



Letter to the editor

Multiple measures of biodiversity change make for the strongest analyses with historical data – Reply to [Guzman et al., 2021](#)


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[Guzman et al.'s \(2021\)](#) comment on occupancy models from [Soroye et al. \(2020\)](#) explores parameterization with a series of simulations and concludes that modelled rates of bumblebee decline in North America were overestimated. We welcome this foray into occupancy modelling approaches. Yet, [Guzman et al. \(2021\)](#) missed the key messages from our work with their dismissal of bumblebee declines based on their choices of occupancy modelling approach, while underestimating species declines known from independent data sources. There is a recipe for justifying conservation inaction despite multiple lines of evidence that many species are threatened or soon will be.

In [Soroye et al. \(2020\)](#), we used a large dataset of historic North American and European bumblebee observations to link recent species- and assemblage-level change to climate change, having created a new method to identify the frequency and severity of extreme weather relative to species' realized niche limits. We supported discoveries with multiple, robust tests of historic change in assemblage richness, observed extirpation, observed colonization, and change in probability of occupancy, while accounting for variation in sampling effort in various ways. While we presented continental estimates of observed declines, we discovered relationships between biodiversity change and climate change. We did not present species-level estimates of change or endangerment. As [Guzman et al. \(2021\)](#) note, conservation practitioners require accurate estimates of species decline for listing species for protection. This process of estimating absolute levels of population change must be rigorous, including expert insight and independent tests of recent and historic change (e.g. IUCN Red List), something we did not attempt.

[Guzman et al. \(2021\)](#) conclude from occupancy models that there is little evidence of widespread declines in bumblebees, despite the fact that declines in North American and European bumblebees have been observed in detail over recent decades (e.g. [Goulson et al., 2008](#); [Kerr et al., 2015](#)). The IUCN Red List indicates that 8 of 33 North American species in our study (24%) face different levels of endangerment and are listed as “decreasing”. Yet, quantitative analysis in those expert assessments also indicates that about 51% of species (17/33) may have experienced decline over recent decades ([Table 1](#)). Comparing

occupancy results from both [Soroye et al. \(2020\)](#) and [Guzman et al. \(2021\)](#) to these expert assessments ([Table 1](#)) suggests that while [Soroye et al. \(2020\)](#) overestimate declines, [Guzman et al. \(2021\)](#) underestimate them by a similar margin (for species for which measurements exist). It is clear that there remains a need to further refine species-specific estimates of decline among bumblebees and these analyses should be rigorous and consider information from independent sources, like IUCN Red List assessments.

Precise knowledge of population trends is missing for many species, which underscores the vital need for expert estimates of population change to ground pure modelling results of the sort advocated by [Guzman et al.](#) While IUCN Red List assessments for North American species provide detailed quantitative information on population declines and species trends (e.g. [Table 1](#)), no such information is available for European Red List assessments, even for threatened species (e.g. [Rasmont et al., 2015](#)).

In focusing on the occupancy modelling methods of our original paper, [Guzman et al. \(2021\)](#) ignore the key discoveries in that work and the multiple lines of evidence that support them. Importantly, they find that occupancy estimates in Europe appear robust to changes in model parameterization, further supporting the interpretation of our results there (aside from estimates of uncertainty). [Guzman et al. \(2021\)](#) also incorrectly state that we removed *Bombus distinguendus* from analyses. As noted in [Soroye et al. \(2020\)](#), *B. distinguendus* was omitted from calculations of continental change in North America because it was not observed in the historic period. This species was retained otherwise.

Occupancy modelling offers a useful way to understand biodiversity change, but given the observed sensitivity of occupancy models to differences in model parameterization and rapidly evolving modelling techniques, large-scale studies of biodiversity change with historic data should focus on measuring change in multiple ways (i.e. not just with occupancy models) and comparing model results to independently generated estimates. Proposing and testing “risky predictions” (after Karl Popper) with multiple, independent lines of evidence (e.g. from species richness change, observed extinction, observed colonization), and accounting for sampling in various ways (e.g. measuring richness

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Table 1

Species-specific estimates of decline compared to independently assessed estimates of average decline as reported in species' IUCN Red List assessments. No European species had detailed information provided, and 16 North American species with an average decline of zero were excluded (this typically indicated suspected stable or increasing population trend). Details are all from global assessments, except for *B. bohemicus* (which occurs in Europe as well) where we used the North American-specific details provided in the assessment.

Species	IUCN status	IUCN population trend	IUCN average decline	Guzman et al occupancy change	Soroye et al occupancy change	Guzman et al difference	Soroye et al difference
Affinis	CR	Decreasing	-69.36	-69.00	-83.72	0.36	-14.36
Appositus	LC	Unknown	-28.40	8.74	-45.79	37.14	-17.39
Auricomus	LC	Stable	-24.11	0.81	-47.59	24.92	-23.48
Bohemicus	CR	Decreasing	-94.77	-72.58	-27.07	22.19	67.70
Borealis	LC	Stable	-7.11	-42.05	-68.22	-34.94	-61.11
Centralis	LC	Stable	-7.55	16.42	-42.63	23.97	-35.08
Fervidus	VU	Decreasing	-30.69	-1.89	-40.86	28.80	-10.17
Fraternus	EN	Decreasing	-59.96	-42.30	-68.59	17.66	-8.63
Huntii	LC	Stable	-25.34	-8.11	-56.45	17.23	-31.11
Insularis	LC	Stable	-12.37	-12.68	-56.28	-0.31	-43.91
Melanopygus	LC	Stable	-15.78	13.51	-44.88	29.29	-29.10
Morrisoni	VU	Decreasing	-57.74	-50.81	-74.07	6.93	-16.33
Nevadensis	LC	Stable	-15.48	3.81	-52.35	19.29	-36.87
Occidentalis	VU	Decreasing	-40.32	-20.62	-53.19	19.70	-12.87
Pennsylvanicus	VU	Decreasing	-51.38	-44.30	-65.77	7.08	-14.39
Sylvicola	LC	Stable	-10.23	-1.52	-55.52	8.71	-45.29
Terricola	VU	Decreasing	-49.94	-38.95	-66.10	10.99	-16.16
Mean (\pm SE)			-35.32 (5.91)	-21.26 (6.75)	-55.83 (3.31)	14.06 (3.85)	-20.50 (6.36)
Median			-28.40	-12.68	-55.52	17.66	-17.39

change in well-sampled areas, using sampling as a covariate in models, testing various thresholds for inferred absence of a species, testing multiple spatial resolutions of analysis, using occupancy models) allows for robust tests of theory that can reveal potential cross-continent mechanisms of decline, as done in Soroye et al. (2020). Approaches like this, which use occupancy models as one among many tools, provide far stronger bases for proposing and testing mechanisms of global change-related biodiversity change.

Declaration of competing interest

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

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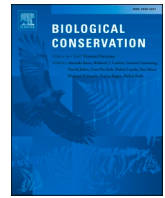
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1. Introduction

Soroye et al. (2021) state that the novelty of Soroye et al. (2020) is the link between climate change and climate sensitivity as drivers of biodiversity change. This may be true, but that analysis hinges on accurate estimates of species' occupancy. Unfortunately species' occupancy and the underlying data are biased in such a way that they cannot be used without properly accounting for these biases.

In a response to our article on occupancy models and bumblebee declines, Soroye et al. (2021) argue that the more lines of evidence validating scientific claims, the better. We agree. In the case of Soroye et al. (2020), however, the “multiple lines of evidence” are based on a single dataset that was processed and collapsed by binning specimens in time and space. Such data processing could introduce systematic biases (e.g., when binning specimens in time, the authors used time intervals of different lengths in their historic and modern eras). In addition, observations themselves have high variation through time and space in their collection, curation, and identification (Ascher et al., 2020). Thus, any line of evidence based on these data *must* account for these potential biases. We have shown that such biases did exist (e.g., known site visitation dropped dramatically between the historic and modern time period, particularly in North America) and that the method Soroye et al. (2020) used did not correct this bias. We then made methodological improvements that we showed reduced bias and subsequently applied this improved method to their data-set. Soroye et al. (2021) did not comment on the differences between our methodologies, but rather compared model-estimated declines to IUCN estimates (IUCN, 2020; Hatfield et al., 2015).

This is, indeed, a potential model validation test — i.e., a new line of evidence. However, Soroye et al. (2021) only considered species that

IUCN has listed as having an average decline different from zero (17/34 species; IUCN does not list population increases). Soroye et al. (2021) then calculated the mean of the differences between model predictions and IUCN estimates. However, this is not meaningful, as negative and positive discrepancies can effectively cancel one another, falsely giving the appearance that a model is producing good estimates. We have expanded Soroye et al. (2021)'s Table 1 to include the species whose average decline is listed by the IUCN as 0 (17/34), calculated the average increase for these species, and then used RMSD (root-mean-squared deviation instead of the mean) to evaluate model performance (see Table 1).

Our method was closer to IUCN estimates than their original method on the 17 species presented in their response (RMSD for Guzman et al. = 20.99, RMSD for Soroye et al. = 33.36; Table 1), and even more so on the complete set of 34 species (RMSD for Guzman et al. = 24.95, RMSD for Soroye et al. = 55.27; Table 1). Further, of the 17 species that are estimated to be stable or increasing by the IUCN, Soroye et al. (2020) estimated 16/17 to be declining, whereas our method estimated 14/17 to be stable or increasing.

Soroye et al. (2021) also state that “occupancy estimates in Europe appear robust to changes in model parameterization.” We found an average 6% decline (95% BCI = [−21%, 2%]), compared to (Soroye et al., 2020) who found a 17% decline (SE = 5%). We feel that a revised effect size that is one third in magnitude is a noteworthy change. The response by Soroye et al. (2021) does not challenge any of the claims made in our analyses. Given the biases we have documented there, we stand by our assertion that the climate analyses in Soroye et al. (2020) need re-evaluation before conservation action can be prioritized. Effective biodiversity stewardship continues to be our overarching goal.

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Table 1

Including all species, the comparison between the estimates from the IUCN Assessment for North American *Bombus* spp. and the estimates from occupancy modeling by Soroye et al. (2020) and Guzman et al. (2021) show that the RMSD of Soroye et al. (2020) is much greater more than the RMSD of Guzman et al. (2021). The values of “current range size relative to historic range”, “persistence in current range relative to historic occupancy”, “current relative abundance relative to historic values” and “IUCN average decline” are all taken from the IUCN assessment. We note however, that IUCN average declines are bounded at zero, and do not show population increases. We have used the same calculation that IUCN uses, to calculate average increases which we label average change. The values highlighted in orange indicate those included in Soroye et al. (2021)’s Table 1. Since *Bombus distinguendus* has no values for persistence or current range relative to the historic period, we do not include it in community calculations.

Species	IUCN status	Current range relative to historic	Persist	Relative abun	IUCN aveg decline	Average change	Soroye change occu	Guzman change occu	Soroye diff	Guzman diff
affinis	CE	54.68	29.77	7.46	-69.36	69.36	-83.72	-69.00	14.36	-0.37
appositus	LC	82.57	85.57	46.65	-28.40	28.40	-45.79	8.74	17.39	-37.14
auricomus	LC	88.62	88.98	50.08	-24.11	24.11	-47.59	0.81	23.48	-24.92
bifarius	LC	119.78	94.13	126.53	0	13.48	-51.16	-5.06	64.64	18.54
bimaculatus	LC	96.56	204.77	188.19	0	63.17	-14.02	49.73	77.20	13.44
bohemicus	DD	3.72	9.16	2.82	-94.77	94.77	-27.07	-72.58	-67.69	-22.19
borealisis	LC	92.98	98.78	86.91	-7.11	7.11	-68.22	-42.05	61.11	34.94
centralis	LC	107.48	88.61	81.27	-7.55	7.55	-42.63	16.42	35.09	-23.96
citrinus	LC	98.54	124.16	130.12	0	17.61	-42.67	4.84	60.28	12.76
cryptarum	DD	28.94	268.16	71.87	0	22.99	-14.72	71.22	37.71	-48.23
distinguendus	DD	0	NA	3789.38	0	1163.13	719.67	22.07		
fervidus	VU	86.98	85.84	38.04	-29.71	29.71	-40.86	-1.89	11.15	-27.83
flavifrons	LC	79.45	131.91	161.79	0	24.38	-43.42	8.34	67.80	16.05
fraternus	EN	71.38	43.33	14.4	-56.96	56.96	-68.59	-42.30	11.63	-14.67
frigidus	LC	39.69	161.88	116.34	0	5.97	-58.59	-12.28	64.56	18.25
griseocollis	LC	90.21	166.87	215.25	0	57.44	-18.15	40.62	75.59	16.82
huntii	LC	92.49	60.98	70.51	-25.34	25.34	-56.45	-8.11	31.11	-17.23
impatiens	LC	97.5	158.62	294.17	0	83.43	-33.58	20.74	117.01	62.69
insularis	LC	89.31	86.28	87.29	-12.37	12.37	-56.28	-12.68	43.90	0.30
jonellus	DD	35.83	198.31	294.02	0	76.05	-24.10	41.03	100.15	35.02
lucorum	NA						-40.16			
melanopygus	LC	71.48	99.33	81.85	-15.78	15.78	-44.88	13.51	29.10	-29.29
mixtus	LC	51.94	141.35	263.61	0	52.30	-41.97	8.21	94.27	44.09
morrisoni	VU	81.87	27.49	17.43	-57.74	57.74	-74.07	-50.81	16.34	-6.93
nevadensis	LC	104.71	84.78	64.08	-15.48	15.48	-52.35	3.81	36.87	-19.28
occidentalis	VU	77.96	72.56	28.51	-40.32	40.32	-53.19	-20.62	12.87	-19.70
pennsylvanicus	VU	81.18	53.24	11.44	-51.38	51.38	-65.77	-44.30	14.39	-7.08
perplexus	LC	93.97	166.31	92.19	0	17.49	-33.55	29.53	51.04	-12.04
rufocinctus	LC	91.45	106.47	154.88	0	17.60	-51.20	-10.61	68.80	28.21
sylvicola	LC	90.03	82.86	96.41	-10.23	10.23	-55.52	-1.52	45.29	-8.71
ternarius	LC	69.05	103.61	162.21	0	11.62	-58.80	-19.83	70.42	31.45
terrestris	NA						22.67			
terricola	VU	63.69	67.32	19.17	-49.94	49.94	-66.10	-38.95	16.16	-10.99
vagans	LC	106.68	103.69	108.97	0	6.45	-47.68	-4.99	54.13	11.43
vandykei	LC	72.82	59.99	163.71	0	-1.16	-55.14	-0.69	53.98	-0.47
vosnesenskii	LC	89.3	76.5	122.3	0	-3.97	-42.20	-4.30	38.23	0.34
Mean		76.55	103.99	105.17	-17.55	-3.99	-45.64	-3.62		
RMSD (declining sp.)							33.36	20.99		
RMSD (all sp.)							55.27	24.95		

Declaration of competing interest

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

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