 4.4.2010 – C-241/08, margin number 50–56.

measures that serve other goals in the area (e.g. economic interests or promote tourism), on the other hand, are not covered by the exemption, as considerable impairment of the conservation objectives cannot generally be ruled out here.[40] The same applies to site-defining management measures, such as the land-use in agriculture and forestry or hunting, when they are not carried out on behalf of the site administration, but down to private economic interests[41] or if such measures have not been designed as part of an integrated management plan in order to ensure that they are compatible with the respective conservation objectives.[42] Not every management plan is a plan within the meaning of Art. 6(1) HD.[43]

3.2. Definition of 'Project'

Neither the HD nor the Birds Directive define what a project is. According to the ECJ, it is, nevertheless, a term under European law that is not defined by the Member States individually.[44] Due to the high level of protection, the Court adopts a wider interpretation of the term 'project'.[45] Here, the Court gets its bearing from the definition of the term in Article 1(2)(lit. a) Directive 2011/92 on the assessment of the effects of certain public and private projects on the environment (EIA Directive); according to which, in addition to the erecting of structures or other facilities, projects include all other interventions into nature and the landscape including those geared towards the extraction of mineral resources.[46] In its decision of 7 November 2018, the ECJ emphasised that the definition of a project under habitat law also includes non-physical interventions (e.g. substantive or acoustic emissions).[47] Projects are, therefore, not only measures for which approval or notification requirements exist.[48] The only decisive factor is the

40 European Commission 2021a, p. 13; European Commission 2018a p. 16–20, 38; Sobotta Journal for Nature Conservation 2018, p. 263; Epiney/Gammenthaler 2009, p. 93 et seq.

41 ECJ, adjudication of 4.4.2010 – C-241/08, margin number 39, 56; adjudication of 13.6.2002 – C-117/00, margin number 22–33.

42 ECJ adjudication of 17.4.2018 – C-441/17, margin number 122–127; European Commission 2021a, p. 13 et seq.; European Commission 2015a. Cf. Cortina/Boggia Journal of Environmental Management 2014 et sqq.

43 For the forest management plan in Białowieska, see ECJ adjudication of 17.4.2018 – C-441/17, margin number 122–127. With regard to the practical synergies Trentanovi et al. Journal for Nature Conservation 2018.

44 ECJ, adjudication of 10.1.2006 – C-98/03.

45 ECJ, adjudication of 7.9.2004 – C-127/02, margin number 23 et sqq., 34; adjudication of 10.1.2006 – C-98/03, margin number 41–45; adjudication of 26.5.2011 –C-538/09, margin number 45; adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 61–73; adjudication of 29.7.2019 – C-411/17, margin number 122 et sqq.

46 Settled case law ECJ, adjudication of 14.1.2010 – C-226/08, margin number 38 with further references; adjudication of 10.1.2006 – C-98/03, margin number 40 et seq.; adjudication of 7.9.2004 – C-127/02, margin number 24 et sqq.

47 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 50–73 and 1st recital.

48 ECJ, adjudication of 14.1.2016 – C-399/14, margin number 68 et seq.; adjudication of 10.1.2006 – C-98/03, margin number 40 et sqq.

potential impact of human activities on Natura 2000-sites.[49] Due to impact interplay, projects and plans outside of a Natura 2000-site may also require an appropriate assessment, if they impair the conservation objectives in the area (e.g. the input of emissions via the air, water, runoff or due to fragmentation effects).[50] The indirect effects of a project are also relevant to the assessment (e.g. via respective food chains)[51] provided that they can be clearly assigned, since the appropriate assessment does not require evidence of causality, since it is rather sufficient that a project probably exerts significant effects.[52]

The term 'project' thus includes all activities in or in the vicinity of Natura 2000-sites that are likely to have significant effect on a Natura 2000-site individually or in conjunction with other plans and projects.[53] The nature, size and location of a project are only of significance when assessing impact in the appropriate assessment. Whether there is a project that needs to be assessed must be established as part of an official screening.[54] The summary assessment of the likelihood of significant impairment must be made on the basis of objective circumstances, taking into account the special characteristics and environmental conditions of the Natura 2000-site concerned.[55]

In Germany, the Federal Administrative Court (BVerwG)[56] raised the question of whether this impact-focused concept of a project requires restricting planned impacts on

49 Cf. ECJ, adjudication of 29.7.2019 – C-411/17, margin number 134 et sqq.; adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 50–73; adjudication of 14.1.2016 – C-399/14, margin number 68 et seq.; adjudication of 10.1.2006 – C-98/03, margin number 40 et sqq. BVerwG, adjudication of 12.11.2014 – 4 C 34.13, margin number 29; adjudication of 19.12.2013 – 4 C 14.12, margin number 28; decision of 18.5.2004 – 7 B 18.04, margin number 24.

50 ECJ, adjudication of 29.7.2019 – C-411/17, margin number 136 et sqq., adjudication of 24.11.2011 – C-404/09, margin number 146 et sqq., 166 et sqq.; adjudication of 11.9.2012 – C-43/10; adjudication of 13.12.2007 – C-418/04, margin number 256 et seq.; adjudication of 20.10.2005 – C-6/04, margin number 34; BVerwG, adjudication of 18.12.2014 – 4 C 35.13, margin number 34, 43 et seq.; adjudication of 28.3.2013 – 9 A 22.11, margin number 84, 88 et seq. On genetic exchange ECJ, adjudication of 24.11.2011 – C-404/09, Slg. 2011, I-11853 margin number 146 et sqq., 166 et sqq.; BVerwG, adjudication of 14.4.2010 – 9 A 5.08, margin number 32–34; decision of 23.1.2015 – 7 VR 6.14, margin number 16; adjudication of 14.7.2011 – 9 A 12.10, margin number 93.

51 Cf. e.g. BVerwG, adjudication of 9.7.2009 – 4 C 12.07, margin number 11.

52 Cf. ECJ, adjudication of 10.11.2016 – C-504/14, margin number 29; adjudication of 14.1.2016 – C-399/14, margin number 42; adjudication of 14.1.2016 – C-141/14, margin number 58; adjudication of 24.11.2011 – C-404/09, margin number 142.

53 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 67. In detail European Commission 2018a, p. 35, 39 et sqq. and Möckel Nature Conservation 2017d.

54 ECJ, adjudication of 7.9.2004 – C-127/02, margin number 43 et seq.; adjudication of 26.5.2011 – C-538/09, margin number 39; adjudication of 21.7.2011 – C-2/10, margin number 41 et seq.; adjudication of 12.4.2018 – C-323/17, margin number 31–40. Cf. European Commission 2021a, p. 11 et sqq.

55 ECJ, adjudication of 7.9.2004 – C-127/02, margin number 44, 49; adjudication of 26.5.2011 – C-538/09, margin number 39; BVerwG, adjudication of 18.12.2014 – 4 C 35.13, margin number 33, 48; adjudication of 17.1.2007 – 9 A 20.05, margin number 61.

56 From 2002 onwards, BVerwG decisions can be located, based on their case number and can be freely accessed under: https://www.bverwg.de/suche. References to the locations of earlier decisions are provided in this article.

protected areas to distinguish Article 6(3) HD from the general prohibition of changes and disturbances in Article 6(2) HD.[57] According to the BVerwG, activities that are subject to a mandatory assessment would only exist if there was an opportunity to check whether they were compatible with the conservation objectives of the protected area on the basis of plans, concepts or established practice, which implies that ad hoc measures do not constitute projects.[58] In this way and contrary to the impact-focused approach, the Court includes considerations of practicality when determining what a project is — a line of argument that has not been recognised in the ECJ's case law up to now.[59]

3.3. Land-use in agriculture and forestry as projects

In their law or regulations for protected sites, Member States have repeatedly exempted land-use in agriculture and forestry from the appropriate assessment or did not classify it as a project within the meaning of Article 6(3) in their justifications or a rule in general.[60] The ECJ has opposed this practice repeatedly.[61] Following the ECJ, the intended general anticipated exemption of certain activities and projects is only permitted if the provisions justifying an exemption ensure systematically and in each individual case that the exempted project and activities do not cause disruptions that could significantly impair the protected areas.[62] For this purpose, the mere existence of general protective regulations is not sufficient, if they only reduce, but not exclude, the risk of considerable impairment.[63] In particular, the assessment of cumulative effects of other projects and plans[64] required under Article 6(3) HD can hardly be captured fully and adequately by anticipating regulations — which is why their scope is severely limited.[65]

With regard to land-use in agriculture and forestry, the ECJ has repeatedly affirmed its classification as a project and, therefore, the applicability of the appropriate

57 BVerwG, adjudication of 13.4.2013 – 4 C 3.12, margin number 30.

58 BVerwG, adjudication of 8.1.2014 – 9 A 4.13, lead sentence 6 and margin number 55. Confirming BVerwG, decision of 24.3.2015 – 4 BN 32.13, margin number 34.

59 In detail Möckel Nature Conservation 2017d, p. 42 et seq.

60 So in Germany until 2007: since its condemnation by the ECJ (adjudication of 10.1.2006 – C-98/03, margin number 39–45), the Federal Government indicated in its explanation to the amended Federal Nature Conservation Act that land-use in agriculture, forestry and fishing are usually not a project under the rules of good subject specific practice (BT-Drs. 16/6780, p. 13; BT-Drs. 16/12274, p. 65). See also Schumacher/Fischer-Hüftle, BNatSchG, 2021, p. 885 e seq., 943; Mühlenberg/Möckel/Sattler 2021.

61 ECJ, adjudication of 26.5.2011 – C-538/09, margin number 41 et sqq.; adjudication of 4.3.2010 – C-241/08, margin number 36; adjudication of 10.1.2006 – C-98/03, margin number 41.

62 Cf. ECJ, adjudication of 26.5.2011 – C-538/09, margin number 41 et sqq.; adjudication of 4.4.2010 – C-241/08, Slg. 2010, I-1697 margin number 36; adjudication of 10.1.2006 – C-98/03, margin number 41.

63 ECJ, adjudication of 26.5.2011 – C-538/09, margin number 63; Cf. ECJ, adjudication of 4.4.2010 – C-241/08, margin number 39, 56. Cf. Möckel Nature Conservation 2017d, p. 47 et seq.; Schoukens JEEPL 2014.

64 See Möckel Nature Conservation 2017a, p. 65–68.

65 In detail Möckel Nature Conservation 2017d, p. 47 et seq.

assessment. In 2018, on the basis of a preliminary Dutch inquiry,[66] the ECJ detailed that agricultural management measures, such as fertilisation or grazing with their physical effects (e.g. fencing, feeding and treading), but also non-physical inputs (e.g. nutrients), can be projects, unless it can be ruled out with certainty that they can significantly impair Natura 2000-sites individually or in conjunction with other projects and plans.[67] The same applies to pesticide use in agriculture and forestry, the choice and intensity of agricultural and forestry crops or animal husbandry,[68] soil cultivation measures (e.g. grassland ploughing) including drainage and irrigation[69] and the manner of agricultural, forestry and fishery harvesting and extraction,[70] as well as plans that foresee such agricultural or forestry measures without being an integrated part of the site management plan.[71,72] The court reaffirmed its strict requirements for anticipatory exemptions, even when combined with compensatory mitigation measures as part of a programmatic approach.[73] It also stressed that mere average parameters are not enough for an exemption[74] and that unfavourable conservation statuses — as exist in many habitats and species of Community importance (see 2) — limit the admissibility of projects and plans.[75]

According to the Court of Justice, only recurrent agricultural land uses do not require an appropriate assessment that were permitted under national law before the Directive came into force (1992) and that continue to be classified as one and the same project.[76]

3.4. The categorisation of recurring measures as projects requiring assessment

According to the ECJ, the protective purpose of the HD requires that each intervention must be assessed separately as a matter of principle.[77] However, the ECJ recognises that

66 On the background, hopes and legal consequences of the ECJ decision in Denmark, see Kegge/Drahmann JEEPL 2020; Schoukens JEEPL 2018.

67 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 59–73, 113–120.

68 Cf. ECJ, adjudication of 18.12.2007 – C-186/06, margin number 26 et sqq.; 13.6.2002 – C-117/00, margin number 22–33; adjudication of 25.11.1999 – C-96/98, margin number 29, 45 et seq.

69 Cf. ECJ, adjudication of 25.11.1999 – C-96/98, margin number 29, 45 et seq.

70 Cf. ECJ, adjudication of 4.3.2021 – C-473/19 and C-474/19; adjudication of 4.4.2010 – C-241/08, margin number 39, 56; adjudication of 7.9.2004 – C-127/02, margin number 27.

71 ECJ, adjudication of 17.4.2018 – C-441/17, margin number 106–193. Cf. Sobotta Journal for Nature Conservation 2018, p. 263.

72 Similarly European Commission 2015a, p. 76 et sqq.; European Commission 2012, 2012, p. 45 et sqq.; European Commission 2014, p. 29 et seq.

73 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 92–104, 116–120. Cf. Squintani RECIEL 2020.

74 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 118 et seq.

75 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 103.

76 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 86. Cf. ECJ; adjudication of 29.7.2019 – C-411/17, margin number 127–129; adjudication of 14.1.2010 – C-226/08, marginal number 47.

77 ECJ, adjudication of 14.1.2010 – C-226/08, margin number 37–41; adjudication of 7.9.2004 – C-127/02, margin number 28.

recurring measures can be regarded as a single project within the meaning of Article 6(3) Habitats Directive if, due to their nature or the circumstances in which they are carried out, they can be regarded as a uniform measure, in particular, when they always follow the same purpose.[78] In the case of a uniform project, Natura 2000-sites are protected during the entire duration of the project by the general ban on deterioration and disturbance under Article 6(2) HD, which, according to the ECJ, requires a level of protection that is similar to Article 6(3) HD[79] and obliges Member States not merely to be reactive, but also to take preventative and repressive action.[80] While the permanent operation of facilities constitutes only one project despite long periods of time (e.g. a highway), in the case of intermittent, but regularly recurring measures (e.g. maintenance measures;[81] agricultural, forestry or fishing management measures[82]), the question arises about whether this is a coherent project or several successive individual projects and whether a Member State may respectively adopt procedural rules in this regard.

In principle, in view of the impact-focussed concept of a project and the existing dynamics in the Natura 2000-site concerned, like the development of species and climate change or changing emission situations, new interventions have to be assessed on the basis of the situation at the time of the planned measures, even if the latter are comparable with old interventions in terms of type and extent.[83]

Using the example of agricultural fertilisation and grazing, the ECJ showed, in 2018, that land-use in agriculture and forestry can only be classified under certain conditions as permanent projects for which no Natura 2000-appropriate assessment is necessary.[84] The Dutch court had asked whether a long-term project is constituted, when the quantities and techniques of fertilisation 'which themselves evolve over time as a result of technical and regulatory changes and, secondly, nitrogen deposition caused by the application of fertilisers has not, overall, increased after the entry into force of that provision'.[85] While the second aspect is irrelevant according to the ECJ, since it does not exclude the possibility that nitrogen deposits have increased in the individual protected areas, the changes in management are of definite relevance.[86]

Specifically, according to the ECJ, the obligation to assess land-use in agriculture and forestry depends on whether it has had a common purpose, since the HD came into force in 1992 and whether the location and circumstances of its practices (amongst others

78 ECJ, adjudication of 14.1.2010 – C-226/08, margin number 47–51; adjudication of 29.7.2019 – C-411/17, margin number 127–129.

79 Settled case law ECJ, adjudication of 14.1.2016 – C-399/14, margin number 52 with further references.

80 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 85, 134; adjudication of 13.12.2007 – C-418/04, margin number 208, 217.

81 ECJ, adjudication of 14.1.2010 – C-226/08, margin number 35 et sqq.

82 ECJ, adjudication of 7.9.2004 – C-127/02, margin number 21 et sqq.; adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 74–86.

83 Cf. ECJ, adjudication of 9.9.2020 – C-254/19, margin number 59; adjudication of 14.1.2016 – C-399/14, margin number 58–62; BVerwG, adjudication of 12.3.2008 – 9 A 3.06, margin number 89.

84 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 74–86.

85 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 81.

86 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 82–84.

types, measures and techniques) have remained the same.[87] However, particularly in the case of agricultural land use, cultivation has changed significantly in the last 28 years due to, for example, new crops, varieties, livestock species and machinery, changes in the management of fertilisers and crop protection and legal changes; up to 2017, intensification measures have increased yields significantly.[88] Studies around the world show that agricultural intensification is regularly accompanied by the deterioration of the affected biotopes and wild species.[89] In addition, many of the pesticides, biocidal products and their active ingredients used today were only developed, approved and placed on the market after 1992, which is why reference to unchanged management cannot be made.

Agricultural and forestry land-use within and in close vicinity to a Natura 2000-site that is categorised as a permanent project does not require a Natura 2000-appropriate assessment; however, such activities fall under Article 6(2) HD, according to which Member States must ensure that the latter do not cause disturbances that can significantly impair the objectives of the HD and the conservation objectives of the Natura 2000-site concerned.[90] The probability or risk of an impairment is sufficient here. The ECJ considered it to be sufficient if national law allows the monitoring and controlling of facilities whose activities cause inputs or interference in Natura 2000-sites and if the possibility of sanctioning exists that can go as far as the closure of the respective facility.[91] Since Article 6(2) HD provides the same level of protection as Article 6(3) HD,[92] the protection of integrity may also require a retrospective appropriate assessment, based on the current situation, if area protection cannot be achieved otherwise or if an exception under Article 6(4) HD is meant to be granted despite significant impairments.[93]

If individual management measures in agricultural and forestry land-uses change within and in the vicinity of Natura 2000-sites (e.g. due to different fertilisers or pesticides respectively or their quantities), this constitutes a new project,[94] which is to be subjected to a screening and, if necessary, a comprehensive appropriate assessment by the competent nature conservation authority.

3.5. De minimis thresholds

The Dutch court had also asked the ECJ whether 'certain projects which do not exceed a certain threshold value or a certain limit value in terms of nitrogen deposition from the requirement for individual approval, since the cumulative effects of all the plans

87 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 86.

88 European Commission 2018b, p. 6 et sqq. For Germany BLE 2017, p. 72; BMEL 2018, p. 13.

89 Beckmann et al. Global Change Biology 2019 et sqq.

90 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 85.

91 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 137 and 7th recital.

92 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 87; adjudication of 14.1.2016 – C-399/14, margin number 52; adjudication of 15.5.2014 – C-521/12, margin number 19.

93 ECJ, adjudication of 14.1.2016 – C-399/14, margin number 33, 42–46, 54–62. Cf. BVerwG, decision of 6.3.2014 – 9 C 6.12, margin number 28, 35, 39.

94 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 86 and 2th recital.

or projects likely to create such deposition were subject in advance to an "appropriate assessment" [of the Programma Aanpak Stikstof 2015–2021] within the meaning of Article 6(3) of that Directive'.[95] In Germany, too, so-called substance- and area-related de minimis thresholds have been established by the authorities[96] and recognised by the Federal Administrative Court (BVerwG)[97] with reference to the proportionality principle; in case these thresholds are not violated, an appropriate assessment is not required provided that the project does not cause any further impairments.[98]

In continuation of its case law on anticipated exemptions, the ECJ has now emphasised that exempting de minimis thresholds must not lead to projects being admitted without an appropriate assessment, although significant negative effects on a Natura 2000-site cannot be ruled out without scientific doubt.[99] National courts must, therefore, thoroughly and comprehensively examine the scientific validity of the threshold values with regard to the requirements under Article 6(3) HD. In particular, it must be assessed whether there is already a risk that below the respective de minimis thresholds, individual projects alone or in conjunction with other plans and projects can cause considerable impairments.[100] Only if there is no reasonable scientific doubt that there will be no significant effects on Natura 2000-sites in each individual case, such exempting de minimis thresholds can be reconciled with Article 6(3) HD. The mere reference to average values is not sufficient in this context, since impairments depend, in particular, on the scope and intensity of activities, the distance between the place of activities and the protected area in question, as well as special conditions.[101]

According to the ECJ, also certain areas of activity or types of facilities cannot be excluded based on, for example, their small size or low cost of activities,[102] even if they have already shaped the area, like for instance, agriculture, forestry, fishing or hunting,[103] or if the observance of conservation objectives has been agreed

95 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 105.

96 Lambrecht/Trautner 2007 and Balla et al. 2013. Cf. Wulfert et al. 2015.

97 Cf. BVerwG, adjudication of 23.4.2014 – 9 A 25.12, margin number 45 m.w.N.; decision of 6.3.2014 – 9 C 6.12, margin number 23; adjudication of 28.3.2013 – 9 A 22.11, margin number 65; adjudication of 6.11.2012 – 9 A 17.11, margin number 62, 93 and lead sentence 3; decision of 5.9.2012 – 7 B 24.12, margin number 7, 12; adjudication of 29.9.2011 – 7 C 21.09, margin number 42–44; adjudication of 12.3.2008 – 9 A 3.06, margin number 124; adjudication of 17.1.2007 – 9 A 20.05, margin number 49 et seq.

98 Sceptical about compliance with the directive Schumacher/Fischer-Hüftle, BNatSchG, 2021, p. 916–922; Möckel Nature Conservation 2017a. Cf. also Tegner Anker et al. JEEPL 2019.

99 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 105–112 and 4th recital.

100 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 111.

101 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 116–119.

102 Cf. ECJ, adjudication of 26.5.2011 – C-538/09, margin number 55 et seq.; adjudication of 21.9.1999 – C-392/96, margin number 66; adjudication of 10.1.2006 – C-98/03, margin number 43 et seq.; adjudication of 4.4.2010 – C-241/08, margin number 31.

103 ECJ, adjudication of C-241/08, Slg. 2010, I-1697 margin number 39, 56; adjudication of 17.4.2018 – C-441/17, margin number 127.

contractually.[104] After all, whether an activity or project has significant negative effects depends not only on its type and extent, but also on the sensitivity and condition of the habitat types and species protected in the respective area, as well as on previous pollution and additional cumulative projects and plans.[105] As a result, even small projects can have significant negative effects. In addition, the overall effect of cumulative impacts on habitats and species does not increase in a linear, but exponential manner due to decreasing resilience.[106]

As a result, the requirements, specified by the ECJ for anticipated exemptions and de minimis thresholds, can only be ensured for individual Natura 2000-sites and, due to the dynamics in the area, only for a limited period of time.

4. Requirements for the Natura 2000-appropriate assessment

The appropriate assessment requires an analysis of the relevant impact factors of a project, as well as a forecast of its negative impacts on the Natura 2000-site in question.[107] Due to the required certainty, uncertainties in the assessment of the potential effects and their significance are at the detriment of the project.[108] According to the ECJ, the assessment must not be incomplete and must contain complete, precise and definitive statements that are suitable to dispel any reasonable scientific doubt as to the effects of the plans or projects proposed in the relevant protected area.[109] Both must be assessed for the potential impacts on the habitat types and species and the related conservation objectives for which the Natura 2000-site has been designated. Furthermore, according to the ECJ, the effects on all HD species and HD habitat types occurring in or outside the area must be listed and discussed, insofar as these effects are likely to impair the conservation objectives of the area.[110]

Whether an agricultural and forestry land-use or an individual management measure can lead to a significant impairment of a Natura 2000-site requires an individual assessment, which depends largely on nature conservation specific findings and assessments of the area and project in question.[111] The assessment must be based on the current scientific state-of-the-art and the best relevant scientific insights, including

104 ECJ, adjudication of C-241/08, Slg. 2010, I-1697 margin number 55.

105 Cf. ECJ, adjudication of 26.5.2011 – C-538/09, margin number 55 et seq.; adjudication of 21.9.1999 – C-392/96, margin number 66. In detail Möckel Nature Conservation 2017a, p. 65–68.

106 Cf. Siviter et al. Nature 2021; Liess et al. Scientific Reports 2016.

107 In detail European Commission 2021a, pp. 25–51; European Commission 2018a, pp. 39–52 and Möckel Nature Conservation 2017a.

108 Explicit ECJ, adjudication of 11.9.2012 – C-43/10, margin number 112.

109 Settled case law ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 98, 117 with further references.

110 ECJ, adjudication of 7.11.2018 – C-461/17, margin number 39.

111 Settled case law BVerwG, decision of 7.2.2011 – 4 B 48.10, margin number 6; adjudication of 12.3.2008 – 9 A 3.06, margin number 68 and adjudication of 17.1.2007 – 9 A 20.05, margin number 43.

generally recognised empirical principles and research methods,[112] although the German BVerwG also sees limits in the obligation to investigate.[113] With regard to substantial inputs, the BVerwG has stated several times that, in principle, any project-related violation of recognised critical load thresholds represents a significant impairment in the case of habitat types or species protected in the area.[114]

5. Exceptional approval despite Incompatibility

In case of an incompatibility, the authorities can approve a project as an exception under Article 6(4) HD, if there are compelling reasons of overriding public interest, no reasonable alternatives exist with regard to 'how' and 'where' to achieve the project and the coherence of the Natura 2000-network is safeguarded by compensation measures.[115] Regarding priority habitat types and species under Article 6(4) subparagraph 2 HD, an exception is only permitted for reasons related to human health and public safety in connection with significant beneficial effects on the environment or after a positive opinion by the Commission. Furthermore, an exemption requires that a complete appropriate assessment and documentation of the same has been carried out beforehand, as knowledge of the effects on the Natura 2000-site and its conservation objectives is an essential prerequisite for the application of Article 6(4) HD.[116]

As a rule for exception, Article 6(4) HD and the requirements specified there must be interpreted strictly.[117] It is necessary to balance project-related impairments to the area against public interests, as well as to assess alternatives and coherence measures specific to nature conservation.[118] A diverging approval is to be made with the greatest possible protection of the affected area.[119] If the prerequisites are met, the decision to

112 Settled case law ECJ, adjudication of 11.9.2012 – C-43/10, margin number 113; adjudication of 26.10.2006 – C-239/04, margin number 20; BVerwG, adjudication of 23.4.2014 – 9 A 25.12, margin number 48.

113 BVerwG, adjudication of 17.1.2007 – 9 A 20.05, margin number 66.

114 BVerwG, adjudication of 14.4.2010 – 9 A 5.08, margin number 91; decision of 10.11.2009 – 9 B 28.09, margin number 6; adjudication of 17.1.2007 – 9 A 20.05, margin number 44.

115 In detail European Commission 2021a, pp. 67–91; European Commission 2018a, pp. 53–70 and Möckel Nature Conservation 2017b.

116 ECJ, adjudication of 17.4.2018 – C-441/17, margin number 191; adjudication of 14.1.2016 – C-399/14, margin number 56 et seq.; adjudication of 15.5.2014 – C-521/12, margin number 36; adjudication of 11.4.2014 – C-258/11 margin number 35; adjudication of 16.2.2001 – C-182/10 margin number 74 et seq.; adjudication of 24.9.2011 – C-404/09, margin number 109, 157.

117 ECJ, adjudication of 17.4.2018 – C-441/17, margin number 189; adjudication of 14.1.2016 – C-399/14, margin number 73; adjudication of 20.9.2007 – C-304/05, margin number 83.

118 Cf. ECJ, adjudication of 11.9.2012 – C-43/10, margin number 114; BVerwG, adjudication of 1.4.2015 – 4 C 6.14, margin number 28.

119 BVerwG, adjudication of 9.7.2009 – 4 C 12.07, margin number 15. Similar BVerwG, decision of 3.6.2010 – 4 B 54.09, margin number 9; adjudication of 6.11.2013 – 9 A 14.12, margin number 79.

deviate is at the discretion of the competent authority,[120] although this authority has no discretion and, in principle, no margin of appreciation with regard to whether the requirements are met in the first place.[121]

While coherence measures are usually possible regarding operating areas belonging to the project when it comes to land-use in agriculture and forestry, the other two preconditions raise larger issues. Even if there is a great public interest in maintaining food security and preserving forests as carbon sinks and as a source for numerous ecosystem services, the individual areas used for agriculture and forestry are unlikely to be of any particular public interest, since due to the many agricultural and forestry areas within and outside of a member state, individual areas are dispensable. This may have to be assessed differently, however, if intervention is required to address an acute pest and disease infestation in order to avoid large-scale spread and, thus, greater social damage. An overriding interest could also come into consideration, if the agricultural and forestry land-use is necessary for the preservation of the protected habitat types or species (e.g. in the case of open land biotopes). However, this is unlikely to be the case, if management results in significant adverse effects on a Natura 2000-site. Mere private interests (e.g. economic profits), on the other hand, do not suffice as justification for exceptions,[122] since the public interest in protecting private property relates to the institution of private property, but not to individual property positions.[123]

The question of the local and factual lack of alternatives to agricultural and forestry land-use also raises major difficulties. On the one hand, at least in individual cases, the agricultural and forestry land-use can regularly be carried out on other areas outside of a Natura 2000-site and its protective area by purchasing or leasing corresponding areas. On the other hand, generally recognised and proven extensive ways of agricultural and forestry land-uses are established (e.g. organic farming,[124] including animal husbandry in accordance with EU regulation 834/2007/EC; forestry in accordance with FSC criteria), with which significant adverse effects can usually be avoided. Insofar as, as with the Dutch Programma Aanpak Stikstof, an attempt is made to claim an exemption from Art. 6(3) HD for the entire agricultural sector in a region or in the whole Member States on the basis of state-planned management measures and the classification of the entire agricultural sector and its profitability as an overriding reason in the public interest,[125] the ECJ has not followed this.[126] This is to be agreed with. On the

120 ECJ, adjudication of 4.4.2010 – C-241/08, margin number 72; adjudication of 26.10.2006 – C-239/04, margin number 25; adjudication of 21.7.2016 – C-387/15 and C-388/15, margin number 63.

121 BVerwG, adjudication of 6.11.2013 – 9 A 14.12, margin number 74; adjudication of 9.7.2009 – 4 C 12.07, margin number 15; decision of 3.6.2010 – 4 B 54.09, margin number 9. Different for national defence projects BVerwG, adjudication of 10.4.2013 – 4 C 3.12, margin number 19.

122 ECJ, adjudication of 16.2.2012 – C-182/10, margin number 77.

123 Misjudging Sobotta Journal for Nature Conservation 2018, p. 263.

124 Thünen-Institut 2019.

125 Cf. considerations of Advocate General J. Kokott in C-293/17 and C-294/17, margin numbers 158 et sqq.

126 Cf. ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 105–137.

one hand, Art. 6(3) HD serves the protection of the individual Natura 2000 site and not merely an average improvement in the entire network of protected areas. On the other hand, the individual agricultural and forestry projects with potential negative impacts are subject to appropriate assessment; however, they are not relevant to the public interest in food security as well as for securing jobs and the overall prosperity.

Overall, the requirements for an exception under Article 6(4) HD are rarely met fully, when it comes to land-use in agriculture and forestry.

6. Conclusion

The ECJ decision of 7 November 2018 has far-reaching legal and practical implications for agricultural and forestry land-use in the EU. In practice, a large number of agricultural and forestry land-uses and management measures within and in the vicinity of Natura 2000-sites require an official screening and, if necessary, a full appropriate assessment under Article 6(3) HD and are only permitted if significant effects can be excluded. All agricultural and forestry measures in and in the vicinity of Natura 2000-sites are subject to the reservation of examination and prohibition, if their application has changed since 1992 or is being implemented for the first time and, according to general experience, is likely to prejudice the conservation objectives of the Habitats Directive (e.g. by dint of the use of fertilisers and plant protection products, the conversion of permanent pastures and other near-natural or extensive areas, drainage measures).

In legal terms, national laws or protected area-regulations for Natura 2000-sites may no longer exempt agricultural and forestry land-use generally from the prohibition of deterioration and disturbance, as well as from the applicability of an appropriate assessment, since such general exemption clauses violate Article 6(2) and (3) HD.[127] Rather, it must be ensured that, in the event of changes to the agricultural and forestry land-use that was carried out before the HD came into force, an official screening and, if necessary, a full appropriate assessment is carried out.[128] Changed land-use and management measures are only permitted if a significant adverse effect to Natura 2000-sites can be ruled out with certainty. Here, an unfavourable state of conservation restricts admissibility.[129] If there are no preconditions for an exemption as listed in Article 6(4) HD, agricultural and forestry land-uses must be adapted and incompatible management measures terminated.

According to the ECJ, no deviating standards can be derived from the fact that implemented projects have previously been unregulated or approved by the authorities, since proportionality is ensured by the exception options in Article 6(4) HD and, furthermore, no preservation of legitimate expectations exists in an unchangeable legal

127 ECJ adjudication of 21.6.2018 – C-543/16 margin number 91–94; adjudication of 10.1.2006 – C-98/03, margin number 39–43; adjudication of 17.9.1978 – 412/85.

128 Cf. Kelleghan et al. Atmospheric Environment 2021.

129 ECJ, adjudication of 7.11.2018 – C-293/17 and C-294/17, margin number 103.

situation.[130] This is in line with the jurisdiction of the German Federal Constitutional Court (BVerfG),[131] according to which the trust in an unchanged legal situation is not protected and the legislator is authorised to regulate hitherto unregulated matters and to restrict existing private benefits that are based on an ecologically questionable use of public goods.[132]

There is also no right under European law or human right to intensive agricultural and forestry land-use with maximum yields, given that economic efficiency of private land-use is given even with extensive cultivation (e.g. due to higher market prices for organic products, European direct or agri-environmental payments). In addition, restrictions on agriculture and forestry in Natura 2000-sites can be compensated by Member States, based on Article 30 EAFRD Regulation 1305/2013/EU (from 1.1.2023 according to Art. 72 CAP Regulation 2021/2115/EU).[133] Rather, the ownership and use of land are subject to greater social responsibility due to the limited availability of land and its integration into ecosystems and in the respective spatial situations in comparison to, for instance, mobile property or indoor uses.[134] Ecological conditions, such as the occurrence of rare or endangered animal and plant species, are intrinsic natural limitations to private ownership of land or as the German Federal Administrative Court put it:

'If the natural or spatial features of a plot of land are worth preserving in the general interest and require protection, this results in a kind of immanent, i.e. property innate restriction of the owner's powers, which is only given manifestation by nature and landscape protection regulations'.[135]

Even if, in view of the considerable spatial extent of agricultural and forestry land-uses within and in the vicinity of Natura 2000-sites, as well as the multitude of

130 Cf. ECJ, adjudication of 10.11.2016 – C-504/14, margin number 41; adjudication of 14.1.2016 – C-399/14, margin number 69–71, 74–78; adjudication of 11.9.2012 – C-43/10, margin number 136 et seq.; adjudication of 14.1.2010 – C-226/08, margin number 42–46; adjudication of 7.9.2004 – C-127/02, margin number 37.

131 From 1998 onwards, BVerfG decisions can be located, based on their case number and can be freely accessed under: https://www.bundesverfassungsgericht.de/EN/Entscheidungen/Suche/suche_node.html.

132 BVerfG, adjudication of 3.4.2001 – 1 BvR 1681/94; decision of 9.12.2003 – 1 BvR 558/99; decision of 25.7.2007 – 1 BvR 1031/07, margin numbers 35–37; adjudication of 20.4.2004 – 1 BvR 1748/99, margin number 41.

133 Cf. German jurisdiction of the Federal Constitutional Court (BVerfG) and the Federal Administrative Court (BVerwG): BVerfG, decision of 10.10.1997 – 1 BvR 310/84, NJW 1998, 367; decision of 22.11.1994 – 1 BvR 351/91, BVerfGE 91, 294 (310); BVerwG, adjudication of 25.10.2018 – 4 C 9.17, margin number 29.

134 Cf. German jurisdiction of the Federal Constitutional Court and the Federal Administrative Court: BVerfG adjudication of 22.05.2001 – 1 BvR 1512, 1677/97, BVerfGE 104, 1 [12]; adjudication of 12.01.1967 – 1 BvR 169/63, BVerfGE 21, 73 [82 et seq.]); BVerwG adjudication of 24.06.1993 – 7 C 26/92, BVerwGE 94, 1 [4].

135 BVerwG adjudication of 24.06.1993 – 7 C 26/92, BVerwGE 94, 1 [4] (translated by Harry Bauer). The original quote is: 'Wenn die natürlichen oder landschaftsräumlichen Gegebenheiten eines Grundstücks im Interesse der Allgemeinheit erhaltenswert sind und des Schutzes bedürfen, so ergibt sich hieraus eine Art immanenter, d. h. dem Grundstück selbst anhaftender Beschränkung der Eigentümerbefugnisse, die durch natur- und landschaftsschutzrechtliche Regelungen lediglich nachgezeichnet wird.'

different management methods and measures, it is understandable that Member States have hitherto refrained from subjecting these land-uses to an appropriate assessment as far as possible, as the interest in limiting assessing efforts for land users and authorities cannot justify a weakening of the protection standards required by European law.[136] According to the ECJ, anticipated, sweeping exemptions without an assessment of individual cases are only permitted to a very limited extent and, in particular, not permitted on the basis of general protective regulations or average values.

In order to meet the legal and practical requirements of European law by applying a reasonable amount of effort, it is advisable to subject agricultural and forestry land-uses that were previously free of approval to a general reservation of permission with a concentration effect, instead of assessing each individual management measure.[137] In addition, it is necessary to examine the extent to which certain agricultural and forestry land-uses can be exempted from protected area-regulations for individual Natura 2000-sites on the basis of early appropriate assessments.[138] The latter is only permissible under European law, however, if it is ensured that significant adverse effects cannot occur by means of specific legal requirements with regard to the type and extent of the management and its official monitoring. Furthermore, in view of constant ecological changes to protected habitats and species (e.g. due to population dynamics or climate change), as well as changes in cumulative effects (e.g. due to new projects or remote effects), such area-related exemptions must be subject to renewed appropriate assessments at periodic intervals in order to evaluate their impact, based on the current situation.

Conflicts of interest

The author declares no conflicts of interest. The founding sponsors had no role in the design of the study; in the collection, analyses or interpretation of data; in the writing of the manuscript and in the decision to publish the results

Acknowledgements

The author thanks Harry Bauer for translating the vast majority of the manuscript, as well as Prof. Dr. Chris Backes and Prof. Dr. Hendrik Schoukens for their reviews and helpful comments.

The work was funded by the German Federal Ministry of Education and Research (BMBF) in the framework of the funding measure “Soil as a Sustainable Resource for the Bioeconomy—BonaRes”, project “BonaRes (Module B): BonaRes Centre for Soil Research, subproject A” (grant 031A608A).

136 Cf. ECJ, adjudication of 21.6.2018 – C-543/16, margin number 113 et seq.

137 Cf. also Möckel et al. 2014, p. 453 et sqq.

138 Cf. Kelleghan et al. Atmospheric Environment 2021; Tegner Anker et al. JEEPL 2019.

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Nature Conservation 48: 185–212 (2022)
doi: 10.3897/natureconservation.48.85179
https://natureconservation.pensoft.net

RESEARCH ARTICLE

A peer-reviewed open-access journal
Nature Conservation
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Spatial and temporal dynamics of habitat quality in response to socioeconomic and landscape patterns in the context of urbanization: A case in Zhengzhou City, China

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Academic editor: M. Nazre | Received 11 April 2022 | Accepted 31 May 2022 | Published 23 June 2022

https://zoobank.org/412AD17D-30F0-4A81-B065-71BE9D9CB2B5

Citation: Zhao M, Tian Y, Dong N, Hu Y, Tian G, Lei Y (2022) Spatial and temporal dynamics of habitat quality in response to socioeconomic and landscape patterns in the context of urbanization: A case in Zhengzhou City, China. Nature Conservation 48: 185–212. https://doi.org/10.3897/natureconservation.48.85179

Abstract

With the rapid development of urbanization, the habitat quality (HQ) in urban areas has been eroded. This phenomenon is destroying the balance of ecosystems, triggering the reduction of biodiversity and the decay of ecosystem service functions. The study of the relationship between urbanization and HQ in Zhengzhou City is beneficial for the reference of sustainable urban ecological planning and management. Based on landscape classification data and socioeconomic data for three years, this study analyzes the spatial correlations between socioeconomic and landscape pattern factors and HQ, compares the dynamic changes in the explanatory power of different factors, and explores the joint effects between multiple factors. The results show that: (1) The overall value of HQ index in Zhengzhou City decreased by .10 during 2000–2020, mainly occurring in suburban areas, with a small amount of HQ improvement occurring in the core areas of ecological protection, such as mountains and river channels. (2) The spatial autocorrelation of all influencing factors with HQ increased during this period, while the negative impact from socioeconomic sources was stronger than the positive impact from landscape patterns. (3) Intensive human activities lead to a single habitat type, which reduces HQ; rich landscape types and complex landscape composition can enhance HQ. Improving the connectivity of blue-green landscapes helps to attenuate the negative effects of urbanization on HQ. (4) Changes of HQ in the study area and the development of multi-factor effects on HQ are driven by the Zhengzhou Metropolitan Area Plan. Urban development policies and management can build idyllic complexes at the edge of urban development, preserving pristine blue-green patches to avoid their homogenized distribution and thus slowing the decline of HQ. The above results provide new ideas for the development of sustainable urban ecology.

Keywords
Landscape pattern, policy, socioeconomic, urbanization, Zhengzhou Metropolitan Area Plan

1. Introduction

Habitat refers to the environment in which organisms live, and habitat quality (HQ) measures the ability of an ecosystem to provide conditions for individuals and populations to survive and reproduce (Hall et al. 1997; Nelson et al. 2009). HQ is indicative of the reflection of biodiversity status. Global urbanization rates are continuing to grow, with urban areas already doubling in 2020 compared to 1992, and may expand to 180% in 2100 (Knapp et al. 2021); the rapid expansion of cities is eroding the natural habitats where plants and animals live. The study shows that the fragmentation of the landscape and the complexity of the landscape structure continue to affect HQ as the expansion of towns and cities is accompanied by rapid changes in the surface pattern (Goldstein et al. 2012; Rosenberg et al. 2019; Chang et al. 2021). Areas with high HQ are more likely to have towns and cities, and the range of negative impacts of urbanization is much greater than that of ecosystems in urban areas (Knapp et al. 2021). Urbanization is considered to be an important cause of degradation and the loss of pristine habitats and thus a threat to ecosystem stability (Van Dolah et al. 2008; Mcdonald et al. 2009; Song et al. 2020). As a basic component of the ecosystem, changes in the quality of habitat are important for protecting biodiversity, building ecological security patterns, and enhancing ecosystem service functions (Termorshuizen and Opdam 2009; Krauss et al. 2010). In order to maintain the balance of the regional ecosystem, to create a near-natural and diverse habitat and promote a healthy symbiotic relationship between human and nature, research related to HQ is one of the hotspots in the field of urban ecology (John et al. 2019; Lanfredi et al. 2022).

Achieving regional ecological sustainability requires exploring the mechanisms by which urbanization affects ecosystem structure and function. Therefore, the responsive relationship between urbanization and HQ has attracted the attention of many scholars. The InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) model is commonly used to quantify HQ in recent studies (Moreira et al. 2018; Wu et al. 2021); this model can be used to obtain the HQ index based on the ecological suitability of the habitat and its sensitivity to different threat sources, and to derive spatial distribution maps of HQ index based on the composition of multiple habitats even when complete species distribution data are not available (He et al. 2017). The manifestations of urbanization can be divided into two forms, indirect and direct. The landscape pattern index is considered as an indirect representation of urbanization; it can express the changes in landscape patterns under the influence of human activity aggregation and land use change (Suo et al. 2016; Dadashpoor et al. 2019). To a certain extent, it reflects the impact of the urbanization process on the ecological environment. Several international scholars have conducted studies on the relationship between landscape pattern and HQ,

and uncovered regional differences in the effects of landscape pattern indices on HQ (Sallustio et al. 2017; Dadashpoor et al. 2019; Chang et al. 2021). However, the indicative role of the landscape pattern index is limited because the causes of landscape pattern changes are very complex (Li et al. 2004). Socio-economic indicators are seen as a direct manifestation of urbanization (Zeng et al. 2022), visually reflecting the prosperity and expansion intensity of cities. Several studies have indicated a significant correlation between socioeconomic indices and HQ (Sun et al. 2019; Zhu et al. 2020), factors such as population density (POP) and Gross Domestic Product (GDP) have a negative impact on HQ (Bai et al. 2019), their model simulations predicted that the intensive development of cities and towns could slow habitat degradation (He et al. 2017; Chu et al. 2018; Li et al. 2018). Fewer studies have combined the two in a multi-temporal analysis and quantified the spatial correlation between socioeconomic indicators, landscape pattern indices and HQ. This paper will examine this perspective.

In related studies, when analyzing the influence of multiple influencing factors on HQ, SPSS correlation analysis was applied to screen the influencing factors with strong influence on HQ (Zhu et al. 2020), combined with statistical analysis models such as the ordinary least squares model (OLS) and the geographically weighted regression model (GWR) to infer the degree of association between different influencing factors on HQ in geographic space and to determine the relationship between multiple variables and HQ (Sun et al. 2019; Wu et al. 2021). These research methods reflect the relationship between individual influencing factors and HQ; they cannot reflect the intensity and magnitude of the aggregation of correlations in space, and cannot analyze the joint effect of different influencing factors on HQ. The bivariate autocorrelation analysis with GeoDA software (Huang et al. 2020a; Chang et al. 2021) and the interaction detector with Geodetector software (Wang et al. 2022) can solve the above mentioned problems.

As China's new first-tier city and one of the country's major transportation hubs, Zhengzhou City is a typical example of urbanization development with its high population flow and rapid urban renewal (Feng et al. 2005). Zhengzhou City is a core area for metropolitan development and also has a Yellow River Wetland Nature Reserve; it covers a wide range of landscapes including large rivers, mountains, hills, and plains. Because of its urban expansion rate, landscape pattern changes, and ecosystem composition, Zhengzhou City is an ideal study area for conducting research on urbanization and HQ change. This paper evaluated Zhengzhou's HQ from 2000 to 2020 through the InVEST model based on three phases of landscape classification data. With the technical support of GIS10.2 software, a grid cell of 1 km × 1 km was used to resample the study area, and GeoDA software was applied to analyze the spatial correlation between landscape pattern factors, socio-economic factors and HQ. Geodetector software was used to compare the influence of different factors on HQ and analyze the common effect between the influencing factors. The objectives of this study are: (1) to analyze the spatial and temporal evolutionary characteristics of HQ in Zhengzhou in multiple time series, (2) to reveal the spatial coupling relationship between socio-economics, landscape pattern and HQ, (3) to explore the optimization strategies of urban ecology in the context of urban regionalization development.

2 Materials and methods

2.1. Study area

Zhengzhou City is the capital of Henan Province (34°16'N–34°58'N, 112°42'E–114°14'E) and is located in the central-northern part of Henan Province. With a continental monsoon climate and four distinct seasons, it is hot and rainy in summer, but cold and dry in winter. The terrain is high in the west and low in the east, with plains and inclined plains dominating the whole territory, while the western mountainous areas belong to the Funiu Mountains and the rivers in the territory belong to the two major water systems, the Yellow River and the Huaihe River (Feng et al. 2005; Lei et al. 2012). Zhengzhou City is in charge of Zhongyuan District, Erqi District, Jinshui District, Huiji District, Shangjie District, Guanchenghuizu District, Xinzheng City, Dengfeng City, Xinmi City, Xingyang City, Gongyi City, Zhongmou County (Wang et al. 2021). In 2020, Zhengzhou's GDP exceeded EUR 0.17 trillion for the first time, ranking 16th among China's top 100 cities. According to the results of the seventh national census, Zhengzhou's resident population jumped into first place in Henan Province, attracting 74% of the province's new population over the last 10 years, demonstrating superb economic growth and population absorption capacity. As a national central city and a national ecological garden city, Zhengzhou City is gradually growing into the core city of the Central Plains City Cluster. Location and elevation image of Zhengzhou City is as follow (Fig. 1).

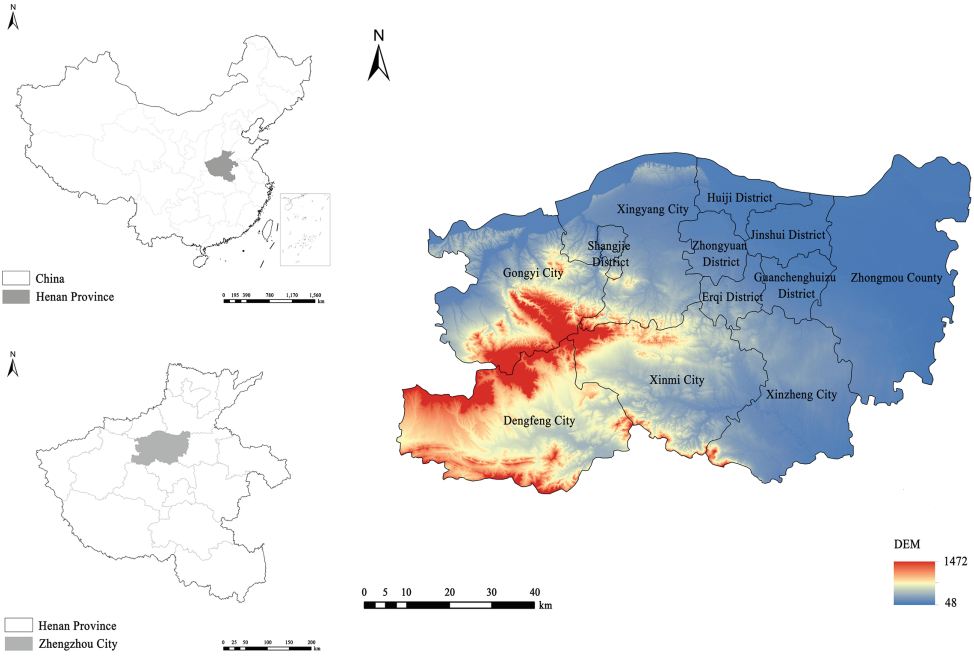

Figure 1. Location and elevation image of Zhengzhou City.

2.2. Data sources and pre-processing

The 30 m resolution landscape classification data for 2000, 2010 and 2020 were obtained from GlobleLand30 (http://www.globallandcover.com, accessed on 29 November 2021) released by the Ministry of Natural Resources of China, using the multispectral images without or with few clouds in the vegetation growing season as the information source, and classifying the land use types according to land use attributes and natural attributes. It is divided into 10 primary land use types, and after data merging and clipping, a total of 6 primary land use types are covered in the study area, namely, arable land, forest, grassland, wetland, water, and construction land, with a classification accuracy of more than 83%. The specific classification description is shown in Appendix 1. The nighttime light data come from the joint product developed by the GIS development and urban research team of the College of Geographical Sciences of East China Normal University and others (https://doi.org/10.7910/DVN/YGIVCD, accessed on 29 November 2021) (Chen et al. 2021b), using DMSP-OLS and NPP-VIIRS NTL as data sources, with the advantages of high spatial resolution of 500 m and long time span through cross-sensor calibration, verified by random pixel, with good accuracy in pixel-level (R^2: 0.87) and city-level (R^2: 0.95) (Chen et al. 2021a). Population data was obtained from worldpop's 100 m resolution demographic data set (https://www.worldpop.org/, accessed on 29 November 2021), the raster data were corrected by combining the population's numbers from Zhengzhou City Yearbooks and census results. The rural settlements were obtained from the Resource and Environmental Science and Data Center (https://www.resdc.cn/, accessed on 29 November 2021); it is used as a reference to extract the data of land urbanization space. The elevation data was obtained from the ALOS DEM data on the official NASA website (https://search.asf.alaska.edu/#/, accessed on 29 November 2021) with a spatial resolution of 12.5 m.

2.3. Methods

2.3.1. Habitat quality (HQ) evaluation

The InVEST model assesses the variability and distribution of HQ in the study area based on the sensitivity of different habitat types to stressors and the intensity of external threats to them, and evaluates the biodiversity service function of ecosystem in the study area by the level of the HQ index (Peng et al. 2018; Sun et al. 2019); these can replace a large number of field surveys and facilitate the optimization of biodiversity conservation strategies. The calculation formula is as follows:

$$Q_{xj} = H_j \left[1 - \left(\frac{D_{xj}^z}{D_{xj}^z + k^z} \right) \right] \quad (1)$$

$$D_{xj} = \sum_{i=1}^{R} \sum_{y=1}^{Y_r} \left(\frac{\omega_r}{\sum_{r=1}^{R} \omega_r} \right) r_y i_{rxy} \beta_x S_{jr} \quad (2)$$

where Q_{xj} is the HQ of raster image element x in landscape type j, H_j is the habitat suitability, D_{xj} denotes the habitat threat level, k is the half-saturation constant, usually taken as half of the maximum value of D_{xj}, z is the normalization constant, usually taken as 2.5, R denotes the number of threat factors, y is all raster image elements of threat r, Y_r indicates the total number of raster image elements occupied by r, ω_r is the weight, β_x is the is the legal accessibility of raster image element x, S_{jr} is the sensitivity of land cover j to threat factor r, i_{rxy} means the coercive effect of raster image element y on habitat raster image element x.

In landscape classification, the more primitive, complex and large continuous ecosystems have higher suitability and stability, while land types with high intensity of human activities are more likely to threaten the surrounding habitats with strong expansiveness and need to be extracted as threat sources (Chang et al. 2021). Referring to the existing research results (Wang et al. 2020; Zhu et al. 2020; Chen and Li 2021) and the actual situation in the study area, the maximum impact distance, weight of threat factors, and the sensitivity of each type of habitat to threat factors were set as Table 1 and Table 2.

Table 1. The weight for threat factors.

Threat factor	dr_max/km	Weight/ω_r	Distance-decay function
Cropland	4	0.5	exponential
Construction Land	8	0.9	exponential

Table 2. The sensitivity of habitat types to threatening factors.

Landscape code	Habitat type	Habitat suitability	Cropland	Construction Land
10	Cropland	0.5	0	0.5
20	Forest	1	0.6	0.4
30	Grassland	0.8	0.8	0.6
50	Wetlands	1	0.4	0.9
60	Water area	0.9	0.4	0.4
80	Construction Land	0	0	0

2.3.2. Selection of impact factors

The landscape pattern indicators reflect the dynamic changes of the ecosystem under the influence of urbanization as indirect influence factors, and the socio-economic indicators reflect the direct influence of socio-economic development on the ecosystem as direct influence factors. Referring to the relevant literature (Huang et al. 2020a; Chang et al. 2021; Zeng et al. 2022), the following indicators were selected as impact

factors (Table 3). In this paper, population density (POP) is selected to characterize the aggregation of population, night time light (NTL) to characterize the frequency of socio-economics, and land urbanization rate (LUR) to characterize the intensity of urban development, so that they represent the direct impact of urbanization (Chan and Vu 2017; Zeng et al. 2022). In the landscape pattern indices, SHDI and PD express the diversity of landscape patches and are used to characterize landscape types, CONTAG and ED express the shape and connectivity of landscape patches and are used to characterize landscape structure (Satir and Erdogan 2016; Zeng et al. 2022). The socioeconomic indicators are obtained from the corrected raster data, and landscape pattern indicators are calculated by Fragstats 4.0. The values of all factors are assigned to the grid with the help of ArcGIS's spatial analyst. The description of the factors' calculation formula is shown in Appendix 2.

Table 3. Descriptions of the impact factors.

Category	Metrics	Abbreviation	Description
Landscape pattern	Edge density	ED	Reflects the degree of differentiation or fragmentation of the overall landscape patches.(Xia et al. 2021)
	Contagion index	CONTAG	Reflects the degree of agglomeration or extension trend of the plaque.
	Shannon's diversity index	SHDI	Reflects landscape heterogeneity.(Li 2011)
	Patch density	PD	The number of patches in unit area.
Socio-economic	Population density	POP	The number of people per square kilometer.
	Night time light	NTL	Reflects the activity and agglomeration of socio-economic activities.
	Land urbanization rate	LUR	Proportion of urban land to urban-rural construction land.(Gao et al. 2018)

2.3.3. Grid analysis

The application of grid analysis can describe, compare, and analyze regional geographic phenomena in equivalent spatial conditions. 1 km × 1 km grid scale is often applied in articles for studying land use change (Zhu et al. 2020; Xia et al. 2021; Wang et al. 2022), so this paper uses this grid scale as the basic research scale for analysis and discussion.

2.3.4. Construct the spatial weight matrix

The spatial weight matrix is constructed by GeoDA software to define the spatial relationship between grids, and the queen contiguity is selected to construct the spatial weights with the grid number as the variable, with the following rules:

$$W = \begin{bmatrix} w_{11} & w_{12} & \cdots & w_{1n} \\ w_{21} & w_{22} & \cdots & w_{2n} \\ \vdots & \vdots & \vdots & \vdots \\ w_{n1} & w_{n2} & \cdots & w_{nn} \end{bmatrix} \quad (3)$$

where n denotes the number of spatial units, w_{ij} denotes the adjacency between region i and j. If they have a common boundary or point, the value is 1, otherwise, the value is 0.

2.3.5. Bivariate spatial autocorrelation

Bivariate spatial autocorrelation analysis can reflect the degree of association between two attribute values of a spatial unit (Anselin 1995); the relationship is characterized by the Moran's *I* index, while the Moran's *I* scatter plot is generated. LISA (the Local Indicators of Spatial Association) clustering maps can characterize the degree of correlation between a unit and its neighboring units on the geographic space. There are generally four types of spatial patterns in the LISA clustering map: high-high (H-H), high-low (H-L), low-high (L-H), and low-low (L-L). The Moran's *I* index and LISA clustering map can show the degree of spatial association of different indicators with HQ and the distribution of clustered areas. The calculation formula is:

$$I = \frac{n\sum_{i=1}^{n}\sum_{j=1}^{n} w_{ij}(x_i - \bar{x})(x_j - \bar{x})}{\sum_{i=1}^{n}\sum_{j=1}^{n} w_{ij}\sum_{i=1}^{n}(x_j - \bar{x})^2} = \frac{\sum_{i=1}^{n}\sum_{j=1}^{n} w_{ij}(x_i - \bar{x})(x_j - \bar{x})}{S^2\sum_{i=1}^{n}\sum_{j=1}^{n} w_{ij}} \quad (4)$$

$$S^2 = \frac{1}{n}\sum_{i=1}^{n}\left(x_i - \bar{x}\right)^2 \quad (5)$$

$$\bar{x} = \frac{1}{n}\sum_{i=1}^{n} x_j \quad (6)$$

where I is the Moran's *I* index; n is the number of spatial cells, x_i and x_j are the observed values of cells i and j, respectively, and w_{ij} is the spatial adjacency of cells i and j. S^2 is the variance of the observed values. I takes values between [-1,1], and values less than 0 indicate negative spatial correlation, greater than 0 indicate positive spatial correlation, where equal to 0 indicates no correlation and random distribution in space.

2.3.6. Geodetector

The Geodetector can avoid the covariate interference of multiple factors and compare the magnitude of the driving force or explanatory force of multiple influencing factors on the geospatial distribution of something based on spatial heterogeneity (Wang et al. 2022). The Geodetector can not only reveal the influencing factors with important driving forces behind HQ, but also compare the magnitude of the explanatory power of the factors and evaluate the co-action among them (Wang and Xu 2017). The results of the Geodetector's analysis can be used to obtain influence factors that are more helpful to improve HQ and provide reference for urban planning adjustments. The formula is:

$$q = 1 - \frac{1}{n\sigma^2}\sum_{h=1}^{L} n_k \sigma_k^2 \tag{7}$$

where: q is the explanatory power; n_k and n are the number of samples within type h of factor A and within the entire study area, respectively; σ_k^2 and σ^2 are respectively the discrete variance within type h of factor A and within the entire study area. q takes values between [0,1], and larger values of q indicate greater explanatory power of factor A.

3. Results

3.1. Spatial distribution and dynamics of HQ

As shown in Fig. 2, the spatial pattern of HQ in Zhengzhou City changed significantly from 2000 to 2020. According to the values of HQ index, there are high quality zones (>0.75), relatively high quality zones (0.5~0.75), relatively low quality zones (0.25~0.5), and low quality zones (<0.25). Overall, the distribution of HQ in Zhengzhou City from 2000 to 2020 is "high in the northwest and low in the southeast". In conjunction with the landscape classification map of Zhengzhou City (Fig. 3), the northwestern and northeastern parts of the study area serve as the edge of the main urban area and the nature reserves; there are rich landscape compositions with intermingled agriculture and forestry, and concentrated high quality areas. The southeastern part of the study area is an agglomeration of arable land with lower quality zones distributed. Low quality areas are distributed in the main urban area, which is dominated by man-made surfaces in the central north.

From 2000 to 2020, the area of the low HQ changed greatly, increasing by 1451.68 km^2, with a percentage increase of 19.15%; the relatively low HQ zone and high HQ zone showed a decreasing trend, decreasing by 1401.16 km^2 and 70.42 km^2, with a percentage decrease of 18.45% and 0.92%, respectively, where the high HQ zone showed fluctuating changes. The relatively high HQ zone had the smallest change with an increase of 17.42 km^2 and a percentage increase of 0.23% (Table 4).

Table 4. HQ classification and area change.

Classification	Value	2000		2010		2020	
		Area/km^2	Percentage/%	Area/km^2	Percentage/%	Area/km^2	Percentage/%
Low habitat quality	<0.25	615.63	8.12	1237.70	16.32	2067.30	27.27
Relatively low habitat quality	0.25~0.5	6021.31	79.39	5360.43	70.68	4620.15	60.94
Relatively high habitat quality	0.5~0.75	52.40	0.69	54.37	0.72	69.81	0.92
High habitat quality	>0.75	895.17	11.80	932.00	12.29	824.75	10.88

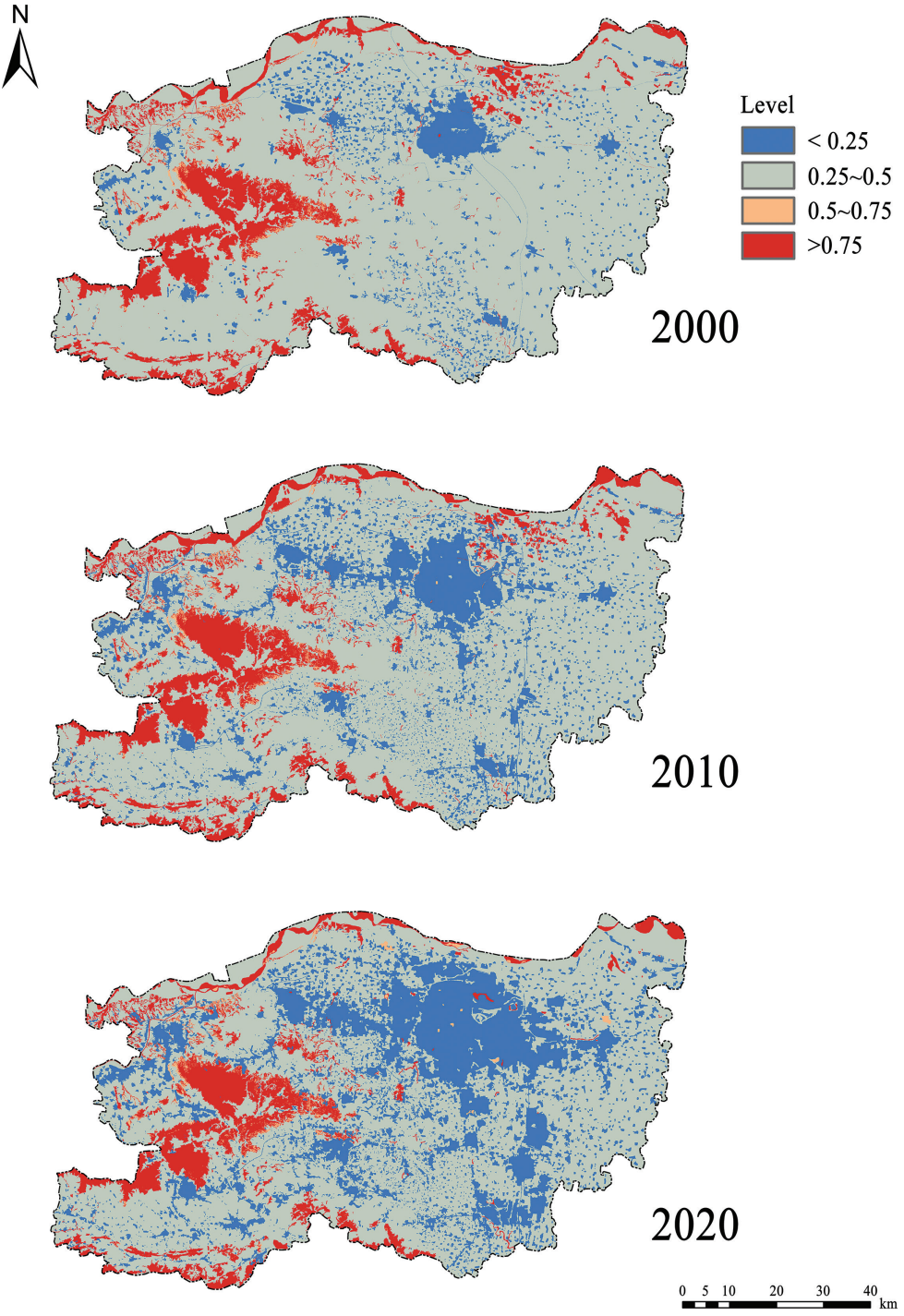

Figure 2. Spatial pattern of HQ.

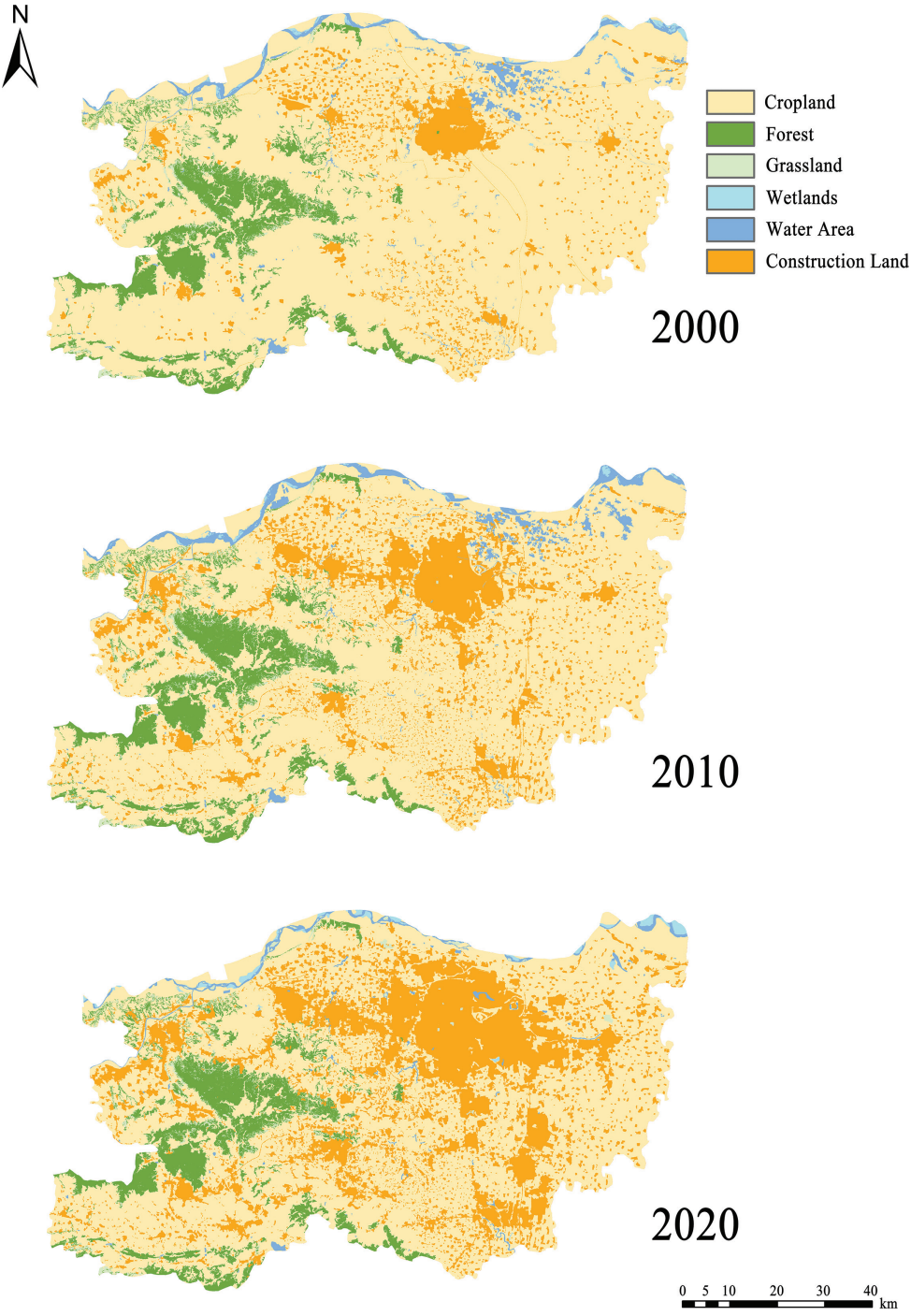

Figure 3. Landscape classification map of Zhengzhou City.

HQ changes in Zhengzhou City from 2000 to 2020 were calculated by using the ArcGIS software, through the natural break method the results were classified into five categories: significant decrease, slight decrease, no significant change, slight increase, and significant increase (Fig. 4). The northeastern part of the main urban area, where the urban construction activities are most concentrated, has significantly reduced HQ. The area centered on the urban to the distant suburbs with a slight decrease in HQ. The quality of habitats in places adjacent to natural woodlands and rivers improved slightly. The mountainous zone in the west and the Yellow River basin in the north have high HQ themselves, with little overall change and significant improvement in some areas.

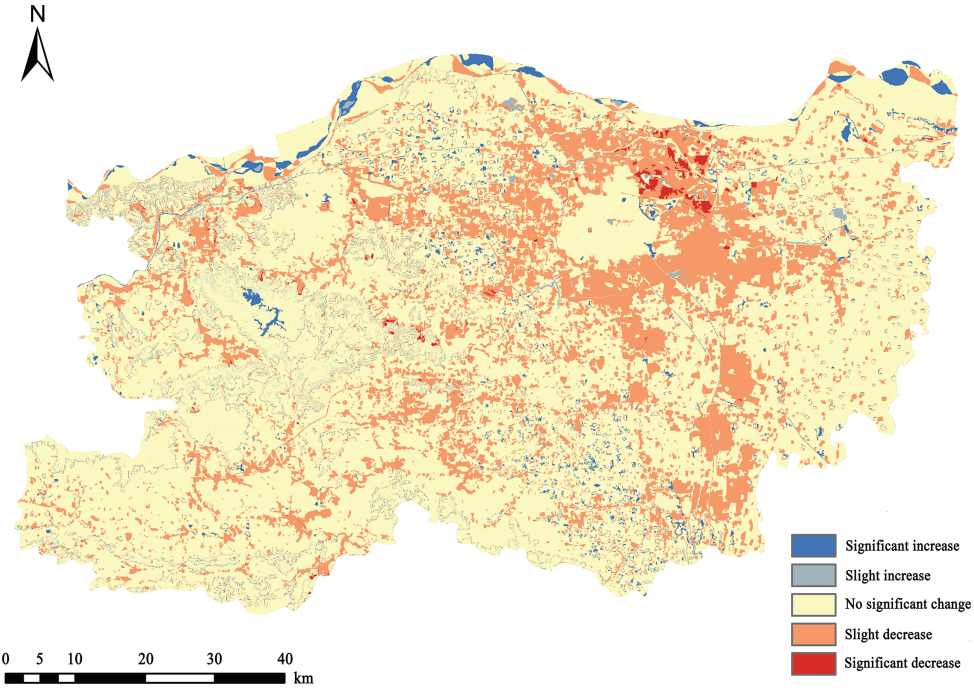

Figure 4. The variation of HQ.

3.2. Spatial autocorrelation analysis of HQ and different impact factors

3.2.1. Global autocorrelation

Moran's *I* indices for seven sets of bivariate variables were obtained using GeoDA software, after 999 random permutations, all of them passed the z-test ($p = 0.001$), indicating a significant spatial autocorrelation between the bivariate variables at the 99.9% confidence level.

As shown in Table 5, the Moran's *I* indices of three socioeconomic indices, POP, NTL, and LUR, and HQ are negative, indicating negative spatial correlations; the Moran's *I* indices of four landscape pattern indices, PD, CONTAG, SHDI, and ED,

and HQ are positive, indicating positive spatial correlations. Comparing the Moran's *I* indices of each year, it can be found that the absolute values of NTL, POP, LUR, and PD are all higher, indicating that the spatial aggregation of NTL, POP, LUR, PD and HQ in the study area is strong. The Moran's *I* indices of NTL, LUR, and PD showed an increasing trend, except POP which showed a decreasing trend. In 2020, LUR and NTL are strongly negative (-0.518, -0.513) impact factors, and PD is a strongly positive (0.320) impact factor.

Table 5. Moran's *I* indices of HQ and impact factors.

Year	ED	CONTAG	SHDI	PD	POP	NTL	LUR
2000	0.246	0.151	0.184	0.302	-0.347	-0.320	-0.300
2010	0.218	0.180	0.127	0.279	-0.366	-0.428	-0.439
2020	0.272	0.277	0.176	0.320	-0.324	-0.513	-0.518

3.2.2. Local autocorrelation

From Fig. 5, it can be obtained that the spatial aggregation effects of different impact factors and HQ are significantly different. The landscape pattern indices mainly showed H-H cluster and L-L cluster, and the socio-economic factors mainly showed H-L outlier and L-H outlier.

Among the landscape pattern factors, the distribution and development trend of PD and ED are similar, the H-H cluster is mainly in the western mountainous area, the H-H cluster is surrounded by the H-L outlier in 2000, the H-H cluster gradually expands and the H-L outlier gradually decreases in 2010, and the H-H cluster has been distributed in a continuous pattern in the western part of Zhengzhou City in 2020. There were also many similarities between CONTAG and SHDI. CONTAG and SHDI were dominated by H-L outlier in 2000, which were scattered in the study area, and H-H cluster appeared in the western and northeastern parts of the study area, and then turned out to be dominated by H-H cluster. H-H cluster of CONTAG developing to the southwest and H-H cluster of SHDI clustering steadily in the west, the H-L and L-H outlier scattered at their edges. The L-L cluster of all four landscape pattern indices are increasing in size with the direction of urban expansion and moving to the southeast.

Among the socio-economic factors, the NTL and LUR aggregation area development is more consistent. In 2000, their L-H outlier was mainly distributed in the central part of the study area to the north, and in 2010, they expanded to the south, and in 2020, they were concentrated in the study area in a south-central direction, and a small number of H-H clusters appeared in the suburban areas at the edge of the city. The H-L outlier was distributed around the L-H outlier in 2000, gradually decreasing in size in 2010, then becoming concentrated in the western and northern parts of the study area in 2020. There is less variation in POP, with the L-H outlier mainly in the

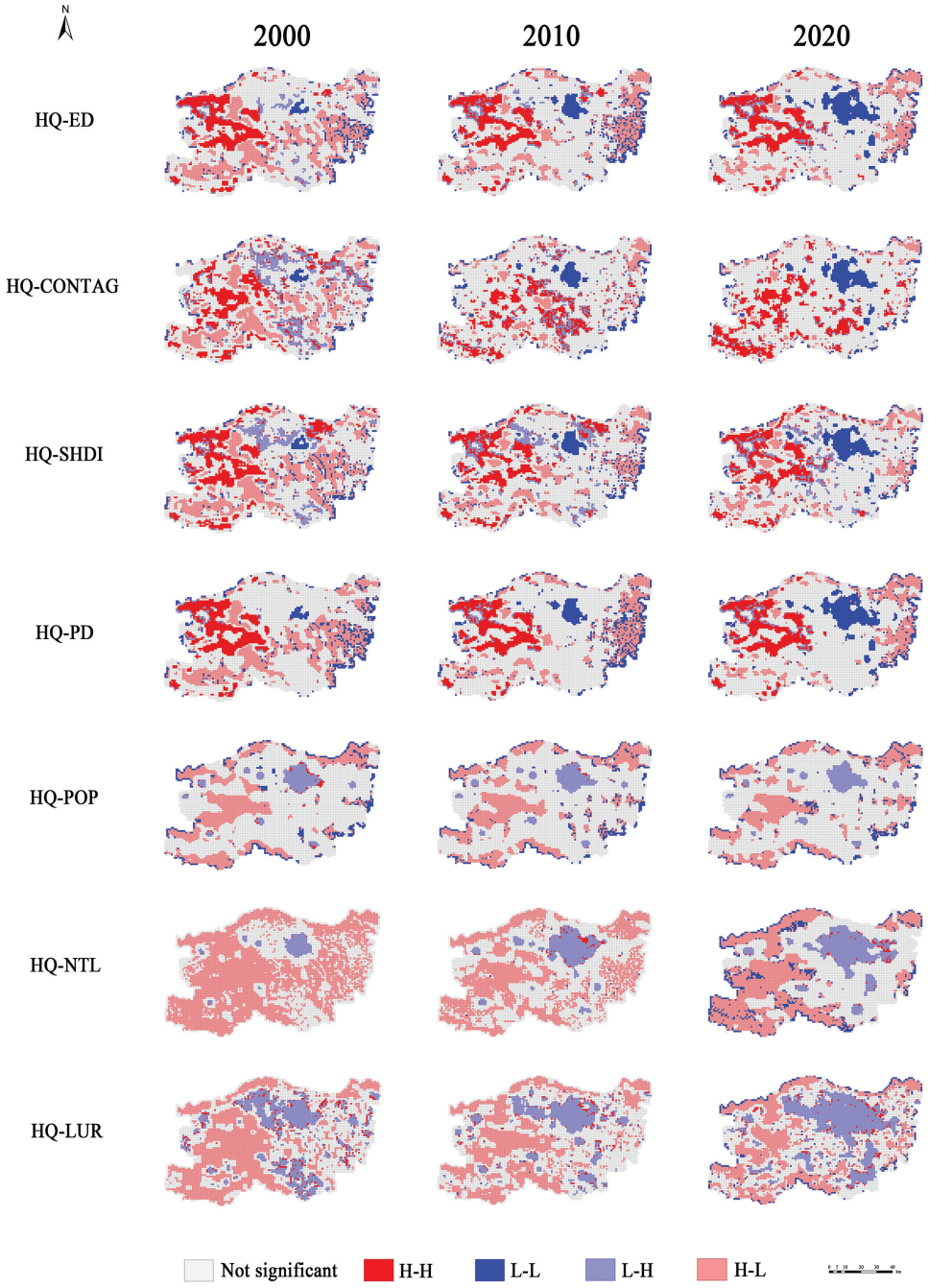

Figure 5. LISA clustering map of HQ with different impact factors.

central part of the study area to the north and the H-L outlier mainly in the western, southwestern and northern parts of the study area, with a significant decrease in the H-H cluster and a small expansion in the other agglomerations over the 20-year period.

The above shows that from 2000 to 2020 the development intensity of the landscape pattern factor, which is positively correlated with HQ, is lower than that of the socioeconomic factor, which is negatively correlated. Besides, the influence of socioeconomic and landscape pattern on HQ has different development direction and magnitude in space and time. The west and the north are the main sites for HQ protection, while the southeast is the key area for urban expansion and intensive development. In the future, metropolitan construction requires zoning plans for the development direction of different areas.

3.3. Driving force analysis of different impact factors

According to the results of the factor detector in the Geodetector, the average deterministic powers (q-value) of the seven driving factors were ranked in descending order: NTL > LUR > PD > POP > ED > SHDI > CONTAG.

In Fig. 6, except for POP, the explanatory power of the other six drivers is increasing over the period 2000–2020. From 2000 to 2010, SHDI and ED are relatively stable, the explanatory power of NTL, LUR, and CONTAG is growing, with increases of 20.49%, 40.78%, and 38.00%, while the determining power of POP and PD is decreasing, with declines of 25.33% and 22.79%. From 2010 to 2020, the influence of all factors except POP has increased, NTL and CONTAG have increased significantly, 45.36% and 69.55% respectively. The average determinant q values of NTL, LUR and PD were above 0.1 as the main drivers. The average decision force q values of POP, ED, and SHDI ranged from 0.05 to 0.1 for the secondary drivers. The mean q-value of CONTAG was below 0.05, with a small explanatory power. This suggests that NTL, LUR and PD have the greatest influence on HQ in the study area during 2000 to 2020.

Overall, the mutual gap between NTL and LUR is narrowing, and the growth trend of landscape pattern indices is similar. During the 20-year period, the determinants of NTL and LUR respectively increased by 0.21 and 0.20, the determinants of the four landscape pattern indices increased by less than 0.05, and the determinants of

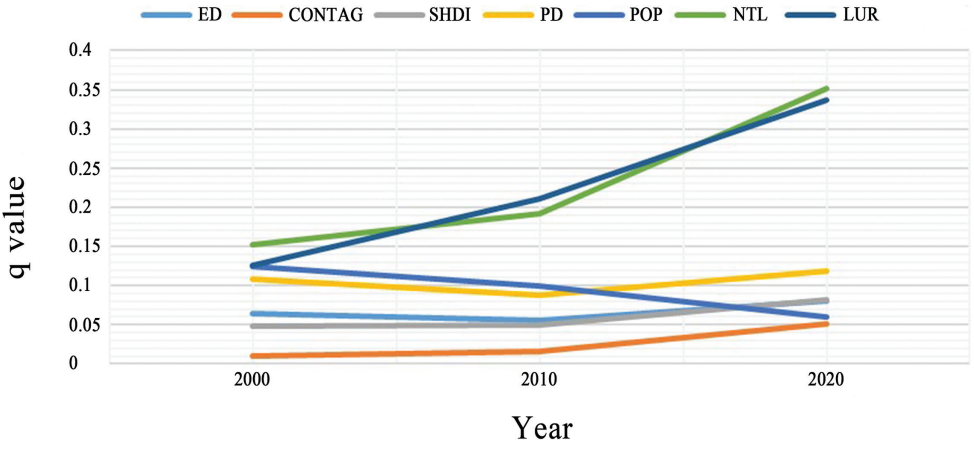

Figure 6. Changes of q-value during 2000–2020.

POP decreased by 0.06. The overall influence of the socio-economic factors was greater than the landscape pattern factors, denoting that the socio-economic factors have a more prominent influence on HQ.

The results of the ecological detector and interaction detector are shown in Table 6. By examining the differences in the effects of the seven drivers on the spatial distribution of HQ through the ecological detector, combined with the results of the factor detector, it can be confirmed that LUR, NTL, and PD have the greatest influence on HQ, and the other factors have a weaker influence. The joint effect between the seven drivers was detected by the interaction detector, and Table 6 shows that all drivers two-by-two showed a non-linearly enhanced or bi-factorially enhanced effect on the HQ distribution, indicating that the joint effect of each two drivers was stronger than the effect of the individual factors. The strongest joint effect is NTL ∩ PD in 2000 (0.2621), in 2010 the strongest is LUR ∩ PD (0.2885), and in 2020 the strongest is NTL ∩ LUR (0.4315). The nonlinear enhancement effect is greater than the two-factor enhancement. In 2000 and 2010, the co-action of the five factors CONTAG, SHDI, LUR, POP, and NTL with other factors is basically nonlinear enhancement, and only the co-action of CONTAG with other factors is nonlinear enhancement in 2020, denoting that although the co-action has been shown to be enhanced, the enhancement effect is weakening.

Table 6. Ecodetector and interaction detector results.

Year		ED	CONTAG	SHDI	PD	POP	LUR	NTL
2000	ED							
	CONTAG	N†						
	SHDI	N	Y†					
	PD	Y	Y†	Y				
	POP	Y†	Y†	Y†	N			
	LUR	Y†	Y†	Y†	N	N		
	NTL	Y†	Y†	Y†	Y†	Y	Y	
2010	ED							
	CONTAG	N†						
	SHDI	N	Y†					
	PD	Y	Y†	Y				
	POP	Y†	Y†	Y†	N			
	LUR	Y†	Y†	Y†	Y	Y		
	NTL	Y†	Y†	Y†	Y	Y	N	
2020	ED							
	CONTAG	N†						
	SHDI	N	Y†					
	PD	Y	Y†	Y				
	POP	N	N	N	N			
	LUR	Y	Y†	Y	Y	Y		
	NTL	Y	Y	Y	Y	Y	N	

† indicates that the interaction of the two factors is nonlinearly enhanced and blank indicates the interaction of the two factors is bilinearly enhanced (Appendix 3)(Wang and Xu 2017). Y means that the influence of the vertical column factor is stronger than the horizontal column factor in the ecological detector and N means that the vertical column factor is weaker than the horizontal column.

In summary, the spatial and temporal distribution of HQ in Zhengzhou City is influenced by a combination of socioeconomic and landscape pattern factors, and the influence of most factors is increasing year by year, but the influence of socioeconomic factors is dominant.

4. Discussion

4.1. Mechanisms influencing changes in HQ distribution

4.1.1. The variation of HQ

HQ in the study area showed a distribution as "high in the northwest and low in the southeast". With the expansion and construction of Zhengzhou metropolitan area, the urban land gradually evolved from point distribution to continuous distribution in patches, and the agricultural land and forest land at the edge of the city were transformed into construction land. The suburban area is also the main area of reduced HQ, as the flat topography of the central to southeastern part of the study area facilitates the laying and upgrading of traffic routes (Wang et al. 2021) which accelerates the fragmentation of the landscape. The western mountainous areas are gradually surrounded by successive towns, the degree of threat to pristine habitats has increased, and fragmentation of marginal habitats has occurred, therefore the HQ has been reduced. HQ at the northern edge of the study area showed an interwoven distribution of enhanced and degraded areas, indicating that the Yellow River basin is highly sensitive, with low ecosystem stability and HQ prone to fluctuating changes. In recent years, Zhengzhou City has focused on ecological protection and has drawn ecological red lines, which have seen an improvement in HQ in natural mountains, woodlands and rivers. In the process of building garden city and sponge city, the ecological environment of river networks and urban green areas has been maintained and improved, the new blue and green patches have been added, and hence patches of improved HQ appear within the main city. This is consistent with related studies showing that rapid urbanization significantly affects the distribution of HQ (Haddad et al. 2015), that topographic and protected area constraints can inhibit the negative effects of human activities (Huang et al. 2020a), and that increasing landscape richness and ecosystem complexity has a facilitative effect on HQ (Bai et al. 2019).

4.1.2. Changes in the correlation between different indicators and HQ

The results showed that the socio-economic factors in the study area had a negative relationship with HQ, and the landscape pattern factors had a positive relationship with HQ. Besides, the deterministic power and spatial aggregation of all influencing factors was increasing year by year, with the strongest explanatory power of NTL, LUR, and PD. The NTL represents the degree of gathering of human activities, and

the LUR represents the urbanization ratio per unit area. The higher the NTL and LUR, the more intensive the human activities, the larger the artificial surface area, the more homogeneous the habitat type and the lower the HQ, and vice versa. PD represents the number of patches, the more blue and green patches per unit area indicates the proximity to the natural habitat gathering area, low urban development, high ecological land preservation and good HQ, while the more impervious patches indicate the proximity to the main urban area, high urban development, high ecological land destruction and low HQ, the larger total number of patches the more complex the landscape composition and the higher HQ. CONTAG represents the connectivity of patches, and in the study area CONTAG in combination with either factors showed an effect of increased explanatory power, indicating that blue-green landscape connectivity has an important contribution to HQ.

In a similar study, four landscape pattern indices, including ED and SHDI, also showed significant positive correlations with HQ in the Beijing-Tianjin-Hebei region of China, although the strength of the correlations was weakening year by year (Chang et al. 2021). In Changchun City, the most significant negative correlation was found between POP and HQ (Bai et al. 2019). These denote that the coupling relationship between socioeconomic and HQ, landscape patterns and HQ is complex and variable in different regions. Vega and Küffer (Vega and Küffer 2021) found that for dense urban green infrastructure patches, connectivity is associated with a beneficial effect on species richness, which is an important expression of HQ and ecosystem service value, which, combined with this study, suggests that increasing blue-green landscape connectivity is beneficial in weakening the negative effects of urbanization on ecosystems.

4.2. Policy's driver and suggestions for urban planning

Changes in socioeconomic indicators and landscape pattern indices mainly originate from policy formulation and implementation, and reasonable policy planning can balance regional development and ecological environment protection (Le Roux et al. 2014; Huang et al. 2020b). Ruan et al (Ruan et al. 2016) found that the ecological condition of Chongming Island was improved, and ecosystem services were enhanced under the intervention of ecological conservation policies. Waylen et al (Waylen et al. 2019) found that in Europe the ecological enhancement of agricultural land due to the support of rural development programs (AES) had a positive impact on wildlife on farmland. Françoso et al (Françoso et al. 2015) noted that the establishment of protected areas has been effective in protecting habitats and biodiversity.

The response of HQ to urbanization in the study area also corresponds to the content of policy implementation during the same period. After the approval and implementation of the General Land Use Plan of Zhengzhou City (1997–2010), the government has increased the protection of nature reserves, forest parks, wetland parks and water source protection areas based on the existing Songshan Mountain National Forest Park and Yellow River Wetland, and has improved the level of watershed management based on the Yellow River and Huaihe River water system. It has been vigorously

promoting the integration process of counties (cities) and districts such as Zhongmou County, Xingyang City, Shangjie District and Xinzheng City with the central city, and accelerating the development of Zhengdongxinqu (it is an independent economic zone) to the east (Wang et al. 2021). Therefore, from 2000 to 2010, small areas of low HQ were evident in Shangjie District, Xingyang City and Zhongmu County, and the area of high HQ areas increased in northern Zhongmu County. The H-H aggregation area of landscape pattern indices and HQ gradually formed a convergence pattern in the western and northern parts of the study area, and the H-L outlier of socioeconomic indices and HQ increased in size. Later, the General Plan of Zhengzhou City (2010~2020), the Ecological Construction Plan of Zhengzhou National Central City (2016~2025), and the Spatial Plan of Zhengzhou Metropolitan Area (2018~2035) (The People's Government of Zhengzhou Municipality http://www.zhengzhou.gov.cn/) were issued one after another, the goal of regional centralized development in Zhengzhou City is clarified, and the spatial structure of "one core, four axes, three belts and multiple points" is proposed, while the integration of Zhengzhou-Kaifeng, Zhengzhou-Xinzheng, Zhengzhou-Jiaozuo and Zhengzhou-Xuchang is deeply integrated. Therefore, the low HQ areas in the study area from 2010 to 2020 are interconnected into pieces and expanded toward Zhongmou County on the basis of the original ones, and new low HQ blocks have also appeared in Xinzheng City. The high HQ of the northeastern part of the main city in Zhengzhou City has been internalized as large urban green areas, and the increased intensity of development has led to a decrease in HQ. Due to the effective implementation of the ecological protection plan, there has been an improvement in HQ in both the western mountains and the northern water system. The H-H cluster of landscape pattern indices and HQ basically formed a continuous cluster in the northwestern part of the study area, and the H-L outlier of socio-economic indices and HQ showed a clear trend of expansion to the southeast.

Excessive resource exploitation and economic growth will inevitably lead to an ecological crisis, which will in turn lead to the collapse of human society (Daly 1968; Qi and Wang 2016). To ensure the harmonious development of people and nature, from the perspective of urban planning, the adjustment of policies and plans should be based on ecological arguments (Peterson et al. 2005; Fisher et al. 2008). HQ, as an ecosystem service, can influence multiple dimensions of human well-being through its merits and demerits (Hattam et al. 2015). Combined with the analysis results of this study, it is recommended to implement diverse spatial regulation and management to gradually improve the quality of multiple habitats and provide help to enhance the integrated carrying capacity (Kiss and Kiss 2018) and sustainability of ecosystems.

(1) For habitats dominated by natural mountains, woodlands and water bodies, focus on protecting the integrity of the natural landscape and ecological stability, and ecological buffer zones can be installed in bordering areas to reduce ecological sensitivity.

(2) For the main urban areas where the population gathers, the connection and combination of similar patches should be improved. Increasing blue-green space while satisfying socio-economic development, such as the combination of urban greenways

and commercial streets, the connection of medium and large parks, the intensive layout of living space, etc., to avoid the fragmented distribution of landscape patches and gradually improve the quality of urban habitats.

(3) Preserving large blue-green patches at the junction of urban and rural development. Focus on the production red line delineation and ecological protection of farmland, develop field complexes, and flexibly regulate the Sansheng Spaces (production, living and ecological space) in response to changes in landscape patterns and HQ.

(4) Actively play the role of landscape pattern indicators to promote HQ, especially to enhance blue-green landscape diversity and connectivity, and to improve urban habitats with diverse management measures that maintain natural succession combined with human intervention, thereby increasing ecosystem service functions and enriching biodiversity.

4.3. Limitation

Since the choice of research scale affects the development of urban planning schemes (Guo et al. 2012; Yue and Liu 2017), follow-up analyses at multiple grid scales can be conducted by applying high spatial resolution data sources to improve the accuracy of habitat assessment results while investigating in depth the scale effects of the relationship between urbanization and HQ.

5. Conclusions

This paper assesses the change of HQ in Zhengzhou City from 2000 to 2020, analyzes the spatial correlation between HQ and different influencing factors, and compares the magnitude of the explanatory power and the strength of the joint effect of the influencing factors, finally obtaining the following conclusions:

(1) HQ in Zhengzhou City shows a spatial condition of "high in the west and low in the southeast", and the overall HQ shows a decreasing trend from 2000 to 2020. According to the evaluation results of the InVEST model, the average HQ index decreased from .51 to .41, and the low-HQ area increased by 1451.68 km^2, the proportion increased by 19.15%, mainly from the fragmentation and disappearance of agricultural and forest land in peri-urban.

(2) The high value areas of HQ are stably distributed in natural habitats, such as western mountains, southern woodlands, and northern waters. The low value areas are distributed in the main urban area of Zhengzhou City, and have a tendency to spread to the southeast.

(3) The influence of socio-economic and landscape patterns on HQ from 2000 to 2020 has different directions and magnitudes in space and time. The relationship between landscape pattern indices and HQ mainly shows H-H cluster and L-L cluster, the relationship between socio-economic factors and HQ mainly shows H-L outlier

and L-H outlier. Besides, the intensity of the influence of the landscape pattern factors is weaker than those of the socio-economic factors.

(4) Based on the value of the average influence, NTL (0.23), LUR (0.22), and PD (0.11) are the main determinants. The more intensive human activities, the larger the artificial surface area, the more homogeneous the habitat type, and the lower the HQ. The richer the landscape type, the more complex the landscape composition, and the higher the HQ. Analysis of the joint effects of the influencing factors revealed that blue-green landscape connectivity has a strong promoting effect on HQ.

This study provides a clearer picture of the differences in landscape patterns and socioeconomic development on HQ, and denotes that the synergistic construction of construction land and blue-green space driven by policies will contribute to the improvement of HQ, which has important implications for the planning and design of urban regionalization and the sustainable development of ecosystems.

(1) It is recommended that the planning of habitat is not limited to cities, and that a combination of natural maintenance and artificial intervention is implemented depending on the composition of the ecosystem type.

(2) In the ecological protection areas, the original landscape composition should play a role in promoting HQ, and a buffer zone should be established at the junction with the main urban area to reduce the risk of habitat fragmentation.

(3) At the boundary of urban sprawl development, there is a need to plan construction land intensively, enrich landscape diversity, protect large blue-green patches, such as natural habitats, wilderness and so on, enhance the connectivity of high-quality patches, guarantee the ecological stability of farmland, and thus avoid habitat degradation and loss.

(4) In cities with mainly impervious surfaces, blue-green patches with richer species diversity should ensure their integrity, avoid over-artificialization of blue-green spaces, and re-wild the habitats according to the habitat needs of plants and animals to gradually enrich the ecosystem service functions within the city.

Data availability statement

The data presented in this study are available on request from the first author.

Acknowledgments

The authors would like to thank the support of the International Joint Laboratory of Landscape Architecture, Henan Agricultural University, for their infinite help.

This research was funded by the National Natural Science Foundation of China (grant number: 31600579), Key Technology Program of Henan Province (grant number:

162102310093), 2020 Training Program for Young Backbone Teachers in Higher Education Institutions in Henan Province "Impact of Multi-scale Green Space Planning and Design on Public Health" (grant number: 2020GGJS049), International cooperation research program of Henan province (grant number: HNGD2021035), Research on the whole process of online and offline hybrid international joint training model for graduate students in Landscape Architecture (grant number: 2021SJGLX162Y), and International Joint Laboratory of Landscape Architecture in Henan Province.

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Appendix 1

Table A1. Description of landscape classification in Zhengzhou City.

Code	Classification	Description
10	Cropland	Land used for growing crops, including paddy fields, irrigated dry land, rain-fed dry land, vegetable land, pasture land, greenhouse land, land with fruit trees and other economic trees between mainly planted crops, as well as tea plantations, coffee plantations and other shrubs for cash crops.
20	Forest	Land covered by trees with more than 30% canopy cover, including deciduous broadleaf forest, evergreen broadleaf forest, deciduous coniferous forest, evergreen coniferous forest, mixed forest, and open forest land with a canopy cover of 10–30%.
30	Grassland	Land covered by natural herbaceous vegetation with a cover greater than 10%, including grasslands, meadows, savannas, desert grasslands, and urban artificial grasslands, etc.
50	Wetlands	Land located in the border zone between land and water, with shallow standing water or excessively wet soil, mostly with boggy or wet plants growing. Includes inland bogs, lake bogs, river floodplain wetlands, forest/shrub wetlands, peat bogs, mangroves, salt marshes, etc.
60	Water area	The area covered by liquid water in the land area, including rivers, lakes, reservoirs, ponds, etc.
80	Construction land	The surface formed by artificial construction activities, including towns and other types of residential land, industrial and mining, transportation facilities, etc., excluding continuous green areas and water bodies within the construction site.

Appendix 2

Table A2. Calculation formula of impact factors.

Abbreviation	Metrics	Calculation formula	Notes
ED	Edge density	$ED = \frac{E}{A}$	E is the total edge length of the patches within the landscape; A is the total area of the landscape. P_i is the percentage of area occupied by type i patches; g_{ik} is the number of type i patches and type k patches adjacent to each other; m is the total number of landscape patch types. NP is the number of patches.
CONTAG	Contagion index	$CONTAG = \left[1 + \frac{\sum_{i=1}^{m}\sum_{k=1}^{m}\left[(P_i)\left(\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right)\right]\left[\ln(P_i)\left(\frac{g_{ik}}{\sum_{k=1}^{m} g_{ik}}\right)\right]}{2\ln(m)}\right]$	
SHDI	Shannon's diversity index	$SHDI = -\sum_{i=1}^{m}(P_j)(\log_2 P_j)$	
PD	Patch density	$PD = \frac{NP}{A}$	
POP	Population density	$POP = \frac{r}{S}$	r is the population size; S is the area.
LUR	Land urbanization rate	$LUR = \frac{ul + il + tl}{ul + il + tl + rl}$	ul is the scale of urban land use; il is the scale of industrial and mining land use; tl is the scale of transportation land use; rl is the scale of rural settlement land use.

Appendix 3

Table A3. The interactive types of two factors and the description.

Interactive Types	Description
$q(x_1 \cap x_2) > q(x_1) + q(x_2)$	Nonlinearly enhanced
$q(x_1 \cap x_2) = q(x_1) + q(x_2)$	Independent
$q(x_1 \cap x_2) > \text{Max}(q(x_1), q(x_2))$	Bilinearly enhanced
$\text{Min}(q(x_1), q(x_2)) < q(x_1 \cap x_2) < \text{Max}(q(x_1), q(x_2))$	Unique nonlinearly weakened
$q(x_1 \cap x_2) < \text{Min}(q(x_1), q(x_2))$	Nonlinearly weakened

Supplementary material 1

Notes on the data

Authors: Mengqi Zhao

Data type: pdf. file

Explanation note: This file contains link to download the datas used in the paper and the description of the datas.

Link: https://doi.org/10.3897/natureconservation.48.85179.suppl1