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Insectivorous bat activity over swimming pools retrofitted for wildlife

Year 6: 2022

Report prepared for Ku-ring-gai Council

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Prepared by the NSW Department of Primary Industries (Forest Science)

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Report prepared August 2022

More information

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Acknowledgments

Sampling of converted pools, non-converted pools, backyards, golf course dams, sediment ponds/wetlands and bushland was undertaken by Ku-ring-gai Council. In 2022, creeks were also sampled by Ku-ring-gai Council. We thank all residents and land managers for providing access to land for bat monitoring.

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Executive summary

Within the urban matrix, green spaces play a valuable role in mitigating the detrimental impacts of urbanisation on biodiversity. Swimming pools converted to wildlife ponds ('converted pools') may provide a refuge for fauna, including birds and frogs, but also foraging habitat for bats. Bats occupy high trophic levels and are considered indicators of environmental change and can reflect wider-scale impacts on other biota. Initial assessment of converted pools in year 1 of monitoring revealed value of these pools for insectivorous bats, with nightly bat species richness and activity almost two-times greater than creeks in spring. Here we report on years two to six (2018-2022) of monitoring that assessed the value of converted pools relative to other habitat elements (natural creeks, non-converted pools, sediment ponds/wetlands, golf course dams, backyards/parks and bushland) in the urban matrix using acoustic surveys, all of which were carried out in autumn unlike year one. This is the first study of its kind that documents spatial and temporal trends for urban insectivorous bats across multiple habitat elements. In 2022, 57 sites were sampled. Across six years of monitoring, sixteen bat taxa were recorded, including eight threatened species. Nightly bat species richness and activity was greatest at golf course dams and lowest at bushland creeks. Species richness fluctuated among years at all treatments but these fluctuations were not significant, with the exception of backyards/parks which had a 60 % reduction in species richness in 2021 compared to 2018. The trajectory of trends for total bat activity also fluctuated among years at all treatments. Golf course dams had 67 % greater activity in 2019 (drought) compared to 2018 whereas sediment ponds/wetlands had 69 % higher activity in 2019 compared to 2022, suggesting these areas may be especially important during periods of drought. Activity at converted and non-converted pools was mostly attributed to urban adapted generalist species, Gould's Wattled Bat *Chalinolobus gouldii* and Eastern Bent-wing Bat *Miniopterus orianae oceanensis*, whereas the specialist bat species Large-footed Myotis *Myotis macropus* was only detected at golf course dams, sediment ponds/wetlands, large waterways and creeks. Although three species (*C. dwyeri*; *R. megaphyllus* and *S. rueppellii*) were recorded at converted pools but not at non-converted pools or backyards/parks, detections were uncommon. The annual trend for activity was species- and habitat-specific in years 2-6, except *M. orianae oceanensis*, which maintained low activity or showed a >56 % decline in activity after black summer fires in 2020 and 2021, with some recovery in 2022. We recommend that monitoring be continued to track trends in

bat activity, particularly for *M. orianae oceanensis* that roosts within the LGA, but also to assess how the value of converted pools and other habitat types changes over time for bats, including the influence of recent wet years of 2021-2022. We suggest that sites used in the current study should be resurveyed annually and that systematic annual monitoring and targeted research is undertaken at key *M. orianae oceanensis* roosts. Targeted research is also needed to identify roosting habitat for other species via radio-tracking or acoustic surveys of subterranean structures (e.g., stormwater culverts). These sites that are considered critical habitat for bats may then be incorporated into annual monitoring.

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Introduction

A major cause of decline in biodiversity is the loss and fragmentation of habitat resulting from urban development (Garden et al. 2006). Remnant bushland in urban areas often occurs in small patches and can be highly fragmented, with a lack of connectivity between these patches (New and Sands 2002; Stenhouse 2004). However, it is in these remnants that urban-sensitive species are often restricted (How and Dell 1993, 1994; Garden et al. 2006). Conversely, within the urban matrix, generalist species tend to dominate, while habitat and dietary specialists tend to decline or become locally extinct (How and Dell 1993, 2000; White & Burgin 2004; Tait et al. 2005; Garden et al. 2006).

Within the urban matrix, green spaces play a valuable role in mitigating the detrimental impacts of urbanisation on biodiversity (Goddard et al. 2010). Gardens form a major component of green space in urban areas and can contribute considerable habitat elements, including trees, nest boxes and ponds (Davies et al. 2009). The structural complexity of gardens is closely associated with bird (Daniels and Kirkpatrick 2006) and invertebrate (Smith et al. 2006) abundance and diversity. Consequently, supplementing existing gardens with additional habitat elements may be beneficial for urban biodiversity. However, space required to implement these features can be scarce, particularly in high-density urban areas.

Swimming pools in urban areas represent an opportunity to create an additional habitat element in gardens, via conversion to wildlife ponds. Maintenance or removal of swimming pools can require significant time and money. Conversion of swimming pools to wildlife ponds (hereafter 'converted pools') can provide supplemental habitat for fauna, including frogs, bats and birds. Furthermore, conversion of a pool does not destroy the asset, as it is possible to restore its function as a swimming pool if required in the future. In the Ku-ring-gai local government area (LGA), it is estimated that there are 16,000 swimming pools. A 'Pool to Pond' program was developed by the Council, providing residents advice regarding conversion of swimming pools to wildlife ponds as well as provision of native fish and aquatic plants. In the 16 years of operation, Council have provided advice or supported the transition of over 50 swimming pools to wildlife ponds through the 'Pool to Pond' program.

Insectivorous bats are a diverse fauna group that use echolocation to navigate their habitat and detect prey (mostly insects such as moths, beetles and flies) (Churchill 2009). Bats occupy high trophic levels and are considered indicators of environmental change (Jones et al. 2009). Furthermore, the importance of water to bats is well-established (Korine et al. 2016). Consequently, bats represent key taxa that may respond to the provision of converted pools in urban gardens.

This is the 6th year of monitoring to assess the value of converted pools to bats in the Ku-ring-gai LGA. In year one (2017), we recorded the diversity and activity of bats at converted pools, non-converted pools and creeks in the austral summer. However, since some species are more abundant in northern Sydney in autumn (Gonsalves and Law 2018), monitoring in subsequent years (2018-2022) was undertaken in autumn and also sampled other habitat types within the urban landscape (backyards/parks, golf course dams, sediment ponds/wetlands, large waterways and reference bushland) to provide context for converted pool sites. In year four (2020), restrictions associated with the COVID-19 pandemic only allowed for limited sampling of some habitat types. In 2021 and 2022, sampling was more extensive, though for the former still with limited access to some converted pool sites. In this report, we assess the value of converted pools to bats by comparing diversity and bat activity at converted pools with other habitat types available to bats in the Ku-ring-gai LGA. We also report on trends for years 2-6 of monitoring as these sampled a major component of Sydney's urban bat fauna (i.e., *M. orianae oceanensis*) that were under-sampled in year one of monitoring (Gonsalves et al. 2018). This is the first study to document spatial and temporal trends for bats in Sydney.

Methods

Study area and design

The study was carried out in the Ku-ring-gai LGA, situated just 16 km North of Sydney's CBD. The LGA is moderately large (~8544 ha) and 'leafy', extending from Roseville in the south to Wahroonga in the north, and from St Ives in the east to Lane Cove National Park in the west. Natural area reserves represent approximately 1,150 ha of the LGA and many of these are contiguous with National Parks including Ku-ring-gai Chase, Garigal, Lane Cove and Dalrymple-Hay Nature Reserve (Ku-ring-gai Council 2016). The LGA also spans three of Sydney's major catchments: Lane Cove River, Middle Harbour and Cowan Creek. These catchments are drained by approximately 220 km of creek lines that occur in the LGA, with many of these in semi-natural to natural condition, particularly those that occur in private easements, parkland and bushland reserves. The LGA has a population of ~126,550 residents within the built area, of which 72% is low density housing, 13% medium density and 15% high density (Ku-ring-gai Council 2020).

In 2018 and 2019, a total of 65 sites (Fig. 1, Table 1) were sampled, representing bushland creeks (Fig. 2a), swimming pools that were converted to wildlife ponds (hereafter 'converted pools', Fig. 2b), non-converted pools (Fig. 2c), golf course dams (Fig. 2d), wetlands/sediment ponds (Fig. 2e), backyards (Fig. 2f) and bushland (Fig. 2g). Most sites were sampled in both years, with four sites sampled in one of the two years. In 2020, COVID-19 restrictions allowed for a subset (56 %) of sites to be sampled, with access to some backyards and public spaces (golf courses) unavailable. Urban parks were sampled as an alternative to backyards that were unable to be sampled, and large waterways were included as an additional habitat type (Fig. 2h, Table 1). In 2021 and 2022, more extensive sampling was possible (n=62 sites and n=57 sites, respectively), though some habitat types were under sampled compared to years 2 & 3 due to access limitations associated with the COVID-19 pandemic in 2021.

Characteristics of each site were recorded *in situ* during bat sampling or later extracted using spatial data and ArcGIS (ESRI). For all sites, distance (m) to nearest natural creek line, geology (% sandstone and % shale) and amount of bushland within 500 m of sites, and projective foliage cover (% cover) within a 20 m radius of the detector was recorded. For sites on water (converted pools, non-converted pools, golf course dams, sediment ponds/wetlands, large waterways and creeks), cover of emergent vegetation (% cover) and pool size (m^2) were also recorded. Creek and constructed pools ranged in size from 10-5500 m^2 and were 4-687 m from the nearest natural creek line. Emergent vegetation cover over waterway sites ranged from 0-100 %, while bushland cover within 500 m of each site ranged from 7-69 %, with most bushland located in areas predominantly comprised of sandstone geology, as was the case for backyards with and without pools (non-converted and converted). Golf course dams and sediment ponds/wetlands were also generally located on sandstone geology or at the interface of sandstone and shale.

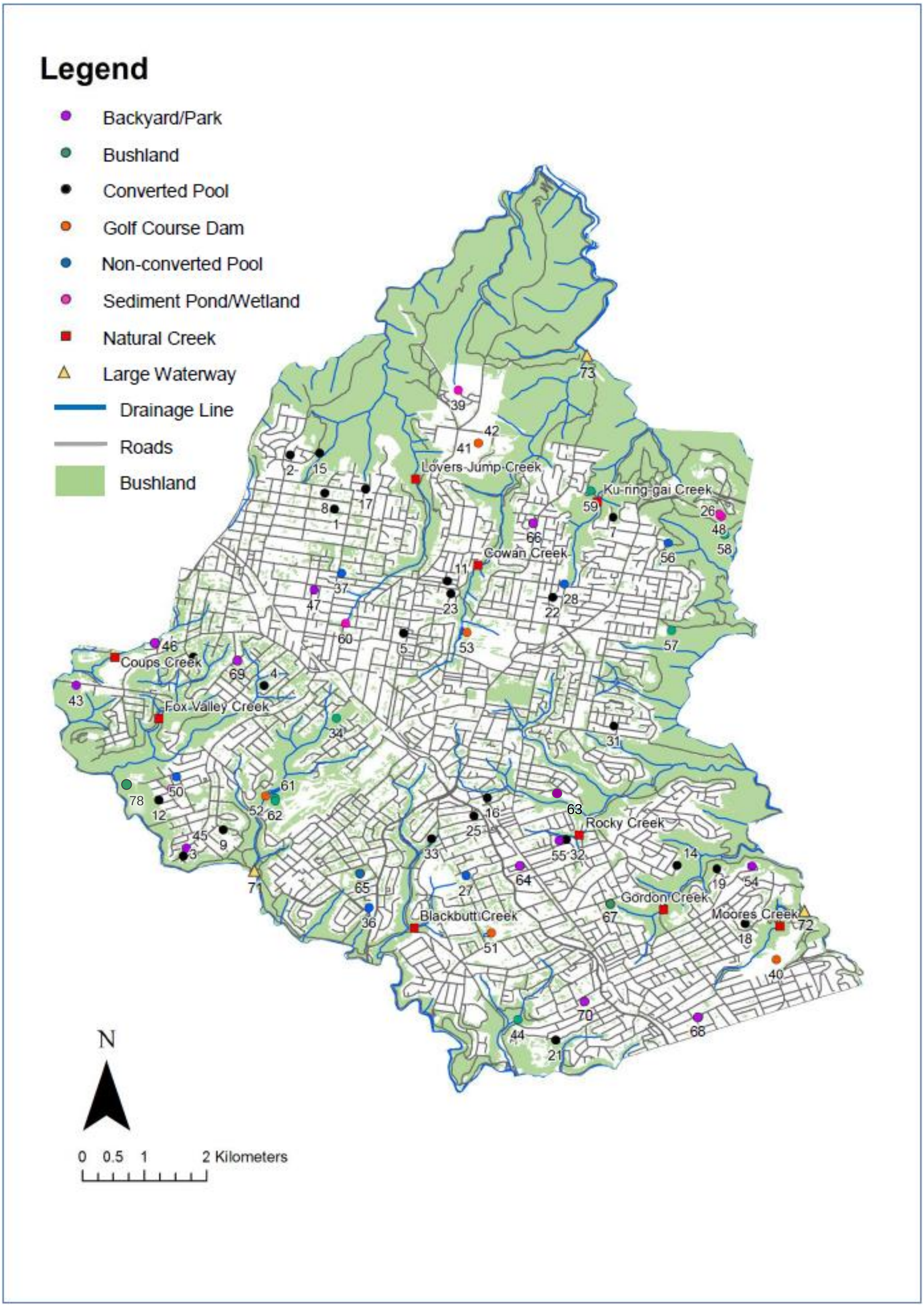


Fig. 1. Location of all (2017-2022) sampling sites in the Ku-ring-gai LGA.


a
b
c


d
e
f

Fig. 2. Habitat types surveyed for insectivorous bats in the Ku-ring-gai LGA: a) Creek (Ku-ring-gai Creek); b) Converted Pool (site 32); c) Non-converted Pool (site 37); d) Golf Course Dam (site 40); e) Sediment Pond/Wetland (site 26); f) Backyard (site 75); g) Bushland (site 34); and h) Large Waterway (site 72).

Bat surveys

Bat activity was surveyed for 2-5 nights in autumn in each year between 2018 and 2022. A single AnaBat Express detector (Titley Scientific, Brendale QLD) was deployed at all sites, except for creek sites that were also sampled with AnaBat II + Z-CAIM, AnaBat SD1 and AnaBat SD2 detectors in years 2-5. For sites on water, detectors were set with microphones facing the water surface from a height of <0.5 m as the Large-footed Myotis, *Myotis macropus*, usually flies 15-100 cm above the surface of water bodies (Churchill 2009). For sites without water, detectors were secured with microphones ~1 m from the ground. For bushland sites, detectors were located off tracks. Each detector recorded bat calls from dusk until dawn.

Since bat activity can be significantly reduced during heavy rain, sampling avoided these conditions where possible or was extended to sample nights without rain. All recorded bat calls were identified to species using automated call identification software, AnaScheme (Adams et al. 2010), in association with an identification key for bats of Sydney (unpublished data – B. Law). Bat calls with fewer than three valid pulses (i.e., minimum of six data points and model quality of ≥0.9) were not analysed by AnaScheme. Because multiple bat species may call simultaneously, calls were assigned to a species only if >50 % of pulses within the sequence were attributed to that species and only passes with a minimum of three pulses classified to the same species were identified. All bat calls that could not be assigned to a bat taxon were included in counts of total bat activity but were labelled as 'unidentified'. Since linear calls of *M. macropus* and Long-eared Bats, *Nyctophilus* spp., can be difficult to distinguish using automated software, all linear calls were assigned an identification of 'linear bat' and were subsequently manually checked to verify whether calls were produced by *Nyctophilus* spp. or *M. macropus*.

Data analyses

The number of bat calls for each species and all species combined (hereafter total bat activity) was tabulated for each site and night of sampling. Generalised Linear Mixed Models (GLMMs) were carried out in R (R Core Team 2014) using the lme4 package (Bates et al. 2015) to test for the main effects of habitat type, year and the interaction of these two factors (habitat type*year) on bat species richness, total bat activity and the activity of individual species for which sufficient data were available. For each model, site and night was used as a random factor and minimum daily temperature (BOM weather station: Terrey Hills AWS, 066059) and total daily rainfall (BOM weather station: Macquarie Park Villandra Village, 066156) was used as a covariate as bat activity can be positively correlated with nightly temperature (O'Donnell 2000; Erickson and West 2002) but negatively associated with rainfall among and within nights (Appel et al. 2019; Perks and Goodenough 2020). All bat data were $\log_{10}(x+1)$ transformed prior to analysis. All plots for GLMMs present back-transformed estimated marginal means. Distribution maps for nightly species richness, total bat activity and activity of individual species with sufficient data were generated for the Ku-ring-gai LGA using the Inverse Distance Weighting (IDW) tool (Spatial Analyst Tools) with a variable search radius and maximum power (3) in ArcGIS (ESRI). Maps were generated separately for each year between 2018 and 2022 to identify spatio-temporal trends.

Canonical Correspondence Analysis (CCA) was undertaken to identify environmental variables associated with activity of bat taxa. Separate analyses were carried out for each year of monitoring as some environmental variables changed between years. (i.e., pool size, water flow and emergent vegetation cover). Only species that were recorded at ~≥10 % of sites were included in analyses. Prior to running each CCA, all variables were $\log_{10}(x+1)$ transformed as suggested by Palmer (1993). A Pearson correlation analysis was then undertaken to identify those variables with significant collinearity. When two or more variables were correlated, only one was selected for inclusion in the CCA, except if the strength of the correlation was considered weak ($r<0.5$) or if the variables are known to significantly influence bat activity elsewhere. Environmental variables included in the analysis were: emergent vegetation cover at waterbodies, pool size, presence or absence of water flow (using active pumps as a proxy for pool and converted pool sites), surrounding bushland cover within 500 m, distance to nearest drainage line, and the % of sandstone and shale within 500 m. CCAs were run using PAST (PAleontological Statistics) version 3.0.

A canonical analysis of principal coordinates (CAP) was carried out in PERMANOVA+ for PRIMER (PRIMER-E, Plymouth, UK) to visualise differences between habitat types based on bat assemblages using data from years 2-6. This analysis plots sites on the basis of how similar bat assemblages are at each site. Sites that are plotted closer together have more similar bat assemblages than sites plotted further away. Prior to analysis, data were log-10-transformed and standardised, and a Bray-Curtis similarity matrix was constructed.

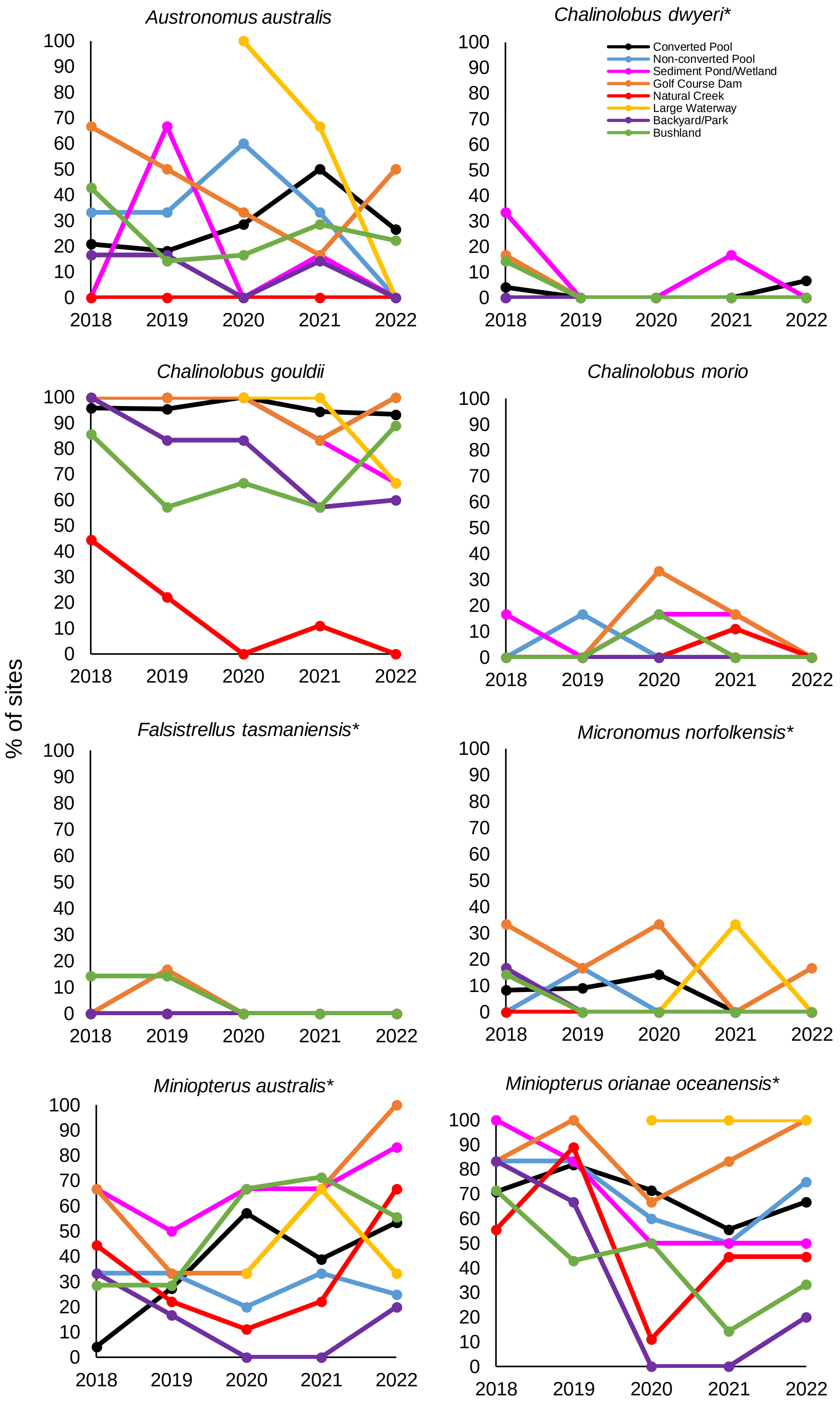
Results

In all, 46,663 bat calls were recorded in autumn between 2018 and 2022. Of these, 28,909 (62 %) were identified to one of 16 taxa (Table 1; Fig. 3). All other calls were usually poor quality and of short duration, and therefore could not be assigned a species-level identification. Across all years of monitoring, eight taxa were commonly recorded (Table 1). In 2022, Greater Broad-nosed Bat *Scoteanax rueppellii* was recorded for the first time since 2020, whereas Eastern False Pipistrelle *Falsistrellus tasmaniensis*, Chocolate Wattled Bat *Chalinolobus morio* and Yellow-bellied Sheathtail Bat *Saccolaimus flaviventris* were not recorded.

Table 1. Number of sites at which bat species were detected in the Ku-ring-gai LGA (2018-2022).

Species	**Common Name**	**Converted Pool (24,22,7,18,15)**	**Non-converted Pool (6,6,5,6,4)**	**Sediment Pond/ Wetland (6,6,6,6,6)**	**Golf Course Dam (6,6,3,6,6)**	**Natural Creek (9,9,9,9,9)**	**Large Waterway (0,0,3,3,3)**	**Backyard/Park (6,6,6,7,5)**	**Bushland (7,7,6,7,9)**
Austronomus australis	White-striped Freetail Bat	5,4,2,9,4	2,2,3,2,0	0,4,0,1,0	4,3,1,1,3	0,0,0,0,0	ns,ns,3,2,0	1,1,0,1,0	3,1,1,2,2
*Chalinolobus dwyeri**	Large-eared Pied Bat	1,0,0,0,1	0,0,0,0,0	2,0,0,1,0	1,0,0,0,0	0,0,0,0,0	ns,ns,0,0,0	0,0,0,0,0	1,0,0,0,0
Chalinolobus gouldii	Gould's Wattled Bat	23,21,7,14	6,6,5,5,4	6,6,6,5,4	6,6,3,5,6	4,2,0,1,0	ns,ns,3,3,2	6,5,5,4,3	6,4,4,4,8
Chalinolobus morio	Chocolate Wattled Bat	0,0,0,0,0	0,1,0,0,0	1,0,1,1,0	0,0,1,1,0	0,0,0,1,0	ns,ns,0,0,0	0,0,0,0,0	0,0,1,0,0
*Falsistrellus tasmaniensis**	Eastern False Pipistrelle	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0	0,1,0,0,0	0,0,0,0,0	ns,ns,0,0,0	0,0,0,0,0	1,1,0,0,0
*Micronomus norfolkensis**	East-coast Freetail Bat	2,2,1,0,0	0,1,0,0,0	1,0,0,0,0	2,1,1,0,1	0,0,0,0,0	ns,ns,0,1,0	1,0,0,0,0	1,0,0,0,0
*Miniopterus australis**	Little Bent-wing Bat	1,6,4,7,8	2,2,1,2,1	4,3,4,4,5	4,2,1,4,6	4,2,1,2,6	ns,ns,1,2,1	2,1,0,0,1	2,2,4,5,5
*Miniopterus orianae oceanensis**	Eastern Bent-wing Bat	17,18,5,10,10	5,5,3,3,3	6,5,3,3,3	5,6,2,5,6	5,8,1,4,3	ns,ns,3,3,3	5,4,0,0,1	5,3,3,1,3
*Myotis macropus**	Large-footed Myotis	0,0,0,0,0	0,0,0,0,0	2,1,1,1,0	3,4,2,5,4	1,1,1,1,1	ns,ns,3,3,3	0,0,0,0,0	0,0,0,0,0
Nyctophilus spp.	Long-eared Bats	4,5,1,0,0	0,0,0,0,0	2,3,2,2,0	2,2,0,0,0	3,4,2,1,1	ns,ns,0,0,0	0,0,1,0,0	0,1,1,0,0
Ozimops ridei	Eastern Freetail Bat	18,20,5,13,12	5,6,4,3,2	4,3,5,3,3	5,6,2,5,6	3,0,0,0,0	ns,ns,3,3,1	4,5,3,2,3	2,4,3,1,5
Rhinolophus megaphyllus	Eastern Horseshoe Bat	4,3,0,1,1	0,0,0,0,0	1,1,3,2,3	0,0,1,0,0	2,2,2,4,2	ns,ns,0,0,1	0,0,0,0,0	1,0,0,1,0
*Saccolaimus flaviventris**	Yellow-bellied Sheathtail Bat	0,0,0,0,0	0,0,0,1,0	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0
*Scoteanax rueppellii**	Greater Broad-nosed Bat	2,2,1,0,1	0,0,0,0,0	0,0,0,0,0	0,1,1,0,0	0,0,0,0,0	ns,ns,0,0,0	0,0,0,0,0	0,0,0,0,0
Scotorepens orion	Eastern Broad-nosed Bat	0,0,0,0,0	0,0,0,0,0	0,0,0,0,0	0,0,0,0,1	0,0,0,0,0	ns,ns,0,0,0	0,0,0,0,0	0,0,1,0,0
Vespadelus vulturnus	Little Forest Bat	3,1,0,3,2	0,3,3,1,0	4,3,2,3,2	0,2,2,3,2	1,0,0,0,1	ns,ns,1,0,2	1,0,1,1,0	0,2,2,0,2

* Indicates species listed on the threatened species schedules of the NSW *Biodiversity Conservation Act 2016*


% of sites
Austronomus australis
Chalinolobus dwyeri*
Converted Pool
Non-converted Pool
Sediment Pond/Wetland
Golf Course Dam
Natural Creek
Large Waterway
Backyard/Park
Bushland
Chalinolobus gouldii
Chalinolobus morio
Falsistrellus tasmaniensis*
Micronomus norfolkensis*
Miniopterus australis*
Miniopterus orianae oceanensis*
0
10
20
30
40
50
60
70
80
90
100
2018
2019
2020
2021
2022

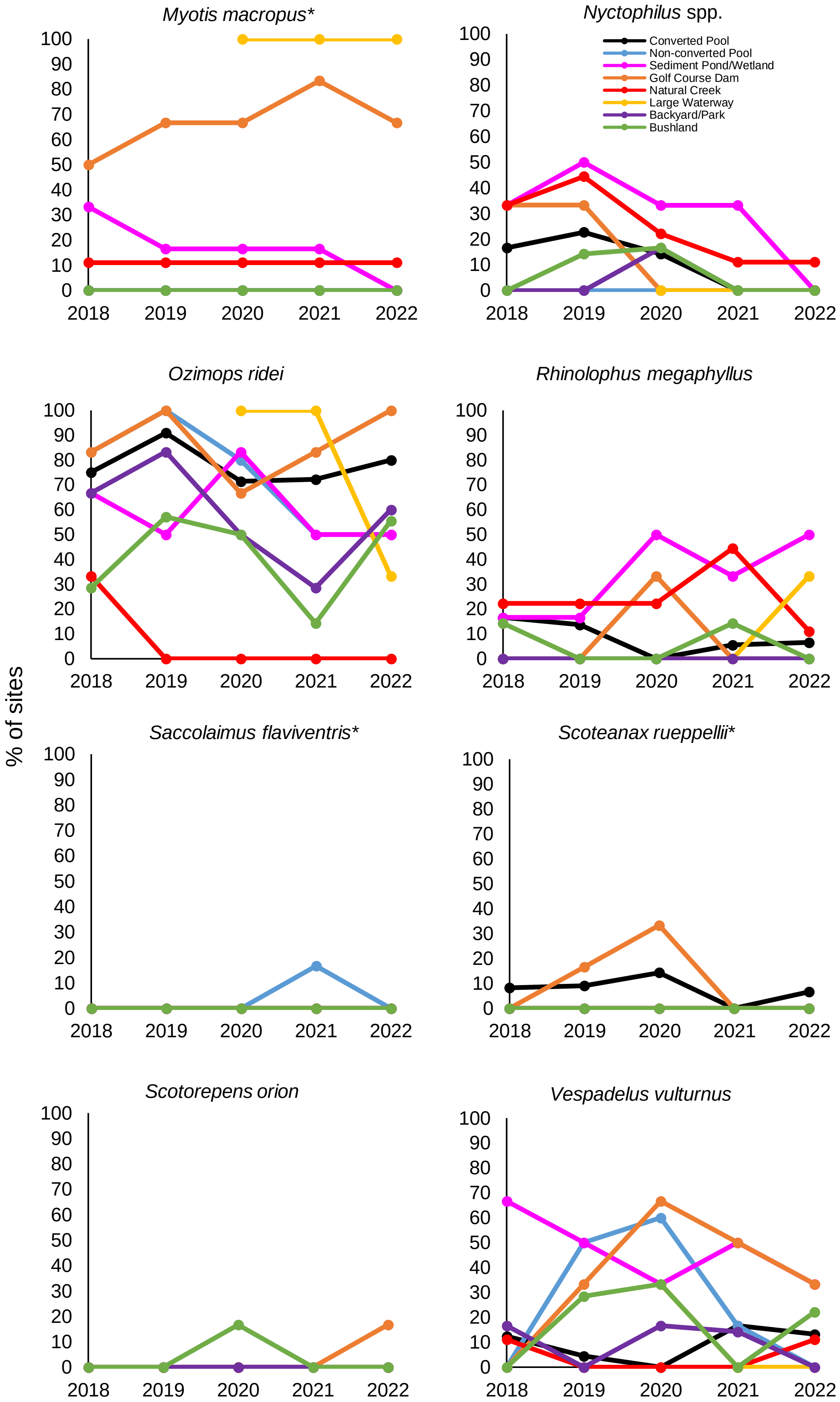


Fig. 3. Naïve occupancy of bat species in the Ku-ring-gai LGA between 2018 and 2022.

Comparison among habitat types

Nightly bat species richness in 2022 was highest at golf course dams and lowest at creeks. Bat species richness was significantly affected by the interaction of habitat type by year ($F_{26,593.14}$=2.6808, $P<0.001$; Fig. 4; Appendix 1a-1e). Species richness fluctuated among years at all treatments but these fluctuations were not significant, with the exception of backyards/parks which recorded a 60 % reduction in species richness in 2021 compared to 2018.

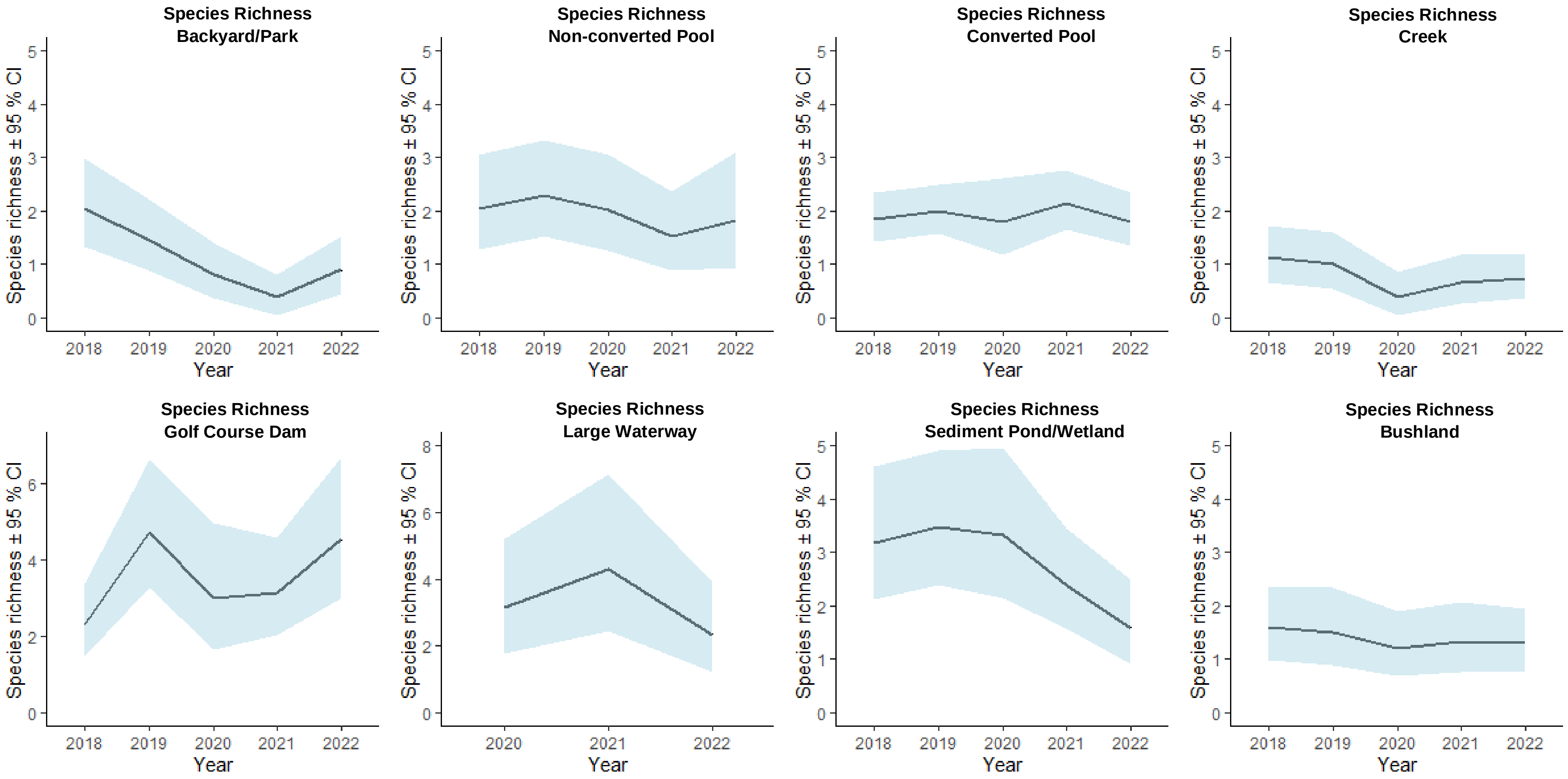


Fig. 4. Nightly bat species richness recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

Total bat activity (no. passes night^{-1}) ranged from 5.6-502.5 passes night^{-1} and was on average lowest at creeks and backyards/parks but greatest at golf course dams among all years. Total bat activity was significantly affected by the interaction of habitat type and year ($F_{26,641.56}$=2.2842, P=0.006; Fig. 5; Appendix 2a-2e). The trajectory of trends for total bat activity fluctuated among years at all treatments but these fluctuations were not significant, with the exception of golf course dams which had 67 % greater activity in 2019 compared to 2018 and sediment ponds/wetlands that had 69 % higher activity in 2019 compared to 2022.

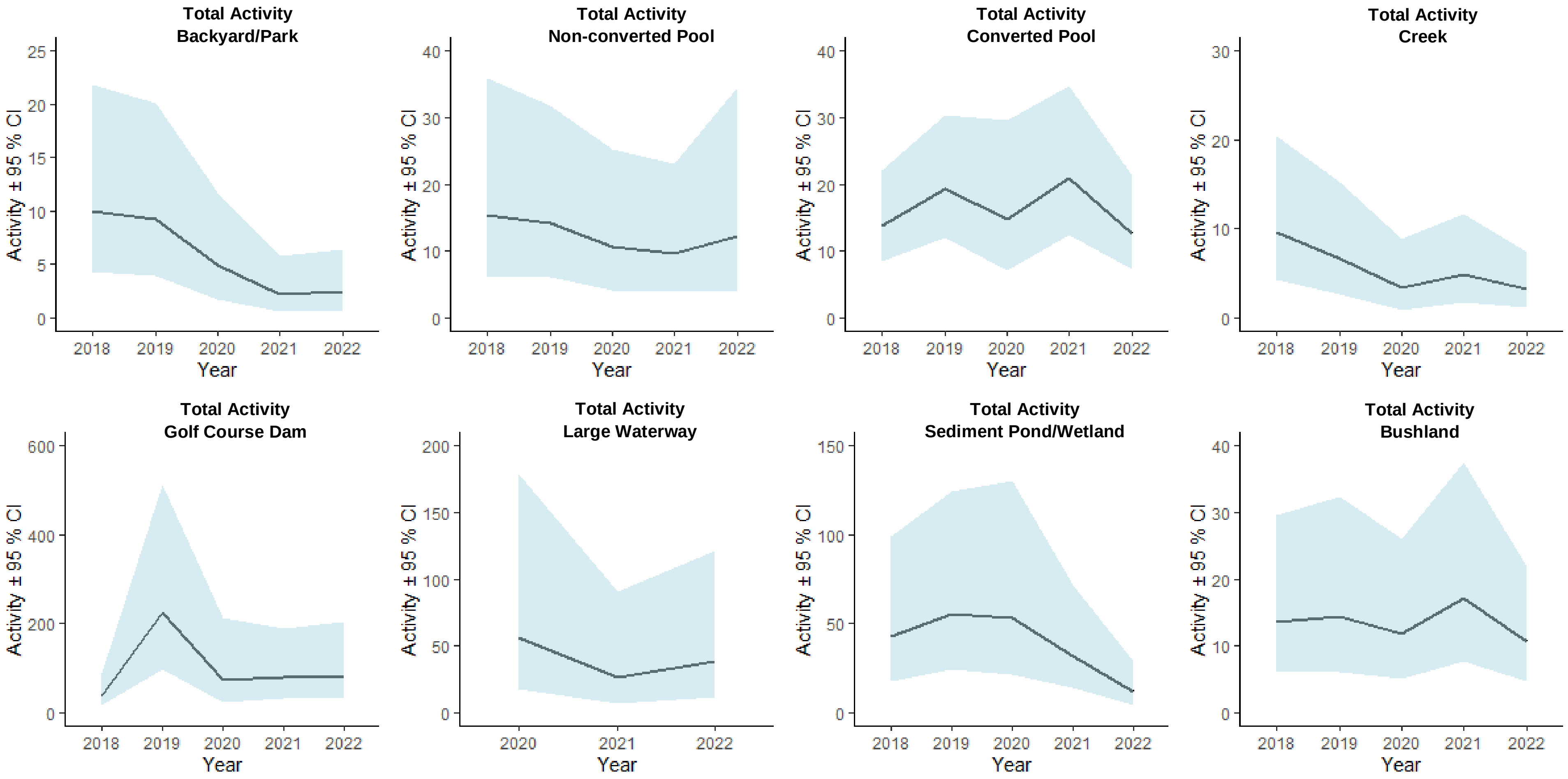


Fig. 5. Nightly bat activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

Activity of Individual Species

The White-striped Freetail Bat (*Austronomus australis*) was not recorded on creeks and was not commonly detected in the other habitat types during the study (Fig. 3). Nightly activity of *A. australis* was significantly affected by the interaction of habitat type by year ($F_{26,726.22}$= 1.7480, P=0.012; Fig. 6; Appendix 3a-3e). Activity was generally low in all treatments and remained stable despite minor fluctuations among years, with the exception of golf course dams that had a significant reduction (~90 %) in activity in 2021 compared to 2018.

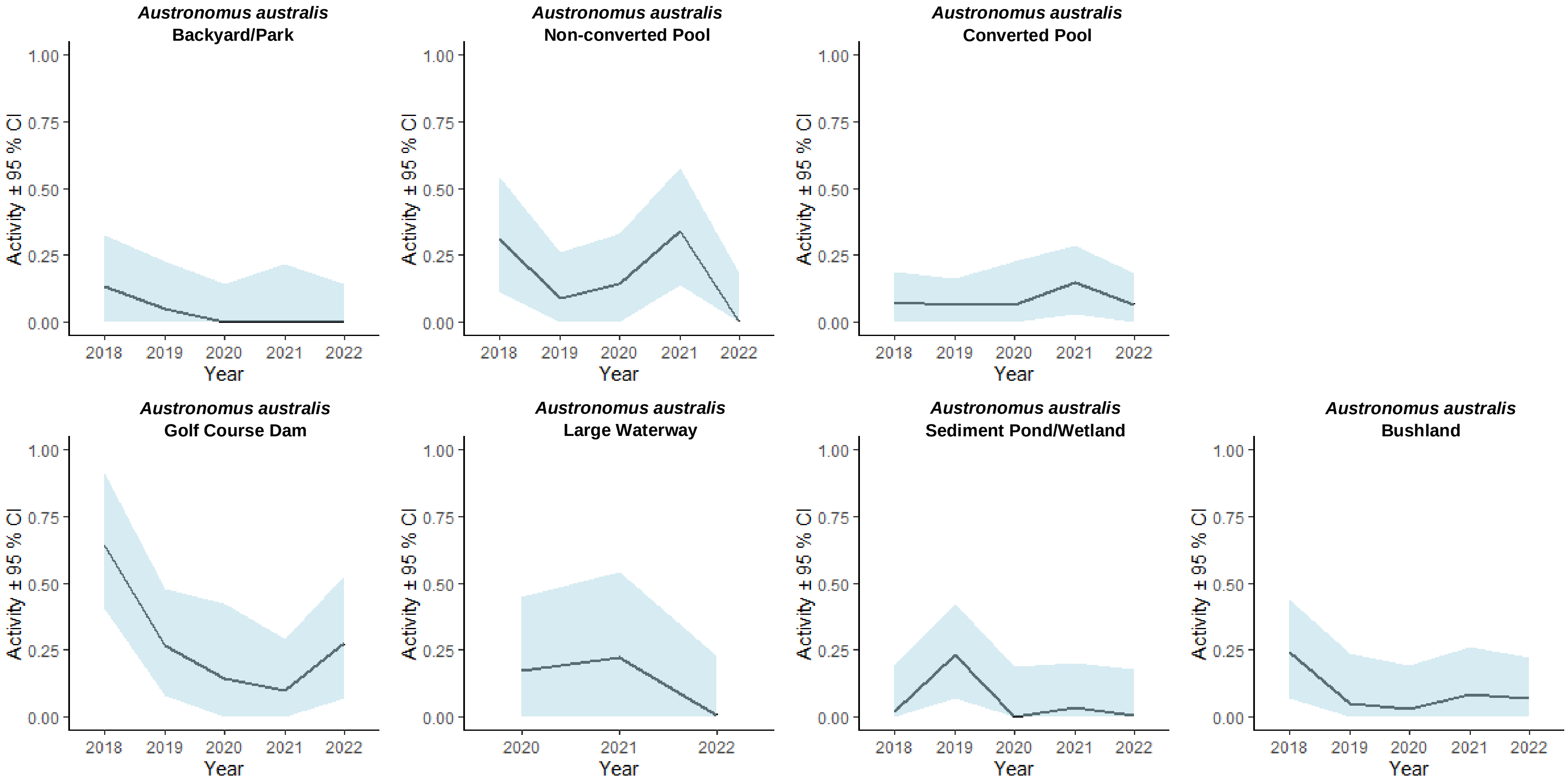


Fig. 6. Nightly *Austronomus australis* activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

Gould's Wattled Bat (*Chalinolobus gouldii*) was commonly detected in all habitat types during the study, except for creeks where the species was recorded at <50 % of sites or undetected (Fig. 3). *Chalinolobus gouldii* activity was significantly affected by the interaction of habitat type by year ($F_{26,662.04}=3.7721$, $P<0.001$; Fig. 7; Appendix 4a-4e). Activity at golf course dams was ~85 % lower in 2018 and 2021 compared to 2019, and activity at sediment ponds/wetlands was ~80 % lower in 2022 compared to 2020.

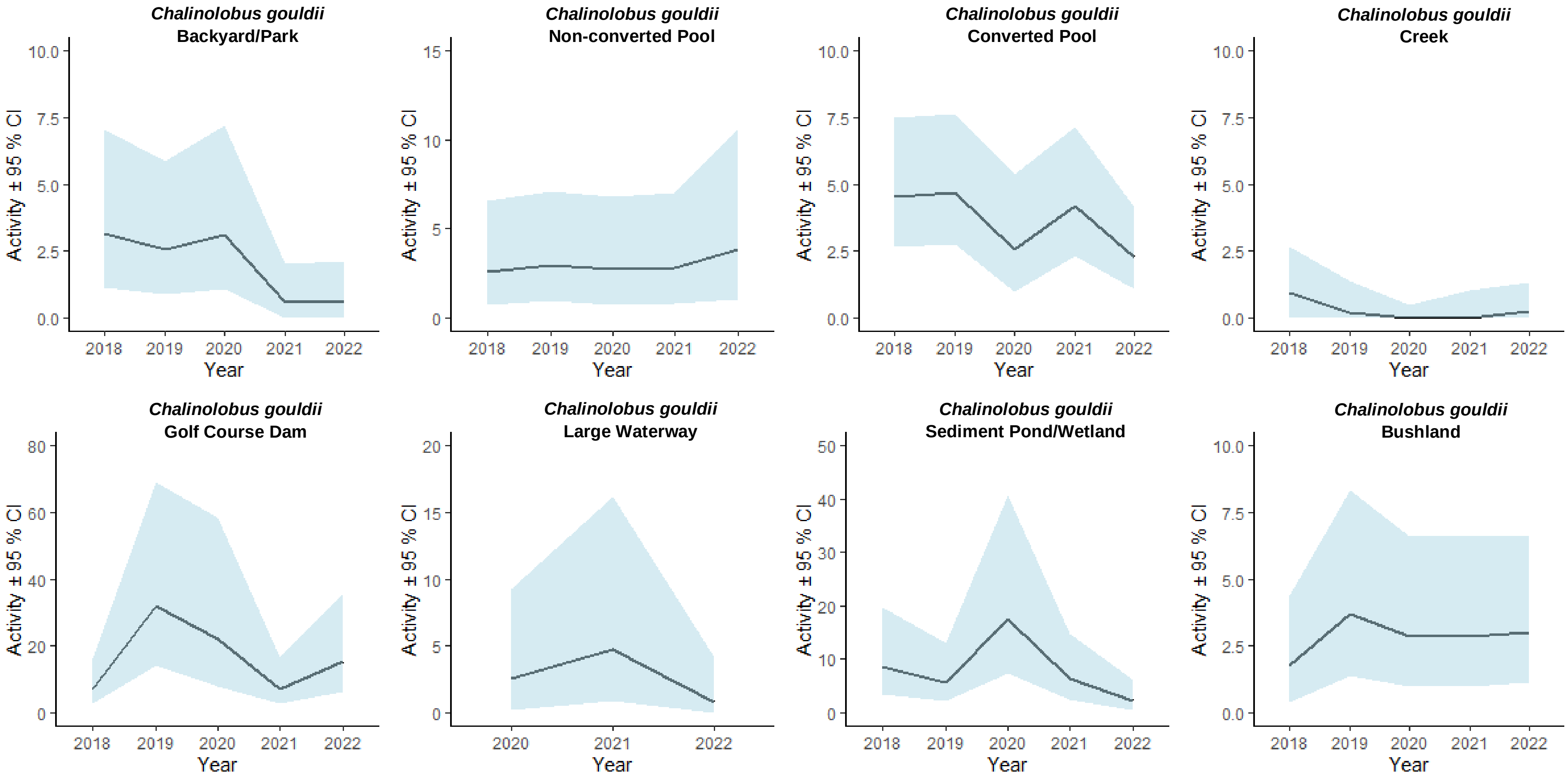


Fig. 7. Nightly *Chalinolobus gouldii* activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

The Eastern Bent-wing Bat (*Miniopterus orianae oceanensis*) was detected at ~75-80 % of sites sampled in 2018 and 2019 and was recorded in each habitat type (Fig. 3). The species was less common in 2020-2022, detected at ~44-58 % of sites and was not detected in backyards/parks in 2020 and 2021 (Fig. 3). Nightly activity was significantly affected by the interaction of habitat type by year ($F_{26,692.81}=3.8634$, $P<0.001$; Fig. 8; Appendix 5a-5e). Activity at bushland sites in 2018 was >90 % greater than 2019, 2020 and 2021. Activity at golf course dams in 2018 was >95 % higher than 2020 and 2021, whereas activity at golf course dams in 2019 activity was >72 % higher than all other years. Activity at sediment ponds/wetlands in 2018 was >90 % higher than 2021 whereas in 2019 activity was 56-93 % greater than 2021 and 2022. Activity at converted pools in 2018 and 2019 was >90 % greater than 2021. In 2022, there was an upward trend in activity across all treatments following a decline in 2020-2021.

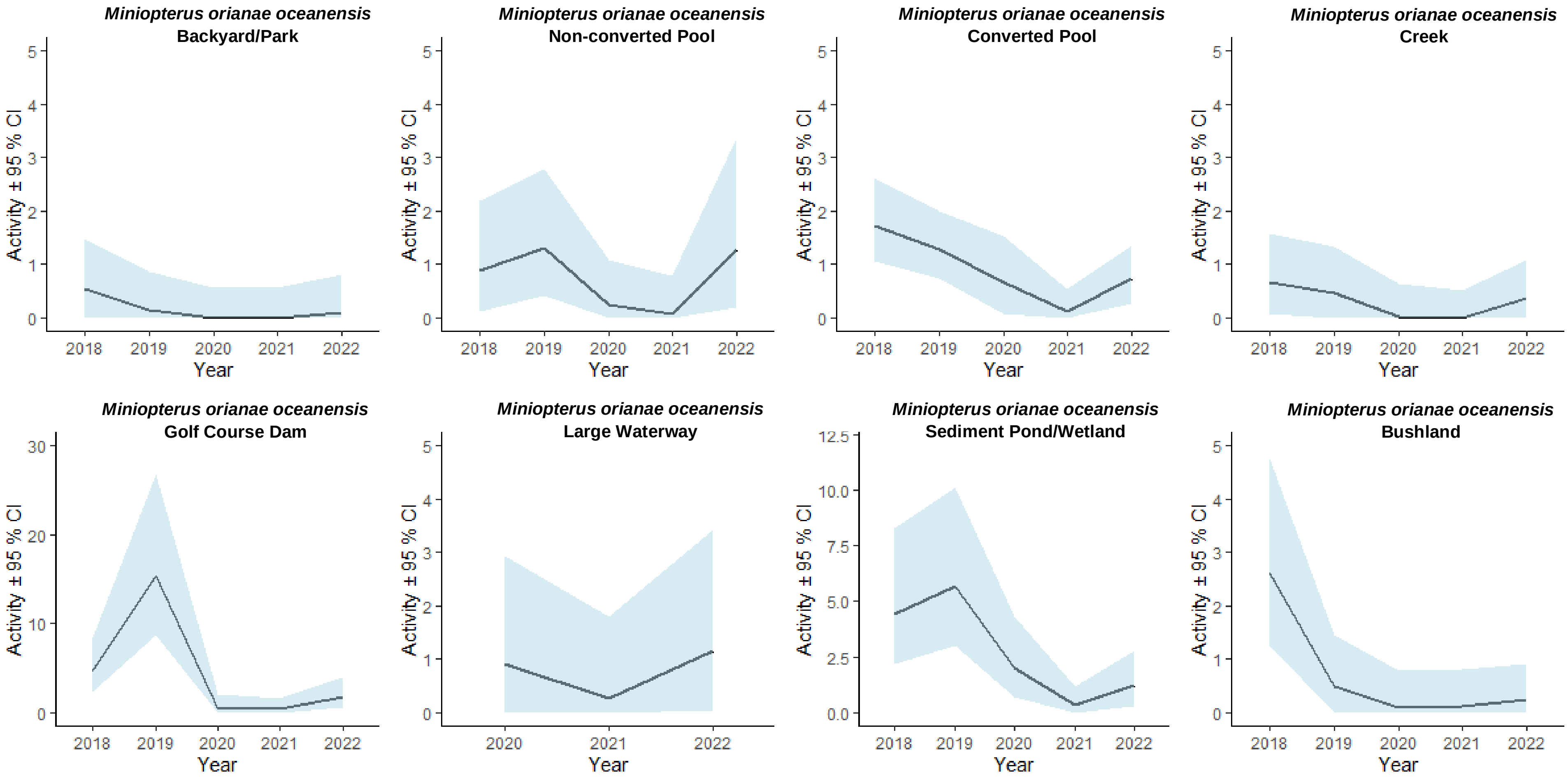


Fig. 8. Nightly *Miniopterus orianae oceanensis* activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

Large-footed Myotis (*Myotis macropus*) was only detected at golf course dams, sediment ponds/wetlands and a single creek site between 2018 and 2022 (Fig. 3). In all years, *M. macropus* activity was greatest at golf course dams and lowest at creeks. The interaction of habitat type by year significantly affected *M. macropus* activity ($F_{26,587.15}=2.2081$, $P<0.001$; Fig. 9; Appendix 6a-6e). Activity at golf course dams in 2019 and 2021 was 61-70 % greater than 2018, whereas activity in 2020 and 2022 was intermediate. Activity of *M. macropus* in sediment ponds/wetlands and creeks was low in all years of monitoring whereas activity at large waterways was comparable to golf course dams.

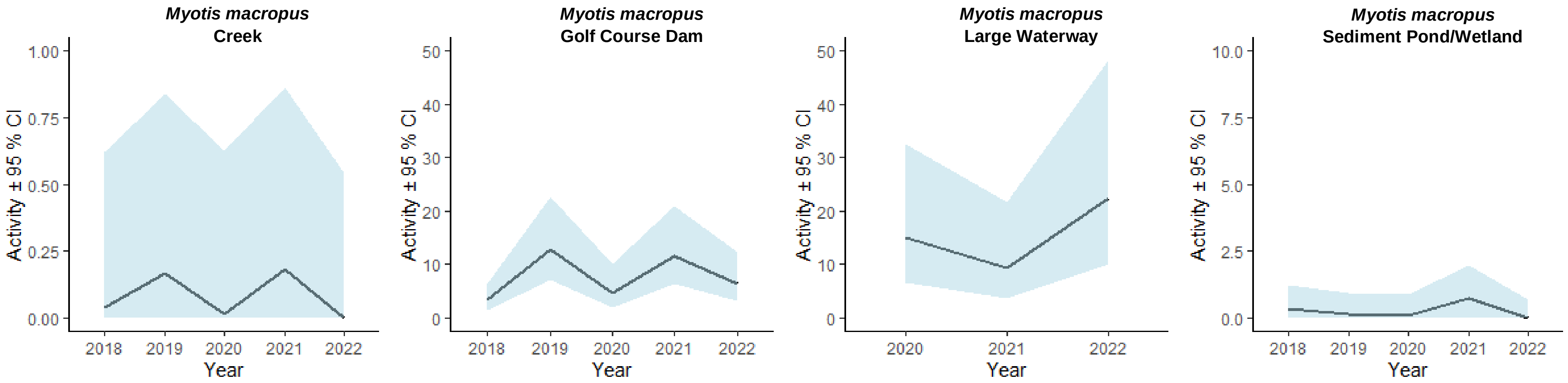


Fig. 9. Nightly *Myotis macropus* activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

The Eastern Freetail Bat (*Ozimops ridei*) was detected at 48-69 % of sites across all years, but was not recorded on creeks since 2019 (Fig. 3). *Ozimops ridei* activity was significantly affected by the interaction of habitat type by year ($F_{26,668.83}$=1.6898, P=0.018; Fig. 10; Appendix 7a-7e). Activity at golf course dams in 2021 was 70 % lower than 2019. Activity at all other habitat types was low and similar among years.

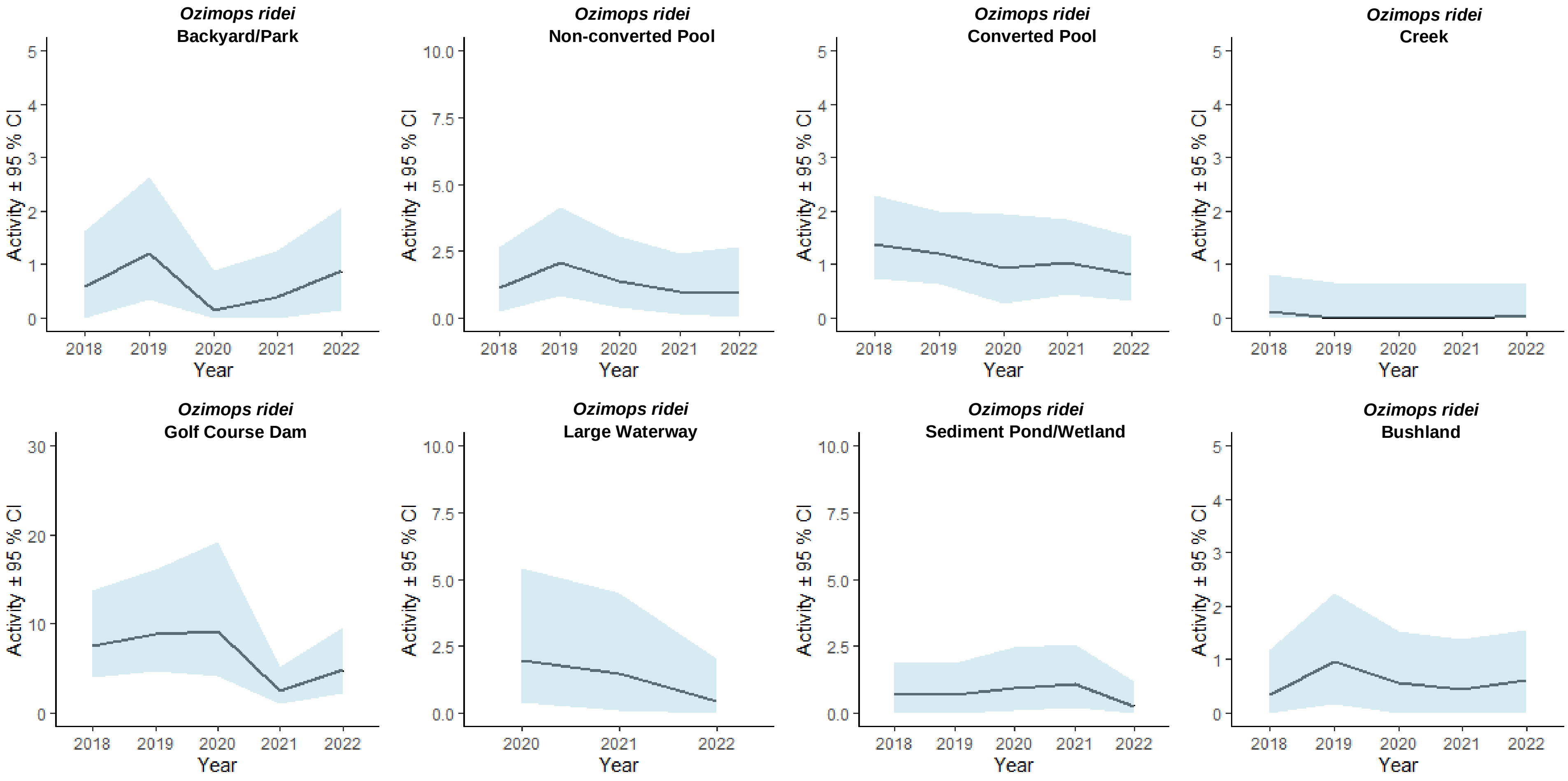


Fig. 10. Nightly *Ozimops ridei* activity recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2022. Back-transformed estimated marginal means are displayed ± 95 % confidence intervals.

Large-eared Pied Bat *Chalinolobus dwyeri*, *C. morio*, *F. tasmaniensis*, Little Bent-wing Bat *M. australis*, East-coast Freetail Bat *Micronomus norfolkensis*, *Nyctophilus* spp., Eastern Horseshoe Bat *R. megaphyllus*, *S. flaviventris*, *S. rueppellii*, Eastern Broad-nosed Bat *S. orion* and Little Forest Bat *Vespadelus vulturnus* were too infrequently recorded to allow for statistical comparison among habitat types and among years.

A Canonical Analysis of Principal Coordinates (CAP) using data from 2018-2022 revealed that bat assemblages differed among habitat types (Fig. 11). The bat assemblage of creek sites was generally distinct from all other habitat types and was characterised by low levels of *C. gouldii* and *M. ridei* activity. There was considerable overlap in bat assemblages for non-converted pool, converted pool and backyard/park sites, whereas most golf course dams and large waterways had different bat assemblages that were characterised by greater activity of *C. gouldii*, *M. australis*, *M. orianae oceanensis*, *O. ridei* and *M. macropus*. Bushland sites supported bat assemblages that were not distinct from all other habitat types except creeks, golf course dams and large waterways.

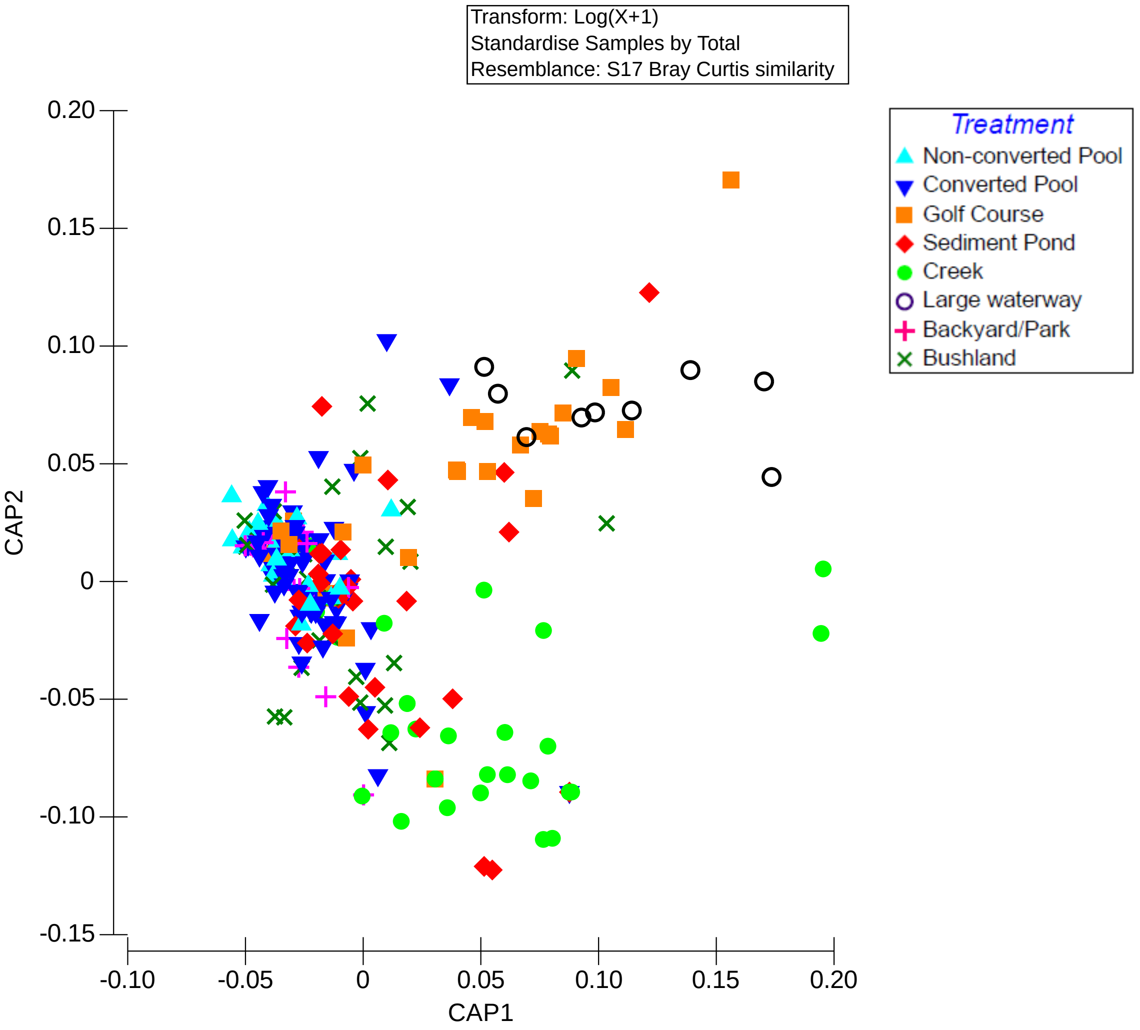


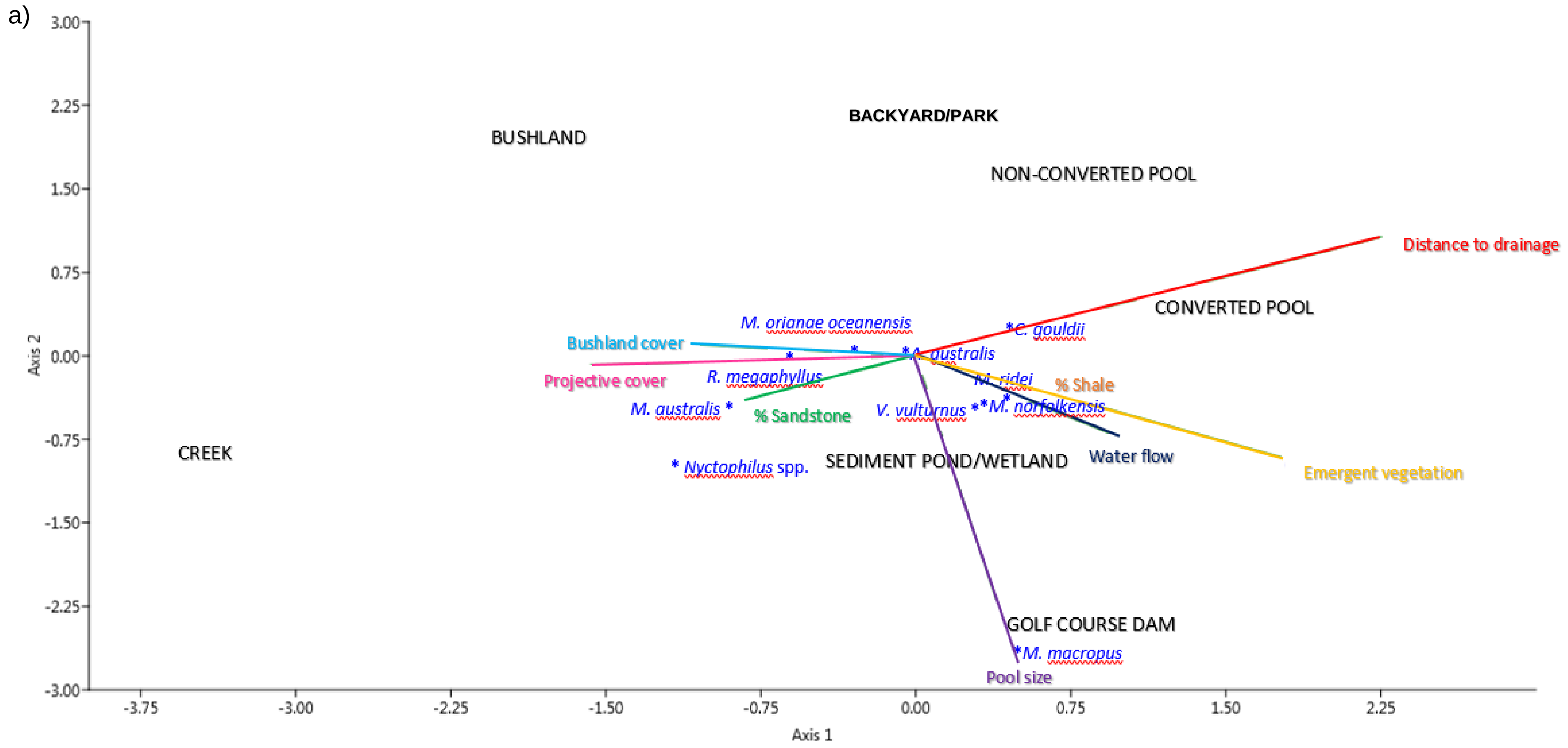
Fig. 11. Canonical analysis of principal coordinates (CAP) plot illustrating differences between habitat types on the basis of bat species assemblages.

Relationship between bat activity and environmental variables

Bushland cover was closely associated with sandstone geology (r=0.622, P<0.001) as most bushland was co-located in areas with sandstone geology. However, given bat activity is known to be strongly influenced by geology (Basham et al. 2010; Threlfall et al. 2012), both variables were retained in CCAs.

CCA biplots for 2018-2022 data revealed that creek and bushland sites were associated with sandstone geology, greater bushland, projective foliage cover, and were near drainage lines (Fig. 12a-e). Backyards/parks, non-converted pools and converted pools varied greatly in distance from drainage lines (range= 40–555 m^2) and the latter was characterised by greater emergent vegetation cover. Sediment ponds/wetlands and golf course dams were associated with moderate-large pool sizes (mean; 1354 m^2), though pool sizes varied considerably in 2020-2022 (range; 70-5500 m^2) (Fig. 12a-e). Large waterways were near bushland with underlying sandstone geology and had lower levels of emergent vegetation than golf course dams and sediment ponds/wetlands (Fig. 12c-d).

Associations between bat species and environmental variables were generally consistent among years. *Myotis macropus* was strongly associated with large pools provided by some golf course dams, sediment ponds/wetlands and large waterways (from 2020 onwards) (Fig. 12a-e). *Nyctophilus* spp., *R. megaphyllus* and *M. australis* were positively associated with projective foliage cover, bushland cover, sandstone geology, and were negatively associated with distance from drainage lines. *Miniopterus orianae oceanensis* was generally associated with low projective foliage cover and low bushland cover (Fig. 12a-e). *Chalinolobus gouldii*, *M. norfolkensis and O. ridei* were associated with shale geology and low-moderate distances from drainage lines, whereas *V. vulturnus* associations varied considerably among years but tended to be weakly associated with shale geology (Fig. 12a-e). Although in 2018 *A. australis* was not associated with any environmental variable quantified in this study, the species was associated with areas of low bushland cover in 2019-2022 and pools of a moderate size in 2019-2021 (Fig. 12a-e).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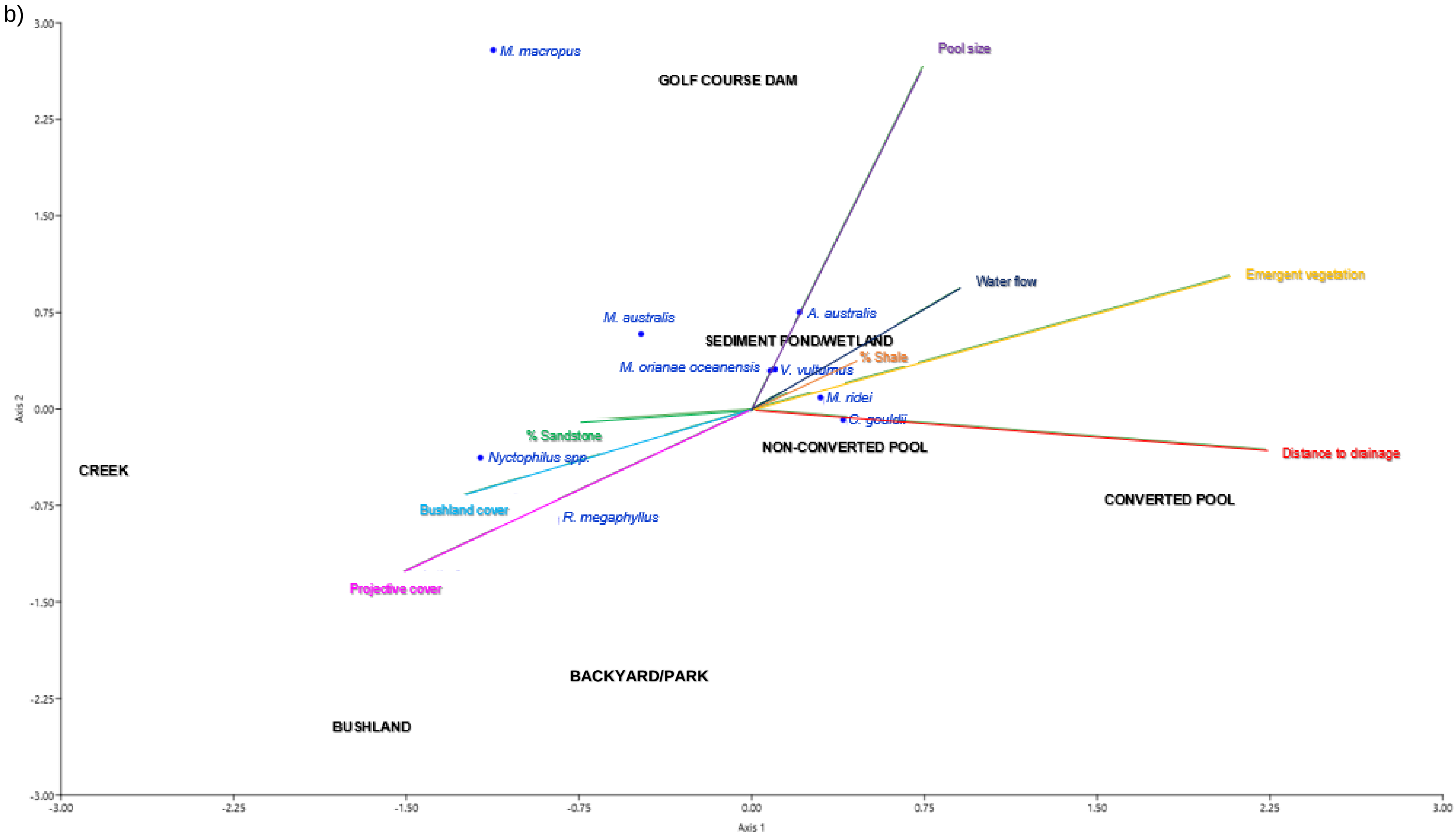
BUSHLAND
BACKYARD/PARK
NON-CONVERTED POOL
Distance to drainage
CONVERTED POOL
M. orianae oceanensis
*C. gouldii
Bushland cover
*A. australis
Projective cover
R. megaphyllus
M. ridei
% Shale
M. australis *
% Sandstone
V. vulturnus
**M. norfolkensis
CREEK
Water flow
Emergent vegetation
* Nyctophilus spp.
SEDIMENT POND/WETLAND
GOLF COURSE DAM
*M. macropus
Pool size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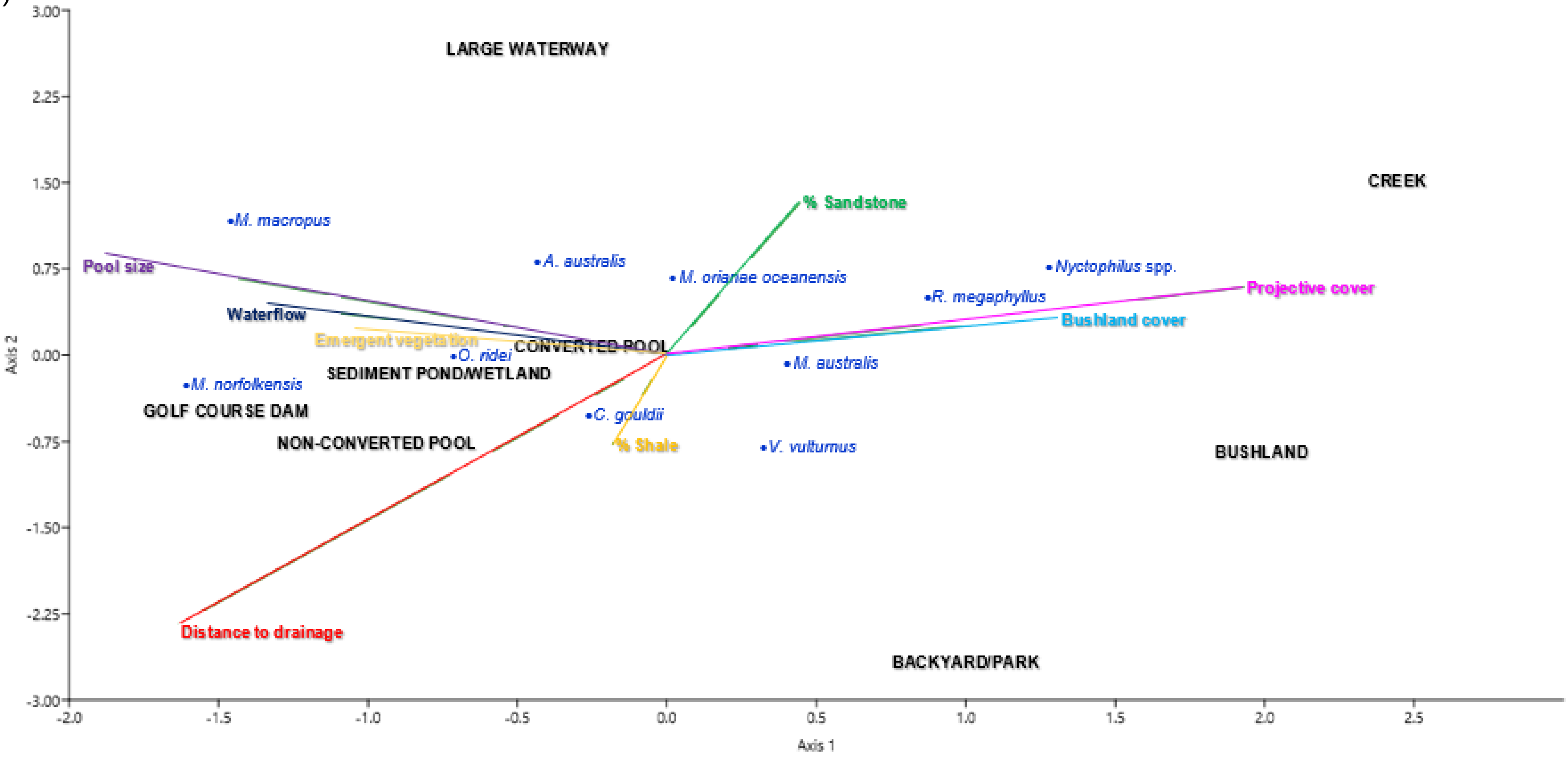
2.25
3.00
Axis 1
M. macropus
GOLF COURSE DAM
Pool size
Water flow
Emergent vegetation
M. australis
A. australis
SEDIMENT POND/WETLAND
% Shale
M. orianae oceanensis
V. vulturnus
M. ridei
C. gouldii
% Sandstone
Nyctophilus spp.
NON-CONVERTED POOL
Distance to drainage
CREEK
CONVERTED POOL
Bushland cover
R. megaphyllus
Projective cover
BACKYARD/PARK
BUSHLAND

c)


LARGE WATERWAY
CREEK
% Sandstone
M. macropus
A. australis
Pool size
M. orianae oceanensis
Nyctophilus spp.
R. megaphyllus
Projective cover
Waterflow
Bushland cover
Emergent vegetation
CONVERTED POOL
O. ridei
M. australis
SEDIMENT POND/WETLAND
M. norfolkensis
GOLF COURSE DAM
C. gouldii
NON-CONVERTED POOL
% Shale
V. vulturnus
BUSHLAND
Distance to drainage
BACKYARD/PARK
Axis 2
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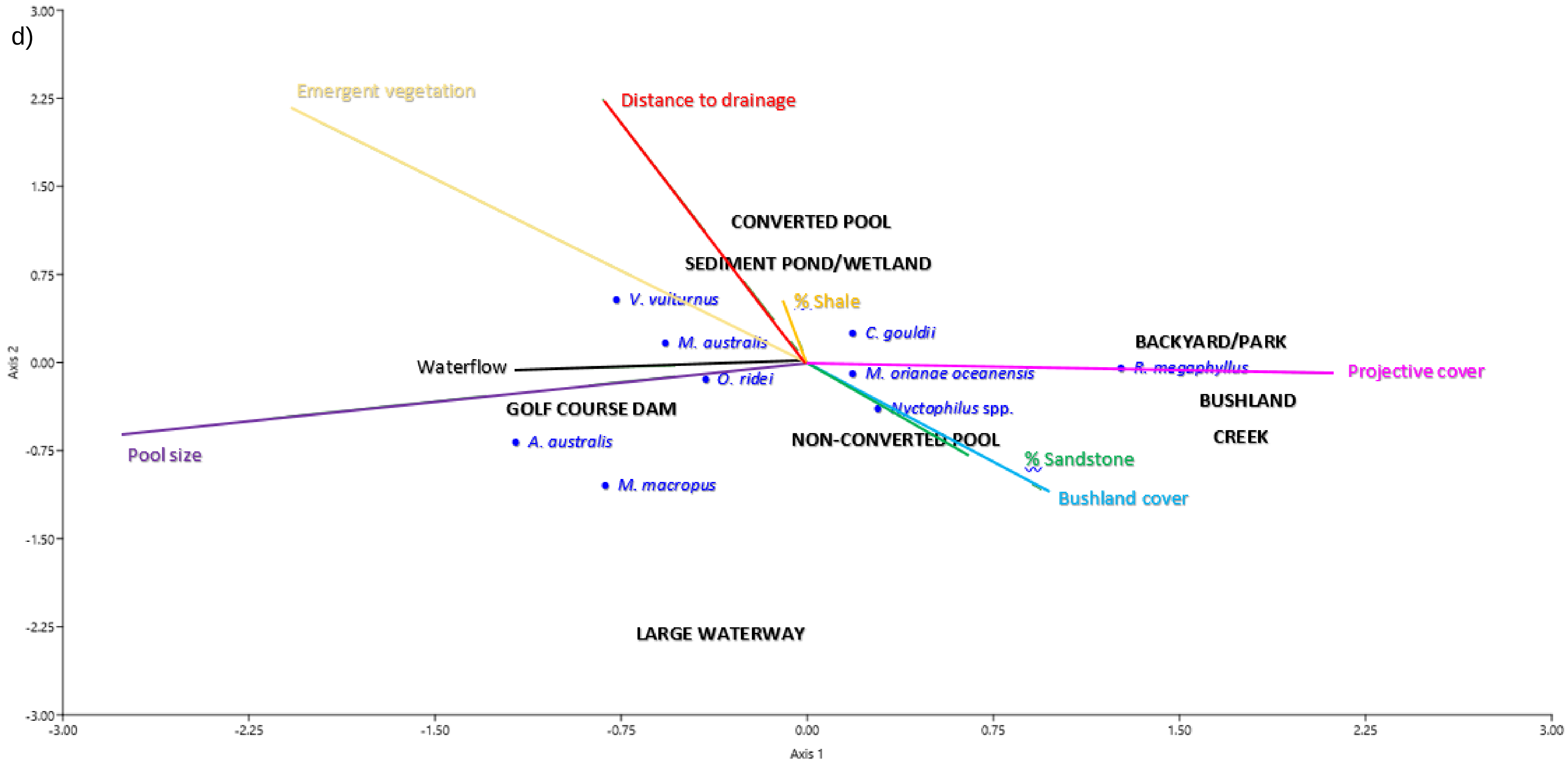
d)
Emergent vegetation
Distance to drainage
CONVERTED POOL
SEDIMENT POND/WETLAND
V. vulturnus
% Shale
C. gouldii
M. australis
BACKYARD/PARK
Waterflow
O. ridei
M. orianae oceanensis
R. megaphyllus
Projective cover
GOLF COURSE DAM
Nyctophilus spp.
BUSHLAND
A. australis
NON-CONVERTED POOL
CREEK
Pool size
% Sandstone
M. macropus
Bushland cover
LARGE WATERWAY
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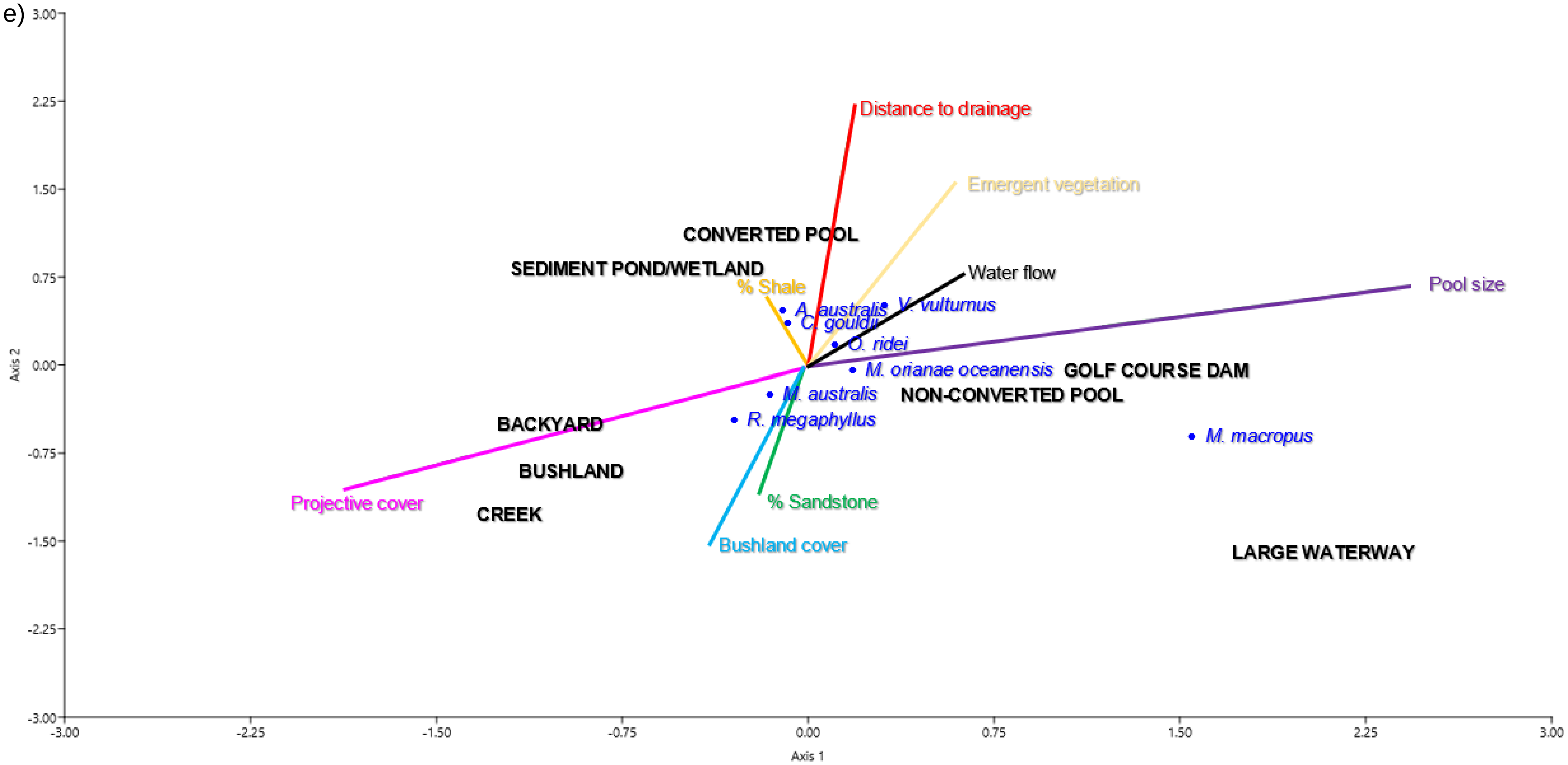

Fig. 12. Canonical correspondence analysis bi-plot illustrating associations between activity of bat taxa and environmental variables in a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022. Centroids for the different habitat types are also plotted.

Discussion

This is the first study to annually monitor insectivorous bats across broad habitat types in an urban, but leafy local government area in Sydney. Six years of monitoring have documented some clear and consistent results about the relative value of wildlife ponds for bats. Furthermore, this study reports on the first trends for urban bat species in Sydney. Nightly bat species richness and activity was greatest at golf course dams and lowest at creeks, with the former variable remaining similar among years for all habitat types except backyards/parks which had 60 % lower nightly species richness in 2021 compared to 2018. Activity at golf course dams and sediment ponds/wetlands was 67-69 % lower in 2018 and 2022 compared with 2019, respectively. Greater relative activity at golf course dams and sediment ponds/wetlands was primarily driven by three species - *M. macropus*, *O. ridei* (formerly *M. ridei*) and *C. gouldii*. Species richness and activity at converted pools was generally greater than creeks, but lower than golf course dams and sediment ponds/wetlands, and comparable to non-converted pools, reference backyards/parks and bushland. Furthermore, up to three taxa (*C. dwyeri*; *R. megaphyllus* and *S. rueppellii*) recorded at converted pools were not detected at non-converted pools or backyards/parks. The specialist trawling bat, *M. macropus* was most active on larger pools with low levels of emergent aquatic vegetation cover present in golf course dams and large waterways, whereas activity on sediment ponds/wetlands was low-moderate, but highly variable. Small pools on creeks were infrequently used by *M. macropus*. Activity of most species fluctuated among years, though differences were habitat-specific. *Miniopterus orianae oceanensis* activity declined or remained low across all habitat types between 2018 and 2021 but showed an upward trend at all habitat types in 2022. Ongoing monitoring will provide data required to track trends for insectivorous bat populations in the LGA as converted pool habitats continue to develop.

The value of waterbodies for insectivorous bats

Nightly bat species richness across six years of monitoring was greatest at golf course dams, sediment ponds/wetlands and large waterways, with 1-2 more species per night relative to other habitat types. Species richness at converted pools (~2.4 species per night), non-converted pools (~2.3 species per night), creeks (~1.0 species per night), backyards/parks (~1.3 species per night) and bushland sites (~1.8 species per night) was mostly stable among years, though there was a 60 % reduction between 2018 and 2021 at backyards/parks. The level of species richness recorded during the study was comparable to the levels recorded in bushland (Gonsalves et al. 2018) and backyards in the leafier parts of Sydney (Basham et al. 2011) and other vegetated landscapes within the Sydney Metropolitan region (Threlfall et al. 2011; Gonsalves and Law 2017). Three species (*C. gouldii*, ~96 % of sites; *O. ridei*, ~77 % of sites; *M. orianae oceanensis*, ~68 % of sites) were commonly recorded at converted pool sites between 2018 and 2022, though the latter was recorded at ~20 % fewer sites in 2021 compared to the previous three years with some recovery (~15 % fewer sites relative to 2018-2020) in 2022. This contrasts with a considerable increase (~23 %) in the occurrence of *M. orianae oceanensis* from 2017 to 2018 (Gonsalves et al. 2017), when the timing of surveys was changed from summer to autumn when *M. orianae oceanensis* is more abundant in Sydney as the species prepares to overwinter (Gonsalves and Law 2018). It is unclear why there was a substantial reduction in the proportion of sites with *M. orianae oceanensis* in 2021, but the species also exhibited declines in activity across the LGA in 2020. This occurred 1-2 years after the black summer fires of 2019-2020 and the decline may reflect broader landscape effects on the species. While there was no recovery in activity of the species in 2021, an upward trend in 2022 was associated with some recovery in the occurrence of the species in the LGA.

All three commonly recorded species at converted pools were also frequently detected in backyards/parks with or without pools in our study and in other leafier suburbs of Sydney (Basham et al. 2011), except in 2020 and 2021 when *M. orianae oceanensis* was not detected in backyards/parks without pools and was detected at 23-33 % fewer sites at backyards/parks with swimming pools. In 2022, the species was detected at 20 % of backyards/parks and all swimming pool sites. *Chalinolobus gouldii* and *O. ridei* were also less commonly recorded at backyards/parks without pools in 2021, with some recovery in 2022. It is unclear whether these differences among years are due to site-level differences, with most backyards sampled in 2018 and 2019 unable to be sampled in 2020 due to COVID-19 restrictions. Urban parks were sampled as an alternative in 2020 to continue to provide context for converted and non-converted pools within the urban matrix, and in 2021 and 2022 there was a return to sampling backyards, though a small number of these (n=2) were sampled for the first time in 2021 or 2022.

Between 2018 and 2022, on average activity was greatest at golf course dams (295 passes per night) and sediment ponds/wetlands (58 passes per night), with activity considerably lower in all other habitat types (13-37 passes per night). Activity at large waterways that were sampled since 2020 was intermediate (43 passes per night). Total activity at golf course dams increased by 67 % between 2018 and 2019, but was intermediate in subsequent years. Activity of three commonly recorded species (*C. gouldii*, *M. orianae oceanensis* and *M. macropus*) increased in 2019 but remained stable in 2020, except *M. orianae oceanensis* which declined or remained low at golf course dams and all other habitat types with no recovery in 2021. No changes were made to the management of golf courses (e.g., increases in use of fertilisers) between 2018 and 2019, but the latter year was during drought. Elevated activity at golf course dams in this year may reflect a contraction by bats to an area of reliable moisture. It is also possible that there was a mass emergence of aquatic insects from golf course dams. Elsewhere, bat activity has been shown to be positively associated with the abundance of aquatic insects in riparian forests (Fukui et al. 2006).

Activity at backyards/parks without or with converted pools or non-converted pools was comparable to activity recorded for backyards in vegetated parts of metropolitan Sydney (Threlfall et al. 2011), but not in shale-dominated areas in northern Sydney where bat activity was approximately four times greater (Basham et al. 2011). Nightly activity at creeks in our study was lower than backyard/park sites (without or with converted or non-converted pools), except in 2020 and 2021 when the difference was reduced or reversed for backyards/parks, perhaps due to site-level differences for this habitat type or reductions in activity of particular species (i.e., *M. orianae oceanensis*) that were infrequently detected at creeks. Creeks were located deep within sandstone gullies that are associated with low activity on adjacent small flyways (Basham et al. 2011). Activity at gully sites in the study of Basham et al. (2011) was considerably lower than backyards that were often on shale geology in this area, highlighting the importance of geology and productivity to bats in the Sydney area (Threlfall et al. 2012). In our study, backyards/parks were not associated with any one geology and comprised a mix of sandstone- and shale-dominated geologies.

Patterns of bat activity among habitat types were species-specific. *Chalinolobus gouldii* was most active at golf course dams and sediment ponds/wetlands, with activity ~2-31 times greater than all other habitat types. This species is an edge-space bat (Adams et al. 2009) that can forage in open spaces and along edges that are prominent in the urban matrix. Furthermore, the species is known to tolerate lights and exploit insect concentrations at lights (Kirsten and Klomp 1998; Adams et al. 2005; Haddock et al. 2019). Golf course dams and sediment ponds can often contain high-nutrient runoff that supports high insect abundance, particularly emerging aquatic insects (Fukui et al. 2006). Similarly, wetlands can also support high insect abundances and are productive foraging areas for bats (Gonsalves et al. 2013a, 2013b).

Miniopterus orianae oceanensis is one of the most commonly recorded bat species in northern Sydney during autumn (Gonsalves and Law 2018). In the Ku-ring-gai LGA, the species was detected at 75 % (n=64) and 80 % (n=61) of sites in 2018 and 2019, respectively. However, in 2020 and 2021 *M. orianae oceanensis* was recorded at just 44 % (n=45) and 47 % (n=62) of sites, respectively. In 2022, there was a recovery with the species detected at 84 % of sites (n=57). In 2018 and 2019, there was significantly greater activity for the species at golf course dams and sediment ponds/wetlands relative to most other habitat types sampled, whereas in 2020 and 2021 there was a significant reduction in activity observed at all habitat types except creeks, backyards/parks and bushland where activity was typically low. It is unclear why there was a decline in activity and proportion of sites where the species was detected in 2020/2021. Given that declines occurred at multiple habitat types within the LGA, it is unlikely that they are associated with changes to management at a more localised level. This trend could reflect fluctuations in climate (e.g., winter rainfall) as has been found for bats in the semi-arid Pilliga Forests (Law et al. 2020). The decline in occurrence and activity of the species was evident immediately after the 2019 drought and black summer fires in 2019-2020 and given the species is known to migrate (Dwyer 1963), it is likely to be more affected by events that occur at a large landscape scale than other species. Indeed, declines of this nature were not evident for other commonly recorded bat taxa in 2020. It is also possible that a disturbance event at a roost site may have influenced the decline seen for *M. orianae oceanensis* in 2020. Adult females of this species migrate to maternity caves outside of Sydney just before summer, where they give birth and care for young (Dwyer 1963). One of these caves north of Sydney was affected by the black summer bushfires of 2019-2020, with a significant area of forest surrounding the cave being burnt. It is possible that this fire may have impacted the population size of the species.

Annual census of the population at two other maternity sites, including one that was also impacted by fire, revealed a 20 % reduction in the numbers of flying adults pre-fire from 2018 to 2019, with a further 3 % decline in numbers of flying adults post-fire in 2020, though this smaller change was within the bounds of sampling error (Pers. Comm. D. Mills). Furthermore, winter surveys at a local roost site within the Ku-ring-gai LGA revealed stable activity between 2019 and 2020 (unpublished data – C. Costello), though winter surveys may not be a reliable indicator of population size given bats will enter torpor and not all individuals may leave the roost on a given night. It was hypothesised in 2020 that rather than impacting population size, the spring/summer bushfires in 2019 may have delayed the return of *M. orianae oceanensis* to Ku-ring-gai and other parts of Sydney (Gonsalves et al. 2020), resulting in LGA-wide reductions in autumn activity but stable trends at an overwintering roost site in winter. However, a lack of recovery in the activity of the species in 2021 did not support this hypothesis. A more widespread distribution of the species within the LGA as well as an upward trajectory for activity in all habitat types in 2022 may be early signs of recovery for this species. The population structure of *M. orianae oceanensis* can be quite complex at overwintering roosts at an LGA-level, with the population made up of individuals from broad geographic areas (i.e., multiple maternity sites) as revealed by bat banding (Hoye 2000). Furthermore, regular shifting of individuals between several local roosts is known to occur in accordance with prevailing weather conditions (White 2011). As such, it is difficult to assess how changes to the population size at maternity sites corresponds to more localised patterns in areas used for overwintering. Systematic monitoring of *M. orianae oceanensis* roost sites within the LGA is needed to provide a clearer picture of the status of the species in the LGA.

Despite having a close association with waterways, *M. macropus* was rarely recorded on creeks between 2018 and 2022 and was not detected at converted or non-converted pools. This is consistent with monitoring in 2017, albeit undertaken in a different season (Gonsalves et al. 2017). The species was recorded at Moores Creek in each year (2017-2022) of monitoring, with this site located <600 m from a golf course dam (Roseville Golf Course) that recorded the highest level of *M. macropus* activity in all years of monitoring. Furthermore, the species was recorded nearby (<500 m) at a large waterway on Middle Harbour where moderate levels of activity have previously been recorded (Gonsalves and Law 2017). Converted pools generally had moderate amounts of vegetation cover over pools as well as emergent vegetation which represents physical and acoustic clutter that can affect the ability of trawling bats to locate prey (e.g., Frenckell and Barclay 1987; Boonman et al. 1998), whereas non-converted pools were chlorinated and likely to support few aquatic invertebrate prey. In a study across metropolitan Sydney, the species was negatively associated with backyard elements within the urban matrix (Threlfall et al. 2012). In our study, *M. macropus* was recorded at >50 % of all golf course dams (n=6, 2018, 2019, 2021 & 2022; n=3, 2020) and was less commonly detected at sediment ponds/wetlands, with activity ~28-times greater at golf course dams. Golf course dams were generally larger than sediment ponds/wetlands and the species is known to be most associated with larger, more permanent waterways (Anderson et al. 2006). The apparent rarity of *M. macropus* in the Ku-ring-gai LGA was paralleled in freshwater environments on underlying sandstone, where the species was recorded at ~38 % of sites (n=24) (Asplet 2016). These results for northern Sydney contrast with autumn surveys of waterways in western Sydney, where *M. macropus* was recorded at ~77 % of sampled sites (n=26), which included natural and artificial creeks and wetlands situated on shale soils of the Cumberland Plain (Gonsalves and Law 2016). Elsewhere in the Port Jackson estuary, the species was also widespread, though with hotspots of activity identified (Gonsalves and Law 2017). Since 2020, large waterways in the upper reaches of the Port Jackson estuary were also sampled as part of this study and moderate levels (22 calls per night) of *M. macropus* activity were recorded. The trend for *M. macropus* activity was generally quite stable at sediment ponds/wetlands and creeks, with low activity recorded at these habitat types, whereas at golf course dams, activity of the species fluctuated among years.

Relationships between bat activity and environmental variables

At the species level, associations with environmental variables were broadly consistent among years and were generally explained by the ecomorphology of bats, but also geology-associated productivity (Basham et al. 2011; Threlfall et al. 2012). *Chalinolobus gouldii* uses constant frequency-frequency modulated (CF-FM) echolocation that is suited to flying along edge habitats but also in open spaces (Adams et al. 2009). Activity of *C. gouldii* was moderately, positively associated with distance from drainage lines and negatively associated with projective foliage cover, indicating the species was most active away from drainage lines and in areas with lower levels of vegetative clutter, suited to the species ecomorphology.

Miniopterus orianae oceanensis is suited to flying along vegetation interfaces and open areas that are bordered by an edge (Adams et al. 2009; Gonsalves and Law in 2018). This species was associated with drainage lines, but in areas with lower bushland and foliage projective cover. *Ozimops ridei* is a fast-flying species suited to open spaces (Adams et al. 2009) and activity of the species was greatest at golf course dams and in areas with low bushland cover and foliage projective cover.

For other less commonly recorded species, CCA bi-plots highlighted associations for *Nyctophilus* spp., *R. megaphyllus* and *M. australis* with bushland cover on sandstone geology, often near drainage lines, though the latter was more associated with sites on shale in 2021. *Nyctophilus* spp. was recorded at <10 % of sites in 2022 and was not included in the corresponding CCA. *Rhinolophus megaphyllus* and *M. australis* use subterranean roosts (e.g., caves, deeper overhangs, culverts) and these may be more available in the deep gullies of drainage lines. Furthermore, *R. megaphyllus* and *Nyctophilus* spp. are able to fly and forage in high-clutter using high-constant frequency or broadband frequency modulated echolocation calls and manoeuvrable flight. *Myotis macropus* was strongly associated with larger pool sizes and these were generally present as golf course dams with low levels of emergent aquatic vegetation. This pattern was consistent among years. *Micronomus norfolkensis* was associated with sediment ponds/wetlands and golf course dams situated away from drainage lines and bushland on sandstone geology, though the species was rarely recorded in 2021 and 2022. An association with wetlands has previously been found for this species on shale geology on the Cumberland Plain (Gonsalves and Law 2016).

Management recommendations

Waterbodies surveyed across the Ku-ring-gai LGA comprised a suite of bat taxa that included eight threatened species, though bat activity varied among waterbodies and nearby sites. Nevertheless, these waterbodies represent important habitat and foraging elements for insectivorous bats in an urbanised landscape (Blakey et al. 2018). Consequently, appropriate management is needed to continue to provide and enhance existing habitat at waterbodies in the Ku-ring-gai LGA. Below we provide recommendations to enhance habitats for bats.

Waterbodies with large pools and low projective foliage cover provide bats with open areas and with low levels of clutter that are more efficient for foraging by edge- and open-adapted bats (Fenton 1990; Gonsalves et al. 2013a). In our study, golf course dams and sediment ponds/wetlands typically provided these features for bats and these waterbodies had the highest levels of bat activity in the LGA. Four species (*C. gouldii*, *M. orianae oceanensis*, *O. ridei*, and *M. macropus*) contributed to most of the activity recorded at these two habitat types over all years of monitoring, though note the LGA-wide reduction in activity of *M. orianae oceanensis* in 2020 and 2021. Both habitat types retain water via runoff and are likely to capture pollutants (e.g., nutrients, heavy metals, insecticides and herbicides). It is known that heavy metals in sediments of ponds and other waterbodies can be substantial (Karlsson et al. 2010) and a pathway for uptake by bats has been established (Clarke-Wood et al. 2016; Asplet 2016). Given this, periodic monitoring of these waterbodies should be undertaken to assess pollutant levels and establish a threshold at which management is triggered. Management could take the form of dredging contaminated sediments, which can benefit some bats species (Flache et al. 2016).

We also suggest that land managers are made aware (e.g., via workshops) of the relative importance of golf course dams and sediment ponds/wetlands for bats and they should be provided with specific guidance/advice on how to sensitively manage these habitat elements for bats. For example, the need to limit the amount of emergent vegetation at dams/ponds used by *M. macropus* as well as limiting the use of water features that generate surface acoustic clutter for the species. This may be possible by turning off water features at night. Consideration should also be given to lighting around dams, as *M. macropus* is considered sensitive to white lights (Straka et al. 2016; Unpublished data – J. Haddock). Land managers may also be more encouraged to conserve high-value bat habitat if they are able to be part of the monitoring process (e.g., by deploying detectors) and/or have the opportunity to observe bats using their land (e.g., via a bat talk/walk). There is also potential for engagement with members of the public that use the golf courses in the LGA. Interpretive/educational signs may be installed near dams or at the tee off area for selected holes to highlight the presence of threatened species at golf course dams (and the courses more generally).

Other than large waterways, all other habitat types had lower bat species richness and activity than golf course dams and sediment ponds/wetlands, though sampling on wide tracks in bushland sites is likely to increase species richness and bat activity at this habitat type as activity is known to be focused along such linear features. Nevertheless, these habitats are still used by a suite of bats, including some that were not recorded at golf course dams or sediment ponds/wetlands, and require suitable management. For example, the threatened species *F. tasmaniensis* was detected at a single bushland site (ID44 – Sir Phillip Game Reserve) in both, 2018 and 2019 but was not recorded at sediment ponds/wetlands. Furthermore, these habitat types are used for foraging by bats, albeit at low levels (Gonsalves et al. 2020). Productive bushland on shale soils is known to have high conservation value for bats (Basham et al. 2011) and it is important that these patches are protected and managed.

Given its close association with waterways, *M. macropus* is a species that was thought to potentially respond to the provision of converted pools. However, the species is yet to be recorded at converted pools after six years of monitoring (Gonsalves et al. 2017, 2018, 2019, 2020, 2021) and was rarely recorded at creeks. Activity of *M. macropus* was greatest over waterbodies with large pools, mainly golf course dams. The failure to record *M. macropus* at converted pool sites suggests that habitat provided by these sites is not currently suitable for the species. Converted pools can have relatively high cover of emergent aquatic vegetation and this can negatively affect the ability of trawling bats to detect prey (Boonman et al. 1998). Conversely, golf course dams generally had <10 % emergent vegetation cover and lacked algal cover, though leaf litter was sometimes present. Increasing the size and area of open water (i.e., reducing emergent aquatic vegetation and algal mats) will make converted pools more suitable for use by *M. macropus*, however this is unlikely to be feasible. *Myotis macropus* also appears to be sensitive to artificial light (unpublished data – J. Haddock) and most converted pools had lighting within 10 m, though the impact of lights on pools was considered to be minimal (Pers. Comm. C. Foulis/H. Henning). Golf courses tended to have no light or very low levels of light around dams. Red lights should be used when artificial lighting is needed for safety/security purposes, as bats, including *M. macropus*, are known to be less sensitive to this wavelength of light (Spoelstra et al. 2017; unpublished data – J. Haddock). Future investigation of the availability of roosting habitat for *M. macropus* (e.g., via radio-tracking or acoustic surveys of subterranean structures) in the urban matrix and near green spaces may provide more context for trends revealed during acoustic monitoring of waterbodies in the LGA.

Myotis macropus was rarely recorded at pools on creeks, with the species only detected at a single creek (Moores Creek) in each year. Pools on creeks were relatively small compared to golf course dams and sediment ponds/wetlands and were located deep in drainage gullies that had a high level of projective foliage cover. This cover likely contributes to acoustic and physical clutter around pools on creeks, making them less suitable for foraging by *M. macropus*. Reducing cover of riparian weeds may serve to improve access to creek pools for *M. macropus*.

Future research

Six years of monitoring in the Ku-ring-gai LGA have identified future areas of research to assist with the management of bats, including threatened species. In particular, further research is required to identify potential causes for the fluctuations in activity recorded at golf course dams among years. Similarly, research is required to identify potential drivers of decline for the activity of *M. orianae oceanensis*. This may first require an assessment of fine-scale habitat use by the species (e.g., via GPS tracking – see Gonsalves et al. 2021) to identify areas of high importance that may be subject to disturbance (e.g., fuel reduction fire, changes to lighting, etc.).

Investigation of the availability of roosting habitat for *M. macropus* in the urban matrix and near green spaces is needed to provide more context for trends revealed during acoustic monitoring of waterbodies in the LGA. Ongoing annual monitoring is also required to continue to map trends in bat activity that may fluctuate in response to significant disturbance events (e.g., high severity, broad-scale bushfires or localised disturbances at roost sites). We suggest that the current monitoring program should be expanded to include other LGAs within Sydney and supplemented with systematic annual monitoring of roost sites. This may be done relatively cost-effectively via acoustic surveys to reveal emergence activity patterns (i.e., activity in the first hour after dusk) or counts using thermal cameras and automated software for counting bats. If knowledge of roost sites within the LGA is limited, a broad acoustic survey of potential roost sites could be undertaken to identify new roosts.

Converted pools provided value for bats in the LGA. Three species (*C. dwyeri*; *R. megaphyllus* and *S. rueppellii*) were recorded at these sites but not at non-converted pools or backyards/parks, though detections were uncommon at converted pool sites. Nevertheless, bats have also been recorded feeding at these pools (Gonsalves et al. 2020), albeit at low levels. Future research should seek to examine whether converted pools provide additional prey resources than non-converted pool and backyard/park sites. It is also unclear whether bats use converted pools for drinking, particularly during drought. Future research may seek to specifically examine the extent to which bats use converted pools and other waterways for drinking. This will require more detailed analysis of ultrasonic data collected at monitoring sites with water (i.e., converted pools, swimming pools, golf course dams, wetlands and creeks). A trial analysis should be undertaken at a subset of sites where activity is high and therefore drinking activity may be more likely to be detected.

Conclusion

Bats occupy high trophic levels and are considered to be good indicators of environmental change (Jones et al. 2009). Furthermore, they can be cost effectively monitored using acoustic sensors (Hourigan et al. 2008) with 90 % power to detect upward or downward trends of up to 30 % within 10 years (Law et al. 2015). In six years of monitoring converted pools alongside other waterbodies and habitat elements (backyards/parks without pools and bushland) in the urban matrix, we found patterns of total activity were fairly consistent, though with some fluctuations among years and habitat types. We also identified species-specific fluctuations in annual trends for activity. We suggest that the monitoring sites used in the current study should be resurveyed annually to continue to track trends in bat populations especially through periods of extreme weather events that are predicted to increase with climate change. This will facilitate a long-term assessment of the value of converted pools and other habitat types to bats in the Ku-ring-gai LGA.

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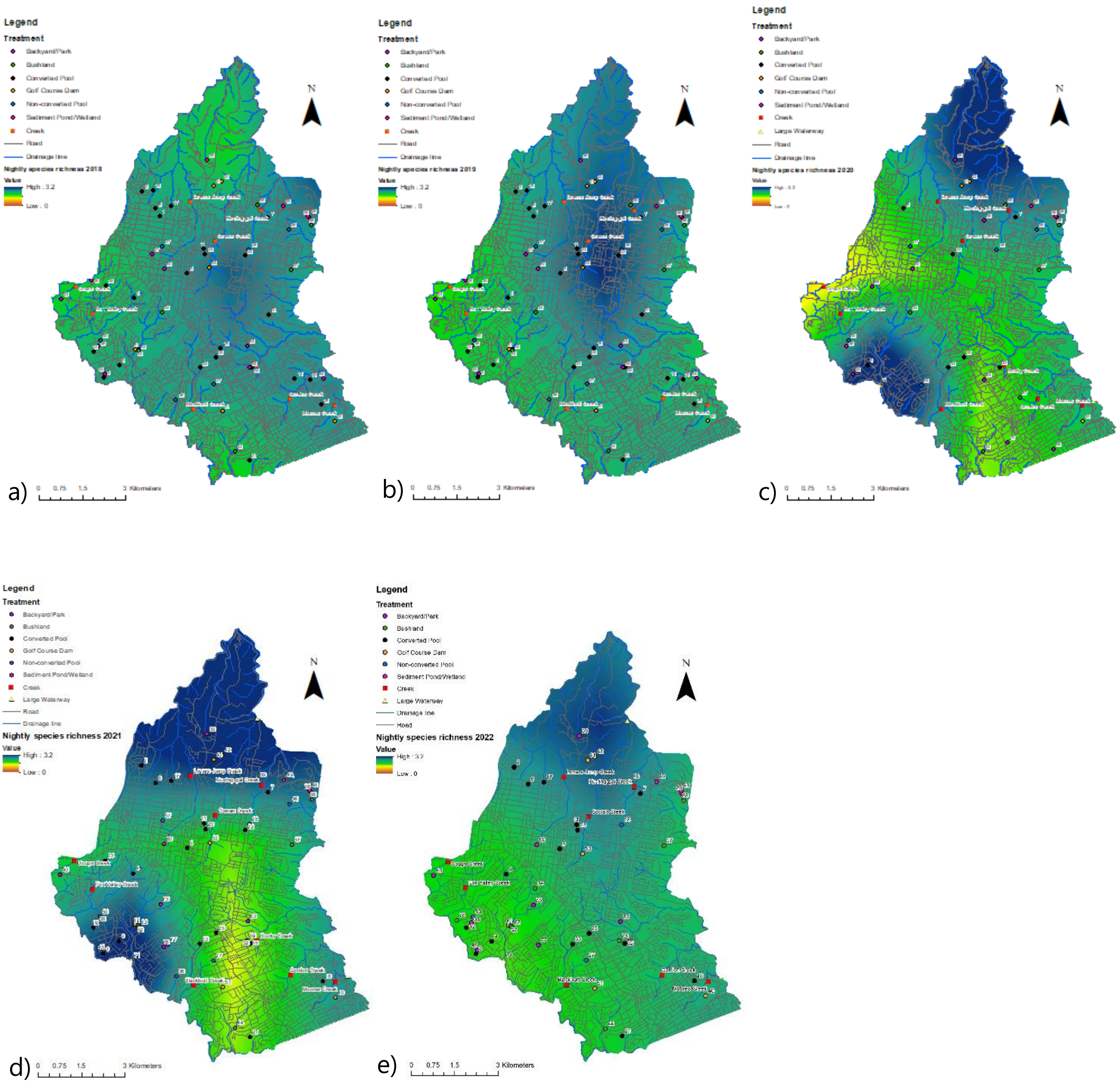
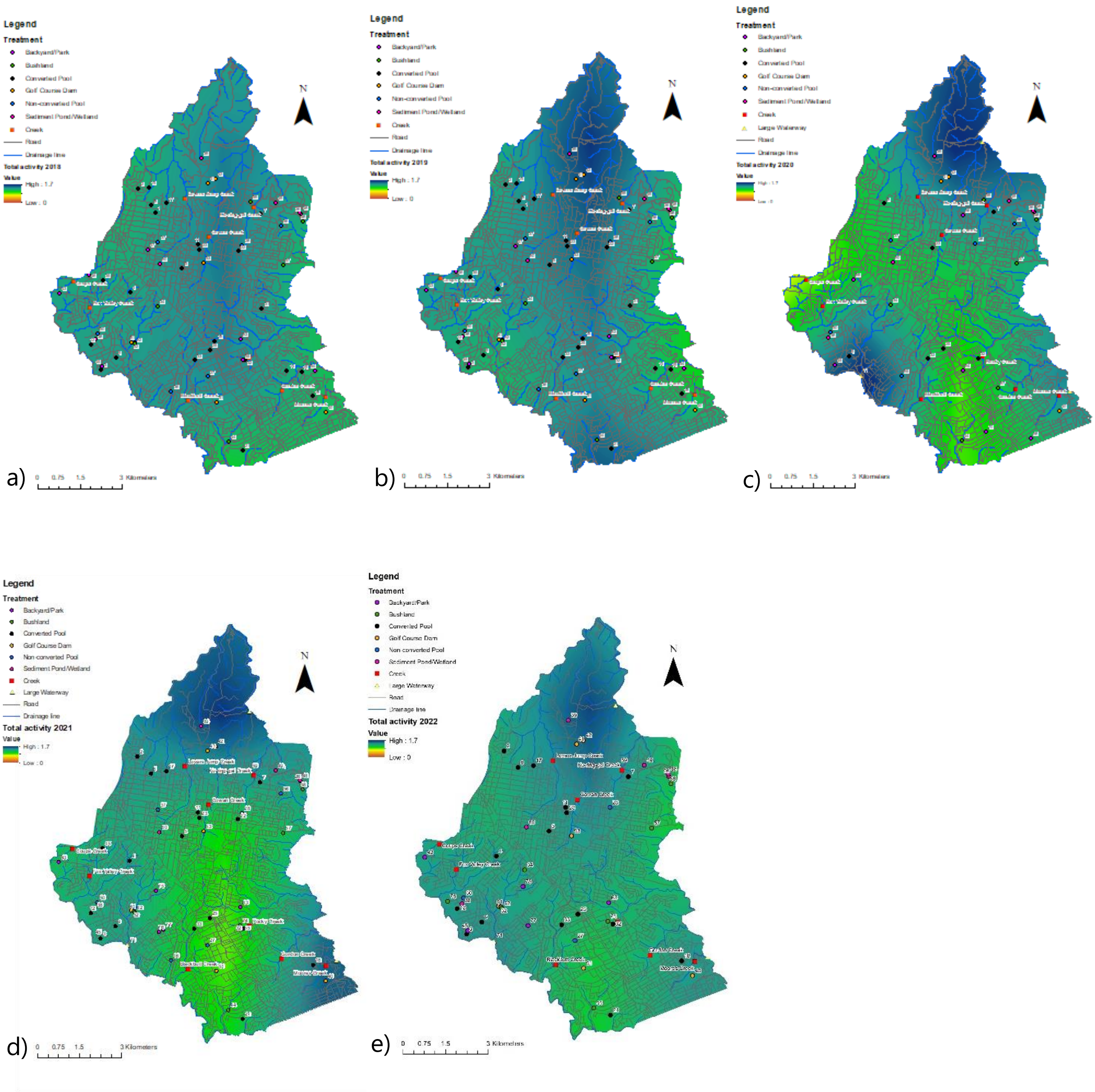
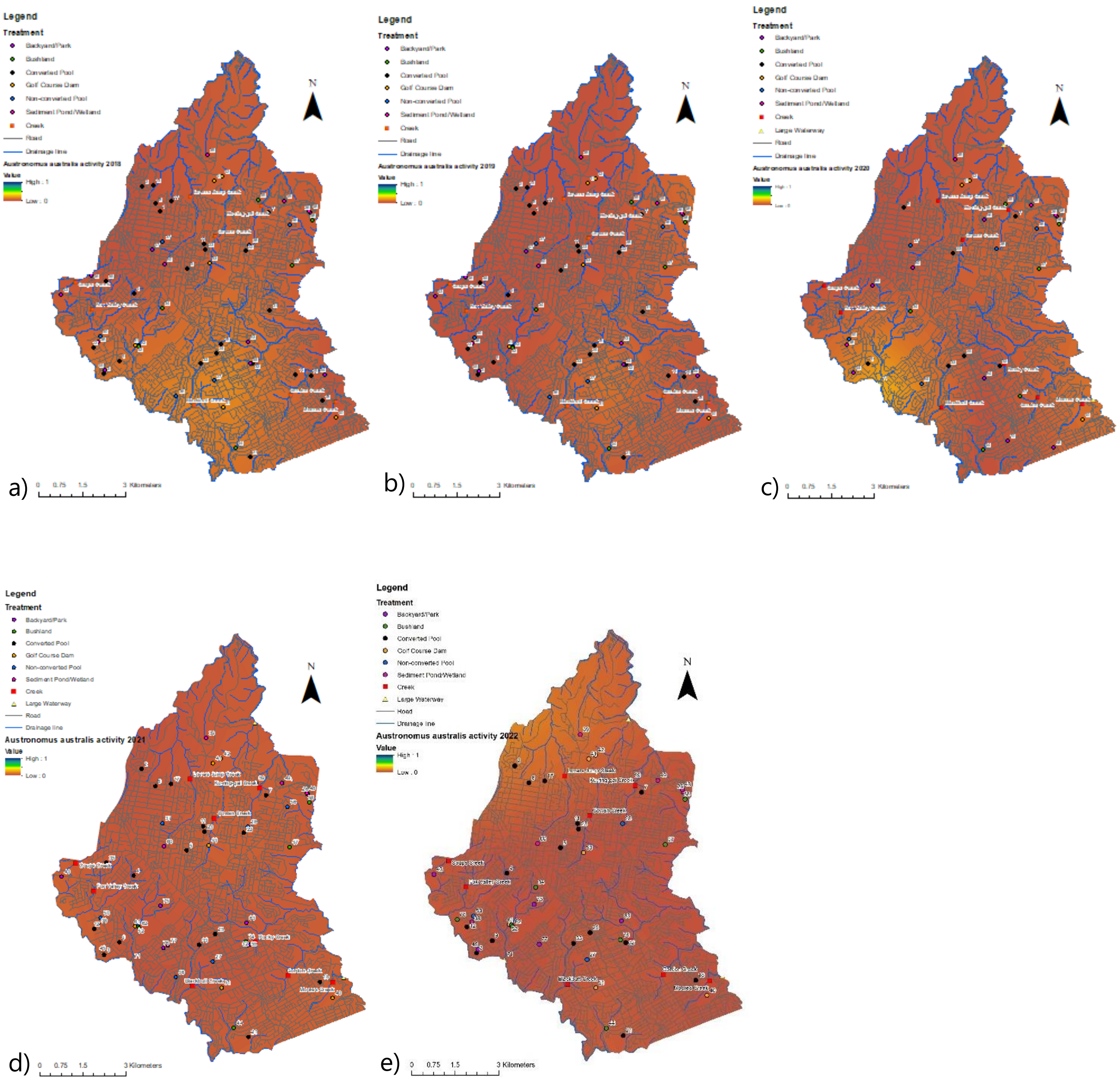
Appendix 1 – Night species richness interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



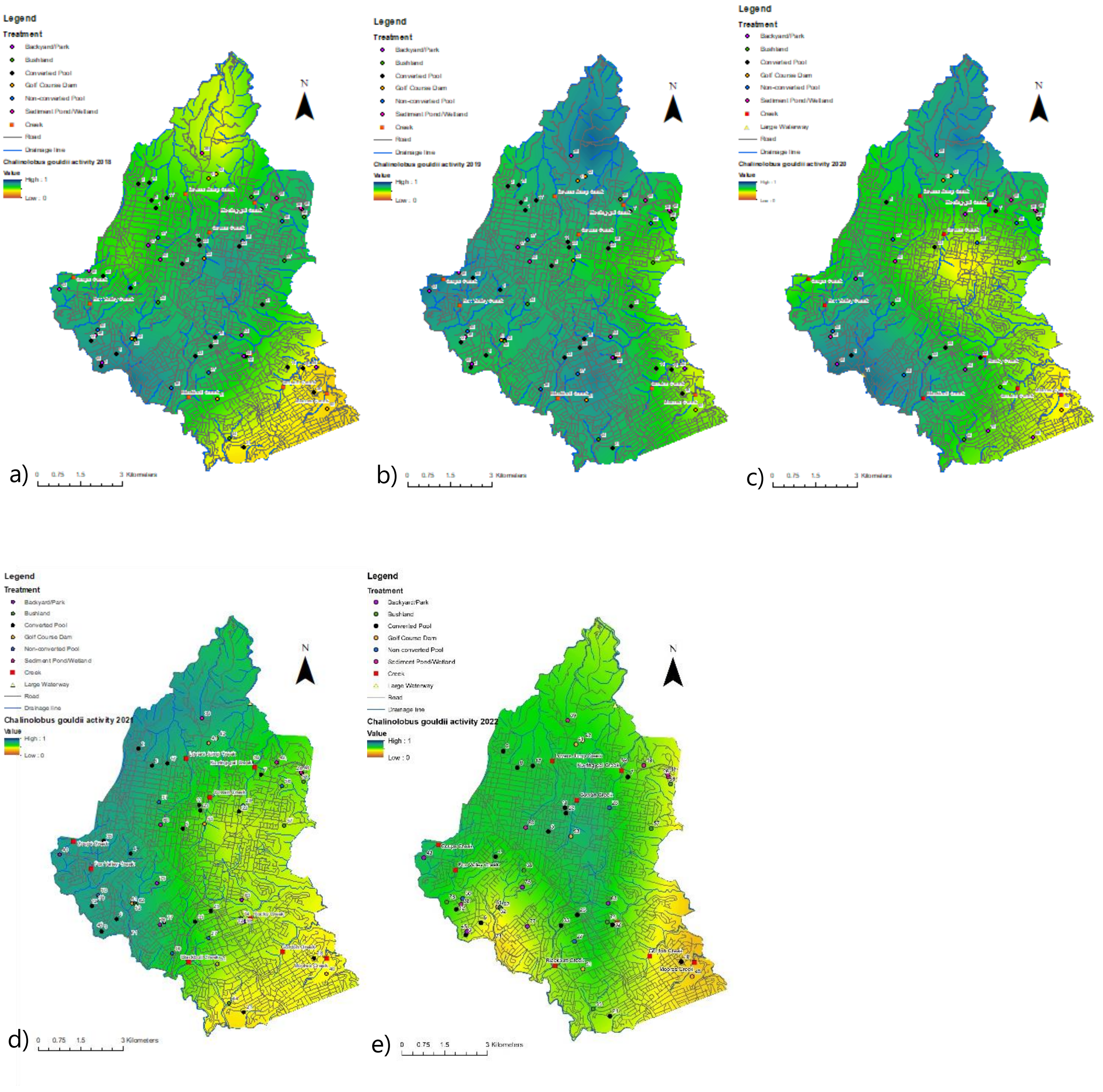
Appendix 2 – Night bat activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



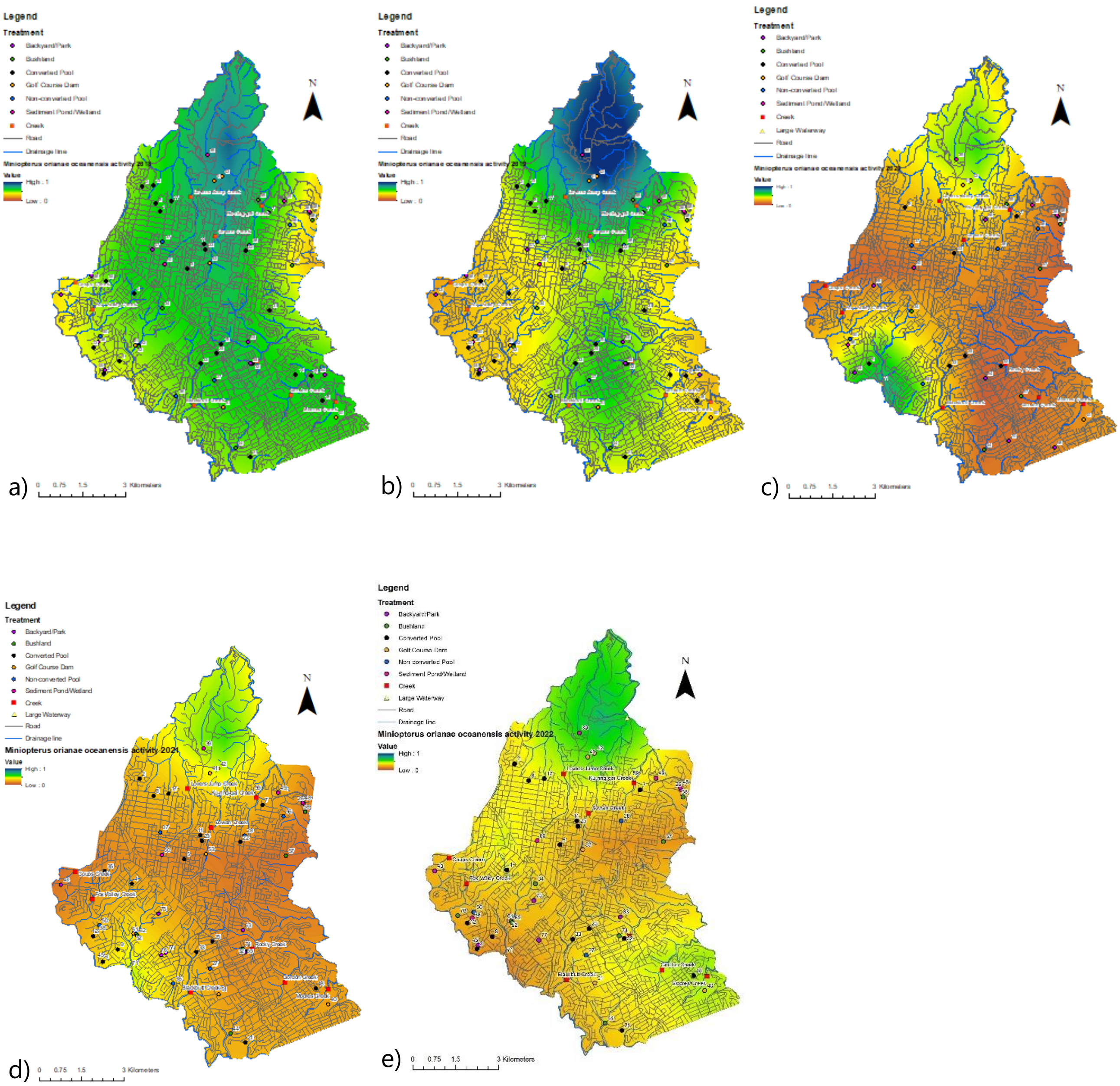
Appendix 3 – *Austronomus australis* activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



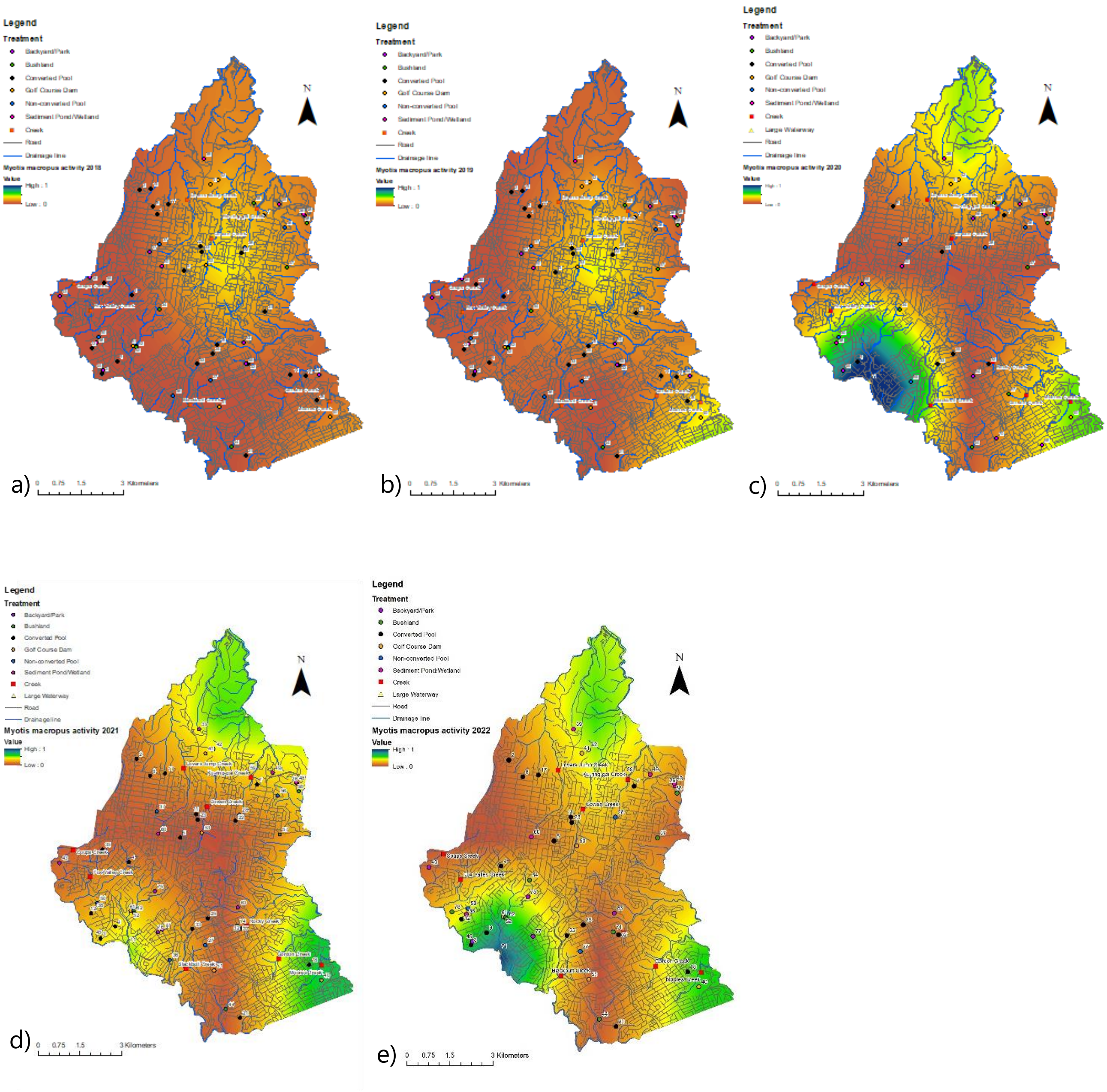
Appendix 4 – *Chalinolobus gouldii* activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



Appendix 5 – *Miniopterus orianae oceanensis* activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



Appendix 6 – *Myotis macropus* activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.



Appendix 7 – *Ozimops ridei* activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019, c) 2020, d) 2021 and e) 2022.