METHOD

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Poor-quality monitoring data underestimate the impact of Australia's megafires on a critically endangered songbird

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Abstract

Aim: Catastrophic events such as south-eastern Australia's 2019/20 megafires are predicted to increase in frequency and severity under climate change. Rapid, well-informed conservation prioritization will become increasingly crucial for minimizing biodiversity losses resulting from megafires. However, such assessments are susceptible to bias, because the quality of monitoring data underpinning knowledge of species' distributions is highly variable and they fail to account for differences in life history traits such as aggregative breeding. We aimed to assess how impact estimates of the 2019/20 megafires on the critically endangered regent honeyeater *Anthochaera phrygia* varied according to the quality of available input data and assessment methodology.

Innovation: Using Google Earth Engine Burnt Area Mapping, we estimated the impact of the megafires on the regent honeyeater using six monitoring datasets that differ in quality and temporal span. These datasets are representative of the variable quality of monitoring data available for assessing fire impact on 326 other threatened species; most are poorly monitored, and few have standardized, species-specific monitoring programmes. We found that assessments based on area of occupancy (AOO), extent of occurrence (EOO) and public sightings underestimated the fire impact relative to recent, targeted monitoring datasets: a MaxEnt model, sightings from a national monitoring programme and nest locations since 2015. Using an impact threshold of 30% of habitat burned, regent honeyeaters would not meet these criteria using estimates derived from EOO, AOO or public sightings, but would exceed the cut-off based on estimates derived from the targeted monitoring data that account for population density.

Main conclusions: To ensure that conservation prioritization has the greatest capacity to minimize biodiversity losses, we highlight the need to improve targeted, threatened species monitoring. We demonstrate the importance of using recent, standardized monitoring data to estimate accurately the impact of major ecological disturbances, particularly for declining, nomadic species undergoing range contractions.

1 | INTRODUCTION

Given impending climate predictions, extreme events such as Australia's 2019/20 megafires will become more common in coming decades (IPCC, 2018; Boer et al., 2020). Between 1 August 2019 and 31 March 2020, 12.6 million hectares of bushland burned and more than a billion animals are estimated to have perished in eastern Australia (Wintle et al., 2020). Initial assessments estimate 327 threatened species have been impacted by the fires, defined as taxa having >10% of their known or predicted range burnt. These figures include 31 species listed as "critically endangered" under federal biodiversity legislation (Commonwealth of Australia, 2020a, 2020b). The development of rapid and strategic conservation responses to such unprecedented events will play an increasingly crucial role in limiting future global biodiversity loss (Wintle et al., 2020).

Rapidly and accurately assessing the relative impact of extreme events on threatened species is crucial not only for prioritizing emergency conservation investment (Wintle et al., 2020) but also for informing how, where and when these funds should be utilized to maximize conservation returns (Bottrill et al., 2008). In this instance, it is desirable to devise a quick, simple and repeatable metric of relative impact assessment that decision-makers can use to help identify those taxa in most need of conservation effort (Ward et al., 2020). In other words, how is conservation prioritization in response to Australia's bushfires best implemented, given the unprecedented scale of the fires and the short time frame required to commence on-ground action?

Determining how to rapidly and accurately prioritize conservation funds across entire ecosystems is challenging. In such circumstances, an attractive option is to determine a species' predicted distribution using area of occupancy (AOO), extent of occurrence (EOO) or species distribution models (SDMs), overlay remotely sensed fire severity mapping and calculate the percentage of a species' range that has been fire-affected (Ward et al., 2020). The principal advantage of utilizing these methods as a first-phase assessment is that they can be implemented using data that already exist for the vast majority of affected species. Consequently, the Australian Government have adopted these methods, alongside consideration of pre-fire imperilment and other threats, to help guide the conservation response to the bushfires (Commonwealth of Australia, 2020a, 2020b; Legge et al., 2020).

Despite practical advantages, using species distribution data to assess bushfire impacts may be subject to systematic biases (Legge et al., 2020). The quantity and quality of baseline monitoring data varies substantially across fire-affected species, but is poor for the majority of taxa (Legge et al., 2018; Scheele et al., 2019). Interspecific differences in life history traits such as ecological niche, dispersal capacity, sociality and competitive ability (i.e. body size) will also affect species' capacity to recover from the impacts of bushfire, both with and without conservation assistance (Legge et al., 2020; Woinarski & Recher, 1997). Highly mobile species, whose functional habitat for survival and reproduction in any given year may be only a tiny proportion of their entire range (Runge et al., 2014; Webb et al. 2017), will be particularly at risk if bushfire affects those critical areas (at least until habitats in such areas have recovered). If high-quality monitoring data on breeding activity for such species are not available, bushfire impacts may be underestimated using species distribution methods. Underestimating bushfire impact due to variability in the quality of available monitoring data could cause prioritization decisions to misallocate conservation resources, potentially overlooking actions that could help prevent extinction (Woinarski et al., 2017).

Here, we assess how estimates of fire impact vary with the quality of available monitoring data for a critically endangered, nomadic songbird, the regent honeyeater. Specifically, we assessed fire impact using six monitoring datasets that differ in quality and temporal span: EOO, AOO, a MaxEnt SDM, public sightings, targeted monitoring and nest monitoring. We predicted that fire impact would be underestimated using methods that draw on datasets with longer temporal spans and opportunistic monitoring data such as public sightings. These datasets may fail to account for habitat functionality (e.g. annual rainfall and dynamic resource distribution) and species-specific life history traits (e.g. flocking and group nesting). Consequently, they contain biases in survey effort that could result in overestimation in habitat availability or the size of the species' contemporary range.

2 | METHODS

2.1 | Study species

The regent honeyeater is endemic to Australia's eastern seaboard and was abundant and widespread as recently as 60 years ago (Franklin et al., 1989). Extensive land clearing has led to a rapid population decline, with fewer than 350 individuals estimated to persist in the wild today in a range exceeding 600,000 km^2 from Victoria to southern Queensland (Crates et al., 2021). Regent honeyeaters nest primarily in association with flowering events in a small number of *Eucalyptus* tree species, which show very high spatio-temporal variation in flowering phenology (Birtchnell & Gibson, 2006; Franklin et al., 1989). Regent honeyeaters have evolved a highly nomadic life history to track these dynamic nectar resources. Individuals can travel hundreds of kilometres and typically nest in loose aggregations when flowering conditions allow. Aggregative nesting could help optimize settlement decisions, antipredator defence and the cultural transmission of information amongst conspecifics (Crates

et al., 2017). The entire population represents a single genetic management unit, but the core remaining population persists within the greater Blue Mountains area of central/eastern New South Wales (Crates, Olah, et al., 2019; Crates, Rayner, et al., 2019).

2.2 | Fire severity mapping

We used the Australian Google Earth Engine Burnt Area Map (GEEBAM, Commonwealth of Australia, 2020a, 2020b), derived remotely from Sentinel 2 satellite imagery. GEEBAM calculates the difference between pre-fire (April 2018 to April 2019) and post-fire (November 2019 to May 2020) normalized burn ratio using near-infrared and shortwave infrared spectral data. Fire severity classes reported in GEEBAM include low (little change), medium (crown unburnt), high (crown partially burnt), very high (crown fully burnt) and unclassified (i.e. non-native vegetation or areas outside of the fire footprint). Further details of the fire severity mapping are available at https://www.environment.gov.au/system/files/pages/a8d10ce5-6a49-4fc2-b94d-575d6d11c547/files/ageebam.pdf.

2.3 | Regent honeyeater monitoring datasets

We used six monitoring datasets based on varying degrees of data quality (Table 1). Since 2015, we have used these datasets to establish a standardized, targeted and range-wide monitoring programme for the regent honeyeater—the National Regent Honeyeater Monitoring Program (NRHMP). The aim of the NRHMP is to increase the quality and quantity of monitoring data available for the regent honeyeater, with the ultimate goal of informing targeted conservation action to help prevent the species' extinction in the wild. Developed over the past 6 years through extensive field surveys, habitat modelling and expert elicitation, the NRHMP now surveys over 1,300 sites stratified in known or potential breeding areas throughout the species' contemporary breeding range (Figure S1) during the Austral spring and early summer (Crates, Rayner, et al., 2019). The NRHMP

sampling regime aims to account for both the nomadic life history and the breeding biology of the regent honeyeater (Crates, Terauds, et al., 2017). The six monitoring datasets based on wild birds are as follows:

1) *Extent of occurrence (EOO)*: minimum convex polygon of verified regent honeyeater sightings since 1990, sourced from BirdLife Australia and used for IUCN classification.

2) *Area of occupancy (AOO):* 1 km × 1 km grids containing a verified regent honeyeater sighting since 1990, sourced from BirdLife Australia.

3) *Public sightings*: Location of wild regent honeyeaters detected by the general public and reported to BirdLife Australia between January 2015 and December 2019. We refined the database to remove duplicate records of the same individuals in each year. The final database contained 152 point locations for 290 individuals, with potential for duplicate records of the same individuals across different years.

4) *MaxEnt species distribution model*: A raster model with 1-km^2 resolution developed using public sightings of regent honeyeaters between 2000 and 2010, and 6-month lagged rainfall data (Stojanovic et al., 2021, in revision). SDMs were individually run for 6-month periods between July and December, or "time slices" ($n = 22$). The final consensus model flags all cells modelled as suitable breeding habitat in at least 11 of the 22 seasons (i.e. probability of suitability greater than 50%; see File S1 for further details on the modelling process).

5) *Public and NRHMP sightings:* Location of wild regent honeyeaters detected either by the general public or through the NRHMP between 2015 and 2019. We refined the database to remove duplicate records of the same individuals in each year. The final database contained 416 point locations for 899 individuals, with potential for duplicate records of the same individuals across years.

6) *Nests*: Location of regent honeyeater nesting attempts that reached the egg stage, located either through the NRHMP ($n = 138$) or by members of the public ($n = 6$) since August 2015 (Crates, Rayner, et al., 2019). We also included three nests involving a captive-released regent honeyeater if the partner was of wild origin.

TABLE 1 Components of data quality for regent honeyeater monitoring datasets

Component of data quality	Poor quality	High quality
Spatial extent	Isolated, random or only part of a species' range covered. Mostly on publicly accessible land	Sampling covers full extent of the range, regardless of land tenure
Spatial resolution	Low	High
Temporal span	Long (>10 years old). Contains potentially outdated records	Short (<5 years old). Contains only contemporary records
Sampling regime	Random, opportunistic, does not account for species-specific traits	Stratified, systematic sampling accounts for species-specific traits
Data source	Mostly derived from public sightings	Mostly derived from a targeted monitoring programme
Data resolution	Presence/absence	Abundance
Life history resolution	Does not include breeding information	Includes breeding information

2.4 | Spatial and statistical analysis

We used ArcGIS Desktop 10.8 (ESRI, 2020) for all geoprocessing. Prior to spatial analysis, we projected all spatial data to EPSG: 3,577 (GDA94—Australian Albers), ensuring equal area between raster cells. We resampled the GEEBAM data to 40-m resolution during projection, then converted from raster to polygon for later use in analysis. We clipped the EOO minimum convex polygon to the coastline and converted each pixel of the MaxEnt raster (i.e. 0 | 1; unsuitable | suitable) into a distinct polygon using a three-step procedure involving: i) raster to point conversion; ii) creation of a fishnet based on the raster dimensions; and iii) conversion of point feature to polygon with raster values, using the raster to point output for labelling the final polygons. We projected all point data for known nests, public sightings and NRHMP sightings from WGS84 to Australian Albers, buffered by 500-m radius, and converted from circular polygons to square polygons. We used a 500-m buffer to account for foraging and dispersal movements of regent honeyeaters (Geering & French, 1998) and created square buffers around point data to ensure consistency and comparability between reporting of impacts based on raster data (1 km² cells) versus vector distribution data.

Because the sightings and nesting databases confer data at the individual level, some of the buffered polygons overlapped. We therefore created a complementary fire impact estimate for the sightings databases by dissolving the boundaries around overlapping buffered sighting locations.

To calculate the proportion of habitat within each dataset impacted by the 2019/20 bushfires, we overlaid each projected vector distribution dataset (EOO, AOO, the MaxEnt model, public sightings, NRHMP plus public sightings, and nests) with the GEEBAM fire severity mapping using a "*Summarize Within*" function in ArcPro (ESRI, 2020).

Using R version 3.4.3 (R Core Team, 2017), we fitted a logistic regression model via package *lme4* v1.1–21 (Bates 2019) to compare the proportion of fire-affected 1-km grid cells in each database.

3 | RESULTS

The 2019/20 bushfires burned 71,011 square kilometres of bushland within the extent of occurrence of the regent honeyeater (based on records since 1990), representing an estimated 13% of the species' EOO (Figures 1 and 2, Tables 2 & Table S1). Burn severity was
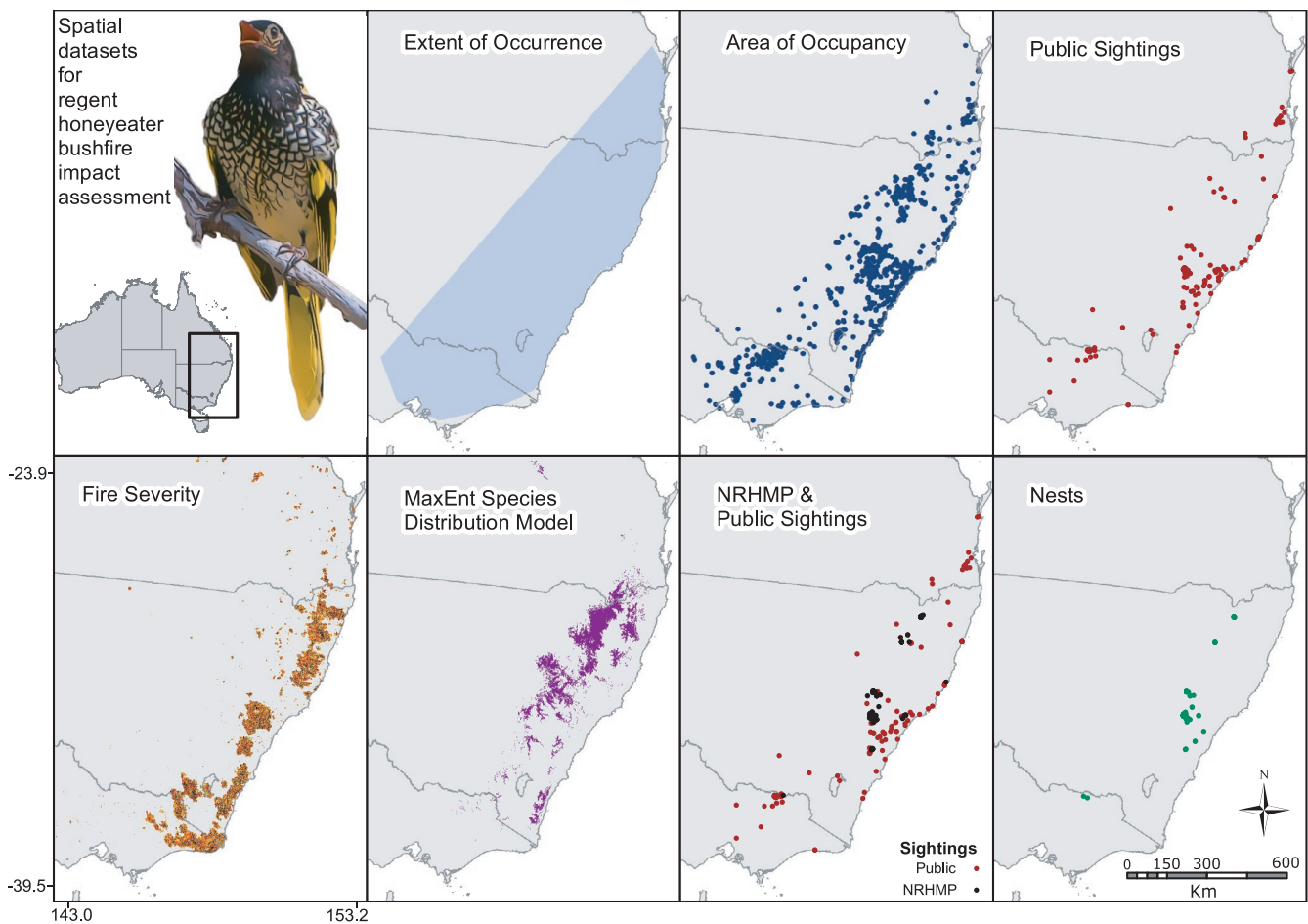
FIGURE 1 Spatial relationship between 2019/20 Google Earth Engine Burnt Area Mapping and regent honeyeater monitoring datasets. For sightings and nests, points denote locations. Due to map scale, not all points are visible and points shown appear much larger than their actual spatial extent. Inset in top left panel denotes study area on a national scale. Numbers bordering bottom left panel denote decimal latitude and longitude extent of the study area to the nearest 0.1 degree

estimated to be high or very high for 54% of the burned area. For the species' area of occupancy, 15.6% of 1-km^2 grid cells containing a regent honeyeater record since 1990 and 10.6% of the total area within those cells were affected by fire. On average, 39% of the area within fire-affected AOO cells had burned with high or very high severity. For public sightings of regent honeyeaters since 2015, 17% of 1-km grid cells buffering the location of a sighting were fire-affected. Of fire-affected public sightings grid cells, on average 27% of the area had burned at high or very high severity (Figures 1 and 2, Table 2). For the MaxEnt species distribution model, 24% of 1-km grid cells modelled as suitable habitat were fire-affected, of which a mean area of 47% burned at high or very high severity (Figures 1 and 2, Table 2). Combining public sightings with sightings from the NRHMP, the proportion of fire-affected cells increased to 37%, with 22% of fire-affected cells burnt at high or very high severity. The assessment based on regent honeyeater nest locations since 2015 returned the most severe fire impact estimate, with 44% of nest cells having been affected by fire (Figures 1-3). Twenty-three per cent of habitat near fire-affected nests burned with high or very high severity. Logistic regression revealed the proportion of fire-affected cells in each database differed significantly (Table 3, Figure 2).

4 | DISCUSSION

Minimizing biodiversity losses from catastrophic events such as Australia's megafires depends on the implementation of urgent conservation actions for severely affected species (Wintle et al., 2020). Implementing effective prioritization to identify priority species from an estimated 327 fire-affected threatened taxa (Legge et al., 2020)

FIGURE 2 Estimated fire severity impact by regent honeyeater monitoring database. Columns denote proportion of 1-km^2 fire-affected cells within each spatial database. Colours within columns denote mean proportion of 40-m^2 GEEBAM cells by burn severity class within 1-km^2 fire-affected cells

requires accurate estimates of bushfire impact, alongside accurate information on pre-fire imperilment, other threats and population trajectory. This in turn ideally requires high-quality, species-specific monitoring data (Boer et al., 2020; Bottrill et al., 2008). However, available occurrence records and monitoring data for many fire-affected taxa are poor (Scheele et al., 2019), which reduces confidence in the results of conservation prioritization and potentially misallocates conservation resources. We demonstrate with our case study of the critically endangered regent honeyeater that estimates of fire impact can vary dramatically depending on the input dataset. Specifically, we found that methods to estimate species distributions that are available for many vertebrate species, such as EOO and AOO, underestimated fire impact by at least half in comparison with records derived from a targeted, contemporary monitoring dataset. We examine some of the underlying discrepancies between the estimates obtained from the different datasets and outline recommendations for improvement.

The datasets that led to the most severe underestimation of impact were AOO and EOO, which utilized verified sightings data from the 1990's onwards (including the targeted contemporary monitoring data), broadly in line with the temporal period considered in other assessments (Legge et al., 2020). However, rapid population decline has seen a concurrent range contraction in regent honeyeaters during this period (Commonwealth of Australia, 2016), meaning the species now rarely occurs in areas on the southern and western edge of its EOO, which were largely unaffected by fire in the summer of 2019/20 (Figure 1). Hence, fire impact estimates were lower for these datasets. In contrast, estimating fire impact using the contemporary occurrence records for the species collected since 2015 provides a more accurate assessment of fire impact on the species' contemporary range. While initial assessments based on AOO or EOO represent a good first step and are achievable for many species, they need to be treated with caution, especially when species have experienced recent range contractions. When augmented with additional information such as pre-fire imperilment (as done by Legge et al., 2020), such methods can identify priority species such as the regent honeyeater despite relatively low estimates of fire overlap.

The databases used to calculate EOO and AOO are underpinned by public sightings data, much of which is spatially biased towards areas that are publicly accessible and where regent honeyeaters are known to occur (Commonwealth of Australia, 2016). The bias in public sightings data is highlighted by comparing the number and distribution of contemporary regent honeyeater sightings (2015 to 2019) contributed by the public versus those contributed by the National Regent Honeyeater Monitoring Program. Less than 37% of contemporary sighting locations and less than 33% of individual bird sightings were obtained from the public, and many of these were clustered around suburban areas, encompassing mostly non-breeding birds moving through the landscape. Extensive land clearing combined with competition from larger honeyeater species means regent honeyeaters are now severely restricted in their breeding locations (Crates, Rayner, et al., 2019; Ford et al., 2001; Figure 1). Many
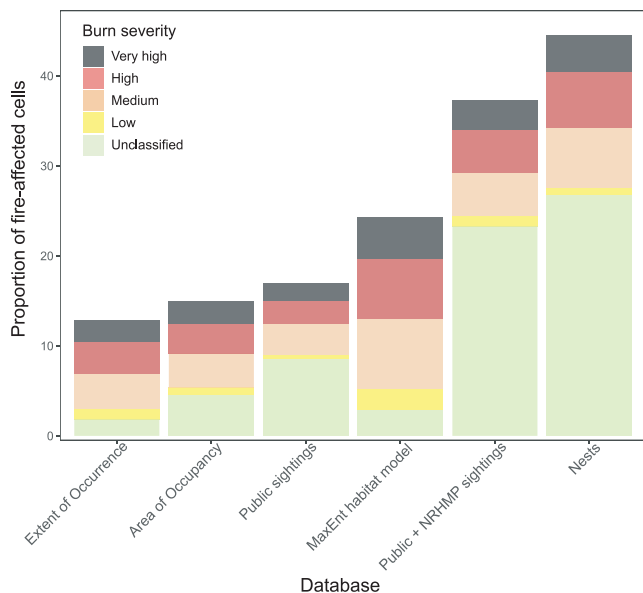
TABLE 2 Summary of the spatial databases used to evaluate the impact of Australia's 2019/20 megafires on regent honeyeater habitat. See Table S1 for area of occupancy and extent of occurrence impact estimates based on data from 1996 onwards

Dataset	Data quality	Availability for other taxa	No. of 1-km^2 cells	% of cells fire-affected	Area (km^2)	% area fire-affected[a]
Extent of occurrence (EOO)	Poor	Broad	606,382	14	605,690	9
Area of occupancy (AOO)			1,226	15	1,226	11
Public sightings			155	17	121	9
MaxEnt model			50,075	24	50,075	21
Public+NRHMP sightings			416	37	199	13
Nest locations	High	Sparse	144	44	53	22

[a]Differences between % of fire-affected cells and % fire-affected area is because it was not always the case that 100% of the area within fire-affected cells was burnt.

important breeding areas, including the Burragorang, Goulburn and Capertee River Valleys, are mostly inaccessible to the public (Crates, Rayner, et al., 2019). Since 2015, less than 5% of regent honeyeater nests were found by the public, meaning without a targeted monitoring programme, the distribution of contemporary breeding activity would be largely unknown. These breeding data are critical for not only assessing accurately the impact of the megafires on regent honeyeaters, but also informing where to best implement post-fire recovery actions such as habitat restoration, nest protection and pest species management (Wintle et al., 2020).

Public sightings data also often underpin species distribution models, including our MaxEnt model (Araújo et al., 2019). Although SDMs are useful for estimating bushfire impacts (Ward et al., 2020), they are prone to overestimating habitat availability if sampling biases are not accounted for (Araújo et al., 2019; Kramer-Schadt et al., 2013). Species distribution models are particularly likely to overestimate habitat availability for wide-ranging habitat specialists (Webb et al., 2017), which form a large component of Australia's vertebrate fauna (Runge et al., 2016; Welbergen et al., 2020). Many of these nomadic species rely on the co-occurrence of multiple dynamic habitat features, such as eucalypt blossom, water and tree hollows, for potential habitat to become functional habitat (Webb et al., 2017). In any given year, only a tiny proportion of a nomadic species' breeding range may be suitable (Webb et al., 2014), and if these areas are fire-affected, the capacity for such species to breed successfully in the years during and following fire may be seriously limited (Runge et al., 2014). If these critical areas are unknown due to a lack of systematic, spatially extensive monitoring, then bushfire impacts on nomadic species could be substantially underestimated.

Our results raise a question central to efforts to assess the biodiversity impact of major catastrophic events such as Australia's recent megafires: *What is the appropriate input dataset upon which to base assessments?* Outdated, incomplete or inaccurate input datasets risk biasing assessments, which could either over- or underestimate impacts. As we observe for the critically endangered regent honeyeater, underestimated fire impacts may be common in species that have undergone rapid declines and for which occurrence records that are not yet a decade or two old are already redundant because the population has continued to decline. This is worrying,
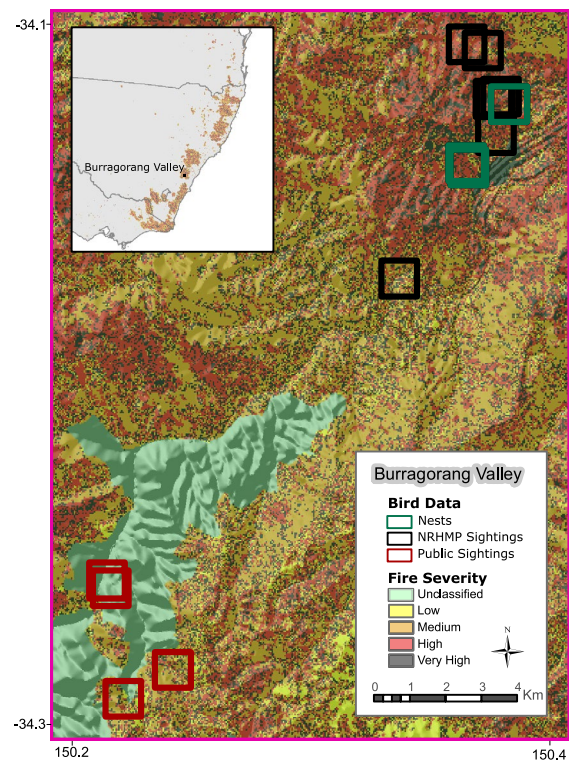
FIGURE 3 Location of regent honeyeater public sightings, National Regent Honeyeater Monitoring Program (NRHMP) sightings and nests relative to bushfire impacts in the Burragorang Valley. Squares denote 1 km × 1 km buffer around each sighting or nest location for fire impact analysis. Axis numbers denote decimal latitude and longitude of the study area to the nearest 0.1 degree

as these species may be the ones most vulnerable to extinction due to stochastic events (Melbourne & Hastings, 2008). For species experiencing rapid declines, we recommend assessments use only reliable contemporary distribution data (ideally less than a decade old and preferably less). For species without targeted monitoring programmes, available data within this time period are likely to be insufficient (i.e. since 2015, only 5% of known regent honeyeater nests were found by the public). In the case of rapidly declining species, the assumption that earlier occurrence records mean previously occupied areas still contain functional habitat that can be exploited

TABLE 3 Logistic regression summary of the proportion of fire-affected 1-km cells within each regent honeyeater monitoring database. Effect sizes shown are relative to the extent of occurrence database

Database	Beta	SE	Z	p
Area of occupancy	0.12	0.08	−1.53	.13
Public sightings	0.26	0.22	1.19	.24
MaxEnt habitat model	0.70	0.02	44.20	<.01
Public & NRHMP sightings	1.33	0.10	12.97	<.01
Nests	1.69	0.17	10.08	<.01

cannot be met, as species declines are generally non-random in environmental space, because threats or species capacity to tolerate threats varies across the environment (Scheele et al. 2017).

Our assessment—which focused on one of Australia's flagship species for conservation, the regent honeyeater—revealed that bushfire impact estimates are highly sensitive to the input datasets used. This result highlights the pressing need for improved threatened species monitoring and reinforces the need for better knowledge on species ecology and distribution (Scheele et al., 2018; Legge et al., 2020). That such large discrepancies were observed for a critically endangered bird species—a class for which monitoring information is typically most available—indicates that such issues are likely to be widespread. As one in four of Australia's threatened vertebrate species are not monitored at all (Scheele et al. 2019), it is likely that even greater discrepancies would be found in less well-known groups, such as plants, fish and invertebrates. This is a major shortcoming because many threatened species require targeted monitoring programmes that account for their specific life history traits and ecological requirements to provide robust information on their contemporary distribution (Cottee-Jones et al., 2016). While such monitoring programmes require adequate resourcing, targeting conservation actions for rare and nomadic species in space and time is extremely challenging without accurate information on their distribution and the threats jeopardizing their persistence (Crates et al., 2018). Our study exemplifies the multifaceted benefits that can be derived from quality threatened species monitoring programmes and demonstrates the conservation value of investing in such programmes for other poorly monitored species.

PEER REVIEW

The peer review history for this article is available at https://publons.com/publon/10.1111/ddi.13385.

DATA AVAILABILITY STATEMENT

Summary tables of fire analysis for each spatial dataset and R code are available via the Dryad digital repository: https://doi.org/10.5061/dryad.7m0cfxpv1. AOO / EOO mapping and public sightings data are managed by BirdLife Australia. MaxEnt models and NRHMP datasets are available from the lead author upon request.

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BIOSKETCH

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SUPPORTING INFORMATION

Additional supporting information may be found online in the Supporting Information section.