



# Prioritising species for monitoring in a South African protected area and the Red List for plants

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## Abstract

Protected areas (PA) in biodiversity hotspots face the challenge of monitoring large numbers of locally rare and threatened plant species at times with limited budgets. Prioritising species according to their local extinction risk could help PA managers to decide which species to monitor. However, there is often very little information available on the species occurrence and extinction risk in the PA. Because of this, PA managers often rely on the national or global Red List for prioritising species at the PA level. Here, we evaluate the effectiveness of using the Red List for species prioritisation and examine the robustness of extinction probability equations for 74 fynbos species in Table Mountain National Park (TMNP). We conducted in-field surveys to verify the persistence of subpopulations previously recorded, following a detection protocol adapted for rare and cryptic plant species. We found that most targeted species were extant within TMNP but with a substantially reduced number of subpopulations. Twenty-six species only had one or two subpopulations remaining. Critically Endangered (CR) species lost on average 4 subpopulations more than Least Concern (LC) species. However, species persistence in TMNP was largely independent of their Red List status. Half of the species represented by just one or two subpopulations were listed as LC. This work shows that prioritising monitoring according to the Red List status is not appropriate at the scale of the individual PA. We suggest that more in-field data and monitoring is required to prevent extinctions occurring in PAs.

**Keywords** IUCN Red List · Protected area · Fynbos · National Park · Biodiversity

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## Introduction

The main purpose of Protected Areas (PAs) globally is to protect and nurture nature within their boundaries and mitigate the negative effects of external pressures so as to ensure the functioning of ecosystems and the continuation of natural ecological processes (Leroux and Kerr 2012; Pimm et al. 2014) whilst some also aim to provide benefits to people (Juffe-Bignoli et al. 2018; Gannon et al. 2017). Protected areas are one of the main mechanisms used to combat species loss (Pimm et al. 2014). Defined here as legally declared areas managed for conservation, PAs are a cornerstone of biodiversity conservation (Watson et al. 2014; Dudley et al. 2014). They also aim to conserve heritage and natural sites of value such as archaeological landscapes, biological or geological formations (Yui 2014; Phillips 2002).

In order to ascertain whether establishment objectives of PAs are being achieved, an assessment of the management effectiveness should be regularly conducted (Hockings et al. 2006; Geldmann et al. 2018). Information and knowledge of the state of biodiversity within PAs are required to inform these assessments. To develop such knowledge PAs are required to undertake research and monitoring of species and ecosystems under threat (Juffe-Bignoli et al. 2018; Bacon et al. 2019).

Limited budgets are a common challenge in PA management mainly in the Global South (Mansourian and Dudley 2008; Comerford et al. 2010; Evans et al. 2012; Juffe-Bignoli et al. 2018) and are particularly problematic for conservation monitoring programmes faced with large numbers of nationally and even locally endemic and threatened plant species (Thompson et al. 2011), such as in a megabiodiverse country like South Africa. For accurate decision-making, reliable field data is essential to PAs (Alexander et al. 2012). There is an urgent need to develop new systems for monitoring to get a clearer and more detailed picture of biodiversity loss through quantitative changes in the dynamics of habitats and populations (Carlson et al. 2019; Hochkirch et al. 2020). With so many species under threat, one of the greatest challenges faced by PAs is to prioritise which species to target and monitor.

Often the charismatic megafauna and bird species are known in a region and prioritised within a PA while the majority of species in the PA remain undocumented (L. Joppa pers. comm.). In some instances, historic collections provide a baseline for species lists (e.g. Moreira et al. 2020a, b) but more often field based studies are required to identify species and generate lists (Darbyshire et al. 2017). Providing an area with protected status and managers with a list of species occurring in a PA may not suffice to prevent the decline and extinction of species; on-the-ground actions are still needed (Mora and Sale 2011). Protected Areas require a list of species that is prioritised for monitoring and can be used to make informed management decisions with associated implementable actions. One means to do this is to determine the extinction risk for each species and prioritise accordingly (Cowell 2018). An understanding of species extinction risk validated by accurate in-field surveys will help address conservation managers' challenges of time and funding.

International conservation targets have been set by Parties to the Convention on Biological Diversity (CBD 2010). Known as the Aichi targets, these are very broad in scope, but specific targets exist for PAs (Target 11) and for reduction of extinction risk (Target 12) (Zafra-Calvo et al. 2019; Juffe-Bignoli et al. 2016; Gannon et al. 2017). The CBD Aichi targets are due to be revised in 2022 with more emphasis on monitoring population change over time and reducing extinction with a view to establishing more measurable and impactful targets. For instance the South African government has been a signatory of,

and Party to, the CBD since 1995. It has since enacted a series of conservation acts and regulations relying mainly on PAs to achieve the conservation goals for the country. The National Environmental Management: Biodiversity Act (NEMBA) requires monitoring of South Africa's biodiversity assets and the National Environmental Management: Protected Areas Act (NEMPAA) requires the South African National Parks (SANParks) to conduct this monitoring (Sect. 50, paragraphs 1–3). Protected Areas are required to monitor and report on progress towards strengthening South Africa's PA estate and the conservation status of threatened biomes, vegetation types and species. We chose a PA in South Africa as a case study to consider the potential of monitoring within PAs to achieve international and national conservation targets.

South Africa is one of the few megadiverse countries with a National Red List category for all plant species which aligns with the IUCN (International Union for the Conservation of Nature) Red List, here after referred to as the Red List. Many of the plant species listed on the South African Red List remain unsupported by a formal assessment (Raimondo 2019; Raimondo et al. 2009). There is a perceived need to use the South African (and at times the IUCN ) Red List as a measure of PA performance and PA managers in South Africa are asked to report results on all species listed on the South African Red List (<http://redlist.sanbi.org/redcat.php>) under their custodianship including rare species (Cowell et al. 2020; Rebelo et al. 2011). This requirement reflects the reality that the Red List is at times the only tool readily available to conservationists for identifying species with a high likelihood of going globally extinct in the face of the numerous and growing threats to plant species. However, the Red List is not developed or intended as a conservation prioritisation tool at local scale but rather intended as an aid to prioritisation at the global scale (Collen et al. 2016). The use of the threatened species on the South African Red List, by PAs in South Africa, as a checklist of species to be monitored, can have negative outcomes for PA management.

The IUCN extinction risk assessments are generated at a global (IUCN 2022b), regional (IUCN 2022a) or national scale. The scale of these assessments is therefore rarely a true reflection of a species status within a single PA. Red List assessments allocate species to fairly broad categories of extinction risk related to population size, geographic range and rate of decline of both of these metrics. As conservation managers are responsible for ensuring species do not go extinct within their PA, the Red List is of limited use as a conservation tool for monitoring given the scale at which it is generated (global, regional and national). We acknowledge that this is certainly not the aim of the IUCN Red List but it is a reality that in many countries it is the only list of information on species for conservation and is misused. At the local scale, the use of historic data to assess threats and estimate the probability of survival in the future is an important step in determining which species require urgent attention within the PA. However, the spatial resolution of current plant data is usually insufficient to address conservation concerns at the scale of an individual PA, because of an absence of empirical data for many plant populations (Bachman et al. 2019).

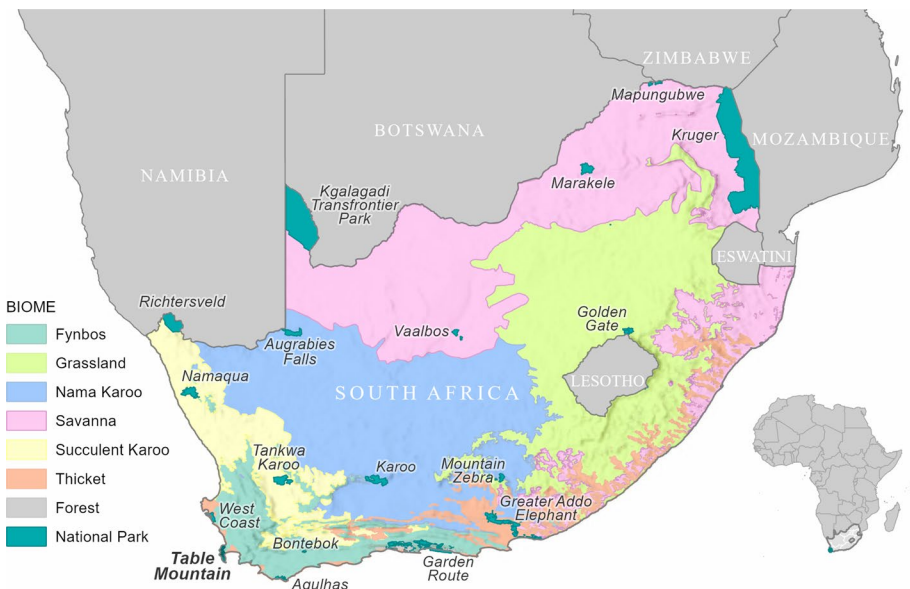
A potential solution to the lack of spatial resolution for many species is to use a combination of herbarium and survey datasets for the PA and then ground-truth what is happening at a finer scale. This strategy could make a powerful contribution to locating and learning about species in the field (Thompson et al. 2013). Herbarium collections are large, time-sensitive records, and have increasingly been recognised around the world as having significant value beyond simple taxonomic purposes (Greve et al. 2016; Davis et al. 2015; Nic Lughadha et al. 2019a). Botanical survey data (survey data) are data from formal in-field vegetation surveys that do not necessarily have an herbarium specimen collected with the data but are verified by specialists in field or with photographs. Survey data are

increasingly being used to address ecological and conservation management questions, yet they are less frequently used in IUCN Red List assessments (Nic Lughadha et al. 2019b). S. Bachman pers. Comm. and J. Plummer pers. Comm)

Previous work (Cowell 2018) combined the longer-term temporal datasets offered by herbarium collections and shorter-term botanical survey datasets to determine their potential to estimate the persistence of plant species in a PA. The use of extinction probability equations has generally focussed on estimating the date of extinction of a species already thought to be extinct (McInerney et al. 2006). Recent studies applying extinction probability equations to sightings records reached mixed conclusions concerning their applicability and called for the approach to be tested on other datasets (Chong et al. 2012; Roberts and Jaric 2020; Thompson et al. 2019, 2020; Brook et al. 2019). Cowell's (2018) analyses of datasets comprising herbarium and botanical survey data (excluding citizen science datasets which were unavailable at the time of the study) individually yielded insufficient data for application of extinction probability equations. Combining herbarium and botanical survey data yielded extinction probability results that required in-field validation.

Field work is vital to validate extinction probability inferences from desktop studies (Clements 2014). If extinction probability predictions are to be adopted by PA managers as a basis for prioritising monitoring effort, with a suitable degree of confidence, species predicted to be extant (or extinct) need to be detected (or not), and the predictions verified or refuted. Protected area managers can then be informed of the current status of plant species in their PAs. If the approach proves reliable, we anticipate that PAs could use the species information to assess the long-term trends of the vegetative ecosystems for planning purposes. We tested this proposition on one of the South African PAs declared to protect its floristic diversity.

Five of the six floristically informed South African National Parks occur in the Fynbos Biome (Fig. 1) due to the high proportion of locally endemic plant taxa in this biodiversity



**Fig. 1** South African National Parks and the respective Biomes in which they are found. The case study park, Table Mountain National Park is shown in bold text

hotspot (Cowling et al. 1994; Linder et al. 2012). As a case study, we use Table Mountain National Park (TMNP), which provides a unique research prospect as it has a high number of both globally and locally threatened plant species on the South African Red List. This provides an opportunity to test extinction probability predictions with in-field methods addressing both international and local conservation targets and assisting PA managers to achieve objectives specific to their PA. Although extinction probability equations are widely advocated in the literature, they have yet to be fully investigated in the Fynbos Biome. South Africa National Park managers in the Cape have little fine-scale data available to support the management of indigenous plant species (Cowling et al. 1994). As a result, there is a total reliance of PA managers on the IUCN Red List to provide extinction risk of species and priorities for management action. This places pressure on managers to ensure that all the species listed as threatened on the IUCN Red List and occurring in the PA(s) for which they are responsible are monitored and conserved. Many PA managers only report occurrence and not temporal or spatial aspects of populations.

The purpose of this paper is to evaluate the effectiveness, of the current practice, of using the South African Red List (as a proxy for the Global Red List) to prioritise species for monitoring at the PA level and to examine the robustness of extinction probability equations, by ground-truthing the persistence of subpopulations with in-field surveys, using detection methodologies adapted for rare and cryptic plant species (Alexander et al. 2012; Kery and Schmidt 2008). Here we followed the IUCN Red List definition of subpopulations and geographically distant groupings were treated as subpopulations in the absence of data on immigration rates or genetic exchange. Using this methodology, we seek to determine if global extinction probabilities and extinction dates estimated from sightings records predict the persistence of species within a PA.

## Methods

### Study site

The fynbos (Fig. 1) of the Cape Floristic Region (CFR) has a high proportion of endemic (meaning they occur nowhere else on earth) and rare plant taxa (Linder et al. 2012; Rebelo et al. 2011). For the purposes of this paper, we follow the South African Red List Assessment definition of ‘Rare’. A rare species meets at least one of four criteria for rarity but does not qualify for a threat category according to one of the five IUCN criteria. The four South African criteria are as follows (<http://speciesstatus.sanbi.org/about/>) :

- “Restricted range: Extent of Occurrence (EOO) < 500 km<sup>2</sup>, OR.
- Habitat specialist: Species is restricted to a specialised microhabitat so that it has a very small Area of Occupancy (AOO), typically smaller than 20 km<sup>2</sup>, OR.
- Low densities of individuals: Species always occurs as single individuals or very small subpopulations (typically fewer than 50 mature individuals) scattered over a wide area, OR.
- Small global population: Less than 10, 000 mature individuals.”

Consisting of mainly evergreen, fine-leaved shrubs, sub-shrubs and geophytes known collectively as fynbos, a term also used for the dominant vegetation type, the CFR has a very high species diversity which makes it unique globally. A high percentage of fynbos

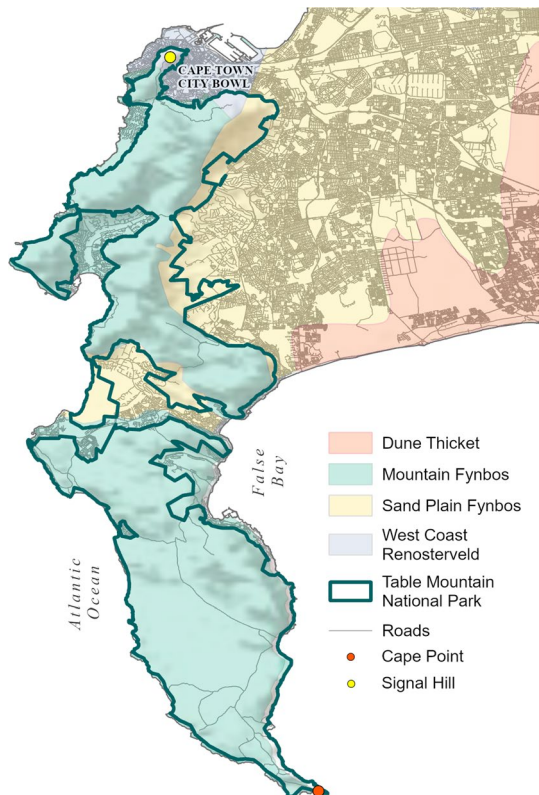
species are rare and difficult to identify due to the large number of congeneric species found in this biome. These closely related, morphologically similar and cryptic species are often overlooked, thus demanding an increase in detection efforts and associated costs (McInerney et al. 2006).

Our study area, TMNP, encompasses approximately 25,000 hectares and is situated on the Cape Peninsula in South Africa. Signal Hill is in the northern limit of the park ( $33^{\circ} 54' S$ ,  $18^{\circ} 24' E$ ) and Cape Point in the south respectively ( $34^{\circ} 21' S$ ,  $18^{\circ} 29' E$ ) (Fig. 2). The vegetation of the Park consists of fynbos with a small percentage of Afromontane forest and coastal dune veld. TMNP has 544 plant species assessed on the IUCN Red List, of which 250 are listed as threatened (categorised as Critically Endangered, Endangered or Vulnerable (Table 1) (Rebello et al. 2011).

Categories marked with <sup>n</sup> are non-IUCN, national Red List categories for species not in danger of extinction but considered of conservation concern. The IUCN equivalent of these categories is Least Concern (LC). DDD is assigned to a species where there is insufficient information to do an assessment and the species is well defined. DDT is assigned to a species when there are taxonomic problems which affect defining of the distribution range and habitat rendering an assessment not possible

Species categorised as Critically Rare and Presumed Extinct, Critically Rare or Rare are prioritised for monitoring within TMNP (Table 2). However, currently TMNP staff manage to monitor fewer than a dozen of these (Rebello et al. 2011). As this study focused on data

**Fig. 2** The Cape Peninsula showing the Table Mountain National Park bordered by the Cape Town City Bowl to the north, the industrial and urban areas of the Cape Flats, and False Bay to the east of the Park. Cape Point in the south and the Atlantic seaboard in the west



**Table 1** South Africa Red List categories for Table Mountain National Park plant species as available in 2016 (Rebello et al. 2011, <http://redlist.sanbi.org/>)

| IUCN Red List category                          | Number of species |
|---|-------------------|
| Critically rare & presumed extinct <sup>n</sup> | 10                |
| Critically Endangered (CR)                      | 55                |
| Vulnerable (VU)                                 | 113               |
| Endangered (EN)                                 | 82                |
| Near Threatened (NT)                            | 46                |
| Data Deficient (DDD/DDT)                        | 54                |
| Rare (including declining) <sup>n</sup>         | 60                |
| Total number of species of conservation concern | 420               |

**Table 2** Grouping of selected fynbos families according to Red List status. Group 1 is given the highest priority for monitoring in Table Mountain National Park, with Group 3 and 4 currently not monitored at all. Number of species per Group given in brackets

| Group Number  | South African Red List categories                               |
|---------------|---|
| Group 1 (29)  | Critically Rare Presumed Extinct (CR PE), Critically Rare, Rare |
| Group 2 (120) | Critically Endangered (CR), Endangered (EN), Vulnerable (VU)    |
| Group 3 (31)  | Near Threatened (NT)  |
| Group 4 (302) | Least Concern (LC)  |

availability it was decided to remove DDD or DDT species from the group analysis as they are already known to lack data and could potentially bias the results.

## Study species

A total of 74 taxa from three of the iconic plant families with the greatest numbers of taxa in the fynbos were selected at random (<https://www.randomizer.org/>) for field sampling, regardless of their threat status (supplementary data). This facilitated herbarium- and desk-based research to gather earlier records, to focus on particular sections of herbaria and the published literature more than would have been the case with a random sample of the plant species in the PA. It also enabled the field team to focus on developing expertise in reliably distinguishing and identifying the many species in these groups, with a view to maximising in field detection rates and accuracy of identifications. Our study sample comprised 25 Ericaceae taxa, 24 Proteaceae taxa, 24 Restionaceae taxa, and included a total of 7 infraspecific taxa. The infraspecific taxa were treated as species throughout the analyses performed and are referred to as ‘species’ hereafter. Information on locality (latitude and longitude) within the Park boundaries was obtained from herbarium specimen labels and survey data when available. Locality information was converted to GIS spatial coordinates using georeferencing methods (Culley 2013). Records were analysed and different localities for the same species identified as separate subpopulations within the Park varying from a minimum of 1 subpopulation to a maximum of 34 per species, making a total of 325 subpopulations (supplementary data). The habitat of each species was determined using literature (Goldblatt and Manning 2000) and from notes on the herbarium and survey data.

## Field detection

The detectability of a species is the probability it will be found whilst undertaking a survey and its presence verified (Alexander et al. 2012). Some factors affecting a species' detectability can be controlled, such as the method used to detect the target species, the frequency of field visits and the time spent searching for the species (Tingley and Beissinger 2009). However, other factors such as weather, terrain and wildfires are more challenging to anticipate. Detectability depends on the season of the search, as plants may be in a vegetative state only and hard to identify (Brummitt et al. 2015). This can increase costs if repeat visits are required to search for species. To mitigate costs and repeat sampling, field visits should be conducted during the peak flowering or growth time of the target species (Kery et al. 2006) suggest the minimum sampling effort of 1.65 visit per species on average that are 'easy' to detect, such as large trees, shrubs or species that flower prolifically. For species that are difficult to detect, such as cryptic and rare species, the recommended average number of visits is 2.2.

We adapted field detection methods (Kery et al. 2006; Alexander et al. 2012) to consider the large number of cryptic species, erring on the side of high sampling effort and visited each species twice. For each subpopulation of each species, we sampled three 100 m x 4 m transects placed 8 m apart. This resulted in a total of 509 transects, each visited twice in 2016 to confirm with confidence the presence or absence of a species at a locality (MacKenzie and Royle 2005). Transects were placed to capture as much of the habitat as possible and divided into 20 m intervals and marked with wooden droppers. Transects were mapped out on GIS maps, using previous localities, as a desk top exercise and the waypoints recorded, these were then duplicated in field where possible or moved slightly and new waypoints taken when needed. Transects were placed parallel to each other. Opportunistic sightings of target species did not influence transect placement. However, where a transect targeting one species was found another on our target list, the latter was recorded as a new subpopulation. Two observers experienced in fynbos field botany walked 2 m apart within the transect and spent 8 min per 20 m section searching for the target plant species (Garrard et al. 2008). Positively identified and located species were recorded and marked with a GPS waypoint.

Due to the high diversity of species within the fynbos, subpopulation localities for two different species often coincided, so some transects were re-used to detect more than one species. Habitat and population condition data was recorded for all subpopulations, in the form of written text, photographs and specimens taken. Once a species was confirmed at a site, the subpopulation was recorded as extant (more than one individual plant present). All recorded data were added to the SANParks Data Repository System (<https://www.sanparks.org/scientific-services/data-information-resources/data-repository>) for ongoing monitoring.

## Data analysis

### Quantifying subpopulation decline

Decline was measured as the difference between the number of subpopulations found in the 2016 field survey and the number of subpopulations previously recorded in TMNP some dating back to the 1800s. Values of decline deviated from a normal distribution. Thus, we



used paired one-sided Wilcoxon signed rank tests (Zar 2010; Team 2022) to test for significant declines in numbers of subpopulations of the species in our sample as a whole (74 taxa) and within each family (Ericaceae 25 species; Proteaceae 24 species; Restionaceae 25 species). In addition, we used a bootstrap approach, resampling the data 1000 times with replication, to estimate the median reduction in subpopulations and confidence intervals (Canty and Ripley 2021; James et al. 2021).

### Associations between subpopulation persistence and red list status

We estimated persistence by the number and proportion of previously recorded subpopulations found in the 2016 field survey. We used Wilcoxon rank sum tests to test for differences in number and proportion of subpopulations re-found between species assessed as threatened on the Red List (i.e. categorised as VU, EN or CR) and those assessed as not threatened (i.e. categorised as LC or NT). We also tested for differences in subpopulation number between species assessed in different Red List categories using Analysis of Deviance under a Poisson Generalized Linear Model framework (Fox and Weisberg 2011), and we used ANOVA (Fox and Weisberg 2011) to test for differences in the proportion of subpopulations persisting in 2016 between species in different Red List categories. All analyses above were conducted in R version 4.1.1 or above (Team 2022).

### Associations between subpopulation persistence and extinction probability metrics

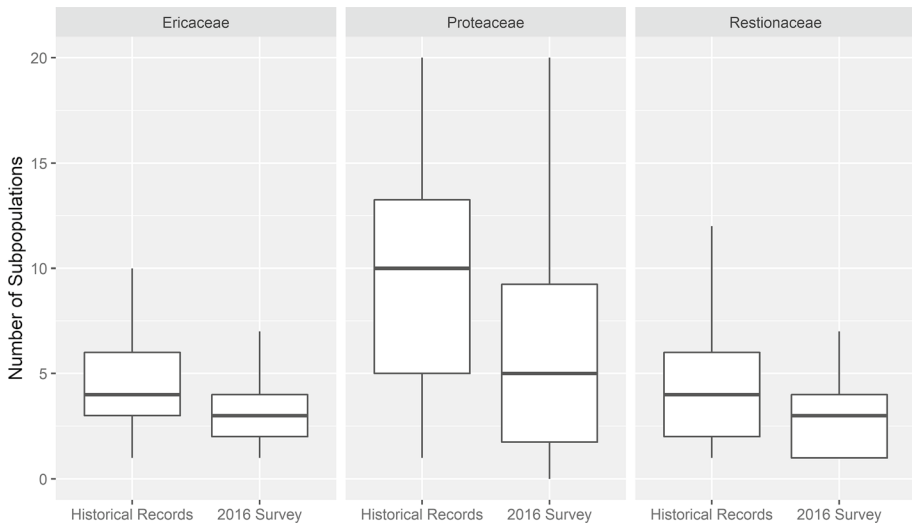
We assessed whether there is a Spearman's rank correlation between the persistence of subpopulations of the 74 randomly selected taxa and two estimates of extinction probability generated from applying the Solow (1993) equation. The Solow (1993) extinction probability equation assumes a constant collection rate over time, used to infer extinction following abrupt cessation of collections. We applied the Solow (1993) equation to a combined set of herbarium and survey data (Cowell 2018) to estimate the probability that the species is still extant and expected extinction year. The assumption of a constant collection rate over time was not met for most of the species, thus field work was required to gather new data.

We also tested for linear relationships between number of subpopulations found in 2016 and each of the two extinction probability estimates by fitting a Poisson regression, conditional on the number of subpopulations originally found. In all tests, validation of Solow (1993) required a significant ( $p < 0.001$ ), positive association (correlation or regression coefficient).

## Results

### Quantifying subpopulation decline

We found a decline of known subpopulations across families ( $V = 1378$ ,  $p < 0.001$ ) and within each family (Ericaceae:  $V = 136$ ,  $p < 0.001$ ; Proteaceae:  $V = 253$ ,  $p < 0.0001$ ; Restionaceae:  $V = 105$ ,  $p < 0.0001$ ; Fig. 3). Median survival proportions were approximately 75%. Bootstrap results showed on average two subpopulations were lost per species (95% CI 1 to 3 subpopulations lost per species). Species lost up to 15 subpopulations. Ericaceae averaged a reduction of one subpopulation per species (95% CI 1 to 3 subpopulations lost



**Fig. 3** The number of subpopulations in the historical records ( $\geq 1800$ ) and refound in the 2016 field survey for the 74 randomly selected taxa of Ericaceae, Proteaceae and Restionaceae, showing minima, maxima, medians, and upper and lower quartiles

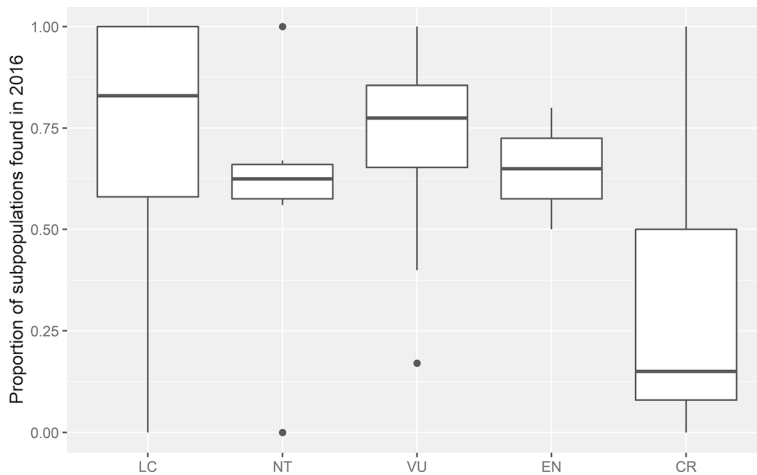
per species); Proteaceae averaged a reduction of five subpopulations per species (95% CI 2 to 8.5 subpopulations per species); Restionaceae averaged a reduction of one population per species (95% CI 0 to 2 subpopulations lost per species).

### Associations between subpopulation persistence and red list status

We found no evidence of significant differences between threatened and non-threatened species in terms of number of subpopulations present in 2016 or proportion of known subpopulations persisting until 2016. Similarly, numbers of subpopulations present in 2016 did not differ between species assigned to different South African Red List categories. When comparing subpopulation persistence across individual Red List categories, we noticed no difference between LC, VU and EN categories (Fig. 4). However, the proportion of known subpopulations persisting until 2016 was notably smaller for CR species than for other Red List categories, suggesting consistency between CR status and low persistence within TMNP. The proportion of subpopulations of CR species persisting until 2016 was 53% less than for LC species ( $p < 0.01$ ) and median persistence of subpopulations of CR species was just 15% (mean = 35%) compared to the LC median of 83% (mean = 75%). The number of subpopulations lost by CR species also differed significantly from that lost by LC species. On average, CR species lost 4 subpopulations more than LC species ( $p < 0.001$ , 95% CI CR lost 2 to 6 subpopulations more than LC).

### Associations between subpopulation persistence and extinction probability metrics

We found no relationship between subpopulation persistence in TMNP and the ‘probability that the species is still extant’ or ‘expected extinction year’ derived from the application of Solow (1993; Fig. 5). Both Spearman’s rank correlation and the Poisson regression model



**Fig. 4** The proportion of subpopulations found in 2016 by IUCN Red List categories. LC, Least Concern; NT, Near Threatened; VU, Vulnerable; EN, Endangered; CR, Critically Endangered. The proportion of subpopulations for CR species was smaller than for other Red List categories, indicating a relationship between CR status and low persistence within TMNP.

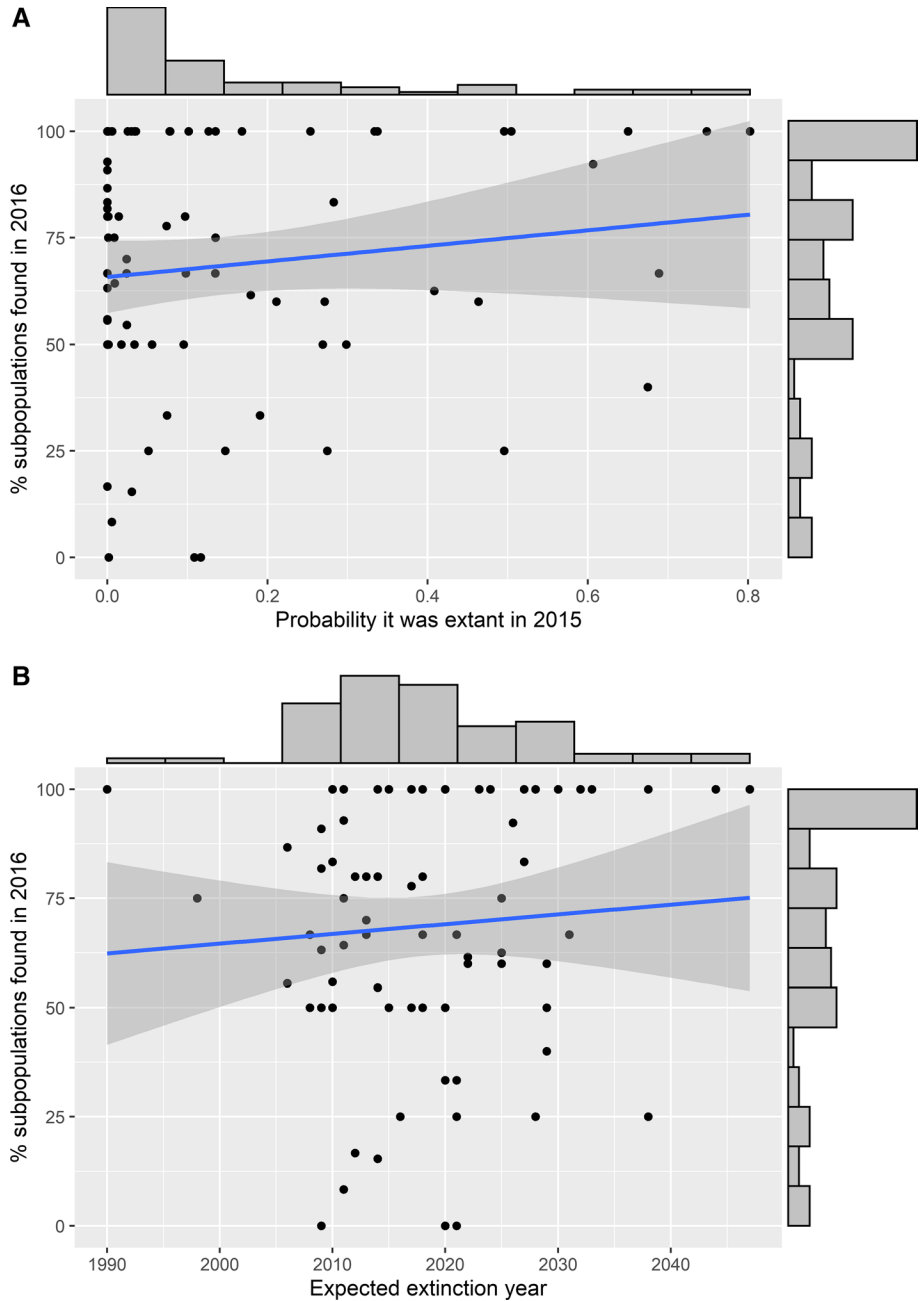
conditional on the number of subpopulations suggested no relationship between observed persistence and estimated extinction probability.

Of the Proteaceae taxa, 21 (88%) were predicted to be globally extinct by 2015, as were 8 Ericaceae and 5 Restionaceae taxa. Yet, our fieldwork refound 20 of the Proteaceae and all of the Ericaceae and Restionaceae taxa predicted to be extinct by 2015. These taxa were represented by an average of 5 subpopulations in the PA. Only three species were not detected at any of their targeted subpopulation locations: [*Protea grandiceps* (NT), *Leucadendron levisanus* (CR), and *Aulax cancellata* (LC)] (all Proteaceae). Failure to detect *Aulax cancellata* at its last known locality was due to a large wildfire in 2015. Although this species is not a post-fire resprouter, its seeds could be within the soil seed bank (Rebello 1995). For 26 species confirmed as extant within the Park we found only one or two subpopulations remaining; these included 13 species in the Least Concern category (this is both at the global and national scale as the species are endemic to the Cape Floristic Region and have the same threat status).

## Discussion

The Red List is an invaluable tool at the global and regional scales to conservationists around the world. It has played and continues to play a major role in guiding conservation policy and identifying areas for protection (Bachman et al. 2019). However, our work concurs with the IUCN guidelines not to use the Red List (global, regional or national) to prioritise species for monitoring a site is ineffective at the level of an individual PA. Yet in our experience it is still currently used in exactly this way.

Our field detection protocol, informed by locality descriptions on herbarium specimens, enabled us to verify that most targeted species were extant within TMNP but with a substantially reduced number of subpopulations. There was some compensation in the recording of new subpopulations, which supports the need for more and consistent field



**Fig. 5** The non-significant relationships between observed persistence (% subpopulations found in 2016) and the two extinction probability estimates: **a** probability that a species could be found in the wild in the year 2015; and **b** expected extinction year. The lines show the least-square fit and the shaded areas show the 95% confidence intervals. Histograms show the marginal distributions of the variables on their respective axes

work. We also concluded that, with the exception of species categorised as CR, species persistence in TMNP was largely independent of the species' Red List status. Species classified as LC, VU or EN on the South African Red List showed indistinguishable persistence in TMNP. Species categorised as CR had low persistence in TMNP and had, on average, lost four subpopulations more than LC species. Twenty-six species only had one or two subpopulations remaining and, of these, 13 species were listed as LC. The overall low correspondence between Red List status and species persistence in the PA confirms our expectations that the Red List is insufficient for prioritising species for monitoring and management within the PA. By using our field detection protocol, individual species can be prioritised for monitoring to assess the extent of the loss and mitigate further loss in TMNP. Our work illustrates that, independently of the Red List status, the future of a species may be far from secure, and it could indeed go extinct within the PA if the loss of subpopulations is not addressed.

To implement efficient conservation management strategies, PA personnel need to understand the distribution and threats to species within their borders, enabling them to target the species most likely to go extinct and allocate resources for maximum returns (Rout et al. 2010). In-field detection addresses the concerns that locality data contains biases and gaps resulting in the monitoring of those species in less need of monitoring and erroneously declaring a species extinct or extant in a PA (Moerman and Estabrook 2006). It addresses the temporal and spatial requirements of monitoring and data collection through ground-based sampling (Wintle et al. 2012). It meets the requirements of PAs with lower costs by reduced time and manpower while retaining integrity of the sampling data. Unfounded assumptions that a species is already extinct can have serious consequences for PAs and conservation actions: a species may be removed from a monitoring list and funding no longer received for its conservation, resulting in it inadvertently going extinct, a phenomenon known as the Romeo error (Collar 2016; Ungricht et al. 2005).

Through in-field surveys we corrected certain errors and problems with the historical herbarium data, including species localities that had been incorrectly georeferenced, which were replaced with accurate GPS waypoints (Rhoads and Thompson 1992). Similarly, errors in the survey data, including taxonomic and locality details, were addressed. Large citizen science survey datasets could also be validated through regular in-field surveys. Newly collected data from structured, in-field detection work on the habitat condition and threats to subpopulations of plant species can not only inform status of actions for the species within the PA but also be fed upwards into national or global assessments.

Our results indicate that there was little congruence between persistence and extinction values from the extinction equations regarding the expected survival and extinction dates of species in TMNP using the available data. This suggests that these extinction probability estimates were not reliable on our specific dataset. However, the extinction probability equations have been demonstrated to be robust and compatible with other datasets (Jaric and Ebenhard 2010; Gotelli et al. 2012). The performance of other extinction probability equations, though less generally agreed on as useful (Rivadeneira et al. 2009; Caley and Barry 2014), may generate different results. A different equation may work better with an increased number of sighting records, should more field surveys be made. Some of the probabilistic models which may yield better results include; Burgman (Burgman et al. 1995), Partial Solow (McCarthy 1998), Solow/Roberts (Solow and Roberts 2003) and Sighting Rate (McInerny et al. 2006). In a situation similar to TMNP it would seem they need to be refined and ideally foster methods that do work when the available data are very sparse (Roberts and Jaric 2020; Thompson et al. 2020).

In making the case for more investigations into possible relationships between extinction probability results, IUCN Red List status and in-field survey results, we do not intend to cast doubt on the utility of the probability equation used here. We caution that the poor performance of the Solow (1993) equation may be due to the dataset used. The Solow (1993) Optimal Linear Estimation was chosen because of the sparse dataset: the Solow (1993) equation works with a minimum of five records, while other approaches require more records to run optimally. A weakness in the Solow (1993) estimation is it assumes a constant rate of collection before an abrupt stop and then infers a collapse to extinction (Hamer et al. 2009). The combined dataset used in our study is very sparse, with the frequency of collections dropping off rapidly for periods of time for each family tested. The rapid decline in sightings records has been cited as the main reason for false or misleading extinction probability results (Rivadeneira et al. 2009; Solow 2005).

The Proteaceae lost more documented subpopulations than the other fynbos families in our study (Ericaceae and Restionaceae). Such temporal variation in the records can be caused by increased botanical interest in a species. Although species of the Proteaceae are being seen in the wild, they are not being collected or recorded on databases where they can be used in extinction probability models as up-to-date sightings records. This could be as a result of the Protea Atlas Project (Rebello 1995) which was initiated and completed from 1991 to 2001, to bolster the few sighting records of protea species prior to this. The role of research interests and funding commitments is evident here as sightings records were constant for a period while there was interest and funding. However, sightings declined after 2001 with the end of the project and the consistent use of the dataset. The Protea Atlas Project saw the publication of a very successful field guide that is used extensively. Little additional information has been collected since, as researchers and botanists rely on the Protea Atlas data and only a few records of the Red Listed or rare species are now captured. In the case of the 13 LC species (across the three families), for which we re-found only one or two subpopulations within the Park, it is entirely possible that other subpopulations persist in surviving remnants of native vegetation outside the Park, contributing to the persistence of the population as a whole but some may be undocumented and unprotected on these remnants.

## Recommendations

Our field detection results have implications for management and conservation and inform immediate management action such as habitat rehabilitation and risk prevention measures including path closure and invasive alien plant clearing (Farnsworth and Ogurcak 2006). We caution that these results should be used carefully and, as new data are acquired, they should be used to update and inform further analysis, within an adaptive feedback loop for management (Biggs and Rogers 2004).

Within a PA a metapopulation of a species consists of subpopulations (Hanski 1998). Conservation management actions aim to reduce the extinction risk of a species, yet effective actions to conserve the integrity of the metapopulation cannot be taken without knowing where the subpopulations occur and what threats they face. The establishment of thresholds of subpopulation reduction for narrow endemics requires exploration to help identify which species are in imminent decline and prioritise species to allocate limited resources for management. Management actions include monitoring and rehabilitating habitat or system, restoration of a species and attending to threats. Many international strategies and plans have been developed to help identify and prioritise species for management

attention and action (IUCN 2012). There is intense pressure on PAs, as with TMNP which is an urban park (Fig. 2) with factors such as recreational use of the natural area (paths, roads and tourist facilities), housing, waste dumping, invasive alien vegetation, wildfires and climate change affecting the survival of native species within the park.

Our results are a sobering reminder that long-term biodiversity monitoring and recording is extremely important and poor and sparse datasets have negative impacts for the use of conservation tools such as probabilistic models and Red List assessments. That the persistence of biodiversity may be directly affected by the lack of data is vitally important and action is needed to remedy this dearth of data (Schuttler et al. 2018). Shortcuts cannot be taken in using desktop analyses alone. Field verification is required, particularly when data used are sparse. Fieldwork provides insights and augments poor data for future use in conservation and research. There is an urgent need to build the capacity of conservation practitioners within countries and PAs to monitor and manage their sovereign biodiversity.

Key actions include:

- The importance of monitoring and data collection must be made explicit to those supporting conservation and PAs, thereby fostering an appreciation by funding agencies of the connections between basic field botany and critical conservation management.
- Substantial increases in financial support for field collections and monitoring are needed to supplement the long-term data already available.
- Collection efforts need to be improved with more botanically skilled people getting out into the field and recording species data in order to ensure that sufficient botanical data are available for evidence-based decision-making.

Given the challenges faced by conservationists, notably a lack of resources such as funds, people and time to undertake surveys and monitoring, there is an urgent need to find an effective solution.

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**Data availability** The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

**Code availability** Not applicable.

## Declarations

**Conflict of interest** All authors declares that they have no conflict of interest to disclose.

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