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Seeking consensus in German forest conservation: An analysis of contemporary concepts

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Abstract

Setting operational conservation objectives is a major challenge for effective biodiversity conservation worldwide. To analyse forest conservation objectives in Germany in a transparent manner and to achieve a consistent and consensual framework, we systematically classified conservation objectives suggested in concepts by different stakeholders. We analysed 79 biodiversity and forest conservation concepts of different stakeholder groups at various scales and applied textual content analysis and Dirichlet regression to reach a high degree of transferability and applicability. Our analysis revealed a broad consensus concerning forest conservation across stakeholders and scales, albeit with slight differences in focus, but we detected a scale-related mismatch. A wide array of conservation objectives covered social, biotic and abiotic natural resources. Conservation of species, ecosystems and structural elements in forests were found to be of primary importance across stakeholders and scale levels. Shortcomings in the conservation concepts were found in addressing genetic diversity, abiotic resources and socio-cultural objectives. Our results show that problems in forest conservation may be rooted in trade-offs between aims, targeting mismatch across scale levels and insufficient implementation of objectives.

Keywords

biodiversity, conservation concepts, conservation objectives, Dirichlet regression, forest conservation, stakeholders, spatial scales, scale mismatch, targets

Introduction

Twenty-five years after coming into force, the targets of the Convention on Biological Diversity (CBD) are yet to be reached. National and international strategy papers on nature conservation and sustainability have proliferated in the meantime (BMU 2018; Hagerman and Pelai 2016). However, implementation is often controversial and not all measures have been successful in achieving CBD targets. There is general agreement amongst conservationists, that biodiversity and its services to human well-being are still at high risk and that many actions have not succeeded in reducing these risks. For instance, Tittensor et al. (2014) concluded that, by 2020, the pressures affecting biodiversity will still be increasing and Ripple et al. (2017) warned that the global state of biodiversity conservation is more than worrying. Human-induced biodiversity loss is a matter of concern for all societal groups and from global to local levels (Masood 2018). It is beyond doubt that biodiversity decline is driven chiefly by unbridled habitat destruction and land-use intensification (Vellend et al. 2017; Tittensor et al. 2014; CBD 2010; Millennium Ecosystem Assessment 2005).

Effective conservation needs a consistent and comprehensive framework of conservation objectives. Such a framework should aim at preserving wildlife species, as well as ecosystems as a whole. Moreover, the sustainable production and use of natural products such as food, timber, minerals and other resources for human needs, as well as the non-material benefits of recreation, amenity, culture and science, are to be considered (Harley 1977). Perrings et al. (2011) emphasised that frameworks should indeed reflect and consider human well-being and the benefits people enjoy and gain by protecting biodiversity and securing its ecosystem services. To enhance biodiversity-friendly land-use, it is crucial to develop nested knowledge systems (Cornell et al. 2013), which are harmonised across scales and groups of stakeholders (Peterson et al. 2018).

The limited success of nature conservation efforts can also be attributed to scale mismatches within frameworks of conservation objectives (Guerrero et al. 2013). Scale mismatches (temporal, functional or spatial) arise when social-ecological functions are disrupted across the scales of the managing social and environmental organisations and when environmental problems are the result of mismatches between the scales of human responsibility and natural resources (Cumming et al. 2006; Lee 1993). Within stakeholder groups (e.g. administrations, conservation associations, forest enterprises), conservation objectives should ideally be nested and harmonised across scales, enabling unimpeded conceptual transfer and exchange of knowledge. As ecological processes and ecosystem functions vary across scales (Peterson et al. 1998), overcoming scale mismatches is of particular importance for the successful implementation of conservation objectives (Ahlborg and Nightingale 2012; Paloniemi et al. 2012). It is essential to reveal framework inconsistencies and whether conservation objectives deviate amongst stakeholders and between spatial scales and, if so, in which respect (Guerrero et al. 2013). Several studies found that insufficient definitions of objectives and inconsistencies in frameworks are major obstacles for effective nature conservation (Butchart et al. 2016; Meyer et al. 2016; Maxwell et al. 2015; Stafford-Smith 2014; Marquard et al.

2013; Heink and Kowarik 2010; Kapos et al. 2008; Tear et al. 2005). Different stakeholder expectations may be a major reason for such deficiencies. This study aims at bridging these obstacles by providing a conceptual contribution to the ongoing debate in nature conservation.

Multiple approaches exist to frame nature conservation, provide tools and justify actions (Mace 2014). The People and Nature approach tries to encompass ideas and disciplines by interrelating the protection of nature with the services it provides for human well-being (Carpenter et al. 2009; Mace 2014). In contrast, the Nature's Contribution to People approach, developed by the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES), and the Provisioning, Regulating and Maintenance and Cultural Ecosystem Services (CICES; Haines-Young and Potschin-Young 2018), look at nature conservation in a more utilitarian way. These approaches have a wider focus than the general ecosystem service framework (Millennium Ecosystem Assessment 2005), as they also include social and cultural standards (Díaz et al. 2018).

In fulfilment of the obligations of CBD, article 6, Germany adopted a National Strategy on Biological Diversity, comprising 330 targets and around 430 measures (BMUB 2007), many of them involving forest ecosystems. Subsequently, individual German federal states as well as state forestry enterprises and non-governmental organisations published separate regional biodiversity and nature conservation concepts. In Germany, where forests cover approximately one third of the land area, close-to-nature forest management and sustainable use of forest products are priority components of these concepts. With respect to forest conservation, our overall objectives were

(1) to derive a comprehensive and conceptual reference framework of German forest conservation objectives based on contemporary concepts to classify and systematically analyse the conservation objectives in terms of completeness and consistency;
(2) to reapply the results to conservation concepts in search of commonalities and differences and to examine the comprehensive nature of concepts.

Assuming a wide range of different interests, we hypothesised considerable variation between the conservation concepts and objectives provided by different stakeholders but, nevertheless, scale-independent consensus (meaning a balanced knowledge transfer) within particular groups of stakeholders. Another aim was thus to verify unimpeded conceptual transfer of knowledge within stakeholder groups across scales.

Methods

Deriving a reference framework of conservation objectives

We define a conservation objective (CO hereafter) as the combination of a physical object of conservation, e.g. organisms, biotopes, soil or water resources and the properties of its desired state (target). We derived a comprehensive reference framework of COs

Table 1. Classification framework of conservation objectives (for a detailed list see Suppl. material 1: Table S1).

Level	Conservation objective	Specification	Code
1	General field of conservation	Socio-political	S
		Nature conservation *sensu stricto*	N
2	Field of natural resources	Abiotic environment	A
		Biotic environment	B
3	Mainly abiotic targets	Soil	S
		Water	W
		Climate	C
	Mainly biotic targets	Genetic diversity	G
		Species	S
		Ecosystems and biotopes	E
		Landscapes	L
4	Categories of natural resources	Processes	P
		Structures, elements	S
		Functions = cross-connecting various levels	F
5	Qualities and properties of natural resources	Diversity	D
		"Typicalness"	T
		Completeness, integrity	C
6	Management dependency	Self-sustaining	S
		Management-dependent, culture-bound	M

by referring to the CBD (United Nations 1992a) and the German Nature Conservation and Landscape Management Act (BNatSchG, as amended on 29 July 2009). The BNatSchG, in its Article 1 (1), defines the purpose of nature conservation and landscape management as to "permanently safeguard (1) biological diversity, (2) the performance and functioning of the balance of nature, including the ability of natural resources to regenerate and lend themselves to sustainable use and (3) the diversity, characteristic features and beauty of nature and landscape, as well as their recreational value" (BMU 2010). According to both CBD and BNatSchG, biological diversity is defined as the variability amongst living organisms, terrestrial, marine and freshwater and the ecological complexes of which they are part; this includes interactions within species, between species and communities, ecosystems and biotopes (United Nations 1992a).

For each objective, we defined six levels of potential hierarchical classification depth of COs (Table 1). Relationships between levels of COs were understood as functions and indicated separately. Each single observation within the framework of COs was described as a target. For instance, the target "forest bog ecosystem" was described by the cross-connected code NBEF(NAC), as bogs are ecosystems functioning as important long-term carbon sinks (Moore and Knowles 1989), hence contributing to climate protection. With this approach, we identified and described even rather complex and interlinked relationships, reflecting multi-layered environmental patterns and processes. Each single target received a code (a combination of letters) representing a certain level of the framework of COs.

At the first level of differentiation (general field of conservation), COs were classified into the categories socio-political (e.g. recreation, enhancement of tourism, stimulating financial funding for conservation, legal issues, awareness-raising) or nature conservation *sensu stricto*. For socio-political COs, no further differentiation was deemed necessary, but cross-connections were possible (Suppl. material 1: Table S1). COs of nature conservation *sensu stricto* were grouped into abiotic and biotic objectives. The latter were further grouped to cover genetic, species and ecosystem diversity (in accordance with the CBD) and landscape diversity, as this is stressed in the BNatSchG. Our differentiation of abiotic and biotic natural resources is compatible with the CICES themes and classes of ecosystem services: provisioning, regulating and maintenance and cultural (Haines-Young and Potschin-Young 2018; Haines-Young and Potschin 2011).

To give each objective more detail, we developed further levels concerning categories of natural resources, qualities and conditions of existence (Table 1). We distinguished between COs related to processes, structures or functions and further, by CO addressing diversity as such, typical features or integrity/intactness. At the final level, we differentiated between self-sustaining and management-dependent systems.

A specific code was assigned to each CO (Suppl. material 1: Table S1). However, as the classification system had to deploy an operational level, some specific targets fall under the same generalised category and could not be detected separately. The code NBESTS, for example, comprises all targets concerning self-sustaining ecosystem structures.

Finally, individual target keywords were added to address more specific cases. For instance, the code NBESCS, addressing the integrity of self-sustaining ecosystems, was further detailed by the target keyword "protection of beech forest ecosystems". A detailed list of all target keywords and their assigned codes can be found in Suppl. material 1: Table S2.

Textual content analysis

We conducted textual content analyses of 79 biodiversity and forest conservation concepts (for a detailed list of concepts, see Suppl. material 1: Table S3). The concepts were collected via web-based literature research on the websites of different stakeholders. We selected and gathered all current concepts and strategies published until 2016, covering all relevant stakeholder groups. Single forest owners or private forest enterprises were not analysed, as they did not develop their own valid forest conservation concepts. Furthermore local or municipal groups were excluded as well to ensure comparability amongst all stakeholders.

We classified the stakeholders into three pre-defined groups; administrative institutions (e.g. ministries), nature conservation NGOs and state forestry enterprises (Table 2). Furthermore, each concept was assigned to a specific concept type: general nature and biodiversity conservation related concepts; specific forest conservation concepts; concepts addressing forest management and silviculture; general forest programmes; and specific concepts addressing veteran tree and deadwood management.

In terms of scale, the concepts were referable to international, national (Germany) or regional (federal states) levels (Table 2). For the definition of scale, we refer to Gibson et al. (2000) and Cash et al. (2006), who state that scale has many different

Table 2. Categorisation of concepts with their abbreviations (Abbr.) and numbers of concepts per stakeholder group and jurisdictional scale level (Int = International, Nat = National, Reg = Regional).

Stakeholder	Abbr.	Concept type	Jurisdictional scale levels		
			Int	Nat	Reg
Concepts published by administrative or governmental institutions (e.g. ministries)	Instit	Biodiversity	3	2	14
		Forest conservation	1	–	2
		Forest management	–	–	3
		Forest programme	–	1	4
		Veteran trees and deadwood	–	–	–
Concepts originated under the leadership of state forestry enterprises	StateF	Biodiversity	–	–	–
		Forest conservation	–	–	10
		Forest management	–	–	14
		Forest programme	–	–	2
		Veteran trees and deadwood	–	–	6
Concepts published by environmental and nature conservation NGOs	NGO	Biodiversity	–	1	1
		Forest conservation	–	8	4
		Forest management	–	1	–
		Forest programme	–	–	1
		Veteran trees and deadwood	–	–	1

dimensions (e.g. spatial, temporal, jurisdictional, institutional), each having different levels, “units of analysis that are located at different positions on a scale” (Cash et al. 2006). The international, national and regional levels refer to the jurisdictional scales (administrations) (Cash et al. 2006).

Textual content analysis was used to identify and interpret the COs. Content analysis is a standard research method in social sciences and is used to gather and scrutinise text, the content of which “can be words, meanings, pictures, symbols, ideas, themes or any communicated message” (Neuman 2014). Qualitative (descriptive) and quantitative (numerical) content analyses can be distinguished and the former may be “defined as a research method for the subjective interpretation of the content of text data through the systematic classification process of coding and identifying themes or patterns” (Hsieh and Shannon 2005). To ensure scientific transparency and reproducibility throughout the process of content analysis, all concepts were read twice. During that second stage, falsely assigned COs were reassigned to another code and neglected ones were newly described. Each identified CO was categorised according to the classification system.

Data analysis

For each concept, all individual CO code assignments were treated as single observations and each hierarchical level of classification (Table 1) was analysed separately. The relative importance of a certain objective was determined by dividing the number of targets assigned to the CO by the overall sum of targets registered in the concept. This procedure generates vectors of shares of targets, distributed along the CO. Each vector contains non-independent elements and must be treated as one observation per concept. We used Dirichlet distribution as a statistical model suitable for describing the mechanisms underlying such observations. Dirichlet regression (Maier 2014) is a statistical method for working out differences in the expected composition of such vectors – that is, the collection of expected values (EV) of the vector elements – according to differences in explanatory variables. Presented results are based on the estimated EV and their uncertainties quantified in uncertainty intervals. If one of the observed vectors – belonging to one specific CO – contains an element that has a value of zero, this CO had no target mentioned in a concept. As all concepts in this study are related to nature conservation, we assumed that each of these underlie minimal shares of concern for each CO. Based on this assumption, we treated zero observations as "rounded zeros" (Martín-Fernández et al. 2003), which enabled us to lift zero values up to small positive values based on the transformation proposed by Maier (2014). This makes the use of Dirichlet regression possible, as it requires values between 0 and 1. We ran the Dirichlet regression model (Maier 2014) with concept type as categorical explanatory variables. All analyses were performed using the STAN Bayesian inference environment (Carpenter et al. 2017). For technical details of the model fitting process, see Sennhenn-Reulen (2018). Of the several prior choices described by Sennhenn-Reulen (2018), we used the N (0.5) prior for all model coefficients. Results are displayed as posterior means (Jaynes 2003) in percentages. With this standardised method, the relevance for forest conservation of each CO level was ensured for all concepts. Furthermore, the motivation for protecting and securing forest biodiversity of each stakeholder group could be assessed and evaluated.

With respect to orthogonality, it is critical that not all stakeholders are represented on all jurisdictional levels (Table 2). Thus, the analysis of the effect of the stakeholder group was conducted only at the regional level, reducing the sample size to 62 concepts. To analyse the effect of scale, only administrative concepts were assessed, reducing sample size to 30. In this stakeholder group, we expected content-related harmonisation across the levels.

To further analyse the degree of specification within the stakeholder group of administrative institutions, a level-of-detail-analysis was conducted. To allow for sufficient specification, we restricted the analysis to biotic COs (genes, species, ecosystems and landscape, see Table 1) at the third level. The level of detail was equal to the maximum hierarchical level reached (Table 3). The analysis was conducted for each biotic CO separately and mean specification degrees were calculated for each concept. For the analysis of the keywords, counts or mentions (presence/absence) per concept were calculated.

Table 3. Specification degree of conservation objectives.

Conservation objective	Level of Detail
No further specification of biotic objective	0
Categories of natural resources	1
Qualities and properties of natural resources	2
Management dependency	3

Results

Commonalities and differences between conservation concepts

The textual content analysis of 79 concepts revealed a broad range of single COs. In total, 170 individual targets (keyword combinations) were detected, with between 14 and 85 (mean 50) targets per concept. On average, a single concept covered 30% of the overall number of targets.

All stakeholders clearly prefer nature conservation *sensu stricto* instead of socio-political COs (Table 4). The EV for the social-political targets ranged between 8% and 11%. The highest values were found in the concepts of nature conservation NGOs (NGO) and administrative-governmental institutions (Instit), the latter significantly differing from state forestry enterprises (StateF). The highest percentages, albeit insignificant, of socio-political targets were found in national and international concepts.

Our results show that COs consider protecting the biotic environment generally more important than abiotic resources (Table 4). Even though biotic targets are pursued at all spatial scales, regional institutions have significantly higher percentage values than international institutions.

Ecosystem and species diversity are the main biotic COs in all analysed concepts, followed by, but with considerably lower percentages, the protection of landscape elements (Figures 1 and 2). In contrast, the protection of genetic diversity and of all elements of abiotic resources (soil, water and climate) is considered as of minor relevance. Within regional stakeholders (Figure 1), Instit had significantly lower proportions for the most frequently mentioned targets (protection of ecosystem and species diversity) than NGO and StateF. Regarding the protection of landscape diversity, Instit concepts had significantly higher values than the other stakeholder groups. Targets for the protection of soil, water, climate and genetic diversity were scarcely mentioned by all stakeholder groups, with EV mainly lower than 5%. Apart from soil-related COs, where Instit had lower proportions than the other two groups, no significant differences were found between the stakeholder groups. However, this difference is based on lower sample size and not discussed further.

Regarding the scale effect, regional concepts exhibited a smaller range than the other levels (Figure 2). For international institutions, the protection of ecosystems turned out to be significantly less important than for national and regional institutions. Species and ecosystem protection were similarly relevant in international concepts, whereas in national or regional concepts, the protection of species was less frequently mentioned. The protection of landscape elements was found to be of minor importance at all levels. With decreasing scale level, the necessity for protecting genetic diversity and abiotic resources was noted decreasingly, although this effect was not significant.

The results concerning the category (Table 5), quality (Table 6) and conditions of existence (Table 7) showed that the general focus in all concepts – regardless of the specific stakeholder group or scale level – lies in protecting diverse and naturally self-sustaining structures of forest ecosystems. Targets for the protection of processes or natural dynam-

Table 4. Proportions (expected values, in %) of the first and second classification level of conservation objectives.

		General field of conservation			Natural resources		
		Socio-political	Nature conservation	*	Abiotic	Biotic	*
Regional stakeholder	Instit (n=23)	10.3	89.7	*a*	8.0	92.0	*a*
	NGO (n=7)	11.1	88.9	*ab*	6.3	93.7	*a*
	StateF (n=32)	7.6	92.4	*b*	6.5	93.5	*a*
Jurisdictional scale	Int (n=4)	13.7	86.3	*a*	14.2	85.8	*a*
	Nat (n=3)	13.1	86.9	*a*	11.6	88.5	*ab*
	Reg (n=23)	9.5	90.6	*a*	6.9	93.1	*b*

Instit = Administrative-governmental institutions, NGO = Non-governmental organisations or nature conservation associations, StateF = State forestry enterprises, Int = International, Nat = National, Reg = Regional, * = different letters indicate significant differences between stakeholder groups and between scale levels.

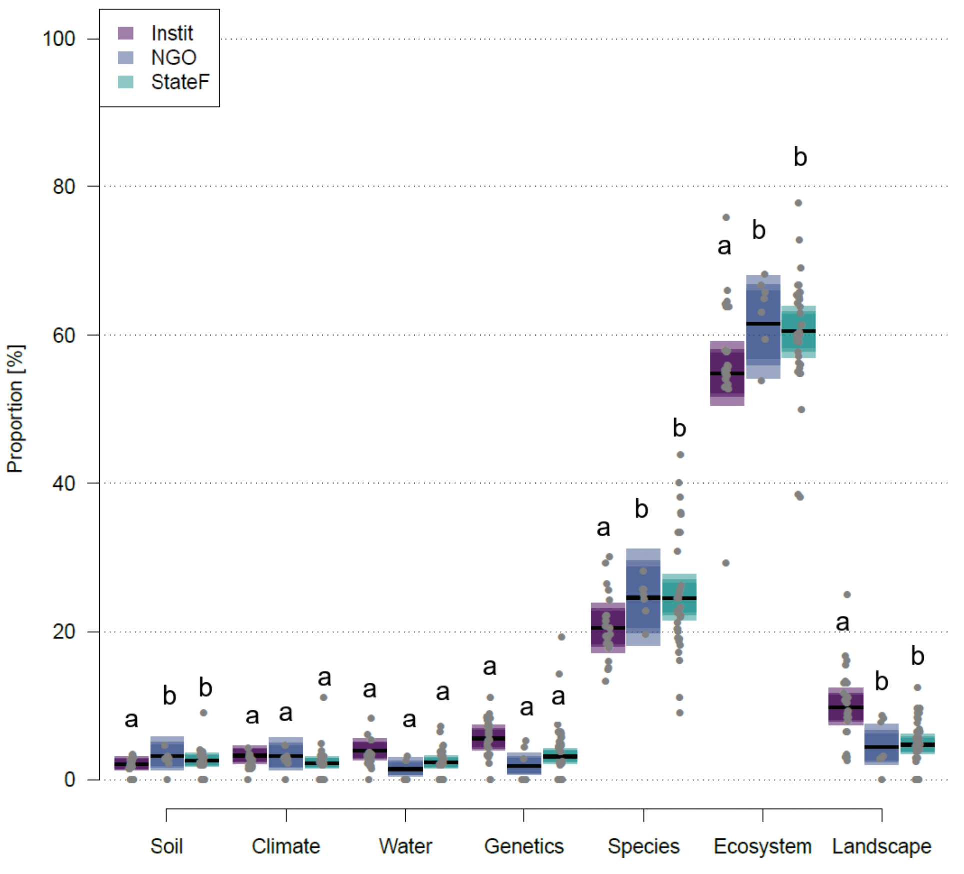

Figure 1. Stakeholder impact – posterior means for the third level of COs for the three stakeholder groups (n = 62). Different letters indicate significant differences between stakeholder groups (Instit = administrative-governmental institutions, NGO = environment or nature conservation NGOs, StateF = State forestry enterprises). Displayed are the expected value (black line), the 99% (light), the 95% (medium) and the 90% (dark) uncertainty intervals.

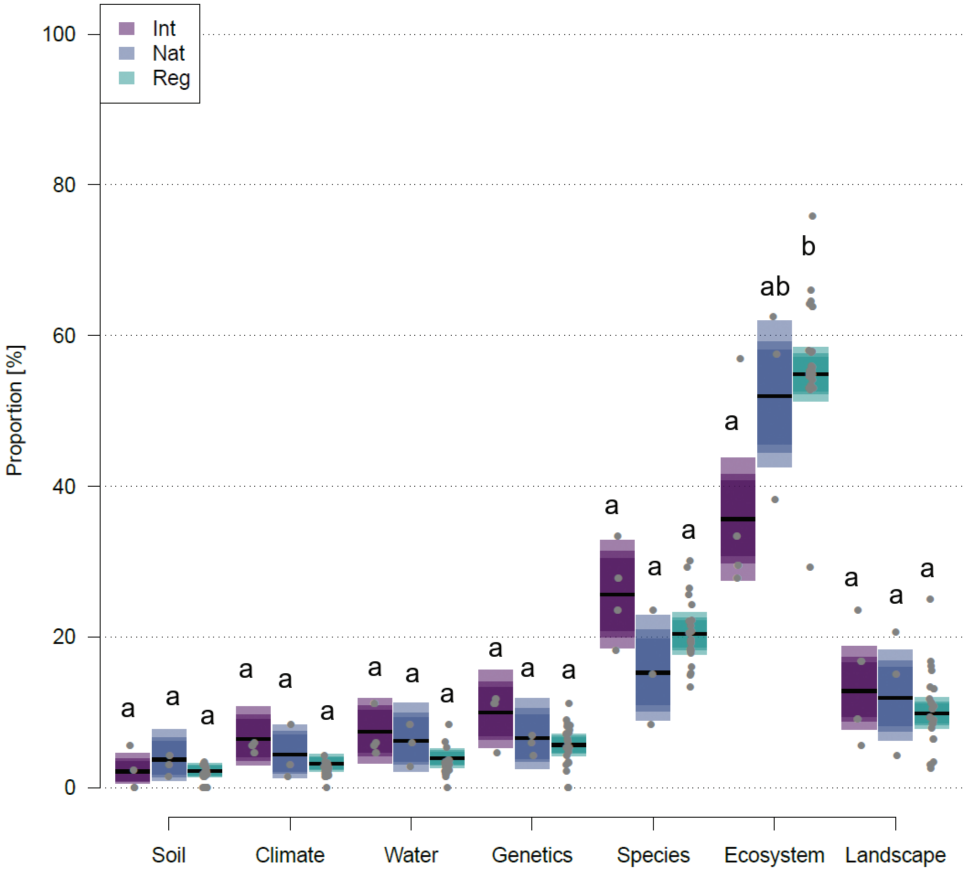

Figure 2. Jurisdictional scale effect – posterior means for the third level of COs for the three spatial scales (n = 30). Different letters indicate significant differences between scales (Int = International, Nat = National, Reg = Regional). Displayed are the expected value (black line), the 99% (light), the 95% (medium) and the 90% (dark) uncertainty intervals.

ics (fourth level: e.g. natural forest cycles; natural forest regeneration; habitat continuity) were the least mentioned by the stakeholders, with NGO having significantly higher percentage values than Instit and StateF (Table 5). For international institutions significantly lower values regarding natural dynamics protection were found than for regional ones.

The significantly highest percentages of targets with functions/cross-connections to other CO levels were found in international concepts. Cross-connections were either in relation to socio-political targets (e.g. a social responsibility to protect species; forest habitats as a place for recreation and tourism) or to abiotic targets (e.g. preservation or development of climate-resilient forest stands; water supply by forests). Here, StateF had significantly lower percentages than NGO. In general, protecting particular elements and structures (e.g. specific forest or species communities; habitat trees; biotope

Table 5. Proportions (EV, in %) of the fourth level to describe the categories of conservation objectives.

		Functions/ cross-connections	*	Processes	*	Structures, elements	*
Regional stakeholder	Instit (n=23)	24.7	*ab*	14.0	*a*	61.3	*a*
	NGO (n=7)	28.3	*a*	20.8	*b*	50.9	*b*
	StateF (n=32)	22.6	*b*	13.9	*a*	63.5	*a*
Jurisdictional scale	Int (n=4)	49.4	*a*	3.3	*a*	47.4	*a*
	Nat (n=3)	30.3	*b*	12.3	*ab*	57.5	*ab*
	Reg (n=23)	25.0	*b*	14.6	*b*	60.4	*b*

Instit = administrative-governmental institutions, NGO = environmental and nature conservation NGOs, StateF = State forestry enterprises, Int = International, Nat = National, Reg = Regional, * = different letters indicate significant differences between stakeholder groups and between scale levels.

Table 6. Proportions (EV, in %) of the fifth level to describe the qualities of conservation objectives.

		Diversity	*	"Typicalness"	*	Completeness	*
Regional stakeholder	Instit (n=23)	53.6	*a*	40.7	*a*	5.6	*a*
	NGO (n=7)	47.2	*a*	49.5	*a*	3.3	*a*
	StateF (n=32)	56.2	*a*	39.9	*a*	3.9	*a*
Jurisdictional scale	Int (n=4)	76.7	*a*	20.7	*a*	2.6	*a*
	Nat (n=3)	46.5	*b*	43.1	*a*	10.5	*b*
	Reg (n=23)	53.6	*b*	40.6	*a*	5.8	*b*

Instit = administrative-governmental institutions, NGO = environmental and nature conservation NGOs, StateF = State forestry enterprises, Int = International, Nat = National, Reg = Regional, * = different letters indicate significant differences between stakeholder groups and between scale levels.

Table 7. Proportions (EV, in %) of the sixth level to describe the conditions of existence of conservation objectives.

		Management-dependent	Self-sustaining	*
Regional stakeholder	Instit (n=23)	22.2	77.9	*a*
	NGO (n=7)	9.2	90.9	*b*
	StateF (n=32)	20.5	79.5	*a*
Jurisdictional scale	Int (n=4)	7.4	92.6	*a*
	Nat (n=3)	25.2	74.8	*b*
	Reg (n=23)	21.5	78.5	*b*

Instit = administrative-governmental institutions, NGO = environmental and nature conservation NGOs, StateF = State forestry enterprises, Int = International, Nat = National, Reg = Regional, * = different letters indicate significant differences between stakeholder groups and between scale levels.

types; single species) plays a major role across almost all stakeholders and levels. However, StateF and Instit emphasise the protection of structural elements significantly more than NGO. This was also true at the regional level and partly so at the national level.

The fifth level describes particular qualities of COs (Table 6), focusing either on diversity (e.g. habitat or species diversity), qualitative characteristics (particular forms or features) or on attempting completeness, integrity or intactness of the CO. Such targets were commonly mentioned in all concepts. Significant differences were found between scale levels but not between stakeholder groups. At the international level, the main target was to protect a maximum degree of diversity. At national and regional levels, significantly lower percentages of this target were found. Generally, the aim to protect complete qualities of COs was found to be of relatively low priority at all levels, with the significantly lowest EV at the international level (Table 6).

On the sixth level, protecting self-sustaining biodiversity features was given priority across all stakeholders and scales (Table 7). This was particularly true for concepts by NGOs or at international level, which had the significantly highest percentage (EV) values. The maintenance of culture-bound and management-dependent systems was considered particularly important for Instit and StateF. Within institutions, it is more often addressed at the national and regional than at international level.

Degree of specification for administrative concepts

We assumed that the degree of specification would increase from the international to the regional level. However, this was not the case for COs related to genetic diversity and only weakly so for species and landscape diversity (Figure 3). Here, levels of detail mainly remained at the fifth overall level (Table 1). A clear, scale-dependent increase of specification could only be confirmed for the CO ecosystems. With respect to the CO landscape, the range is prominently higher at the regional than at the national and international levels.

Assessment of forest conservation target keywords

We distinguished a total of 107 target keywords in the concepts (Suppl. material 1: Table S3). While concepts of international administrations cover only 18% of all possible keywords, national ones included 40% and regional ones 44%. NGO and StateF generally cover about 30% and Instit 44% of all possible keywords. The protection of habitats was the most frequently mentioned target included in all concepts (Table 8). Targets such as the maintenance of deadwood in forest ecosystems, sustainable forestry, the social obligation to protect and secure species habitats, the implementation of a close-to-nature forest management and the protection of habitat trees were also very frequently mentioned. With on average approximately 60 mentions, the preservation of protected areas, as well as of habitats and species in the EU Natura 2000 network of conservation areas, also played a major role in the concepts. Keywords concerning the protection of particular forest biotopes (e.g. wooded heathland or fir forests) and of forest attributes with carbon sink functions (e.g. deadwood and old-growth forests) were comparatively rarely mentioned.

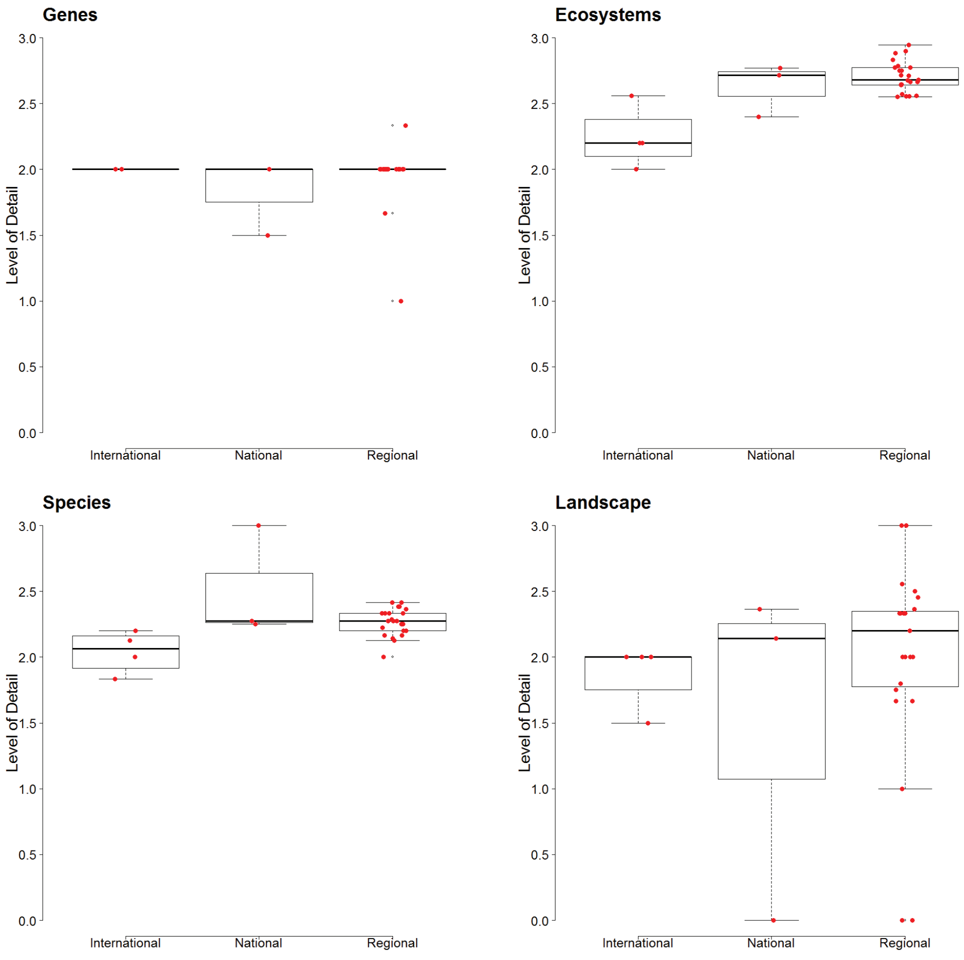

Figure 3. Level of detail (specification degree) for the four elements of biodiversity, genes, species, ecosystems and landscape, in relation to their scale levels (international n = 4, national n = 3 and regional n = 23).

Certain differences between administrative-governmental concepts (found at all scale levels) and between regional concepts (found in all different stakeholder groups) are worth mentioning. Regional concepts pay more attention to the protection of specific forest elements, such as habitat trees, deadwood-dependent species and old-growth forests. Administrative-governmental concepts, on the other hand, stress the importance of landscape- and connection-related elements, such as biotope networks, species stepping stones and riverine systems, while emphasising the need to finance forest conservation. Although not shown in Table 8, some keywords were non-exclusively claimed by all members of a specific stakeholder group or scale level. International institutions invariably mentioned habitat protection, sustainable forestry and ecosystem services. Likewise, national institutions all claimed sustainable forestry, biotope

Table 8. Absolute and percentage frequency of the most important keywords for all concepts, for administrative-governmental concepts at all levels and for regional concepts of all stakeholder groups, respectively (only keywords with > 40 mentions for all concepts are listed).

Keyword	All concepts (n = 79)	%	Administrative concepts (n = 30)	%	Regional concepts (n = 62)	%
Habitat protection	75	94.9	28	93.3	59	95.2
Deadwood in forest ecosystems	67	84.8	24	80.0	53	85.5
Sustainable forestry	65	82.3	27	90.0	50	80.6
Social obligation for habitat protection	65	82.3	21	70.0	51	82.3
Close-to-nature forestry	63	79.7	23	76.7	52	83.9
Habitat trees	63	79.7	20	66.7	52	83.9
Protected areas	62	78.5	26	86.7	46	74.2
Natura 2000 habitats	61	77.2	25	83.3	49	79.0
Natura 2000 species	59	74.7	24	80.0	47	75.8
Near-natural forests	59	74.7	24	80.0	46	74.2
Rare species	59	74.7	20	66.7	46	74.2
Forest structures	58	73.4	19	63.3	48	77.4
Naturally developing forests	58	73.4	20	66.7	46	74.2
Natural regeneration	54	68.4	19	63.3	45	72.6
Hunting	53	67.1	22	73.3	43	69.4
Natural forest reserves	52	65.8	18	60.0	42	67.7
Biotope network	51	64.6	26	86.7	40	64.5
Wetlands	51	64.6	20	66.7	41	66.1
Deadwood-dependent species	49	62.0	15	50.0	42	67.7
Forests developing stages	49	62.0	17	56.7	42	67.7
Old-growth forest	49	62.0	16	53.3	39	62.9
Species stepping stones	49	62.0	20	66.7	37	59.7
Forest edges	48	60.8	16	53.3	39	62.9
Beech forests	46	58.2	16	53.3	37	59.7
Mixed forests	46	58.2	22	73.3	41	66.1
Rare tree species	45	57.0	16	53.3	38	61.3
Bogs	44	55.7	21	70.0	38	61.3
Riverine systems	44	55.7	24	80.0	36	58.1
Traditional forest management	44	55.7	16	53.3	36	58.1
Certification	42	53.2	19	63.3	32	51.6
Forest conservation financing	42	53.2	18	60.0	28	45.2

networks and the maintenance of protected areas, wildlife species and near-natural forests. All NGOs pursue the purpose of habitat protection, protecting natural forest development and designating protected areas. Regional concepts emphasise specific forest conservation related keywords of local scope, such as the protection of deadwood and habitat trees, as well as close-to-nature forestry. This was particularly true for StateF and NGO. In the concepts of regional institutions, more general nature conservation statements were made, such as protecting Natura 2000 habitats and expanding biotope networks.

Discussion

Deriving and applying frameworks of conservation objectives

Many researchers examined and reviewed nature conservation concepts in general and the implementation of nature and forest conservation objectives in particular (Ulloa et al. 2018; Morales-Hidalgo et al. 2015; Moilanen et al. 2014; Pullin and Stewart 2006; Pullin et al. 2004; Sutherland et al. 2004). Amongst their findings was that it requires interdisciplinary collaboration, the integration of all fields of biodiversity research and a unifying frame of reference to be effective in conservation. As there is no review of forest conservation that could be used as a generalised reference frame, the framework of forest COs we derived may serve as such a reference system and moreover contribute to an improved communication of this often emotionally discussed topic (Meyer 2013; Winkel et al. 2005; Scherzinger 1996).

The framework proved suitable in reviewing 79 concepts of different stakeholder groups and across different scale levels. Universal validity with respect to German nature conservation in forests is achieved due to the fact that our analysis is firmly based on the common ground of the CBD and the BNatSchG. The frame may be used to encompass all possible objectives in nature conservation and cultural and natural objectives alike. It may be adopted in various fields of conservation science, despite its presently narrow focus on German forests. Our framework is in line with the initially-mentioned approaches to widely conceive nature conservation (CICES, People and Nature, Nature's Contribution to People). It is, however, constrained to an overall level, requiring further implementation in practice.

The assignment of keywords helps to acquire higher degrees of detail and to overcome the disadvantage of abstraction and is important in specifying COs, making the framework more applicable. Nevertheless, some constraints remain, as further implementation also means setting priorities and identifying synergies or trade-offs between single COs and hierarchical levels. This process, however, defies generalisation, as additional criteria need to be evaluated, such as the local or regional conservation status or the level of protection already gained. Thus, priority setting and the identification of trade-offs are not included in our framework of COs. However, the functional relationships can be regarded as an indication of existing synergies.

Commonalities and differences amongst forest conservation concepts

Our analyses of forest COs show that, in general, there is a broad consensus concerning forest conservation amongst different stakeholders in Germany. A wide variety of targets was found, covering social, biotic and abiotic natural resources. All stakeholder groups emphasised the protection and maintenance of diverse and self-sustaining structures, forest ecosystems, species and natural forest elements. Genetic diversity, landscape elements and abiotic resources are less considered. However, apart from this detected consensus amongst stakeholders and across scales, some differences in prioritising conservation objectives were identified, which do not fully accord with a comprehensive approach to nature conservation. The preamble of the CBD in 1992 already recognised the importance of comprehensive nature conservation concepts in postulating that the contracting parties are "conscious of the intrinsic value of biological diversity and of the ecological, genetic, social, economic, scientific, educational, cultural, recreational and aesthetic values of biological diversity and its components" (United Nations 1992a).

National and international administrations take more account of social demands and the protection of abiotic resources. Since abiotic resources and their regulating services are an essential part of the natural environment (Dewulf et al. 2015), their protection and maintenance is crucial for the sustainable development and use of global biodiversity, including all elements of ecology, economy and society (United Nations 1992b). As the conservation of abiotic natural resources is scarcely mentioned by most stakeholders, conservation efforts in this field could be intensified. For internationally orientated concepts, the percentages found at the third level of COs (climate, soil, water, genes, species, ecosystems and landscape) were more balanced, underlining their more encompassing scope and validity. Although regional stakeholders consider the protection of landscape diversity more than others, COs concerning the protection of landscape and its components were rarely represented. Our results, concerning the under-representation of landscape protection and social-political requirements in the concepts, are in accordance with Petereit et al. (2017), who analysed the implementation of nature conservation in public forests in a manner analogous to ours. Their findings show that the main forest conservation target in concepts was the maintenance of biodiversity in general and that targets for the protection of natural resources were of marginal importance. Securing landscape and recreational values were the least claimed targets.

On the whole, concepts with a wider scale level turned out to be more balanced and consider functional relations. Regional concepts focus on concerns to be tackled by approved forest conservation methods and are more aware of management-dependent systems. Nevertheless, our results demonstrate that there is a lack of focus on the maintenance of culture-bound and management-dependent COs (e.g. cultural heritage and management-related habitat tradition). Even state forestry enterprises focus on natural and self-sustaining ecosystems, although initially we assumed they would pay more attention to management-dependent systems.

For an effective forest biodiversity conservation, it is important to identify synergies and trade-offs (Di Marco et al. 2016; Perrings et al. 2010). Our analyses of biodiversity and forest conservation objectives showed that COs with functions/cross-connections to other levels of COs, while indeed common in some concepts, could be more frequently considered by regional stakeholders. Providing and addressing these synergies is essential for fostering biodiversity protection. Our degree-of-specification analysis within administrative-governmental concepts confirmed the expected increase in specific COs with decreasing scale level for ecosystems only. The weaker response of species and landscape COs can be neglected, as the protection of ecosystem diversity was, with few exceptions, the most common COs in the concepts. Lindenmayer and Franklin (2002) stated that preventing species loss can be achieved by preventing ecosystem loss through maintaining habitat connectivity, landscape heterogeneity and stand structural complexity. Therefore, it seems wise to lay the primary focus on the conservation and restoration of forest ecosystem diversity, which simultaneously contributes to some extent to the protection of species and genetic diversity and serves the purpose of carbon storage in forest ecosystems.

The most frequently mentioned forest conservation keywords (e.g. protecting deadwood in forest ecosystems) reflect topics recently discussed amongst forest conservationists in Germany. The differences between the concepts concerning the frequency of specific keywords are, with few exceptions, not very pronounced, supporting the detected consensus amongst stakeholders in terms of forest conservation.

Knowledge transfer within stakeholder groups and across scales

As ecosystem functions, species and ecosystem processes occur at different temporal and spatial scales (Paloniemi et al. 2012; Peterson et al. 1998), the political and societal challenges are to consider these complex and multi-dimensional processes during governmental decision-making and biodiversity conservation planning (Lee 1993). Our analysis revealed that COs considering societal obligations, e.g. environmental education for effective biodiversity conservation, are under-represented in most concepts, especially surprisingly at the regional level. This imbalance is the more astonishing, as regional stakeholders, in particular, should be aware of what is needed to reconcile the local population with nature conservation. International administrative institutions follow more general nature conservation goals and differ markedly from regional administrations. The challenging transferability of national or regional level CO, on the one hand and broader scales (Europe or worldwide) on the other, can lead to an implementation mismatch.

The detected imbalance in target-consistency prompts us to reject our hypothesis that frameworks of COs within stakeholder groups are scale-independently consensual and confirms rather a slight scale mismatch indicating possibly insufficient transfer and exchange of knowledge. One-to-one transmissions of CO set at the international level may be problematic (Guerrero et al. 2013). The EU Habitats Directive, for example, has a broad spatial range of validity and aims at the conservation of species and habitats of Community concern, many of which are vulnerable. It is implemented at the local or regional level, though, with possible bottom-up consequences (Paloniemi et al. 2012). To overcome trade-offs between aims and targeting inconsistency across scale levels, stakeholders need to stress their conceptual clarity and facilitate an unimpeded transfer and exchange of knowledge.

Conclusions

Paloniemi et al. (2012) put in a nutshell where nature conservation needs to improve on: "analyzing, understanding, and overcoming […] ecological scale-sensitivities requires combining ecological knowledge with information, awareness and experience of actors at various governance levels thus directly bridging science and policy discourses". Furthermore, it requires addressing the importance of protecting all types of ecosystems and their services within nature conservation concepts (Faith 2011; Perrings et al. 2011) as focal species and ecosystems differ in their response towards environmental changes and land-use management intensities at different scales (Nilsson 2009). Our study confirms the importance of integrating the various stakeholders, instruments and scales into conservation practices, taking into account their specific needs and requirements. With the increasing complexity of successfully implementing conservation actions across scales and different stakeholder groups, our framework of COs might qualify as a common basis for conservation priority targeting even beyond the context of German forest conservation and can help to manifest a consensual, precedential and long-term forest conservation.

Our analysis identified shortcomings concerning the unbalanced design of the concepts, where social-cultural demands and societal obligations, as well as the protection of landscape, genetic diversity and abiotic resources are not always covered adequately. These objectives might have been considered as subsidiary COs, implemented per se in the wake of ecosystem and species diversity conservation (umbrella effect). This study suggests to stakeholders that they reassess their conservation concepts in these fields. Improving the awareness of biodiversity and its values is essential to convince residents and other people concerned of the ecological and economic justification and the necessity and consequences of conservation actions.

Forest stakeholder concepts describe the purpose of conservation and restoration measures, such as to secure veteran and habitat trees, forest soil care, management of protected biotopes and species conservation programmes. The next step, specifying how to implement the measures, was taken only in 48 out of 79 concepts which provided information to this effect for certain forest COs. Without practical how-to recommendations, however, even well-founded objectives run the risk of remaining wishful thinking, a long way from implementation.

If, as our results indicate, stakeholders largely agree on the conservation objectives, the question remains why there are still considerable discrepancies in German forest conservation. Implementing forest conservation measures usually involves various stakeholders (owners, inhabitants, users, nature conservationists, administrators) with diverse and sometimes incongruent requirements. Therefore, the procedure of integrating all parties, which is so essential for the successful conservation and sustainable use of forest biodiversity, is to be improved. Mutual respect should be strengthened.

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Supplementary material 1

Tables S1–S3

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Data type: supplementary tables

Explanation note: **Table S1.** Framework for conservation objectives and its application. **Table S2.** List of all target keywords, their German equivalent and their assigned codes according to the framework of conservation objectives. **Table S3**. List of all concepts analyzed in this study with their names, references, type of concept, assigned stakeholder group, jurisdictional scale level allocation and their year of publication.

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RESEARCH ARTICLE

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Multi-decadal surveys in a Mediterranean forest reserve – do succession and isolation drive moth species richness?

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Abstract
Isolated fragments of semi-natural habitats are often embedded in a landscape matrix that is hostile to organisms of conservation concern. Such habitat islands are prone to changes in their biota over time. For insects, few studies on long-term trends in species richness within conservation areas are available, mainly due to the lack of historical data. We here use moths in the coastal pine wood reserve Pineta san Vitale (Ravenna, NE Italy) to assess how local fauna has changed over the last 85 years. This reserve has experienced massive changes in vegetation structure due to secondary succession. We compared historical collections (1933–1976: 107 species; and 1977–1996: 157 species) with our own samples (1997–2002: 174 species; and 2011+2012: 187 species). Over the last 85 years, the proportion of habitat generalists in relation to all recorded moth species increased from 20 to 33%. The fractions of woodland and open habitat species concomitantly decreased by 10 percentage points, respectively. Amongst woodland and habitat generalist species, gains outnumbered losses. In contrast, 18 species of open habitats and 10 reed species were lost over the decades. We attribute these changes to vegetation succession and to the isolation of the reserve. Generalist species are presumably better able to pass through anthropogenically exploited landscapes and colonise isolated habitat fragments than habitat specialists.

Keywords
Anthropogenic influence, biodiversity, conservation biology, generalists, habitat specialists, Italy, Lepidoptera, long-term changes, moth diversity

Introduction

Mediterranean coastal pine forests are vulnerable and rare ecosystems. They provide habitat for many species, but have experienced massive contractions over centuries and are nowadays strongly at risk through anthropogenic land-use intensification (Gasparella et al. 2017). Only few coastal pine forests have persisted on the Italian Peninsula. Some of them have been awarded high legal conservation status (Gasparella et al. 2017), but the current status of their biodiversity is often imperfectly known (Scarascia-Mugnozza et al. 2000). In general, isolated nature reserves, embedded in a landscape matrix dominated by intense human land-use, run the risk of losing over time those organisms for which they had been established (Mora and Sale 2011). Even for large-sized conservation areas, such detrimental trajectories have been documented from a range of biomes (Gauthier et al. 2015, Hautier et al. 2015, Uhl et al. 2016). Some species may persist for decades in conservation areas, but eventually get lost over time, be it due to ecosystem degradation or just attributable to stochasticity. This notion has led to the concept of an extinction debt and, only in the long term, can it be evaluated whether organisms are really safe in the reserves that have been set aside for their conservation (Carroll et al. 2004, Halley et al. 2016). Especially in areas where anthropogenic influence has been severe over decades, long term studies are of great interest because they can mirror changes in an ecosystem best and shed light on the steady erosion of biodiversity (Habel et al. 2016).

To understand long-term changes of the insect fauna in an isolated nature reserve, we investigated moth communities in Pineta san Vitale (hereafter PsV). Nowadays protected as a Natura 2000 site (Montanari 2010) and listed in the Convention of Ramsar (Ramsar Convention Secretariat 2013) as "wetland of international importance", the coastal pine forest PsV, since the year 1988, forms part of the Parco Regionale del Delta del Po which is also covered under the EU life programme (http://ec.europa.eu). Therefore it is of high interest which long term changes might have taken place in this reserve.

Originally PsV is believed to have been covered by dune vegetation that later changed to a humid forest. Around the 12th century, Italian stone pine trees (*Pinus pinea*) were planted for wood and pine nut production. Timber extraction and commercial pine nut harvest were abandoned in 1988 when the Parco Regionale del Delta del Po was established (Enrica Burioli, pers. communication). In general, Italian coastal areas with wooded dune habitats show strong signs of vegetation succession during the past decades. In particular, the grassland fractions in these areas have prominently declined since the 1960s (Prisco et al. 2016). Comparisons of old photographs with the current vegetation status also indicate that, in PsV, shrub and tree cover has massively increased at the expense of grassland that historically provided a habitat for numerous non-forest species (Fig. 1).

Apart from succession, there are multiple external sources of environmental stress acting on the nature reserve. These include the neighbouring industrial harbour of Ravenna (Lucialli et al. 2007) as well as a surrounding landscape dominated by intense agriculture (Benini and Pezzi 2011). The heavy use of pesticides in Ravenna's

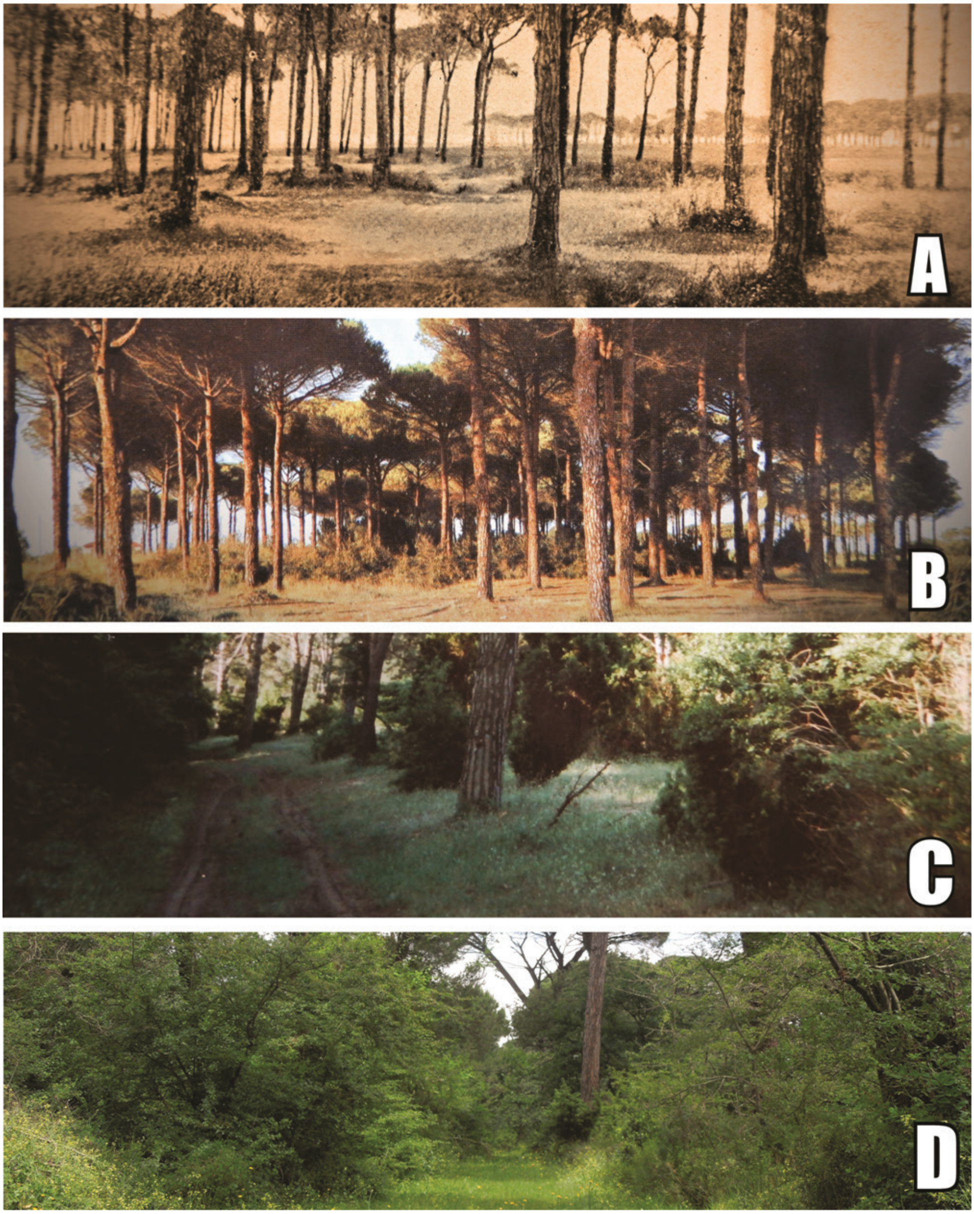

Figure 1. Succession of vegetation in PsV. **A** 1910 **B** 1970 **C** 1996 **D** 2017.

agriculture is also well documented (Paris et al. 2016). Therefore, pesticide drift must be considered to possibly affect organisms inside the reserve. Furthermore, the whole region is subject to subsidence induced soil salinisation, raising salt concentrations in soil water up to 22 g/l in PsV due to continual groundwater pumping and offshore gas production (Antonellini et al. 2008).

Finally, climate change is also evident in the region of Ravenna. Surface solar radiation in northern Italy decreased from 1959 until the mid-1980s, followed by an increase later on (Manara et al. 2016). Mean annual temperatures increased between 1961 and 2010 (Antolini et al. 2015), accompanied by substantial changes in precipitation levels. As a consequence, the overall aridity in the region has increased, particularly near the coastline (Appiotti et al. 2013).

Against this background, it is expected that – apart from mere stochastic effects on local colonisation and extinction promoted through isolation (Haddad et al. 2015) – natural succession, air pollution, soil salinisation, pesticide drift and climate change may have triggered long-term directional ecosystem changes in PsV.

We here combine multi-annual data from our own observations with a rare set of historical records to assess the transformation of the insect fauna in a Mediterranean coastal pine forest over the last 85 years. Regional lepidopterists visited PsV over many decades to conduct light-trapping, thereby collating faunal data for the area. We consider macro-moths as suitable focal organisms for this type of study since they occur in high numbers, are rich in species and have short generation times (usually one year or less), rendering quick responses to environmental change visible. Moreover, moth species span a wide range of trophic affiliations, especially during their larval stages. Many species are quite specialised to particular host plants or habitat structures, which renders them susceptible to anthropogenic habitat changes (Fox 2013). With these data, we test the following hypotheses:

1. Species richness has overall decreased because of manifold anthropogenic influences and stochastic extinctions;
2. Species numbers of forest-bound moths have increased and non-forest moths decreased, due to the succession towards more forest cover within the reserve.

Material and methods

Historical moth data

To analyse multi-decadal changes in moth assemblages, data spanning a period of about 80 years were collated (Table 1). Two historical collections of moth specimens, accompanied by reliable data on sampling localities and sampling dates, were traced in regional natural history museums in Italy, viz. the Museo di Storia Naturale di Venezia (vouchers from 1933–1968) and the Museo Civico delle Cappuccine, Bagnacavallo (vouchers collected between 1966–1976). Further data were made available by a private collector, providing samples from the years 1977–1996 (E. Bertaccini, pers. communication). All moths, reliably labelled to have been collected in PsV, were considered for analysis. Identifications were cross-checked and corrected according to up-to-date taxonomy, whenever required. These historical voucher collections were qualitative in the sense that we have no information as to the reasons why collectors decided to keep or discard observed specimens. For sure, no large voucher series of common species were assembled at these earlier times. We consider it likely that early collectors always kept vouchers of species that appeared to be 'new' to them for the site, whereas they may have ignored common species after their first observations.

Extant moth data

Data on the extant moth assemblages in PsV (expansion from north 44°31'39.15"N, 12°14'19.82"E to south 44°27'48.09"N, 12°13'43.67"E and west 44°29'51.96"N, 12°13'22.79"E to east 44°29'50.50"N, 12°14'15.56"E) were sampled by means of light-trapping in two time periods between 1997 and 2012, but in different manners. From 1997 to 2002, moths were attracted to one single light trap and manually sampled, mainly in early summer and early autumn, at seven locations within different types of vegetation (viz. downy oak forest, hygrophilous forest and, occasionally, reed or remaining open habitats). In the years 2011 and 2012, we more systematically collected moths in spring, early summer, high summer and autumn. This was done in four different habitat types prevalent in PsV (viz. reed, hygrophilous forest, downy oak forest and open habitats) to cover the moth community of the entire reserve as completely as possible. In 2011, automated light-traps were run at 20 sites rather equally distributed within PsV, which allowed sampling multiple habitats simultaneously. In 2012, the light trap employed from 1997 to 2002 was used again at nine locations. Moths were manually collected at this trap, but due to the high demand of manpower, this could be realised only at a smaller number of sites (see Table 1 for further details). All vouchers, sampled since 1997, are stored in the private collection of Mirko Wölfling (Niederwerrn, Germany).

Data management and analysis

Our primary target group were species of the monophyletic clade Macroheterocera sensu Regier et al. (2017), augmented by a handful of larger-sized representatives of Cossidae and Limacodidae that have traditionally been treated as 'macro-moths' by earlier European lepidopterists. Since our extant data were exclusively derived through light-trapping, we removed all strictly diurnally active Macroheterocera from the historical data to improve comparability. We also omitted species from the historical records that are on the wing only during the cold seasons, since we have no recent data for these parts of the year. Finally, we took out from all time periods species that only show up in NE Italy as sporadic or seasonal long-distance migrants, but which are not able to build up persistent populations there.

In a couple of cases, cryptic species diversity has been uncovered amongst moths represented in our data in recent years, while in the historical collections, these were still treated as just one species each. We then adopted the older (more inclusive) taxonomic species delineations for our analyses, since it was not possible to re-examine all historical records by means of anatomical or DNA-sequence based methods.

Table 1. Overview of the moth collections from PsV, available for evaluation. Only those moth species which qualified for a comparative analysis are considered in this tabulation (see Methods section).

Collection	Period	Number of recorded species	Number of trap locations	Type of trap and lamps
Callegari + Martinasco combined	1933–1976	107	Unknown	Unknown
Bertaccini	1977–1996	157	Unknown	Unknown
Wölfling Early extant data	1997–2002	174	7	500 W HWL, manual
Wölfling Recent extant data	2011–2012	187	20 (2011); 9 (2012)	2011: Sylvania 15 W BL + 15 W white BL, automated 2012: 500 W HWL, manual

For analysis, we partitioned our data into four time horizons. The first time horizon covered collections from 1933–1976 (collections Callegari and Martinasco), the second one refers to the period 1977–1996 (collection Bertaccini). The third time horizon was represented by our own samples from 1997–2002 and the fourth group by our own samples collected in 2011 and 2012.

For sample-based species richness estimation, each of the four time horizons had to be subdivided into sampling units. Since we have no information about the details of moth sampling in the old collections, we instead used individual calendar years as proxy for sampling units. This way, the moth data of the first time horizon were allocated to 33 subsamples and those of the second time horizon to 20 subsamples.

For the time horizon from 1997–2002, we instead used sampling nights as units. The same was done for the year 2012. In 2011, when automated light traps were used, we decided to choose sampled habitats per season rather than sampling nights. As the four automated light traps sampled four different habitats in one sampling night, choosing sampling nights as a unit would mean pooling data from different vegetation types. With manual light trapping, as it was performed in all the other years of our own sampling, just one vegetation type per night could be sampled. To adjust automated to manual samples by number of sampling units, choosing sampled habitats rather than sampling nights therefore seemed to be the most logical approach. As a consequence, we came up with 30 subsamples for the 1997–2002 timespan and 41 subsamples for the most recent timespan 2011/12.

For comparisons of moth species richness between the four temporal layers, we then analysed species accumulation by incidence data using the programme iNEXT online (Chao et al. 2016).

We further partitioned observed moth species into inhabitants of wooded habitats, species of open habitats, reed habitats and habitat generalist species, respectively. In three of these subsets, we again checked for temporal changes in species richness by means of species accumulation analysis across the four temporal layers, as described above. Reed species were too few to allow for a meaningful analysis through species accumulation statistics.

Finally, we used the information of species incidence counts to calculate the proportions of these for classes of habitat affiliations across the four time horizons. Using

χ^2-tests, we checked for significant differences in the representation of species per category of habitat use over the four time horizons. With these data, we also created pie charts to visualise the relationships between the different habitat users and how these might have changed proportionally over time.

Information about habitat affiliations of moth species was compiled from Ebert (1994–2003), Hausmann and Viidalepp (2012), Redondo et al. (2009), Rákosy (1996) and from various internet sources (www.lepiforum.de; www.pyrgus.de; www.euroleps.ch). The resulting classification of moth species into the four groups of habitat use can be found in Suppl. material 1: Table S1.

Results

In total, we assembled records of 403 macro-moth species for PsV. From these species, 103 species recorded in older collections had to be deleted from the analyses presented below, as they are either on the wing during the cold seasons only, show exclusively diurnal flight activity or only reach the area as sporadic long-distance migrants, leaving exactly 300 species of macro-moths for the present analyses. With the above adjustments, historical records could be traced for 219 macro-moth species that were observed in PsV during the 20th century. The two sets of historical collections comprised 107 (1933–1976: Callegari + Martinasco) and 157 species, respectively (1977–1996: Bertaccini). Our own samples covered in total 174 species from 1997–2002 and 187 species in the years 2011+2012 (237 species in total since 1997).

Altogether, 63 of the 219 species covered by historical records (28.8%) have never been observed again by us in PsV since 1997. We consider these below as 'lost' species. On the other hand, our data comprise records of 81 species that were not represented in the earlier collections ('gained' species). These gross figures indicate a substantial turnover in moth species composition over time, but they need to be controlled for sampling intensity prior to interpretation.

An incidence-based comparison of older time layers with the more recent datasets clearly shows a substantial increase in total macro-moth species richness after correcting for sampling intensity (Fig. 2). When extrapolated to a standardised number of 40 sample units, an estimated plus of about 67 species has occurred. In particular, 119 macro-moth species (± 13) were estimated for the oldest data, 193 species (± 21) for the time period from 1977–1996, 190 species (± 14) for the years 1997–2002 and 186 species (± 10) for the newest data. This corresponds to an increase by 56.3–59.7% in total moth species richness over the course of the entire time span, whereas almost no change is apparent if only the more recent data from the Bertaccini collection are taken as the basis.

Concerning the three classes of moth species according to their habitat use (Fig. 2), our comparisons at a standardised number of 40 samples revealed the following results. In total, 75 generalist species were observed over the last 80 years. From the oldest to the most recent time horizon, habitat generalists showed a plus of 28 species (1933–1976: 24 ± 6, 1977–1996: 56 ± 12, 1997–2002: 51 ± 4, 2011+2012: 52 ± 7), viz. an increase

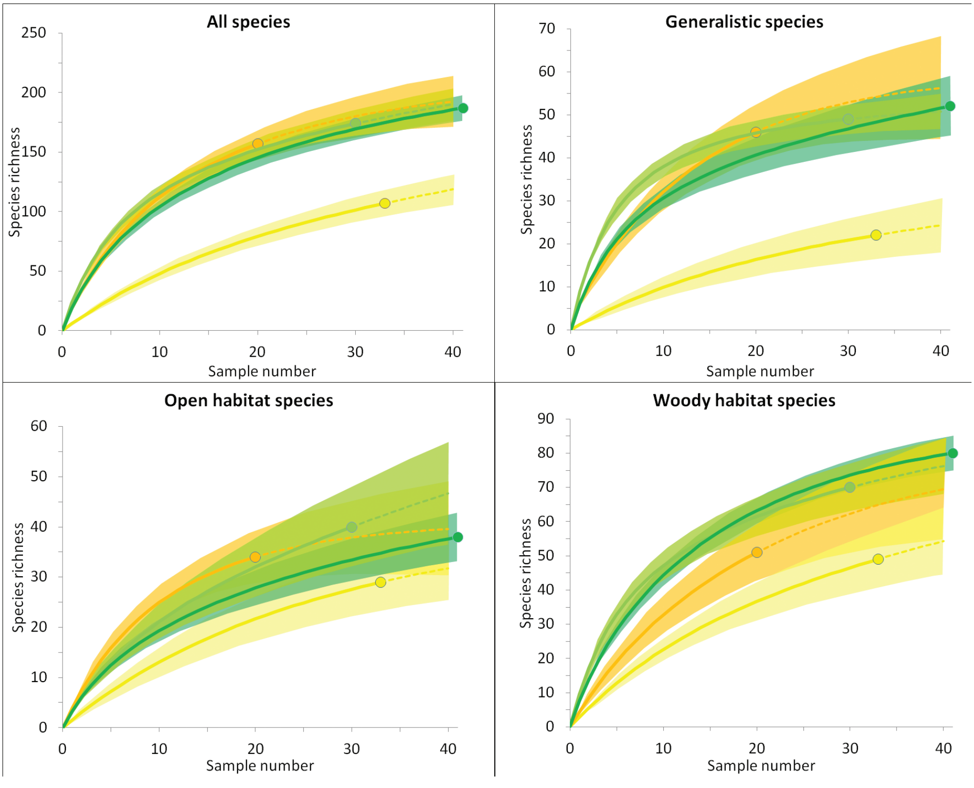

Figure 2. Species richness accumulation of macro-moths in PsV according to their habitat use, across four time horizons, as a function of the number of sampling units calculated in iNEXT. Shaded areas: 95% confidence limits. Yellow = 1933–1976, orange = 1977–1996, light green = 1997–2002, dark green = 2011/2012. Filled circles indicate observed species numbers at the respective number of available sampling units.

by 112.5–116.7%. Altogether, 123 woodland moth species were represented in the records. Woodland moths showed an estimated plus of 26 species over the full timespan (1933–1976: 54 ± 10, 1977–1996: 70 ± 14, 1997–2002: 76 ± 9, 2011+2012: 80 ± 5), which means an increase by 40.7–48.1%.

Overall, 73 open habitat species have thus far ever been recorded from PsV. With an estimated plus of 6 species, which refers to an 18.8–46.9% increase, open habitat users had the lowest increase in species numbers (1933–1976: 32 ± 7, 1977–1996: 40 ± 9, 1997–2002: 47 ± 10, 2011+2012: 38 ± 5). Observed species counts and the respective estimates for a standardised number of 40 sample units of macro-moths at the four time horizons in PsV, including segregation into classes of their habitat use, are listed in Table 2.

The contribution of moth species associated with individual habitat types (Fig. 3) revealed an increase in the proportion of generalist species, from 20.5% in the earliest samples to 32.3–37.4% around the year 2000 and later on. In contrast, the proportion of woodland species slightly decreased from 48.2% to 40.5%. Similarly, moth species

Table 2. Species counts and species richness estimates of macro-moths in PsV, segregated according to temporal layers and habitat use.

Habitat use	Old data	1970s	1997–2002	2011 & 2012
All species observed	107	157	174	187
Estimated species total	119	193	190	186
Generalist species observed	22	46	49	52
Generalist species estimated	24	56	51	52
Woodland species observed	49	51	71	81
Woodland species estimated	54	70	76	80
Open habitat species observed	29	34	40	38
Open habitat species estimated	32	40	47	38
Reed species observed	7	26	14	16
Reed species estimated	–	–	–	–

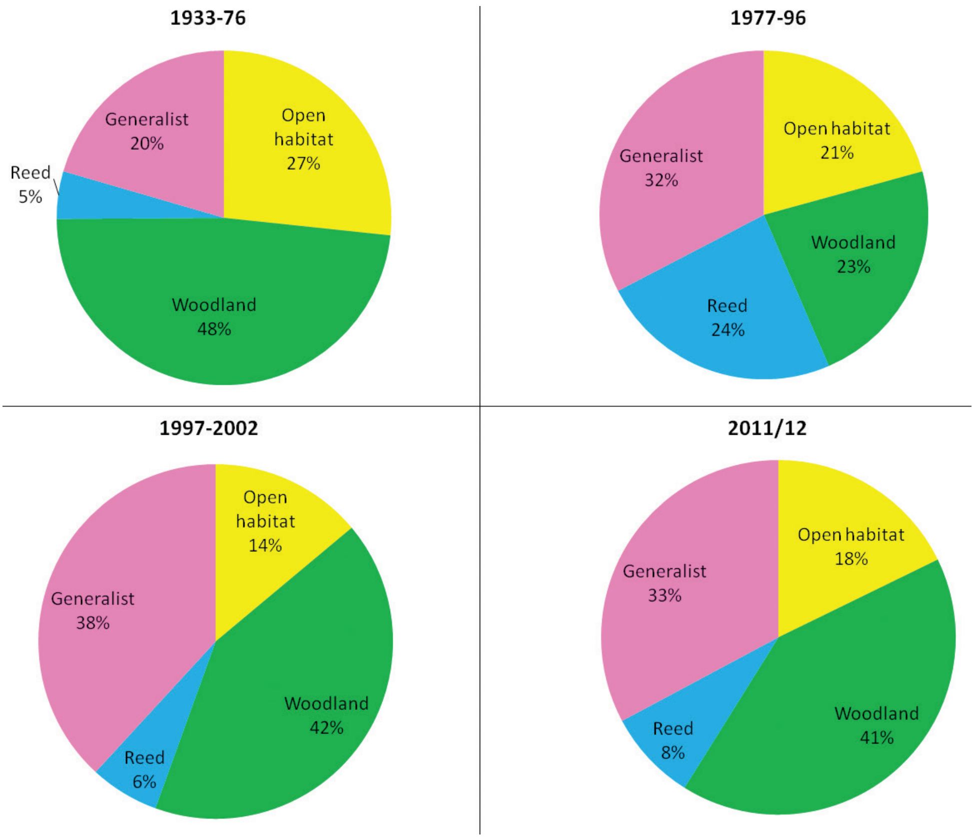

Figure 3. Proportions of macro-moth species in four classes according to habitat use, in the four time horizons.

of open habitats decreased in relative prevalence from 26.7% down to 13.6–17.5% of observed species. Reed species contributed only a minor fraction of 4.6–9.7% of the observed species richness per time horizon, except for the decades spanned by the

Table 3. Numbers of moth species no longer observed after 1995 in the reserve Pineta san Vitale ('lost'), only observed after 1997 ('gained') and present in historical as well as recent surveys ('persistent'), according to their major habitat affiliations.

Habitat use	Lost	Persistent	Gained
Open habitats	18	35	20
Woodland	25	57	41
Generalist	10	46	19
Reed	10	18	1

Bertaccini collection, when almost one quarter of the observed macro-moth species were reed dwellers. However, these differences of species numbers in the different types of habitat affiliations and time periods were just not significant (χ^2_{9df}=16.73, p=0.055).

A comparison of the number of species which disappeared during the last 80 years with those that were newly recorded since the mid-1990s revealed a substantial turnover in all four classes of moths according to their habitat use (Table 3). Amongst woodland species and habitat generalists, gains were almost twice as large as losses. In contrast, moth species of open or reed habitats were disproportionally prone to losses. These differences were statistically significant (χ^2_{6df}=15.78, p=0.015).

Discussion

Our study revealed that (1) contrary to expectation, total species richness of macro-moths did not decline obviously over the past 85 years; yet (2) indeed a substantial species turnover has occurred, favouring generalist and, to some extent, woodland species, while macro-moths of dry open grassland became far less prevalent than before and also reed species suffered from losses. The first observation is surprising, given the numerous stressors that act heavily on the isolated nature reserve PsV from its immediate surroundings. Observations in German nature reserves, embedded in landscapes of intensive agriculture, indicated that insect biomass has undergone severe reductions in the last decades (Hallmann et al. 2017). The same trend should be expected for PsV.

Apart from pressures exerted by the surrounding land-use, a severe extinction debt in isolated nature reserves such as PsV should be expected from demographic and environmental stochasticity alone (Bommarco et al. 2014). Hence, one might have anticipated a strong erosion of species richness over time (Halley et al. 2016). On the contrary, overall species richness of macro-moths appears to have increased over time, either when considered at a standardised sampling intensity or using the raw species counts. This richness pattern was also largely consistent across all groups of moths according to their habitat affiliations, though this apparent increase was strongest when all moths or only generalist species were considered.

We attribute this apparent, unexpected increase in moth species richness to two complementary reasons, viz. secondary succession and sampling intensity. After the definitive abandonment of land-use following the implementation of the current conservation status of PsV, succession has changed the vegetation of the area towards a more complex suite of woodland habitats, at the expense of dry open grassland (Fig. 1). Similar vegetation developments have also been observed elsewhere in northern Italy (Prisco et al. 2016). An increase in species richness should therefore be expected, since the number of niches available in an area usually increases with succession (Hilmers et al. 2018). Indeed, species accumulation analysis suggests that the number of woodland species steadily increased in PsV from the 1930s to the end of the 20^{th} century, but has subsequently remained on the same high level over the past 20 years.

In this context, the species thriving in open and often xeric habitats are also informative. Richness of this group of species has increased the least and even decreased in the most recent collections, although our own quantitative light-trap samples were much larger and thus more comprehensive than earlier records available from PsV. Specifically, our own collections from the years 1997–2002 comprised 1655 moth individuals and those from 2011 and 2012 even 3192 individuals, as opposed to the Bertaccini collection (1459 specimens) and the oldest data (454 specimens). Hence, despite a higher likelihood of detecting open habitat species in these much larger samples, their contribution was low in our data. This well matches the fact that open xeric habitats have shrunk considerably in PsV over the last decades.

Moth species of reed habitats contributed only a minor fraction to the moth fauna of PsV, even though this particular nature reserve is part of a wetland national park of international relevance (Montanari 2010, Ramsar Convention Secretariat 2013). This habitat-specialist group of insects, like open habitat species, appears to have experienced disproportional biodiversity losses in recent decades. Despite the much larger size of our own moth samples, only one single additional reed species could be detected, whereas 10 moth species of reed habitats, present in old collections, have never been observed since 1997. However, our sampling efforts were not specifically targeted to surveying wetland species, so this might also be an effect of preferred sample locations of old collectors, for which we do not have concrete information.

In contrast, generalist species have increased in absolute species numbers, as well as in their relative contribution to the local fauna. This might indicate that generalists are better able to colonise isolated semi-natural areas than some specialised groups (Slade et al. 2013).

Apart from the overall increase in woodland, open and generalist species richness, which might be due to succession and increased sampling effort, the change in the proportions of the groups over time indicates that PsV, as an isolated nature reserve, might today favour the colonisation by generalist species and therefore fail in conserving specialised species (Rossetti et al. 2017). In fact, the proportion of generalists compared to the whole community increased most. These trends are in line with the notion that increasing human pressure on habitats favours generalists over specialists, thereby contributing to biotic homogenisation (Mangels et al. 2017).

Apart from true species turnover, the apparent increase in species richness may partially be due to the way in which historical collections have been assembled. We do not know which kind of light trap was used by early lepidopterists, but in the 1930s, collectors did not have access to lamps powered by electricity with substantial light emission in the near-UV range. They instead often used petrol lanterns with lower efficiency in attracting nocturnal insects. Moreover, lepidopterists with a keen interest in faunistic research tended to be biased towards keeping records preferentially of the 'more interesting' species, i.e. those that are regionally rare or otherwise charismatic. In hindsight, it is impossible to safely tell which species, lacking in old collections, are 'false negatives' (i.e. species that were present, but went unnoticed or no vouchers were kept). However, a number of conspicuous species like *Hemithea aestivaria*, *Opisthograptis luteolata* and *Lacanobia w-latinum*, which collectors of the old data would surely have taken, only appeared in the new data. In contrast, small and 'uncharismatic' species like *Idaea straminata* and *Deltote pygarga* were sampled by old collectors. In order to compensate for differences in sampling effort, we allocated the old data into two time horizons yielding subsets of roughly similar size. We consider comparing two 'historical' periods of low effort sampling with six years of medium and two years of high sampling effort, suitable to facilitate comparisons. We acknowledge that analysis of data from non-standardised sampling by extrapolation is always prone to critique. However, even by comparing the raw data, old collections comprised fewer species than the newer ones (about 80 species). We therefore conclude that the lower richness of the older collections is not only due to sampling effects, but indeed reflects the appearance of new species in the reserve over time.

Even though our results might indicate that preservation of moth biodiversity works quite well within the reserve PsV, many moth species have apparently completely disappeared. We never observed 63 (out of 219) species recorded at least once between 1933 and 1996 during our own light-trapping campaigns. These lost species include conspicuous species (e.g. *Calophasia lunula*, *Plusia festucae*, *Diachrysia chryson* and *Sphinx ligustri*) that are very unlikely to have gone undetected by chance in the period between 1997 and 2012. Lost species also comprise a few species of high conservation concern (e.g. *Calyptra thalictri* and the very rare and localised wetland geometrid *Chariaspilates formosaria*). Even though one can never be entirely sure whether 'lost' species are really locally extinct or whether 'gained' species had not existed earlier in PsV, yet escaped discovery, our analyses show that species turnover in the reserve was non-random.

Overall, these considerations indicate that (a) a substantial extinction debt still remains a risk for the fauna of PsV: more local species extinctions are to be expected, just as the losses that have occurred in earlier decades; and that (b) the process of biotic homogenisation (Newbold et al. 2018) is likely to proceed here as well. Generalist species already play a larger role in faunal composition than was the case with the historical data. Similarly, moth communities across many regions in Europe tend to become ever more homogeneous, with generalist ubiquitous species replacing specialists (Mangels et al. 2017; Franzén and Betzholtz 2012).

Our analyses indicate that, apart from an apparent increase in recorded species numbers, this area of high legal conservation status is indeed threatened by further erosion of its biodiversity, mainly due to the risk of a strong extinction debt, as well as by landscape-level constraints on recolonisation once species have locally gone missing. In the long run, even though the vegetation in PsV may continue to converge to a more 'natural' structure, the insect fauna in this highly isolated area might be prone to further homogenisation. Therefore, active conservation management is most desirable, for example with focus on wetland or open habitat fractions remaining as niches for specialist organisms, in order to safeguard the function of PsV in the context of preserving biodiversity.

Conclusions

To understand changes in insect diversity, there is a strong need for long term analyses. Yet, long-term data from standardised monitoring are largely lacking. Historical collections not only provide an opportunity to gain an insight into community change, but also pose challenges, such as selective or variable sampling effort and gaps in time series. We tried here to extract valuable information on the long term development of biota in an isolated nature reserve by analysing such old collections.

In contrast to our expectations, species richness increased although isolation effects, increased salinity and pesticide use in nearby agricultural areas might have affected the reserve. Therefore, succession might even override these negative effects and conceal possible influences on moth species richness. Open habitat specialists have been lost to a disproportionate extent, whereas generalist and woodland species have increased. These trends reflect both the succession inside the forest reserve, as well as constraints on species dispersal in fragmented landscapes. From a conservation perspective, enhancing connectivity between such reserves is of the highest importance for protecting specialised and rare species.

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Supplementary material 1

Table S1. Species of Pineta san Vitale splitted into different types of habitat use

Authors: Mirko Wölfling, Britta Uhl, Konrad Fiedler

Data type: species data

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The utility of Sentinel-2 Vegetation Indices (VIs) and Sentinel-1 Synthetic Aperture Radar (SAR) for invasive alien species detection and mapping

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Abstract

The threat of invasive alien plant species is progressively becoming a serious global concern. Alien plant invasions adversely affect both ecological services and socio-economic systems. Hence, accurate detection and mapping of invasive alien species is valuable in mitigating adverse ecological and socio-economic effects. Recent advances in active and passive remote sensing technology have created new and cost-effective opportunities for the application of remote sensing to invasive species mapping. In this study, new generation Sentinel-2 (S2) optical imagery was compared to S2 derived Vegetation Indices (VIs) and S2 VIs fused with Sentinel-1 (S1) Synthetic Aperture Radar (SAR) imagery for detecting and mapping the American Bramble (*Rubus cuneifolius*). Fusion of S2 VIs and S1 SAR imagery was conducted at pixel level and multi-class Support Vector Machine (SVM) image classification was used to determine the dominant land use land cover classes. Results indicated that S2 derived VIs were the most accurate (80%) in detecting and mapping Bramble, while fused S2 VIs and S1 SAR were the least accurate (54%). Findings from this study suggest that the application of S2 VIs is more suitable for Bramble detection and mapping than the fused S2 VIs and S1 SAR. The superior performance of S2 VIs highlights the value of the new generation S2 VIs for invasive alien species detection and mapping. Furthermore, this study recommends the use of freely available new generation satellite imagery for cost effective and timeous mapping of Bramble from surrounding native vegetation and other land use land cover types.

Keywords
Alien species invasions, Sentinel-1, Synthetic Aperture Radar (SAR), Sentinel-2, Vegetation Indices (VIs), American Bramble, Fusion, Support Vector Machine (SVM)

Introduction

Global biodiversity is increasingly becoming susceptible to pressure from invasive species (Butchart et al. 2010). Specifically, the rapid spread of invasive alien plants in several regions of the world has adversely impacted ecosystem health, native species diversity and local and national economies (Pysek et al. 2012; Schirmel et al. 2016; Convention on Biological Diversity 2009). Brooks et al (2006) highlight the imperative need for the protection of native biodiversity, a need further emphasised by the United Nations (UN) that declared the period between 2010 and 2020 as the decade of biodiversity (UNEP 2010). Moreover, increased costs associated with invasive alien species eradication and management programmes puts further pressure on biodiversity (Marbuah et al. 2014). The severity of the problem has increased the impetus on development of efficient and cost-effective approaches for the control and management of invasive alien plant species.

In South Africa, approximately two million hectares of land have been invaded by invasive alien plant species (van Wilgen et al. 2012). The south western, southern and eastern coastal and interior regions have been identified as highly vulnerable to invasion (Kotzé et al. 2010; van Wilgen et al. 2012; Clusella-Trullas and Garcia 2017). In KwaZulu-Natal (KZN) province, for instance, Erasmus (1984) notes that the cool and moist conditions favour a range of invasive alien plant species. The American Bramble (*Rubus cuneifolius*) has particularly thrived in the province's western mountain ranges (Henderson 2011). Originating from North America, Bramble belongs to the *Rosaceae* family and has adverse direct and indirect impacts on biodiversity that include changes in nutrient cycling, increase in soil erosion, reduction in rangeland carrying capacity and viability, as well as effects on natural plant succession, fire patterns and behaviour and hydrological processes (Henderson 2001).

To develop optimal mitigation of spread and eradication approaches, determination of spatial cover and extent of Bramble infestation is paramount. Traditionally, surveys have been adopted for mapping and monitoring of invasive alien plant species (Tan et al. 2012; Shah and Reshi 2014). However, reliance on field-based surveys is often restrictive, as they are commonly time consuming, labour and resource intensive and unsuitable in inaccessible sites. Hence, the adoption of remotely sensed imagery for invasive alien species detection and mapping has recently gained popularity. Huang and Asner (2009) attribute this increase to improved sensor technology, facilitating detailed and large scale landscape mapping and monitoring. In the recent past, the majority of invasive alien plant species detection and mapping applications have relied on remotely sensed image spatial and spectral characteristics (Mirik et al. 2013; Müllerová et al. 2013). Other studies have proposed object-based textural and contextual characteristics (Zhou et al. 2008) and landscape thermal characteristics (Eisavi et al.

2015). However, the advent of new sensors with radar scanning capabilities provides new opportunities for invasive plant species detection and mapping (Bradley 2014). For instance, radar's ability to determine surface structure and roughness, dielectric constant (moisture content) and slope angle and orientation offer great opportunities for invasive species mapping. The European Space Agency's (ESA) sentinel constellation is a recent satellite that consists of the Sentinel-1 (S1) and Sentinel-2 (S2) earth observation instruments. Both sensors disseminate freely available multispectral optical (S2) and multi-polarised SAR (S1) data. The unique S1 and S2 sensor characteristics, such as large swath widths, medium to fine scale spatial resolutions, short re-visit times and additional bands (Frampton et al. 2013; Sentinel-1 User Handbook 2012) provide numerous opportunities to evaluate the potential of the sensors to improve the reliability of remote sensing approaches for invasive alien plant species mapping.

Conventional remote sensing of invasive alien species utilises spectral wavelengths of absorbed and reflected light by distinguishing certain pigments in leaves and inflorescence (Huang and Asner 2009; Mirik et al. 2013; Weisberg et al. 2017; Müllerová et al. 2013; Bradley 2014). Hence, the potential of S2 to detect and map invasive alien species exists (Rajah et al. 2018). Specifically, the senor's improved spectral resolution can be used to derive numerous band ratios and indices, useful for vegetation mapping. For example, spectral vegetation indices (VIs), derived from remotely sensed data, have become valuable in mapping and monitoring vegetation species (Jamali et al. 2014; Zhang et al. 2015; Orhan et al. 2014). VIs have several advantages over stand-alone spectral bands that include reduced effect of atmospheric conditions, canopy geometry and shading, decreased effect of soil background on canopy reflectance and enhanced variability of spectral reflectance of target vegetation (Liu et al. 2004; Viña et al. 2011). On the other hand, the unique characteristics of S1 SAR imagery could provide additional variables that could improve invasive alien species detection and mapping. SAR data can operate at wavelengths irrespective of cloud conditions or lack of illumination and is capable of acquiring data during day and night (Sentinel-1 User Handbook 2012). SAR offers detailed information on the often difficult to detect characteristics of vegetation such as shape, moisture and roughness (Chen et al. 2010). However, despite this potential, previous adoption of SAR imagery for invasive alien plant species mapping has been limited by high acquisition cost, limited area coverage and complex data pre-processing (McNairn et al. 2009). Hence, the provision of freely available SAR imagery from the S1 satellite provides new prospects for advancing the mapping and detection of invasive alien plant species.

Asner et al. (2008) and Zhang (2010) note that the fusion of imagery from various sensors, while applying appropriate methodologies, may be valuable for invasive alien species detection and mapping. Furthermore, conventional optical imagery and SAR are commonly believed to be complimentary (Zhu et al. 2012). Considering the above-mentioned advantages, as well as S2s improved spatial, spectral and temporal characteristics valuable for generating VIs, the fusion of these datasets provides a unique opportunity to investigate the value of new generation sensors such as S1 and S2 in mapping alien species. Accordingly, this study sought to determine the performance of conventional stand-alone S2 optical imagery, stand-alone S2 derived VIs and fused S2 VIs with S1 Synthetic Aperture Radar (SAR) imagery in detecting and mapping the American Bramble.

Methodology

Study site

This study was conducted at the uKhahlamba Drakensberg Park (UDP), a UNESCO proclaimed world heritage and nature conservation area. The area is situated along the western edge of the KwaZulu-Natal province of South Africa (Figure 1). The area experiences wet and humid conditions during summer (November to March) (Nel 2009), with rainfall ranging from 990–1130 mm (Dollar and Goudy 1999). Winters (May to August) are dry and cold, with common occurrence of snow and frost (Mansour et al. 2012). Mean annual temperatures average 16° C and annual rainfall averages 1000 mm and 1800 mm at lower and higher elevations, respectively (Tyson et al. 1976). The landscape is predominantly natural grassland with wiregrass – *Aristida purpurea*, weeping lovegrass – *Eragrostis curvula* and the common thatch grass – *Hyparrhenia hirta* as dominant species. According to Everson and Everson (2016), the UDP is one of the most valuable remnant grassland in the country. The area is also characterised by patches of natural shrubs (*Erica* spp.) and isolated dense groups of bushes and trees. In the recent past, Bramble has emerged within the UDP and has invaded significant portions of the landscape (Bromilow 2010).

Field data collection

Field data collection was conducted during spring and summer of 2016. A purposive sampling technique was utilised to record ground truth points of four major land cover classes (Bare rock, Bramble, Forest and Grassland). These seasons were chosen for field data collection as Bramble patches are most phenologically discernible from native vegetation. Ground control points were recorded as close to the centroid of Bramble patches as possible. Collected Bramble patches ranged from 15 m × 15 m to 50 m × 50 m. Ground truth point data collected from Bramble patches were spatially independent from each other to compensate for the spatial resolution of the satellite imagery utilised. This ensured that each Bramble patch fell within a single image pixel and could be associated with the unique spectral reflectance of a specific pixel. Due to the area's steep and mountainous terrain, hence restricted accessibility, only Bramble patches that could be accessed by foot were considered for this study. In addition, aerial photographs at a 0.5 m spatial resolution captured in 2016 were used to supplement and verify selected land cover ground truth points. In total, 15, 40, 45 and 60 ground truth points were used for Bare rock, Forest, Grassland and Bramble, respectively.

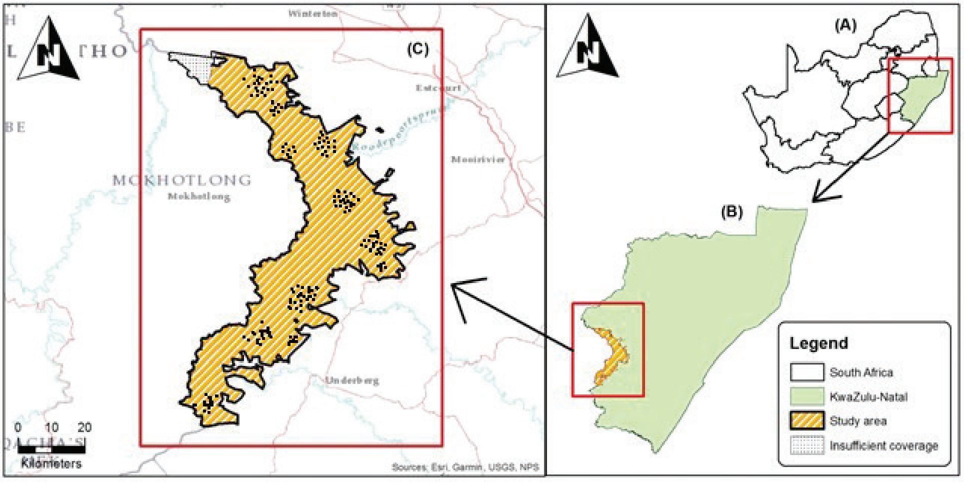

Figure 1. The uKhahlamba Drakensberg Park (UDP) (C) located within the KwaZulu-Natal Province (B) of South Africa (A) (Points within the map represent GPS cordinates of ground truth points).

Image acquisition

Optical Imagery

The Sen2Cor plugin ESA SNAP toolbox 3.0 (European Space Agency 2018) was used to convert summer Sentinel-2 level-1C raw products to surface reflectance values in the Sen2Cor plugin. Images were corrected for topographic effects to remove shadows associated with mountainous areas using the System for Automated Geoscientific Analyses SAGA (2.1.2) terrain analysis lighting tool within the Quantum GIS (QGIS) environment on a band by band basis (Conrad et al. 2015). The correction of topographic effects is a tool within the SAGA software that best adjusts optical imagery for topographic effects of shadow (Conrad et al. 2015).

Sentinel-1 Synthetic Aperture Radar (SAR) Imagery

Summer Synthetic Aperture Radar (SAR) data were downloaded from the Sentinel-1 data hub. Sentinel-1 level-1 Ground Range Detected (GRD) products were multi-looked and projected to ground range using an earth ellipsoid model. SAR Vertical-Horizontal (VH) polarised imagery was acquired using the Interferometric Wide Swath (IW) mode, with a spatial resolution of 20 metres and a 250 km^2 swath width. Pre-processing of SAR imagery was conducted using the ESA SNAP toolbox following the methodology outlined in Bevington (2016). The Bevington (2016) SAR image processing chain consists of 5 steps: (1) Application of orbit file to SAR image; (2) Radiometric calibration; (3) Terrain correction; (4) Application of speckle filter; (5)

Convert SAR DN to Gamma backscatter values. Polarisation of SAR imagery recorded in Vertical Horizontal (VH) acquisition mode was fused with S2 derived VIs. SAR backscatter measurements are believed to be a function of polarisation and target object characteristics, such as geometry, roughness and dielectric properties (Vyjayanthia and Nizalapur 2010).

Sentinel-2 derived Vegetation Indices (VIs)

Sixty-five Vegetation Indices (VIs), selected from the online Index Database (IDB) (www.indexdatabase.de), were calculated from summer Sentinel-2 surface reflectance optical imagery. The IDB is a tool developed to provide a simple overview of satellite specific vegetation indices that are usable from a specific sensor for a specific application (Henrich et al. 2009). All VIs were calculated within a python 2.7.13 environment using listed formulae from the IDB and spectral reflectance Sentinel-2 bands (Table 1). The 10 most influential VIs were selected for stand-alone classification results and subsequent image fusion with SAR imagery in order to produce a fused VIs and SAR classification result. The top 10 VI selections were determined using the Variable Importance in the Projection (VIP) method. Variable Importance in the Projection aims to improve classification accuracy by recognising a subset of all initial variables (VIs) that, if combined, could increase classification accuracies with parsimonious representation (Farrés et al. 2015; Xu et al. 2018). As aforementioned, the study area is pre-dominantly natural grassland, regarded as a valuable economic and environmental resource. Hence, in addition to Bramble, it was necessary to reliably determine the spatial extent of grassland. In this regard, the VIP was used to determine the importance of each VI in increasing the two land use land cover's user's and producer's accuracies.

Table 1. Selected S2 derived VIP vegetation indices subsequently utilized for SAR fusion

VIP Vegetation Indices (VIs)	VI formula (S2 optical bands)	
Datt2 (Simple Ratio 850/710)	Near Infrared (NIR)/Red Edge 1	Datt 1999
PSSRc2 (Simple Ratio 800/470 Pigment specific simple ratio C2)	Near Infrared (NIR)/Blue	Blackburn 1998
RDVI (Renormalized Difference Vegetation Index)	Near Infrared - Red/(Near Infrared + Red)$^{0.5}$	Roujean and Breon 1995
SR520/670 (Simple Ratio 520/670)	Blue/Red	Henrich et al. 2009
SR672/550 (Simple Ratio 672/550)	Red/Green	Henrich et al. 2009
SR800/550 (Simple Ratio 800/550)	Near Infrared/Green	Henrich et al. 2009
SR833/1649 (Simple Ratio 833/1649 MSIhyper)	Near Infrared /Shortwave Infrared1	Henrich et al. 2009
SR860/550 (Simple Ratio 860/550)	Narrow-Near Infrared/Green	Henrich et al. 2009
SRMIR/Red (Simple Ratio MIR/Red Eisenhydroxid-Index)	Shortwave Infrared2/Red Edge 1	Henrich et al. 2009
TM5/TM7 (Simple Ratio 1650/2218)	Shortwave Infrared1/ Shortwave Infrared2	Henrich et al. 2009

Image fusion

Pixel level image fusion, based on ground truth points, was used to merge the ten most influential VIP VIs and Sentinel-1 SAR imagery (a description of image fusion levels can be found in Hong et al. (2014). All VIs were derived from S2 optical bands, at a spatial resolution of 20 m. Extraction of feature pixels (ground truth points) were done separately for optical imagery (spectral reflectance measurements) and SAR imagery (backscatter measurements). Corresponding backscatter measurements were then assigned to the corresponding extracted spectral reflectance of ground truth points. Optical and SAR imagery were then fused using the composite band tool in ArcMap 10.4. This was achieved by stacking optical and SAR imagery on a band by band basis, creating a composite (fused) image containing both spectral reflectance and backscatter measurements at respective ground truth points. The fused image was then used for image analysis.

Image classification

Image classification was conducted post pixel level image fusion as outlined in Pandit and Bhiwani (2015). The Support Vector Machine (SVM) algorithm was run using the scikit-learn package in a Python environment. The SVM algorithm is a supervised statistical learning technique initially developed to handle binary classification (Vapnik 1979). SVM aims to identify a hyper-plane that is able to distinguish the input dataset into a predefined discrete number of classes consistent with training data (Mountrakis and Ogole 2011). Several evaluations of SVM have shown that the algorithm is capable of delineating several classes with a small number of support vectors as training data, without ultimately compromising classification accuracies (Foody and Mathur 2004; Mantero et al. 2005; Bruzzone et al. 2006; Shao and Lunetta 2012; Zheng et al. 2015). Spectra were extracted using ground truth points of the aforementioned major land cover classes. The fused VIP vegetation indices and SAR image measurements were used to define the SVM feature space and a radial basis kernel function used to determine optimal hyperplanes that differentiate the different land cover classes. Waske et al. (2010), for instance, established that the approach is superior to the polynomial function. Furthermore, this approach is known to be fast and computationally efficient, with a two parameter tuning requirement; cost 'sigma (C)' for error adjustment of misclassified instants of training data and kernel width 'gamma (γ) (Waske et al. 2010). As recommended by Hsu and Lin (2002), the one-against-one approach was used to implement a multiclass-based SVM model.

Spatial cover map production and validation

Support Vector Machine classification maps were generated for S2 optical imagery, Vegetation Indices and for the fused VIS and SAR imagery within a Python environment. Training data (70%) of all four considered land cover classes were used as the input for Bramble spatial cover maps. The respective test dataset (30%) was then used to assess classification accuracies across all imagery. A confusion matrix was generated from the SVM process and user and producer accuracies used to quantify the reliability of the resultant Bramble spatial cover maps. In a confusion matrix, the overall accuracy is determined by dividing correctly classified pixels by the total number of pixels checked (Congalton and Green 1999). Two other measures, producer's and user's accuracy, can also be generated from the matrix. Producer's accuracy is determined by dividing the total number of correct pixels in one class divided by the total number of pixels as derived from reference data (Congalton and Green 1999). It is a measure of how well an area has been classified and is expressed as:

$$\textit{Producer's accuracy (\%)} = \textit{100\%} - \textit{error of omission (\%)} \quad (1)$$

User's accuracy on the other hand is a measure of map reliability and provides information on how well a map represents ground features. It is expressed as:

$$\textit{User's accuracy (\%)} = \textit{100\%} - \textit{error of commission (\%)} \quad (2)$$

Results

Sentinel-2 optical bands

The overall classification accuracy using S2 optical bands was 78% (Table 2). Bramble produced the lowest users' accuracy (46%) across all considered classes, while Grassland produced the lowest producers' accuracy (69%) (Table 3). Results produced using only S2 optical bands were used as a benchmark for classification using VIs and VIs fused with SAR imagery.

A large overestimation of Bramble discrimination and spatial cover using S2 optical bands was evident (Figures 2b and 3a). An underestimation in Grassland discrimination and spatial cover was observed, as the SVM algorithm could not effectively distinguish between Bramble and Grassland (Table 3, Figure 2b). An underestimation in the spatial cover of the Bare rock class was also evident, as there was consistent misclassification of Bare rock from Grassland and Bramble (Figures 2b and 3a).

Vegetation Indices (VIs)

Discrimination and mapping of Bramble using vegetation indices produced the highest overall accuracy (82%) when compared to the benchmark of using only S2 optical image bands (Table 3). A users' accuracy of 72% for Bramble surpassed those achieved by S2 optical imagery as well as fused vegetation indices and SAR imagery (Table 1).

Table 2. Support Vector Machine (SVM) confusion matrix using Vegetation Indices for Bramble mapping and discrimination. Where BR = Bare rock; BBL = Bramble; FR = Forest; and GR = Grassland, UA = Users accuracy; PA = Producers accuracy and OA = Overall accuracy.

S2 (Optical bands)	BR	BBL	FR	GR	UA (%)
BR	33	2	0	11	70
BBL	0	24	0	30	46
FR	1	1	51	3	92
GR	2	3	7	94	89
PA (%)	92	81	87	69	
OA (%)	**78**				

Table 3. Support Vector Machine (SVM) confusion matrix using Sentinel-2 optical bands for Bramble mapping and discrimination. Where BR = Bae rock; BBL = Bramble; FR = Forest; and GR = Grassland, UA = Users accuracy; PA = Producers accuracy and OA = Overall accuracy.

Vegetation Indices (VIs)	BR	BBL	FR	GR	UA (%)
BR	51	11	0	0	83
BBL	0	53	19	4	72
FR	1	0	54	0	97
GR	13	7	0	57	74
PA (%)	83	78	76	91	
OA (%)	**84**				

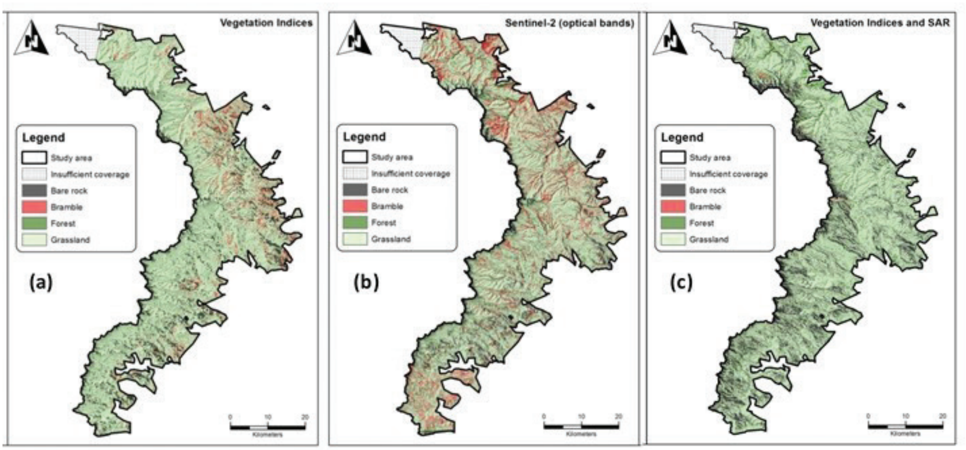

Figure 2. Support Vector Machine (SVM) classification maps produced utilising (**a**) Vegetation Indices; (**b**) S2 optical bands and (**c**) Fused VIs and SAR.

The classification map resulting from fused vegetation indices and SAR imagery showed the most accurate discrimination and spatial cover of all considered land cover classes. The Grassland and Bare rock classes were reliably discriminated (Figures 2a

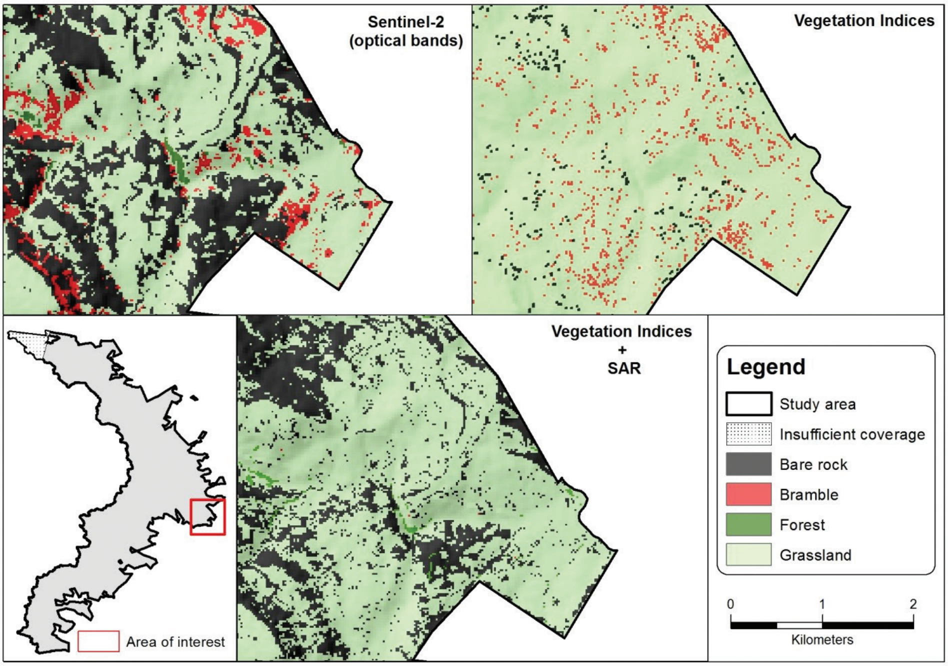

Figure 3. Overestimation and underestimation of land-cover classes within an area of interest where (**a**) = S2 optical bands; (**b**) = Vegetation indices (VIs) and (**c**) = VIs and SAR imagery.

and 3b). In addition, the spatial discrimination and cover of Bramble was reliably discriminated as compared to the S2 optical band benchmark and the fused VIs and SAR imagery (Figures 2a and 3b).

Vegetation Indices (VIs) and S1 SAR imagery

The ten most influential S2 VIs were selected for pixel level image fusion with S1 SAR imagery. Using VIP, the influence of VIs was identified by the importance on increasing Grassland and Bramble's User's and Producer's accuracy, hence, the ten bands that generated the ten highest classification accuracies were selected. Five of the selected VIs incorporated the Near Infrared (NIR) optical band, while three selected VIs were derived using Shortwave Infrared 1 (SWIR1) and Shortwave Infrared 2 (SWIR2) optical bands (Table 1). The SR520/670 and SR672/550 VIs were the only two VIP VIs derived using bands within the visible portion of the electromagnetic spectrum (Table 3).

The fusion of VIs and S1 SAR imagery produced the lowest overall accuracy (55%) when compared to the benchmark of S2 optical band results (Table 4). Bramble users' and producers' accuracies were 29% and 20%, respectively (Table 4), the lowest in all

Table 4. Support Vector Machine (SVM) confusion matrix using fused Vegetation Indices and SAR imagery for Bramble mapping and discrimination. Where BR = Bae rock; BBL = Bramble; FR = Forest; and GR = Grassland, UA = Users accuracy; PA = Producers accuracy and OA = Overall accuracy.

VIs and SAR	BR	BBL	FR	GR	UA (%)
BR	43	2	0	11	79
BBL	0	15	0	38	29
FR	1	0	45	17	73
GR	0	53	0	39	42
PA (%)	97	20	100	37	
OA (%)	**55**				

classes. The Forest (73% and 100%) and Bare rock (79% and 97%) classes were the highest users' and producers' accuracies, respectively.

The SVM classification map, produced using fused vegetation indices and SAR, resulted in an underestimation of the Bramble class, while an overestimation of the Grassland class was observed (Figure 2c). Although the Forest class received high users' and producers' accuracies, the overall distribution and discrimination were overestimated when compared to the benchmark (Figures 2c and 3c).

Discussion

This study sought to determine the potential of derived Vegetation Indices (VIs) and fused VIs and Synthetic Aperture Radar (SAR) imagery to improve invasive alien species detection and mapping. The overall classification accuracy of optical imagery was used as the benchmark for comparison of the results achieved using S2 VIs and fused VIs and SAR. Opposing the expected outcome, fused VIs and SAR imagery produced the lowest classification accuracy (55%) compared to conventional S2 optical imagery (78%). Moreover, S2 derived VIs produced the highest classification accuracy (84%) when compared to conventional S2 optical imagery and fused VIs and SAR.

Poor performance of fused VIs and SAR imagery was unanticipated and opposes research done by Sano et al. (2005), who noted that the combination of VIs and SAR for discrimination within a savannah environment was complementary and improved overall discriminant analysis. Sano et al. (2005) also noted that VIs and SAR were able to easily separate Grassland from woodlands. However, Sano et al. (2005) also reported increased confusion between Grassland and shrub species when utilising fused VIs and SAR. This provides some indication that previous studies have also encountered unanticipated results when combining VIs and SAR for discrimination purposes. Poor overall classification accuracies of fused VIs and SAR imagery can further be attributed to vegetation structure and roughness, as this plays a major role in measured SAR backscatter values. Similar difficulties were documented by Millard and Richardson (2018) who note that, even though it is well established that vegetation roughness influences

SAR backscatter, characterising these variables spatially and temporally within natural environments remains a challenge. Although results from fused VIs and SAR were unexpected, similar poor performance using the same combination of variables is not unprecedented. For example, Torma et al. (2004) also experienced poor performance when fusing VIs and SAR.

Patel et al. (2006) and Srivastava et al. (2009) note that the magnitude of SAR backscatter is dependent on SAR band frequency, for instance, SAR backscatter signatures at high frequency (e.g. X-band SAR) are known to be sensitive to subtle variations in vegetation phenology attributed to deep canopy penetration. Sentinel-1 C-band SAR is considered low frequency (decreased canopy penetration) SAR imagery and could have experienced difficulty in discerning between Bramble characteristics and surrounding native vegetation when using fused VIs and S1 SAR imagery (Khosravi et al. 2017; Duguay et al. 2015; Naidoo et al. 2015; Hajj et al. 2014; van Beijma et al. 2014; Turkar et al. 2012). The influence of sensor incident angle on SAR backscatter is known to be interpreted using the same mechanism, particularly for lower frequencies of SAR. Inoue et al. (2002) notes that correlations to plant physiological characteristics, such as Leaf Area Index (LAI), canopy height and stem density, decrease with an increasing incident angle. This is mainly attributed to the penetration of SAR microwaves responsible for backscatter measurements, as smaller incident angles are able to penetrate deeper into canopy cover, hence extract more physiological information (McNairn et al. 2009). The relatively large incident angle of S1 (46°) (Sentinel-1 User Handbook 2012) could have hindered its ability to distinguish vegetation physiological information, which could serve to justify decreased classification accuracies achieved using fused VIs and SAR imagery (de Almeida Furtado et al. 2016; Naidoo et al. 2015; Frampton et al. 2013; Vyjayanthia and Nizalapur 2010). The influence of soil moisture and roughness on leaf and stalk SAR backscatter measurements is considered a weakness of SAR imagery across specific classification applications (Moran et al. 2002). SAR imagery could have served to increase confusion between Bramble and surrounding native vegetation when fused with S2 VIs.

The use of S2 VIs outperformed the benchmark accuracy achieved by conventional S2 optical imagery. Similar results were achieved by Kandwal et al. (2009), where selected VIs performed well in discriminating Lantana camara (*Verbenaceae*), an invasive alien plant with similar growth pattern and phenology to Bramble. The majority of Vis, selected as VIP indices, were dominated by VIs incorporating the Near Infrared (NIR), Shortwave Infrared (SWIR) and red edge S2 bands. A study by Zhao et al. (2007) produced similar results, where VIs, derived from SWIR, red-edge and NIR bands, were reported to be closely correlated to canopy LAI and canopy chlorophyll density. Chuai et al. (2013), for instance, used NDVI to determine seasonal vegetation correlations with lag-time climatic effects in Inner Mongolia between 1998 and 2007. They established varied seasonal changes and concluded that NDVI provides a reliable measure of vegetation changes attributed to climatic variability. According to Domaç et al. (2004), VIs can extract valuable information by generating a new variable set without inter-band correlation and reduced data dimensionality. Whereas the NDVI has commonly been preferred in vegetation mapping, El-Mezouar et al. (2010) suggests that Soil Adjusted Vegetation Index (SAVI) is more suitable for mapping patchy vegetation characterised by lower percentage cover. A recent study by Tarantino et al. (2019) in Apulia region, southern Italy, concluded that indices like MSAVI on World-View2 imagery are effective in discriminating the invasive *Aillanthus aitissina* species. This superior performance is attributed to the indices' maximal reduction of background soil effect on vegetation reflectance (Qi et al. 1994). Other studies like Groβe-Stoltenbeg et al. (2018) used 15 vegetation indices to determine an *Acacia lonifolia* cove in a dune ecosystem and concluded that the fusion of vegetation indices with LIDAR could effectively determine the effects of species invasion on the dune landscape.

Eight of the ten VIP VIs selected for Bramble discrimination and mapping were derived from at least one of these three spectral bands. The strong relationship between NIR, SWIR and red edge bands to variable vegetation parameters could have resulted in the increased accuracy of Bramble discrimination and mapping. Moreover, reflectance within the visible region of the spectrum is largely determined by vegetation pigments and is commonly used to quantify vegetation physiological properties (Li et al. 2013; Zhao et al. 2007). The collective capability of combined VIs to discriminate various vegetation parameters could further explain the increased overall classification accuracy achieved using stand-alone vegetation indices.

Several studies (e.g. Royimani et al (in press) – *Perthenium*, Matongera et al. (2017) - *Pteridium aquilinum* (L.) *Kuhn*, Robinson et al. (2016) - *Mesquite* (*prosopts* spp.), Peters et al. (1992) - *Gutierrezia sarothrae* and Oumar (2016) - *Lantana camara* have discriminated invasive species from native vegetation using spectral variability. Matongera et al. (2017) and Zhao et al. (2009) attribute this to invasive species' dissimilar biophysical (e.g. texture, canopy, leaf structure and orientation) and biochemical (e.g. chlorophyll and water content) characteristics from the surrounding vegetation species. According to Blossey and Notzold (1995), invasive species are commonly characterised by superior physical development due to disproportionate availability or exploitation of resources. Such differences, particularly volume and height, facilitate their discrimination. Goodwin et al. (1999), for instance, noted that differences in stem heights and flowering periods could be used to discriminate invasive species from native vegetation, while Peerbhay et al. (2016) found that dense infestation, particularly in new habitats, facilitate discrimination. Commonly, the phenology of invasive species differs from native plants. Holland and Aplin (2013) and Page (2010) note that the discreet reflectance of invasive species at different seasons offers great potential for their discrimination. This is consistent with Santos and Ustin (2018) who noted that fennel (*Foeniculum vulgare*) could be effectively discriminated by considering seasonal foliar variability from surrounding grasses. In this study, Bramble's broad leafs could have facilitated its discrimination, a finding in agreement with Hong et al. (2014) who successfully discriminated alfalfa in the Prairie Provinces of Canada.

According to Motohka et al. (2010), the potential of green-red VIs for phenological vegetation discrimination exists. VIs derived from ratios of red and green optical bands are known to be sensitive to variations in canopy colour, where changes in visible characteristics of vegetation canopy are often timeously detected (Motohka et al. 2010). The SR672/550 VI, an index derived solely from S2 red and green optical bands, suggests an agreement with Motohka et al. (2010). The SR672/550 VI could have assisted in the discrimination of Bramble as it produces noticeable white inflorescence during summer, a significant phenological trait that could have been exploited. This potential is further enhanced by S2's higher temporal resolution (five days, with possible higher resolution due to overlap in swaths of adjacent orbits) that facilitates single scene's image turnover. Although the combined potential of VIs and SAR imagery produced the lowest overall classification accuracy, the potential of the latest and advanced spectrally derived VIs was evident when compared to the benchmark set by conventional S2 optical imagery. While the fusion of S2 VIs and SAR showed limited utility with regard to accurately mapping Bramble, the complementarity of these datasets has previously been documented. Our findings are consistent with existing literature. For instance, in lower Magdalena region, Colombia, Clerici (2017) established that Sentinel 1 and 2 fused dataset, classified using SVM, generated the highest classification accuracy of the existing land use land covers. Hence, the study recommended the use of radar sensor due to its all-weather capability and the spectral wealth of the optical sensor. Niculescu et al. (2018) mapped major vegetation types by fusing S1, S2 and SPOT – 6 in Pays de Brest, France by stacking time series data using Random Forest supervised classification. The study achieved a 93% classification accuracy when the major vegetation indices (Normalised Difference Vegetation Index - NDVI, Normalised Difference Wetness Index NDWI, Inverted Red-Edge Chlorophyll Index - IRECI and Sentinel-2 Red-Edge Position - S2REP) were used in the classification process. The study recommends the use of S1 and S2 due to free availability and improved sensor capabilities.

Conclusion

This study utilised freely available advanced Sentinel-1 radar and Sentinel-2 optical imagery, with the aim of evaluating spectrally derived VIs and fusing Synthetic Aperture Radar (SAR) imagery for improving American Bramble (*Rubus cuneifolius*) detection and mapping. This study contributes to the evaluation of economically viable, efficient and large scale invasive alien species detection and mapping. Conventional S2 optical imagery was used as a benchmark for comparison with results achieved using S2 VIs and fused VIs and S1 SAR imagery. The use of S2 VIs increased overall classification accuracies when compared to traditional optical imagery results, while the fusion of S2 VIs and S1 SAR decreased the overall accuracies. Hence this study demonstrated that new generation S2 VIs have the potential to increase the detection and mapping of Bramble from surrounding native vegetation. Results further indicate that the fusion of VIs and SAR imagery for Bramble detection and mapping failed to increase overall classification accuracies, hence have limited utility when applied to Bramble detection and mapping. The new generation satellites, such as S1 and S2, possess unprecedented sensor characteristics like higher temporal and spatial resolution, as well as tandem acquisition of SAR data, hence valuable for improved landscape mapping. This study concludes that the recently launched Sentinel satellite, with optical and radar capabilities, holds great promise in landscape delineation and vegetation mapping.

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Translocations of European ground squirrel (*Spermophilus citellus*) along altitudinal gradient in Bulgaria – an overview

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Abstract

The European ground squirrel (*Spermophilus citellus*) is a vulnerable species (IUCN) living in open habitats of Central and South-eastern Europe. Translocations (introductions, reintroductions and reinforcements) are commonly used as part of the European ground squirrel (EGS) conservation. There are numerous publications for such activities carried out in Central Europe, but data from South-eastern Europe, where translocations have also been implemented, are still scarce.

The present study summarises the methodologies used in the translocations in Bulgaria and analyses the factors impacting their success. Eight translocations of more than 1730 individuals were performed in the period 2010 to 2018. These included 4 reinforcements, 3 reintroductions and 1 introduction. Two of the translocations are still ongoing. Five of the completed six (83%) translocations were successful, although in two cases the number of individuals was critically low. The relatively higher success in Bulgaria than in Central Europe is probably due to using the gained experience. Most of the translocations (6) used a soft release approach. In 6 cases, the animals settled 100 to 720 metres away from the release site, implying management and protection of suitable habitat beyond the translocation area. In 7 of the translocations, the altitude between the donor colony and the release site varied from 470 to 1320 m which could have a hindering effect on the adaptation of animals due to the specific conditions in the mountains. The main reasons for failure are probably poorly selected and maintained habitats and bad climatic conditions (rainy and cool weather) during the translocation action. European funds are of critical importance for translocations, with only two translocations funded by other sources. Based on the gathered data, the current paper also gives some recommendations for improvement in translocation activities.

Keywords
repatriation, reintroduction, reinforcement, *Spermophilus citellus*, european souslik, high altitude, mountain

Introduction

The European ground squirrel (*Spermophilus citellus*, also known as souslik), hereafter EGS, is a diurnal rodent living in colonies in the open habitats of Central and South-eastern Europe. Its distribution and population numbers are decreasing significantly throughout all of its range (Coroiu et al. 2008), including throughout Bulgaria (Koshev 2008, Stefanov 2015). It is listed in the European Community Directive 92/43 Appendix II and IV, the IUCN Red List (Coroiu et al. 2008) and the Bulgarian Red Data Book (Stefanov 2015) as "Vulnerable". This population status and trend has prompted a variety of conservation activities. In many countries, the species is protected and its habitats are conserved and maintained through mowing, grazing and cutting bushes and trees. Along with these activities, translocations (or repatriations) have often been implemented in order to save EGS from habitat destruction or to re-establish extinct colonies. In the last 30 years, those have taken place in Czech Republic, Slovakia, Poland and Hungary (Balaz et al. 2008, Matějů et al. 2010, 2012, Gedeon et al. 2011, 2012, Lobbová and Hapl 2014). In total, 15 repatriations have been accomplished in Central Europe (Czech Republic, Slovakia and Poland), relocating more than 3200 individuals (Matějů et al. 2010, 2012). Different methods have been used in these translocations and, in general, they can be separated into "hard" and "soft", according to the method of release (Beck et al. 1994, Matějů et al. 2012). During hard release, animals were released directly on the field, in some cases in artificial burrows, but with poorly closed holes, for example with grass. During soft release, animals are released into enclosures, in abandoned or artificial burrows with a retention cap (grass, bottle or stone) with a food supply.

All these activities were conducted in the north-western part of the species' range. The south-eastern part of the range, particularly Bulgaria, presents some unique ecological challenges and conservation opportunities. It is separated from the north-western part of the species' range by the Carpathians and the Djerdap Canyon of the Danube river (Ramos-Lara et al. 2014). The EGS population within the territory of Bulgaria has the highest genetic variability and is most likely to be the centre of the ancestral range (Říčanová et al. 2013, Chassovnikarova et al. 2015). The country's territory of 110.993 km^2 is unevenly distributed along the altitudinal gradient: 72% of the area is low elevation (0–600 m) and 27% is in the range 600–2925 m (Kopralev et al. 2002). This topography reduces the opportunities for selection and transfer of individuals in mountainous habitats, where the main protected areas are located. The EGS habitats in the mountainous regions are threatened by the reduction of grazing and pasture succession of shrubs, juniper and high grass vegetation (Koshev 2008). Conservation measures, such as pastures maintenance, have been implemented, especially after the accession of Bulgaria to the European Union in 2007. Eight EGS translocations were

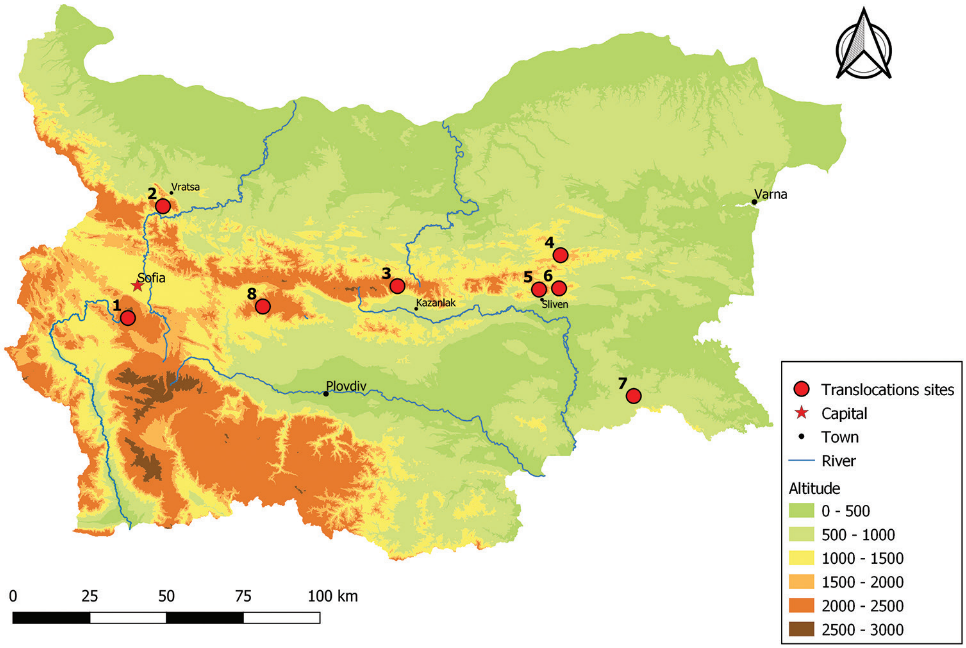

Figure 1. Map of the translocation sites of European ground squirrel in Bulgaria Legend: **1.** Vitosha NP; **2.** Vrachanski Balkan NP; **3.** Bulgarka NP; **4.** Kotlenska planina protected site; **5**. Sinite Kamani NP (Karakyutyuk and Karierata); **6.** Sinite Kamani NP (Golyamata chuka and Lokvata); **7.** Western Strandzha protected site; **8.** Luda Yana dam.

launched between 2010 and 2018. Not all the methods used have been published nor have their outcomes been analysed. The effectiveness of the translocations has not been systematically monitored.

The purpose of this article is to summarise existing information on EGS translocations in Bulgaria and to provide critical comments on the field methods and results. Four categories of information sources were used: 1) scientific publications and reports (Stoynov et al. 2013, Stefanov et al. 2016, Stoeva et al. 2016, Zidarova et al. 2018); 2) news from official websites; 3) personal data of the field crews (two sites in Sinite Kamani Natural Park (NP), one protected site in Kotlenska planina and Luda Yana dam translocations); 4) unpublished data of the authors for Vitosha NP, Vrachanski Balkan NP, Bulgarka NP and Western Strandzha protected site translocations. The unpublished data included description of the field methods and the results from 2-day field visits of 6 of the sites (without Western Strandzha and Luda Yana) in May-June 2017. The field data included burrows mapping and counting, observations and trapping. In 2018 Vrachanski Balkan NP, Vitosha NP and Luda Yana dam were visited.

An overview of the 8 translocations is presented in Table 1 and their locations are mapped in Figure 1.

Table 1. Translocations of the European ground squirrel in Bulgaria.

Location	NP Vitosha	NP Vrachanski Balkan	NP Bulgarka	Kotlenska planina protected zone	NP Sinite Kamani – Karakyutyuk and Karierata	NP Sinite Kamani – Golyama Chuka and Lokvata	Western Strandzha	Luda Yana dam
Natura 2000	Vitosha SCI&SPA[1] BG0000113	Vrachanski Balkan SCI BG0000166, SPA BG0002053	Bulgarka SCI&SPA BG0000399	Kotlenska planina SCI BG0000117, SPA BG0002029	SCI Sinite Kamani BG0000164, Sinite kamani – Grebenets SPA BG0002058	SCI Sinite Kamani BG0000164, Sinite kamani – Grebenets SPA BG0002058	Western Strandzha SPA BG0002066, Derventski vazvisheniya – 2 SCI BG0000219	Sredna Gora SCI BG0001389 SPA BG0002054
Type	Reinforcement	Reintroduction	Reintroduction	Reintroduction	Reinforcement	Introduction	Reinforcement	Rescue translocation, reinforcement
Finance source	OP[2] Environment 2007–2013	OP Environment 2007–2013	OP Environment 2007–2013	NGO own funds	OP Environment 2007–2013 project (Karakyutyuk); Life+ EU programme (Karierata)	OP Environment 2007–2013 project	Life+ EU programme	MRDPW[3]
Responsible organization	NPD[4] Vitosha	NPD Vrachanski Balkan	NPD Bulgarka	Wild Flora and Fauna Fund	NPD Sinite Kamani	NPD Sinite Kamani	Bulgarian Society for protection of Birds	IBER-BAS[5]
Duration	2011–2014	2013–2016	2013–2015	2011–2016, ongoing	2010–2014	2013–2014	2017–2019	2018
Number of locations	1	1	1	more in one area	2	2	1	1
Number of animals released	100	132	149	309	292 in Karakyutyuk 57 in Karierata	206 in Golyamata Chuka, 222 in Lokvata	167 (to be continued in 2019)	96
Method of release	Soft – in adaptation cages	Soft – in adaptation cages	Soft – in adaptation cages	Hard – in artificial burrows closed with dry hay or stone	Hard – in artificial burrows closed with dry hay or stone	Soft – in adaptation cage	Soft – in individual adaptation cages	Soft – in individual adaptation cages
Months of release	April–August	July – August	May - August	July	July–August	July–August	June–July	July
Altitude	1550 m	1420 m	1420 m	660 m	920 m	830 m	300 m	1380 m

Location	NP Vitosha	NP Vrachanski Balkan	NP Bulgarka	Kotlenska planina protected zone	NP Sinite Kamani – Karakyutyuk and Karierata	NP Sinite Kamani – Golyama Chuka and Lokvata	Western Strandzha	Luda Yana dam
Land property	State public	Municipality public and private	State-owned private and private	Municipality public and municipality-owned private, private	State public	State public	BSPB (NGO)	Mainly municipality public
Distance to the source population	36 km	44 km	13 km	27 km (to Topolchane village) 32 km (to Rechitsa and golf course)	13 km (to Topolchane village) 11 km (to Rechitsa and golf course)	13 km (to Topolchane village) 11 km (to Rechitsa and golf course)	75 km	10 km
Difference in the altitude to the source population	850 m	1320 m	970 m	470m	730m	630 m	110 m	780 m
Success in 2017/2018	Yes	Yes	Yes	Yes	Yes	No	Yes	–
Habitat preparation activities	Burrow digging with grass mowing	Burrow digging with grass mowing	Burrow digging	Burrow digging	Burrow digging	Shrubs clearing, grass mowing and burrow digging	Burrow digging	Burrow digging
Habitat maintenance activities	Grass mowing	Extensive horses grazing	Cattle grazing	Bushes removal and extensive livestock grazing	extensive sheep grazing	Shrubs clearing, sheep grazing until 2014	Extensive caws and sheep grazing	Grazing
Food provision	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes
Animal marking	Yes, transponders	Yes, transponders	Yes, transponders	Yes, toe-cutting in the more recent releases	Yes, with transponders and a hair dye	Yes, with transporters	Yes, with transponders	Yes, with hair dye
Guarding of release sites	Yes, for 30 days	Yes, for 3 days	Yes, for 30 days	No	Yes	Yes	Yes, for 10 days	Yes
Main threats and limiting factors	Lack of habitat maintenance	Unfavourable meteorological conditions	Unfavourable meteorological conditions	Decline in extensive grazing	Increased predators presence	Lack of habitat maintenece	Diseases of livestock	–

[1]Sites of Community Importance and Special Protection Areas in the Natura 2000 protected areas network (https://www.eea.europa.eu/themes/biodiversity/natura-2000/the-natura-2000-protected-areas-network#tab-data-visualisations) [2]Operational Programme [3]Ministry of Regional Development and Public Works [4]Nature Park Directorate [5]Institute of Biodiversity and Ecosystem Research, Bulgarian Academy of Sciences

Field methods used in Vitosha NP, Vrachanski Balkan NP, Bulgarka NP and Western Strandzha protected site

All Vitosha NP, Vrachanski Balkan NP, Bulgarka NP and Western Strandzha protected site translocations were implemented following one common methodology. The methodology and the results have not been published before and that is the reason why they are described here. The animals were captured with “donski” type traps – perforated (holes of 5 × 5 mm) cylindrical metal box. Trap length was 300 mm and the diameter – 53–55 mm. The rear end of the trap was closed with a lid and, at the front, there was a backflow valve (Figure 2). The traps were placed at the burrow entrance. After being caught, the animals were placed in a dark, cool and airy place until they calmed down.

Standard body measurements were taken (length of ear, tail, head and hind foot) with a vernier caliper with 1 mm accuracy. The weight was measured with portable electronic scales TH-1000A with a maximum load of 1000 g and accuracy 0.5 g. In the Western Strandzha, the animals were measured with scales Joycare JC-405B / P / JC-445 with a maximum load of 5000 g and accuracy 1 g. All the locations in the survey were recorded with the Garmin Dakota 10 GPS device.

There are no requirements and recommendations in literature regarding the age and size of the animals used for reintroductions/reinforcements. The following trapped animals were not used in the reviewed translocations and, if trapped, were released back to the donor colony: animals less than 120 g, visibly sick animals (with external injuries and/or highly infected with ectoparasites such as fleas and ticks) and nursing females. The animals, suitable for translocation, were marked with individual transponders (Datamars 12/2 mm for Vitosha NP, Vrachanski Balkan NP, Bulgarka NP and Animal Microchip Syringe encased in 12/2 mm biodegradable glass for Western Strandzha protected site), injected subcutaneously between the two shoulders. Antiseptics were applied to the area with 70% ethanol prior the manipulation. The Datamars Micromax reader was used to read the transponders. The animals were transported in cylindrical boxes made of PVC tube with a length of 400 mm and a diameter 63 mm. On each box, there are 36 openings, 10 mm in diameter, arranged in 3 rows of 12 holes on each side. The transportation boxes were specially designed to ensure that the animals were not injured (Figure 3).

All trapped animals were released in pre-prepared holes at the translocation sites on the evening after the capture day. The minimum number of holes designated to a released individual was 5. They were located at 3 to 5 metre distance from each other. The holes, made with a motorised drill (Figure 4) were 60–100 mm in diameter, 695–700 mm long and at an angle of 45 degrees. After the animals’ release, the holes were covered with grass or other material to calm down the animals in the burrow and prevent them from escaping.

Animals were released into adaptation enclosures, whose design and size varied amongst the translocation projects (Figure 4, more detailed description below). The goal of using adaptive enclosures, artificial holes and additional feeding is to reduce

Figure 2. A ground squirrel captured in the "donski" type trap.

Figure 3. Transportation boxes.

stress in the first days after animals are released, to avoid the panic displacement of individuals in inappropriate areas (forests, shrub complexes, urbanised territories etc.) and to reduce the risk of predation. Typically, a few days after the transfer, the ground squirrels found a way to leave the enclosures and settle nearby.

In Vitosha NP, Vrachanski Balkan NP, Bulgarka NP and Western Strandzha protected site, activities were implemented in accordance with the ethical recommendations and Guidelines for Reintroductions and Other Conservation Translocations IUCN / SSC (2013 and earlier) (IUCN / SSC 2013). Detailed information about the field practices is provided in the text.

Figure 4. Installation of adaptation enclosure in Vrachanski Balkan NP.

List of translocation projects

Reinforcement in Vitosha Nature Park

Vitosha Nature Park is located south of Bulgaria's capital – Sofia. The park covers the mountain Vitosha with an area of 270.79 km^2 and with an average altitude of 1317 m. The southern slopes of the park are predominantly pastures and meadows.

Prior to the translocation, a feasibility study was carried out by assessing the possible release areas and donor colonies. Four EGS colonies had historically been documented in the park, but the existence of only three of them has recently been confirmed (Koshev 2013). Over the years, there has been a decline in the distribution of the species on the park's territory. One of the densest colonies (near the village of Zheleznitsa) disappeared in 2004–2005, most likely due to lack of connection with other colonies, although the habitats were maintained in good condition by intensive cattle grazing (Stefanov and Markova 2009, Koshev 2008, Koshev 2013). According to Koshev (2013), the EGS population in the park's territory exists as small colonies (about 20–30 individuals) isolated from each other by geographic barriers (forests, gullies).

The activities for moving individuals started in 2011 under the leadership of the Vitosha NP Directorate (DNP). Animals were trapped in a colony near Kremikovtsi, city of Sofia (42.7918N; 23.4935E, 680 m a.s.l.), threatened by reduced grazing intensity, ploughing and covering with soil from the nearby mine. One of the highest densities in the country was previously calculated at this site, based on the number of holes: 15–120 individuals/ha (Stefanov and Markova 2009). The donor colony is

Figure 5. Adaptation enclosure in Vitosha NP (2011–2014).

36 km away from the release site and belongs to the same gene pool (Říčanová et al. 2013). The selected site is located southwest from Kupena peak (42.5185N; 23.2611E, 1530 m a.s.l.). A total of 5–15 ha of the site is maintained by the park through annual mowing with a self-propelled lawn mower (the area varies over the years).

The nearby game station "Vitoshko-Studena" contributes to the stable population of European roe deer (*Capreolus capreolus*) and red deer (*Cervus elaphus*) which graze in the release area. The deer are additionally attracted to the area by rock salt regularly placed by the park authorities.

A 150 m^2 adaptation enclosure was built to prevent attacks by predators (Figure 5). It had a reinforced base (inserted into the grass), walls of metal wire-net and a top of plastic mesh. Artificial burrows were created inside and around the enclosure. One hundred individuals were released inside the enclosure in the period 2011–2014 (Table 2). The sex ratio was 1:1.45 in favour of females. One-year-olds were 60% of all catches (data for 2012 were not available at the time of writing this article, but the proportions were similar).

The animals were released in several sessions with a maximum of 20 in the enclosure per session. The sessions occurred at intervals of 5 to 25 days. This allowed the previous group of animals to adapt and spread in the vicinity. After their release, the animals were observed daily for one month in order to prevent predator and human interactions and to collect data for their initial behaviour and adaptation. Sunflower seeds, wheat and apples were provided as supplementary food.

Reinforcement can be considered successful. In the first year (2012), the animals survived the winter and settled in the new place. In 2012, two individuals were seen at the release site. In 2013, 13 holes and several individuals were reported, with holes

Table 2. Sex and age ratio of the EGS, translocated in Vitosha NP.

Year	Period of realising	Female	Male	Juveniles and subadults	Adults	total
2011	8–9.08.	12	8	13	7	20
2012	25.04.–30.05.	*	*	*	*	40
2013	6–14.07.	11	9	9	11	20
2014	9–10.8.	12	8	15	5	20
Total		**35**	**25**	**37**	**23**	**100**

* data not available

found in a new location. In June 2017, the colony had moved about 150 metres away from the release site, 52 burrows were recorded and 4–5 individuals were observed. A one-year-old unmarked male was trapped. Thirteen holes were counted in 2018. The habitat is a humid meadow with naturally low vegetation and there was no evidence of grazing by livestock or mowing. The observed density was very low and the future survival of the colony is uncertain without further reinforcement and habitat maintenance.

Reintroduction in Vrachanski Balkan Nature Park

The Vrachanski Balkan Nature Park is located south of the regional centre Vratsa in the western Stara Planina mountain. The park has an area of 288.03 km^2 and an average altitude 700 m (Bechev and Georgiev 2016). There are historical data of the species' presence near the release site (Parshevitsa hut), but the species disappeared at the end of the 1950s due to a ban on transhumance (G. Stoyanov – unpubl. record, Nedyalkov and Koshev 2016). Vrachanski Balkan NPD started EGS reintroduction activities in 2013.

The release site (43.1379N; 23.4855E, 1420 m a.s.l.) had previously been maintained through mowing, horse grazing and rarely sheep grazing (Figure 6). Other favourable factors for the site selection were the limestone rock base, good soil permeability and southern exposure. Four adaptation enclosures were installed, each measuring 5 × 7 m or a total area of 560 m^2. They were covered with plastic mesh and their walls of metal wire-net were inserted into the grass. Artificial burrows were dug.

The closest mountainous EGS colonies are in Ponor Mountain and colonies have gradually been decreasing from 2008 to 2011 (Koshev 2014). Therefore, they were found to be inappropriate for donor colonies. In 2013, four individuals were transferred from the colony near the village of Chiren, but future captures were discontinued due the low density of the colony. A new donor colony was identified near the village of Kobilyak (43.5215N; 23.443E) which was situated 36 km away from the release site. This donor colony covered a large area and had a good density of 30–40 ind/ha. A total of 132 animals were captured and translocated during the 4 years of the project (Table 3). Five to seven animals (1 adult male, 1–2 adult females and several juvenile and sub-adults) were released in each enclosure. Oats and sunflower seeds were provided as supplementary food. An increased aggregation of predators (red fox

Figure 6. Picture of 2 of the 4 adaptation enclosures in Vrachanski Balkan NP.

Table 3. Sex and age ratio of the individuals, translocated in Vrachanski Balkan NP.

Year	Period of releasing	Females	Males	Juveniles and Subadults	Adults	Total
2013	4–15.07.	25	15	17	23	40
2014	13.06–12.08.	28	23	34	17	51
2015	19.07.	6	8	11	3	14
2016	30.07.	16	11	19	8	27
Total		**75**	**57**	**81**	**51**	**132**

(*Vulpes vulpes*), common buzzard (*Buteo buteo*), common ravens (*Corvus corax*) and common kestrels (*Falco tinnunculus*)) was observed around the enclosures.

The monitoring showed 2 adult and 1 juvenile animals in 2014, 4 animals in 2015 and 4 animals in 2016. In 2017, it was found that part of the colony had established 190–390 m away from the release site. A total of 58 holes were detected, 10 individuals were observed, 2 individuals were captured – one juvenile female and one adult male previously marked in 2015. In 2018, one individual was observed and 40 holes were counted. In the same year, the park's authorities reported observing ground squirrels 600 m away from the release site (Klyuchni dol locality). This information was not confirmed upon field checking.

The establishment of a colony and the juveniles observed/captured indicate that the translocation was successful. On the other hand, the number of observed individuals is too small and it can have a negative impact over a longer period of time.

Reintroduction in Bulgarka Nature Park

Bulgarka Nature Park is situated on the northern slopes of the Central Stara Planina mountain with an area of 236.9 km^2 and an average altitude of 870 m. A total of 89% of the park area is covered by forests. EGS colonies were previously documented in the area of Uzana hut (V. Popov – personal data for 2003), Karamandra locality, "St. Nikola" (Shipka) and Budzludzha peaks, all situated on the borders of the park (Koshev 2013).

A donor colony was selected near the village of Kran (42.6788N; 25.3770E, 480 m a.s.l.) – 12–13 km away from the release site. Location near the border of the park named Karamandra (42.7410N; 25.2510E, 1410 m a.s.l.) was selected for a release site. It has been inhabited by EGS in the past, according to the local people. The low (10–15 cm) grass vegetation is maintained by about 100 cows. The water source for the animals is located at a nearby hut built by the park authorities.

Four adaptation enclosures were built (5 m × 7 m or a total area of 560 m^2) (Figure 7). They were covered with plastic mesh (to prevent attacks by predators) and their walls of metal wire-net were inserted into the grass. Artificial burrows were dug. For the period 2013–2015, 149 individuals were released (Table 4). The sex ratio was in favour of females (1:1.48). The adult animals were 52%. Five to seven animals (1 adult male, 1–2 adult females and several juvenile and sub-adults) were released in each enclosure. Additional feeding (sunflower seeds, apple) was provided. The area had a permanent guard for one month. Predation by a young imperial eagle (*A. heliaca*), red fox (*V. vulpes*), common buzzard (*B. buteo*), common ravens (*C. corax*) and common kestrels (*F. tinnunculus*) has been observed.

In the period 2013–2014, the animals were released by less experienced externally hired experts. In 2015, the park's authorities independently organised releases on 10 July and 31 August (S. Staykov – unpubl. records). The later release date is close to the start of the hibernation for the species, which poses a threat to their successful adaptation.

In 2014, two individuals were observed and 11 holes counted. The grass was high, owning to the bad weather which forced local shepherds to move livestock to the mountains at a later date. In 2015, several animals were observed before the new release. In 2016, the park's authorities reported observing individuals, but this was not confirmed by the regular monitoring.

No signs of ground squirrels' presence were found at the release site in 2017. The colony was discovered on the southern slopes of Ispolin peak (42.7334N; 25.2520E) which is 720 m (suitable habitat path) away from the original release site at the same altitude (1420 m a.s.l.). Forty seven holes were counted, a minimum of 15 animals were observed and a juvenile female was captured. The habitat is in good condition with high plant diversity and sufficient grazing. The translocation could be considered successful, as the animals formed a colony and there were signs of reproduction. However, the colony's future is uncertain due to ongoing and expected new disturbances. The site is regularly visited by motor vehicles for sightseeing and is being researched for installing wind turbines.

Figure 7. Four adaptation enclosures built in the Bulgarka Nature Park (2013–2015).

Table 4. Sex and age ratio of the animals, translocated in Bulgarka Nature Park.

Year	Period of releasing	Females	Males	Juveniles and Sub-adults	Adults	Total
2013	22–23.06; 9–10.07.	23	17	19	21	40
2014	23.05; 14–28.06; 21–22.07.	21	20	23	18	41
2015	12–24.07; 10–31.08.	45	23	29	39	68
Total		**89**	**60**	**71**	**78**	**149**

Reintroduction in Kotlenska Planina Natura 2000 Site

Kotlenska Planina is a Natura 2000 protected site in Eastern Stara Planina mountain near the town of Kotel, with an area of 690.58 km^2 and highest peak Razboina (1128 m a.s.l.). The main habitat types are deciduous forests and secondary steppe habitats, maintained by livestock grazing (Stoynov et al. 2013).

The 2007 and 2008 assessment of the site's suitability for EGS translocation showed a shallow soil horizon, dense soil and rocky terrain (Y. Koshev – unpubl. records). The closest EGS colony is 25 km away (Stoynov et al. 2013), separated by severely intersected mountainous terrain which greatly hinders the natural exchange of individuals. According to Stefanov et al. (2016), EGS had been considered extinct in the region of Kotel since 1990 due to the decline of the extensive livestock-grazing. However, the historic presence of the species in this area is questionable because the four specimens from the Kotel Natural History Museum were from locations outside the target area (Y. Koshev, D. Ragyov – unpubl. records).

In 2011, the Fund for Wild Flora and Fauna started an EGS reintroduction project in the area Urushki Skaly (42.922N; 26.4617E, 660 m a.s.l.), where the organisation had been protecting and restoring semi-natural grassland habitats through traditional methods of cattle grazing since 2000 (Stoynov et al. 2013). The source colony inhabited a golf course near the town of Sliven (42.6386N; 26.2914E) where the species was unwanted. In 2014, the golf course failed, consequently the colony disappeared and a new source colony was designed (near Topolchane and Kaloyanovo villages, Sliven district, 42.666N; 26.441E, 180 m a.s.l.).

The animals were released in 6 sites that were not geographically isolated and were close to one another. Areas with weak slope, soil layer depth 60–80 cm and grass cover with height under 15–20 cm and projective cover below 80% were accepted as appropriate habitats (Stoynov et al. 2013; Stefanov et al. 2016). The animals were released in artificial burrows with tunnel length of 60–100 cm and a diameter of 5 cm. The translocation took place after the juveniles had attained independence and before the start of their hibernation (Stefanov et al. 2016). The initial releases in 2011 and 2012 were in two sites, 200 m apart from each other. In total, 309 individuals are released. The increase in the number of livestock due to the favourable conditions in 2011–2015 encouraged a new round of translocations in 2015 and 2016 (Stefanov et al. 2016). The number, the sex and the age ratio of the released animals are presented in Table 5 (according Stefanov et al. 2016, V. Stefanov – unpubl. records).

According to the official overview in 2013, 80 burrows were found (Stoynov et al. 2013). The area used is about 10 ha and the animals are concentrated in three distinct plots.

In 2014, there were 300 sheep of the Karakachan breed, 4 cows and about 20 goats in the region and about 30 inhabited holes were found in the area. Only a few holes have been found in some of the areas. Due to the rocky terrain, only in small, separate areas (with a sufficiently deep soil layer) have the animals managed to settle and dig their shelters.

In 2014, two colonies were observed within an area of 0.5 and 2.4 ha. The number of holes remained the same and the density was estimated at 5–10 animals/ha (Stefanov et al. 2016).

In May 2017, all the release sites were visited and interviews with the involved professionals were held. On the 2012, 2015 and on one of the 2016 release sites, fresh burrows were found and animals were observed. Neither holes nor animals were found on the 2011 release site and the second 2016 release site (more remote and dry). The overall habitat condition was very good: the grass height was less than 10 cm and was maintained by the local livestock which numbered about 300 cows, 300 sheep as well as goats and buffaloes.

Predation of ground squirrels in the area was confirmed several times: domestic cat (*Felis catus*), domestic dog (*Canis familiaris*) and common buzzard (*B. buteo*) (Stoynov et al. 2013).

This project involved one of the highest numbers of translocated animals in Bulgaria (Table 1). Due to the fact that new holes were found and juveniles observed, reintroduction can be considered successful. The hard release method is used as there

Table 5. Sex and age of the animals, translocated in Kotlenska planina (Stoynov et al. 2013, Stefanov et al. 2016, V. Stefanov – unpubl. records).

Year	Period of releasing	Females	Males	Juveniles	Adults	Total
2011	July	36	28	43	21	64 (57*)
2012	July	27	29	37	19	56
2015	–	25	19	39	5	44
2016	–	**	**	**	**	145**
Total		**88**	**76**	**119**	**45**	**309**

Note: * the authors give different data on released individuals in 2011 – according to Stoynov et al. (2013), there were 57 and according to Stefanov et al. (2016) – 64, the differences being at the expense of young male and female individuals; ** only the total number was available (V. Stefanov – unpubl. data).

is no evidence of: monitoring the populations from which the animals were caught; no use of enclosures; no data on weather conditions at release; not all individuals were marked; transport boxes were not used; the animals were not guarded from predators; no selection of individuals was performed on the basis of their individual weight, infestation with external parasites, physiological and health status. Under these conditions, it is easy to explain the observations by Stoynov et al. (2013) and Stefanov et al. (2016) that the animals dispersed and rarely inhabited the artificially-made burrows.

Sinite Kamani Nature Park – two main translocations

Sinite Kamani NP, with an area of 113.80 km^2, is located in Eastern Stara planina mountain, north of the town of Sliven. Its altitude is between 300 m and 1181 m. The EGS conservation status became unfavourable on the park's territory in the late 1980s and early 1990s due to the abandonment of extensive farming and pastures (Koshev 2013, Stoeva et al. 2016). This status encouraged a reinforcement project, which started in 2010 and was led by NPD Sinite Kamani.

Prior to the start of the translocation, a study of the potential donor colonies was carried out in the lowland in a perimeter of about 30 km around the town of Sliven. The main donor colonies selected were: the golf course in Rechitsa district (Sliven) (12 km away) and the pastures of the villages of Topolchane and Kaloyanovo (7–12 km away). These populations belong to the same genetic line (Říčanová et al. 2013) and provide appropriate source colonies. Their density has been studied and threats to the colonies have been evaluated. Both colonies are at risk. The animals in the golf course were purposefully killed by the owners and later, when the course stopped functioning in 2013, habitat succession restarted. The pastures around the villages of Topolchane and Kaloyanovo are ploughed for farming and there is a high mortality rate from the nearby busy road Sliven–Burgas.

For the purpose of translocation, the animals were trapped in the golf course until 2014 and near Topolchane village after that. "Donski" type traps and rat traps with apple bait were used. The individuals were marked with standard Felixcan microchip transponders. They were placed in transport boxes and transported to the release sites

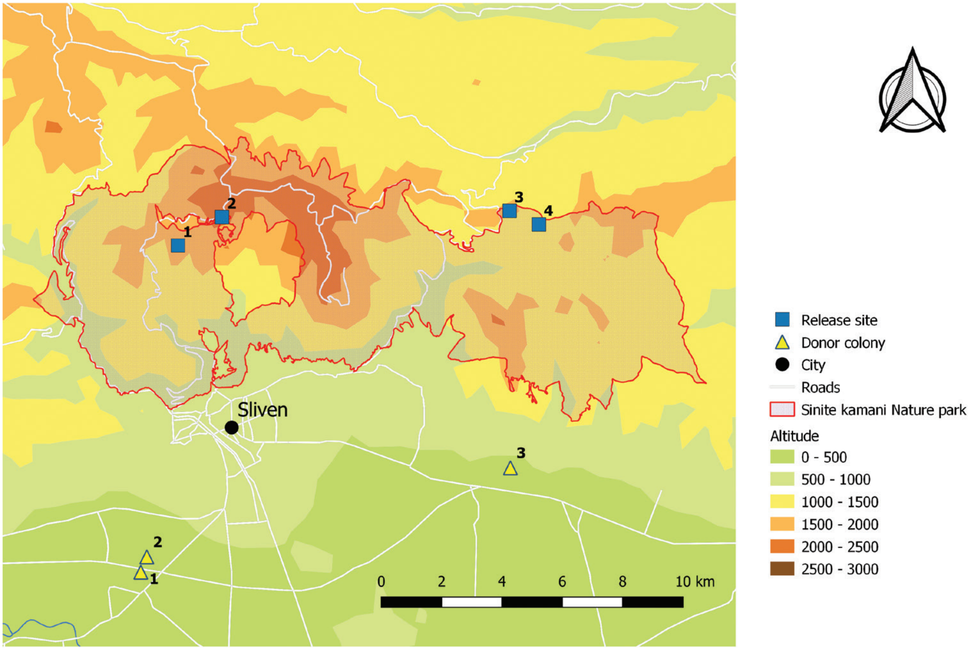

Figure 8. The translocation sites and main donor colonies in Sinite Kamani Nature Park. Legend: Translocations (squares): Reinforcement in the sites Karakyutyuk (**1**) and Karierata (**2**); Introduction in sites Golyamata chuka (**3**) and Lokvata (**4**). Main donor colonies (triangles): (**1**) Rechitsa district, (**2**) golf course, (**3**) villages of Topolchane and Kaloyanovo.

on the day they were caught. All related activities and manipulations were performed in the presence of a veterinarian. Juveniles, underweight individuals and visually unhealthy animals were not translocated (Stoeva et al. 2016).

In the region of "Sinite Kamani" NP, two different type of translocations (reinforcement and introduction) were implemented at four locations (Figure 8). As the release sites were separated by landscape barriers such as forests and gullies and adaptation fences were used on some but not others, they were considered distinct translocations, despite all being on the park's territory.

Reinforcement in the Karakyutyuk and Karierata sites

Karakyutyuk site (42.7375N; 26.3049E, 930 m a.s.l.), has an area of 34.56 ha and harbours a small EGS population of almost critically low abundance (about 20 individuals). In order to increase the habitat quality, the shrub and tree vegetation was cleared and the site was managed through extensive sheep grazing and mechanical maintenance. Additional food was provided. There was a spreading of clover seeds to improve the vegetation. Artificial burrows (80 cm deep at 45°) were dug. The animals were released in the evening after being trapped. The holes are plugged with a large tuft of grass. The

Table 6. Sex and age ratio of the animals, translocated in Karakyutyuk site (Sinite Kamani Nature Park) (Stoeva et al. 2016).

Year	Periods of releasing	Female	Male	Juveniles	Juveniles/ 1st year	Adults/ 1st year	Adults	Total
2010	7–22.07.	27	30	23	9	3	22	57
2011	30.06.–23.08.	67	52	68	8		43	119
2012	25.06.–25.07.	35	31	41	–	–	25	66
2013	19.08.	5	5	–	6	–	4	10
2014	28.6.–22.7.	22	18	–	25	–	15	40
Total		**156**	**136**	**132**	**48**	**3**	**109**	**292**

activities continued annually from 2010 to 2014. The exact periods of release, number, sex and age of the repatriated animals are presented in Table 6 (Stoeva et al. 2016).

The estimation of the sex index for the entire period shows that females prevailed, accounting for 53.4% of the individuals, compared to males – 46.6%. Most of the released individuals were juveniles (45.2%), while the adults comprised 37.3% (Stoeva et al. 2016). The post release guarding was implemented though camera-traps in key locations and supplementary feeding to divert the predators. Guards were also present to monitor the behaviour of the individuals and chase away predators.

A remarkable success was reported in 2014, the number of holes increasing by 520% compared to 2010. With the active holes providing a more precise picture of the relevant population abundance of the species, there was an impressive increase by 1352% compared to the numbers recorded in 2010 prior to the release of the first individuals (Stoeva et al. 2016).

In June 2017, 62 holes were counted through the transect method. Several animals were seen. The habitat was in good condition, with low grass cover. Although the site is rocky, which is generally considered unsuitable for the ground squirrel, there was no apparent preference observed for the less rocky areas. Many burrows were dug directly under the stones and some were even dug under the fence's poles fixed in cement. A possible explanation is that the fence provides a defence against raptors by hindering their flight. The raptors are abundant in the area and a couple of booted eagles (*Hieraaetus pennatus*) were reported feeding regularly on the colony. European wildcats (*Felis silvestris*), golden eagle (*Aquila chrysaetos*), golden jackals (*Canis aureus*), hawks (*Accipiter* sp.) and foxes (*V. vulpes*) were also observed. The increased predator pressure could also be due to the proximity of a vulture feeding site (although a camera trap showed EGS feeding there). Human presence and livestock grazing in the area are prominent. Two hundred sheep are owned by the NGO and they plan to increase this number to 500. This will assure the habitat maintenance.

In conclusion, the reinforcement in the Karakyutyuk region has been successful. An increase in both the area and the number of holes was recorded. The following success factors were identified: initial presence of animals, preparation of the habitat, existence of sustainable grazing, maintenance of the habitat and sufficient number of individuals, properly selected for release, nourished and secured.

Karierata site (42.7417N; 26.3217E, 1070 m a.s.l.) is located at the foot of the highest peak (Bulgarka) in the Natural Park Sinite Kamani, 1.3 km away from the Karakyutyuk. In May 2011, a small EGS colony was found. This colony was near a stone quarry at the foot of the peak. It is connected with the Karakyutyuk site through suitable habitats and maybe the two colonies were part of single colony split after a drop in the population number (E. Stoeva, I. Ivanov – unpubl. records). There were 38 holes in 2011, of which 29 were active. One individual was observed and two others were heard. Researchers suggested that this colony was on the verge of extinction and, therefore, proceeded with its reinforcement in 2016 and 2017 (Table 7), which is still ongoing (E. Stoeva, I. Ivanov – unpubl. records).

Table 7. Sex and age ratio of the animals, translocated in Karierata site (Sinite amani NP).

Year	Period of releasing	Females	Males	Juveniles	Subadults	Adults	Total
2016	8–21.07.2016	15	17	4	–	28	32
2017	–	11	14	8	–	17	25
Total		**26**	**31**	**12**	**–**	**45**	**57**

In 2017, the habitat seemed appropriate – there was a natural low grass cover – although it was stony and near a road. Fresh holes and excrement were found, but no animals were observed. Fifteen holes with faeces were counted. The reinforcement continues, so it is still not possible to evaluate the success of the activities.

Introduction in the Golyamata chuka and Lokvata sites

The places for introduction are selected by the NPD Sinite Kamani without preliminary research. There is no historical data on the EGS presence there. The area consists of several open habitats surrounded by forests with a total area of 45 ha.

Golyamata chuka locality (42.7446N; 26.4396E, 830 m a.s.l.) has an area of 6.8 ha. Initially, part of it (1.36 ha) was cleared in 2013, then the entire area was cleared in 2014, leaving single trees and shrubs (5% of the area). The nearest EGS colonies are 10 km (Karierata) and 12 km (Karakyutyuk) away. A fence, enclosing an area of about 500 $m^{2,}$ was built in 2013 and this was doubled in 2014 to reach an enclosed area of 1000 m^2 (Figure 9). The wire-net was buried 10 cm in the ground. Artificial burrows (40 cm deep, 7 cm in diameter) were dug at an angle of 45° inside and around the enclosed area. Security cameras and signs were installed. The animals were fed with wheat and guarded for 1 month. Sheep grazing by a local farmer was implemented in the summer of 2014 and 2015 and high vegetation was cleared 2 times (E. Stoeva, I. Ivanov – unpubl. records).

A total of 207 individuals were released at that site – 101 in 2013 (caught on a golf course near Sliven) and 106 in 2014 (caught in Topolchane colony and golf course) (E. Stoeva, I. Ivanov – unpubl. records) (Table 8). The estimated survival after the first year of the translocation was 10%. Only 7–8 animals were found in 2015. The population kept decreasing and no signs of ground squirrels' presence were found on the site in 2017.

Figure 9. The adaptation enclosure in Golyamata chuka locality in 2014 (**a**) and 2017 (**b**).

Table 8. Sex and age ratio of the animals, translocated in Golyamata chuka site (Sinite Kamani NP).

Year	Period of releasing	Males	Females	Juv	Subadults	Adults	Total
2013	–	–	–	–	–	–	101
2014	10.06–17.07	52	54	44	–	62	106
Total							**207**

Table 9. Sex and age ratio of the animals, translocated in Lokvata site (Sinite Kamani NP).

Year	Period of releasing	Females	Males	Juveniles	Subadults	Adults	Total
2010	21–22.07	33	18	25	10	16	51
2011	–	59	62	73	7	41	121
2013	August	–	–	–	–	–	10
2014	21–27.07.	–	–	–	–	–	40
Total							**222**

In **Lokvata** locality (42.7394N; 26.4516E, 780 m a.s.l.), with a total area of 2.65 ha, is a former pasture heavily overgrown with shrubs. The release area of 0.74 ha was cleared for the translocation. The area is 1 km away from Golyamata Chuka, 10 km from Karierata and 12 km from Karakyutyuk. It is separated from the former two sites by forests and gullies. In total, 222 individuals were released (E. Stoeva, I. Ivanov – unpubl. records) (Table 9).

The monitoring of Lokvata's activities shows a modest initial success – 12 active holes were mapped in early June of 2013, 66 holes were counted in 2014 and 84 holes again in 2014 after releasing the new animals (E. Stoeva, I. Ivanov – unpubl. records). According to the project staff, the success of the introduction was hindered by the succession of pastures and the lack of remnant colony. The 2017 monitoring showed no presence of EGS and shrubs cover of more than 40%. The cottage and the photo-shelter built on the site were abandoned.

In conclusion, the Golyamata chuka and Lokvata translocation activities could be considered unsuccessful as no sign of ground squirrels was found in 2017. The most likely causes are the rainy and cold weather in 2014 (see Discussion), as well as the lack of data for the EGS presence in the past, the small and fragmented habitats and insufficient habitat maintenance (Figure 9).

Reinforcement in Western Strandzha Natura 2000 Site

The Natura 2000 site Western Strandzha (BG0002066) is situated in a hilly area in south-eastern Bulgaria near the border with Turkey. The EGS populations in the region are declining (Koshev 2013, Y. Koshev, D. Demerdzhiev – unpubl. records) because of pasture abandonment. A reinforcement of the colony near the village of Momina Tsarkva (42.1513N; 27.0061E, 300 m a.s.l) was started in 2017 by the Bulgarian Society for Protection of Birds (BSPB) in partnerships with local farmers. The activities' core area (1.65 ha) is owned by the same non-government organisation. The surrounding area (around 10 km^2) is constantly grazed by sheep and cows. Thirty seven individual cylindrical adaptation cages (51 cm high, 45 cm in diameter) were built to reduce the stress (Figure 10). Each of them was placed above an artificial burrow. Ninety six animals were translocated from the Topolchane and Kaloyanovo colony in 2017 and an additional 71 in 2018 (Table 10). This donor population was chosen because of its size, demography and genetic similarity (Říčanová et al. 2013). Apple, carrots and sunflower

Table 10. Sex and age ratio of the animals, translocated in Western Strandzha.

Year	Period of releasing	Females	Males	Juveniles	Subadults	Adults	Total
2017	6–19.07	54	42	49	14	33	96
2018	21.06–19.07	35	36	34	8	29	71
Total		**89**	**78**	**83**	**22**	**62**	**167**

Figure 10. A ground squirrel released in the individual cage for initial adaptation in Western Strandzha protected site.

seeds were provided as supplementary food. The animals' adaptation progress in the new environment was documented by recording their behaviour through radio-tracking and direct observations. Samples were taken for stress hormones and endoparasites.

Rescue transfer (reinforcement) of EGS from the bottom of "Luda Yana" dam

In 2016, the Ministry of Regional Development and Public Works (MRDPW) resumed the construction of Luda Yana Dam near the town of Panagyurishte and overflowing was planned for 2018. An EGS colony had been reported to exist in the flood-threatened zone in 1997–2018 (S. Lazarov, Y. Koshev, S. Uzunov - unpubl. records). The colony occupied an area of 0.420 km^2 (42.5229N; 24.2044E) at 550–600 m a.s.l. A total of 90% of it was destroyed during the construction of the dam by removal of the humus soil layer, pouring of earth masses into the EGS habitat, disturbance from the construction machinery and other construction-related activities (Figure 11). A small, core part of the colony occupying an area of 0.04 km^2 survived near the river before the flooding of the dam. In June 2018, the colony was estimated to have about 70–150 active holes or about 50–60 adults. Juveniles were observed at that time.

In the period 24.06–20.07.2018, the first EGS rescue translocation in Bulgaria began. About 90 artificial burrows were dug at the release site. They were 60–80 cm deep

Figure 11. EGS colony in the future bed of the Luda Yana dam, June 2018.

and have a 45° slope. Ninety six animals (of which 68 were juveniles) were trapped and temporarily marked with hair dying. They were transferred 10 km away to the Belotrup area (42.6095N; 24.2561E, 1380 m a.s.l.). Individual adaptation cages were placed over the artificial burrows. Additional feeding (seeds, carrots and apples) was provided (World Bank news, Zidarova et al. 2018).

The Belotrup area was inhabited by a viable colony prior to the translocation. The habitat was a maintained pasture with an active hut and artificial ponds (Koshev 2013, Y. Koshev, D. Ragyov – unpubl. records). The population for the whole protected site was estimated at 0.73 burrows/100 m transects (n = 71) (SD = 1.38) and the conservation status was unfavourable-bad (Koshev 2013). There is no evidence that the specific population in Belotrup has been subjected to anthropogenic pressure, negative factors or risk of extinction.

Shortly after the translocation, the release site experienced severe disturbance: an annual gathering involving more than 130 high passable off-road vehicles took place on 5–10 July 2018. The event included building of temporary camps, tents and feeding places, as well as fires and loud music. Historical events were recreated with gunshots and pistols (Srednogorie website, P. Tsvetkov – personal communication). All these activities likely impacted the adaptation and acclimatisation of the animals. In addition, the months of June and July 2018 were extremely rainy. The target area is located on a mountain ridge where rainfall is more frequent and more abundant (NIMH-BAS 2018). The results of this first rescue and EGS repatriation in Bulgaria are yet to be assessed in the coming years.

Discussion

Assessing the success of translocations in Bulgaria

A total of 1730 EGS individuals were translocated in Bulgaria between 2010 and 2018. The success of the activities was assessed according to three criteria: survival (phase I), settlement (phase II) and reproduction of the released animals (phase III) (Letty et al. 2003, Teixeira et al. 2007, Matějů et al. 2012). Therefore 5 out of 6 accomplished translocations (83%) where reproduction was detected can be considered successful. In other parts of Europe (Matějů et al. 2010, 2012), only half of the relocations were successful. The relatively higher success in Bulgaria is probably due to the accumulated experience from Central Europe shared in numerous articles (Adamec et al. 2006, Balaz et al. 2008, Ambros 2008, Matějů et al. 2010, 2012, Tokaj et al. 2012, Lobbová and Hapl 2014), guidelines (Hapl et al. 2006), experimental studies (Gedeon et al. 2011, 2012) and others. Failure is probably due to poor preparation of the new site, lack of further habitat maintenance and/or poor weather conditions.

In some cases, even if the repatriation is reported successful, the number of individuals in the newly formed populations is low, which can lead to inbreeding or population density reduction when catastrophic events occur (heavy spring snowfall or torrential rains). That is why the populations in Vrachanski Balkan NP and Bulgarka NP should be strengthened with more individuals.

Our analysis shows that choosing the right release site is crucial for rescue transfers. In the case of Luda Yana dam, the release site was inhabited by a stable known population, well connected with other populations. There were no data to indicate that it was decreasing or at risk (Koshev 2013). Bringing more individuals to it was expected to lead to a temporary increase in the population size. Then, once the capacity of the environment was reached, the number of the individuals should drop to the pre-release level. Uncertainty remains whether such rescue actions have only short-term positive effects and do not contribute to the species' conservation in the long-term. Although the genetic diversity is expected to increase in the reinforced colony, there is an increased risk of introducing parasites (Golemansky and Koshev 2009) and diseases to a healthy population.

Target areas, funding sources and sustainability

Translocations were conducted in two types of protected areas: Nature Parks and Natura 2000 sites. The Nature Parks are preferred for several reasons. First, the land there is often owned by the state or municipalities. Another advantage is that the park administration manages the grazing and mowing, gain extra funding (OPE), guard the areas etc. A possible drawback is that NPDs tend to prepare similar project proposals for reintroductions/reinforcements without implementing preliminary research or consultation with a specialist, which leads to difficulties with the choice of release place, donor colony etc.

Translocations in Natura 2000 sites include Kotlenska Planina Mountain, Western Strandzha and Sredna Gora (Luda Yana Dam). The Natura 2000 network also could be suitable for such activities – some funding opportunities exist. The main problem is the ineffective protection that only exists "on paper" (Duprey 2014). Examples are the EGS colonies on the pastures of the villages of Topolchane and Kaloyanovo (Figure 12) or Besaparski ridove that are being ploughed despite their conservation status (Nedyalkov and Koshev 2014).

In Kotlenska Planina, the ownership of the land in the release site is not mentioned by the authors (Stoynov et al. 2013, Stefanov et al. 2016), but the organisation leading the activities (Fund for Wild Flora and Fauna) has managed the habitat through sheep herding for several years. In Western Strandzha, the activities have been implemented on land owned by another non-government organisation, the BSPB and the management of the habitats has been undertaken jointly with the farmers – partners of a LIFE project.

Sometimes, the planned and the actual dispersal of released animals differ. For example, the target area in Bulgarka NP is on the border of the park as this was the only suitable habitat with available data for the species' presence in the past. However, the new colonies have settled outside its borders of Bulgarka NP in the Natura 2000

Figure 12. Source colony of Topolchane and Kaloyanovo EGS habitat freshly ploughed (2018).

protected site "Central Balkan – buffer" on land owned by the state where grazing is still subsidised, but a new threat from wind turbines has recently emerged.

Most of the translocations described (6) are financed under the Operational Programme Environment (OPE) of the European Union and one translocation (in the Western Strandzha) is financed under the EU's Life + program. The rescue action is financed by the MRDPW, which is a precedent for Bulgaria. One translocation (that in the Kotlenska Mountains) had no specified funding and was probably undertaken with the responsible NGO's own resources. The main donor – OP Environment – does impose some restrictions, such as the impossibility to postpone the activity for another year in the presence of unfavourable climatic conditions, as those observed in 2014. That lack of flexibility could lead to increased mortality amongst the translocated animals.

EU funds are crucial for conservation activities, such as EGS translocations, because national funding is lacking. Since all funding is project-based, with funds covering only the translocation activities, it is difficult to conduct systematic monitoring that reliably assesses the conservation effect. The same applies to the site management (mowing, grazing that also requires special regular funding). In the cases where such management is necessary, the end of the funding project could also be the end of the habitat maintenance activities (Figure 9b). When the agricultural activities maintaining the habitats in the target areas are stimulated by EU subsidies, changes in EU's common agricultural policy could have huge consequences for the newly established colonies. One possible solution for implementation in the Western Strandzha translocation is to choose an area owned by the organisation implementing the project. Even in that case, there is a certain degree of dependence on the surrounding conditions as the grazing is provided by local farm animals. Another possible solution is practised by the NGO Green Balkans, which has its own livestock permanently based in the area (Sinite Kamani NP).

Choice of source colony, genetic diversity and risks

In 4 translocations (Sinite Kamani NP (2 cases), Kotlenska planina and Western Strandzha) the main donor colony was in the pastures near the villages of Topolchane and Kaloyanovo (Sliven region). This is the Bulgarian EGS population with the highest genetic diversity (Říčanová et al. 2013). It is only partially protected, with a small part being included in the Natura 2000 protected site, designed according to the Birds Directive, but not protected by the Habitats Directive. The estimated density in 2016 is 12.95 holes / 0.05 ha. One of the highest densities of EGS in the country is believed to be here.

Part of the area is recorded as agricultural land in the registers, although it has been used for over 20 years as pasture. Landowners, stimulated by the EU subsidy policy, are taking steps to plough the pastures (Figure 12). Since it is impossible to predict which section will be ploughed next, we consider the translocations to be rescue actions.

The choice of the other donor colonies was mostly driven by the available options. Since the activities are implemented in mountainous areas (on the territories of Vitosha NP, Vrachanski Balkan NP, Bulgarka NP) where no suitable source colonies existed, the donor colonies were from relatively distant regions.

A very important feature of the donor colony is that it should be from the same genetic pool as the area of release or the colony that is amplified. According to Kryštufek et al. (2009), the planning of EGS conservation and translocation activities in Bulgaria should be particularly careful due to the presence of the two genetic lines on the territory of the country.

Difference in altitude between donor colonies and release sites

In Bulgaria, EGS is distributed from the sea level to more than 2500 m. The highest altitude records are for 2593 m in Rila Mountain (Y. Koshev, V. Milushev - unpubl. records). The species also occupies high altitudes in Vitosha Mts, Rodopi Mts, Sredna Gora Mts, Central and Western Stara Planina Mts (Koshev 2008). Most of the countries' conservation areas, such as nature parks, are situated in the mountains. Yet, few suitable donor colonies are available there. For this reason, the altitudinal difference between donor and release sites in seven of the analysed translocations ranged from 470 to 1320 m. Even when high flexibility existed regarding the translocation area

(Luda Yana dam), the animals were still transferred to an area located about 800 m higher than where the initial colony was.

The difference in altitude has several negative effects:

- With every 100 metres increase in elevation, there is a decrease in temperature by 1 °C. Increasing altitude also increases the wind speed. Weather in the mountains is rainier and windier, i.e. less favourable than in the lowlands (Kopralev et al. 2002). In these conditions, the daily activity period of the EGS is reduced (Katona et al. 2002, Koshev and Kocheva 2008).
- Translocations are carried out on the mountain ridges where the open alpine area is usually the suitable habitat. In these areas, the rainfall and the wind speed are especially high.
- EGS has a pronounced life cycle dominated by hibernation. Due to different mountain conditions, the difference in hibernation period may reach up to 2 months (Y. Koshev – unpubl. records).

These altitude differences lead to numerous related problems. When EGS in the lowlands are in a period of growing and the juveniles need to be moved to a higher altitude, the warm season has not yet started there; temperatures are low, especially during night and morning hours. The vegetation is not yet well-developed, the grasses give seeds later in the season etc. The type of habitat and hence the food resources differ. Ružić (1950) found that females from mountain areas in Serbia have more embryos (average 6.1) than females from lowland (average 4.7). Additionally, the reproduction season in mountain populations is shorter (5–12 days) in comparison with lowland populations (about 30 days). Therefore, newly released individuals encounter a number of difficulties that may lead to reduced adaptability, frostbite, higher mortality etc., in addition to the stress related to the transfer itself. The active period in the mountains is shorter and the EGS have less time to accumulate body fat and build suitable burrows.

Attempts have been made to solve these issues by moving the animals earlier (in April, May, June) and avoiding transferring pregnant or nursing females and young animals. This strategy has shown initial positive effects in Vitosha NP and Bulgarka NP (Koshev, Arangelov – unpubl. records).

Climate conditions in the year of release

The weather conditions in two years (2014 and 2018) of the period reviewed (2010–2018) were particularly extreme. All the translocations, with the exception of that in Kotlenska planina, involved activities in one of these years.

For the whole territory of Bulgaria, the monthly precipitation amount in 2014 was above the normal in April, June and September. For the whole period of April to July, rainfall was above normal. Increased rainfall leads to a positive accumulation of water in the soil. In the same two years, temperatures were exceptionally high in February and the temperature anomaly in the spring and summer was negative. The 2014 winter season started with high temperatures, which helped to make the snow melt faster while the summer was relatively cold, which is a prerequisite for the slower soil drying and its several months of water saturation (Tepsizova 2017). Similar climatic conditions were observed in the summer of 2018 (NIMH-BAS 2018).

In 2014, extreme rainfall was recorded in the area of Vrachanski Balkan NP on 15–20.04.2014, 30–31.07.2014 and 2–7.09.2014. There was local intense rainfall and high soil moisture, which led to flooding in surrounding villages. In the area of Bulgarka NP, extreme rainfall and flooding was registered on 28–31.05.2014 (Stoycheva et al. 2015).

The abundant rains flood the holes, cause drowning and hypothermia. EGS are scattering mammals that spend much of their lives underground, making them particularly vulnerable. Numerous cases of mass death caused by floods and rains (Hoffmann et al. 2003) resulted in translocation actions (Lobbová et al. 2012). In Bulgaria, Stoyanov (2001) reported hundreds of drowned EGS in the Ponor Mountains at 1200 m a.s.l. (Western Stara Planina) after heavy rains. This could be especially valid after translocation in artificial burrows which initially have simple structure and cannot yet provide the protection of the animal-made ones. Increased rainfall also causes higher grass cover, which has negative impact for EGS.

Independent resettlement away from the release site

It is notable that, on 4 occasions, the animals settled several hundred metres away from the release site. This has been observed during the translocations in Central Europe, but the distances observed in Vrachanski Balkan NP and Bulgarka NP are the largest reported (Table 11).

Table 11. Distance (m) from the release site to the resettlement point reported in literature compared with the data of the current article.

Translocation	State	Distance from the release site (m)	Reference
Pod Okrúhlou skalou (Tisovec)	Slovakia	30	Lobbová, Hapl 2014
Ponitrie Protected Landscape Area	Slovakia	200	Matějů et al. 2010, 2012
Malé Karpaty Protected Landscape Area	Slovakia	250	Matějů et al. 2010, 2012
Malé Karpaty Protected Landscape Area	Slovakia	100	Matějů et al. 2010, 2012
Malé Karpaty Protected Landscape Area	Slovakia	200	Matějů et al. 2010, 2012
Slavkovský les Protected Landscape Area	Czech Republic	350	Matějů et al. 2010, 2012
Kotlenska Planina Natura 2000 Site	Bulgaria	500	Stefanov et al. 2016
Kotlenska Planina Natura 2000 Site	Bulgaria	100	Stefanov et al. 2016
Vitosha Nature Park	Bulgaria	150	current study
Vrachanski Balkan Nature Park	Bulgaria	290	current study
Vrachanski Balkan Nature Park	Bulgaria	600	current study
Bulgarka Nature Park	Bulgaria	720	current study
Average		290.8	
Min		30	
Max		720	
n		12	

This could be explained by the microhabitat conditions where the slope, the wind, the soil type and depth and the vegetation vary widely and are hard to assess at first sight. In that case, the exact data about the species' occurrence in the past could be precious, but was not present in the investigated cases. This resettlement indicates that the animals' perception of habitat suitability could differ from conservationists' opinion and project restrictions. Thus, the possibility for moving and the conditions in the surrounding territories (300 to 720 m around) should be considered during the process for selection of the release area. For example, the resettlement in the Bulgarka NP led to the current colony being situated outside the park's boundaries and is now threatened by wind power plants. In Vrachanski Balkan NP, after its movement, the colony is partly on private land so its protection is not guaranteed.

Conclusions

1. Between 2010 and 2018, 8 translocations of more than 1730 individuals were performed in Bulgaria for different purposes: 4 for reinforcement of old colonies, 3 for reintroductions and 1 for introduction. Currently two translocations are ongoing.
2. Five or 83% of the translocations were successful, but two had a critically low number of established individuals – Vrachanski Balkan NP and Bulgarka NP.
3. Six translocations used soft release methods and two translocations – hard release. In six cases, released individuals settled from 100 to 720 m away from the place of release, which imposes management and protection of larger areas.
4. In seven cases, there was a difference in altitude between the donor colony and the release site of 470 to 1320 m a.s.l., which could have a hindering effect on the adaptation of animals due to the specific conditions in the mountains and the preparation for hibernation.
5. The main reasons for failure were probably related to poorly selected and maintained habitats, as well as poor climatic conditions (rainy and cold weather).
6. European funds are of critical importance with only two translocations funded by other sources.

Recommendations for the future

1. The IUCN / SSC (2013) recommendations for translocation of individuals and the recommendations of Matějů et al. (2010, 2012) should be followed, for example to undertake a preliminary study to check whether the donor colonies and the translocation sites meet the requirements.
2. Soft release methods should be used (using enclosures, guarding, artificial holes, additional feeding).
3. The number of individuals should be consistent with the initial success [the three phases of Letty et al. (2003), Teixeira et al. (2007), Mateju et al. (2012)], post-

release-monitoring to be a mandatory part of projects. Rules for interruption of the translocation should be adopted when it fails – for example no surviving individuals and breeding.
4. Maintaining the habitat and even the populations, if needed, should continue after the project's end. The funding sources should be diverse so that there is flexibility in the implementation of the activities. Translocation activities should not be undertaken during a given year in the case of unacceptable factors, such as bad meteorological conditions.
5. Larger areas around the translocation sites should be designed and maintained taking into account the migration (movement) of the individuals described in the current article. The releasing sites should be far from the protected areas' edges so that individuals remain under protection despite their dispersal.
6. Moving individuals from low to high altitudes should be avoided, if not necessary. If case it is inevitable, undertake translocation activities only under appropriate meteorological conditions and in a season consistent with the active cycle of individuals, carefully selecting the age, sex and physiological state of the animals.
7. A scientific database should be created hosting detailed information about past, current and future activities related to translocation of EGS (including reintroduction, restocking, translocation, repatriation, restoration, recolonisation etc.) aiming for standardisation and harmonisation of the activities. This need is due to the constantly increasing number of translocations of EGS that, if not planned carefully, could hinder the unique genetic diversity of the species in Bulgaria.

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Supplementary material 1

Translocations of European ground squirrel (*Spermophilus citellus*) along altitudinal gradient in Bulgaria – an overview

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RESEARCH ARTICLE

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Two alternative evaluation metrics to replace the true skill statistic in the assessment of species distribution models

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Abstract
Model evaluation metrics play a critical role in the selection of adequate species distribution models for conservation and for any application of species distribution modelling (SDM) in general. The responses of these metrics to modelling conditions, however, are rarely taken into account. This leads to inadequate model selection, downstream analyses and uniformed decisions. To aid modellers in critically assessing modelling conditions when choosing and interpreting model evaluation metrics, we analysed the responses of the True Skill Statistic (TSS) under a variety of presence-background modelling conditions using purely theoretical scenarios. We then compared these responses with those of two evaluation metrics commonly applied in the field of meteorology which have potential for use in SDM: the Odds Ratio Skill Score (ORSS) and the Symmetric Extremal Dependence Index (SEDI). We demonstrate that (1) large cell number totals in the confusion matrix, which is strongly biased towards 'true' absences in presence-background SDM and (2) low prevalence both compromise model evaluation with TSS. This is since (1) TSS fails to differentiate useful from random models at extreme prevalence levels if the confusion matrix cell number total exceeds ~30,000 cells and (2) TSS converges to hit rate (sensitivity) when prevalence is lower than ~2.5%. We conclude that SEDI is optimal for most presence-background SDM initiatives. Further, ORSS may provide a better alternative if absence data are available or if equal error weighting is strictly required.

Keywords
Species distribution modelling, True Skill Statistic, evaluation, presence-background

Introduction

Species Distribution Modelling (SDM) relates independent environmental variables to species occurrence data and, in turn, predicts a dependent variable such as probability or the relative likelihood of occurrence (Guisan and Zimmermann 2000; Peterson 2001; Guillera-Arroita et al. 2015). Even though SDM predictions mostly range from zero to one, SDM predictions are often discretised into binary presence-absence maps (i.e. comprising only zeros and ones) used to evaluate wildlife management options, to identify appropriate conservation translocation sites and to evaluate model performance (Willis et al. 2009; Fordham et al. 2012; Liu et al. 2013) with confusion matrix-based performance metrics. These confusion matrices (Table 1) summarise the correspondence between predictions and observations, by providing the counts of (a) true presences, (b) false presences (commissions), (c) false absences (omissions) and (d) true absences. However, inherent asymmetric uncertainty levels, particularly for mobile species, between the observed and predicted presence and absence classes, can complicate such comparisons.

Observed absences in presence-absence datasets can be either true, i.e. the species does not occur, or false, i.e. the species does occur but remains undetected (Martin et al. 2005). 'Observed true absences' result from biological processes, such as intolerance to local conditions, competition (Hardin 1960; Leathwick and Austin 2001), general rarity (Gaston 1994), meta-population dynamics, i.e. perpetuating series of local extinctions and recolonisations (Hanski 1998) or other biotic interactions (Bascompte 2009; Wisz et al. 2013; Bulleri et al. 2016). 'Observed false absences', on the other hand, are artefactual in nature, resulting from insufficient monitoring relative to species movement (Tyre et al. 2003) or imperfect detection (MacKenzie et al. 2002). Whereas both true and false absences can lead to 'zero-inflated' datasets (Heilbron 1994) that violate statistical assumptions, the latter are also a source of uncertainty in parameter estimates as artefactual signals (e.g. sampling bias, probability of detection) confounding estimates of probability of occurrence (MacKenzie et al. 2002).

The capability to distinguish observed true and false absences may also dictate the applicability of model evaluation metrics, many of which differ in weights assigned to each of the four categories in the confusion matrix (Table 1). For example, when the observed true and false absences are indistinguishable, omission errors, i.e. excluding known presences (type c in Table 1), may be more problematic than commission errors, i.e. including absences of relatively unknown certainty (type b in Table 1). Although seldom investigated, differential error weighting may also be desirable when considering specific biological questions that relate to population dynamics, such as population

Table 1. Confusion matrix with cell designation as defined by the agreement of predictions (rows) and observations (columns).

Predicted	**Observed**	
	Present	***Absent***
Present	True (a)	False (b)
Absent	False (c)	True (d)

sources versus population sinks (Guisan and Thuiller 2005), or biogeographical processes like invasion source versus colonisation fronts (Ward 2007), particularly when using high confidence absence data.

While it is possible to estimate or model the probability of detection by repeated surveys and, hence, to discern observed true and false absences (MacKenzie and Royle 2005; Guillera-Arroita et al. 2015), it is often logistically impractical. Moreover, presences often include numerous opportunistic observations, whereas absences generally go unrecorded and suffer from greater uncertainty. Therefore, contrasting presences with background points, i.e. pseudo-absences, which are randomly sampled from within the study area (Iturbide et al. 2015), are the only viable option for many situations (Elith and Leathwick 2009). In this context, it has to be stressed that the default number of background points (10,000) in MAXENT, a broadly adopted maximum entropy machine-learning SDM algorithm (Phillips et al. 2004, 2017), is often insufficient and – unless increased sufficiently to capture the range of existing environmental conditions – equates to modelling at lower spatial resolution (Renner and Warton 2013).

When using presence-background occurrence data, the measured performance of modelling techniques, such as Generalised Linear Models (Nelder and Wedderburn 1972) or MAXENT (Phillips et al. 2004, 2017; Philipps and Dudík 2008), generally has a positive relationship with both the number and geographic extent of random background points relative to presence points (Philipps and Dudík 2008; Barbet-Massin et al. 2012). Model performance then, should be interpreted as the result of a complex interplay of artifacts (stemming from data or methods) and biological causes, since performance depends on modeller decisions, data availability and the underlying distribution of species based on dispersal from historical distributions (Barve et al. 2011). For instance, specialist species are, by definition, confined to narrower conditions within a broad landscape, than are generalist species. Unfortunately, the relative ease of characterising narrowly confined species (Jiménez-Valverde et al. 2008) and consequent high model performance scores, may not be due to biological causes, such as ecological specialisation, i.e. dependence on specific environmental conditions. Instead, good model performance may result from model overfitting as species presences simply coincide with specific conditions, combinations or transformations of environmental variables in overly complex models (Merow et al. 2014; Fourcade et al. 2018). Therefore, in order to interpret and compare the performance of models in a meaningful way, modellers must move past simple evaluation metrics (e.g. sensitivity and specificity) and consider confounding effects on model performance. Bias, cell number totals in the confusion matrix and prevalence should be assessed and their effect on model performance minimised by, for example, choosing less susceptible evaluation metrics.

The issue of confounding effects on model performance is particularly important in conservation planning and reserve area selection, since both regularly take SDM predictions into account (Margules and Pressey 2000; Lin et al. 2014; Guillera-Arroita et al. 2015). Unless confounding effects are considered during model evaluation, however, any application of SDM is potentially affected, including estimates of species richness and community composition (Gioia and Pigott 2000; Pineda and Lobo 2009;

Thuiller et al. 2015) or hindcasting past distributions (Franklin et al. 2015). Unfortunately, modellers often assume that model performance constitutes the most objective, if not the best, evidence for model legitimacy, representing not only prediction accuracy, but also underlying biological processes (Jiménez-Valverde et al. 2008). This assumption can be problematic as all evaluation metrics differentially react to modelling conditions. That is, two different evaluation metrics may represent true model performance better under varying circumstances (Lawson et al. 2014). More generally, evaluation metrics are functions of both the model prediction accuracy and the modelling conditions (Woodcock 1976). This means that beyond representing prediction accuracy, model evaluation metrics can conflate both artefactual (e.g. differences in sampling regimes, study extent, resolution, model overfitting) and biological (e.g. degree of species specialisation, population dynamics, autecology) signals.

In summary, the interpretability of measured model performance garnered from presence-background data is limited (Hirzel et al. 2006). The extent to which modelled predictions do reflect the posited goal of most initiatives – namely, identifying the underlying biological processes that dictate species distributions, is less certain. However, any measure of model performance for any given model is the result of a four-part process that includes data collection, model training, threshold setting and the selection of model evaluation metrics. Here, we focus on the fourth part only, the selection of appropriate model evaluation metrics under specific modelling conditions commonly encountered in presence-background SDM initiatives. We also provide some insights into how measured model performance may have resulted from biological signals or artifacts.

In this paper, we use purely theoretical scenarios to compare the responses of three evaluation metrics, the True Skill statistic (TSS; Allouche et al. 2006), the Odds Ratio Skill Score (ORSS; Stephenson 2000) and the Symmetric Extremal Dependence Index (SEDI; Ferro and Stephenson 2011), to three confounding factors on model performance, the cell number totals in the confusion matrix (typically dominated by background points total and dependent on the size of the study area and resolution), bias and prevalence. To our knowledge, the latter two evaluation metrics have not been used in SDM before. We also contextualise our results in terms of SDM initiatives with respect to how well specific evaluation metrics reflect biological signals versus artifacts in particular modelling conditions and discuss this with reference to recently raised concerns about the use of TSS in SDM reported in literature. Additionally, R code is provided for ORSS and SEDI computations.

Materials and methods

Comparison of evaluation metrics

Detailed definitions of some of the more technical terms and a comparison of the mathematical properties of the analysed evaluation metrics are found in Table 2.

Table 2. Comparison of selected properties of binary evaluation metrics compared in this article. 'Consistent at maxSSS' refers to the threshold maximising the sum of sensitivity and specificity (SSS) suggested by Liu et al. (2013) which is generally recommended in literature. Please refer to Somodi et al. (2017) for prevalence effects on maxSSS itself. "+" and "–" indicate that the evaluation metric features or lacks a property, respectively.

Property	Definition	TSS	ORSS	SEDI
Asymptotically equitable	Random predictions yield a score of zero	+	+	+
Prevalence independent	Same result when prevalence changes if both H and F remain unchanged	+	+	+
Complement symmetric	Same result when switching a and c with d and b	+	+	+
Consistent at maxSSS	Maximising SSS maximises the evaluation metric	+	–	–
Fixed range	Minimum/maximum possible values do not depend on prevalence	+	+	+
Hard to hedge	Monotonic increase with H and monotonic decrease with F	+	+	+
Non-degenerate	Meaningful results when prevalence approaches zero	–	–	+
Regular	Isopleths of the evaluation metric pass through the origin	–	+	+
Transpose symmetric	Same result when swapping b and c	–	+	–

Below are the equations of four simple evaluation metrics (variables a, b, c and d according to Table 1). First, the hit rate (H, also termed sensitivity) measures the ratio of true presences to the sum of true presences and omission errors while completely ignoring commission errors and the number of true absences. Second, the false positive rate (F), equal to 1–*specificity*, measures the ratio of commission errors to the sum of commission errors and true absences. Third, bias (also termed bias score or frequency bias) measures the ratio of commissions to omissions and helps to identify over-/under-predicting models. And fourth, prevalence (also termed base rate) measures the ratio of presences (both predicted and omitted) to all cells and hence expresses how common within the study area a species is, according to the available data.

$$H = a/(a + c) \tag{1}$$

$$F = b/(b + d) \tag{2}$$

$$bias = (a + b)/(a + c) \tag{3}$$

$$prevalence = (a + c)/(a + b + c + d) \tag{4}$$

TSS measures the difference between H and F and was first developed as Peirce's skill score in meteorology (Peirce 1884). It was later introduced to other fields, including the field of SDM, where it replaced kappa (Cohen 1960) and its strong unimodal response to prevalence (Allouche et al. 2006).

$$TSS = H - F \tag{5}$$

ORSS measures skill compared to a random prediction, is a synonym of Yule's Q (1900) and was introduced to meteorology by Stephenson (2000). ORSS provides equal error weighting but rapidly converges to one even for imperfect predictions

(Woodcock 1976) and hence, requires significance testing to quantify skill and to discern real skill from chance (Stephenson 2000).

$$ORSS = (ad - bc)/(ad + bc) \tag{6}$$

Ferro and Stephenson (2011) developed the Symmetric Extremal Dependence Index (SEDI) as an improvement on earlier work by Stephenson et al. (2008) and Hogan et al. (2009). SEDI featured greatly reduced sensitivity to prevalence, while retaining most beneficial properties of its predecessors (Ferro and Stephenson 2011). This includes asymptotic equitability, i.e. the ability to distinguish random and skilled predictions at smaller than infinite sample sizes (Hogan et al. 2010). Yet, SEDI is not applicable if any of the four cells in the confusion matrix equals zero (Ferro and Stephenson 2011) since *log*(0) yields infinity. Overfitted or misspecified models in these instances, however, can still be interpreted by adding an infinitely small number to those cells containing zeros. Our implementation of SEDI (see Supplementary Information for repository link) also issues a character string indicating if such approximations were used and how to best interpret the result.

$$SEDI = \frac{\log(F) - \log(H) - \log(1-F) + \log(1-H)}{\log(F) + \log(H) + \log(1-F) + \log(1-H)} \tag{7}$$

Theoretical scenarios

Using two types of extreme prediction settings ("Optimistic" and "Pessimistic") and two types of typical species prevalence settings ("Differential Bias" and "Changing Bias"), we investigated the response of TSS, ORSS and SEDI to increasing cell number total in the confusion matrix, varying commission error rates, omission error rates and species prevalence. These settings were each divided into two theoretical scenarios: "Incorrectly Optimistic" (IO) and "Correctly Optimistic" (CO), "Correctly Pessimistic" (CP) and "Incorrectly Pessimistic" (IP), "Commission Bias" (CB) and "Omission Bias" (OB) and "Low Commission Rate" (LC) and "High Commission Rate" (HC). For each of the eight scenarios (Table 3), we prepared twenty cases, i.e. confusion matrices, to be evaluated (Table 3). In "Changing Bias" scenarios (LC and HC), the term 'logistic' was used to describe model fit which improved with background point totals and cell number totals but plateaued below perfect fit.

Although not mutually exclusive, each scenario is designed to reflect signals that could have arisen from biological signals or artifacts, thereby revealing how susceptible model evaluation metrics are to conflating the two. Below, we briefly describe all scenarios and Fig. 1 visualises selected example cases. Across the theoretical cases in the different scenarios, the cell number total ($a + b + c + d$) approximately ranges from 1240 to 42560 with increasingly large step size (Table 3) to allow the analysis of model evaluation across a large range of modelling conditions without requiring an overly large number of cases. The R code to reproduce our analysis is available upon request.

Table 3. Description of the theoretical scenarios, where IO, CO, CP, IP, CB, OB, LC and HC are abbreviations for scenarios "Incorrectly Optimistic", "Correctly Optimistic", "Correctly Pessimistic", "Commission Bias", "Omission Bias", "Low Commission Rate" and "High Commission Rate". True positives, false positives, false negatives and true negatives are represented by a, b, c, and d, respectively. Total lists the sum of all four cells in the confusion matrix. The formulations are provided in pseudo-R-code, i.e. square brackets ("[" and "]") indicate vectors and colons (":") indicate a series. For example, "[x:y]" represents a vector of integers ranging from x to y. "…" are used to indicate repeating the same number, and n is the case number.

Scenario	a	b	c	d	Total
IO	increasing: 0.005*[1:n]$^{1.25}$*1000+248	increasing: [1:n]$^{1.25}$*1000+248-[a]	constant: [1…1]	constant: [1…1]	min: 1250 max: 42545
CO	increasing: 0.995*[1:n]$^{1.25}$*1000+248	increasing: [1:n]$^{1.25}$*1000+248-[a]	constant: [1…1]	constant: [1…1]	min: 1250 max: 42545
CP	constant: [1…1]	constant: [1…1]	increasing: [1:n]$^{1.25}$*1000 +248-[a]	increasing: 0.995*[1:n]$^{1.25}$ *1000+248	min: 1250 max: 42545
IP	constant: [1…1]	constant: [1…1]	increasing: [1:n]$^{1.25}$*1000 +248-[a]	increasing: 0.005*[1:n]$^{1.25}$ *1000+248	min: 1250 max: 42545
CB	constant: [200…200]	constant: [30…30]	constant: [20…20]	increasing: [1:n]$^{1.25}$*1000	min: 1250 max: 42545
OB	constant: [200…200]	constant: [20…20]	constant: [30…30]	increasing: [1:n]$^{1.25}$*1000	min: 1250 max: 42545
LC	'logistic': [175:189,190…190]	increasing: [c_n:c_1]	decreasing: 200-[a]	increasing: [1:n]$^{1.25}$*1000	min: 1210 max: 42520
HC	'logistic': [175:189,190…190]	increasing: 3*[c_n:c_1]	decreasing: 200-[a]	increasing: [1:n]$^{1.25}$*1000	min: 1230 max: 42570

Scenarios IO, CO, CP and IP were designed to demonstrate how evaluation metrics at essentially constant extreme levels of prevalence react to an increasing cell number total in the confusion matrix. The biological component of these scenarios is analogous to specialist or generalist species that have a constant prevalence of 0.5% or 99.5% of the study area. The artefactual component is related to the implications of study area increases for the number of background points and total number of cells and their effect on the calculation of evaluation metrics. Scenario IO was characterised by large numbers of commission errors as it evaluated an extreme incorrectly optimistic modelling prediction (over-prediction) when true species prevalence is equal to 0.5%, reflecting extreme specialisation or rarity and under increasing background size. Scenario CO was identical to scenario IO in its extreme prediction. However, as true species prevalence was equal to 99.5% (reflecting extremely low specialisation), it no longer resembled an over-prediction and was consequently dominated by true presences. Scenario CP evaluated an extreme correctly pessimistic prediction when true species prevalence was equal to 0.5%, reflecting a high degree of ecological specialisation and species presence was only predicted for a small proportion of the study area, under increasing background size. This scenario was characterised by large numbers of true absences. Scenario IP was identical to scenario CP in its extreme prediction but dominated by false

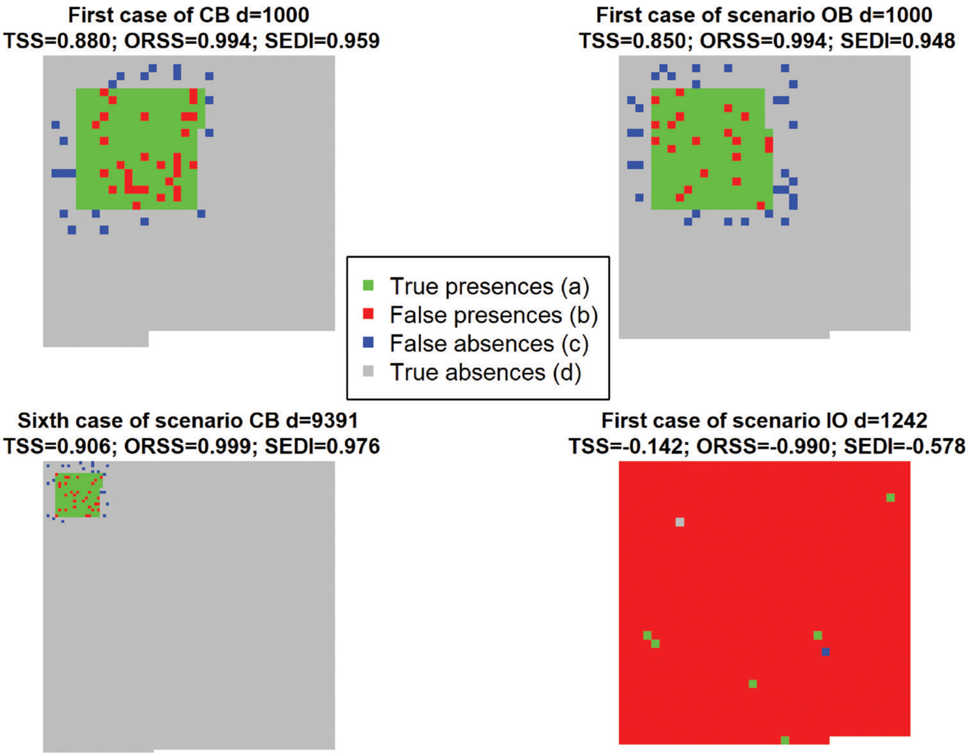

Figure 1. Potential spatial distributions of confusion matrix categories corresponding to values of the True Skill Statistic (TSS), the Odds Ratio Skill Score (ORSS) and the Symmetric Extremal Dependence Score (SEDI) for selected scenario cases.

absences since true species prevalence was now equal to 99.5%, turning it into a gross under-prediction.

Scenarios CB and OB were designed to reveal the effect of bias on evaluation metrics under decreasing prevalence (~17% to ~0.5%) as the study area increased. In these scenarios, evaluation metrics should consistently penalise model predictions according to the degree of their bias, across the whole range of prevalence. Scenario CB was more optimistic (more commission errors and more predicted presences) than scenario OB (more omissions and fewer predicted presences). Therefore, the two scenarios together can be seen as a test of transpose symmetry.

Scenarios LC and HC examined the response of evaluation metrics to changes in bias while model fit (i.e. the number of true positives) and the total number of cells increased as prevalence decreased (~17% to ~0.5%). More specifically, the number of observations was held constant in both scenarios, while the numbers of true positives and omissions increased and decreased, respectively. However, at the same time, commission errors became more frequent. In other words, the bias of the model changed together with prevalence and the size of the study area. Scenarios LC and HC differed only in their rate of commission errors which was three times higher in scenario HC

Table 4. Evaluation scores (rounded to four digits) for all evaluation measures metrics considered across all scenarios. IO, CO, CP, IP, CB, OB, LC and HC are abbreviations for scenarios "Incorrectly Optimistic", "Correctly Optimistic", "Correctly Pessimistic", "Commission Bias", "Omission Bias", "Low Commission Rate" and "High Commission Rate". Total lists the sum of all four cells in the confusion matrix. H, F, TSS, ORSS and SEDI list evaluation metric values for hit rate, false positive rate, True Skill Statistic, Odds Ratio Skill Score and Symmetric Extremal Dependence Score, respectively. Cases #6 and #7 closely resemble typical presence-background modelling conditions in MAXENT.

Cell Total	Scenario	H	F	TSS	ORSS	SEDI
ca. 9,000 – 10,000 (Case #6)	IO	0.9796	0.9999	-0.0203	-0.9900	-0.4050
	CO	0.9999	0.9796	0.020	0.9900	0.4050
	CP	0.0204	0.0001	0.0203	0.9900	0.4050
	IP	0.0001	0.0204	-0.0203	-0.9900	-0.4050
	CB	0.9091	0.0032	0.9059	0.9994	0.9761
	OB	0.8696	0.0021	0.8674	0.9994	0.9659
	LC	0.9000	0.0012	0.8988	0.9997	0.9767
	HC	0.9000	0.0035	0.8965	0.9992	0.9730
ca. 11,000 – 12,000 (Case #7)	IO	0.9831	0.9999	-0.0169	-0.9900	-0.3937
	CO	0.9999	0.9831	0.0169	0.9900	0.3937
	CP	0.0169	0.0001	0.0169	0.9900	0.3937
	IP	0.0001	0.0169	-0.0169	-0.9900	-0.3937
	CB	0.9091	0.0026	0.9065	0.9995	0.9768
	OB	0.8696	0.0018	0.8678	0.9995	0.9668
	LC	0.9050	0.0011	0.9039	0.9998	0.9783
	HC	0.9050	0.0032	0.9018	0.9993	0.9749

than in scenario LC. The biological component could represent increasing specialisation of a given species as the study extent increases; whereas the artefactual component could represent resultant increases in model fit as increasing specialisation makes for easier characterisation (Jiménez-Valverde et al. 2008).

Results

Our results are summarised in Table 4 and presented in Fig. 2. In addition, we provide Fig. 3 which depicts the proportional difference between scores of SEDI and TSS, ORSS and TSS and SEDI and ORSS, for scenarios LC and HC, i.e. the sensitivity to changes in commission errors under decreasing prevalence.

Optimistic and pessimistic scenarios

TSS shows a strong response to increased study area size and, hence, confusion matrix cell number totals and rapidly converges to zero, rendering indifferent useful and random models beyond cell number totals in the confusion matrix of approximately

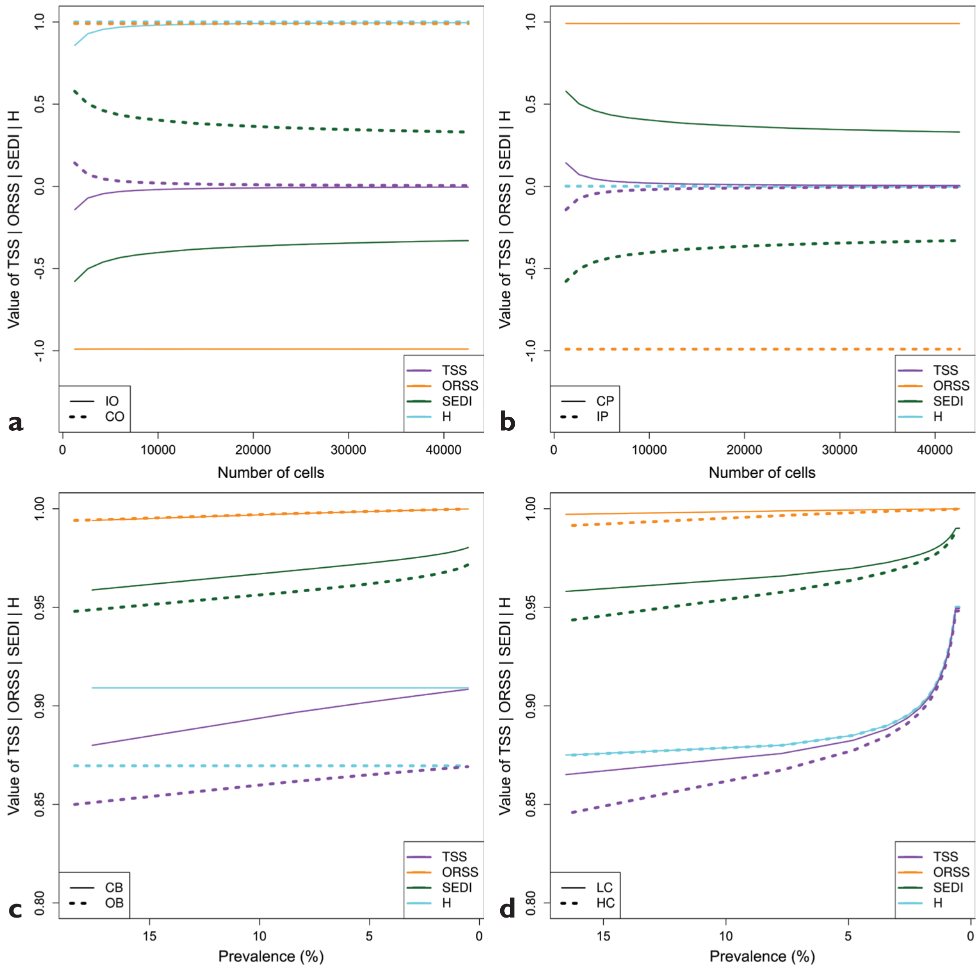

Figure 2. Plots of the values of hit rate (H), the True Skill Statistic (TSS), the Odds Ratio Skill Score (ORSS) and the Symmetric Extremal Dependence Score (SEDI) for all eight scenarios. Panels **a, b, c, d** display scenarios "Incorrectly Optimistic" and "Correctly Optimistic", "Correctly Pessimistic" and "Incorrectly Pessimistic", "Commission Bias" and "Omission Bias" and "Low Commission Rate" and "High Commission Rate". In panels a and b, the x-axis denotes the log of the total number of cells, i.e. the size of the study area, whereas in panels c and d, the x-axis denotes prevalence (%).

30,000 cells. SEDI shows only a moderate response and converges much later to zero. Of note, H completely fails in this respect since both incorrectly optimistic predictions (IO) and correctly pessimistic predictions (CP) converge to one and zero, respectively and yield scores very similar to those of their correct (CO) and incorrect (IP) counterparts. Finally, SEDI has stronger discriminatory power than TSS at intermediate study areas yet, only ORSS is expected to correctly assess model performance as study area size converges to infinity (Fig. 2a, b).

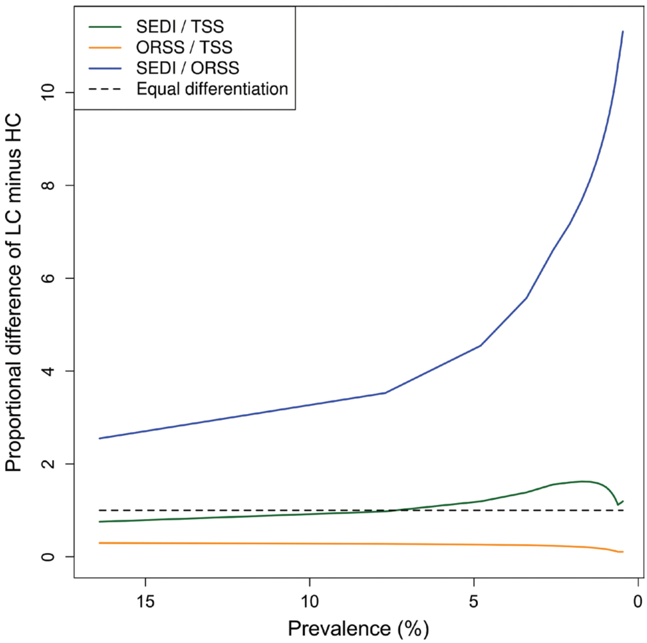

Figure 3. Proportional difference between scenarios "Low Commission Rate" (LC) and "High Commission Rate" (HC) for (in dark green) the Symmetric Extremal Dependence Score (SEDI) vs. the True Skill Statistic (TSS), (in orange) the Odds Ratio Skill Score (ORSS) vs. TSS and (in blue) SEDI vs. ORSS. The black, horizontal, dashed line represents equal differentiation. As there are slight differences in prevalence between scenarios LC and HC, the x-axis shows the mean prevalence for given cases across both scenarios.

Differential bias scenarios and changing bias scenarios

TSS quickly converges with H and always favours over-predictions to under-predictions. However, the degree to how much over-predictions are favoured increases as prevalence decreases. Although SEDI also favours over-predictions, it does so to a much smaller degree and is not significantly affected by prevalence. Just as in scenarios CO and CP, ORSS rapidly converges to one (Fig. 2c). Under increasing study area and background point totals and so decreasing prevalence, TSS converges quickly with H for the most realistic scenarios (LC and HC in Fig. 2d) and models that predict increasing amounts of both true presences and commission errors, as omission errors, decrease. At higher levels of prevalence, TSS can still discern the quality of models differing only in their rate of commission errors, but once prevalence falls below approximately 2.5%, their difference becomes indistinguishable. SEDI can assess the quality of models differing only in their rate of commission errors as prevalence decreases to almost zero. As in previous scenarios, since ORSS rapidly converges to one, model scores differing only in their rate of commission errors become indistinguishable even faster than when using TSS. In addition, the proportional difference between LC and HC SEDI scores and TSS scores are lower at the start though increase as prevalence decreases (Fig. 3). This indicates that, although TSS identifies differences in commission error levels at high prevalence (greater than approximately 7%) better than SEDI, the reverse is true at the low prevalence levels typically encountered in presence-background modelling.

Discussion

Using eight theoretical scenarios, we have shown that TSS, ORSS and SEDI, as well as their underlying evaluation measures (H and F, see F in Table 2), show distinct responses to: 1) increasing size of the study area and, hence, growing numbers of background points, even when prevalence is kept constant (scenarios IO, CO, CP and IP), 2) to the direction of bias as prevalence decreases and the extent of the study area and cell number totals increase (scenarios CB and OB) and 3) to changes in bias as prevalence decreases and the extent of the study area and cell number totals increase (scenarios LC and HC).

Our analysis confirmed a very problematic property of TSS. That is, a very large number found in any of the four cells of the confusion matrix (Table 1) leads to the marginalisation of the other entry in the same column (Stephenson 2000). This means that, when assessing rare events, such as rare species presence, TSS quickly converges to H (Doswell et al. 1990). Less apparent responses of TSS to prevalence have also been discussed in the field of SDM, for instance, by Somodi et al. (2017) who found that small sample size exacerbates the effects of prevalence on TSS. We also evaluated two alternative evaluation metrics from the field of meteorology, SEDI and ORSS. The former appears to be ideal for typical low prevalence presence-background SDM conditions, whereas the latter may be useful for high confidence presence-absence data or if strictly equal error weighting is required.

Optimistic and pessimistic scenarios

By grossly over- or under-predicting the distribution of a hypothetical target species, we observed the response of evaluation metrics to extreme biases in less realistic scenarios. These extreme scenarios, however, have also shown that discernment of strongly and weakly performing models greatly differs amongst evaluation metrics and modelling conditions. While these scenario results support the use of ORSS for large study extents, because of its rapid convergence to one, even for imperfect predictions (Woodcock 1976), significance testing is required in order to determine the quality of and to allow comparisons across models (Stephenson 2000). Therefore, SEDI is preferable as it allows direct assessments of model quality across a much larger spectrum of study extents. Further, SEDI assessments are made with higher discriminatory power than TSS, which rapidly converges to zero for extreme predictions.

Differential bias and changing bias scenarios

These scenarios have been designed to reflect common modelling conditions in order to observe the response of evaluation measures to differential (CB and OB) and changing (LC and HC) biases, under decreasing prevalence as the size of the study area and, hence, the number of background points increased. Analysis of these scenarios revealed very distinct responses to the differing modelling conditions. Results for scenarios CB and OB and LC and HC suggest the use of SEDI since: 1) TSS encourages over-predictions due to its strongly biased treatment of errors which increases as prevalence decreases; 2) TSS quickly loses the discriminatory power to differentiate between models, differing only in their commission rate as it always converges to H; and 3) ORSS converges to one so rapidly (Stephenson 2000) that such differences vanish at even higher prevalence levels than when using TSS.

Modelling conditions and research questions

Our analysis reaffirms the importance of selecting model evaluation metrics corresponding with modelling questions and conditions (Woodcock 1976). Important modelling conditions include the extent of the study area (Termansen et al. 2006), prevalence (Doswell et al. 1990; Stephenson 2000; Somodi et al. 2017; Leroy et al. 2018) and the relative severity of model error types due to varying degrees of biological and artefactual causes. Important biological factors, related to the extent of the study area and species prevalence, include the degree of specialisation within the given landscape (Jiménez-Valverde et al. 2008); population sinks (Guisan and Thuiller 2005); and species equilibrium (Václavík and Meentemeyer 2012). This latter factor is also important in the context of invasive species (Ward 2007). Artifacts can originate from the cell number total in the confusion matrix which is largely driven by background points and hence dependent on modelling resolution (Seo et al. 2008) and study extent. Artefactual signals can also be caused by prevalence and its interactions with sample-size (Somodi et al. 2017) and the degree of confidence in absence data (Leroy et al. 2018) which is influenced by species mobility (Jaberg and Guisan 2001) and a multitude of other factors.

Do commission errors matter?

Our results suggested a limited capacity of TSS to provide consistent performance comparisons across varying modelling conditions. This is worrying because TSS may yield misleading estimates of model fidelity, which can lead to the selection of inadequate models. Although it may be tempting to assume that researchers would recognise anomalous conditions where TSS scores are misleading (such as those presented here), this is not necessarily the case – as demonstrated by the broad and seemingly uncritical application of TSS in presence-background SDM over the last decade. Complications, owing to the relative inability of TSS to provide information on commission errors as prevalence approaches zero, are more nuanced. Ultimately, such complications are only problematic in as much as commission errors matter, which depends on *bias* > 1, the question, the available data and the biology of the species. In fact, it is widely acknowledged that even absence data from professional surveys have greater degrees of both sampling and ecological uncertainty than presence data (MacKenzie et al. 2017). Therefore, one could argue that errors of omission, i.e. predicting absences where there are species observations, are far more grievous than commission errors in model evaluation, particularly when using presence-background data. This is because commission errors can be relatively obscure and should be given less weight than omission errors in model evaluation (Braunisch and Suchant 2010; Liu et al. 2013). Taking the above reasoning to its logical extreme then, commission errors are irrelevant when considering pseudo-absence data.

While the above reasoning is persuasive, simply ignoring commission errors in presence background data by limiting evaluation to H, is not a viable option under all but a small subset of questions, modelling conditions and biological assumptions. More specifically, doing so would be incongruent with the biological circumstances, sampling realities and the intents of most modelling initiatives. Further, evaluation scores would become more vulnerable to artificial inflation. From a biological perspective, model evaluation metrics that ignore commission errors are equivalent to assuming that all background points are locations where the species is present but unobserved. That is, assuming that observed presence locations may represent the subset of relative high use or occupied conditions within local settings (Elith et al. 2011). While this may be a good assumption for situations where habitat use of a generalist species is considered (particularly if survey effort has been uniform across the entire study area), it is unrealistic and impractical for many modelling initiatives that attempt to characterise either the fundamental or realised niche (Elith and Leathwick 2009), based on incomplete sampling regimes.

Furthermore, even when the above biological assumptions and survey prerequisites are valid, explicitly choosing to ignore commission errors further assumes that unobserved locations are irrelevant—an assumption that is seldom the case since these locations may correspond to low population density areas (Guisan and Thuiller 2005). Models (and subsequent thresholds) that commit lower numbers of commission errors may better differentiate between high and low population density areas, a highly desirable characteristic for assessing population status (Tôrres et al. 2012). Of course in rarer situations, strictly considering omission errors in model evaluation may be desirable such as when assessing the impacts of decisions on vulnerable species (Karl et al. 2000). In addition, ignoring commission errors can lead to unintended consequences as seen in Fig. 2a where H is deceptively high for incorrectly optimistic outputs, i.e. over-predicting models. Although commission errors should be weighted less than omission errors in most SDM initiatives, as accomplished by both TSS and SEDI, this does not mean that they are irrelevant or become irrelevant when prevalence decreases. Exceptions to this are very specific circumstances.

This study demonstrated that more consistent commission error weighting (as with SEDI) also circumvents a number of potentially artefactual signals as prevalence approaches zero. We also discussed the relative inability of TSS to compare performance across modelling conditions. For these reasons, whereas maximising TSS may be instrumental when presence-absence thresholds are required (Liu et al. 2013), TSS may not perform well in presence-background model evaluations (Somodi et al. 2017; Leroy et al. 2018). Unless circumstances require down-weighting of commission errors as prevalence decreases, SEDI's ability to take into account information on both error types across a wide range of modelling conditions makes it a better choice for low prevalence conditions, characteristic of presence-background modelling approaches.

The use of similarity measures as an alternative to TSS has recently been suggested by Leroy et al. (2018). However, similarity measures are only applicable when there is a known truth such as when modelling virtual species (see Hirzel et al. 2001; Leroy et al. 2016 for an introduction to virtual species). One might also consider the use of bootstrapping and related techniques (Efron 1983) such as sub-sampling in model evaluation (Verbyla and Litvaitis 1989), i.e. the repeated sub-sampling of background points to the number of presences and averaging of the resulting scores. Sub-sampling would remove the bias caused by large numbers of background points towards true absences from confusion matrices altogether, albeit at the cost of being computationally intensive. Therefore, alternative measures such as SEDI or, for some specific cases, ORSS, appear superior as they are neither limited to virtual species nor costly in terms of computation.

Conclusions

In our study, we focused on the importance of model evaluation in the context of ecology and conservation. The problems discussed are particularly relevant in systematic conservation planning (Margules and Pressey 2000; Lin et al. 2014; Guillera-Arroita et al. 2015) but will likely cause issues in any application of SDM. This is, since in any application, different scores favour different models (e.g. over-predictive vs. under-predictive, as they differentially respond to modelling conditions (Woodcock 1976), which inevitably affects outcomes.

Our results indicate that ORSS is a suitable evaluation metric for high-confidence presence-absence data, high prevalence situations or if strictly equal error weighting is required. SEDI and to a lesser degree TSS, are suitable evaluation metrics for presence-background SDM initiatives, since the error weighting of the evaluation metrics better reflects low-confidence pseudo-absence data. However, since SEDI provides more consistent performance scores and weighting of commission errors over a wide range of study extents (and background point totals) and prevalence, it is better suited for presence-background SDM, which is applied over a wide range of modelling conditions (i.e. to common or rare species and across single protected areas or whole continents). Finally, we strongly recommend abstaining from the use of TSS whenever prevalence is lower than approximately 2.5% or when a large number of background points is used that drives the total number of cells in the confusion matrix to more than roughly 30,000 cells since TSS will not distinguish between low and high commission error rates or useful and random models.

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