REPTILES & AMPHIBIANS

Post-translocation Monitoring of an Egyptian Spiny-tailed Lizard, *Uromastyx aegyptia* (Forskål 1775), Population Displaced by a Global Giga-Project

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Abstract.—Infrastructure development programs and giga-projects (massive development initiatives) on the Arabian Peninsula are implementing large-scale faunal translocation efforts to promote nature recovery. Consequently, many species that have been locally extirpated are being restored. However, extant species also can be subject to translocations when populations are threatened by infrastructure development. Such is the case with the vulnerable Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) population in the boundary of the NEOM giga-project spanning 26,500 km^2 in the Kingdom of Saudi Arabia (KSA). Surveys were conducted to quantify Egyptian Spiny-tailed Lizard distribution and home range overlap within the infrastructure development footprint. Individual animals detected within that footprint were then captured and released in a reserve outside the path of construction. Post-release monitoring is critical to promote establishment of populations of translocated animals. We herein present a novel method of active burrow counts, measured over time, as a proxy for monitoring trends in population size of translocated lizards. We conducted ground surveys of the translocation site during the hot season to achieve a complete count of lizard burrows within a 5.95 km^2 fenced release area within NEOM's Mneifa Wildlife Reserve (MWR). We found that 74% of the 244 burrows that we detected were active and made 31 direct observations of adult animals. Thus, we detected strong evidence of an established population one year after translocation. The active-burrow count system that we developed is not unique to burrowing lizards and is suitable for all subterranean species of conservation concern. Novel techniques such as this will be essential to quantifying the establishment of post-translocation populations of species of conservation concern throughout the Arabian Peninsula and beyond.

Habitat loss is one of the greatest factors threatening species with extinction (Foley et al. 2005; Hogue and Breon 2022) and, as such, extensive conservation efforts have been implemented to protect and manage habitat. Notably, more than one hundred countries (HAC for N&P 2024) have committed to protect at least 30% of their marine and terrestrial area by 2030 in commitment with the "30 by 30" initiative (Dinerstein et al. 2019). Considerable progress to this end has been achieved with the achievement of designating 17% of global terrestrial area as protected by 2020 in line with Aichi Biodiversity Target 11 (UNEP-WCMC and IUCN 2021). In conjunction with protected area establishment, translocation is one of the most widely used means of recovering biodiversity; however, population establishment and self-sustained persistence (some of the key indicators of translocation success) are challenged by several factors. Factors associated with the small population sizes, such as allee effects and demographic stochasticity, can prevent population establishment (Robert et al. 2007; Armstrong and Seddon 2008). The number of individuals released can have the greatest influence over translocation success, compared with other factors (Morris et al. 2021). Population persistence is challenged largely by the genetic impacts of small populations such as inbreeding depression (Robert et al. 2007) and the loss of local adaptations due to gene flow (Lenormand 2002) as well as habitat unsuitability (Griffth et al. 1989). Management-related issues including inadequate feasibility plans and absent monitoring programs are also leading factors contributing to failed translocations (Bubac et al. 2019). These shortcomings, as well as areas for

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improvement in "mitigation-driven" translocations, as in the context of this paper, are outlined by Germano et al. (2015).

Mitigation-driven translocations are translocations that attempt to reduce mortality in a species of conservation concern as a response to the direct threat of human activities such as development projects (Germano et al. 2015). Conservation-driven translocations are those that relocate species specifically for conservation applications (Gaywood et al. 2022). The use of and investment in mitigation-driven translocations has increased rapidly in recent years, exceeding conservation-driven translocations (Hill and Arnold 2012). However, the majority of scientific research concerning translocations and post-release monitoring has been focused on conservation-driven translocations, despite the prevalence and generous funding of mitigation-driven translocations (Germano et al. 2015). In the absence of attention from the scientific community to mitigation-driven translocations, scientific best practices (e.g., proper documentation and scientifically informed implementation protocols) are not regularly applied in designing and documenting these translocations, resulting in high failure rates (Germano et al. 2015). Furthermore, post-release monitoring often is not required or implemented in mitigation-driven translocations (e.g., Lewis 2012) and, even when performed, the results are often not publicly accessible.

In addition to the expansion of protected areas via the 30-by-30 commitments, cross-sector global giga-projects are also promoting sustainability transitions, including biodiversity net gain. Mega-projects have characterized much of the recent urban and infrastructure development on the Arabian Peninsula. In effort to minimize the impacts of these mega-projects on biodiversity loss, among others, large-scale translocations of flora and fauna are widely used as mitigation strategies (Gardner and Howarth 2009). One of the largest giga-projects in the Kingdom of Saudi Arabia (KSA) is NEOM. This giga-project involves no less than ten major infrastructure development projects surrounded by a nature reserve of 25,000 km^2 (NEOM 2024). The use of translocations to mitigate the impacts of development is expected to become more common, so conducting and documenting them properly is important. Translocation efforts that are poorly planned with inadequate post-release monitoring have experienced high rates of animal lethality (e.g., Struhsaker and Siex 1998; Seddon et al. 1999; Fischer and Lindenmeyer 2000). Post-release monitoring, in general, has long been a lacking component in translocation efforts (Fischer and Lindenmeyer 2000; Ewen and Armstrong 2007; Hill and Arnold 2012). This is largely due to the attitude toward translocations as one-off management exercises (Armstrong and McLean 1995) without a focus on answering *a priori* questions. While efforts have been made to standardize (Sutherland et al. 2010) and stress the importance of post-release monitoring (Fischer and Lindenmeyer 2000), much remains to be done. Therefore, robust post-release monitoring is critical for the viability, growth, and establishment of translocated populations (Armstrong and Seddon 2008).

The Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) (hereafter STL) is a large, heavy-bodied agamid (Fig. 1). Classified as vulnerable (VU) on the IUCN Red List of Threatened Species (Wilms et al. 2012), the STL is a protected species in the KSA (Al-Fadhli 2020). This species faces threats from hunting and habitat loss and degradation (Wilms et al. 2012) as well as international trade in exotic pets and traditional medicine (Knapp 2004; Gondhali and Petrossian 2023). Current threats to STLs within the NEOM

Figure 1. An Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) at the entrance of its burrow in the Mneifa Wildlife Reserve, NEOM, KSA. Photograph by Muhammad Elmirghani.

region are hunting and habitat degradation from intensive livestock grazing and off-road driving. To protect STL populations from infrastructure development in NEOM, The Kingdom's National Center for Wildlife (NCW), requires the translocation of any STL in the path of development (Boug et al. 2022). We herein report on the effectiveness of this translocation after one year and propose a survey method to track post-translocation STL populations and those of other burrowing species of conservation concern.

We first sought to confirm the presence of STLs within the Mneifa Wildlife Reserve (MWR) approximately one year after the initial translocation. Given that no observations or records of STLs or active burrows in MWR existed prior to translocation, we assumed that the presence of STLs indicated that at least some released lizards survived for a year. Secondly, we determined that, if indications exist that released STLs are established (defined as surviving in numbers, suggesting that the habitat and conditions are favorable and predation is not excessive), that the population is equilibrating to levels (dispersion, density) within an estimated carrying capacity and natural ranges of variation, and that they are reproducing. Finally, we tested different monitoring methods in order to recommend robust and cost-effective survey methods for monitoring the status of large-scale STL translocations in NEOM and elsewhere. We focused on testing and recommending monitoring approaches that can allow conservation managers to confirm the presence of STLs and track trends in STL populations in release areas over time. This provides a framework to facilitate accurate and efficient monitoring of STL populations throughout their range. In addition, the post-release monitoring principles underlying this method are applicable to other translocated wildlife beyond the KSA.

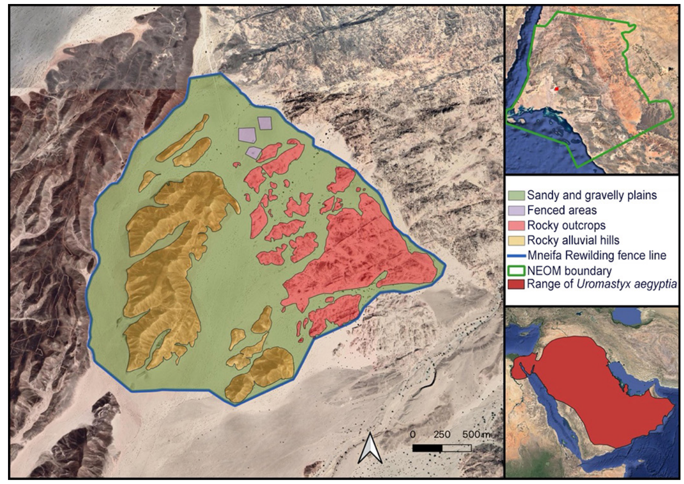

Figure 2. Google Map© showing the location of the Mneifa Wildlife Reserve, NEOM, KSA, and primary habitat types in the reserve.

Table 1. Areas of different habitat types in the Mneifa Wildlife Reserve, NEOM, KSA.

Zone	Area (km^2)	Number of Burrows	Suitable Habitat
Rocky alluvial hills	1.34	3	No**
Granite foothills/rocky outcrop	1.13	4	No
Sand and gravel plain	2.43	215	Yes
Plains and lower alluvial hills*	3.42	235	Yes
Fenced-off pens	0.04	0	No
Entire MWR rewilding area	5.94	244	—

*All is suitable habitat.
**Except some soil pockets adjacent to plains.

Materials and Methods

Study Area.—We established our study area in the MWR within NEOM, KSA. The reserve is located 17 km southeast of Al Bad' (28.34325, 35.14732) (Fig. 2) in the coastal lowlands adjacent to the foothills of Jabal Zuduh and is the first receptor site for STLs translocated from other construction sites within NEOM. The reserve consists of gravel and sandy plains, rocky outcrops, and large, rocky alluvial hills (Table 1; Fig. 1). Flat sandy or gravelly plains are considered favorable habitat as they are consistent with areas that STLs use for burrows (Al-Sayegh et al. 2020; Wilms et al. 2010). The vegetation structure in MWR is largely coastal, non-halophytic vegetation, dominated by *Haloxylon salicornicum*. *Vachellia tortillis*, *Iphiona scabra*, *Senna italica*, *Zilla spinosa*, and *Tephrosia purpurea* also occur. Seasonal grass (*Stipagrostris hirtigluma*) covers the rocky outcrops in spring. The mean annual temperature in MWR is 23 °C with a mean maximum temperature of 37.2 °C (August). Monthly precipitation averages around 1.3 mm. No rain fell in the MWR during the period of this survey, which is typical of the mean monthly precipitation in July and August (0.0 mm) (Vincent 2008; NEOM Environment 2024).

STLs.—Adult STLs can exceed 700 mm in length and 2.5 kg in mass (Wilms et al. 2009). With a characteristic thick spiny tail comprised of 20–24 whorls, STLs are generalist herbivores and feed on a variety of annual and perennial plants as well as the *V. tortilis* tree (Bouskila 1986, Appendix 5; Cunningham 2001). Invertebrates, such as beetles and scorpions, comprise a small part of the species' diet (Cunningham 2001; Castilla et al. 2011a). The species' range occurs across arid desert and semi-desert environments, with a tendency to select for open plains with a coarse sand and gravel substrate type (Wilms et al. 2009). These plains are typically sparsely vegetated by shrubs and small trees. STLs prefer to burrow in compacted, sandy soils, although they will burrow in rocky habitat where soil accumulations facilitate burrowing.

NEOM STL Translocation Program to Date.—The IUCN/SSC (2013) Relocation Guidelines and the KSA NCW translocation guidelines (Boug et al. 2022) were followed in the NEOM STL relocation effort. STLs were captured from receptor sites in the region surrounding the MWR, primarily in the coastal lowlands. Capture methods involved netting to reduce injury and stress to the animals (Boland and Smithson 2023). Following capture, biometric data were recorded for each STL. No markings or tracking devices were applied to the lizards. One hundred and twenty-five individuals were kept in captivity for as long as 11 months prior to release while the MWR was being prepared. Spiny-tailed Lizards were released into artificial burrows to provide initial refuges from predators and extreme ambient temperatures. These constructed burrows were comprised of a half PVC pipe angled into the ground at 45° and extending to a depth of 75 cm. However, based on camera-trap observations at five of these burrows, most lizards appear to have departed these burrows within 48 hours. Supplementary vegetables were provided at the entrance to each burrow for the first two weeks following translocation (NEOM Nature Region 2022).

As of 27 July 2023, 161 STLs had been translocated. Five STLs were released into a small, fenced enclosure in May 2022 and monitored by camera traps to determine whether the lizards would use the fabricated burrows. The first major translocation of 60 individuals was on 23 June 2022. A further 60 individuals were translocated on 3 July 2022. The remaining 36 individuals were translocated in groups of one to five during the period from 3 July 2022 to 27 July 2023. STLs less than 33 cm and below 250 g were considered juveniles. Thirty-two juvenile STLs were released into the translocation site during the entire release period.

Monitoring Population Trends versus Full Counts.—Ideally, one could monitor the status of each released individual over time, supported by close observation and tracking devices. However, this would require an inordinate amount of time and resources if hundreds of lizards are being translocated into extensive natural landscapes. In addition, direct observation of all individuals was not possible in MWR as the released animals were not tagged with tracking devices and direct observations of all individuals for this burrowing species is challenging, at best. Thus, we focused on counting the number of active burrows (defined below) as a proxy for the abundance of STLs. Establishing a baseline (one year post-translocation) abundance estimate will allow managers to track trends in the STL population over time, determining if populations are increasing, decreasing, or stabilizing after a year. We propose that this approach provides robust indicators of translocation success. For key assumptions underlying this approach and the implications for our survey methods, see Appendices 1 and 2.

Surveying for STL Presence.—We confirmed the presence of STLs at MWR using four lines of evidence. First, we used active STL burrows as an indicator of STL presence in the area. We defined active as those burrows that had a distinctive elliptical, oval, or semicircular entrance, typically at least 10 cm wide and 8 cm high and with relatively clean edges (i.e., smooth and with little debris) (Wilms et al. 2010). Active burrows generally matched known reference burrows and had clear unobstructed entrances and looked as if an animal had been moving in and out of the burrow as indicated by drag marks, claw marks, and lack of spider webs and debris at the entrance. Active burrows often had aprons of sediment spreading out from the entrance and feces that appeared recent or fresh near the entrance. STL feces are 1–3 cm in length and roughly 1 cm thick and occur in elongated, rounded cylinders or pellets, often with a chalky urinary bolus comprised primarily of urates (Castilla et al. 2011b). Plant fibers often are visible in fecal pellets and urinary boluses vary from white to yellow or green. The feces are diagnostic and not easily confused with those of other wildlife in the area. For each burrow, activity status was determined with the binary active or inactive. All criteria were not necessarily assigned to a burrow. For example, fresh tracks are less obvious in coarse gravel than in sand. Only burrows with distinct entrances were recorded, meaning long-abandoned burrows visible as mounds or raised sections of earth were ignored. Long-abandoned burrows were not relevant to our primary objectives of determining STL presence and number of active burrows. In addition, a lack of distinguishing features in many of these older mounds would have resulted in low confidence in classifying them as STL burrows. Small mammal burrows and burrow complexes also create raised soil platforms and aprons.

Secondly, the presence of STL feces both around burrow entrances and, although not quantified in this survey, were photographed and served as evidence of STL presence. Thirdly, we assessed camera-trap photographs of STLs after deploying camera traps at 20 different burrow entrances for three days. These camera traps were able to record STLs entering and leaving burrow entrances at any time of day. Finally, when possible, we photographed STLs observed in the field.

Assessing Abundance of STLs at MWR during July/August 2023.—The count of mature STLs in MWR was estimated using the number of active burrows encountered. For confident estimates, the number of active burrows must be correlated with population size when the coefficient of correlation is known or, if possible, the creation of a regression model to describe the relationship between the number of active burrows and abundance. Alternatively, the number of active burrows could be used as an index of abundance, providing information on population trends over time rather than actual abundance estimates. Noting the potential limitations of this latter approach is important. Factors other than population size could determine the number of active burrows. A study by Carthy et al. (2005) on Gopher Tortoises (*Gopherus polyphemus*), another herbivorous reptile that uses burrows, discussed the use of number of active burrows as an index of abundance, suggesting that declines in habitat quality, which can result in increased burrow usage as a result of home-range expansion, could give false evidence of a positive trend in population size. Despite this, the determination of population trends based on changes in active burrow counts can be performed under the assumption that the only factor affecting number of active burrows is population size. No major changes in vegetation were observed over the last two years in MWR, due perhaps to very low rainfall and high grazing pressure from larger reintroduced native herbivores (i.e., Arabian Gazelle, *Gazella arabica*; Sand Gazelle, *Gazella marica*; Arabian Oryx, *Oryx leucoryx*; North African Ostrich, *Struthio camelus camelus*) within the relocation area.

Survey Protocol.—Surveys during which burrows were counted were conducted in favorable habitat across the area (Fig. 3). We estimated the visual field viewed from the vehicle on each side was 25 m. We found no evidence that mature STLs had multiple entrances to their burrows. We recorded the tracks of our surveys and precise point locations of burrows using a hand-held GPS unit. When a burrow was detected, a GPS pin was dropped at the entrance of the burrow and activity and other features were assessed while on foot. Some areas of unsuitable habitat were also surveyed on foot to determine the validity of assumptions of unsuitability. The following features were recorded: whether an STL was directly observed and time of day (Fig. 4), signs of STL activity (an active burrow), burrow entrance height and width, burrow entrance orientation, air temperature, time of day, wind speed, habitat type (e.g., rocky outcrop, rocky alluvial hill), local topography (flat, undulating, or hilly), soil type (sandy, sandy gravel, sand with large gravel, sand with large rocks, or rock), general topography, and distance to nearest tree or shrub (see Appendices 3 and 4 for summaries).

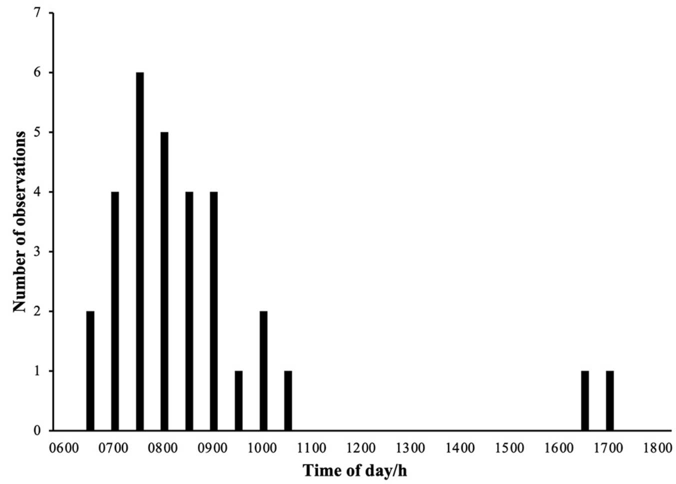

Figure 4. Times of day when Egyptian Spiny-tailed Lizards (*Uromastyx aegyptia*) were observed during this survey (July and August 2023) in the Mneifa Wildlife Reserve, NEOM, KSA.

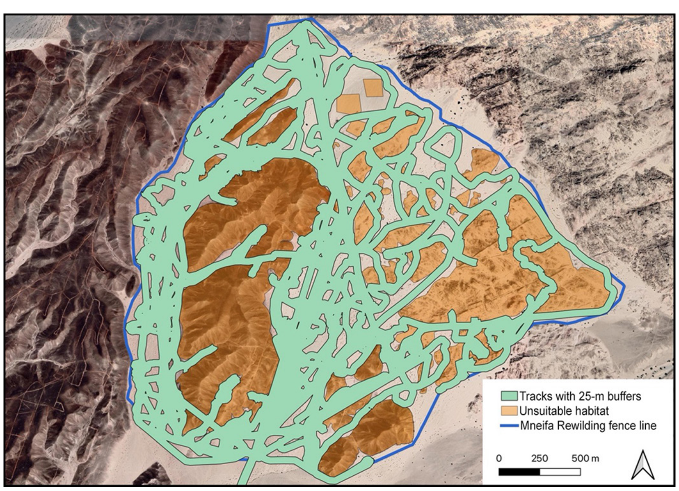

Figure 3. Google Map© showing survey tracks in the Mneifa Wildlife Reserve, NEOM, KSA, used to monitor Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) burrows. All tracks are shown within a 50-m diameter buffer to reflect the total area covered.

Assessing Survey Efficacy.—Burrow detection might have been biased in favor of larger burrows and burrows in more easily traversed terrain, factors that affect detectability. Similarly, burrows that were directly beneath trees or shrubs were less visible than those in open areas and were therefore less detectable. Effects of bias due to visibility issues were mitigated by calculating a margin of error for the number of active burrows based on a subsequent investigation evaluating survey-effort efficacy.

To evaluate survey-effort efficacy once all data had been recorded and all planned tracks travelled, we identified and resampled six 350-m stretches of previously travelled track in habitats identified as having the greatest potential for missed burrows due to dense vegetation or fields of rocks. Three of these transects were in bushy habitat; two were along a boundary of a rocky, alluvial hill; and one was along the edge of a rocky outcrop. These stretches were revisited on foot with every bush and every rocky section searched thoroughly for 20–30 minutes by a single trained observer, equivalent to more than doubling of the sampling effort in these areas. Any burrows that were encountered for the first time were recorded. We then calculated how many burrows might have been missed and incorporated this into a correction factor for our estimates of the number of mature STLs.

Burrow misclassification and undetected burrows could have resulted in inaccurate estimates of the number of active STL burrows. The misclassification of burrows accounted for much of the potential error. Classification of inactive burrows as active would lead to an overestimate of the true total number of active burrows, with the inverse leading to an underestimate. Counting burrows by animals other than STLs would lead to an overestimate while disregarding some STL burrows as not dug by STLs would lead to an underestimate. The assumption of a successful count (i.e., 100% of STL burrows were detected) would lead to an underestimate of the true number if any STL burrows present in the search area were not detected.

To mitigate the effects of these factors, we adhered closely to the activity criteria, noting, however, that some of those criteria can be affected by differences in substrate type. For example, tracks are much more visible in sandy substrate than in gravelly substrate. Confirmed active STL burrows were used as references to evaluate burrows that were unlikely to have been constructed by STLs, although this was sometimes difficult as non-STL burrows were often interspersed with those of STLs. Although we likely failed to detect some STL burrows, a 100% detection rate would require an inordinately intensive search, which would have required more time and resources than were available, especially in light of health and safety considerations as daytime temperatures can exceed 45 °C. Using the GPS points of all burrows detected in the survey, we conducted a nearest-neighbor analysis (NNA) in QGIS to reveal possible clustering of STL burrows.

Results

One year following translocation, we detected evidence of an established population. Adults, juveniles, and hatchlings were observed during the day. Presence also was inferred by the identification of 181 active burrows (Fig. 5). In addition, 62 inactive burrows were documented and one burrow that we were unable to categorize. Thus, 74% of burrows that we encountered were active. Burrow locations were widely distributed in favorable sand and gravel plain habitats and were entirely absent in rocky outcrops and rocky alluvial hills. Sandy substrate was most commonly associated with active burrows, accounting for 41% of all active burrows (75/181). Most burrows were recorded in areas where topography was flat (152/181 or 84%). All burrows were in the proximity of shrubs or trees. The farthest an active burrow was from a tree or shrub was 16 m (mean = 3.95 ± 3.49 m) (Fig. 6). The nearest neighbour analysis (NNA) showed that burrows had a mean nearest neighbour distance of 39.2 ± 41.2 m (NNA Index = 0.486; Z = -15.3), which was indicative of clustering.

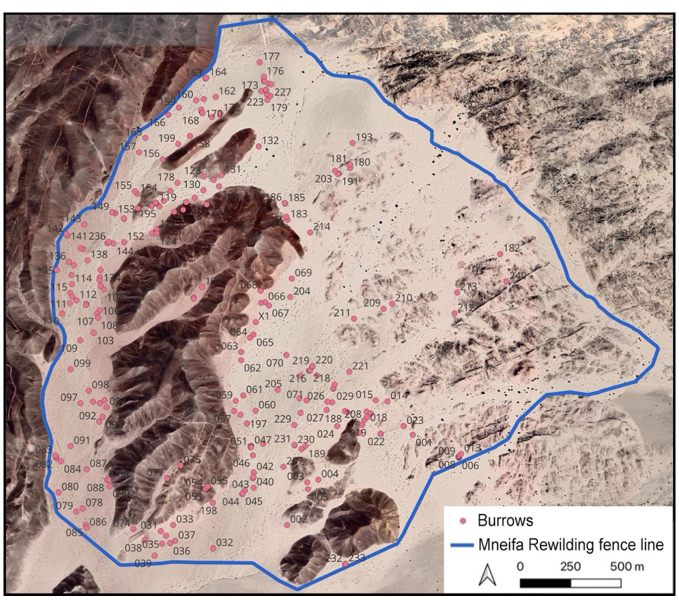

Figure 5. Google Map© showing locations of all Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) burrows located during July and August 2023 in the Mneifa Wildlife Reserve, NEOM, KSA.

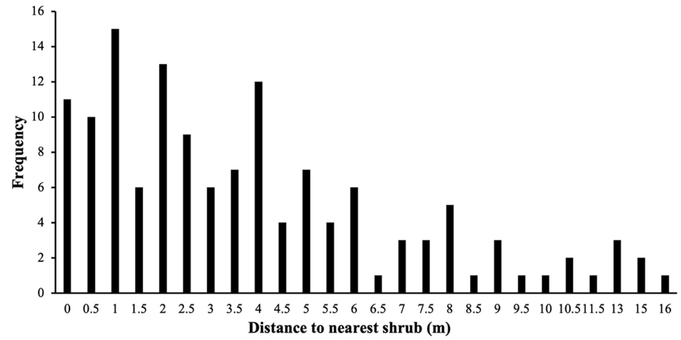

Figure 6. Frequency distribution of distances of Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) burrows to the nearest tree or shrub in the Mneifa Wildlife Reserve, NEOM, KSA.

During our further intensive surveys of the six transects, we found one additional active burrow in bushy habitat and one in rocky boundary habitat. This corresponds to an additional 22% of burrows that were missed in the initial count. However, since those sampled areas had a greater-than-average probability of missed burrows, the percentage of missed burrows in these areas could be higher than for the entire survey area. As a result, we determined that an additional 10% be added to the number of active burrows as a correction factor, bringing the total to 199 active burrows. Under our assumption that an STL uses an average of 1.2 burrows at any one time, we arrived at an estimate of approximately 156 mature STLs at MWR in July/August 2023. This estimate does not include some juveniles and hatchlings (both have been observed at MWR and some juveniles were released) that were not utilizing burrows.

Discussion

Although our survey methods were effective at determining the presence of STLs, they were less effective at calculating an accurate count of extant mature STLs. Nonetheless, they were capable of establishing a baseline of abundance that could be used for monitoring simple populations trends (i.e., stable, increasing, declining) over time within a particular release area. We were unable to derive a confident one-year survival rate by comparing the current STL estimate of 156 individuals with the translocated population size of 161. This was due to the uncertainty of our estimated number that was derived by using active burrows as a proxy for live STLs and that the 1.2 average burrow use by STLs is applicable at MWR. Nevertheless, we propose that NEOM's first STL translocation to MWR was successful as STLs are still present and becoming established in the MWR, whereas they were assumed not to be present or very rare prior to translocation. In addition, hatchlings have been observed, suggesting released STLs are reproducing.

Recommendations for Future Surveys.—In order to make recommendation regarding future surveys of translocated STLs, we evaluated the utility of different methods available for estimating abundance. These include distance sampling, plot/transect sampling, complete count, mark-recapture/mark-resight, and, potentially, indices of abundance. We believe that one of the most robust methods for this STL population would be mark-recapture. However, this type of survey was not practical for multiple translocation populations of hundreds of individuals.

Distance sampling was deemed unsuitable as several key assumptions were violated. When faced with small deviations from the assumptions, minor adjustments are possible (Borchers 2005). However, that was not possible in this case as the assumptions were violated to a degree that could not be rectified by such adjustments. Specifically, in distance sampling, one assumption is that objects with a distance of 0 from the observer are detected with a certainty of 1 (Buckland et al. 1993). In the case of STLs, a lizard inside a burrow with a distance of 0 from the observer would be undetectable. An additional assumption is that objects are detected in their initial location (i.e., animals did not move prior to detection in response to the observer) (Buckland and York 2002). STLs are cryptic and extremely wary of humans, so such movements likely are very common. This method also is impractical as the detectable proportion of STLs is usually very low because animals are in their burrows most of the time.

Complete counts are the most thorough and the most accurate determination of abundance as no estimation is necessary if a 100% detection rate is assumed. However, due to the wary nature and burrowing habits of STLs, observing every individuals would be impossible. In addition, the survey effort required to conduct a complete count of individual STLs in an area with about 160 translocated individuals would be impractical.

Eliminating those possibilities led us to using burrows as proxies for individual STLs, in effect as an index of abundance, providing information on population trends over time rather than actual estimates of numbers, combined with plot/transect sampling. We initially established a set of transects 100 m apart designed to cover the entire survey area. However, after a preliminary survey, we determined that this would not be feasible. Burrows would have to be detected at distances as far as 50 m, which a preliminary survey determined that the detection probability of a burrow was very low, largely due to the presence of vegetation or rocks that reduced detectability. One possible solution for this could have been to reduce transect spacing to 50 m, for example. This would have decreased the maximum distance at which burrows could have been detected to 25 m. However, this would have also doubled the number of transects from 28 to 56 and the total transect length from 57 km to 114 km. Travelling that transect length and making a thorough search within the timeframe available would not have been achievable.

The preliminary survey also highlighted the challenges of traversing the terrain, which was exacerbated by attempting to follow the straight lines of transects. However, because of the high proportion of challenging terrain within the survey area, this approach would have resulted in omitting large areas from the search. Our solution was to follow tracks ≤ 100 m apart and not straight and parallel like transects. This approach, although less systematic than transects, resulted in greater efficiency and search intensity and detection of more burrows. In addition, when the restriction of following straight line transects was removed, we were able to follow natural and established paths, searching areas with a greater focus on favorable habitats.

Using this method, we obtained a complete data set of coordinates of all located burrows, facilitating the testing of different sampling methods to evaluate optimum sampling effort. Plot sampling using plots randomly placed within suitable habitat was tested first (Fig. 7). Sampled habitat was

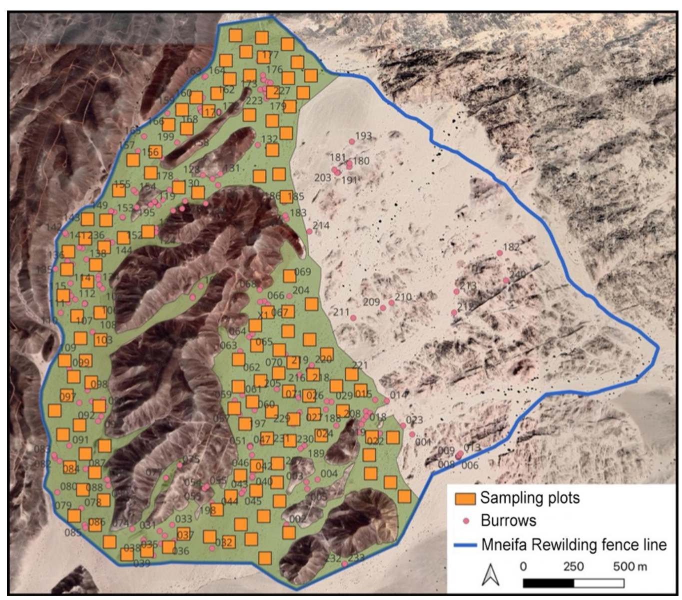

Figure 7. Google Map© showing sampling plots generated in QGIS representing 25% of suitable Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) habitat (sandy and gravelly plains indicated in green) in the Mneifa Wildlife Reserve, NEOM, KSA.

restricted to the western section of the survey area, primarily due to topography, which allowed for a greater proportion of that area to be sampled. Also, the habitat in that area was largely more favorable for STLs, reflected by 88% of all burrows in that section.

Different proportions of the total suitable habitat area (2.5%, 5%, 7.5%, 10%, 12.5%, 15%, 17.5%, 20%, 22.5%, 25%) were sampled three times using the same plot dimensions and the numbers of active burrows were recorded in each iteration. The percentage difference between the proportion of the total suitable habitat area covered and the proportion of the total burrows found was calculated (Fig. 8) and plotted against the proportion of suitable habitat covered

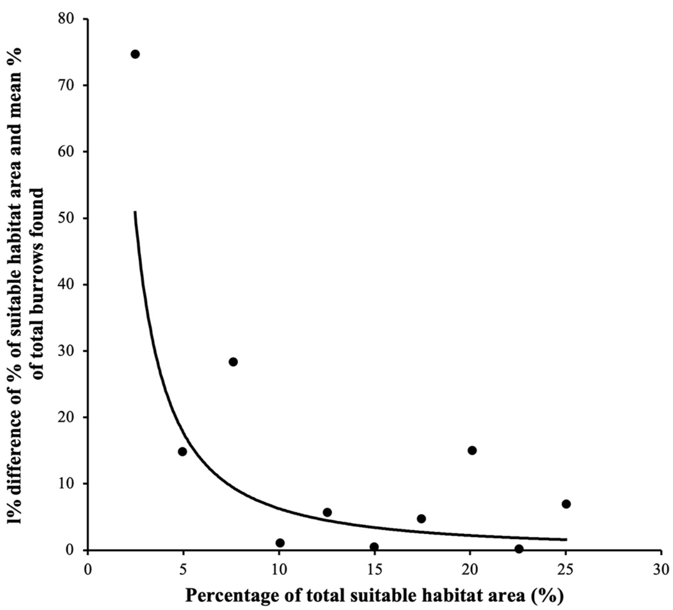

Figure 8. The relationship between the proportion of suitable habitat sampled and the proportion of burrows per percent of suitable habitat as the change in the absolute percentage difference between the proportion of area sampled and proportion of burrows sampled in that area. These data were generated in QGIS based on surveys of Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) burrows in the Mneifa Wildlife Reserve, NEOM, KSA.

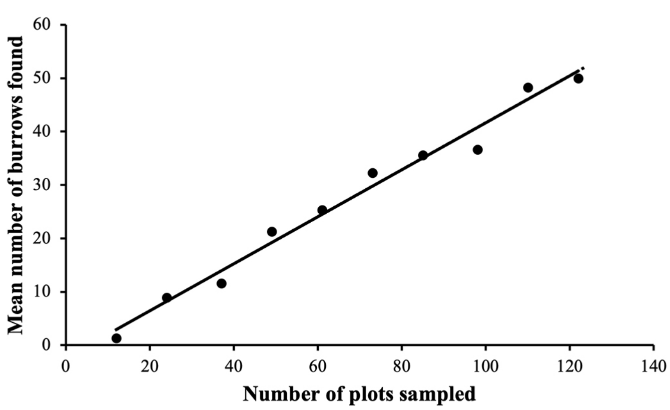

Figure 9. The number of plots surveyed for Egyptian Spiny-tailed Lizard (*Uromastyx aegyptia*) in suitable habitat in the Mneifa Wildlife Reserve, NEOM, KSA, and the number of burrows found in each (modelled in QGIS).

with a best-fit line (Fig. 9). When 7.3% of total suitable habitat was sampled, the percentage difference between proportion of suitable habitat sampled and proportion of total burrows found was 10%. That error can be reduced to 6.2% if 10% of the area is sampled. Assuming that the distribution of burrows in suitable habitat in the survey area is comparable to burrow distribution at other translocation sites, we can calculate the area that corresponds to the suggested 7.3% or 10% of total area that are sampled in future surveys. Sampling the mean of these two percentages (8.65% of total area) results in an error of 8%, which we deem to be acceptable. In this case, 8.65% of the total area equals 211,000 m^2.

While our survey efficacy testing used square plots, we recommend the use of strip transects because of a greater ability to detect environmental variation (e.g., presence of substrates favorable for burrowing). As a result, transects should yield more accurate estimates of total burrow counts, especially if burrows are clustered (Alrashidi et al. 2021). However, if the distribution of favorable habitat is complex (e.g., with many complicated rocky outcrops), one could substitute square plots of comparable survey area.

We recommend that only translocation release areas (i.e., the zone where STLs were released) are sampled when measuring trends in population. This is to reduce overall sampling effort if a translocation site is very large as well as increasing sampling effort in the area where active burrows are more likely to be present (provided that STLs were released into favorable habitat). If a release area contains unfavorable habitat, only favorable habitat should be sampled. Not only will conclusions from such sampling be stronger, sampling unfavorable habitat unnecessarily increases sampling effort. For example, provided that the same criteria for suitable and unsuitable habitat used in this survey are applied, one would expect a similarly low proportion of total burrows (2.9% or 7/244 burrows) to occur in unfavorable habitat. So, sampling unfavorable areas is unnecessary for tracking population trends.

We recommend strip transects with dimensions 250 x 20 m be used in surveys for active burrows. This reduces the distance travelled while maintaining a low strip width, which increases detection probability. The number of transects should be determined by the percentage of the total area that would need to be sampled with an error of < 8% (i.e., in this case, 42 transects of 250 m x 20 m were necessary to achieve the 211,000 m^2 mentioned previously). Transects should be dispersed throughout favorable habitat across a release area to encompass local variations in habitat. Orientation of transects should vary, for example, north to south, east to west, north-east to south-west, north-west to south-east, to avoid any bias resulting from environmental gradients.

To ensure that the widest range of habitats are sampled to detect potential environmental gradients, plots should be placed randomly or evenly but as widely separated as possible in a configuration that allows for them to be accommodated within the translocation release area.

In translocation release sites with high habitat heterogeneity or that would otherwise be too small to accommodate four 0.78 km x 0.78 km plots within favorable habitat, transects may be placed equidistant from one another in the suitable habitat area. Transects should be placed with a density of approximately 17 transects per 1 km^2 to reflect the transect density within the 0.78 km x 0.78 km plots. Transect orientation should still be varied as recommended. If the translocation release site is too small and/or heterogeneous in habitat type to accommodate the previous sampling method, as is the case in the MWR, a plot sampling method may be applied.

To determine the recommended plot sampling method, further GIS-based sampling method testing was performed. A Nearest Neighbour Analysis (NNA) was conducted on five different random configurations of 42 square plots, each of area equal to that of a 250 x 20 m transect (0.005 km^2), placed within the sample area. A mean distance of 187.2 m between center points was calculated. From this, we recommend that the distance between center points of plots should be approximately 187 m. Plots may be placed systematically for ease of implementation. Due to the relatively high density of plots in the sample area and the stipulation that plots may not intersect one another, the distribution of randomly placed plots is quite uniform. As such, we can assume that the use of evenly spaced plots would yield similar results to the random configuration used in the above testing.

We recommend that surveys for translocated STLs be conducted in the hot season (June to September) when lizards are most active and as close to one-year after release to allow for the population to disperse and adjust to the new conditions. Subsequent status surveys should be implemented annually for three years because the first four years post-translocation are when most failures occur (Bubac et al. 2019). This generally would allow for population status trends to be assessed, however, in the case of extreme weather (no rainfall in a given year) that could affect STL populations, the survey period could be extended. The ultimate goal is to assess whether the translocated STL population has become established.

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Appendix 1. Key Assumptions.

Carrying Capacity of STLs at MWR.—A survey of active STL burrows at Mahazat as-Sayd Protected Area in central KSA located 206 active burrows in 4 km^2 (Wilms et al. 2009). This reserve is comprised of flat sandy plains with shrubs, which is favorable habitat for STLs. This is roughly a carrying capacity of 50 STLs per km^2. The MWR has 3.42 km^2 of flat, sandy (favorable) habitat available and one can estimate an STL carrying capacity at roughly 171 mature STLs. Note that some STL burrows occur in rocky alluvial hills in Mneifa, therefore, our estimated carrying capacity might be low.

Mature STLs cannot disperse from the fenced MWR.—Only smaller juvenile STLs are capable of dispersing through the galvanized mesh fence. Thus, we assume that the majority of mature STLs remained in the MWR. However, if mature STLs burrow under the fence they can leave the area, but we have not documented any STLs moving out of the fenced relocation area.

Some natural predation occurs.—We observed a mature STL carcass. Arabian Red Foxes (*Vulpes vulpes arabica*), possibly Rüppell's Foxes (*Vulpes rueppellii*), and Arabian Wildcats (*Felix silvestris gordoni*) in the MWR are likely to prey on STLs. Raptors and owls also have been observed in the Reserve. No feral dogs or cats, Arabian Wildcats, or Arabian Wolves (*Canis lupus arabs*) have been observed inside the fenced area.

D. *Mature STLs use multiple burrows and do not share burrows*.—While no studies have addressed the use of multiple burrows by STLs, prior observations suggest that mature STLs may utilize two or more burrows at the same time (Wilms et al. 2010). Given that we are using the number of active burrows as a proxy for the number of mature STLs present in the Reserve, the average number of burrows used is important in the estimates of STLs present. We used an estimate of 1.2 burrows used by individual STLs (Wilms et al. 2010). We do not know if this usage differs between males and females. We also assume that adult STLs do not share burrows, although this remains to be confirmed. The five STL burrows excavated by Wilms et al. (2010) only had one entrance, although unconfirmed reports suggest that a single burrow can have multiple entrances, with the primary entrance typically facing east. We herein assume that one burrow has one entrance.

E. *The favorability of habitat at the MWR will improve over time with restoration*.—Currently, the natural vegetation of MWR has not been fully restored following the cessation of intensive grazing by livestock two years previously. Furthermore, native herbivores have been introduced in November 2022, and little rain has fallen during the last year. Thus, the MWR is likely able to sustain more STLs than are currently present once the natural vegetation is restored.

F. *Monitoring the STL population at MWR after a year has allowed for equilibration*.—Approximately one year has passed since the major translocations of STLs (June/July 2022). During this time, STLs likely have dispersed throughout all favorable habitat in the MWR, some juveniles might have dispersed outside of the mesh fencing, some STLs have been eaten by predators, some juveniles have grown and are now considered mature, and mature STLs have established a distribution within a natural range of variation (see Alrashidi et al. 2021; Kechnebbou et al. 2019a, 2019b, 2025). Thus, we assume the population has had time to equilibrate to the new environment to some extent and the one-year post-translocation survey offers a more accurate reflection of how STLs have become established at the new site than a survey conducted sooner after the release of lizards into the area.

G. *Conducting field surveys in the hot season (June–September) enhances accuracy*.—Given that STLs are most active during the hot season, STL presence/establishment surveys at translocation receptor sites should be conducted during that season. For example, burrow entrances may not appear active in the cool season even if an STL is present in the burrow.

H. *All active burrows were sampled*.—We assume that all burrows were sampled for the purposes of estimates. We carried out a sampling efficacy analysis to test this assumption.

I. *Very few to no STLs were present within or adjacent to the MWR prior to STL translocation*.—While the MWR was being developed, Nature Reserve staff and MWR managers did not observe any STLs or active burrows either inside or near the MWR. Prior to the Reserve being established, this area was heavily degraded by intensive livestock (camels, goats, donkeys) grazing, and local residents are likely to have hunted mature STLs. Prior to activation of the MWR, many livestock camps were in the area and vehicular traffic was common, so few STLs are likely to have gone unnoticed and escaped hunting.

J. *Changes in active burrow counts are strongly correlated with trends in the population size of mature STLs.*—Active burrows reflect recent lizard activity suggesting that lizards are alive and behaving naturally; this includes adult STLs using the average number of burrows determined in prior studies (Wilms et al. 2010). Under these assumptions, any major changes observed in active burrow counts between surveys effectively track gross changes or relative stability in local STL populations.

K. *The density and distribution of burrows in the MWR is similar to that found in favorable habitats of other areas where STLs have been released within NEOM.*—While some variation is likely among the number of burrows dug by STLs in different substrates (e.g., fewer in rocky plains versus loamy-sandy substrates), within favorable habitats (i.e., flat plains with loose gravel and coarse sand substrates) of NEOM, burrow densities and distributions will not vary sufficiently to compromise trend estimates. This is because we assume that dispersion of individual STLs and active burrows within favorable habitat will therefore be similar to that of the MWR one-year post-translocation.

Appendix 2. What Was Not Measured.

We did not: (A) Attempt a complete direct count of individual STLs due to inherent challenges (see above); (B) evaluate the survival of individual lizards; (C) measure age, size, and sex structures of the population in the MWR except for the lizards at the time of release; (D) assess movements of lizards and patterns of burrow use; (E) measure reproduction, other than anecdotal observations of hatchlings, or movement of lizards outside the fenced MWR other than anecdotal reports; (F) assess mortality, other than an anecdotal observation of fox predation on an STL.

Appendix 3. Testing for Trends in STL Burrow Orientation.

Chi-squared test of the distribution of burrow orientation. H_0 = The observed frequencies of burrow orientations and the expected frequencies of burrow orientations that cannot be explained by chance will not differ significantly. H_1 = The observed frequencies of burrow orientations and the expected frequencies of burrow orientations will differ significantly (df = 8–1 = 7). Critical value at significance level 0.01 = 18.48. 23.61 > 18.48, therefore H_0 is rejected at the 0.01 significance level. Burrows were more frequently oriented toward the west and northwest. The reasons for this are uncertain but could be related to prevailing winds and orientation relative to the sun for temperature regulation.

Burrow Orientation	Expected frequency (E)	Observed frequency (O)	O-E	$(O-E)^2$	$(O-E)^2/E^2/E$
N	30.5	24	-6.5	42.25	1.3852459
NE	30.5	23	-7.5	56.25	1.8442623
E	30.5	24	-6.5	42.25	1.3852459
SE	30.5	28	-2.5	6.25	0.2049180
S	30.5	28	-2.5	6.25	0.2049180
SW	30.5	24	-6.5	42.25	1.3852459
W	30.5	49	18.5	342.25	11.2213110
NW	30.5	44	13.5	182.25	5.9754098
Total	**244**	**244**	**0**	**720**	**23.6065570**

Appendix 4. Burrow metric frequencies.

The frequency of soil type at burrow entrances.

Substrate type	Frequency
Sandy	75
Sandy gravel	68
Sandy with large gravel	28
Sandy with large rocks	4
Rock	6

Frequency of topography type at burrow entrances

Topography	Frequency
Flat	152
Undulating	15
Hilly	14

Appendix 5. Additional information on STL diet compiled from Cunningham (2001) and Bouskila (1986). All plants are native to the KSA. Dashes (—) indicate minor components of the diet.

Species	Dietary component
Pennisetum divisum	Major
Stipagrostis plumosa	Major
Aerva javanica	—
Leptadenia pyrotechnica	—
Moltkiopsis ciliata	—
Heliotropium kotschyi	—
Haloxylon salicomicum	—
Citrullus colocynthis	—
Taverniera cuneifolia	—
Polygala erioptera	—
Fagonia sp.	—
Zygophyllum sp.	—
Aristida plumosa	—
Astragalus gyzensis	—
Horwoodia dicksoneae	—
Launaea capitata	—
Neurada procumbens	—
Plantago boissieri	—
Vachellia tortilis	Major
Hammada salicornica	Major
Aaronsohnia faktorovskyi	Major
Zygophyllum simplex	—
Plantago ovata	—
Erodium bryoniipholium	—
Erucaria boveana	—
Echium rauwolfii	—
Launaea angustifolia	—
Trigonella stellata	—
Launaea nudicaulis	—
Pulcaria undulata	—