



A targeted study to determine the conservation status of a Data Deficient montane lizard, the Eungella shadeskink (*Saproscincus eungellensis*)

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ABSTRACT

Context. Combatting biodiversity loss is often hamstrung by a lack of species-specific knowledge. Species considered Data Deficient (DD) on the IUCN Red List are poorly understood and often neglected in conservation investment, despite evidence they are often threatened. Reptiles have the highest percentage of DD species for any terrestrial vertebrate group. **Aims.** We aimed to assess the conservation status of the DD Eungella shadeskink (*Saproscincus eungellensis*), which is endemic to Eungella National Park, Queensland, Australia. **Methods.** A combination of a targeted field survey, ecological studies, and species distribution modelling were used. **Key results.** *Saproscincus eungellensis* typically occurred within 25 m of streams, at elevations between 700 and 1000 m. The species is thigmothermic, with a low active body temperature (~23–26°C) and was predominantly observed on rocks and fallen palm fronds. The species has a highly restricted distribution with an estimated Area of Occupancy of 36 km² and Extent of Occurrence of 81.7 km², comprising one location (defined by the threat of climate change) with an estimated 16,352–52,892 mature individuals. The main threats are fire, invasive alien species and climate change, with the species forecast to lose all suitable habitat by 2080 under all climate change scenarios. **Conclusions.** The species meets listing criteria for Critically Endangered under Criterion B of the International Union for Conservation of Nature. **Implications.** Our results support recent studies indicating that some DD species are highly threatened. Our approach provides a template for conducting targeted studies to determine the conservation status of DD species, especially those with restricted ranges.

Keywords: climate change, Eungella National Park, extinction risk, fire, invasive species, IUCN Red List, Scincidae, SDM, species distributional modelling.

Introduction

The Data Deficient (DD) category of the International Union for Conservation of Nature (IUCN) Red List is considered the most controversial and misunderstood (Butchart and Bird 2010). The IUCN discourages the liberal use of this category, and recommends that Data Deficient species receive the same research attention as species listed as threatened (Morais *et al.* 2013; Bland 2014). However, in practice this is rarely followed, with conservation investment, planning and assessments generally neglecting Data Deficient species (Jarić *et al.* 2016; Bland *et al.* 2017). Alarming, studies have shown that numerous DD species may be more threatened than data sufficient species (Parsons 2016; González-del-Pliogo *et al.* 2019; Gumbs *et al.* 2020; Borgelt *et al.* 2022). Many governments consider the DD category similar to the Least Concern category, effectively placing these species out of sight and out of mind (Parsons 2016; Braby 2018). However, if a key aim of conservation is to protect species from declines or extinction, we need to be proactive, rather than reactive, and take steps towards early identification of threats, declines in abundance, and contracting distributions (Lindenmayer *et al.* 2011; Martin *et al.* 2012). Given the large

number of species listed as DD, it is increasingly important to prioritise research to remove them from this problematic category (Jarić *et al.* 2016).

Globally, high-elevation ecosystems, and especially montane forest biomes in tropical and subtropical areas, are of high conservation concern. In these areas, the rapid compression of climatic zones and habitats over elevational gradients have catalysed speciation, and ultimately, resulted in high species richness and endemism (Williams *et al.* 2003; Ashton *et al.* 2016). Species in these biomes often exhibit altitudinal stratification and very small ranges due to their thermal adaptations (Ashton *et al.* 2016; Hamilton *et al.* 2021). Such specialist species tend to have increased extinction risk, with a small population size and a restricted geographical range causing a synergistic double jeopardy (Colles *et al.* 2009; Williams *et al.* 2009). In the face of warming climates, mobile species specialising in high-elevation biomes may adjust their ranges through migration to neighbouring cooler latitudes. However, lowland warmer, and often drier, habitats separating mountains can severely restrict species immigration, especially for species that also have limited mobility (Laurance 2015; Ashton *et al.* 2016). Where mountains are isolated, low mobility species can typically only respond to warming by moving up in elevation (Peters and Darling 1985). For species restricted to the tops of mountains, this is expected to result in progressively smaller ranges over time, until the required climatic envelopes reach the elevational limits and extinction follows (Colwell *et al.* 2008; Román-Palacios and Wiens 2020). Encroachment of lower-elevational predators, competitors, and pathogens may also increase extinction risk (Walther *et al.* 2002; Seastedt and Oldfather 2021).

Montane species in regions with relatively low topographic relief, such as Australia, have few options to adjust their ranges in response to climate change (Hughes 2011; Torkkola *et al.* 2022a, 2022b). Australia is the world's flattest continent (Ebach and Michaux 2020) with the lowest mean elevational gradient and 99% of its land area under 1000 m altitude (Hughes 2011). Nonetheless, montane biomes across the continent have high levels of endemic and specialist species, all of which have been predicted to be at a particularly high extinction risk (Williams *et al.* 2003; Hughes 2011; Hagger *et al.* 2013; Torkkola *et al.* 2022a).

Reptiles are the most species-rich group of terrestrial vertebrates globally (Cox *et al.* 2022; Meiri *et al.* 2023). However, their conservation status is more poorly understood than that of other terrestrial vertebrate taxa, with reptiles holding both the highest percentage and highest raw numbers of DD classified species on the IUCN Red List (Cox *et al.* 2022; Meiri *et al.* 2023). Australia is a global hotspot of reptile diversity (holding ~10% of the world's lizards and snakes [vs 5% of the world's land area]; Tingley *et al.* 2019), including an exceptional diversity of skinks (~460 species) (Squamata: Scincidae; Wilson and Swan 2020; IUCN 2024a). Many Australian skinks are listed as DD (17

of 444 assessed species) and have not been studied since their description (Chapple *et al.* 2019), highlighting significant knowledge gaps in skink ecology (e.g. population sizes, geographical ranges, trends, and threats; Chapple *et al.* 2019). With current trends in biodiversity loss Australia stands to lose significant proportions of skink diversity (Cox *et al.* 2022; Graham *et al.* 2023) and Chapple *et al.* (2021) found that 20% of skink species globally are threatened by extinction. Recent field-based studies have revealed three previously DD Australian skink species are threatened (DCCEE 2023a, 2023b; Farquhar *et al.* 2023a; Graham *et al.* 2023; L. Bonifacio, J. E. Farquhar, A. Pili, J. Walsh, D. G. Chapple, unpubl. data). Thus, urgent targeted field research is required on other DD Australian skink species to reveal their true conservation status.

The Eungella shadeskink (Scincidae: *Saproscincus eungellensis*, Sadlier *et al.* 2005) (Fig. 1) is restricted to high elevations in the Clarke Range, with all but one observation from the Dalrymple Heights area within Eungella National Park (herein Eungella NP) of central-eastern Queensland (Fig. 2). It was described in 2005 (Sadlier *et al.* 2005) from 14 voucher specimens with limited ecological data, although an association with upland rainforest streams was apparent. The species has not received any targeted research effort since, although it was sporadically recorded during frog surveys reported by Meyer *et al.* (2020) (H. B. Hines, unpubl. data), including the only additional specimen collected since its description. Thus, it was assessed as DD by the IUCN Red List (Hoskin *et al.* 2018) and is not listed under the federal *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). It is listed as Vulnerable under the *Queensland Nature Conservation Act (1992)*. It shows traits that are positively correlated with extinction risk (e.g. narrow-range, high altitude) and lives in an area impacted by multiple threats (Sadlier *et al.* 2005; Hines *et al.* 2020), raising the possibility that it is threatened, with Sadlier *et al.* (2005) and Wotherspoon *et al.* (2024) considering it to meet IUCN Red List criteria for Endangered.



Fig. 1. Adult *Saproscincus eungellensis*, Eungella National Park, Queensland (Photo: Jordan Mulder).

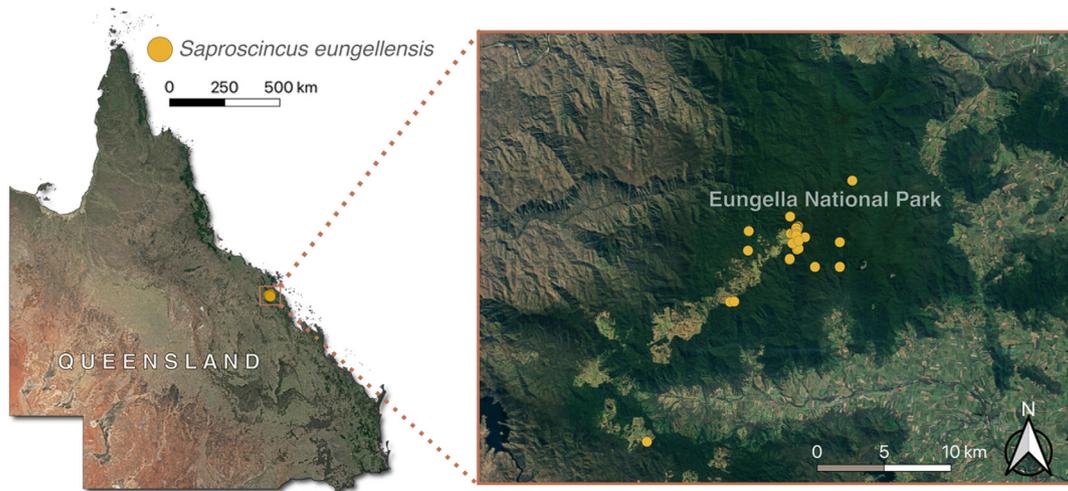


Fig. 2. The geographical range of *Saproscincus eungellensis* within the Eungella study area (using unvetted ALA (Atlas of Living Australia) 2023 records), prior to our study, relative to Queensland, Australia.

Climate change and the associated increased risk of severe drought, heatwaves and fire (Climate Change in Australia 2020) are likely to pose significant conservation risk to *S. eungellensis*. The species has limited dispersal capabilities, and the nearest climatically similar habitats are hundreds of kilometres away. They occur close to the maximum elevation of Eungella NP (1250 m a.s.l.) raising the possibility that changing climate may leave the species with no available habitat (Sadlier *et al.* 2005; Ashton *et al.* 2016). Further, closely related *Saproscincus* species display low critical thermal maxima (CTMax) compared to other Australian skink species, suggesting a high vulnerability to increasing temperatures or loss of cool rainforest microhabitats (Table 1; Greer 1980, 1989; Sadlier *et al.* 2005; Muñoz *et al.* 2016). Severe drought leading to extreme fire has also already extensively impacted fire-sensitive rainforests at Eungella NP (Hines *et al.* 2020).

Table 1. Critical thermal maximum (CTMax) temperatures for *Saproscincus* species, and other high elevation skink species in North Queensland.

Species	CTMax (°C)	Source
<i>Saproscincus hannahae</i>	35.5	Greer (1980)
<i>Saproscincus mustelinus</i>	38.6	Greer (1980)
<i>Saproscincus tetradactylus</i>	35.52	Muñoz <i>et al.</i> (2016)
<i>Saproscincus czechurai</i>	37.75	Muñoz <i>et al.</i> (2016)
<i>Saproscincus basiliscus</i>	38.04	Muñoz <i>et al.</i> (2016)
<i>Carlia rubigularis</i>	43.31	Muñoz <i>et al.</i> (2016)
<i>Gnypetoscincus queenslandiae</i>	36.85	Muñoz <i>et al.</i> (2016)
<i>Lampropholis similis</i>	42.57	Muñoz <i>et al.</i> (2016)
<i>Lampropholis bellendenkerensis</i>	43.05	Muñoz <i>et al.</i> (2016)

Here we conduct a targeted field study and niche modelling (e.g. Le Breton *et al.* 2019) for *S. eungellensis* in order to better understand its distribution, thermal requirements and conservation status. Specifically, we aim to (1) determine the current geographical distribution (i.e. Extent of Occurrence [EOO], Area of Occupancy [AOO]), and elevational limits of *S. eungellensis*; (2) estimate population size, (3) better understand the species' ecology and thermal biology; (4) model potential future distribution under various climate scenarios; (5) evaluate known and potential threats; and (6) assess conservation status using the IUCN Red List Guidelines.

Materials and methods

Field surveys to determine the habitat preferences of *S. eungellensis*

To assess the influence of time of day, temperature, and solar radiation on the activity times of *S. eungellensis*, we conducted systematic visual surveys ($n = 56$) from 18 November to 6 December 2023. Four 150 m long transects were established along two high-elevation tributaries of Cattle Creek (900–1000 m a.s.l.) at Eungella NP known to support the species (Supplementary material Fig. S1). We conducted 60 min visual surveys along these transects during four time windows (09:00–10:00, 11:00–12:00, 13:00–14:00, 15:00–16:00 hours AEST). Each site was surveyed on three separate occasions for every time window (i.e. a total of 12 surveys per site). The order of surveys was randomised with respect to site and time window. Active searches involved two researchers slowly traversing the transects in both directions and visually scanning habitat for lizards. Once detected, the time and location of each lizard observation was recorded on EcoGIS (iPhone app, UgMedia). For each *S. eungellensis*

observation, microhabitat temperature was recorded using a digital thermometer (Omega HH508 digital thermometer, Omega Engineering Inc.) with a K-type thermocouple probe placed approximately 5 cm above the substrate the lizard was observed on. The temperature of the substrate at occupied sites (T_{surf}) and the maximum temperature (T_{surfMax}) of substrate found within a 1 m radius of the lizard was recorded using an infrared temperature gun (Fluke 566 IR Thermometer). To assess the species' relationship to open water, distance from water was recorded (in m) for each individual using a tape measure. Microhabitat preference was recorded as the perch substrate the lizard was first observed on (wood = fallen coarse woody debris, rock, litter, palm = fallen palm frond, flood debris).

Saprosincus eungellensis individuals were hand captured where possible, and field-selected body temperatures (T_b) were recorded within 10 s of capture using a digital thermometer with a K-type thermocouple probe inserted approximately 5 mm into the cloaca. Sex was determined through eversion of hemipenes in males and inspection of females for obvious signs of gravidness. Once caught, the following linear morphological variables were measured using digital callipers (Mituyo 0–200 mm) and a ruler: snout-vent length (SVL), head length, head width, head depth and tail length. Upon each unique capture, lizards were marked on each flank using non-toxic white paint (Born Inc. acrylic paint) to allow re-captured animals to be recorded through the study period. Life stage (juvenile, adult), and lizard colour (light, dark) was also recorded for all observations.

In addition to these four creek-line survey sites we conducted surveys at four transects ~30 m upslope (Fig. S1) from each of them (for a total of eight transects) to assess the species' use of denser rainforest habitat at short distances from its assumed preferred creek habitat. These surveys followed similar survey procedures as the creek line sites (150 m transect at each site, surveyed by two researchers for 60 min) except that these rainforest sites were surveyed on only two occasions each, in the two time windows that lizards were found to be most active (11:00–12:00, 13:00–14:00 hours AEST).

To quantify solar radiation experienced during surveys a solar radiation pyranometer (SR05-A1 solar meter, Hukseflux Inc.) and data logger (LI19 handheld data logger, Hukseflux Inc.) were deployed in a standard position in every transect during each survey and set to record incident solar energy (W/m^2) for every minute of each survey period (Fig. S2).

Distributional surveys to determine elevational extent and habitat associations

To clarify the distributional and elevational extent of *S. eungellensis* in Eungella NP we conducted timed visual surveys ($n = 6$) from the 7 December to 15 December 2023. We undertook 4–5 h visual surveys between 09:00 and 16:00 hours AEST along each of the following creek lines:

Cattle Creek high elevation (800–920 m a.s.l), Cattle Creek low elevation (100–200 m a.s.l), Cattle Creek mid elevation (620–750 m and 620–590 m a.s.l) and Broken River (690–740 m a.s.l) (a catchment with a single previous non-specimen backed record of the species – Sadlier *et al.* 2005). Active searches involved two researchers traversing the creek lines scanning habitat for lizards. If a *S. eungellensis* was observed, thermal and morphological data were taken as described above. We also included any additional records of *S. eungellensis* detected incidentally when traversing to/from sites.

Microhabitat use and thermal biology

At each of the survey sites, vegetation density was measured vertically using the touch pole method (e.g. Lees *et al.* 2022) with a 1.8 m touch pole separated into three height brackets (0–50, 51–100, 101–180 cm). Every 10 m along each transect the touch pole was stuck into the ground and all vegetation touches for each height bracket were recorded. No attempt was made to distinguish between live and dead plant matter (Godínez-Alvarez *et al.* 2009). The substrate type (litter, rock, water, bare soil, wood, palm, or flood debris) beneath the pole at each position was recorded (Godínez-Alvarez *et al.* 2009). At each 10 m intercept we also sampled at three points either side of, and at right angles to, the transect line at 1 m intervals, making a total of seven touches/intercept and 105 touches/site. To quantify canopy openness every 20 m along the transect a photo was taken of the canopy from a height of approximately 140 cm and analysed using Percentage Cover (iPhone app, Mark Mignanelli). The canopy cover percentages were then averaged across the transect.

Quantifying thermal regimes of habitat

We deployed 32 iButton temperature data loggers (Thermocrom DS1921G, Maxim, San Jose, USA) set to record hourly ambient air and refuge temperatures ($T_a \pm 0.5^\circ\text{C}$) from 19 November to 15 December 2023. At each transect ($n = 8$), four iButtons were deployed, each in a zip-lock bag, which was then taped to the end of a thin wooden stake. For air temperature the stake with the iButton approximately 40 cm above the ground was inserted in the creek line near where lizards had been observed (see Fig. S3). Within 0.5–5 m, a second iButton was inserted into a potential refuge (in a rock crevice, beneath a boulder, or under a log). This paired deployment was repeated at a second location on the transect approximately 100 m away. As *Saprosincus* species are primarily diurnally active, only daytime temperatures (07:00–17:00 hours AEST) were considered for analysis.

To quantify spatial variation in thermal environments within the Cattle Creek catchment of Eungella NP we similarly deployed eight iButtons across two lower elevation tributaries (elevation 270 m and 735 m) from 25 November to 12 December 2023. At each site, two paired air-refuge iButton

stations were established approximately 200 m apart along the creek line. Sites were chosen based on having similar characteristics to the higher elevation tributaries of Cattle Creek (i.e. rainforest, rocky creek).

Statistical analyses

All analyses were conducted in the R statistical environment (2023.12.1 + 402; R Core Team 2023) unless stated otherwise. A principal component analysis (PCA) was performed using the packages *FactorMineR* (v2.9; Lê *et al.* 2008) and *factoextra* (v1.0.7; Kassambara and Mundt 2020) to visualise patterns of habitat change between the sites (creek and rainforest). Transect touch counts of palm and flood debris were excluded from the model due to infrequent occurrence (<30 observations out of a total of 2381). PC loadings were inspected and the Kaiser-Guttman criterion (eigenvalue >1) was used to determine PCs for further analysis (Jackson 1993). We used permutational multivariate analysis of variance (PERMANOVA; Anderson 2017) with the package *vegan* (R Core Team 2023) on the scores of PC1 to assess differences in habitat characteristics between the sites. Euclidean distance in the PERMANOVA analysis was used, as many habitat characteristics were correlated with each other (Grady *et al.* 2013).

To assess which variables affect T_b , we applied a linear mixed effect model where T_b was the response variable, survey number was used as the random factor, and SVL, mass, T_{surf} , $T_{surfMax}$, air temperature, average refuge temperature and solar energy were considered as fixed effects (Sannolo *et al.* 2014; Ortega and Pérez-Mellado 2016). A saturated model with all potential explanatory variables was considered first. Akaike Information Criterion (AIC; Burnham and Anderson 2004) was then used to remove non-significant variables and interactions ($P > 0.05$) in a stepwise procedure (Ortega and Pérez-Mellado 2016). The simplest significant model was then chosen to explain the observed active body temperatures of *S. eungellensis*. We tested whether *S. eungellensis* showed between-sex differences in morphological variables by conducting a series of one-way ANOVAs using each morphological feature as dependent variables and sex as the factor with the *car* (Companion to Applied Regression) package. Prior to analysis, parametric assumptions were tested and, where morphological variables deviated too far due to small sample sizes, a Wilcoxon rank sum test was used instead.

To determine if ambient temperatures in the air and in refuges differed across or among sites or elevations, *t*-tests were conducted using the *car* (Fox and Weisberg 2019) package.

The AOO and EOO of *S. eungellensis* were estimated using the Geospatial Conservation Assessment Tool (GeoCAT; Bachman *et al.* 2011). Existing records were downloaded from the Atlas of Living Australia (ALA (Atlas of Living Australia) 2023), and vetted for locational accuracy and combined with unpublished records from the frog survey dataset of Meyer *et al.* (2020) and our 2023 field surveys

(Fig. S4). GeoCAT estimates the EOO by calculating a convex hull defined as the smallest polygon in which no internal angle exceeds 180° and contains all sites of occurrence, and the AOO by calculating the sum of 2 km^2 grid cells containing the sites of occurrence (Bachman and Moat 2012).

Population size was estimated using QGIS (QGIS Development Team 2022). As our study confirmed that *S. eungellensis* is highly dependent upon upland rainforests (>700 m a.s.l.), we created a shapefile of drainage lines above 700 m asl, buffered by 10 m. KML files containing wetland information were downloaded from the Queensland Government website (Department of Environment, Science and Innovation, Queensland 2024a, 2024b) and converted into a shapefile. Drainages were clipped to the NP boundary, as this excludes cleared and heavily disturbed areas outside the park where the species is not known and unlikely to occur. As our quantitative assessment showed a marked increase in relative abundance of *S. eungellensis* above 950 m we stratified buffered streams in two groups, those above 950 m and those between 700 and 950 m. The area of the buffered creek lines for both shape files was then calculated. The number of mature individuals was calculated using the following IUCN endorsed formula: $d \times A \times p$, where d is an index of population density, A is an estimate of area and p represents the proportion of individuals that are mature (Table 2; IUCN 2024b). Using the highest and lowest number of *S. eungellensis* observed at the creek line survey sites (>950 m a.s.l.), we calculated a high and low range population estimate. For the 700–950 m a.s.l. range, we calculated only one estimate because of the low number of individuals observed in this elevational range. We then used the number of *S. eungellensis* found during our surveys to determine the density for this area using a simple equation: we calculated the estimate based on the assumptions that (1) *S. eungellensis* occur in all creek lines above 700 m in Eungella NP, (2) species density differs between the two elevation bands, and (3) the species' core habitat is restricted to creek lines.

Species distribution modelling

We predicted the potential distribution of *S. eungellensis* using the Maximum Entropy species distribution modelling algorithm (MaxEnt, Phillips *et al.* 2006, adapted from Farquhar *et al.* (2023a); refer to Figs S5 and S6 for the modelling

Table 2. Variables used for calculating *Saproscincus eungellensis* population estimate.

Variable	>950 m high range	>950 m low range	700–950 m
d	17,333	4667	379
A	6.01 km^2	6.01 km^2	15.88 km^2
p	0.48	0.48	0.48

d , index of population density; A , estimate of area; p , the proportion of individual that are mature.

background). We nominated a suite of 27 prospective predictor variables (Supplementary Table S1). Because this species is an ectotherm, and is likely highly affected by temperature and precipitation regimes (Powney *et al.* 2010; Farquhar *et al.* 2023a), we therefore primarily selected bioclimatic variables. The environmental predictor variables were resampled from a spatial resolution of ~ 1 km² to match the resolution of the environmental layers using the *mask* function in the *raster* package (Hijmans *et al.* 2005).

To minimise multicollinearity between predictor variables, we inspected the relationship of the predictor variables through principal components (PC; Fig. S7) and pair-wise Pearson correlation coefficients. We retained variables with the likely greatest ecological relevance for *S. eungellensis* with a Pearson correlation coefficient < 0.7 : Bio1 = Annual mean temperature (°C); Bio6 = Minimum temperature of coldest month (°C); Bio7 = Annual temperature range (°C); Bio12 = Annual precipitation (mm); Euclidean distance to watercourses (Fig. S7).

To compensate for uneven sampling effort (both current surveys and historical records) across the modelling region and spatial non-independence of species occurrence points, we reduced spatial clustering by thinning *S. eungellensis* records so there were no records less than 1 km distant from each other using the *thin* function of the R package *spThin* (Aiello-Lammens *et al.* 2015), resulting in 12 occurrence points. Due to the small extent of the Eungella study area, we then sampled 1150 background points weighted against the likelihood density values of the occurrence points. Thus, background data points were selected with approximately the same sampling bias as the occurrence points (Kramer-Schadt *et al.* 2013; Farquhar *et al.* 2023a).

MaxEnt model calibration was run using *ENMevaluate* function of the *ENMeval* package (v2.0.3; Muscarella *et al.* 2014; Kass *et al.* 2021) which runs MaxEnt models across multiple combinations of feature classes and values of the regularisation multiplier to allow comparisons of model performance. We selected the MaxEnt combination that balanced model fit and predictive ability. Models were built with regularisation multiplier options set to 1, 2, and 3 and with three varying feature class combinations (1. Linear; 2. Quadratic; 3. Linear + Quadratic); resulting in nine parameter combinations.

The MaxEnt models were evaluated using metrics AICc, AUC_{TEST}, AUC_{DIFF}, OR_{MTP}, OM₁₀ and Boyce Index (see Table S2 for definitions and explanations of metrics). We visually inspected the predictions of each model and evaluated the validity of each model with our knowledge of the ecology of *S. eungellensis* and of the Eungella region. We chose the parameter combination that returned an optimal MaxEnt model – that is, the parameter produced a model that performed well against multiple metrics and generated ecologically plausible predictions for *S. eungellensis* (see Farquhar *et al.* 2023a for full modelling approach). The model output was used to produce a map of potentially suitable pre-clearing habitat.

Projecting future suitable habitats

To assess the potential effects of future climate change on the distribution of *S. eungellensis*, we forecast pre-clearing habitat suitability, under different climate change scenarios using the Coupled 106 Model Intercomparison Project Phase 6 (CMIP6) multi-model (Eyring *et al.* 2016; Meinshausen *et al.* 2020; Masson-Delmotte *et al.* (2021)) and Euro-Mediterranean Centre on Climate Change coupled climate model CMCC-CM2, (Cherchi *et al.* 2019; Ortega-Andrade *et al.* 2021) because it falls within the mean of the suite of climate models (Grose *et al.* 2023). The CMCC-CM2 models can also reproduce the main patterns of future temperature and precipitation regimes at a very high spatial resolution (Scoccimarro *et al.* 2021).

CMIP6 uses future emission scenarios, known as shared socioeconomic pathways (SSPs), that consider the expected socio-economic, technological growth and associated greenhouse gas emissions in the scenarios of future climatic conditions (Fan *et al.* 2020; HamadAmin and Khwarahm 2023). To characterise future climate conditions, we obtained bioclimatic data projected under two IPCC concentration pathways emissions scenarios, SSP2.45 (optimistic) and SSP3.70 (high middle) (HamadAmin and Khwarahm 2023). SSP1.45 was excluded from the model as emissions have already exceeded this scenario (Hausfather and Peters 2020; Nazarenko *et al.* 2022). For the Euclidean distance to watercourses, we used the present-day data for all future projections under the premise that these geological features are not likely to change significantly within the next 80 years.

Using the optimal MaxEnt model described above, we predicted *S. eungellensis* habitat suitability across the Eungella study area for SSP2.45 and SSP3.70 scenarios across four time periods (2040, 2060, 2080, 2100). Maps of predicted future pre-clearing habitat suitability were created using QGIS (QGIS Development Team 2022). To estimate area of potentially suitable habitat, binary predictions for each scenario and time period were created using a threshold that maximises sensitivity and specificity (Muñoz *et al.* 2022; Torkkola *et al.* 2022a).

Ethical standards

This research was conducted in accordance with the Queensland state wildlife permits issues by the Department of Environment and Science (within protected areas: P-PTUKI-100483094, P-PTC-100483098, P-SPP-100483101, and non-protected areas: WA0056270) and a Monash University Animal Ethics Committee approval (Approval number: 39971).

Results

Individuals observed

We detected 103 *S. eungellensis* individuals during the survey period, comprising 49 adults (22 female, 9 male,

18 indeterminate), 51 juveniles and 3 indeterminate. A total of 56 individuals were captured allowing for measurements of morphology and body temperature. Adult males ($n = 9$) were slightly, but significantly, larger than adult females ($n = 22$) for mass, tail length, head length and head width (Table S3 and S4).

Thermal biology

Saproscincus eungellensis displayed low active body temperatures ($n = 56$, ~ 23 – 26°C ; Fig. 3). The mean T_b of the species was higher than the mean of all the environmental variables (22 – 23°C) but was lower than the observed maximum values of T_{surf} and T_{surfMax} (Table S5). Model V performed best out of the tested models (Table 3). According to this optimal model, T_{surf} ($P = 0.02348$) had a significant positive effect on the T_b of *S. eungellensis*, whereas average potential refuge temperature ($P = 0.04847$) had a significant negative effect (Table 4). Additionally, T_{surfMax} ($P = 0.07772$) had a marginally significant positive effect (Table 4). Both air temperature ($^\circ\text{C}$) and solar radiation (W/m^2) had insignificant effects on lizard T_b and were removed in the step wise process (Table 3).

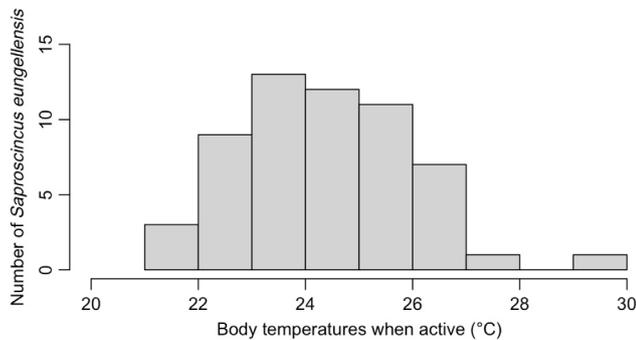


Fig. 3. Bar chart showing distribution of observed *Saproscincus eungellensis* field active body temperatures (T_b ; $^\circ\text{C}$).

Table 3. Comparison of linear mixed effects models on the effects of surface temperature ($^\circ\text{C}$), maximum surface temperature ($^\circ\text{C}$), solar radiation (W/m^2), air temperature ($^\circ\text{C}$), average potential refuge temperature ($^\circ\text{C}$), snout–vent length (mm), mass (g), and site ID (fixed effect variables) on the body temperature of *Saproscincus eungellensis*.

Model	AIC	BIC	d.f.
M I. Saturated model with all predictor variables	133.04	148.3	26
M II. Model excluding air temperature	129.56	143.3	27
M III. Model excluding mass, air temperature	128.56	140.78	28
M IV. Model excluding solar radiation, mass, air temperature	119.62	130.3	29
M V. Model excluding snout – vent length, solar radiation, mass, air temperature	113.88	123.04	30

Chosen model indicated in bold.

AIC, Akaike information criterion; BIC, Bayesian Information Criterion; d.f., degrees of freedom.

Table 4. Summary of the statistics and coefficients from the best linear mixed effects model regarding the influence of surface, maximum surface, and average potential refuge temperatures ($^\circ\text{C}$) on *Saproscincus eungellensis* body temperatures.

Factor	Coefficient	s.e.	t	P
Intercept	21.854	6.3495	3.442	0.00234
T_{surf}	0.5673	0.2355	2.408	0.02348
T_{surfMax}	0.2096	0.1146	1.830	0.07772
Average refuge temperature	−0.7708	0.3710	−2.077	0.04847

Restricted maximum likelihood (REML): 101.9 with 30 degrees of freedom. Significant results ($P < 0.05$) marked in bold.

Habitat

The PCA of habitat structure variables indicated two principal components (PC1 and PC2) explaining a cumulative total of 94.58% of the variation in habitat structure within creek and forest sites (Fig. 4). PC1 explained a high amount of variation (85.63%), describing a gradient of increased water area, rock and flood debris count (Table 5). PC2 explained 8.95% of the variation and described a gradient of increasing woody debris and bare ground. A PERMANOVA was conducted on PC1 only (as only this PC had an eigenvalue >1 ; Jackson 1993), this revealed a significant difference in habitat structure between creek and forest sites ($F_{1,6} = 32.509$, $R^2 = 0.844$, $P = 0.02$).

All but two *S. eungellensis* individuals were found active in shade. Microhabitat preferences for *S. eungellensis* were predominantly rocks and fallen palm fronds, but they were also found on woody and flood debris and on leaf litter (Table S6). Creek site mean potential refuge temperatures were lower than those of adjacent rainforest sites (Fig. S8A); however, the mean air temperature for creek sites was higher than adjacent rainforest sites (Fig. S8B; refuge, $P < 0.0001$; air, $P = 0.03811$). The thermal regime of Eungella NP varies across elevation with the mean temperatures of both air and refuge being much lower at higher elevations (Fig. S9; $P < 0.0001$).

Distribution

All individuals were found within 25 m of a stream (Fig. S10) at elevations between 700 and 1000 m (Fig. S11). No more than 25 individuals were found at any site. Using both the species' historic and new occurrence points, the EOO and AOO were calculated to be 81.7 km^2 and 36 km^2 , respectively (Fig. S4).

Estimate of current population size

The minimum and maximum adult population size estimates for *S. eungellensis* were 17,333 (high range; >950 m elevation), 4667 (low range; >950 m elevation), and 379 (700–950 m) individuals per km^2 , respectively (Table S11). Area calculations from buffered high elevation creek lines between 700 and 950 m ($A = 15.88 \text{ km}^2$) and above 950 m

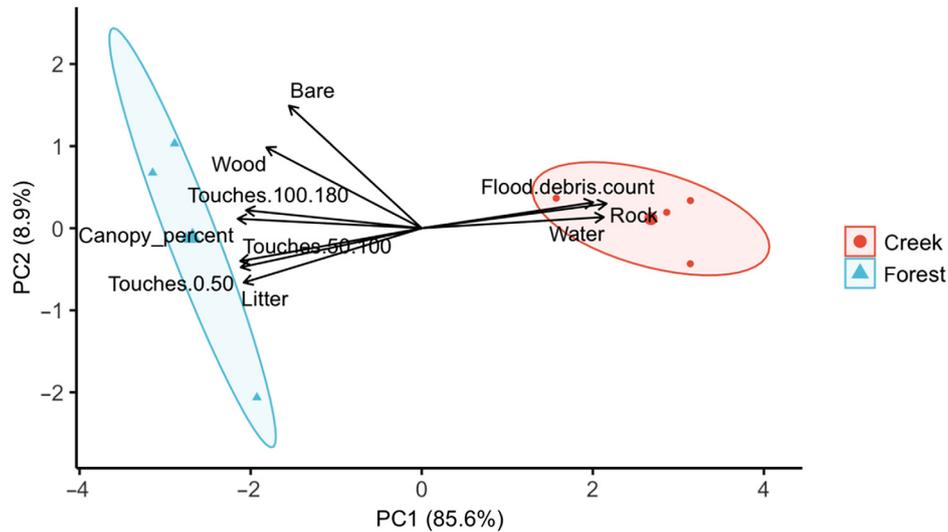


Fig. 4. Principal Component Analysis performed on the mean habitat structure variables at each high elevation survey site, with groups indicating habitat type (creek vs forest). Coloured ovals around each group display 95% confidence ellipses. The first principal component (PC1) describes increases in water, rock and flood debris cover, and decreases in variables related to forest (i.e. canopy percent, litter).

Table 5. Summary statistics and loadings of the principal component analysis (PCA) of the habitat structure variables for PC1 and PC2.

Variable	PC1	PC2
Proportion of variance	0.85	0.09
Eigenvalue	8.56	0.89
Rock	0.34	0.15
Wood	-0.28	0.48
Litter	-0.32	-0.32
Bare	-0.24	0.72
Water	0.33	0.07
Flood debris count	0.31	0.15
Touch pole touches 0–50 cm	-0.33	-0.23
Touch pole touches 50–100 cm	-0.33	-0.19
Touch pole touches 100–180 cm	-0.32	0.11
Canopy cover (%)	-0.34	0.06

($A = 6.01 \text{ km}^2$) resulted in a total A value of 21.9 km^2 . By inputting these values into the $d \times A \times p$ formula, along with the P -value of 0.48, minimum $16,352 (4667 \times 6.01 \times 0.48 + 379 \times 15.88 \times 0.48)$ and maximum $52,892 (17,333 \times 6.01 \times 0.48 + 379 \times 15.88 \times 0.48)$ numbers of mature individuals were calculated for *S. eungellensis*.

Species distributional modelling

The optimal MaxEnt model (regularisation multiplier of one and Linear feature class) performed well, with an AUC of 0.94. The difference between the training and testing AUC

(i.e. AUC_{DIFF}) was small (0.05) suggesting that the optimal model generalises well to novel data and is not overfitted (Warren and Seifert 2011; Farquhar *et al.* 2023a). Both the OR_{10} (0.17) and OR_{MTP} (0.15) scores for the model were also small, further indicating minimal model overfitting (Fielding and Bell 1997). Full model performance metrics of the candidate MaxEnt models are in Table S8. The geographical prediction of the current distribution of potentially suitable pre-clearing habitat for *S. eungellensis* is displayed in Fig. 5. The most important predictors of habitat suitability were annual mean temperature (Bio1; ~70% contribution) and annual precipitation (Bio12; ~25% contribution) (Fig. S12). The model predicted high habitat suitability for relatively low values of annual mean temperatures and high values of annual precipitation. Warm annual temperatures and low annual precipitation were associated with low predicted habitat suitability for the species. Response curves of the environmental variables are supplied in Fig. S13.

Future potential distribution

Projecting models to future climate scenarios predicted large-scale reductions in the extent of potentially suitable habitat for *S. eungellensis* (Figs 6 and 7). The species is projected to lose all available habitat under emission scenario SSP3.70 and SSP2.45 by 2080 and 2100, respectively (Figs 6 and 7).

Threat analysis

We observed evidence of fire, invasive species (feral cats and feral pigs) and climate change at sites where *Saproscincus*

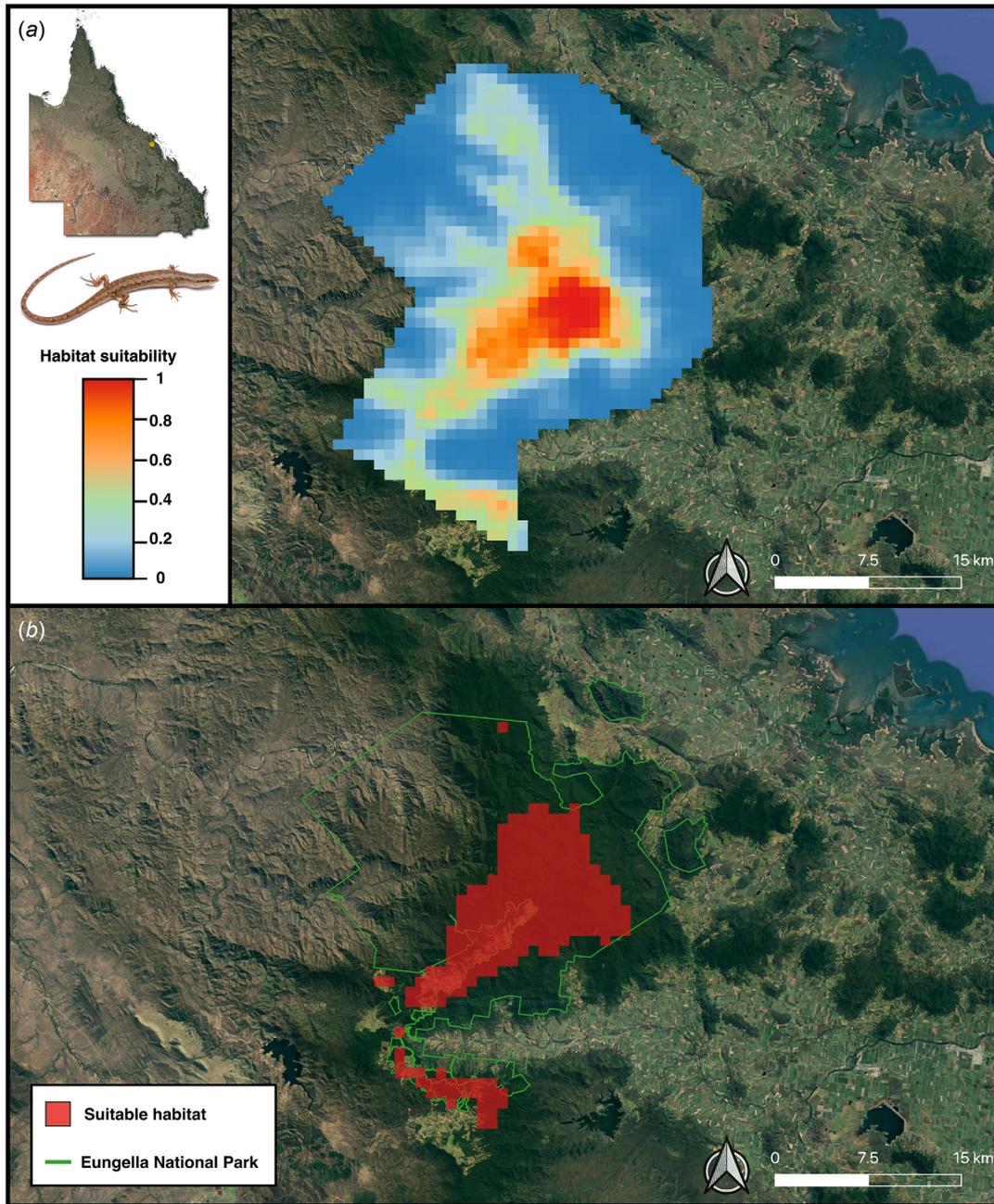


Fig. 5. The optimal MaxEnt model predictions displaying currently climatically suitable areas for the potential occurrence of *Saproscincus eungellensis* across the Eungella study area. Maps show (a) probability of climatic habitat suitability, and (b) a binary prediction based on a probability threshold optimised for ROC; areas highlighted red (where probability of habitat suitability >0.27%) provide the species' approximate realised niche, given that the model included climate variables aligning with the species ecological needs (e.g. temperature, rainfall).

eungellensis occurred. The fires of 2018 burned habitat at, and close to, *S. eungellensis* occurrences (Fig. S14). The extent of the fires within Eungella NP (~209 km²; Hines *et al.* 2020) burned areas over two times greater and five times greater than the species' EOO and AOO respectively (for more information about threats refer to Fig. S14, Tables S7 and S9).

Conservation status assessment

Based on assessment against IUCN Red List criteria (see Tables S7 and S9–S11) *S. eungellensis* meets criteria for A3(c) for Endangered, B1ab(iii) for Critically Endangered, B2ab(iii) for Endangered, Vulnerable D2, and E for Vulnerable. The

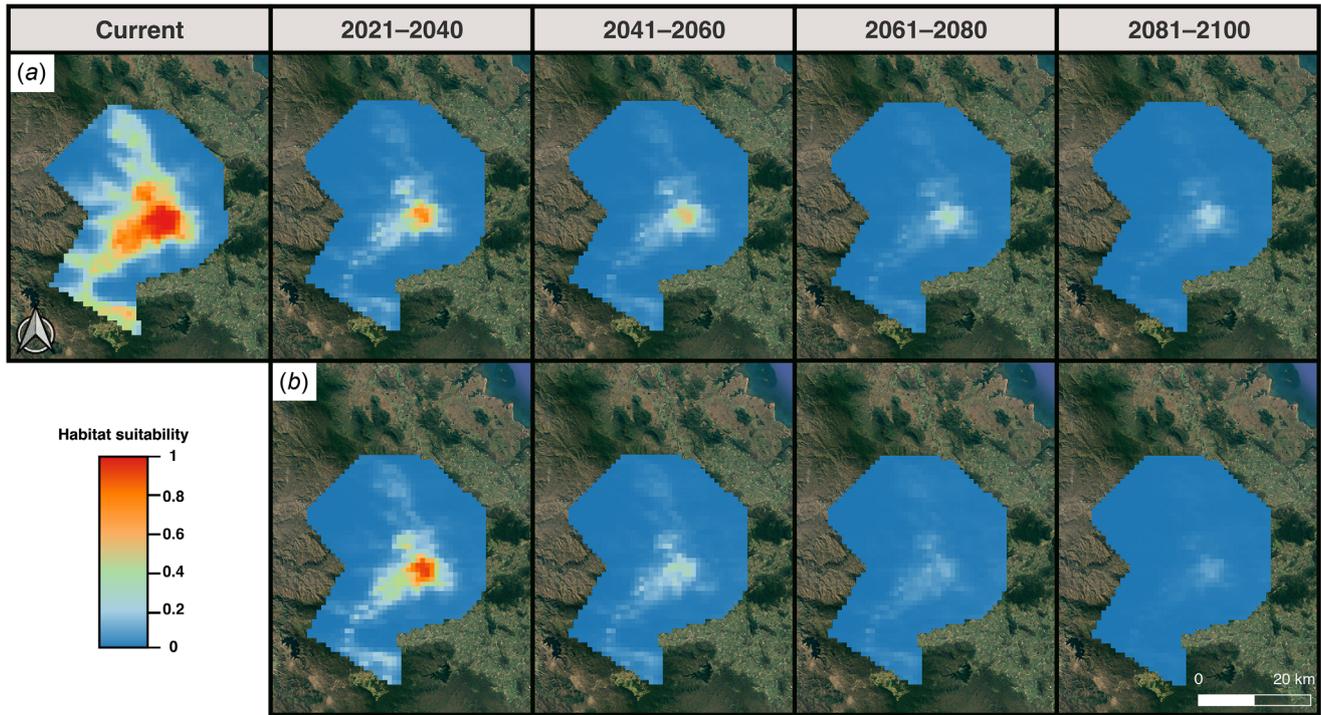


Fig. 6. MaxEnt model predictions of future climatically suitable areas for the occurrence of *Saproscincus eungellensis* in the Eungella study area. Maps show continuous probability of pre-clearing habitat suitability modelled at different time periods under (a) Scenario SPP2.45 and (b) Scenario SPP3.70.

species is not eligible for listing under criterion C. The Criterion A listing is based on projected population reductions $\geq 50\%$ in 10 years. The Criterion B listings are based on EOO $< 100 \text{ km}^2$, restricted to one location threatened by climate change, and an inferred continuing decline in area/extent/quality of habitat. The Criterion D listing is based on ≤ 5 locations and a threat (climate change) that can drive the species to extinction in a short time period. The Criterion E listing was based on $\geq 0\%$ extinction likelihood in 100 years.

Discussion

This study addresses a key problem in conservation biology: clarifying the conservation status of DD species. We show here how a targeted conservation assessment guided by core aspects of the IUCN Red List criteria can collect sufficient data on a poorly-known species to inform conservation assessments. By undertaking targeted field work we confirmed that *S. eungellensis* is a highly specialised, range-restricted species, with a likely high susceptibility to predicted climate change. Predictions of habitat suitability in the future under climate change scenarios indicate that the species will completely lose all suitable habitat by 2080. Below we discuss the implications of our results for the conservation of

S. eungellensis and the corresponding Red List category that the species is eligible for.

***Saproscincus eungellensis* is restricted to high elevation areas of Eungella NP**

Our study has demonstrated that *S. eungellensis* is likely restricted to elevations above 700 m in Eungella NP that are annually subject to relatively cold temperatures and high precipitation. The species distribution model accords broadly with modelling conducted by Moussalli *et al.* (2009) showing that *Saproscincus czechurai* (a related high-elevation species) is dependent on relatively cool and wet conditions. Annual temperature and precipitation have been identified as significant contributing variables influencing distributional limits in other ectotherms (Kearney and Porter 2004; Penman *et al.* 2010; Sopniewski *et al.* 2022). The persistence of short-range endemic taxa is thought to be a result of environmental stability, ecological specialisation, and localised abundance (VanDerWal *et al.* 2009; Torkkola *et al.* 2022a). For example, Torkkola *et al.* (2022b) argued that the persistence of *Calorodius thorntonensis* is a result of small stable upland rainforest habitats acting as an evolutionary refuge. High elevation rainforest refuges provide a stable but typically localised environment, permitting the continued existence of many short-range endemic species. Similarly, Williams *et al.* (2009) found endemic vertebrate biodiversity in the Wet

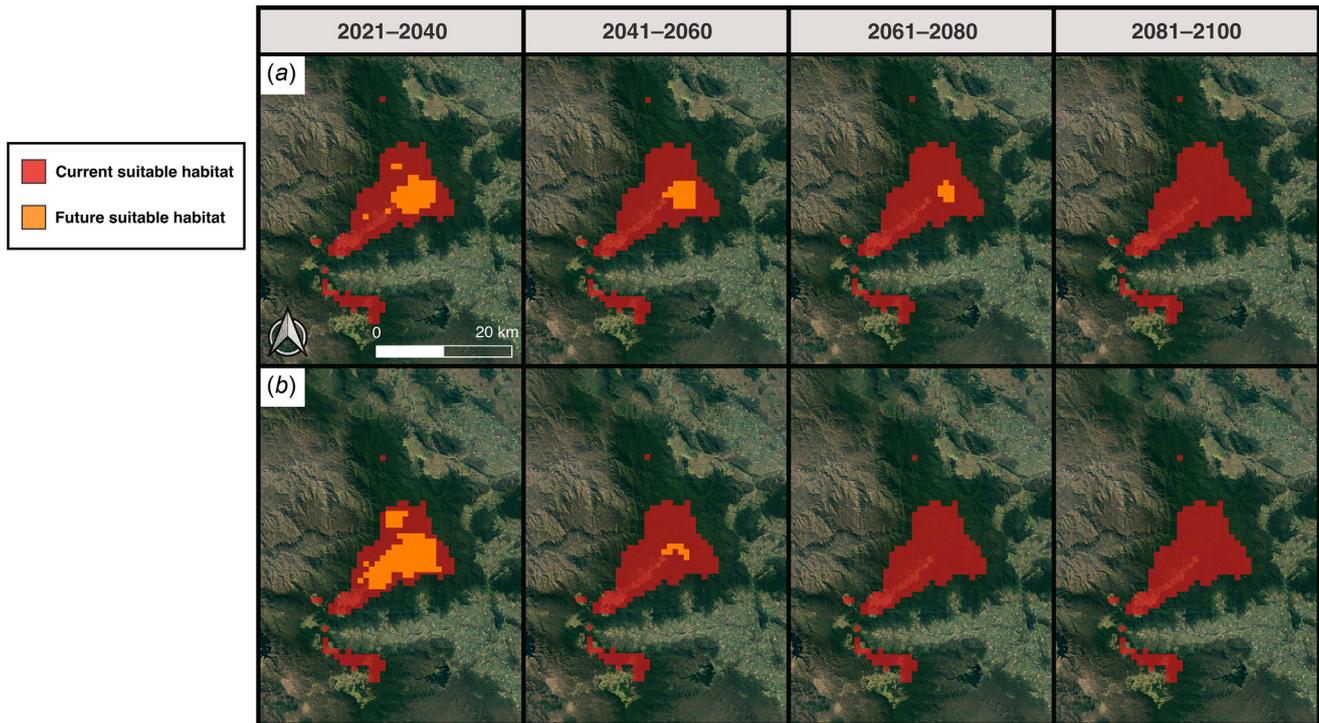


Fig. 7. MaxEnt climate model predictions of future climatically suitable areas for the occurrence of *Saprosinciscus eungellensis* within the Eungella study area. Maps show binary predictions of pre-clearing habitat suitability determined by a probability threshold that optimises ROC; areas highlighted red (where probability of habitat suitability >0.27%) provide the species' approximate realised niche, given that the model included climate variables aligning with the species' ecological needs (e.g. temperature, rainfall); The orange represents an approximation of the species' future realised niche modelled at different time periods under (a) Scenario SPP2.45 and (b) Scenario SPP3.70.

Tropics bioregion to be highest in upland areas. Thus, species distributional modelling indicates that climatic variables likely have a high degree of influence on *S. eungellensis* distributional limits, likely via direct effects of climate on the species' physiological tolerances. High elevations of Eungella NP offer cooler and wetter climatic conditions than lower elevations (Figs. S9A and S9B). Consequently, the relative stability of climatic variables within high elevations of Eungella NP over evolutionary time scales likely shaped the small potential habitat suitability of the species observed through modelling.

***Saprosinciscus eungellensis* is a riparian zone specialist with cold-adapted thermal biology**

Saprosinciscus eungellensis was found to mostly occur in close proximity (<25 m) to creek lines, with the vast majority of observations within 5 m of a water course. Riparian habitats are known to be key refugia for many taxa (Faria *et al.* 2019), as they generally have high humidity and are thermally buffered. As such, riparian habitats often support disproportionately high levels of both species' abundance and richness in ectotherms (Warren and Schwalbe 1985; Faria *et al.* 2019; Bateman and Merritt 2020). The reliance of *S. eungellensis* on riparian habitat accords with the hypothesised association between mesic herpetofauna and habitats with a heterogeneous

structure that provide ample microhabitat refuges (Farquhar *et al.* 2023b). Farquhar *et al.* (2023b) showed that range-edge populations of *Lampropholis delicata* (delicate skink) are associated with habitats that offer deep leaf litter, logs, and rocks. Such microhabitat structure is critical to skink ecology, impacting species' ability to avoid predation, hunt and thermoregulate (Neel *et al.* 2021).

Thermoregulation, along with high humidity, may be a key factor explaining the reliance of *S. eungellensis* on riparian habitat. Morgan (1988) suggests that in heavily shaded environments (such as rainforests), thermally discrete microclimates are critically important for lizard thermoregulation. Complex microhabitat structures related to riparian habitat provide a varying thermal gradient allowing access to distinct thermal regimes. For example, coarse woody debris has been shown to provide North American forest reptiles and amphibians a substrate for both thermoregulation and refuge use that is distinct from ambient air temperatures (Whiles and Grubaugh 1996). High elevation forest areas are of relatively low thermal quality (Díaz 1997; Brazeau 2016), but rocks within riparian systems provide higher substrate temperatures than other microhabitats, allowing high elevation ectothermic species to achieve higher desired body temperatures. For example, Muñoz and Losos (2018) found that in an elevation-stratified assemblage of lizards (*Anolis*) high elevation

individuals were more frequently encountered on rock substrates whereas low elevation individuals were encountered on trees. Closed habitats such as highly shaded forests (where *S. eungellensis* are uncommon) strongly filter sunlight (Scheffers *et al.* 2017) and exhibit little thermal heterogeneity, posing challenges to the maintenance of optimal body temperatures by ectotherms (Huey 1974; Fitzgerald *et al.* 2003). For example, Huang *et al.* (2014), found high elevation forests are too cold to provide suitable habitat to support populations of a high elevation lizard species, *Takydromus hsueshanensis*. Thus, although *S. eungellensis* is not an active basker, it may be seeking out the warmer parts of its microhabitat to achieve its thermal requirements.

Saproscincus eungellensis displayed relatively low field active body temperatures compared to other high elevation skink species (Table 1). This accords with observations that other species of *Saproscincus*, and other thigmothermic species, have lower CTMax values than active basking lizard species occupying similar environments (Table 1; Greer 1980, 1989; Muñoz *et al.* 2016). For example, Muñoz *et al.* (2016) showed a difference of 10°C in the CTMax values of two Australian rainforest lizard species occurring in the same locality, *Saproscincus czechurai* (35.1°C) and *Carlia rubrigularis* (45.2°C). The low field active body temperature of *S. eungellensis* may be explained by considering aspects of their ecology. In the field we found substrate temperature to be the most significant predictor of *S. eungellensis* body temperature, whereas both air temperature and solar radiation were unrelated to body temperature. Further, we only observed two *S. eungellensis* individuals active outside of shade. *Saproscincus eungellensis* belongs to a genus of shade-seeking scincid lizards, which regulate body temperatures via conduction of heat from bodily contact with surface structures (i.e. thigmothermy) rather than direct sunlight (heliothermy; Muñoz *et al.* 2016; Sazima 2023). Heat tolerance in rainforest lizard species likely evolved in two distinct directions, with edge habitat species displaying significantly higher heat tolerances than shade-seeking species (Muñoz *et al.* 2016). Thus, specialisation to use cool and mesic microhabitats likely contributed to the relatively low field active body temperatures displayed by *S. eungellensis*. Future studies should determine the CTMax and thermal preference of *S. eungellensis*. However, Grigg and Buckley (2013) found that thermal tolerances are conserved within evolutionary history rather than being determined by dispersal or local adaptation. Thus, we hypothesise that *S. eungellensis* has both a relatively low CTMax and thermal preference in line with its congeners; indeed, this may explain why the species is geographically restricted to high-elevation creek lines.

Threats to the persistence of *Saproscincus eungellensis*

Saproscincus eungellensis is affected by numerous threats; in the field we observed evidence of invasive species (feral cats

and feral pigs), fire and climate change (additional potential threats to the species are outlined in Tables S7 and S10). These threats are known to interact; for example, both fire and feral pigs simplify habitats by removing vegetation and refuges, which in turn creates conditions favouring cat predation (Doupé *et al.* 2010; McGregor *et al.* 2016). Cats are particularly problematic for Australian native reptile species and it is estimated that cats kill 466 million Australian native reptiles per year (Woinarski *et al.* 2018). The observed threats for *S. eungellensis* are consistent with threat analyses reported by Tingley *et al.* (2019) and Geyle *et al.* (2021), who highlighted invasive species, habitat destruction (i.e. through development), and climate change as major threats acting on terrestrial squamates.

Saproscincus eungellensis displays many characteristics consistent with species considered especially vulnerable to climate change. First, the species is highly specialised and restricted to high-elevation riparian habitat. Tropical high-elevation specialists have been identified as among the species most likely to be impacted by climate change. For example, Cordier *et al.* (2020) predicted total losses for two microendemic high elevation frog species in response to climate change. Similarly, highly specialised species are predicted to be highly susceptible to climate change. For example, Hagger *et al.* (2013) identified habitat specialisation as the second most important predictor of climate sensitivity in an assemblage of subtropical vertebrates. Second, *S. eungellensis* is a relatively recently described narrow-range endemic, impacted by multiple threats. Wotherspoon *et al.* (2024) determined that these traits characterised many Australian threatened squamates. Finally, *S. eungellensis* is a thigmothermic species and consequently has a relatively low CTMax. Muñoz *et al.* (2016) showed closely related shade-seeking *Saproscincus* species with relatively low CTMax's are especially susceptible to climate warming due to low tolerance of heat. Consequently, *S. eungellensis* is likely highly susceptible to future climatic changes.

Despite using an optimistic model (CMCC-CM2) and relatively optimistic scenarios (SSP2.45 and SSP3.70; Grose *et al.* 2023; HamadAmin and Khwarahm 2023), future climatic modelling for *S. eungellensis* showed complete loss of climatically suitable habitat for both climate emission scenarios SSP2.45 and SSP3.70 by 2100 and 2080, respectively (Figs S12 and S13). Our modelling aligns with previous work by Williams *et al.* (2003), which showed extensive losses for Australian tropical high elevation rainforest species in response to climate change. Similarly, Williams *et al.* (2008) predicted that globally many high elevation tropical climates would disappear in response to global warming, essentially leaving these species with no available habitat. In a similar vein Laurance (2015) predicted temperature increases of just 2°C would be enough to drive 1/8th of endemic montane vertebrates in tropical Queensland extinct. Modelling of palaeo-distributions for a related and ecologically similar species (*Saproscincus czechurai*) by Moussalli *et al.* (2009),

indicated a larger distribution when colder and wetter climates were much more extensive in the Pliocene. The differences between prediction of past, present and future distributions of high elevation species highlight the magnitude of the potential impacts of climate warming on these species' persistence. There is some potential that the riparian microhabitat that *S. eungellensis* occurs in may buffer the impacts of climate change, and this process may explain why high elevation areas of the Eungella region host several evolutionarily old faunal lineages (Mahony 2020). However, the speed and trajectory of current climate warming exceeds that most organisms have experienced in their recent evolutionary history, and *S. eungellensis* also has a very narrow niche. For these reasons we argue that our dire predictions for the future of this species under a rapidly and markedly warming climate are realistic and defensible.

In addition, although future climate modelling for *S. eungellensis* predicted drastic losses of climatically suitable areas, it is important to note that other synergistic processes are of concern. First, while CMIP6 climate models predict factors that influence fire likelihood, the model may not show the actual extent and severity of potential future fires (Li *et al.* 2024). A more severe fire weather climate is predicted for Eungella NP (Climate Change in Australia 2020), and fire has already severely impacted high elevation areas of Eungella NP, burning areas in close proximity to core *S. eungellensis* habitat (Hines *et al.* 2020). Modelling by Torkkola *et al.* (2022a) estimated that the 2019–2020 fires burnt about half of the predicted distribution of *Harrisoniascincus zia*, an Australian skink with a similar preference for cool wet upland forest habitats. Second, the modelling does not consider the arrival of lower elevation predators, competitors and pathogens. Warming pushes climatic envelopes towards the poles or higher elevation, warmer climate species are able to track these changes and shift their ranges to accommodate them (Walther *et al.* 2002; Colwell *et al.* 2008; Ashton *et al.* 2016). In the field we saw evidence of skink assemblage composition changes in the Eungella area, with different species occurring in distinct bands of elevation. If these species track climatic envelopes to higher elevations, they may compete with or predate *S. eungellensis*, further contributing to extinction risk. The arrival of lowland competitors has already been observed affecting other Australian high-elevation skink species (DCCEEW 2023c).

***Saproscincus eungellensis* meets the criteria for recognition as Critically Endangered**

Saproscincus eungellensis met the criteria for Critically Endangered (under Criterion B), the most severe IUCN Red List category before extinction. The species also met the criteria for Endangered (A3, B2), and Vulnerable (D2, E), highlighting that the species is severely impacted by multiple aspects of its ecology and threats. As a Critically Endangered

species likely highly susceptible to climate change, conservation (e.g. population monitoring, and mitigation of feral species) for this species should be considered an urgent priority.

These results highlight issues with the IUCN DD category. First, we know that DD species are often at risk of extinction, but have not yet received such diagnosis (Bland *et al.* 2017; González-del-Piiego *et al.* 2019; Caetano *et al.* 2022). Second, the IUCN Red List has only considered climate change as the main threatening process for 4% of assessed skink species (Chapple *et al.* 2021; Torkkola *et al.* 2022a). The case of *S. eungellensis* highlights limitations in the IUCN framework as a previously DD species severely impacted by climate change. Unfortunately, there are many species still listed as DD on the IUCN Red List. The methods used in this study can act as a framework allowing future researchers to survey and assess DD species to remove them from the problematic DD category. Further, this research can act as a case study applying the methods outlined by Mancini *et al.* (2024) to assess the impacts of climate change under subcriterion A3.

Conclusion

With rapidly occurring changes in climate, many of the 21,000 species within the DD category of the IUCN may be faced with the prospect of total habitat loss and corresponding extinction. It is therefore paramount that conservation scientists focus on species in this category to identify those most at risk and better resolve the true conservation status of DD species. The findings from our research add to the growing body of evidence that both DD and high elevation range-restricted species are likely highly susceptible to climatic changes and at a heightened risk of extinction. The methods outlined in this paper can be used as a framework for future researchers to undertake relatively rapid field studies coupled with species distributional modelling of DD species, thereby allowing them to be assigned to a data sufficient IUCN Red List category. In turn, these methods will help to enable conservation managers to target conservation and recovery efforts to the species that require it most.

Supplementary material

Supplementary material is available [online](#).

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Data availability. Our data and R code are in the supplementary online material.

Conflicts of interest. The authors declare that they have no conflicts of interest.

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