REPTILES & AMPHIBIANS

Reptilian Richness and Topographic Heterogeneity in the 'Uruq Bani Ma'arid Protected Area in Southern Saudi Arabia

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Abstract.—An important goal of the Kingdom of Saudi Arabia's Green Initiative is to increase the percentage of protected areas. In terms of Saudi Arabian wildlife, reptiles have been identified as an important element for protection, although their status and distributions are not clearly understood. Herein we report the findings from a survey of reptiles conducted in the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia. Data were collected using a robust-style design and analyzed with multi-species occupancy modelling. Eighteen species were detected, and species occupancy and species richness were positively associated with a topographic heterogeneity index calculated from satellite imagery. Notwithstanding the levels of uncertainty, occupancy by all species and species richness indicated a general proclivity for higher levels of topographic heterogeneity. In the study area, these associations corresponded to a mixture of rocky and sandy habitats. This is a preliminary study, but the results warrant further investigation of the topographic heterogeneity index in arid environments to establish any broader generalities. This will be useful, as a relationship between species distribution and easily accessible satellite information can support the identification of possible sites for protection and will be important for any ongoing management of reptiles within established protected areas.

One important goal of the Kingdom of Saudi Arabia's Green Initiative (https://www.saudigreeninitiative.org/targets/) is to protect 30% of the country's land and sea by 2030. A key step to ensuring the initiative is realized is to quantify the richness and distribution of wildlife across Saudi Arabia, to not only identify key areas for protection but to collect baseline knowledge for ongoing management of those areas. As outlined by Wallace and Jago (2017), managing publicly valued natural areas follows a process of identifying key system elements, determining the required state of those elements (as described and measured by relevant attributes or properties), identifying important risk factors, and appropriately managing key processes to maintain or improve the state of each system element of interest. Key properties used to assess the state of wildlife elements include species occupancy and richness (Kéry and Royle 2008). In terms of Saudi Arabia's protected areas, maintaining or improving the richness and distribution of key wildlife elements will help to ensure that the areas are highly valued well into the future (e.g., Wallace et al. 2020).

One element that has been identified as important in terms of Saudi Arabia's wildlife is the kingdom's reptilian fauna (Alatawi et al. 2020). Unfortunately, the status and distribution of most, if not all, of Saudi Arabia's reptilian species are not as well understood as they could be (AbuZinada et al. 2004; Alatawi 2020; Šmíd et al. 2022). To rectify this situation, the National Centre for Wildlife (https://www.ncw.gov.sa/En/Pages/default.aspx) has embarked on a program to survey reptiles in a series of potentially important areas to build a baseline of information on species richness, occupancy, and distribution. This information will be used to: (1) assist in the identification of important areas for protection; (2) better inform national red listing status of different species; (3) provide baseline data for ongoing monitoring and reporting on the status of this important wildlife element into the future, and (4) better understand key habitat requirements for different species, leaving managers better placed to maintain and, where required, improve the status of species.

Many studies have investigated reptilian distribution and richness in relation to habitat types and variability (e.g., Harings et al. 2014; Berezowski et al. 2015; Gillespie et al. 2015). More recently, satellite-based information has been used to develop habitat indices, which are typically built around climatic, topographic, and/or biological information (Berezowski et al. 2015). For satellite-based assessments of habitat use by reptiles (Berezowski et al. 2015), indices based on vegetation and levels of disturbance are common (Coops et al. 2008; Arenas-Castro and Sillero 2021). However, in

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very arid regions, other metrics of topographic heterogeneity might be more useful, such as those relating to the extent of sandy or rocky areas (Sow et al. 2014). In the more arid regions of the world, satellite-based information has great potential to be useful in understanding where reptiles occur and why, which can assist in identifying new areas where a species is more likely to occur (Šmíd et al. 2021). Such information not only increases capacity to better understand, manage, and map distributions of extant species and their habitat preferences, but also provides opportunities to try to better understand past and future states.

Herein we provide an assessment of a survey of reptile species in the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia (Fig. 1). We estimated site-level species occupancy and richness and related variation in these two key element properties to topographical heterogeneity. We used a broader definition of topography, which is the study of the forms and features of the land surface, and were particularly interested in capturing spatial patterns of variation in regolith and bedrock, as these two topographical features are important for reptilian occupancy in more arid environments (Šmíd et al. 2021). Specifically, we developed a satellite-based index that captured topographic heterogeneity associated with a significant gradient in the protected area from the rocky Tuwaiq Escarpment (a Jurassic limestone massif) to the longitudinal dunes of the Rub' al-Khali (Empty Quarter; Islam et al. 2011). According to the work of Šmíd et al. (2021), we expected a reduction in species occupancy and richness with decreased topographic heterogeneity. This work also builds on a previous survey of reptiles by Aloufi et al. (2022), which resulted in the detection of 15 species across the area.

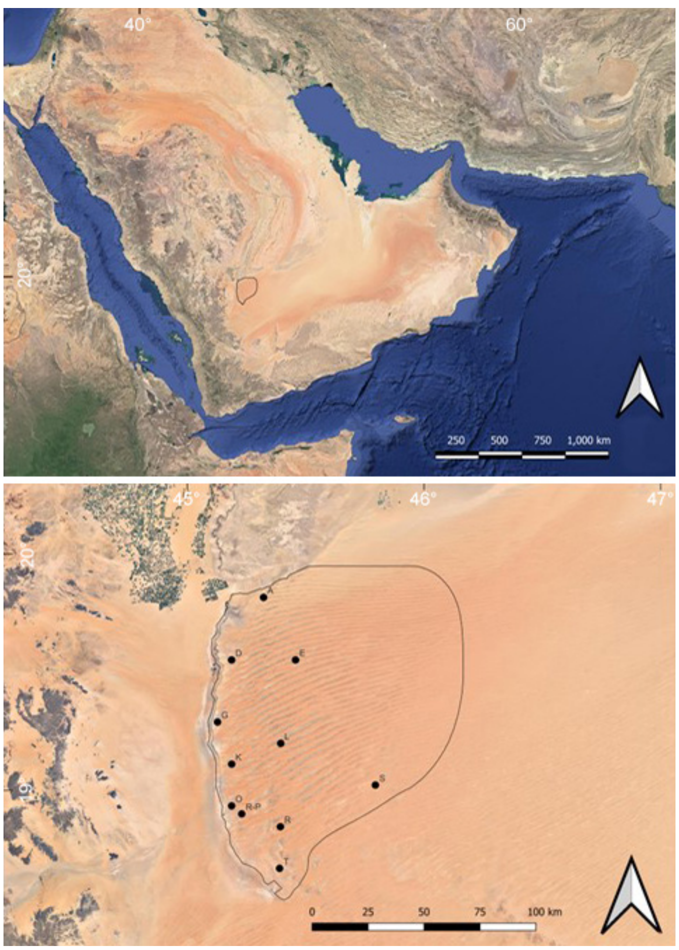

Figure 1. Location of the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia (top) and location of survey transect start points within the Protected Area (bottom). Google maps used for backgrounds.

Methods

Study Area.—The survey was conducted in the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia (Fig. 1). The 'Uruq Bani Ma'arid (total area of 12,787 km^2) varies in elevation from 720 to 940 m and has a mean annual rainfall of around 47 mm (Aloufi et al. 2022). The 'Uruq Bani Ma'arid Protected Area lies on the edge of the world's largest and most arid desert. It is positioned on a plateau created by the Al-Arid escarpment, which marks its western edge. The area is characterized by rocky areas and longitudinal sand dunes (Aloufi et al. 2022) and has a diversity of habitats including vegetated wadis, sandy plateaus, gravel plains, and inter-dune corridors (Islam et al. 2011). Consequently, the area is inhabited by a rich diversity of flora and fauna, including the Arabian Oryx (*Oryx leucoryx*), Arabian Sand Gazelle (*Gazella marica*), Arabian Mountain Gazelle (*Gazella arabica*) (UNESCO 2024), and the vulnerable Spiny-tailed Lizard (*Uromastyx aegyptia*) (Wilms et al. 2012). Additionally, the area is occupied by a unique mammalian carnivore assemblage including the Sand Cat (*Felis margarita*), Rüpell's Fox (*Vulpes rueppellii*), and the Honey Badger (*Mellivora capensis*) (Amin et al. 2021) and the 'Uruq Bani Ma'arid has a high diversity of plant species (including endemic Arabian taxa; Hall et al. 2011). Plant species such as Umbrella Thorn Acacia (*Acacia tortilis)*, Salam (*A. ehrenbergiana)*, Horse Mulga (*A. hamulosa*), *Calligonum crinitum*, White Saxaul (*Haloxylon persicum*), Ben Tree (*Moringa peregrina)*, and African Myrrh (*Commiphora myrrha*) are prominent in the area (Hall et al. 2011; Aloufi et al. 2022). Besides having a 'core area' the 'Uruq Bani Ma'arid Protected Area also has a surrounding buffer zone, which supports a large number of camels and other livestock for the local community, including Bedouins. The area is also a key destination for tourism.

Survey Methods.—We visited each of the 11 sites in the morning (0900–1100 h), afternoon (1600–1800 h), and evening (1900–2200 h) (Table 1). Surveys began at a predetermined starting point prior to 1–3 hours of visually searching for reptiles or their sign. When necessary, animals were captured by hand or with a net. Only individuals that were identified to species are included in the analysis.

The data include only a small number of repeated surveys (maximum of two repeated visits) at three sites (Table 1). Additionally, for two sites, repeated surveys were conducted several months apart (Table 1). We assumed that the occupancy state of the repeatedly visited sites did not change over the survey period and note that this survey approach has low statistical power and will have considerable uncertainty associated with parameter estimates and any associated inferences. However, we consider the analysis to be worthwhile for three important reasons: (1) any information on the reptilian fauna of the UBM and its habitat associations is useful for the area's management even if a level of uncertainty exists — management of the area is of a practical nature and is not an academic pursuit; (2) the analytical approach applied below simultaneously incorporates multiple species that are linked together within the hierarchical formulation of the model — indicating that data are used efficiently with improved precision (Sauer and Link 2002; Goijman et al. 2015); and (3) any information can be used to build knowledge over time; for example, these data can be used as prior information for subsequent analyses within a Bayesian framework, an effective and value-adding approach to building management knowledge and confidence (McCarthy and Masters 2005).

We assumed that differences among sites in time spent searching or pathways taken did not overly bias the results. However, to account for some of the possible temperature effects, we modelled detection on the 'days since the beginning of the survey' with a view that quantifying variation in the survey day would account for some of the possible effects of temperature on our capacity to detect species. As detailed below, we also modelled detection as a function of topographic heterogeneity as we believed a relationship between the two was likely. We do note that the surveys were started at the same times and the primary period constitutes a period during which reptiles likely were similarly active during each secondary survey.

Statistical Methods.—The small size of the data set made it important that we did not over-fit any models and, with that in mind, only a small number of covariates were used. Given the clearly visible transition in the study area from more rocky substrates in the west to those completely dominated by sand dunes in the east, and also bearing in mind the findings of Šmíd et al. (2021), we initially explored relationships between the raw number of species detected at each site and information contained in individual image band and band combinations derived from the Sentinel-2 satellite platform (details provided in Supplementary Material A).

Table 1. The 2021 Survey schedule (gray boxes) for each site in the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia. Refer to Fig. 1 for site locations.

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We used satellite-based information to quantify landform heterogeneity (as no detailed information about the area was collected during the survey and satellite-based information has been shown elsewhere to be useful in terms of quantifying variation in geology and regolith (e.g., Langford 2015). Sentinel 2 information is freely available and provided by the U.S. Geological Survey. Sentinel 2 information was assessed and processed with QGIS (QGIS Development Team 2009) and we made particular use of the Semi-Automatic Classification Plugin (Congedo 2021). Satellite imagery from April 2021 that encapsulated the study area was sourced and the standard pre-processing functions available in the Semi-Automatic Classification Plugin were applied. We assumed that broad topographic heterogeneity would not change significantly from the beginning to the end of the survey period and, as such, we used single satellite images to cover the protected area captured between 10–12 April 2021.

Three frames that were found to completely cover the study area were combined to create a mosaic for bands 2, 3, 4, 5, 6, 7, 8, 11, and 12. Using QGIS, a circular buffer with a diameter of 3 km (area ≈ 7 km^2) was centered on the starting point of each survey site, and the 'Zonal Statistics' function was used to calculate the mean, median, and standard deviation of pixel values for each band across each area. These estimates were combined as required to calculate the various indices that were assessed (Supplementary Material A). The relationship between raw species counts and each satellite image metric was examined visually in Microsoft Excel along with an R^2 estimate from simple linear regression (Supplementary Material A). Of note, comparable estimates were calculated from circles with a 1-km diameter and a 2-km diameter with little to no effect on the mean and median estimates. A 3-km diameter was chosen as it was thought to best represent the area traversed by surveyors while sufficiently capturing topographic heterogeneity in the surrounding area.

Thus, our topographic heterogeneity index (THI) was calculated as:

Eq. (1): THI =
(CVband 4 – SWIRband 12)/(CVband 4 + SWIRband 12)

The THI correlated reasonably well with raw species count (R^2 = 0.53; Supplementary Material A) and clearly related strongly to the protected area's various topographic features (Supplementary Material A, Fig. S.1.3).

The survey data were analyzed with the spOccupancy package (Doser et al. 2022) launched within the R software environment (R Core Team 2013; Version 4.3.0). Given the nature of the data (repeated secondary surveys within a single primary period), multi-species single-season occupancy modelling was applied (Doser et al. 2022). We first applied basic multi-species occupancy modelling and compared models with different combinations of two covariates: (1) a single occupancy covariate, the THI, and (2) combinations of two detection covariates; day of the survey (with the first survey treated as day 1) and the THI index. We hypothesized that species detectability would be affected by topographic heterogeneity.

All model-covariate combinations were compared with the Watanabe-Akaike information criterion (WAIC; Watanabe and Opper 2010), and the preferred models (within 2 units of the smallest WAIC score) were used to examine changes in species occupancy and richness. Given the possibility that spatial autocorrelation might be an issue (Doser et al. 2022), we ran a second series of spatial multi-species occupancy models with the same covariate configurations to account for this potential bias.

In the basic model, and strongly paraphrasing Doser et al. (2022), 'true' presence (1) or absence (0) is denoted by $Z_{(i,j)}$, where i = 1 to number of species and j = 1 to number of sites. A latent occurrence variable was assumed to arise from a Bernoulli process. Thus:

$$Z_{i,j} \sim \text{Bernoulli}\ (\psi_{i,j}),\ \text{logit}(\psi_{i,j}) = x_j^T \beta_i$$

$\psi_{i,j}$ is the probability of occurrence of species 'i' at site 'j', as function of the site-specific covariate and a vector of species-specific regression coefficients $x_j^T \beta_i$. As described by Doser et al. (2022), "the regression coefficients in multi-species occupancy models are considered random effects arising from a common community level distribution" — $\beta_i \sim \text{Normal}(\mu_\beta T_\beta)$. μ_β represents a vector of community level means for each covariate effect and T_β is a diagonal matrix with diagonal elements τ_β^2 accounting for the variability of each covariate effect among reptilian species.

$Z_{i,j}$ is not directly observed and $y_{i,j,k}$ represents the detection (1) or non-detection (0) of species (*i*) at site (*j*) during a replicated visit (*k*). Consequently, $y_{i,j,k}$ can be modelled as arising from a Bernoulli process whereby:

$$y_{i,j,k} \sim \text{Bernoulli}(p_{i,j,k} Z_{i,j}),\ \text{logit}(p_{i,j,k}) = v_{i,j,k}^T \alpha_i$$

$y_{i,j,k}$ is the probability of detecting species (i) at site (*j*) during survey (*k*) and species detection (or not) is a function of a vector of species-specific covariates ($v_{i,j,k}^T \alpha_i$). As with the occurrence regression coefficients, the species-specific coefficients can be thought of as arising from a common community level distribution:

$$\alpha_i \sim \text{Normal}(\mu_\alpha, T_\alpha)$$

In this instance, μ_α constitutes a vector of community level means for each detection covariate effect and is a diagonal matrix with diagonal elements τ_α^2 representing the variability of each detection covariate effect among reptilian species.

A second general model was examined that accounted for residual spatial autocorrelation. Also paraphrasing Doser et al. (2022), the community modelling approach can result in some species having high residual spatial autocorrelation. Thus, the first occupancy models were extended to include a spatial Gaussian Process for each species. As a result, occurrence probability for species (*i*) at site (*j*) takes the form:

$$\mathrm{logit}(\psi_i(s_i)) = x^{T}(s_j)\beta_i + \omega_i(s_j)$$

β_i follows the community-level distribution described above and $\omega_i(s_j)$ is a realization from a zero-mean spatial Gaussian Process,

$$\omega_i(s_j) \sim \mathrm{Normal}\ (0,\textstyle\sum_i\ (s,s^{'},\theta_i))$$

$0,\sum_i\ (s,s^{'},\theta_i)$ is defined as a $J \times J$ covariance matrix and is a function of the distances between any pair of site coordinates s and $s^{'}$ with a set of parameters (θ_i) that regulate the spatial process. Additional details about the modelling approach are in Doser et al. (2022).

Following Doser et al. (2022), standard multivariate normal priors were assigned to the occurrence (μ_β) and detection (μ_α) community-level regression coefficient means in all models. Independent inverse-Gamma priors were assigned to each element of τ_β^2 and τ_α^2 and Pólya-Gamma data augmentation was implemented to improve the efficiency of the Gibbs sampler (Doser et al. 2022).

For the first series of models that did not account for spatial autocorrelation, those with the occupancy covariate (THI) and either days since start of survey on its own or the days since start of survey and the THI were similarly preferred (WAIC scores were within 2 units of each other; Table 3). Given our general interest in the effects of topographic heterogeneity on reptilian distribution and detection, and because the model with both detection covariates was one of the most preferred, we used that model to further explore the relationships between species occupancy, richness, and topographic heterogeneity. This model converged well, and the data were a good fit to the model (Supplementary Material B).

The basic modelling assessment process described in detail by Doser et al. (2022) was followed: visually assessing parameter convergence, using the R-hat statistic (Kéry and Royle 2016) to further assess parameter convergence, and finally estimating Bayesian-p values (ppcOcc function) for each species and the overall community. R-hat estimates <1.1 indicate convergence and Bayesian-p values close to

Table 2. List of reptilian species detected in the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia during this study and by Aloufi et al. (2022).

Species detected in this study	**Also detected by Aloufi et al. 2022**	**Detected by Aloufi et al. (2022) but not in this study**
Bosc's Fringe-toed Lizard (*Acanthodactylus boskianus*)	✓	Aqaba Agama (*Pseudotrapelus aqabensis*)
Arnold's Fringe-fingered Lizard (*Acanthodactylus opheodurus*)		Schokari Sand Racer (*Psammophis schokari*)
Schmidt's Fringe-fingered Lizard (*Acanthodactylus schmidti*)	✓	Rock Gecko (*Pristurus* sp.)
Southern Tuberculated Gecko (*Bunopus tuberculatus*)	✓	
Arabian Horned Viper (*Cerastes gasperetti*)	✓	
Arabian Sandboa (*Eryx jayakari*)	✓	
Blanford's Short-nosed Desert Lizard (*Mesalina brevirostris*)		
Arabian Toad-headed Agama (*Phrynocephalus arabicus*)	✓	
Common Cliff Racer (*Platyceps rhodorachis*)		
Arnold's Rock Gecko (*Pristurus minimus*)		
Fan-footed Gecko (*Ptyodactylus hasselquistii*)		
Eastern Skink (*Scincus mitranus*)	✓	
Doria's Comb-fingered Gecko (*Stenodactylus doriae*)	✓	
Slevin's Sand Gecko (*Stenodactylus slevini*)		
Yellow-spotted Agama (*Trapelus flavimaculatus*)	✓	
Arabian Short-fingered Gecko (*Trigonodactyus arabicus*)	✓	
Egyptian Uromastyx (*Uromastyx aegyptia*)	✓	
Desert Monitor (*Varanus griseus*)	✓	

0.5 indicate good model fit (Kéry and Royle 2016). Model selection was based on a comparison of model WAIC values (using the waicOcc function).

Results

We detected 18 reptilian species during the survey (Table 2), 12 of which were also detected by Aloufi et al. (2022). Similarly, Aloufi et al. (2022) detected two known species and one unknown species not detected in this study (Table 2).

Overall, the mean and median estimates from the red band over each 7-km^2 area correlated more strongly with raw reptilian richness estimates than the other bands and band combinations (Supplementary Material A). This is perhaps not surprising as the red band is used in many soil and vegetation indices (Marzouki and Dridri 2022) and can help to differentiate between the different topographic features in which we were interested. However, we were concerned with topographic heterogeneity, and so we calculated a coefficient of variation from red-band reflectance for each 7-km^2 area and then calculated a ratio with another commonly used band in the measurement of topographic features, shortwave infrared reflectance (SWIR – band 12; centred on 1613.7 nm).

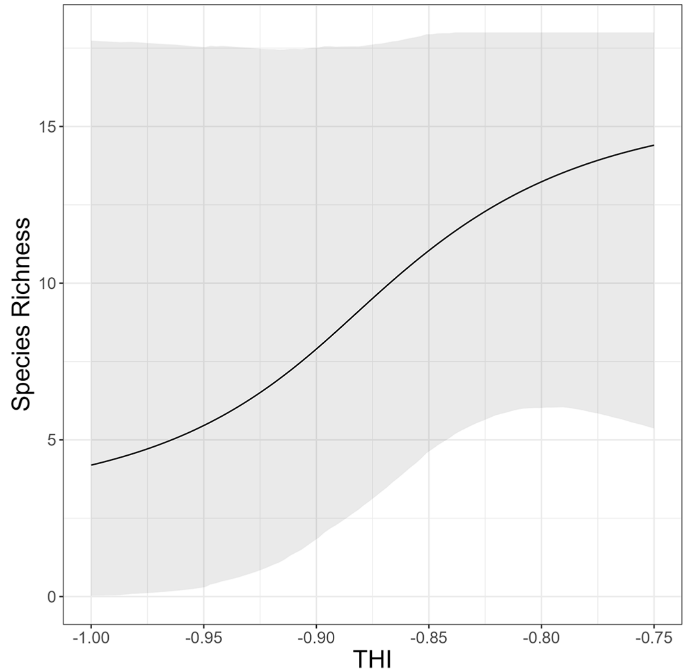

Figure 2. Predicted relationship between reptilian species richness and THI in the 'Uruq Bani Ma'arid Protected Area of southern Saudi Arabia. The solid black line marks mean richness estimates and the gray envelope indicates 95% Credibility Intervals.

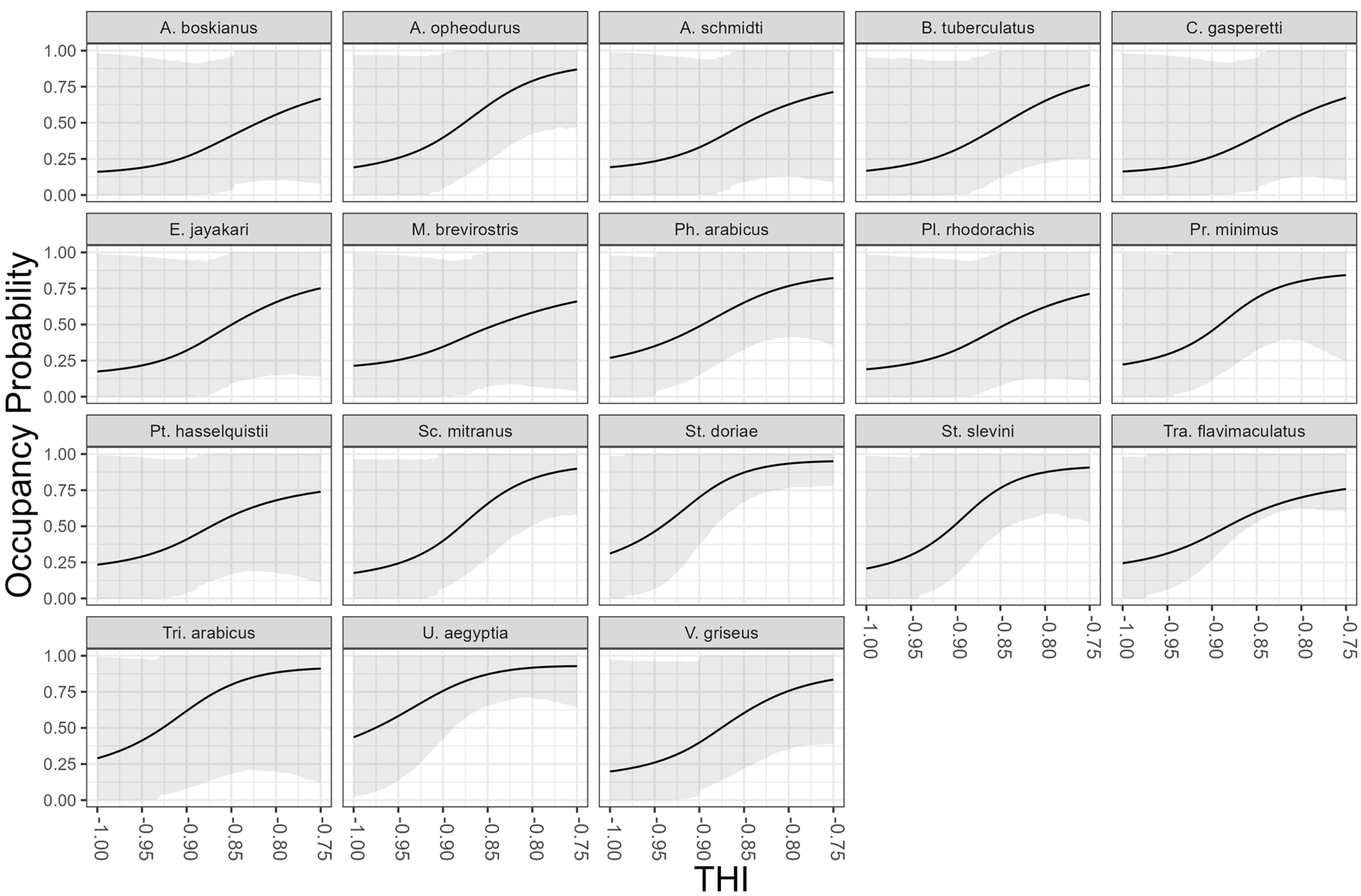

Figure 3. Predicted relationships between individual reptilian species occupancy and the THI in the 'Uruq Bani Ma'arid Protected Area of southern Saudi Arabia. The solid black line marks mean occupancy probability and the gray envelope indicates 95% Credibility Intervals.

At the community level, days since start of survey elicited no detectable effect (mean = -0.0192 (95% CI = -1.11–1.02); Supplementary Material B) or of THI on the probability of detection (mean = -0.34; 95% CI = -1.75–0.99; Supplementary Material B). The probability of detection for each species and its relationship with survey day varied considerably (Supplementary Material B), but mostly unconvincingly (95% CIs strongly included zero). Similarly, the probability of detection for each species and its relationship with THI was variable, but often not strong (95% CI included zero).

An overall positive but equivocal relationship existed between reptilian species richness and the THI (Fig. 2; mean = 1.04; 95% CI = -0.56–2.31; Supplementary Material B), noting also that every individual species was positively associated with the THI (Figs. 3 & 4; Supplementary Material B; again, bearing in mind that the 95% CI included zero).

The preferred models that accounted for spatial autocorrelation were similar when compared to previous models, but with even smaller WAIC scores (Table 3) indicating the importance of accounting for spatial effects and as such were the most preferred models overall. These models converged well, and the data were a good fit (Supplementary Material B).

Parameter estimates for the model that included all covariates were nonetheless similar in direction and magnitude to the comparable models without spatial autocorrelation (Fig. 4; Supplementary Material B) indicating that the basic conclusions hold for both models. In general, we do note that there was considerable uncertainty associated with the various parameter estimates reflecting the low power of the survey methodology that was applied.

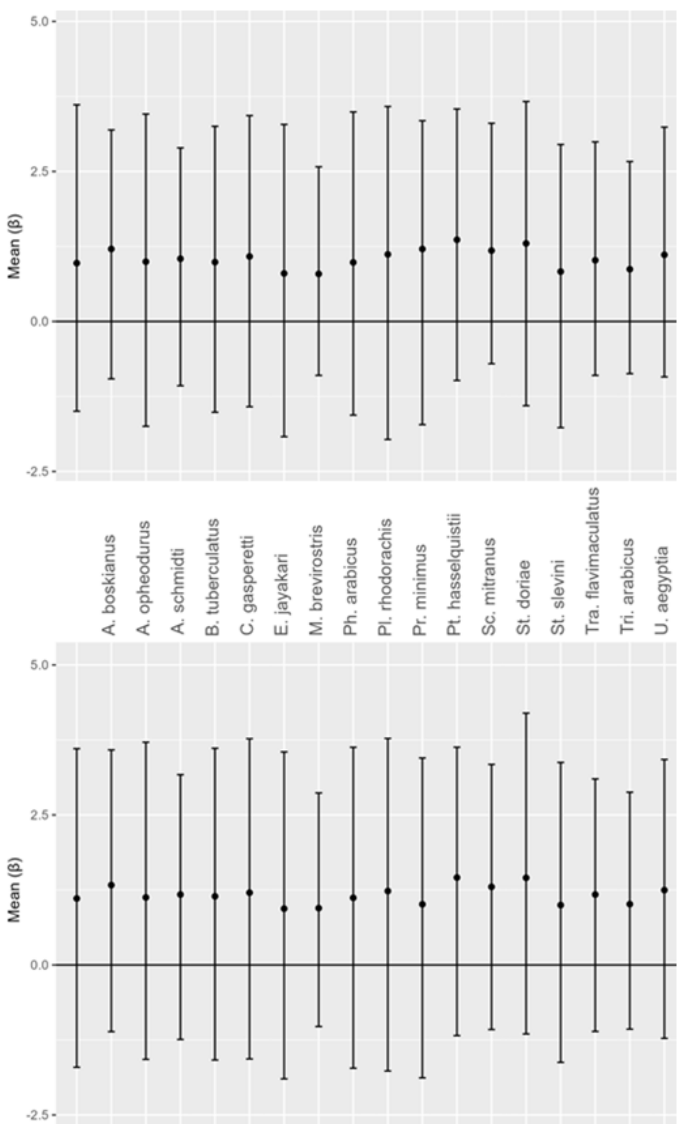

Figure 4. Mean beta (β) estimates for the relationship between occupancy probability and the THI for each species from the most preferred non-spatial model (top) and the most preferred spatial model (bottom). Point indicates mean beta estimate and whiskers indicate 95% Credibility Intervals.

Discussion

A survey of the reptiles of the 'Uruq Bani Ma'arid Protected Area in southern Saudi Arabia (Fig. 1) detected 18 species. Combining information from this survey and a previous survey by Aloufi et al. (2022), the area likely is home to at least 22 reptilian species, demonstrating the importance of the area for the reptilian fauna of Saudi Arabia. Through this work, we expand on the list of reptilian species known to occur in the 'Uruq Bani Ma'arid Protected Area.

Herein we provide an approach that, with further investigation, has a real capacity to improve the outcomes of reptilian surveys and management in arid environments. Specifically, we modelled relationships between an important habitat association expected for the area (topographic complexity) and reptilian distributions and richness. This information is not only of general value to protected-area managers but can now be used as a baseline for future study designs and analyses of data for reptilian surveys in arid Saudi Arabia — providing a level of extra efficiency (e.g., McCarthy and Masters 2005). We also note that protected-area managers often have access to wildlife data that are not collected in the most optimal manner. Therefore, finding ways to associate these data with key habitat metrics and incorporating that information into the primary literature is important. Such efforts help to build knowledge and make optimal use of information that is often underutilized, despite being resource expensive (Dobson et al. 2020).

We provide an easily and freely calculated index of the level of arid-area habitat complexity (available both spatially and temporally) that appears to be informative in terms of reptilian species richness and has the potential to facilitate future surveys and monitoring, in addition to identifying new areas for ongoing protection. The two bands that constituted the heterogeneity index that we applied here are commonly

Table 3. WAIC results for each model. THI indicates Topographic Heterogeneity Index, Day indicates day since start of survey, WAIC indicates Watanabe-Akaike Information Criteria.

Occupancy THI	Detection Day	THI	Type	Model No.	WAIC	Δ WAIC
✓	✓	X	Non-spatial	6	268.13	0.00
✓	✓	✓	Non-spatial	8	268.50	0.37
✓	X	X	Non-spatial	5	269.86	1.73
X	✓	X	Non-spatial	2	270.71	2.58
X	X	✓	Non-spatial	3	271.71	3.58
X	✓	✓	Non-spatial	4	271.71	3.58
✓	X	✓	Non-spatial	7	272.26	4.13
X	X	X	Non-spatial	1	273.83	5.69
✓	✓	X	Spatial	6	263.51	0.00
✓	✓	✓	Spatial	8	264.00	0.49
✓	X	X	Spatial	5	266.52	3.01
X	✓	X	Spatial	2	267.63	4.12
X	X	✓	Spatial	3	268.58	5.07
X	✓	✓	Spatial	4	268.83	5.31
✓	X	✓	Spatial	7	270.26	6.74
X	X	X	Spatial	1	271.81	8.30

associated with sand-based indices (Marzouki and Dridri 2022) and, as such, these preliminary results seem reasonable and align well with the predictions of Šmíd et al. (2021), who suggested that we should see a reduction in species occupancy and richness with decreased topographic heterogeneity. However, this is a study based on small sample sizes, low power, and run over a single primary period. As a result, we suggest that our results are promising, but should be treated as preliminary and subjected to further analysis of validity and assessments of any underlying biases. This analysis is worth investigating further because as a simple metric that is easily acquired, it could be very useful in the process of better understanding the habitat requirements of reptilian species in Saudi Arabia (and other arid areas) and to identify potential areas for future protection.

The likelihood that not all species were detected in the study area is clearly demonstrated by the differences in species lists between the two most recent surveys. Future surveys can be improved by increasing the number and timing of secondary surveys. Additionally, recording important 'metadata,' such as temperatures during surveys, pathways traversed, and start and stop times, will allow for more significant modelling. With robust information, the capacity of land managers to survey and monitor reptilian occupancy, richness, and associated habitat preferences will improve. Importantly, improved survey information will provide more accurate reporting and can facilitate the designation of areas and species for protection and the ongoing management of any risk factors that are identified.

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