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Insectivorous bat activity over swimming pools retrofitted for wildlife Year 4: 2020

Report prepared for Ku-ring-gai Council

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Insectivorous bat activity over swimming pools retrofitted for wildlife. Year 4: 2020

Report prepared June 2020

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 Chelsea Costello (Ku-ring-gai Council). We thank all residents and land managers for providing access to land for bat monitoring.

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 non-converted pools and creeks. Here we report on years two (2018), three (2019) and four (2020) of monitoring that assess 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 in autumn. 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. Fifteen bat taxa were recorded across all sites, including seven threatened species. Nightly species richness and activity of bats was highest at golf course dams, sediment ponds/wetlands and large waterways, though both response variables were greater in 2019 and 2020 relative to 2018 for golf course dams. Activity at converted and non-converted pools was mostly attributed to urban adapted generalist species, Gould's Wattled Bat *Chalinolobus gouldii* and Eastern Bentwing Bat *Miniopterus orianae oceanensis*, whereas the threatened specialist bat species Large-footed Myotis *macropus* was only detected at golf course dams, sediment ponds/wetlands, large waterways and creeks. The annual trend for activity was stable for most species in years 2-4, except *M. orianae oceanensis*, which showed a decline across all habitat types between 2019 and 2020. This species may have been impacted by the 2019 fires that heavily impacted forests surrounding one of its three known major maternity roosts. We recommend that monitoring is extended into the future to continue 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 for bats may change over time. We suggest that sites used in the current study should be resurveyed annually. Targeted research is needed to identify roosting

habitat for some 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 has been 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 15 years of operation, Council have provided advice or supported the transition of 64 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 fourth 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 (2018-2020) 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 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 make comparisons among years 2-4 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).

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 (~8521 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 1,160 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 ~116,000 residents within the built area, of which 95 % is low density housing and 5 % used for business (Ku-ring-gai Council 2016).

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). Characteristics of each site were recorded *in situ* during bat sampling or later extracted using spatial data and ArcGIS (ESRI) (Table S1). 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.

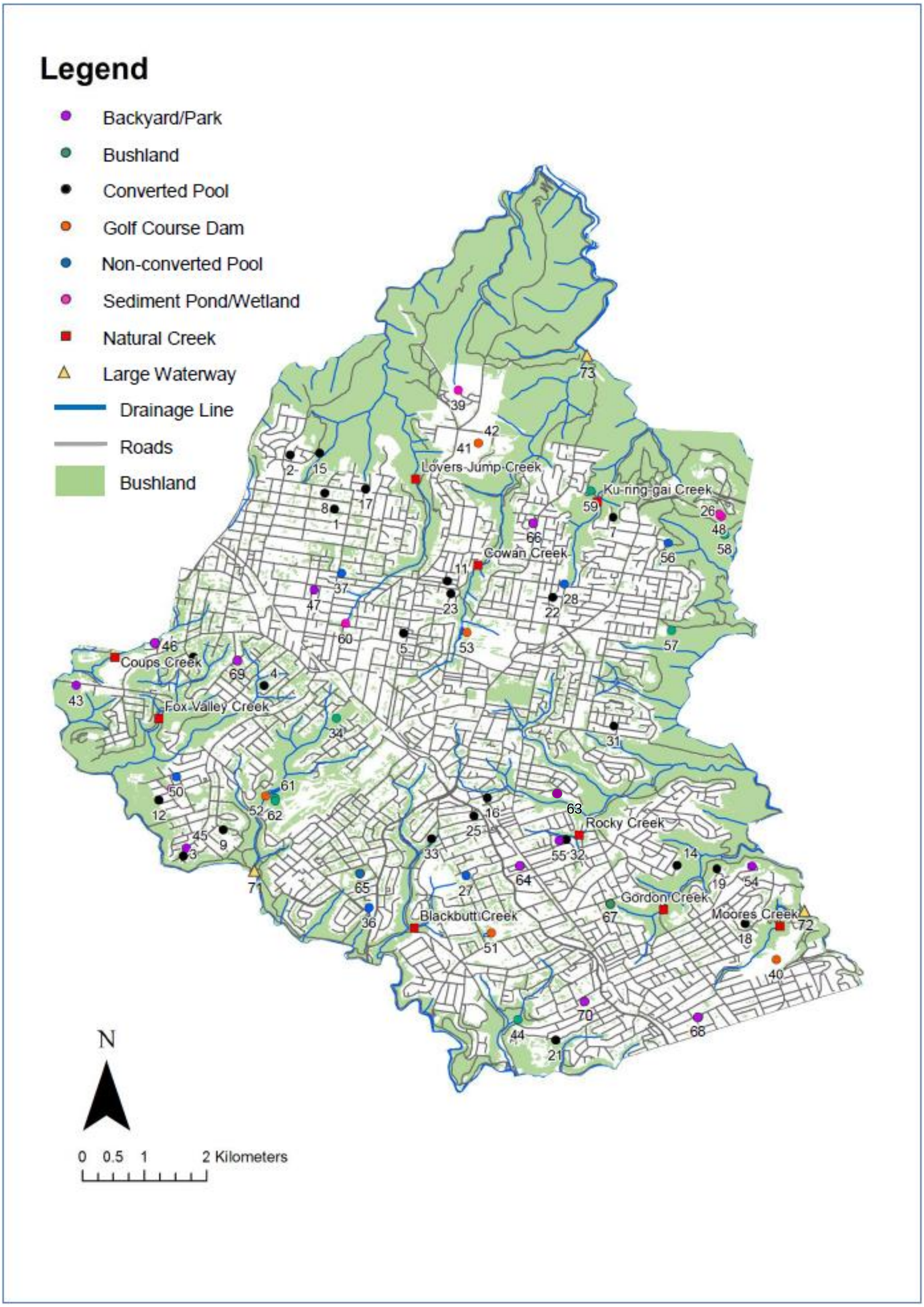


Fig. 1. Location of all (2017-2020) sampling sites in the Ku-ring-gai LGA.


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 55); g) Bushland (site 34); and h) Large Waterway (site 72).

Bat surveys

Bat activity was surveyed for 2-3 nights in autumn 2018-2020. A single AnaBat Express detector (Titley Scientific, Brendale QLD) was deployed at all sites, except for creek sites which were also sampled with AnaBat II + Z-CAIM, AnaBat SD1 and AnaBat SD2 detectors. 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. 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*. In 2020, feeding activity at each site was assessed for the first time by documenting the occurrences of feeding buzzes (Griffin et al. 1960). Feeding buzzes were identified using a feeding buzz filter in AnaScheme and manual verification of files that matched the filter.

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 was used as a random factor and minimum daily temperature (BOM weather station: Terrey Hills AWS, 066059) was used as a covariate as bat activity can be positively correlated with nightly temperature (O'Donnell 2000; Erickson and West 2002). All bat data were $\log_{10}$ (x+1) transformed prior to analysis. All plots for GLMMs were constructed using modelled estimates that accounted for the influence of random effects associated with each site and also variation in minimum daily temperature. 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 Density 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 2020 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 six or more sites (i.e., ~≥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 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 on the basis of bat assemblages. 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, 30,756 bat calls were recorded in autumn between 2018 and 2020. Of these, 17,969 (58 %) were identified to one of 15 taxa (Table 1). All other calls were usually poor quality and of short duration, and therefore could not be assigned a species-level identification. Large-eared Pied Bat *Chalinolobus dwyeri* was recorded in 2018 but not in subsequent years. Eastern False Pipistrelle *Falsistrellus tasmaniensis* was recorded in 2018 and 2019 but not recorded in 2020. Eastern Broad-nosed Bat *Scotorepens orion* was recorded for the first time since 2017 surveys that were carried out in summer. In 2020, the species was recorded at a bushland site that hadn't previously been sampled. All other taxa were recorded in each year of monitoring. In 2020, a total of 34 feeding buzzes were identified from converted pools (n=1 site), sediment ponds/wetlands (n=2), golf course dams (n=3), non-converted pools (n=1), creeks (n=1), large waterways (n=1) and bushland (n=1) (Table 2).

Table 1. Number of sites at which bat species were detected in the Ku-ring-gai LGA (2018-2020).

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

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

Table 2. Number of feeding buzzes recorded in each habitat type in the Ku-ring-gai LGA in 2020.

Species	Common Name	Converted Pool	Non-converted Pool	Sediment Pond/ Wetland	Golf Course Dam	Natural Creek	Large Waterway	Backyard/Park	Bushland
Chalinolobus gouldii	Gould's Wattled Bat	0	0	5	3	0	0	0	0
*Micronomus norfolkensis**	East-coast Freetail Bat	0	0	0	4	0	0	0	0
*Miniopterus australis**	Little Bentwing Bat	1	0	0	1	0	0	0	0
*Miniopterus orianae oceanensis**	Eastern Bentwing Bat	0	0	1	3	0	1	0	1
*Myotis macropus**	Large-footed Myotis	0	0	0	5	1	1	0	0
Ozimops ridei	Eastern Freetail Bat	0	1	0	6	0	0	0	0

Comparison among habitat types

Nightly bat species richness in 2020 was highest at golf course dams, sediment ponds/wetlands and large waterways but lowest at creeks. Bat species richness was significantly affected by the interaction of habitat type by year ($F_{12,440.72}$=2.319, P=0.007; Fig. 3; Appendix 1a-c), with species richness at golf course dams ~28-44 % greater in 2020 and 2019 compared to 2018, with 1-2 additional species recorded per night on average. There was a ~57 % reduction in nightly species richness at backyards/parks between 2018 and 2020. At all other habitat types, nightly species richness remained similar among years. However, 4 taxa (*C. dwyeri*, *Nyctophilus* spp., Eastern Horseshoe Bat *Rhinolophus megaphyllus* and Greater Broad-nosed Bat *Scoteanax rueppellii*) recorded at converted pools were not recorded at non-converted pools and/or backyards/parks (Table 1).

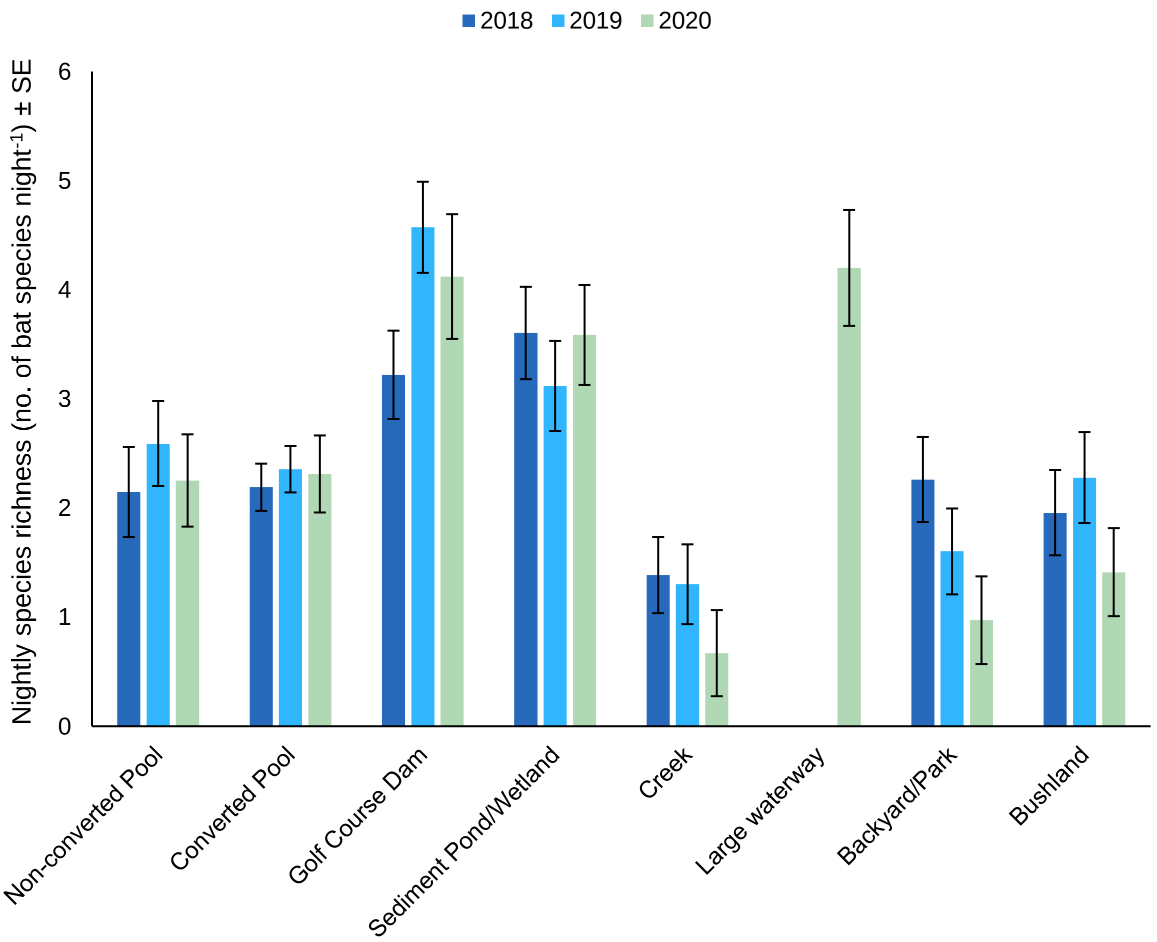


Fig. 3. Nightly bat species richness recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

In 2020, log_{10}-transformed total bat activity (no. passes $night^{-1}$) ranged from 0.7-2.0 passes $night^{-1}$ and was lowest at creeks but greatest at golf course dams. Total bat activity was significantly affected by the interaction of habitat type and year ($F_{12,432.09}=2.059$, $P=0.018$; Fig. 4; Appendix 2a-c). Activity levels were ~44 % higher in 2019 compared to 2018 at golf course dams, and ~32 % lower on creeks in 2020 compared to 2018, whereas activity remained similar among years for all other habitat types. Activity on large waterways that were only sampled in 2020 was comparable to golf course dams and sediment ponds/wetlands.

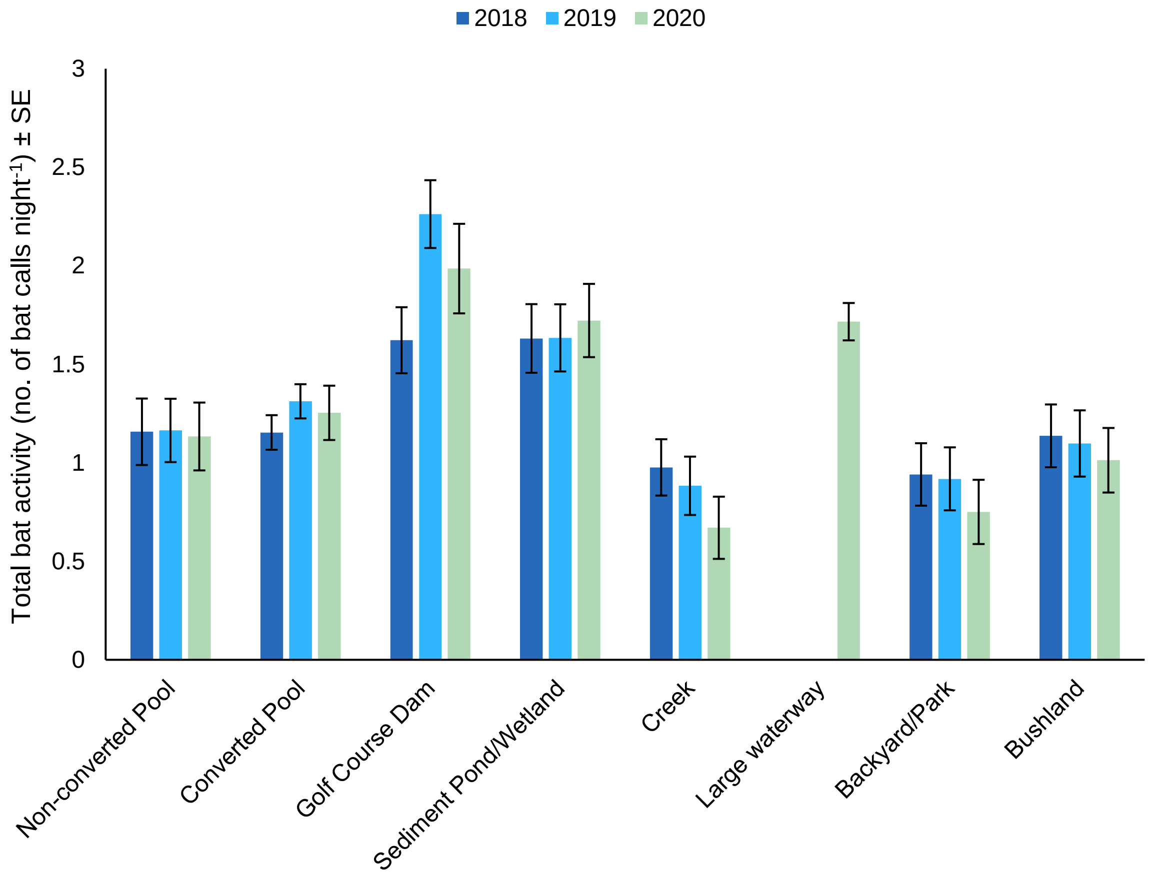


Fig. 4. Nightly bat activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

Activity of Individual Species

The White-striped Freetail Bat *Austronomus australis* was not recorded on creeks and was infrequently detected in the other habitat types during the study but was recorded at all large waterways sampled in 2020 (Table 1). Nightly activity of *A. australis* was significantly affected by the interaction of habitat type and year ($F_{12,406.04}$= 2.438, P=0.005; Fig. 5; Appendix 3a-c). Activity in 2019 and 2020 was ~53-70 % lower than 2018 at non-converted pools and golf course dams, whereas there was no difference among years for converted pools, backyards and bushland sites. In 2020, activity of *A. australis* over large waterways was highly variable.

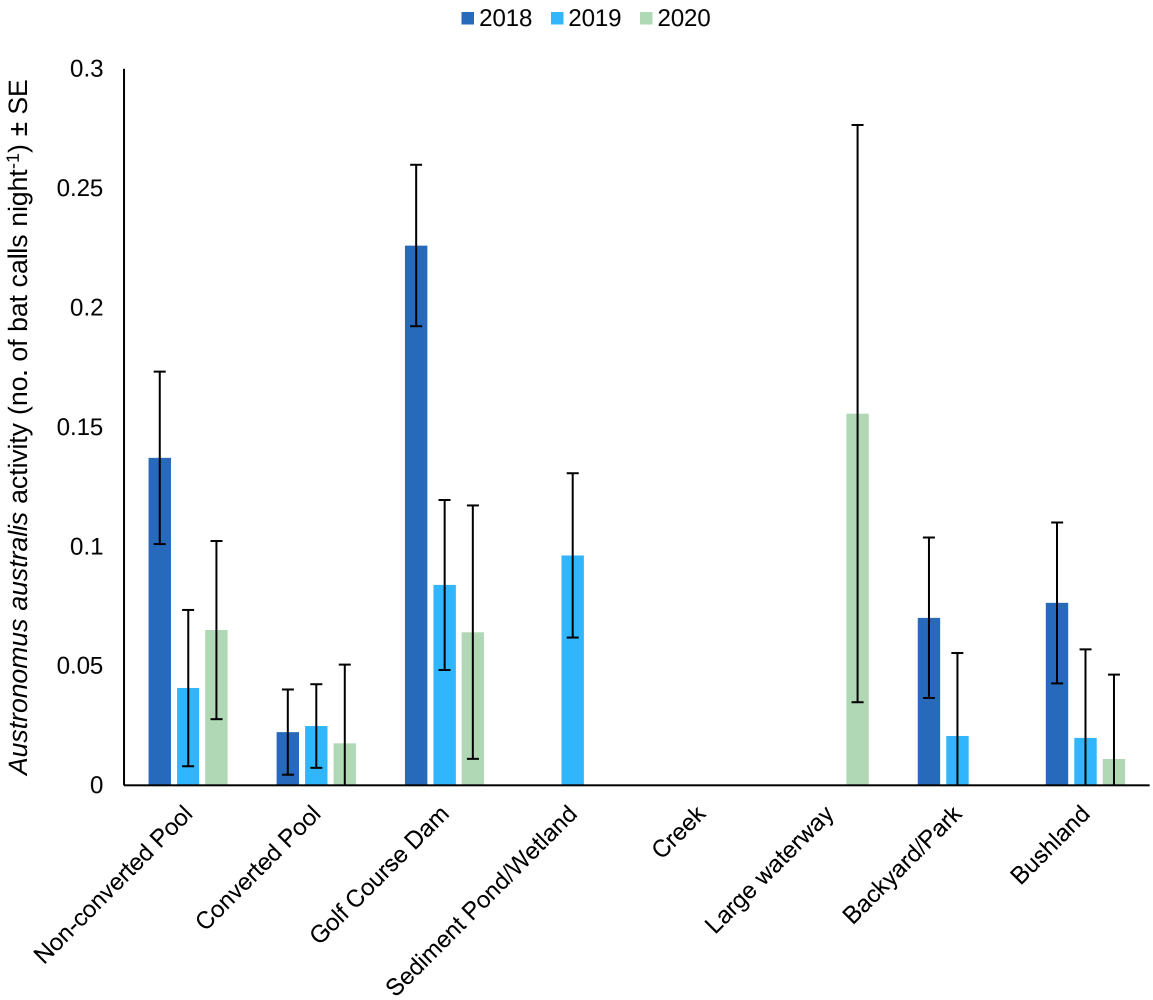


Fig. 5. Nightly *Austronomus australis* activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

Gould's Wattled Bat *Chalinolobus gouldii* was commonly detected in all habitat types during the study, except for creeks where the species was not detected (Table 1). *Chalinolobus gouldii* activity was significantly affected by the interaction of habitat type and year ($F_{12,436.51}$=4.136, P<0.001; Fig. 6; Appendix 4a-c). Activity at golf course dams was ~75 % greater in 2020 and 2019 relative to 2018, whereas activity at converted pool sites was ~33 % lower in 2020 relative to 2019. Activity of *C. gouldii* at all other habitat types remained similar among years. In 2020, activity on large waterways was ~55-60% lower than golf course dams and sediment ponds/wetlands, but comparable to most other habitat types.

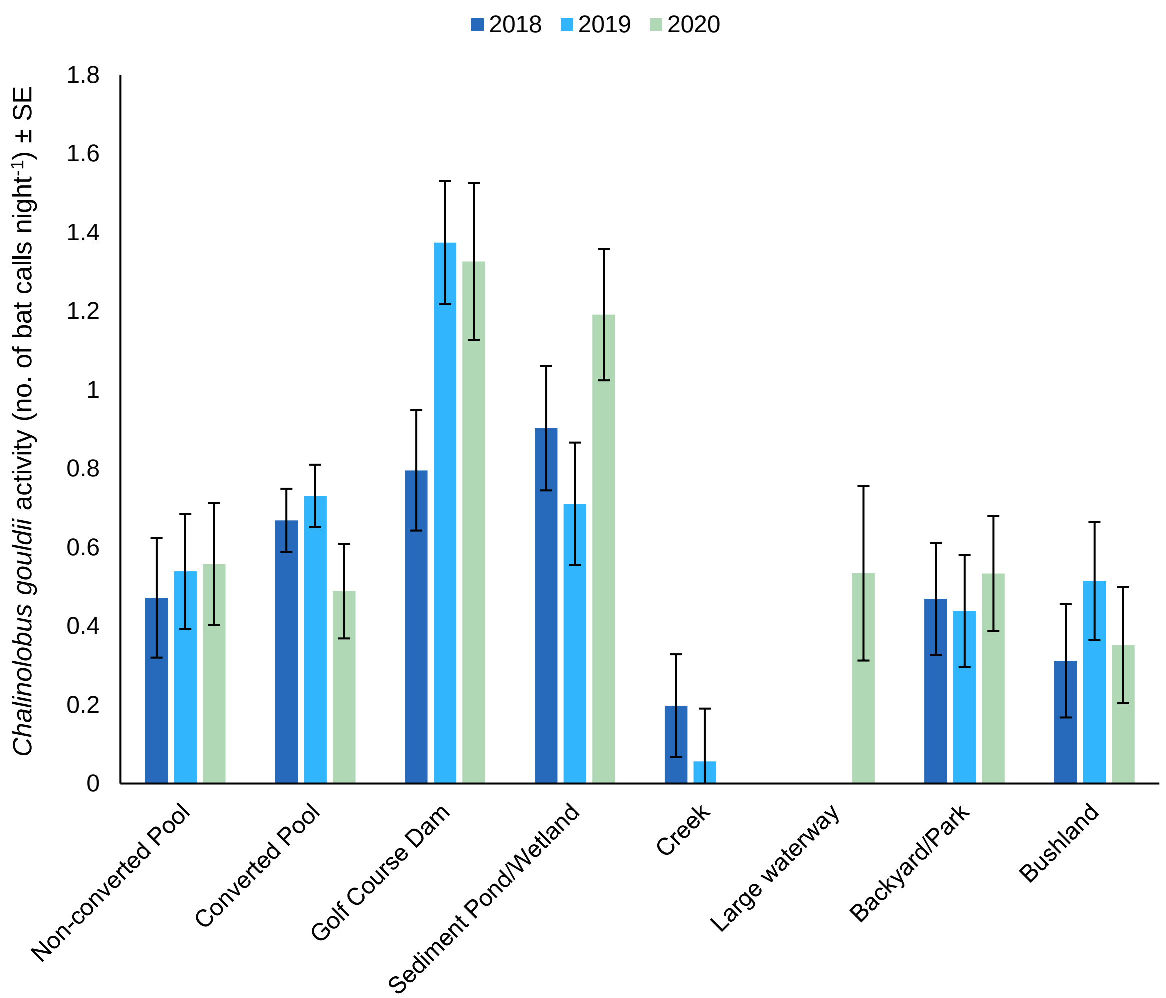


Fig. 6. Nightly *Chalinolobus gouldii* activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

The Eastern Bentwing Bat *Miniopterus orianae oceanensis* was detected at ~75-80 % of sites sampled in 2018 and 2019 and was recorded in each habitat type (Table 1). The species was less common in 2020 and detected at ~44 % of sites but was undetected in backyards/parks (Table 1). Nightly activity was significantly affected by the interaction of habitat type by year ($F_{12,453.19}$=3.544, P<0.001; Fig 7; Appendix 5a-c). Activity at golf course dams was 49 % greater in 2019 relative to 2018, whereas activity in bushland sites was 67 % lower in 2019. In 2020, activity reduced significantly by ~44-83 % at non-converted pools, converted pools, golf course dams and sediment ponds/wetlands and there was a trend for a reduction in all other habitat types, though activity was typically low among years.

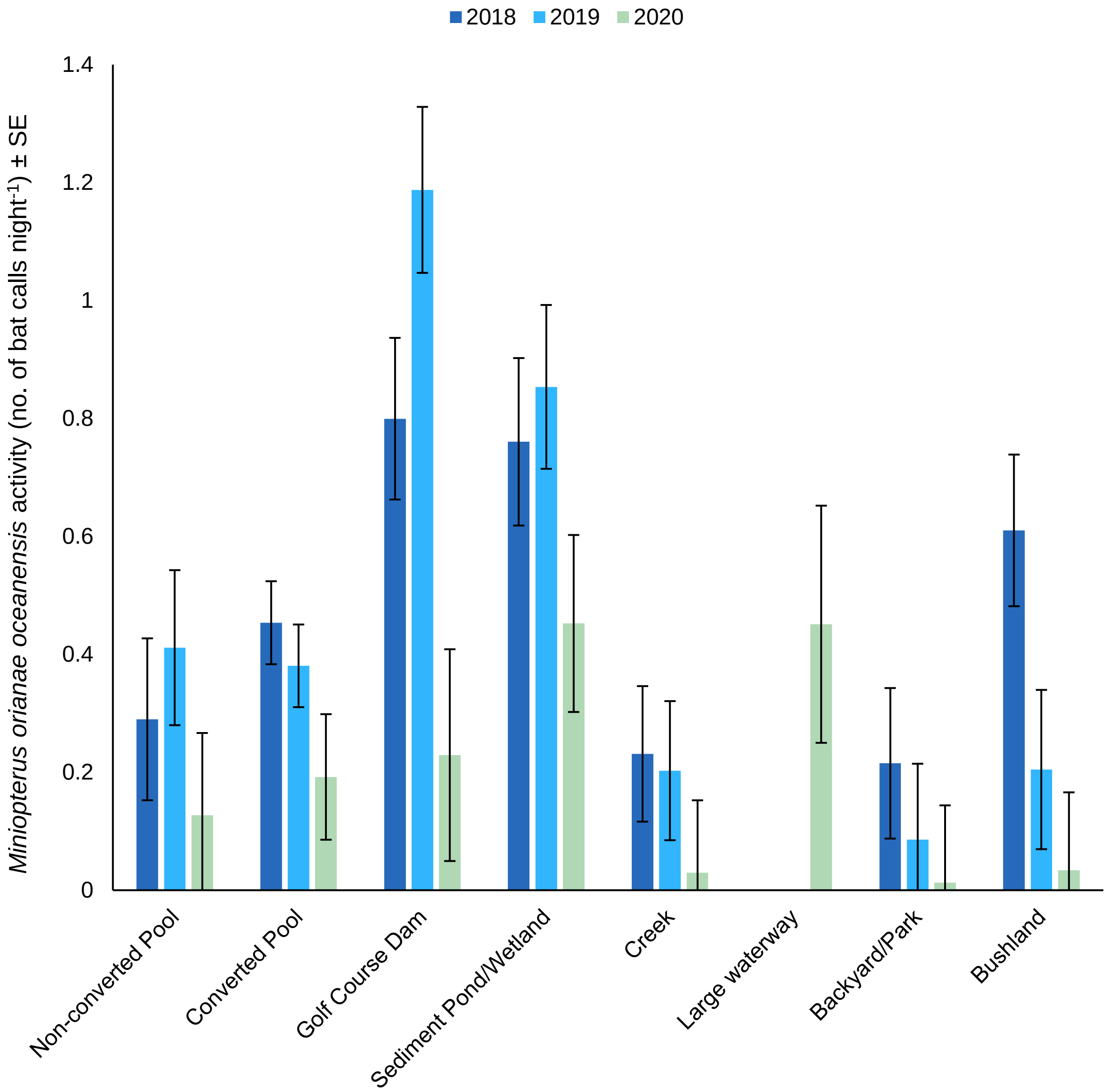


Fig. 7. Nightly *Miniopterus orianae oceanensis* activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

Myotis macropus was only detected at golf course dams, sediment ponds/wetlands and a single creek site between 2018 and 2020 (Table 1). 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 ($F_{8,280.864}$=2.722, P=0.007; Fig. 8; Appendix 6a-c). Activity at golf course dams in 2019 was 59 % greater than 2018, whereas activity in 2020 was intermediate. Activity of *M. macropus* at sediment ponds/wetlands and creeks remained similarly low among years. In 2020, activity on large waterways was highly variable but comparable to golf course dams.

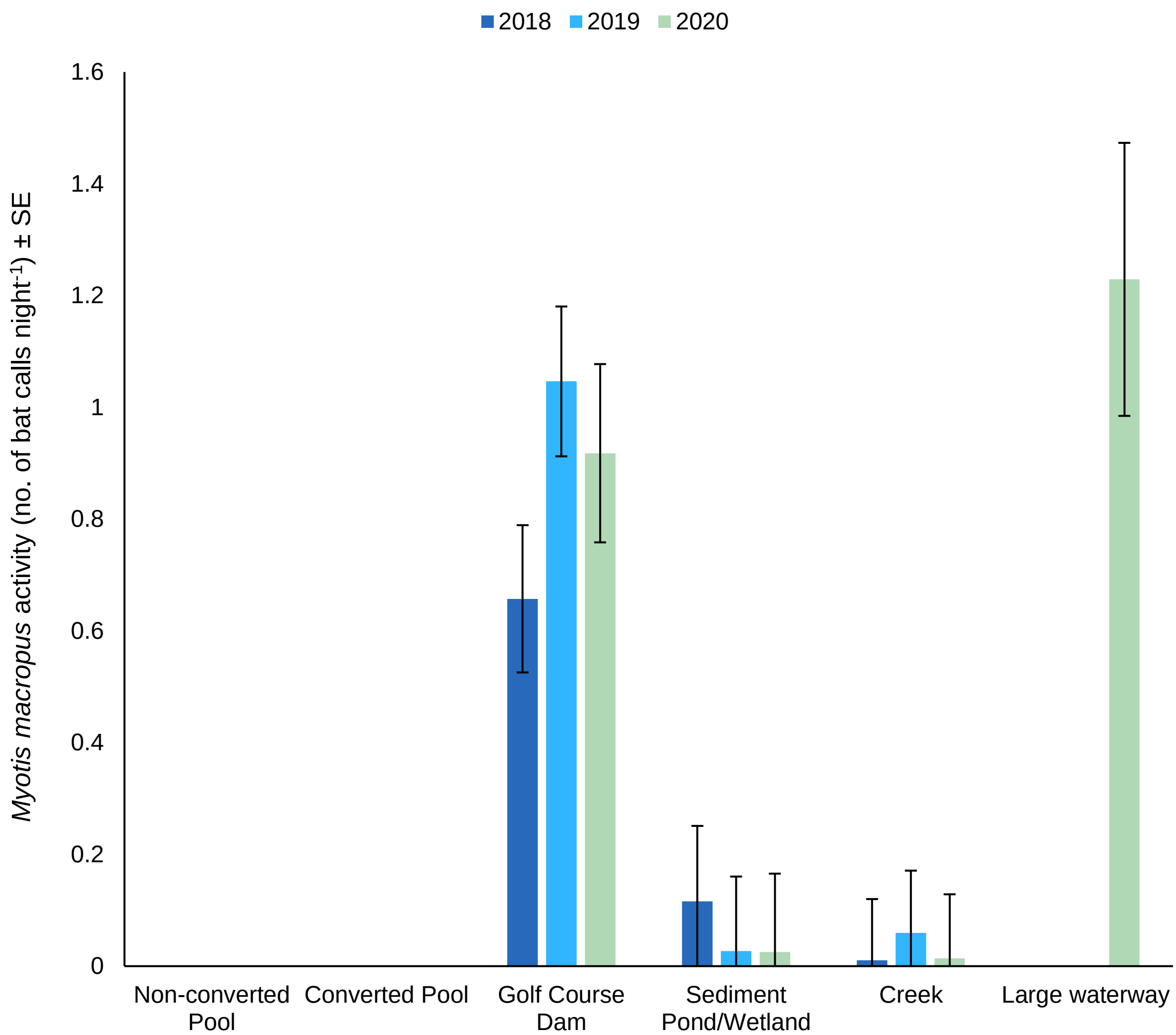


Fig. 8. Nightly *Myotis macropus* activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA between 2018 and 2020.

In 2018, the Eastern Freetail Bat *Ozimops ridei* was detected at 65 % of sites and in each habitat type, whereas in 2020 and 2019 the species was recorded at 56-69 % of sites but not on creeks (Table 1). *Ozimops ridei* activity was not significantly affected by the interaction of habitat type by year ($F_{12,443.08}=1.026$, $P=0.424$; Appendix 7a-c), nor year alone ($F=_{2,374.45}=2.110$, $P=0.123$). However, *O. ridei* activity differed significantly among habitat types ($F=_{6,67.42}=8.924$, $P<0.001$; Fig. 9), with activity at golf course dams 3-24 times greater than all other habitat types (non-converted pools: $t=3.872$, $P=0.005$; converted pools: $t=4.962$, $P<0.001$; sediment ponds/wetlands: $t=4.807$, $P<0.001$; creeks: $t=6.782$, $P<0.001$; backyards: $t=5.601$, $P<0.001$ and bushland: $t=5.570$, $P<0.001$). Activity at converted pools ($t=2.901$, $P=0.05$) and non-converted pools ($t=3.258$, $P=0.028$) was ~10-times greater than creeks, whereas all other habitat types did not differ from each other. In 2020, activity of *O. ridei* on large waterways was ~40 % lower than golf course dams, but comparable to all other habitat types except creeks where the species was not detected.

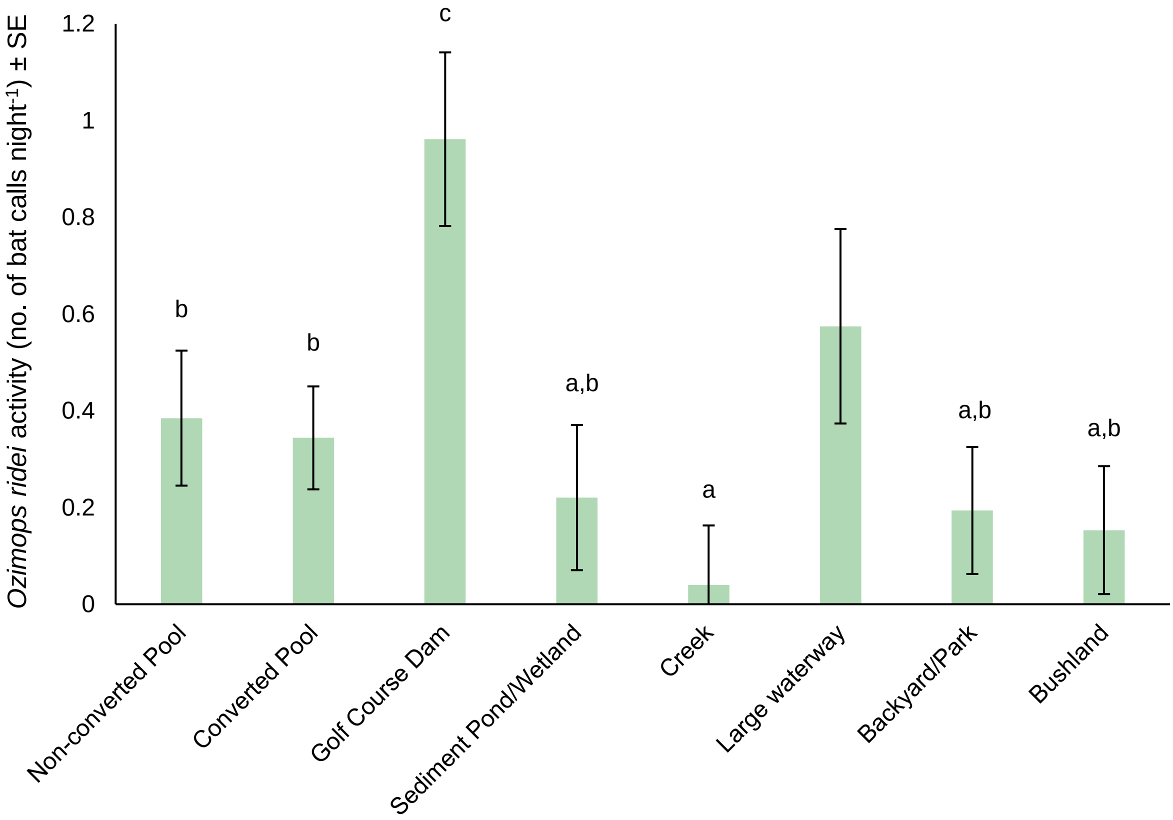


Fig. 9. Nightly *Ozimops ridei* activity (log_{10}-transformed) recorded among habitat types in the Ku-ring-gai LGA using data pooled across 2018, 2019 and 2020. Means denoted by different letters are significantly different from each other. Pairwise comparisons between large waterway and other treatments could not be undertaken as this treatment was not sampled in previous years.

Chalinolobus dwyeri, Chocolate Wattled Bat *C. morio*, *F. tasmaniensis*, Little Bentwing Bat *M. australis*, East-coast Freetail Bat *Micronomus norfolkensis*, *Nyctophilus* spp., *R. megaphyllus*, *S. rueppellii*, *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 2018-2020 revealed that bat assemblages differed among habitat types (Fig. 10). The bat assemblage of creek sites was generally distinct from all other habitat types and was characterised by low levels of *C. gouldii* and *O. 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.

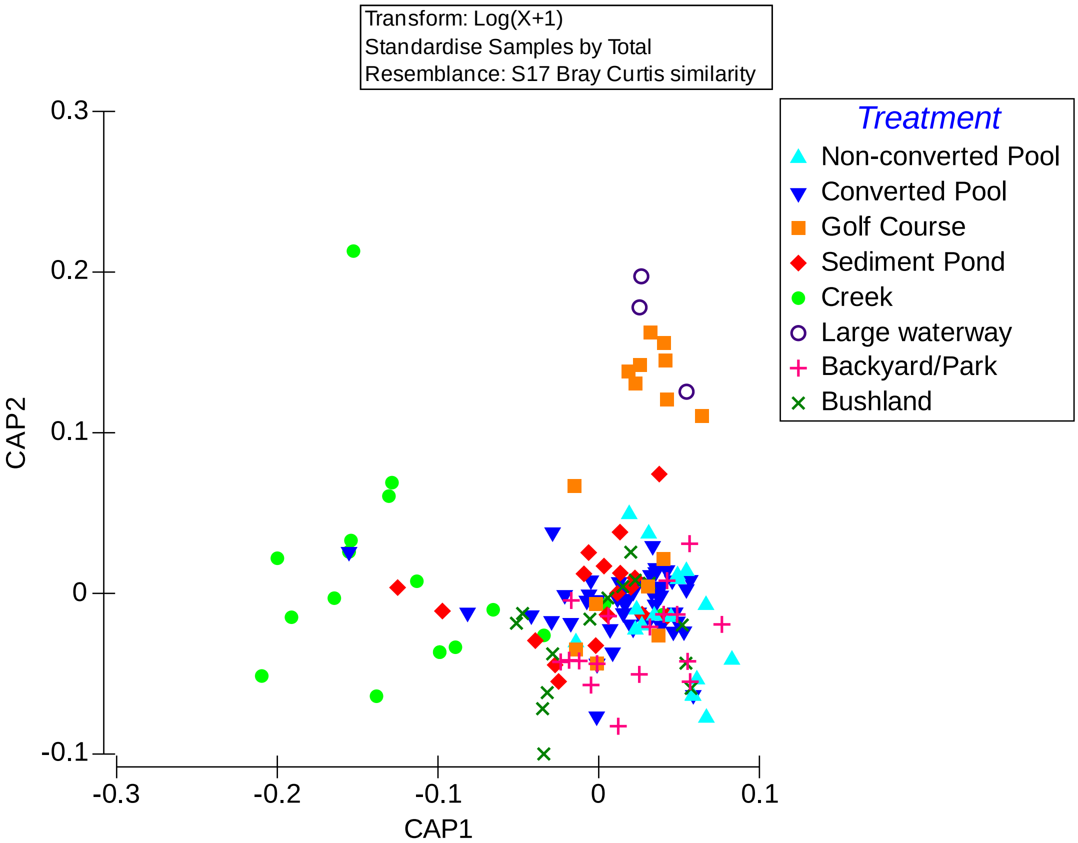


Fig. 10. Canonical analysis of principal coordinates (CAP) plot illustrating differences between habitat types on the basis of bat species assemblages.

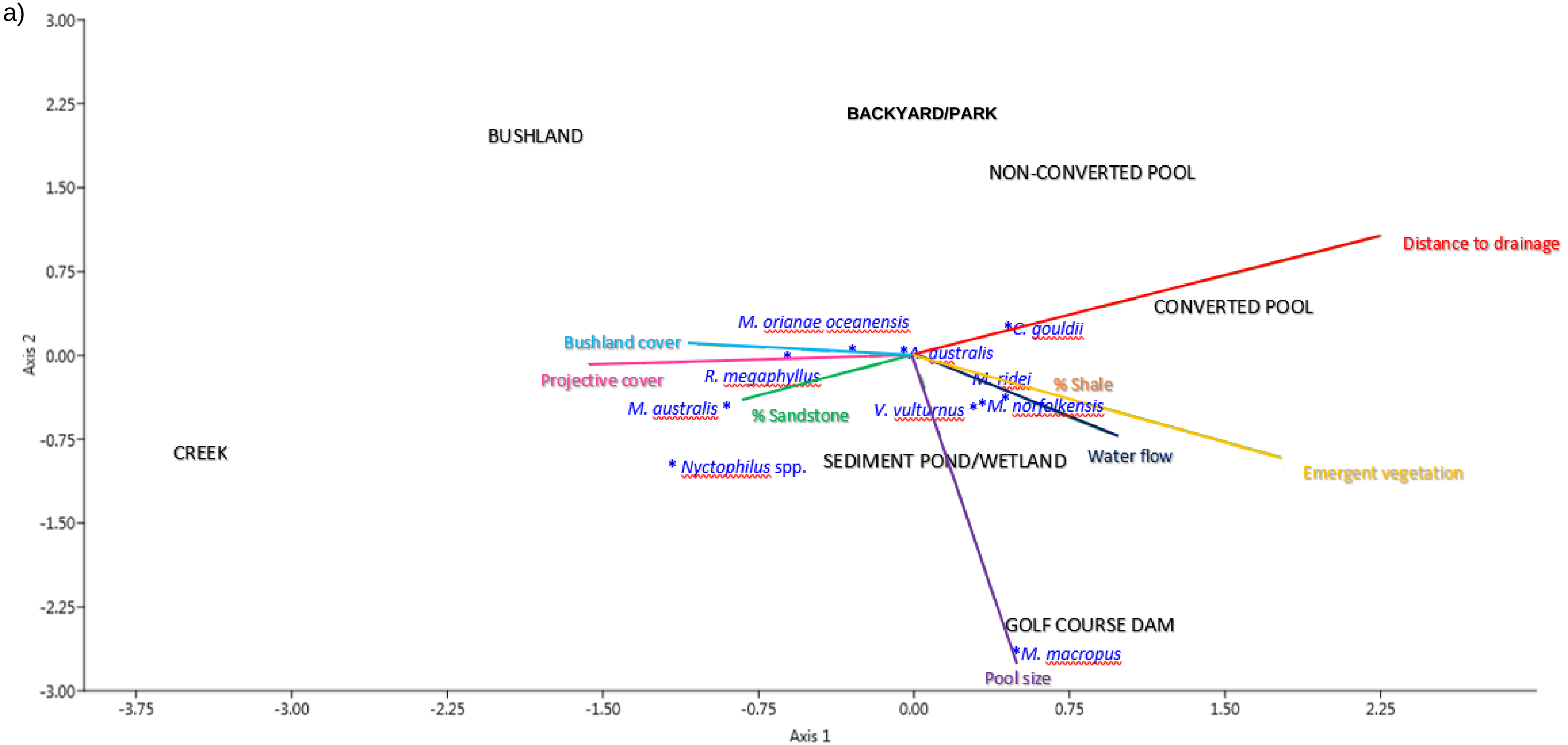
Relationship between bat activity and environmental variables

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.

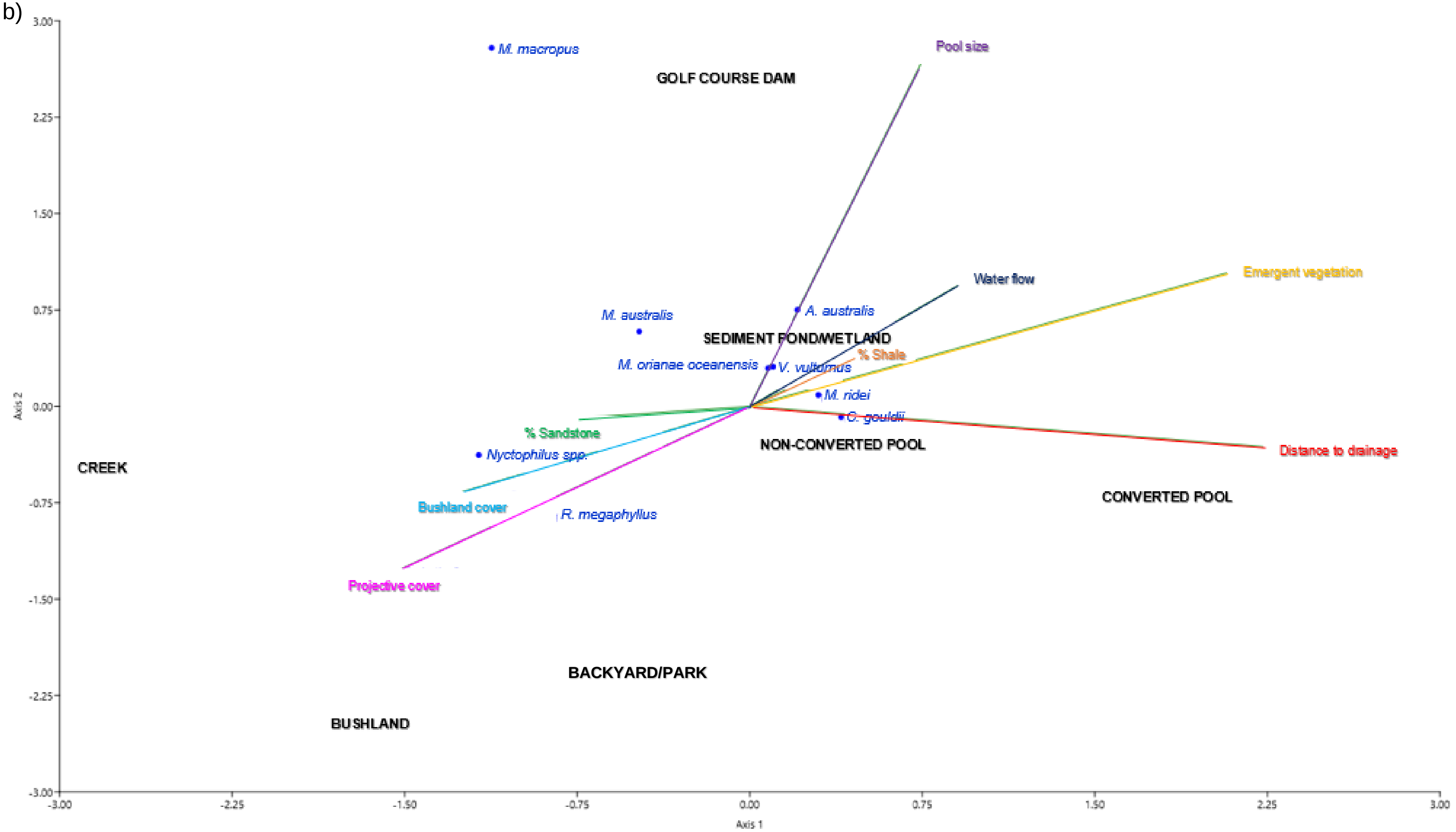
CCA biplots for 2018, 2019 and 2020 data revealed that creek and bushland sites were associated with sandstone geology, greater bushland, projective foliage cover, and were near drainage lines (Fig. 11a-c). Backyards/parks, non-converted pools and converted pools varied greatly in distance from drainage 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 (range; 70-5500 m^2) (Fig. 11a-c). Large waterways were near bushland with underlying sandstone geology had lower levels of emergent vegetation than golf course dams and sediment ponds/wetlands (Fig. 11c).

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 (in 2020) (Fig. 11a-c). *Miniopterus australis*, *Nyctophilus* spp., *R. megaphyllus* were all positively associated with projective foliage cover, bushland cover, sandstone geology, and were negatively associated with distance from drainage lines, whereas *M. orianae oceanensis* was associated with bushland cover and projective foliage cover in 2018 and was weakly associated with pool size in 2019 and sandstone geology in 2020 (Fig. 11a-c). *Chalinolobus gouldii*, *M. norfolkensis and O. ridei* were associated with shale geology and low-moderate distances from drainage lines, whereas *V. vulturnus* was weakly associated with pools size and to a lesser extent, water flow in 2018 and 2019 (Fig. 11a-c). In 2020, *V. vulturnus* was weakly associated with shale geology and low-moderate levels of bushland and projective foliage cover. Although in 2018 *A. australis* was not associated with any environmental variable quantified in this study, the species was associated with pools of a moderate size in 2019 and 2020 (Fig. 11a-c).

a)


BACKYARD/PARK
BUSHLAND
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
Water flow
CREEK
* Nyctophilus spp.
SEDIMENT POND/WETLAND
Emergent vegetation
GOLF COURSE DAM
*M. macropus
Pool size
Axis 1
Axis 2
3.00
2.25
1.50
0.75
0.00
-0.75
-1.50
-2.25
-3.00
-3.75
-3.00
-2.25
-1.50
-0.75
0.00
0.75
1.50
2.25

b)



3.00
2.25
1.50
0.75
0.00
-0.75
-1.50
-2.25
-3.00
Axis 2
-3.00
-2.25
-1.50
-0.75
0.00
0.75
1.50
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
Bushland cover
R. megaphyllus
CONVERTED POOL
Projective cover
BACKYARD/PARK
BUSHLAND

c)

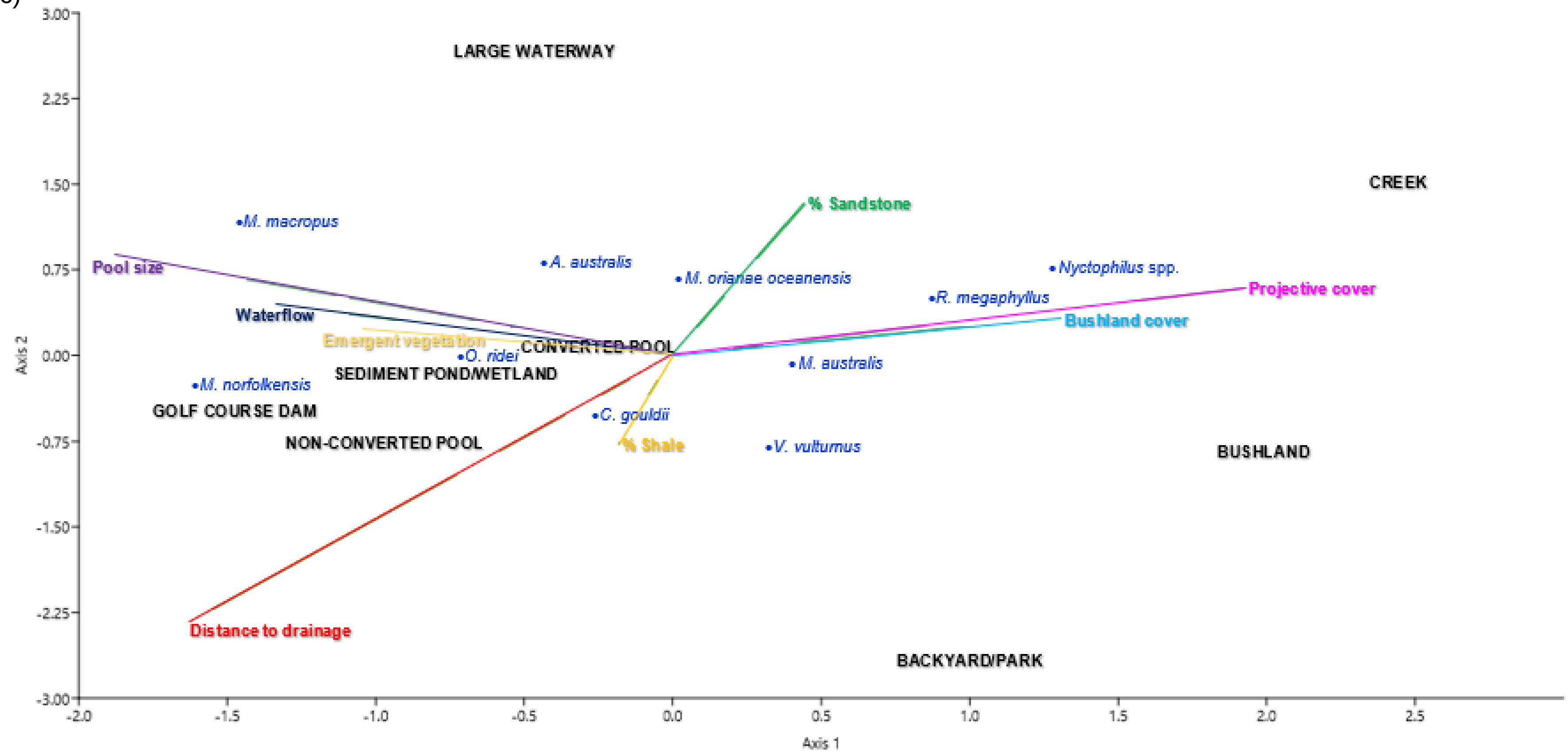


Fig. 11. Canonical correspondence analysis bi-plot illustrating associations between activity of bat taxa and environmental variables in a) 2018, b) 2019 and c) 2020. Centroids for the different habitat types are also plotted.

Discussion

Four years of monitoring activity of insectivorous bats at constructed wildlife ponds in the Ku-ring-gai LGA, with additional context provided by other habitat types that were monitored in the last three years, has provided some clear and consistent results. Nightly bat species richness and activity was greatest at golf course dams and lowest at creeks, with both variables remaining similar between 2018 and 2020 for all habitat types except golf course dams, which had 1-2 more species per night on average and 25-44 % more activity in 2019 and 2020. Greater activity at golf course dams and sediment ponds/wetlands was primarily driven by *M. macropus*, *M. ridei* and *C. gouldii*. Feeding activity in 2020 broadly reflected patterns for activity, with the greatest number of feeding buzzes recorded at golf course dams and sediment ponds/wetlands. 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, except in 2020 when richness and activity was lower in the latter two habitat types. Furthermore, up to three taxa 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 (>3400 m^2) with low levels (<10 %) of emergent aquatic vegetation cover present in some sediment ponds/wetlands and golf course dams but was also infrequently recorded on smaller pools (90 m^2) on creeks. Activity of most species remained stable between 2019 and 2020, except for *M. orianae oceanensis* that had an LGA-wide reduction in activity. Ongoing monitoring is required to assess long-term responses by insectivorous bats as converted pool habitats continue to develop and to track trends in bat activity at other habitats within the LGA.

The value of waterbodies for insectivorous bats

Nightly bat species richness across four years of monitoring was greatest at golf course dams and sediment ponds/wetlands, with 1-2 more species per night relative to other habitat types, except large waterways that supported a similar number of species in 2020. Species richness at converted pools (~2.3 species per night), non-converted pools (~2.3 species per night), creeks (~1.1 species per night), backyards/parks (~1.6 species per night) and bushland sites (~1.9 species per night) was stable among years, though 38-50 % lower in 2020 for the latter three habitat types. The level of species richness recorded during the study was comparable to that recorded for backyards in the leafier parts of Sydney (Basham et al. 2011) and backyards in vegetated landscapes within the Sydney Metropolitan region (Threlfall et al. 2011). Three species (*C. gouldii*, 95-100 % of sites; *M. ridei*, 71-91 % of sites; *M. orianae oceanensis*, 71-82 % of sites) were commonly recorded at converted pool sites between 2018 and 2020. This was a considerable increase (~20-30 %) in the occurrence of *M. orianae oceanensis* from 2017 (Gonsalves et al. 2017). This is likely a reflection of the timing of these more recent surveys, which were undertaken in autumn when *M. orianae oceanensis* is more abundant in Sydney as the species prepares to overwinter (Gonsalves and Law 2018) compared to summer surveys in 2017. Nevertheless, all three commonly recorded species at converted pools were also commonly 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 when the species was not detected in backyards/parks without pools. It is unclear whether non-detection of the species is due to site-level differences, with most backyards sampled in 2018 and 2019 unable to be sampled in 2020 due to COVID-19 restrictions. Instead, urban parks were sampled as an alternative to continue to provide context for converted and non-converted pools within the urban matrix. Sampling of urban parks and backyards in 2021 will help to identify whether any site-level differences may influence the non-detection of the species in backyards/parks in 2020.

Nightly bat activity recorded across the LGA comprised flight and feeding activity as revealed by the presence of feeding buzzes (Griffin et al. 1960), particularly at golf course dams. In 2020, feeding activity (no. of feeding buzzes per night) was quantified for the first time, but was too infrequently recorded to test for differences among habitat types. Notwithstanding, the number of feeding buzzes was greatest at golf course dams (22 buzzes) and sediment ponds/wetlands (6 buzzes), with low levels of feeding recorded at large waterways (2 buzzes) converted pools (1 buzz), non-converted pools, (1 buzz), creeks (1 buzz), and bushland (1 buzz). Between 2018 and 2020, activity was greatest at golf course dams (127-317 passes per night) and sediment ponds/wetlands (75-78 passes per night), with activity considerably lower in all other habitat types (6-42 passes per night). Activity at large waterways that were sampled for the first time in 2020 was intermediate (54 passes per night). Total activity at golf course dams significantly increased between 2018 and 2019, but was intermediate in 2020, whereas activity of most commonly recorded species increased in 2019 but remained stable in 2020, except *M. orianae oceanensis* which declined across all habitat types. Post-hoc investigations with golf course managers revealed no change in management occurred among years (e.g., increase in use of fertilisers). However, water levels in dams were significantly lower in 2018 and at capacity in 2019 and 2020 (Pers. Comm. C. Costello). 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 elsewhere in northern Sydney where bat activity was approximately four times greater (Basham et al. 2011). Nightly activity at creeks was lower than backyard/park sites (without or with converted or non-converted pools), though the difference was reduced for backyards/parks in 2020, perhaps due to site-level differences for this habitat type. 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 2-12 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). 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 *M. orianae oceanensis* was recorded at just 40 % of sites, albeit with fewer sites sampled this year (n=42, excluding large waterways that were not sampled previously). 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 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. Given 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 (unpublished data – Law et al.). However, declines of this nature were not evident for other commonly recorded bat taxa in 2020, suggesting a potential for a species-specific disturbance event. Adult females of this species are known to 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 spring/summer bushfires of 2019, 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. However, annual census of the population at two other maternity sites, including one that was also impacted by fire, revealed a 3 % decline in numbers of flying adults in 2020, though this is 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). Rather than impacting population size, it is possible that spring/summer bushfires in 2019 may have delayed the return of *M. orianae oceanensis* to Ku-ring-gai and other parts of Sydney (Gonsalves and Law 2018), resulting in LGA-wide reductions in autumn activity but stable trends at an overwintering roost site in winter. However, no monitoring data are available for other areas to confirm this.

Despite having a close association with waterways, *M. macropus* was rarely recorded on creeks between 2018 and 2020 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 (2017-2020) 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 2018 and 2019 and moderate levels in 2020. 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 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; n=3, 2020) and was less commonly detected at 17-33 % of sediment ponds/wetlands (n=6, 2018-2020), with activity more than an order of magnitude 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). In 2020, large waterways within the Port Jackson estuary were also sampled as part of this study and moderate levels (22 passes per night) of *M. macropus* activity were recorded.

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 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, areas with lower bushland and foliage projective cover. *Ozimops ridei* is a fastflying species suited to open spaces (Adams et al. 2009) and activity of the species was greatest at golf course dams, which were generally the largest pools sampled and also tended to have flowing water as a result of operating aeration pumps.

For other less commonly recorded species, CCA bi-plots highlighted associations for *Nyctophilus* spp., *M. australis* and *R. megaphyllus* with bushland cover on sandstone geology, often near drainage lines. The latter two species 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. 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. 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 seven 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 between 2018 and 2020. Four species (*C. gouldii*, *M. orianae oceanensis*, *M. ridei*, and *M. macropus*) contributed to most of the activity recorded at these two habitat types. 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 significantly high (Karlsson et al. 2010) and a pathway for uptake by bats has been established (Clarke-Wood et al. 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*. In the case of golf courses, we also suggest that invertebrate assemblages on dams should be sampled periodically as a means of assessing changes in prey abundance that may occur with changes to management practices (e.g., pest control for greens and fairways). 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).

Other than large waterways, all other habitat types had lower bat species richness and activity than golf course dams and sediment ponds/wetlands. Nevertheless, they still provide habitat for 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. Considering the productivity of some bushland embedded in urban areas with underlying shale soils, such patches 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 may respond to the provision of converted pools. However the species is yet to be recorded at converted pools after four years of monitoring (Gonsalves et al. 2017, 2018, 2019) and was rarely recorded at creeks. Activity of *M. macropus* was greatest over waterbodies with large pools, such as 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). Furthermore, >80 % of all converted pools did not have a water pump installed, providing still water that promoted the development of algae that covered parts of or the entire pool. 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, for most converted pools, increasing their size is not likely to be feasible and emergent aquatic vegetation can provide benefits for other fauna (e.g., invertebrates, frogs, fish). *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 in 2020 (Pers. Comm. C. Costello). Golf courses tended to have no light or very low levels of light around dams. If emergent aquatic vegetation was reduced at converted pools, reducing lighting may enhance the suitability of this habitat type for *M. macropus*. 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 in 2018, 2019 and 2020, 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

Four 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 increase in activity recorded at golf course dams in 2019 and 2020 relative to 2018. Similarly, research may be required to identify potential drivers of decline in the activity of *M. orianae oceanensis*.

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). We suggest that the current monitoring program should be expanded to include other LGAs within Sydney.

Converted pools provided value for bats in the LGA, with 3-4 taxa (*C. dwyeri*; *Nyctophilus* spp.; *R. megaphyllus* and *S. rueppellii*) recorded at these sites but not at non-converted pools or backyards/parks. Bats were also recorded feeding at these pools, 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.

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 four years of monitoring converted pools alongside other waterbodies and elements (backyards/parks without pools and bushland) in the urban matrix, we found patterns of total activity were fairly consistent among years, with the exception of golf course dams which had greater activity in 2019. However, we did identify 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 which will facilitate a long-term assessment of the value of converted pools and other habitat types to bats in the Ku-ring-gai LGA.

References

Adams, M.D., Law, B.S., and French, K.O. (2005). Effect of lights on activity levels of forest bats: increasing the efficiency of surveys and species identification. *Wildlife Research*, **32**, 173-182.

Adams, M.D., Law, B.S., and French, K.O. (2009). Vegetation structure influences the vertical stratification of open-and edge-space aerial-foraging bats in harvested forests. *Forest Ecology and Management*, **258**, 2090-2100.

Anderson, J., Law, B.S., and Tidemann, C. (2006). Stream use by the large-footed myotis *macropus* in relation to environmental variables in northern New South Wales. *Australian Mammalogy*, **28**, 15-26.

Asplet, K. (2016). Investigating the sensitivity of *Myotis macropus* (Large-footed Myotis) to heavy metal pollution in urban waterways, Sydney, New South Wales. Unpublished Honours Thesis. Australian Catholic University, North Sydney.

Basham, R., Law, B.S., and Banks, P.B. (2011). Microbats in a 'leafy' urban landscape: are they persisting, and what factors influence their presence? *Austral Ecology*, **36**, 663-678.

Bates, D., Maechler, M., Bolker, B. and Walker, S. (2015). lme4: Linear mixed-effects models using Eigen and S4. R package version 1.1-8.

Blakey, R.V., Law, B.S., Straka, T.M., Kingsford, R.T., and Milne, D.J. (2018). Importance of wetlands to bats on a dry continent: a review and meta-analysis. *Hystrix, the Italian Journal of Mammalogy*, **29**, 41-52.

Boonman, A.M., Boonman, M., Bretschneider, F., and van de Grind, W.A. (1998). Prey detection in trawling insectivorous bats: duckweed affects hunting behaviour in Daubenton's bat, *Myotis daubentonii*. *Behavioral Ecology and Sociobiology*, **44**, 99-107.

Churchill, S. (2009). Australian Bats, 2nd edition. Allen & Unwin, Crows Nest, NSW.

Clarke-Wood, B.K., Jenkins, K.M., Law, B.S., and Blakey, R.V. (2016). The ecological response of insectivorous bats to coastal lagoon degradation. *Biological Conservation*, **202**, 10-19.

Daniels, G.D., and Kirkpatrick, J.B. (2006). Does variation in garden characteristics influence the conservation of birds in suburbia? *Biological Conservation*, **133**, 326-335.

Davies, Z.G., Fuller, R.A., Loram, A., Irvine, K.N., Sims, V., and Gaston, K.J. (2009). A national scale inventory of resource provision for biodiversity within domestic gardens. *Biological Conservation*, **142**, 761-771.

Dwyer, P.D. (1963). The breeding biology of *Miniopterus schreibersi blepotis* (Termminck) (Chiroptera) in north-eastern NSW. *Australian Journal of Zoology*, **11**, 219–240.

Erickson, J.L., and West, S.D. (2002). The influence of regional climate and nightly weather conditions on activity patterns of insectivorous bats. *Acta Chiropterologica*, **4**, 17-24.

Fenton, M.B. (1990). The foraging behaviour and ecology of animal-eating bats. *Canadian Journal of Zoology*, **68**, 411-422.

Flache, L., Ekschmitt, K., Kierdorf, U., Czarnecki, S., Düring, R.A., and Encarnação, J.A. (2016). Reduction of metal exposure of Daubenton's bats (*Myotis daubentonii*) following remediation of pond sediment as evidenced by metal concentrations in hair. *Science of the Total Environment*, **547**, 182-189.

Frenckell, B.V., and Barclay, R.M. (1987). Bat activity over calm and turbulent water. *Canadian Journal of Zoology*, **65**, 219-222.

Fukui, D.A.I., Murakami, M., Nakano, S., and Aoi, T. (2006). Effect of emergent aquatic insects on bat foraging in a riparian forest. *Journal of Animal Ecology*, **75**, 1252-1258.

Garden, J., Mcalpine, C., Peterson, A.N.N., Jones, D., and Possingham, H. (2006). Review of the ecology of Australian urban fauna: a focus on spatially explicit processes. *Austral Ecology*, **31**, 126-148.

Goddard, M.A., Dougill, A.J., and Benton, T.G. (2010). Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology & Evolution*, **25**, 90-98.

Gonsalves, L., and Law, B. (2016). Distribution of the Large-footed Myotis (*Myotis macropus*) in waterways of the Blacktown Local Government Area. Report prepared for Blacktown City Council.

Gonsalves, L., and Law, B. (2017). Distribution and key foraging habitat of the Large-footed Myotis *macropus* in the highly modified Port Jackson estuary, Sydney, Australia: an overlooked, but vulnerable bat. *Australian Zoologist*, **38**, 629-642.

Gonsalves, L., and Law, B. (2018). Seasonal activity patterns of bats in North Sydney, New South Wales: implications for urban bat monitoring programs. *Australian Mammalogy*.

Gonsalves, L., Law, B, and Sife, J. (2017). Insectivorous bat activity over swimming pools retrofitted for wildlife. Report prepared for Ku-ring-gai Council.

Gonsalves, L., Hankin, C., Kerr, I., Law, B., and Sife, J. (2018). Insectivorous bat activity over swimming pools retrofitted for wildlife. Year 2: 2018. Report prepared for Ku-ring-gai Council.

Gonsalves, L., Lamb, S., Webb, C., Law, B., and Monamy, V. (2013a). Do mosquitoes influence bat activity in coastal habitats? *Wildlife Research*, **40**, 10-24.

Gonsalves, L., Law, B., Webb, C., and Monamy, V. (2013b). Foraging ranges of insectivorous bats shift relative to changes in mosquito abundance. *PLoS One*, **8**, e64081.

Griffin, D.R., Webster, F.A., and Michael, C.R. (1960). The echolocation of flying insects by bats. *Animal Behaviour*, **8**, 141–154.

Haddock, J.K., Threlfall, C.G., Law, B., and Hochuli, D.F. (2019). Light pollution at the urban forest edge negatively impacts insectivorous bats. *Biological Conservation*, **236**, 17-28.

Hourigan, C.L., Catterall, C.P., Jones, D., and Rhodes, M. (2008). Comparisons of harp trap and bat detector efficiency for surveying bats in an urban landscape. *Wildlife Research*, **35**, 768-774.

How, R., and Dell, J. (1993). Vertebrate fauna of the Perth metropolitan region: consequences of a modified environment. Pp. 28-47 in Urban Bush Management: Proceedings of a Seminar held at Gosnells, WA, 23 June 1992, edited by Australian Institute of Urban Studies, Greening Western Australia, Department of Conservation and Land Management, and The Tree Society. Australian Institute of Urban Studies, Perth.

How, R.A. and Dell, J. (1994). The zoogeographic significance of urban bushland remnants to reptiles in the Perth region, Western Australia. Pacific Conservation Biology, 1, 132-140.

How, R.A., and Dell, J. (2000). Ground vertebrate fauna of Perth's vegetation remnants: impact of 170 years of urbanization. *Pacific Conservation Biology*, **6**, 198-217.

Jones, G., Jacobs, D.S., Kunz, T.H., Willig, M.R., and Racey, P.A. (2009). Carpe noctem: the importance of bats as bioindicators. *Endangered Species Research*, **8**, 93-115.

Karlsson, K., Viklander, M., Scholes, L., and Revitt, M. (2010). Heavy metal concentrations and toxicity in water and sediment from stormwater ponds and sedimentation tanks. *Journal of Hazardous Materials*, **178**, 612-618.

Kirsten, I., and Klomp, N.I. (1998). Microchiroptera in urban, rural and forest areas of southern NSW. *Australasian Bat Society Newsletter*, **11**, 28-30.

Korine, C., Adams, R., Russo, D., Fisher-Phelps, M., and Jacobs, D. (2016). Bats and water: anthropogenic alterations threaten global bat populations. pp. 215-241 in *Bats in the Anthropocene: conservation of bats in a changing world*, edited by C. Voight and T. Kingston. Springer International Publishing.

Ku-ring-gai Council. (2016). Ku-ring-gai Biodiversity and Riparian Lands Study (version 5). Accessed on 29 May 2018 from: http://www.kmc.nsw.gov.au/files/assets/public/hptrim/information_management_-_publications_-_public_website_-_ku-ring-gai_council_website/environment_and_sustainability/environment_and_sustainability_-_community_programs/ku-ring-gai_biodiversity_and_riparian_lands_study_v5_for_final_principal_and_local_centre_dcp_support.pdf.

New, T.R., and Sands, D.P.A. (2002). Conservation concerns for butterflies in urban areas of Australia. *Journal of Insect Conservation*, **6**, 207-215.

O'Donnell, C.F. (2000). Influence of season, habitat, temperature, and invertebrate availability on nocturnal activity of the New Zealand long-tailed bat (*Chalinolobus tuberculatus*). *New Zealand Journal of Zoology*, **27**, 207-221.

Palmer, M.W. (1993). Putting things in even better order: the advantages of canonical correspondence analysis. *Ecology*, **74**, 2215-2230.

R Core Team (2014). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Smith, R.M., Gaston, K.J., Warren, P.H., and Thompson, K. (2006). Urban domestic gardens (VIII): environmental correlates of invertebrate abundance. *Biodiversity and Conservation*, **15**, 2515-2545.

Spoelstra, K., van Grunsven, R.H., Ramakers, J.J., Ferguson, K.B., Raap, T., Donners, M., Veenendaal, E.M., and Visser, M.E. (2017). Response of bats to light with different spectra: light-shy and agile bat presence is affected by white and green, but not red light. Proceedings of the Royal Society B, **284**, 20170075.

Stenhouse, R.N. (2004). Fragmentation and internal disturbance of native vegetation reserves in the Perth metropolitan area, Western Australia. *Landscape and Urban Planning*, **68**, 389-401.

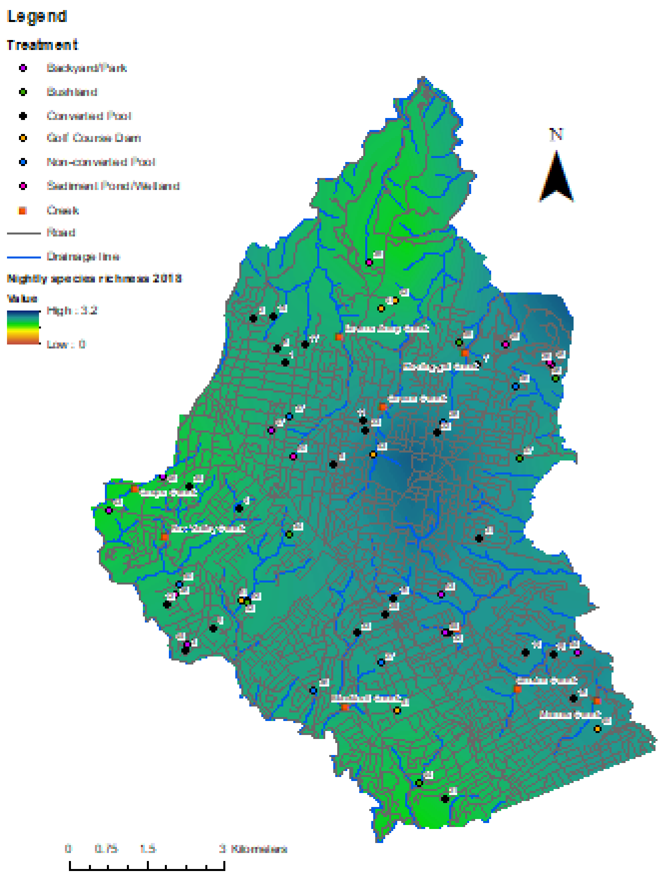
Tait, C.J., Daniels, C.B., and Hill, R.S. (2005). Changes in species assemblages within the Adelaide Metropolitan Area, Australia, 1836–2002. *Ecological Applications*, **15**, 346-359.

Threlfall, C.G., Law, B.S., and Banks, P.B. (2012). Sensitivity of insectivorous bats to urbanization: implications for suburban conservation planning. *Biological Conservation*, **146**, 41-52.

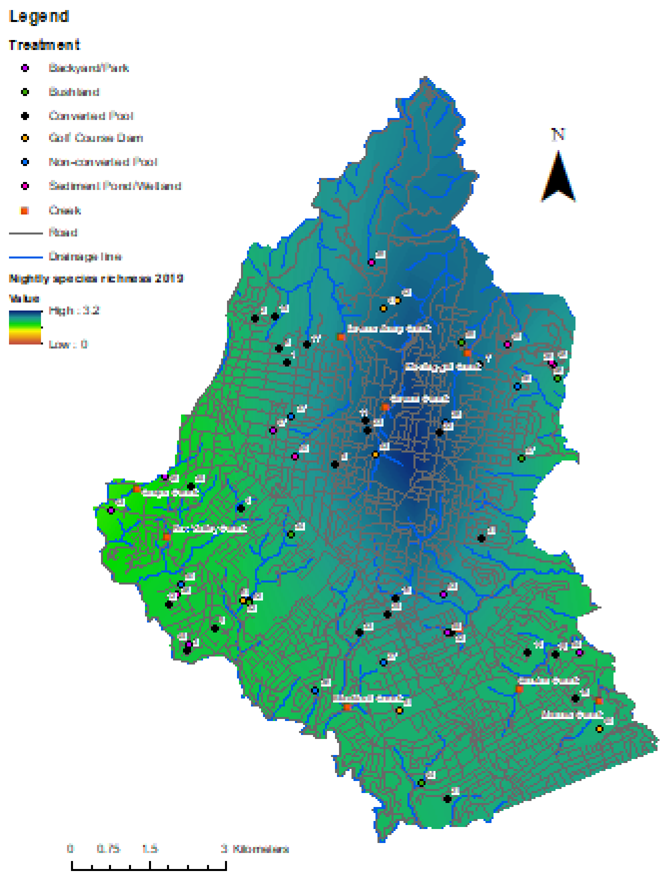
Threlfall, C.G., Law, B.S., Penman, T., and Banks, P.B. (2011). Ecological processes in urban landscapes: mechanisms influencing the distribution and activity of insectivorous bats. *Ecography*, **34**, 814-826.

White, A.W., and Burgin, S. (2004). Current status and future prospects of reptiles and frogs in Sydney's urban-impacted bushland reserves. Pp. 109-123 in Urban Wildlife: more than meets the eye, edited by D. Lunney and S. Burgin. Royal Zoological Society, Mosman.

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



a)



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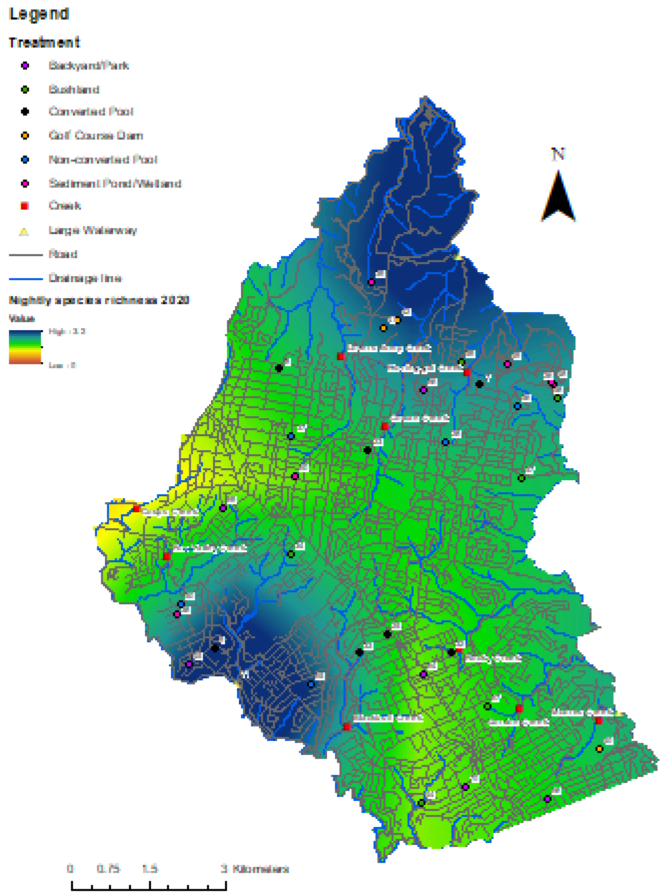
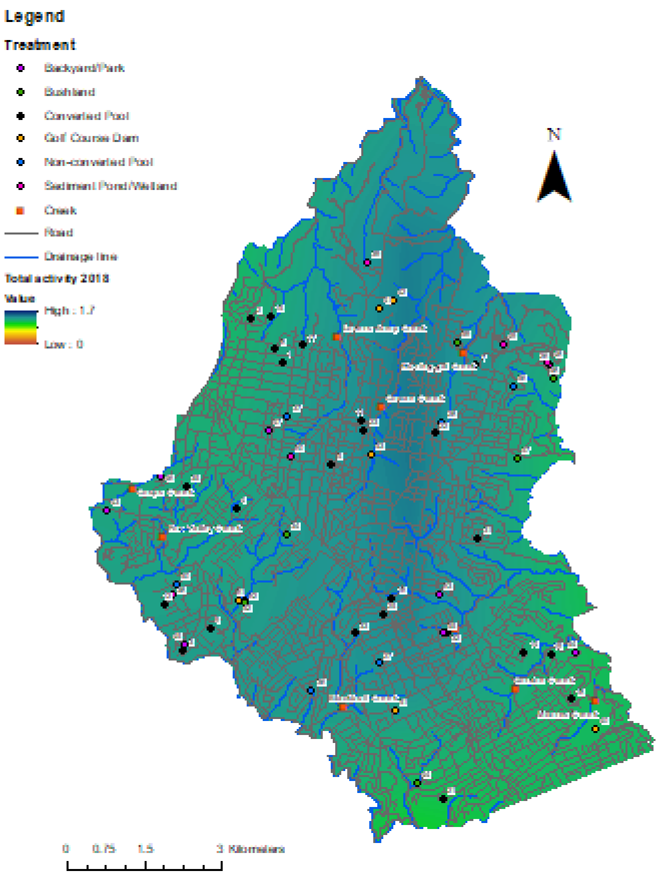
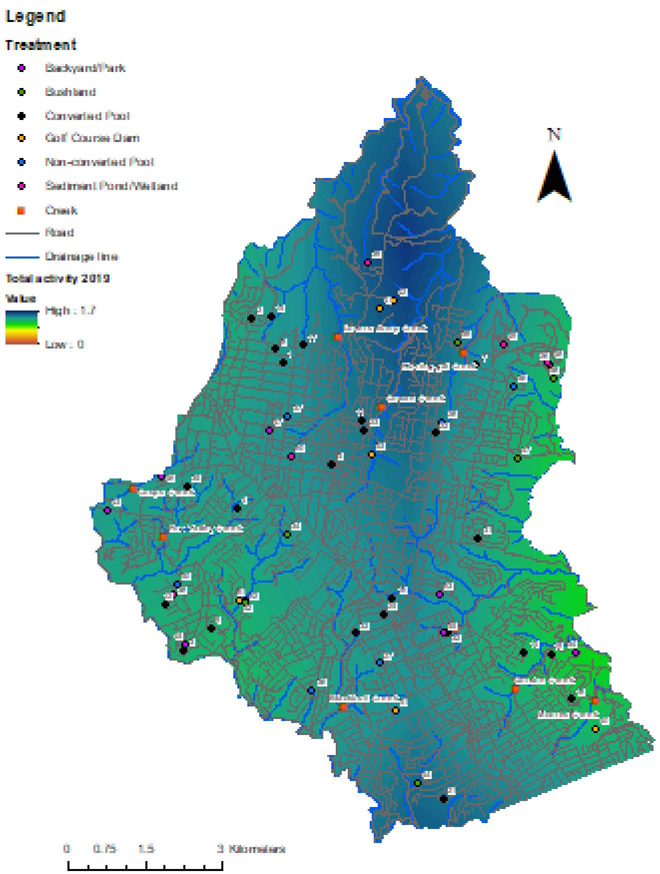


c)

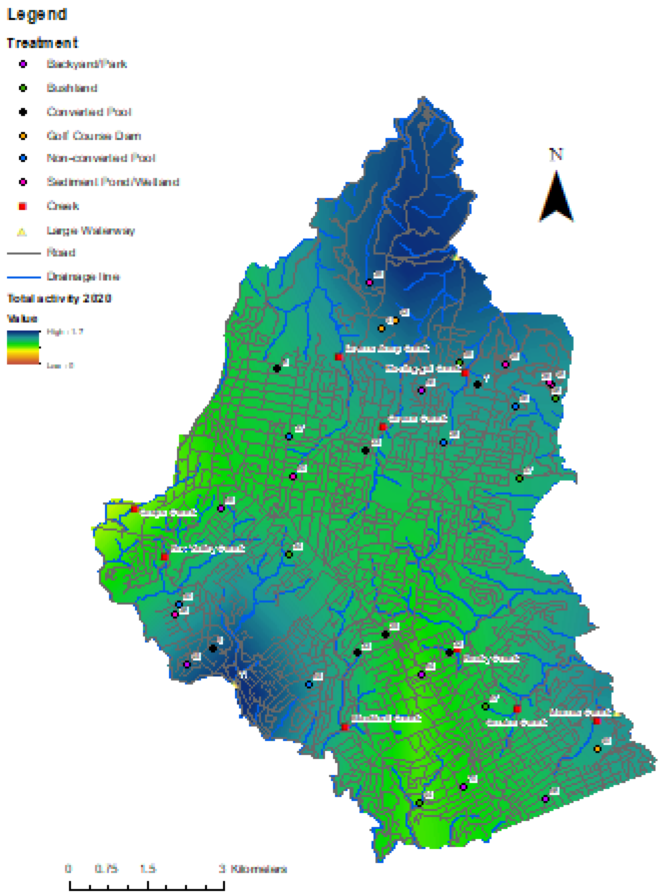
Appendix 2 – Night bat activity interpolation (IDW) map for the Ku-ring-gai LGA in: a) 2018, b) 2019 and c) 2020.



a)

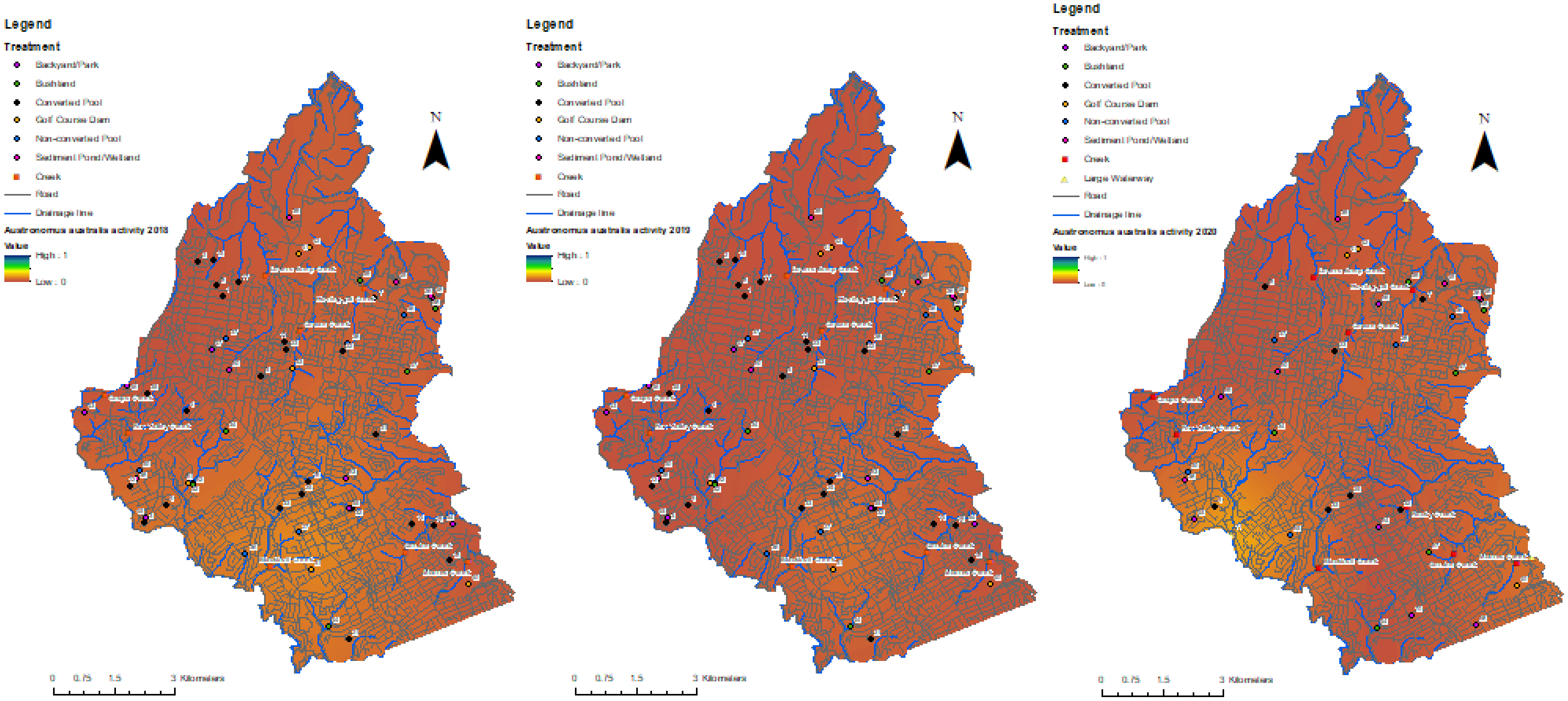


b)



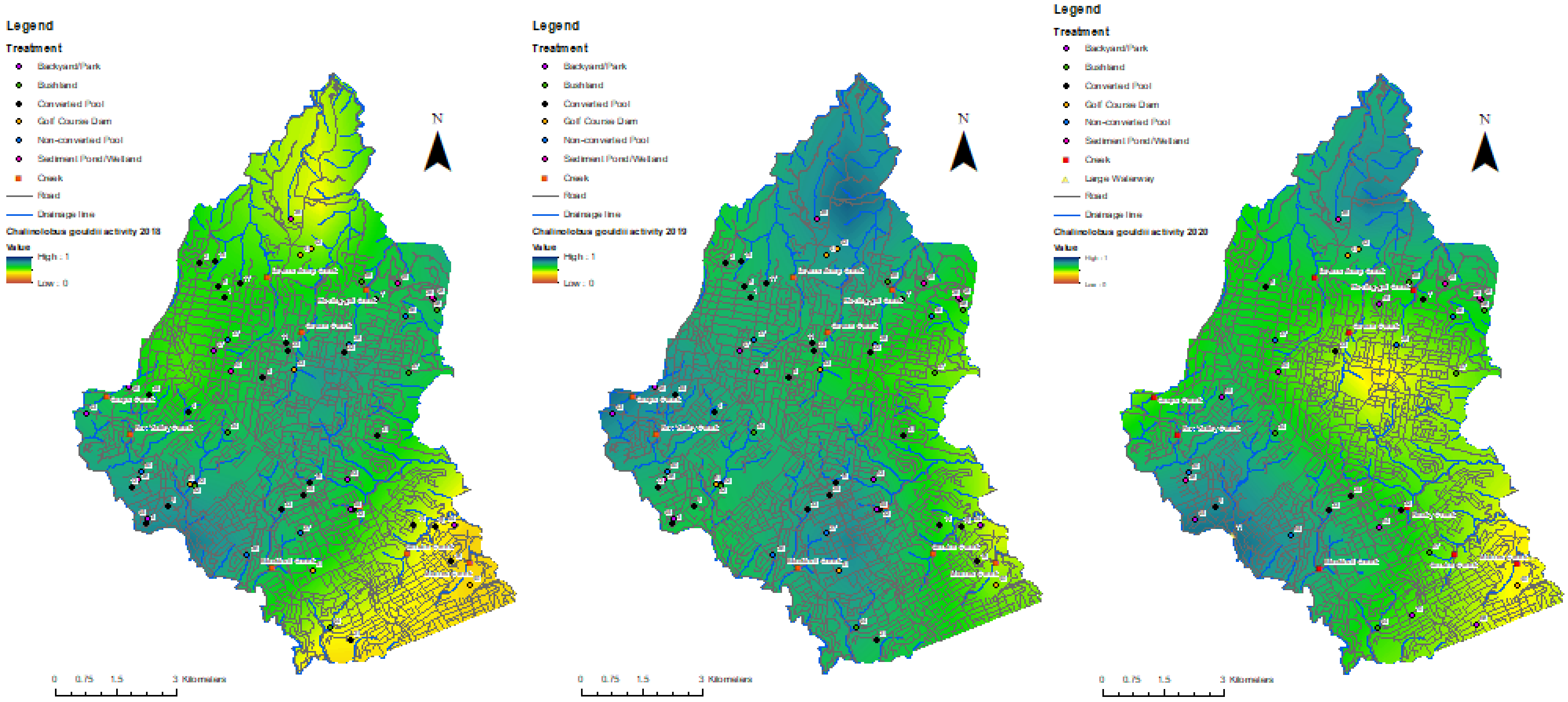
c)

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



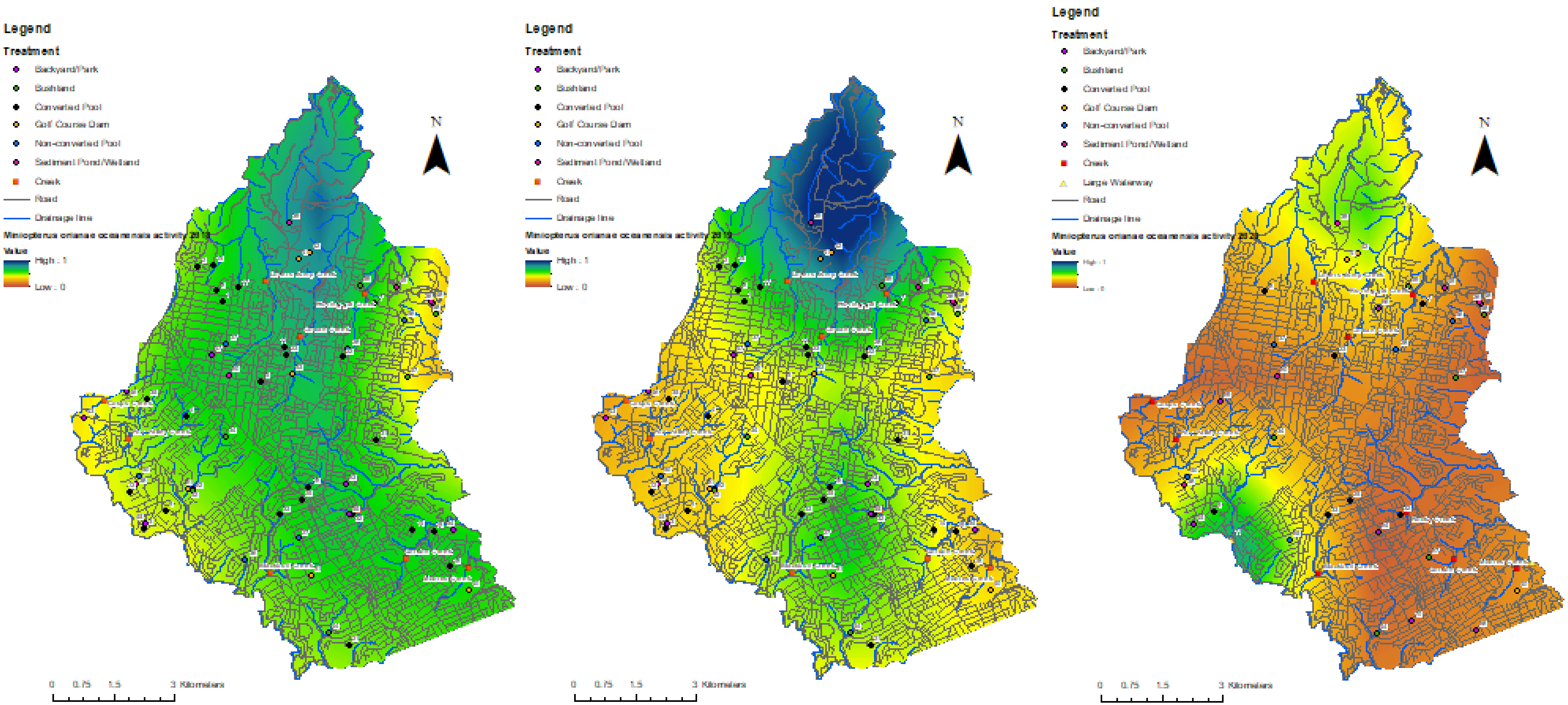
a) b) c)

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



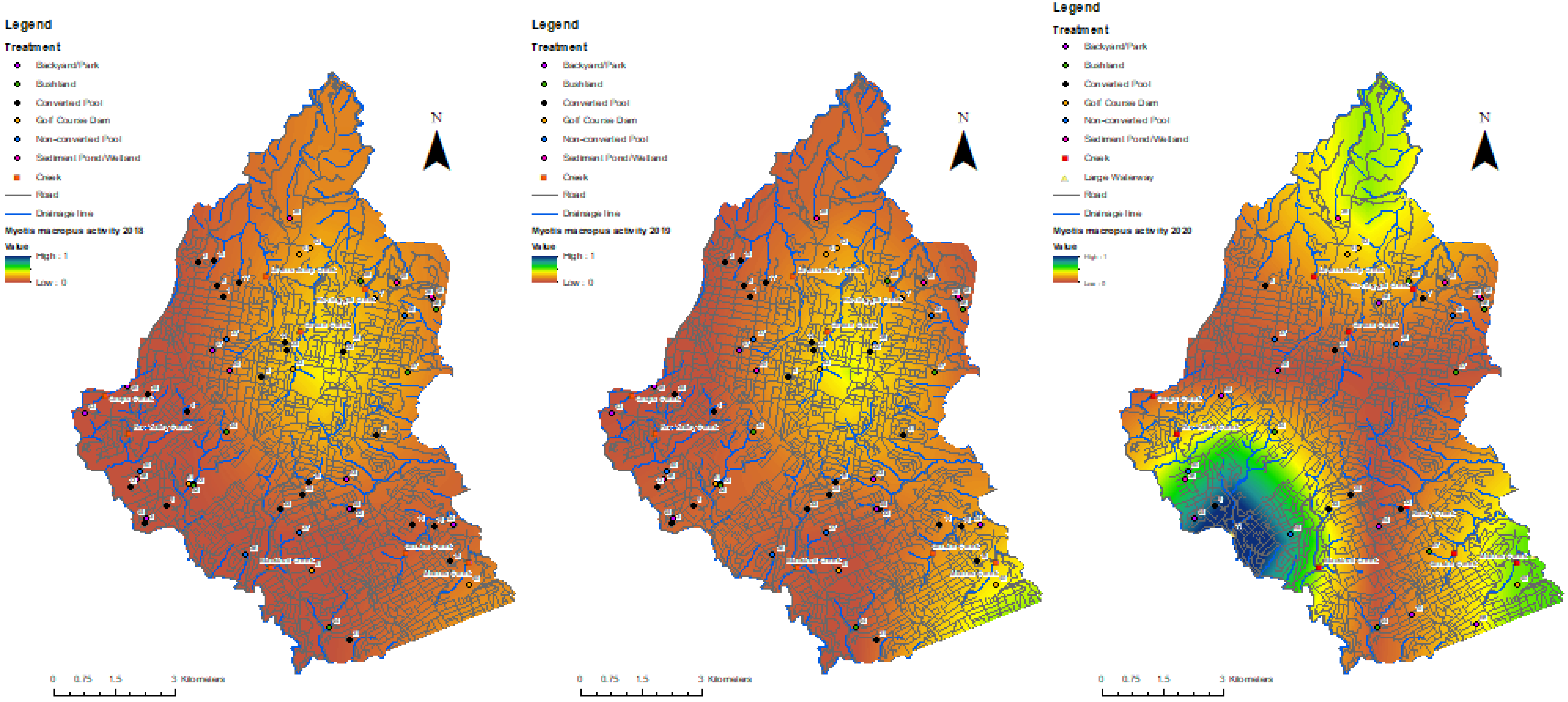
a) b) c)

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



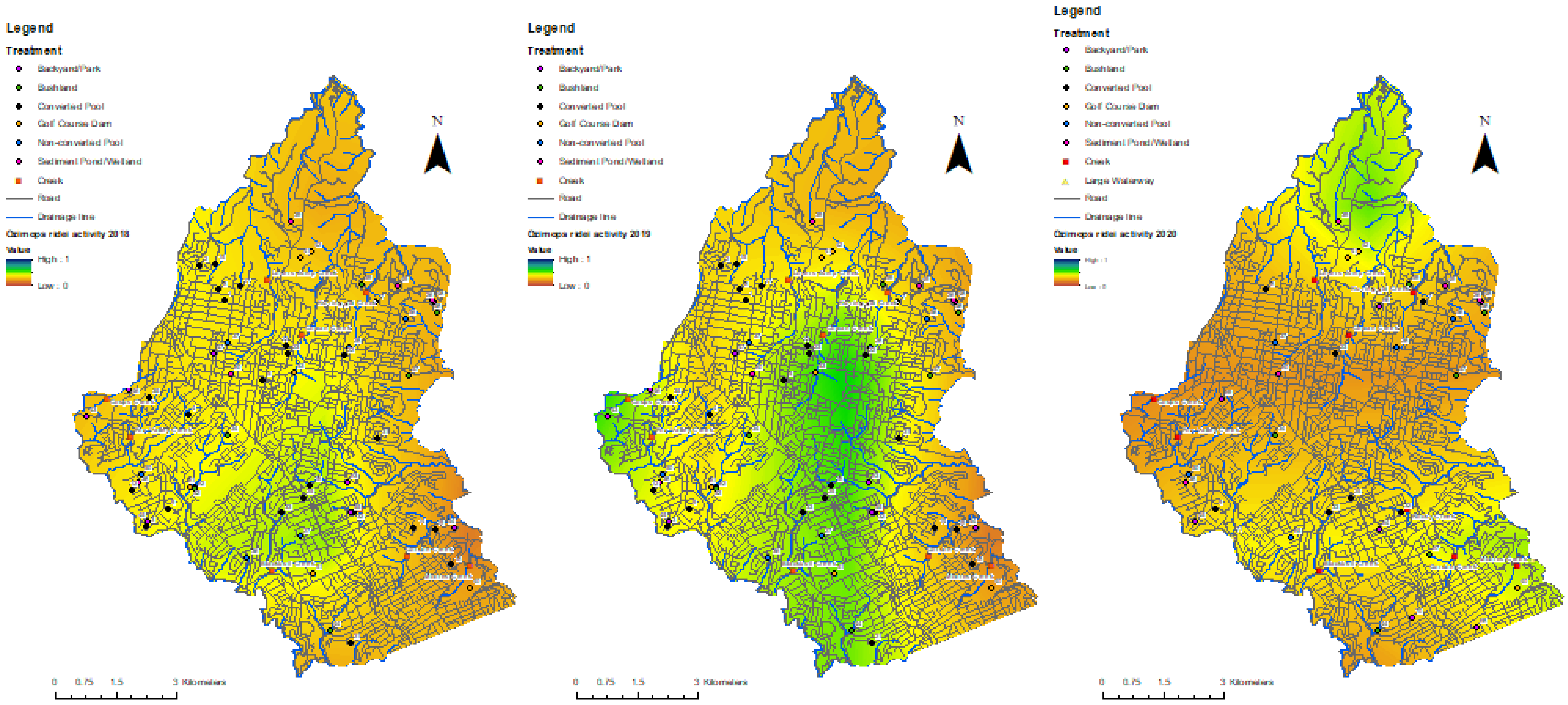
a) b) c)

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



a) b) c)

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



a) b) c)