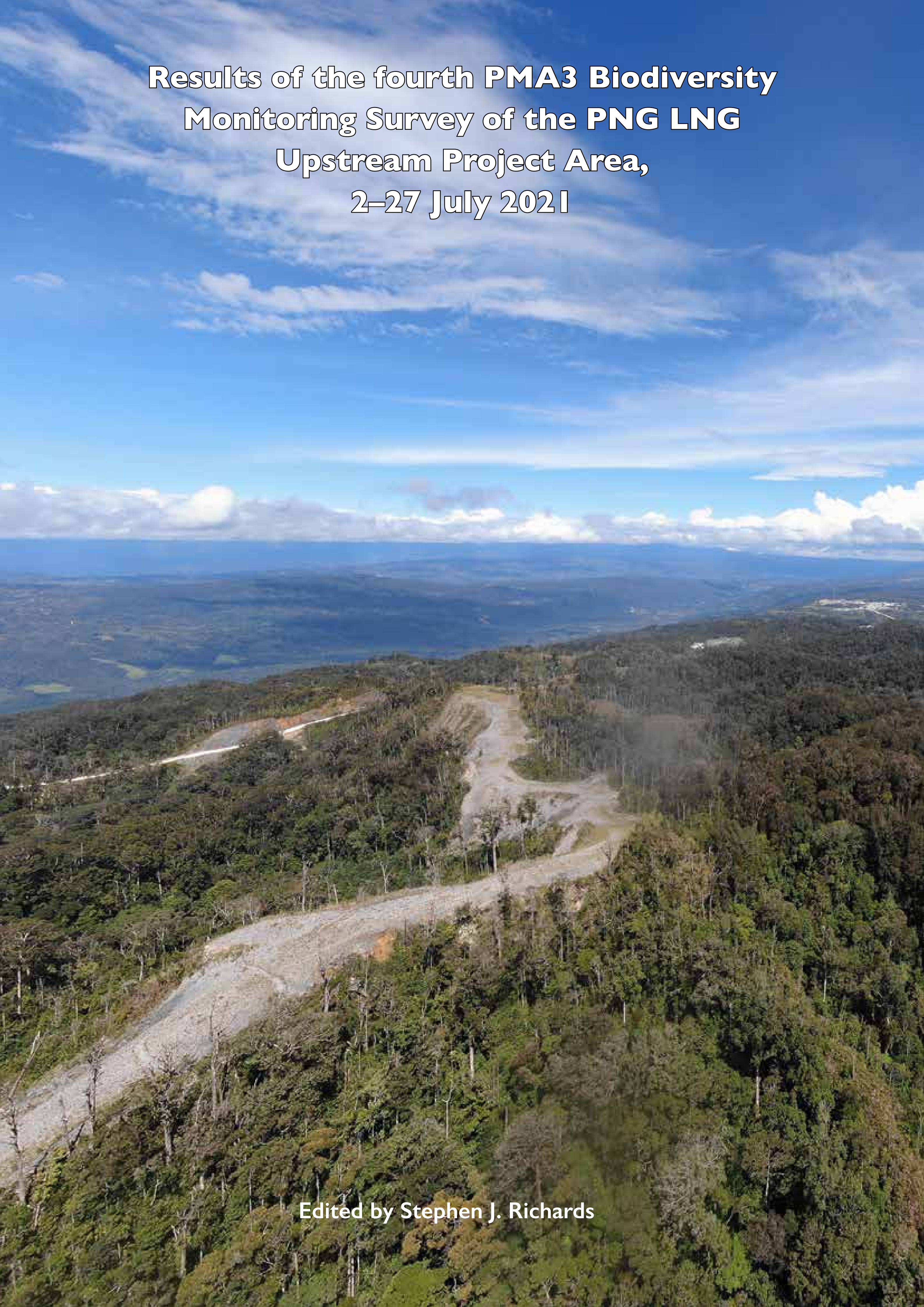
Results of the fourth PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 2–27 July 2021



Edited by Stephen J. Richards

ISBN: 978-0-6484933-7-2

Suggested citation: Richards, S.J. (Editor) 2022. Results of the fourth PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 2–27 July 2021. ExxonMobil PNG Limited. Port Moresby.

Front cover image: The spineline right-of-way (ROW)between Wellpad B and Wellpad C. Photograph by Jerry Albaniel.
Back cover image: Forest interior at Hides Ridge. Photo by D. Okena.

PNG LNG is operated by a subsidiary of ExxonMobil in co-venture with:

Results of the fourth PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 2–27 July 2021

Edited by Stephen J. Richards

A report to ExxonMobil PNG Limited from the 2021 PMA3 Biodiversity Monitoring Program

Table of Contents

Chapters

Participants

Kyle N. Armstrong (Mammals/Frogs)
Australian Centre for Evolutionary
Biology and Biodiversity
The University of Adelaide
South Australia; and
Specialised Zoological
Email: kyle.n.armstrong@gmail.com

Jasper Billy (Mammals)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea

Chris Dahl (Frogs)
Independent Researcher
PO Box 1076, Madang
Papua New Guinea
Email: cd.rokrok@gmail.com

Banak Gamui (Camera Trapping)
Biodiversity advisor
ExxonMobil PNG Limited
Rm: 2-15-A, ExxonMobil Haus,
Jacksons Parade
PO Box 118, Port Moresby, NCD,
Papua New Guinea
Email: Banak.Gamui@exxonmobil.com

Enock Kale (Mammals and field-based Scientific Team Leader)
Ecomate Management
P. O. Box 7146, Boroko, NCD,
Papua New Guinea
Email: enockkale@gmail.com

Elizah Nagombi (Frogs/Mammals)
Independent Researcher
Current address:
Wildlife Conservation Society,
P.O. Box 95, Kavieng, New Ireland, Papua New Guinea
Email: elizah_nagombi@hotmail.com

Daniel Okena (Mammals)
Independent Researcher
Email: dokena25@gmail.com

Luda Paul (Frogs)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea
Email: ludamansa@gmail.com

Frank Philip (Mammals)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea
Email: frankjurgenphilip@gmail.com

Stephen J. Richards (Team leader/Frogs)
Research Associate, Department of Herpetology
South Australian Museum
North Terrace, Adelaide, SA 5000
Australia
Email: richards.steve0@gmail.com

Steven Sau (Mammals)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea
Email: stevenwalimel@gmail.com

Salape Tulai (Camera Trapping)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea
Email: stulai.naturebarter@gmail.com

Iain A. Woxvold (Camera Trapping)
Museum Victoria Sciences Department
GPO Box 666, Melbourne,
VIC 3001 Australia
Email: iainwoxvold@gmail.com

Joachim Yalang (Mammals)
New Guinea Binatang Research Centre
P. O. Box 604, Madang
Papua New Guinea
Email: garanjyalang17@gmail.com

Acknowledgements

Phase 4 of the PMA3 biodiversity monitoring survey presented unique logistical challenges due to the COVID-19 pandemic. To successfully complete the field program the team relied not only on the considerable efforts of our ExxonMobil support team, but also on support provided by the dedicated team of Binatang Research Centre staff.

From ExxonMobil we are extremely grateful to Banak Gamui, Julia Hagoria, Kalyna Taule and Isaac Wama for their organisational skills and ongoing support during the survey. At Hides we are also extremely grateful to Ken Musante and Glen Mitravitz (PICs), Glen Burton and William Baur and their teams (Upstream Security), our CA Officers Paul Mohammed, Peter Kera, Douglas Simala and Anton Waim, VLOs Habe Dewi, Kaehape Heripa and Wai Handale, the drivers Jackson Lai, Hora Koi and Habe Papaya. At Moro we were greatly assisted by Alex Oki, Vicki Watch and Gaius Kave (PICs), Tom Paton and his team (Upstream Security), Dylan Soondrawu (Logistics-Ground Transport), our VLOs Samai Kei and Jack Obea, and drivers Pena Hunts, Thomas Aini, and Ken Naki.

From the Binatang Research Center we are most grateful to Francesca Dem, John Auga and Baibai Ministai, who provided a range of support services in the field and in Madang that were vital to the success of the field program.

The 2021 PMA3 biodiversity survey team. Photo by D. Okena.

Acronyms and abbreviations

asl	Above sea level
BAA	Biodiversity Assessment Area
CEPA	Conservation and Environment Protection Authority
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
DD	Data Deficient (IUCN threat category)
EIS	Environmental Impact Statement
EN	Endangered (IUCN threat category)
GLMM	Generalised Linear Mixed Model – a statistical test
GPS	Global Positioning System
IFC	International Finance Corporation
IUCN	International Union for the Conservation of Nature
km	Kilometers
LC	Least Concern (IUCN threat category)
LNG	Liquefied Natural Gas
m	meters
mm	millimeters
NT	Near Threatened (IUCN threat category)
PNG	Papua New Guinea
Project	PNG LNG Project
RAI	Relative abundance index
ROW	The pipeline Right of Way
sp.	Abbrev. 'species' (singular)
spp.	Abbrev. 'species' (plural)

Glossary of Technical Terms

Central cordillera	Refers to the central mountainous spine of New Guinea that runs from the eastern edge of the Vogelkop Peninsula in Indonesian New Guinea to the eastern tip of mainland PNG.
Community structure	The taxonomic composition of a community; species assemblage.
Conservation listed species	Includes: (1) species listed under the IUCN Red List as threatened (Critically Endangered, Endangered or Vulnerable), Near Threatened or Data Deficient; (2) species listed as Protected under the PNG *Fauna (Protection and Control) Act 1966*; (3) species listed under CITES Appendix I or II.
Diversity	In its broadest sense the concept of biological diversity can refer to multiple organizational levels including (but not limited to) genes, variants and subspecies, species, and ecosystems. In this report the term 'diversity' is restricted to the meaning 'numbers of species' (the most common definition) except where other forms of diversity are also being discussed, when the specific term 'Species Richness' is used.
Endemic	Belonging exclusively or confined to a particular place.
New species	A species new to science, discovered for the first time during the PMA3 surveys
Protected	Species listed as Protected under the Papua New Guinea *Fauna (Protection and Control) Act 1966*.
Restricted range	Species which have a total historical breeding range of less than 50,000 km^2.
Taxa	Plural of taxon; a systematic division (e.g. more than one species, genera, etc.).
Taxonomic	Taxonomy is the science of identifying, naming and classifying living organisms.
Undescribed species	A species that has not yet been formally named. It may be a new species or it may be known previously from other locations.

Kalyna Taule checking a camera trap

Background and aims

The Upstream Project Area of the Papua New Guinea Liquefied Natural Gas (PNG LNG) Project supports considerable biodiversity values. These were summarised in ExxonMobil PNG Limited's (EMPNG) Biodiversity Strategy as (i) extensive intact forest, (ii) high floristic diversity, (iii) high faunal diversity, (iv) endemic species, (v) unique assemblages of species, (vi) species of conservation concern, and (vii) biodiversity of importance to local communities for resource use and cultural and spiritual purposes.

To evaluate the success of its commitment to safeguarding these biodiversity values, and to determine whether the Project is successfully meeting the four major objectives of the Biodiversity Strategy – *Maintain the intactness of the Upstream area as a whole*; *Conserve the priority ecosystems*; *Protect focal habitats*; and *Account for residual impacts* (EMPNG PNG LNG Biodiversity Strategy; publically available on the PNG LNG Website: https://pnglng.com/media/PNG-LNG-Media/Files/Environment/Biodiversity%20strategy/PGGP-EH-SSZZZ-000003_Biodiversity-Strategy_Rev4.pdf) – EMPNG has developed a series of four Programmed Monitoring Activities (PMAs). One of these, Programmed Monitoring Activity 3 (PMA3), provides high-quality information on trends in species diversity and abundance in the Upstream area of the PNG LNG Project in order to detect any changes that may be associated with the development of Project infrastructure.

PMA3 conducts rapid, standardised biodiversity surveys to collect quantitative, repeatable data on species presence, relative abundance and trends in species diversity in two Biodiversity Assessment Areas (BAAs) in areas affected by the PNG LNG Project: the first at Hides Ridge (BAA 1), and the second on the Agogo Range near Moro (BAA 2). The first phase of the PMA3 biodiversity survey program was conducted during June–July 2015, the second phase during May 2017, and the third phase during August 2019. Results of these surveys were presented to EMPNG and subsequently published in three public documents (Richards 2017, 2019, 2021). Those reports found limited evidence for impacts of the linear infrastructure corridor on a suite of faunal groups and presented a series of recommendations for improving the PMA3 monitoring program to ensure that it best supports EMPNG's goal to safeguard biodiversity values in the Upstream Project Area.

The PMA3 monitoring program is scheduled to be conducted biennially. This document reports the results of the fourth phase of PMA3 biodiversity monitoring conducted during July 2021 and compares them with data on species diversity and trends obtained between 2015 and 2019 to identify and interpret any trends in species presence, abundance and diversity in the vicinity of Project infrastructure.

Survey dates

2nd to 27th July 2021 (camera traps deployed 4th July to 27th November 2021).

Brief description of the survey area

Detailed descriptions of environments in the Upstream Project Area are presented in the Project EIS, and the region's biodiversity values are summarised further in the EMPNG Biodiversity Strategy.

Extensive forest cover remains within both BAAs, and gradients in vegetation composition and structure with elevational change are evident. Both BAAs lie within the high-rainfall belt that extends across the southern slopes of PNG's central cordillera, and annual rainfall totals in excess of 4,000 mm with limited seasonality ('continuously heavy'; McAlpine et al. 1983) are typical. A comprehensive description of the local environments in BAA 1 and BAA 2, including forest structure, classification, and illustrations of forest types, is presented in Richards (2017).

The locations of both BAAs are shown in Figure 1.

BAA 1: 02–15 July 2021.

BAA 1 was established on Hides Ridge in Hela Province. It covers elevations between 2,100 and 2,750 m above sea level (asl), and was divided into two elevational bands, with three survey transects located at 2,100–2,400 m asl in the area between Wellpad C and Wellpad D, and three transects at 2,660–2,780 m asl located between Wellpads E and G (Figures 2–4).

BAA 2: 16–27 July 2021.

BAA 2 is located on the Agogo Range near Moro in Southern Highlands Province. Two survey transects were established at elevations of 950–1,080 m asl in the area west of Arakubi Quarry and east of the pipeline ROW, and three survey transects at elevations of 1,340–1,410 m asl in the vicinity of KP107 (Figures 5–7).

Survey approach

Surveys for frogs, non-volant mammals (rodents) and bats were conducted on the 11 permanent transects established during the 2015 PMA3 survey (Figures 2–7): six transects established in BAA 1 along the Hides Ridge access road and pipeline ROW (Figures 2–4), and five permanent transects established in BAA 2 along the pipeline ROW at KP107 (Figures 5–6) and adjacent to the Arakubi Quarry (Figures 5, 7). Each of these 11 transects extended for 220–250 m into the forest and were approximately perpendicular to the ROW or forest edge. Coordinates for all transects are presented in Appendix 1. In addition, camera trapping surveys were undertaken in the same elevational bands in each of BAA 1 and BAA 2 but the activities were carried out at some distance from the transects to avoid regular disturbance of camera trapped areas. Several camera locations were adjusted after the 2017 and 2019 surveys to avoid human disturbance. Locations of camera traps in each sampling year are illustrated in Figures 3–4 (BAA 1) and 6–7 (BAA 2).

A detailed rationale for the use of permanent transects to detect potential impacts of Project activities on species presence and trends is presented in Richards (2017). Perpendicular alignment of transects with respect to linear infrastructure samples a gradient of potential disturbance that is greatest at the forest edge and progressively less so with increasing distance into the forest. The impacts of 'edge effects' on most groups of organisms, including those associated with greater light and wind penetration and dust and noise pollution, are likely to attenuate rapidly and the 220–250 m transects should extend beyond any major impacts.

Survey modifications for 2021 PMA3 monitoring program

Several modifications were made to the PMA3 survey program in response to results of the 2019 survey or to the impacts of COVID-19.

1. After logistical constraints delayed the 2019 survey until August–September (*vs* May–July for the 2015 & 2017 surveys), the 2021 survey was conducted during July to reduce the potential influence of seasonality on survey results.

2. Changes to upstream logistics and team personnel necessitated by the COVID-19 pandemic constrained the ability of the team to undertake night work so Visual and Audio Encounter Surveys for frogs along transects and at Wellpad D were not undertaken during the 2021 survey.

3. Acoustic monitoring of frogs and bats utilised new, lighter recording units, which made equipment deployment easier without compromising recording quality.

4. Due to Covid-19 restrictions fieldwork was conducted by a fully PNG-based team. Remote coordination between the field team and Australia-based senior contract specialists was established via phone, zoom, and WhatsApp for daily reporting, monitoring of progress and troubleshooting to ensure the smooth flow of field work.

Major results

A summary of the major results is presented below.

Taxon accounts

Frogs

A total of 35 species of frogs was documented along permanent transects that run perpendicular to infrastructure clearings in BAA 1 on Hides Ridge and BAA 2 on the Agogo Range near Moro. Five species that were detected during 2015, 2017 and/or 2019 were not detected by the Acoustic Recorders during 2021. Four of these were rarely heard calling during previous surveys so this result was not unexpected because Visual and Audio Encounter Surveys were not conducted during 2021 due to logistical constraints imposed by COVID-19.

Three additional undescribed frog species documented during the PMA3 surveys have now been formally described in the scientific literature.

Analysis of data from the Acoustic Recorders found no consistent evidence for shifts in species diversity or composition with increasing distance from infrastructure clearings across the four survey years. Thus, to date, establishment of the linear infrastructure clearings in BAA 1 on the Hides spine-line and in BAA 2 on the Agogo Range near Moro has had no detectable impacts on local frog populations.

Camera traps

To date, more than 100 vertebrate species have been documented by camera traps during the PMA3 surveys. Results of the 2021 survey include two species newly reported from the BAAs – Masked Ring-tailed Possum (*Pseudochirulus larvatus*) at BAA 1 and Torresian Striped Possum (*Dactylopsila trivirgata*) at BAA 2. Biennial data are provided for seven IUCN listed species, including: four Threatened species – the Eastern Long-beaked Echidna (*Zaglossus bartoni*), Pademelon (*Thylogale* sp.), Ifola (*Dendrolagus notatus*) and Goodfellow's Tree Kangaroo (*D. goodfellowi*); two Near Threatened species – New Guinea Quoll (*Dasyurus albopunctatus*) and Small Dorcopsis (*Dorcopsulus vanheurni*); and one Data Deficient species – Woolley's Three-striped Dasyure (*Myoictis leucura*). The Small Dorcopsis was the most frequently camera trapped of all species with 1,126 photographic events recorded in 2021.

Compared to 2017 estimates, significant declines in activity rates were observed in 2021 for four mammal and six bird species at individual sites. In addition, naïve occupancy measures (proportion of cameras detecting a species) fell sharply for four IUCN listed mammals – the Eastern Long-beaked Echidna and Goodfellow's Tree Kangaroo at Arakubi, and Woolley's Three-striped Dasyure and the Pademelon at Arakubi and KP107. Conversely, an increase in population estimate was recorded for two IUCN listed mammals – the Eastern Long-beaked Echidna at Hides High and the Ifola at KP107.

The rate of daily incursions by humans and/or dogs decreased at all sites compared with 2019 levels. At BAA 1, 2021 incursion rates were also lower than those recorded there in 2017. By contrast, at the BAA 2 sites, both the 2019 and the 2021 incursion rates were higher than those recorded there in 2017. At Arakubi, where the decline in records of IUCN listed species has been strongest, most incursions have been recorded closer to the PNG LNG pipeline ROW than to the Arakubi Quarry and its access road (2019 – 13/16 incursion days; 2021 – 12/17 incursion days), suggesting that most hunting parties are accessing the area via the pipeline route.

Animal activity rate was correlated with distance from infrastructure in 16 species at one or more sites. Edge avoidance patterns were demonstrated by 10 species, being strongest at BAA 2 sites for Raffray's Bandicoot (*Peroryctes raffrayana*), Small Dorcopsis, Collared Brushturkey (*Talegalla jobiensis*), New Guinea Scrubfowl (*Megapodius decollatus*), Pheasant Pigeon (*Otidiphaps nobilis*) and Russet-tailed Thrush (*Zoothera heinei*). Seven species displayed reverse-pattern edge effects, with higher rates of activity nearer to the forest edge. Reverse edge effects were most common at BAA 1 and were the only patterns observed at the Hides High site. Causal factors for these counter-intuitive results for forest-interior species are unclear but are likely to be environmental rather than anthropogenic; candidate parameters include terrain effects (steepness), understorey density and predator avoidance.

Small non-volant (non-flying) mammals

The total number of novel captures of small rodents on the 2021 survey was greater than on all previous surveys but no rodent species new to the study area were detected during the 2021 trapping program.

Statistical tests indicated that Species Richness and the total number of captures were not significantly different amongst the categories defining distance from the ROW, elevation and survey year. Overall patterns in the number of captures were driven by four relatively common species: *Paramelomys* sp. cf. *mollis* AC, *Paramelomys* sp. cf. *rubex* A, *Rattus* sp. cf. *niobe* B and *Rattus* sp. cf. *niobe* D.

No invasive species of rodent was detected in 2021 despite the high capture rate. This is significant because an increasing population of invasive rodent species is likely to be the initial and most conspicuous indication of an edge effect that has the potential to affect the native rodent assemblage.

Together these results indicate that the forest adjacent to the ROW has so far retained its value for a diverse community of rodent species.

Bats

A total of 21 species was detected in the acoustic recordings, which is equivalent to previous surveys (2015: 21 spp.; 2017: 23 spp.; 2019: 19 spp.). Based on combined captures and acoustic recordings from the 2015–2021 surveys (no trapping was conducted in 2021), a total of 27 bat species has now been documented in the PMA3 study area.

No new species for the study area were encountered in 2021, but a growing understanding of variation in the call types of some species that emit low frequency echolocation helped to refine the species list. This led to the removal of the Greater Northern Free-tailed Bat *Chaerephon jobensis* from the list of species present, and the reallocation of some past records of *C. jobensis* and the Bare-rumped Sheath-tailed Bat *Saccolaimus saccolaimus* to an unidentified species of free-tailed bat *Otomops* sp.

No species classified in a threatened category by the IUCN have yet been recorded, but both species of free-tailed bat *Otomops* sp. that could be present are listed as Data Deficient. In addition, two putative new species (*Kerivoula* sp., *Nyctophilus* sp.) that were captured on Hides ridge at c. 2,700 m in 2017 remain undescribed. Additional voucher material is required to facilitate descriptions of these species. A third putative new species remains known only from single echolocation traces (unique call type 172 sCF) at transect M5 in 2015, and at KP 87 adjacent Lake Kutubu in 2017.

The bat communities within each of the BAAs have not changed significantly over the 7-year monitoring period—the communities in 2021 are equivalent to those documented in 2015 and subsequent years. The lack of spatial or temporal shifts in bat diversity or composition in either BAA since 2015 suggests that the forest adjacent to the ROW has so far retained its value for a diverse community of bats.

Table 1. Number of species documented during the 2021 PMA3 Surveys, number estimated to be new to science and/or undescribed, and the number of species holding an IUCN threat classification above Least Concern. Numbers are based on 2021 data, but all undescribed taxa have been recorded on previous surveys.

	Frogs	Camera traps (birds and mammals)	Non-volant Mammals*	Bats	TOTALS
Total Species	35	82+	11	21	**149**
Undescribed Species	10	1	7	2	**20**
IUCN Species	0	7	0	0	**7**

Threats

Two potential ongoing threats to biodiversity values in BAA 1 and BAA 2 were identified during the 2015 survey (apart from risks of mortality to dispersing animals from traffic) and they remained of concern in 2021. These were 1) decreasing habitat quality adjacent to the ROW due to edge effects (e.g. Andrews et al. 2015) and 2) improved access to the forest by humans (for hunting and gardening) and by invasive species, both native and exotic. These and other potential threats are summarised below.

Edge effects

The frog, rodent and bat studies found no consistent evidence for deleterious impacts of edge effects. However, the camera trap study continued to reveal patterns of edge avoidance for some species including Raffray's Bandicoot, Small Dorcopsis, Collared Brushturkey, New Guinea Scrubfowl, Pheasant Pigeon and Russet-tailed Thrush, particularly at the BAA 2 sites. These species may avoid near-edge environments at these sites due to the presence of degraded forest near infrastructure (particularly at Arakubi) and/or an aversion to frequent human activity along roads and the pipeline ROW. By contrast, camera trapping has revealed little evidence that terrestrial mammal or bird species avoid forest edge at BAA 1, suggesting that to date forest quality has not deteriorated at the forest edge.

Hunting, gardening, and predation pressure by dogs

The improved accessibility into formerly remote areas of forest following construction of the linear ROW infrastructure and associated roads has led to an increase in both direct hunting pressure by local people and predation by dogs.

Compared to 2017 data, camera trapping has recorded higher rates of forest incursions by humans and/or dogs at both BAA 2 sites in 2019 and 2021. This corresponds with a sharp decline in records for three IUCN Threatened species – the Eastern Long-beaked Echidna, Pademelon and Goodfellow's Tree Kangaroo – all of which are susceptible to hunting. Importantly, at the Arakubi site, most incursions have been recorded closer to PNG LNG infrastructure than to the Santos/OSL operated Arakubi Quarry, suggesting that most hunters and dogs are entering the Arakubi forest via Project infrastructure. A link cannot be proved, but further sampling is warranted to understand the local status of these high value species and patterns of forest use by local residents and their dogs.

In contrast to the patterns observed at BAA 2, the rate of human and/or dog incursions recorded by camera trapping at BAA 1 has declined steadily across all sampling years (2017–2021). Hunting pressure is difficult to quantify. However, this decline in the measure of hunting activity at BAA 1 was matched in 2021 by an increase in the naïve occupancy of two IUCN Threatened species at Hides High – the Eastern Long-beaked Echidna and Ifola – and the first record of Eastern Long-beaked Echidna at Hides Low since camera trap monitoring began. Although these trends are encouraging, they remain unproved, and the following conclusions from the 2019 monitoring study remain relevant: "a number of lines of evidence indicate that Project operations have resulted in a sustained increase in hunting activity and the harvesting of other resources from Hides Ridge. Direct observations and conversations with Project employees and local residents indicate that: (1) local residents access Hides Ridge more often than they did prior to Project construction; (2) on arriving at Hides Ridge they use the road and pipeline ROW to access broader areas of forest than were covered prior to construction; (3) Project construction and operations have diminished spiritual beliefs that previously contributed to a low rate of visitation; (4) a growing number of shelters and gardens are observed along the Project road, providing a base for hunting activities carried out over multiple days; (5) Project security staff hunt along the ridge (*kapul* are frequently taken), particularly when they stay overnight at wellpads, and carry bushmeat back to be consumed at home or distributed among *wantoks*; (6) security staff use Project vehicles to help unemployed *wantoks* carry timber cut for construction purposes (*Nothofagus* being particularly highly prized)." In response to this EMPNG's Montane Program is conducting ongoing educational awareness and has initiated a hunting survey.

Dieback

Observations during the first three survey monitoring years revealed that, in some areas of BAA 1, canopy trees along the edge of linear clearings were becoming increasingly stressed and, in many cases, dying. This was particularly evident for *Nothofagus* trees along the eastern (lower) half of Hides Ridge. Due to changes in team personnel, limited observations on dieback were obtained during 2021. More rigorous documentation of dieback along Hides Ridge in BAA 1 should be considered in future.

Removal of trees along linear infrastructure

Removal of trees from on or immediately adjacent to transects was not observed during the 2021 survey. However, expansion of gardens in BAA 1 involving the removal of numerous trees adjacent to Project linear infrastructure was noted and is likely to have localised impacts on biodiversity values by removing forest cover.

Exotic rodents

No invasive *Rattus* species were encountered in 2021 but invasive rodent populations are probably established in settlements around HGCP and Moro. ISOS has an existing rodent control program but this does not generate data on numbers of rodents killed so there is no information available for these areas that would allow an assessment of the risk of invasive rodents moving into the BAA study areas.

Major conclusions

1. Results of the 2021 PMA3 survey indicate that both BAAs retain high biodiversity values for all surveyed taxa, with both areas continuing to support rare, conservation listed, restricted range and hunting-sensitive species.

2. Notable declines have been recorded for three hunting-sensitive IUCN Threatened species at BAA 2 over the last two sampling years. Following a sharp decline in records of the IUCN Vulnerable Eastern Long-beaked Echidna and Pademelon at Arakubi in 2019, neither species was again recorded there in 2021. In 2021, novel declines were also recorded for the IUCN Endangered Goodfellow's Tree Kangaroo at Arakubi (first year with no records) and for the Pademelon at KP107 (sharp decline to <10% of previous annual records). By contrast, at Hides High in BAA 1, naïve occupancy of the Eastern Long-beaked Echidna increased sharply in 2021 and that of the Ifola has risen steadily from one camera (5%) in 2017 to three cameras (16%) in 2021.

3. Edge avoidance by multiple species was clearly demonstrated in BAA 2. By contrast, reverse-pattern edge effects were commonly observed in BAA 1. These results are counter-intuitive for interior forest species and causal factors are likely to be environmental rather than anthropogenic. Regardless of the cause, after three years of camera trap monitoring there is little evidence of forest edge avoidance on Hides Ridge.

4. There have been no consistent temporal shifts in frog, rodent or bat species diversity or composition since establishment of the PMA3 monitoring program in 2015 along linear clearings in BAA 1 on the Hides spine-line and in BAA 2 on the Agogo Range near Moro.

5. Bat diversity was significantly greater in open areas at the forest edge compared to the forest interior, which reflects an influx of species that forage in edge and open flight spaces, particularly in BAA 2. These species, mostly small Emballonuridae, have benefitted from creation of additional forest-edge habitats but opening of the forest for linear infrastructure has not impacted the diversity or community composition of forest interior species.

6. Overall, increases in hunting pressure and feral dog predation and the potential spread of exotic rodent species, both associated with installation of the pipeline ROW and associated roads, remain the two major factors most likely to threaten biodiversity values in the BAAs.

General recommendations

1. **Reinstate VAES sampling for frogs**

   Following the 2017 PMA3 survey, Richards et al. (2019) recommended that the use of VAES be reassessed after the 2019 survey due to the logistical difficulties associated with conducting field work at night. Because the VAES surveys continued to generate valuable genetic and acoustic data during the 2019 survey, this reassessment concluded that VAES should be continued in 2021. However, logistical constraints associated with the COVID-19 pandemic prevented the VAES program from proceeding. We recommend that the VAES component of the PMA3 frog monitoring program be reinstated for the 2023 survey.

2. **Continued use of improved genetic assessment methods**

   Genetics-based identification has continued to provide a strong foundation for reliable comparisons between sites, survey years and investigators for multiple taxa during the PMA3 surveys by providing a robust method for species identifications in groups that contain morphologically cryptic fauna. The genomics-based technique should be continued. For frogs, additions to the comparative framework will continue to help resolve the identity and taxonomic classification of several undescribed species. For small rodents, the approach is essential because morphologically diagnostic characters for most species in the study area are inadequate for making consistently accurate identifications whereas the genetic markers provide unambiguous identifications. For bats, it will help to identify future captures of several taxa that are difficult to identify because of unresolved taxonomic issues or a lack of useful diagnostic morphological features.

3. **Assessment of pest invasive rats in human habitation surrounding Project infrastructure.**

   We recommend that consideration be given to a rapid assessment of the presence of *Rattus rattus* and *R. exulans* in inhabited areas around the HGCP to provide context on how common these species are, and how significant a source they might be for invasions along the access road and pipeline on Hides Ridge. ISOS curently has a rodent control program at HGCP but does not keep records of numbers killed so estimates of rodent numbers cannot currently be obtained.

4. **Modification of camera trapping design.**

   We recommend that the camera trapping program continue in 2023 and in subsequent survey years. As far as practical, the biennial schedule should be maintained as fewer data points will extend the time required to make reasonable inferences about population trends. As far as possible, each monitoring survey should take place at the same time of year to control for seasonal effects.

We also recommend that consideration be given to expanding the camera trap sampling design within BAA 1 beyond the site scale, using the same number of cameras placed further apart along the length of Hides Ridge. There is little evidence of edge avoidance by terrestrial birds or mammals on Hides Ridge, and this change in design will improve our ability to assess population change in a variety of priority medium to large-bodied hunting-sensitive species. Additional field assistants may be required to deploy cameras in new positions and to collect environmental covariate data in the time available.

Expansion of the sampling design will necessitate expansion of the occupancy modelling procedure to incorporate data from additional years and the influence of environmental covariates (here applied only to activity rate modelling). We recommend funds continue to be made available for time with the external biostatistician to help expand the analysis protocol.

Sampling design should remain unchanged at BAA 2, given: (1) the strong evidence for edge effects at Arakubi and KP107, and (2) the importance of continuing to monitor the only known population of the new-to-science jewel-babbler (*Ptilorrhoa* sp.) discovered at KP107.

5. **Vegetation assessment of transects and dieback**
   We recommend that the team initiate regular quantitative assessments of vegetation condition along transects to provide a covariate of vegetation condition for future surveys. SImilar assessments should be considered to more rigorously document dieback at regular intervals in BAA 1. Further consideration is required to select an appropriate and rapid assessment method.

6. **Hunting survey with communities near Arakubi**
   We recommend that EMPNG conducts a hunting survey with communities near Arakubi in BAA 2 to capture data on hunting activities from the hunter's perspective. This would also provide the opportunity for EMPNG to raise awareness about the effects of hunting on key wildlife species at that site.

References

Andrews, K.M., Nanjappa, P. & Riley, S.P.D. 2015. *Roads and ecological infrastructure: concepts and applications for small animals.* Baltimore: Johns Hopkins University press.

McAlpine, J. R., Keig, G. & Falls, R. 1983.*Climate of Papua New Guinea*. Canberra: CSIRO.

Richards, S.J. (Editor) 2017. *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. Port Moresby. ExxonMobil PNG Limited.

Richards, S.J. (Editor) 2019. *Results of the second PMA3 Biodiversity monitoring survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. Port Moresby. ExxonMobil PNG Limited.

Richards, S.J. (Editor) 2021. *Results of the third PMA3 Biodiversity monitoring survey of the PNG LNG Upstream Project Area, 8 August–2 September 2019*. Port Moresby. ExxonMobil PNG Limited.

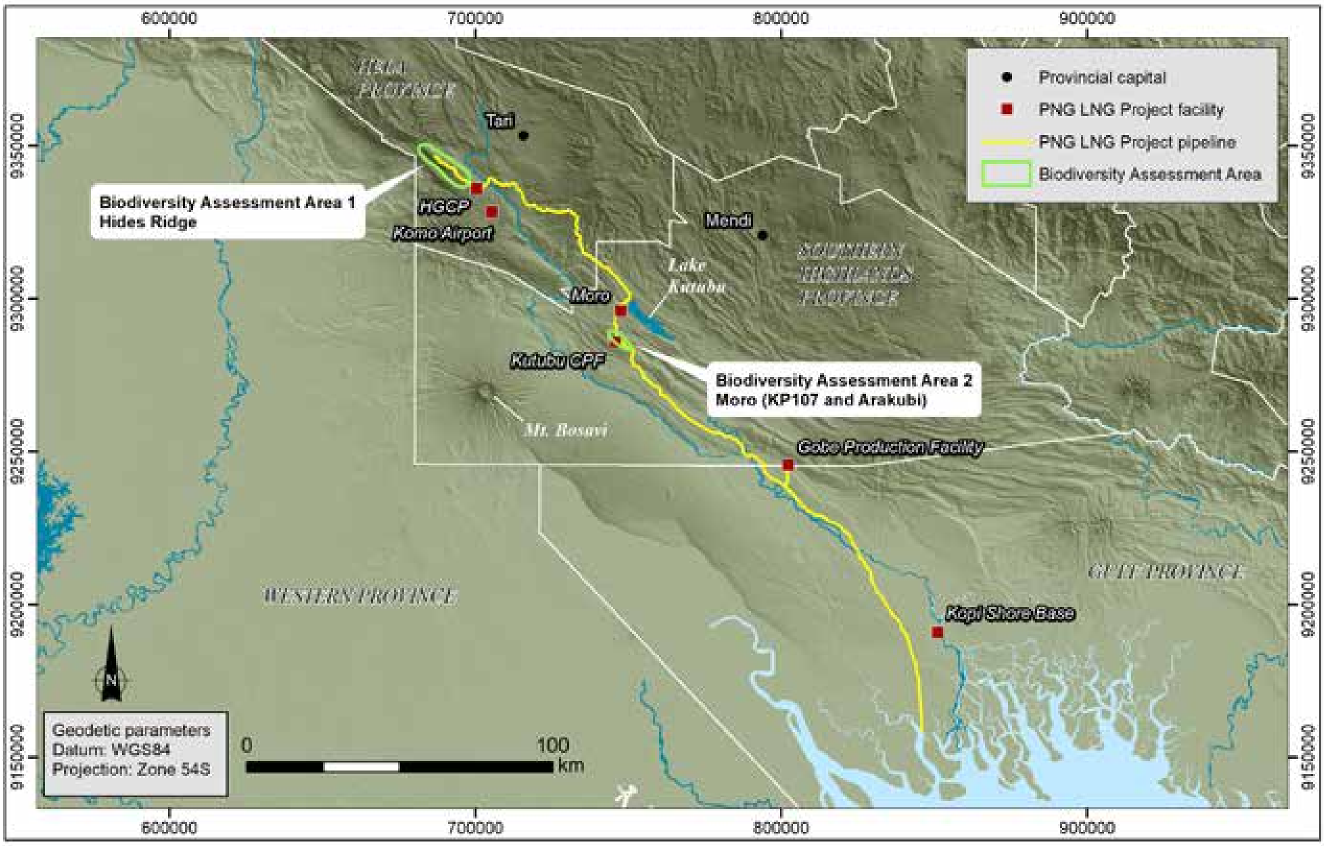


Figure 1. Regional map showing location of the two BAAs surveyed during the PMA3 surveys.

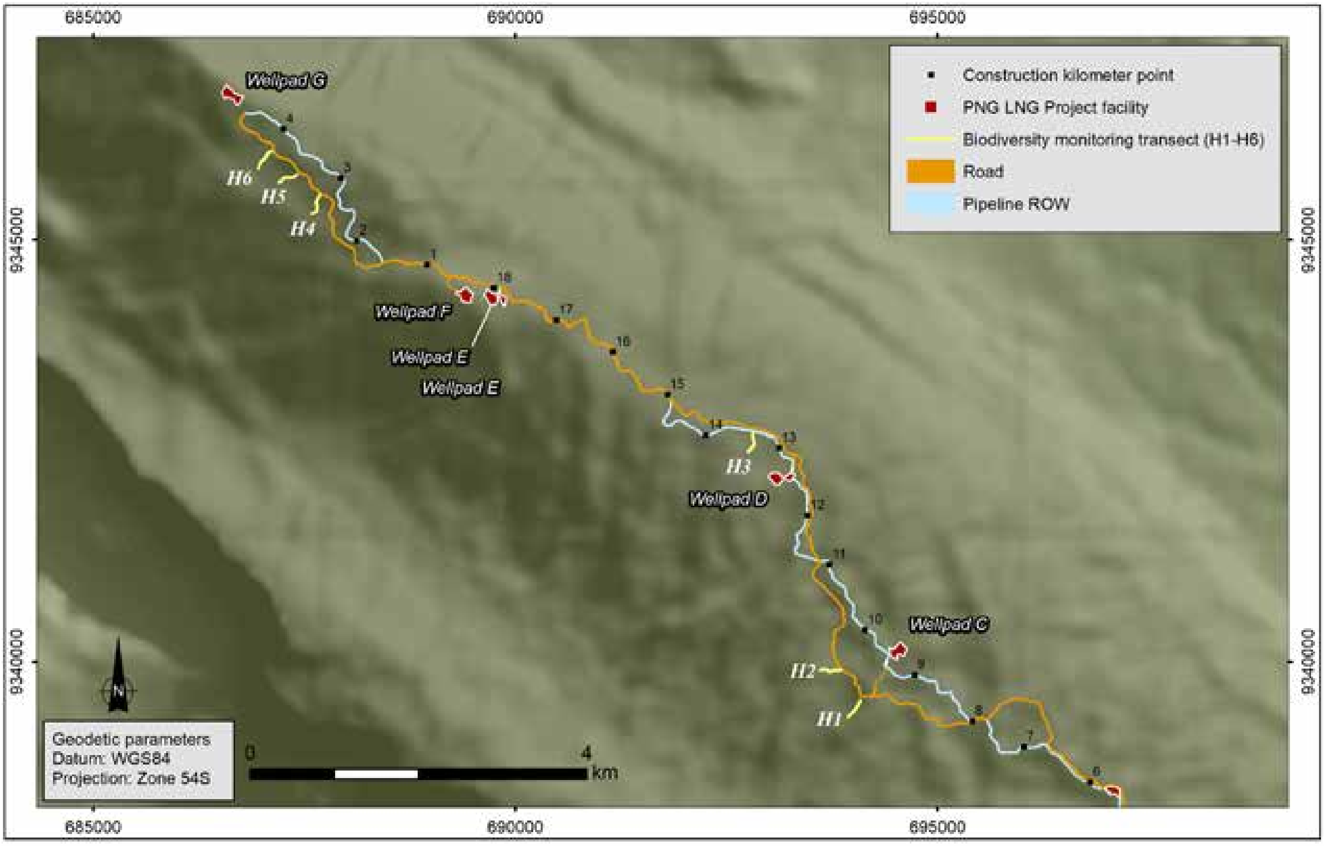


Figure 2. Map showing locations of the six major transects in BAA 1.

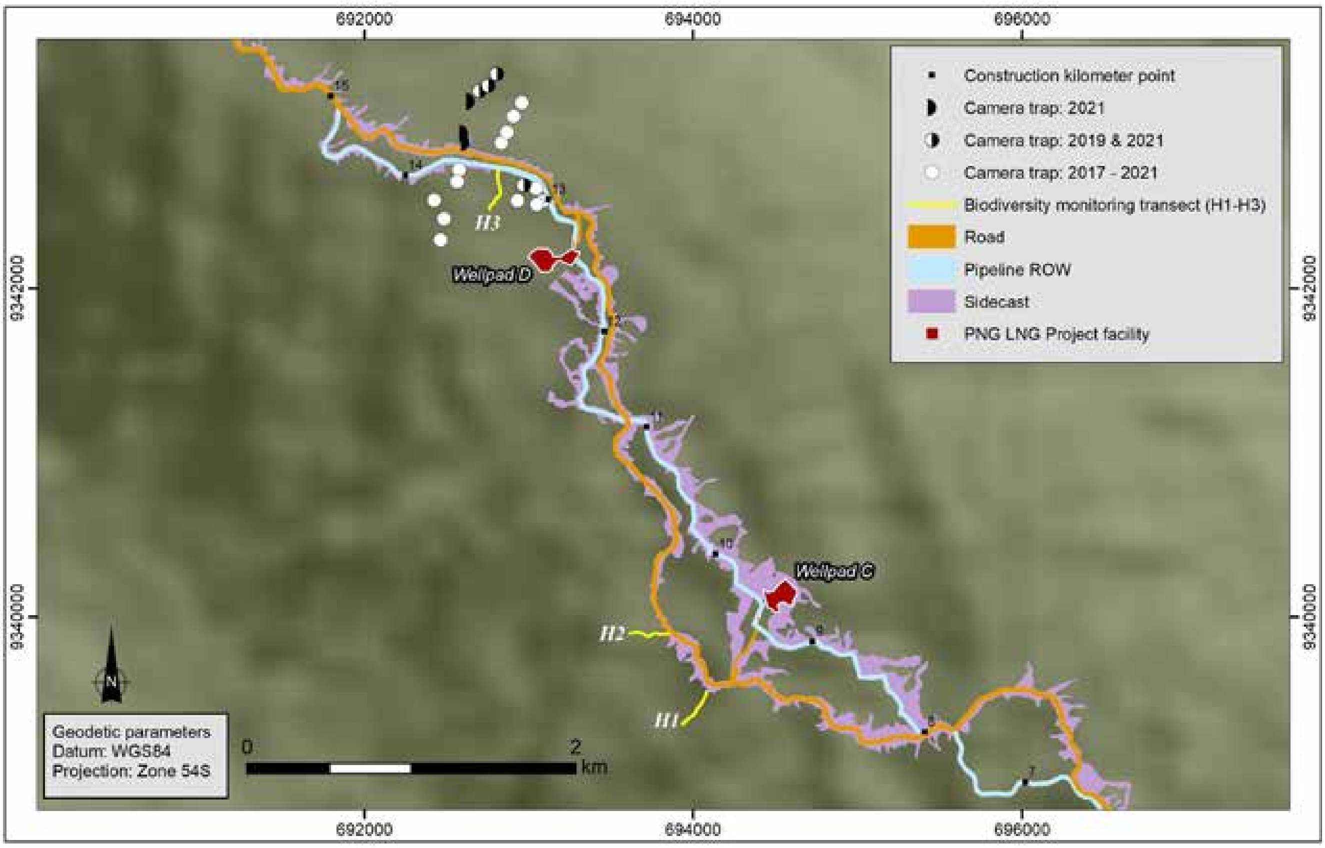


Figure 3. Map of lower elevations in BAA 1 showing details of Transects 1–3, and camera trap arrays.

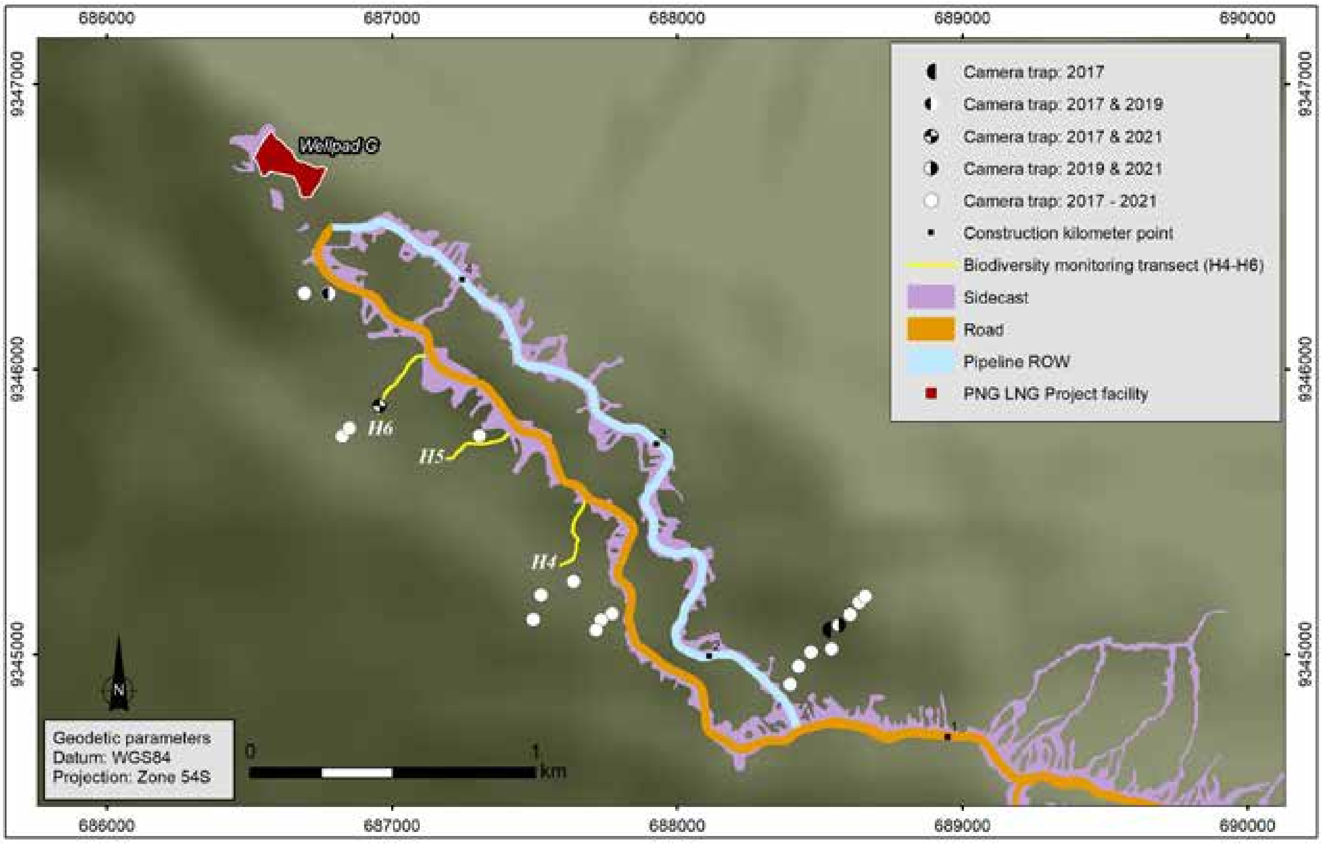


Figure 4. Map of upper elevations in BAA 1 showing details of Transects 4–6 and camera trap arrays.

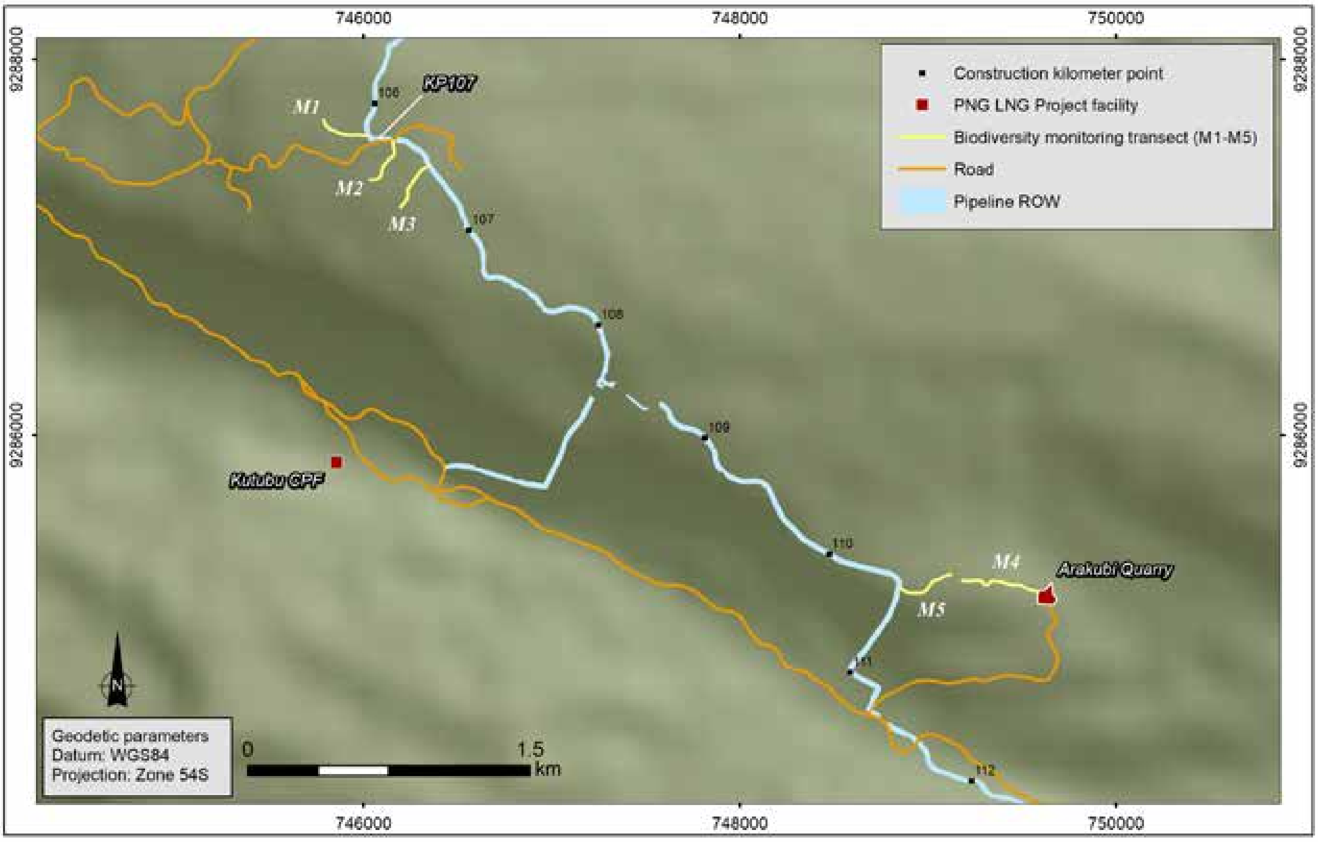


Figure 5. Map showing locations of the five major transects in BAA 2.

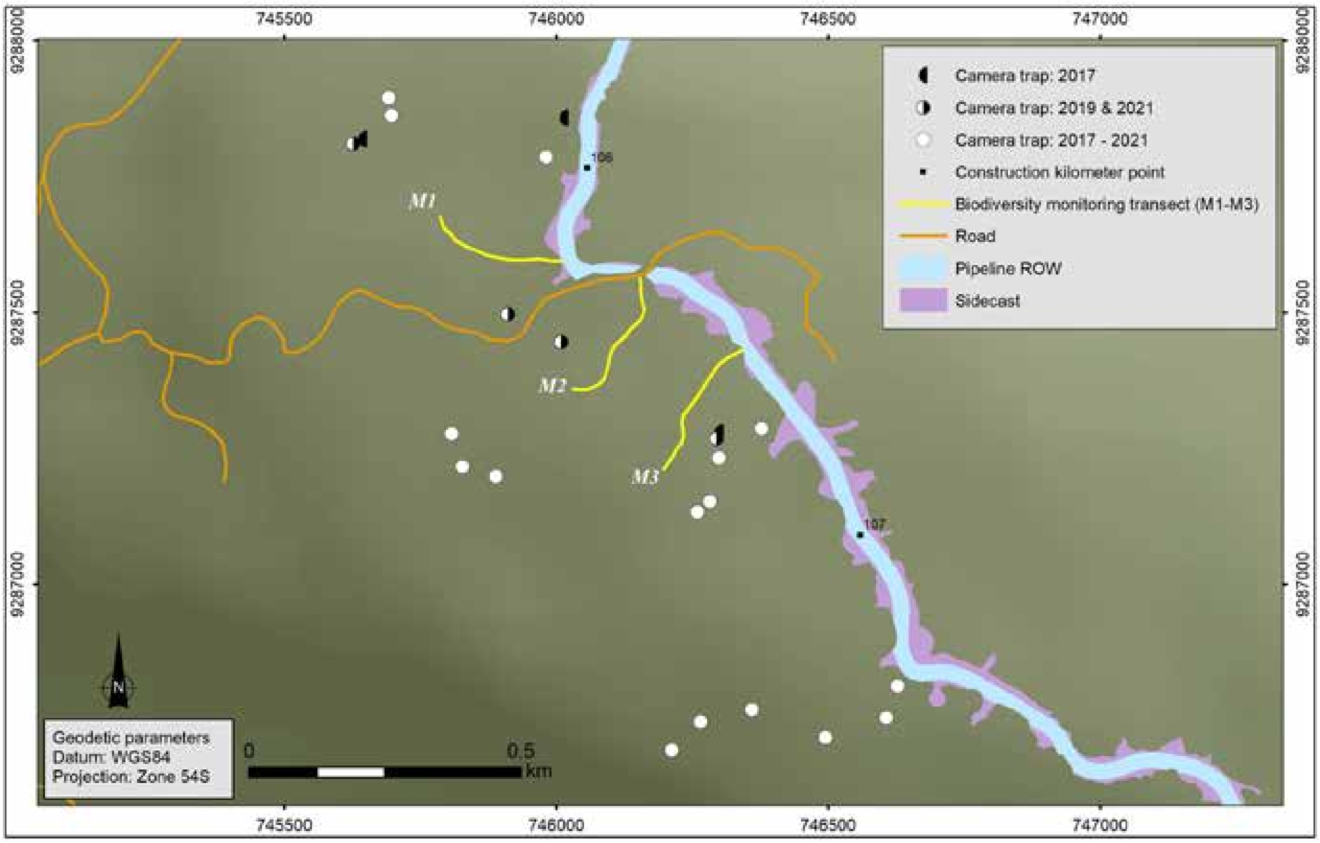


Figure 6. Map showing locations of the three major transects and camera trap arrays at KP107 in BAA 2.

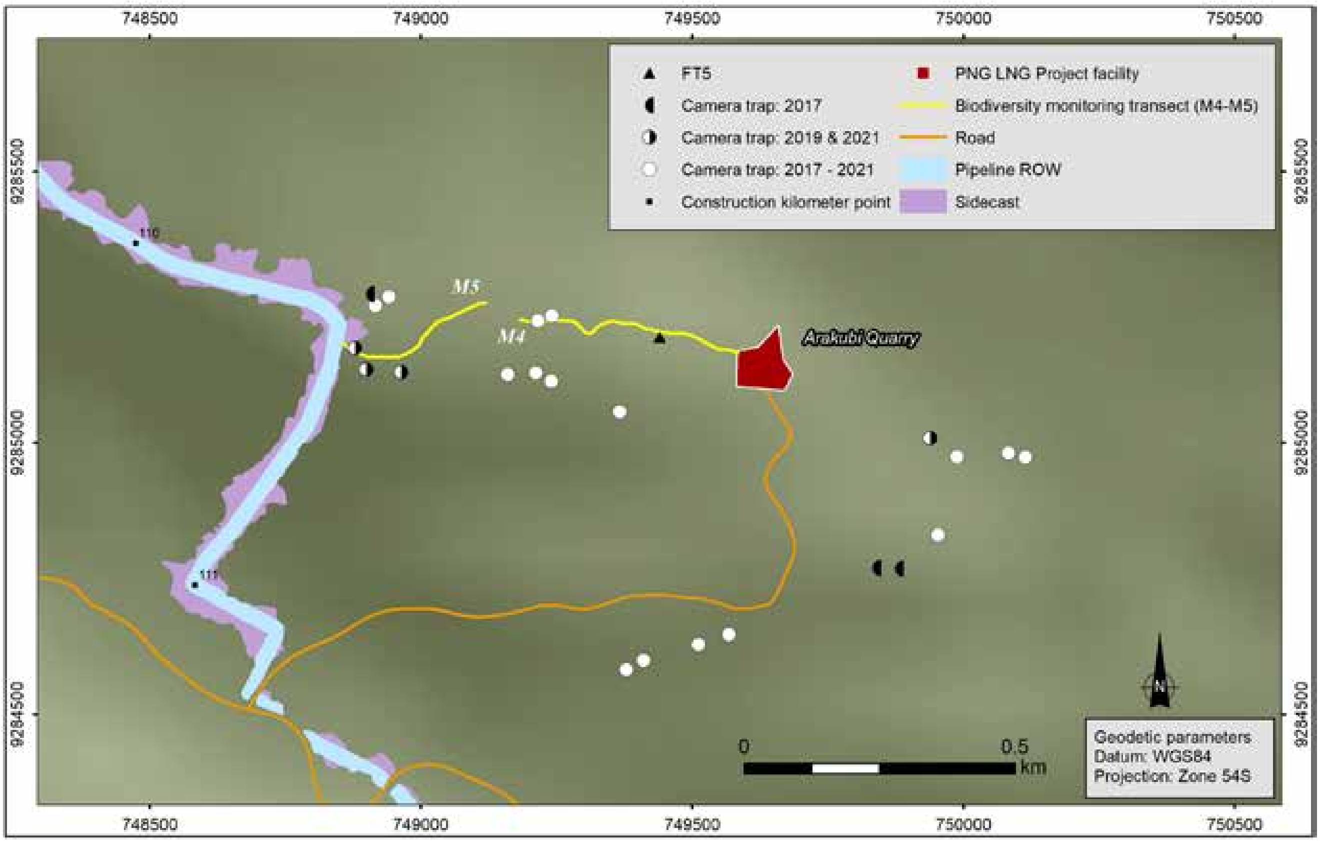


Figure 7. Map showing locations of the two major transects and camera trap arrays at Arakubi Quarry in BAA 2. FT5 is the start of Frog Transect 5.

Appendix 1. Coordinates and elevations (at start) for each of the 11 standard survey transects established in BAA 1 and BAA 2.

BAA	Transect	Position	Coordinates	Elevation
1	H1	Start	S5.97229° E142.75333°	2140
1	H1	End	S5.97416° E142.75198°	
1	H2	Start	S5.96915° E142.75127°	2150
1	H2	End	S5.96913° E142.74908°	
1	H3	Start	S5.94369° E142.74177°	2285
1	H3	End	S5.94579° E142.74132°	
1	H4	Start	S5.91835° E142.69531°	2685
1	H4	End	S5.92036° E142.69456°	
1	H5	Start	S5.91621° E142.69289°	2745
1	H5	End	S5.91699° E142.69095°	
1	H6	Start	S5.91372° E142.69021°	2730
1	H6	End	S5.91553° E142.68877°	
2	M1	Start	S6.44023° E143.22424°	1390
2	M1	End	S6.43950° E143.22221°	
2	M2	Start	S6.44051° E143.22552°	1380
2	M2	End	S6.44236° E143.22442°	
2	M3	Start	S6.44169° E143.22724°	1365
2	M3	End	S6.44368° E143.22594°	
2	M4	Start	S6.46206° E143.25662°	995
2	M4	End	S6.46152° E143.25299°	
2	M5	Start	S6.46124° E143.25242°	1050
2	M5	End	S6.46192° E143.25004°	

Chapter 1 – Frogs

Stephen J. Richards, Kyle N. Armstrong, Elizah Nagombi, Chris Dahl and Luda Paul

Cophixalus melanogenys, a species from KP107 that was named in 2021

Summary

Background and aims

To determine whether linear infrastructure created by ExxonMobil PNG Limited's pipeline ROW and Project roads is having an impact on frogs in the Upstream Project Area, we have established a program to monitor frog populations and communities in two Biodiversity Assessment Areas (BAAs) at Hides Ridge (BAA 1) and on the Agogo Range near Moro (BAA 2). The monitoring program, scheduled to occur every two years, was initiated in 2015 using 1) Visual and Audio Encounter Surveys (VAES) and 2) automated sound recording of frog calls (Acoustic Recorders). The fourth monitoring survey was conducted during July 2021 using only Acoustic Recorders due to constraints imposed by COVID-19 that prevented night work in both BAAs.

This report presents the results of the 2021 monitoring survey and compares them with the 2015, 2017 and 2019 results to assess whether there is currently evidence that Project infrastructure is having an impact on frog populations in either BAA.

Major results

A total of 35 species of frogs was documented along permanent transects that run perpendicular to infrastructure clearings in BAA 1 on Hides Ridge and BAA 2 on the Agogo Range near Moro. Five species that were detected during 2015, 2017 and/or 2019 were not detected by the Acoustic Recorders during 2021. Four of these were rarely heard calling during previous surveys so this result was not unexpected because Visual and Audio Encounter Surveys that can detect non-calling individuals were not conducted during 2021 due to logistical constraints imposed by COVID-19.

Analysis of data from the Acoustic Recorders found no evidence in either BAA for shifts in species diversity or composition with increasing distance from infrastructure clearings across the four survey years. No species was associated significantly with the forest edge.

Our data also reinforce the pattern documented during the 2015–2019 surveys that species diversity is significantly lower at higher elevations. During the 2021 PMA3 survey nine frog species were detected in BAA 1, and 26 species in BAA 2. No frog species were shared between transects in BAA 1 and BAA 2.

Conclusions and recommendations

Results of the 2021 PMA3 survey indicate that there continues to be no detectable temporal shifts in frog diversity and composition since establishment of the PMA3 monitoring program in 2015 along linear clearings in BAA 1 on the Hides spine-line and in BAA 2 on the Agogo Range near Moro.

The biodiversity values of frogs in these areas remain intact.

Richards et al. (2019) argued that, despite logistical issues associated with night work, VAES surveys be continued in 2021 to complement the use of Acoustic Recorders because they facilitate collection of data to improve species identification capacity and enhance the accuracy of call-based monitoring. However, additional safety protocols necessitated by the COVID-19 pandemic prevented this component of the frog monitoring survey from proceeding during the 2021 survey.

We therefore recommend that procedures be developed that will allow the resumption of VAES surveys in 2023.

Given that there is now both an excellent understanding of frog diversity in the study area, and a large resource of acoustic recordings, there is opportunity to streamline aspects of the acoustic analysis by developing a semi-automated identification system. This would increase efficiency of both the static acoustic survey and VAES transect components of the survey.

Introduction

Amphibians are excellent indicators of environmental conditions because their thin permeable skin makes them vulnerable to subtle changes in both aquatic and terrestrial environments. Frogs were identified as a core taxon in EMPNG's Biodiversity Strategy, and the presence of a distinct assemblage of torrential-stream dwelling treefrogs (Family Pelodryadidae) was partly responsible for upland rainforest streams being recognised as focal habitats. However, many frog species in New Guinea do not use aquatic habitats for reproduction, instead depositing large, yolk-filled eggs on plants or under litter on the forest floor where they hatch directly into froglets (Anstis et al. 2011). All New Guinean species in the diverse family Microhylidae are known or expected to reproduce this way (Menzies 2006) and as a result this group dominates the frog faunas of karst habitats in Papua New Guinea.

The karst environments of Hides Ridge in BAA 1 and on the Agogo Range near Moro in BAA 2 are characterised by limited flowing water so the PMA3 frog monitoring program was designed to document the diversity (here also called 'species richness') and composition (which species are present) of microhylid frog communities. The frog monitoring program was initiated in May 2015 to document frog diversity and community composition in both BAAs using quantitative, repeatable sampling techniques that provided baseline data against which future changes in frog diversity and community composition could be measured, and assessed whether frog diversity and community composition changed with increasing distance from Project infrastructure. Results of the 2015 field survey are summarised in Richards and Armstrong (2017), of the 2017 survey in Richards et al. (2019) and results of the 2019 survey are presented in Richards et al. (2021). Here we present the results of the fourth frog monitoring survey, conducted during July 2021.

Methods

Frog surveys in 2021 were conducted along the same permanent transects that were established during 2015: on Hides Ridge (BAA 1) between 4 and 13 July, and on the Agogo Range in the Moro area (BAA 2) between 17 and 24 July (Figure 1 in Report Summary). Each of these BAAs was divided into two survey 'sites' that differed in elevation:

- Hides Ridge (BAA 1):
    - Transects H1–3: between Wellpad C and Wellpad D, at elevations of 2,100–2,400 m asl ('Hides Low' — the 2,200 m asl elevation band).
    - Transects H4–6: between Wellpad E and Wellpad G, at 2,660–2,780 m asl ('Hides High' — the 2,700 m asl elevation band).
- Moro area (BAA 2):
    - Transects M1–3: on the Agogo Range in the vicinity of KP107, at 1,340–1,410 m asl ('KP 107' — the 1,400 m asl elevation band).
    - Transects M4 and M5: west of Arakubi Quarry and east of the pipeline ROW, at 1,000–1,070 m asl ('Arakubi' — the 1,000 m asl elevation band).

Surveys for frogs on transects

Two quantitative methods, Visual and Audio Encounter Surveys (VAES) and audio monitoring with acoustic recorders, were developed for the PMA3 program to document frogs along transects in both BAAs and these are described in detail by Richards and Armstrong (2017). However, logistical constraints associated with the COVID-19 pandemic prevented the team from undertaking VAES during the 2021 survey which was consequently restricted to audio monitoring with acoustic recorders.

Audio monitoring with acoustic recorders

Acoustic Recorders provide high quality data on calling frogs from standardised recording effort at known distances into the forest from the forest edge. During the 2015 survey autonomous acoustic recordings were generated by Wildlife Acoustics Song Meter SM3 recorders and these were used again in 2017. During the 2019 survey we phased in the use of smaller, lighter Frontier Labs Bioacoustic Audio Recorder (BAR) units by placing duplicated units alongside SM3 recorders at each recording locality. All other protocols remained the same. Subsequent comparisons of recordings by paired units did not reveal any differences in call detectability so in 2021 the SM3 units were replaced entirely by BAR recording units.

In 2021, as in previous surveys, Acoustic Recorders were placed at three recording sites at increasing distances from the forest edge (5 m, 70 m and 170 m) on transects H1–6 established in BAA 1 and M1–5 in BAA 2 (Figures 2–7 in Report Summary). Recording units were placed 65 and 100 m apart to reduce the likelihood that an individual frog would be detected by more than one unit. The microphone of the recorder set at the 5 m position on each transect was oriented to maximise reception of signals from the edge habitat adjacent to the open area over the road. Units recorded continuously in WAV format at a sampling rate of 48 kHz for two consecutive nights at each recording site on each transect, giving a total of 36 recording nights over an 8-night survey period for BAA 1, and 30 recording nights over a 6-night survey period for BAA 2 (Table 1.1).

A summary of the design is presented in Table 1.1 and coordinates for each recording location are presented in Appendix 1.1. Unless otherwise specified, for the purposes of this chapter the abbreviation 'ROW' is used to refer to linear infrastructure including both the pipeline right-of way and associated roads.

Table 1.1. Summary of the experimental design and frog acoustic recording site placements.

BAA	Elevation	Transect	Distance from forest edge			Total nights
			5 m	70 m	170 m	
BAA 1	Hides High	H4—2,700 m (2,681–2,696 m)	2	2	2	36
Hides Ridge		H5—2,750 m (2,726–2,756 m)	2	2	2	
		H6—2,730 m (2,725–2,736 m)	2	2	2	
	Hides Low	H1—2,150 m (2,148–2,163 m)	2	2	2	
		H2—2,200 m (2,171–2,229 m)	2	2	2	
		H3—2,300 m (2,296–2,327 m)	2	2	2	
BAA 2	KP107	M1—1,400 m (1,397–1,405 m)	2	2	2	30
Agogo Range		M2—1,380 m (1,315–1,397 m)	2	2	2	
		M3—1,380 m (1,369–1,389 m)	2	2	2	
	Arakubi Quarry	M4—1,030 m (995–1,041 m)	2	2	2	
		M5—1,050 m (1,051–1,073 m)	2	2	2	

Audio monitoring of frogs at Wellpad D on Hides Ridge

An Acoustic Recorder was deployed for two nights at a small pond adjacent to Wellpad D that provides one of the few habitats for aquatic frogs in BAA 1 and, as well as supporting a population of the Rainbow Treefrog (*Litoria iris*), remains the only known locality for *Litoria vivissimia*, a spike-nosed treefrog discovered during the EIS surveys. The resulting data were examined using the same methods described for Acoustic Recorders placed on transects. We were unable to conduct VAES night surveys around the edge of the pond due to logistical constraints but during the day we noted the presence and abundance of gelatinous egg masses of the Rainbow Treefrog hanging from low vegetation (0, 1–10, >10 clumps) (Figure 1.1).

Figure 1.1. Gelatinous egg masses of the Rainbow Treefrog, *Litoria iris*, hanging on leaves above water at a small pond on Hides ridge.

Data synthesis and statistical analyses

Acoustic data

Sixty-six nightly recordings collected from the 11 transects were analysed. Frog presence in 2021 was scored using the modified method developed in 2017 and used again in 2019, as follows: for each 24-hour recording period at each site we analysed the five 1-hour sound files starting at (or closest to) 19:00 to 23:00 inclusive (recording time 19:00 to 00:00). The entire 60 minutes of each of the five 1-hour files was scanned visually in 30 s blocks noting the presence/absence of calls for each species.

Statistical Analysis

Statistical analyses were conducted only on data obtained from the Acoustic Recorders. Frog diversity was compared across elevations and distances from linear infrastructure between years by fitting a Linear Mixed Effects Model by Maximum Likelihood to the data. Variation in community composition (i.e., the mix of species found on each transect) was explored by calculating the Bray-Curtis Dissimilarity Index and then performing Non-metric Multidimensional Scaling (NMDS). The NMDS is an ordination that grouped sites in two-dimensional space based on the similarity/dissimilarity of the mix of their component species.

To best understand which individual species might be associated with particular habitat characteristics, or responding to any change happening in the study area, the Indicator Species analysis used in previous reporting was refined. Indicator Species indices (Dufrêne and Legendre 1997; De Cáceres and Legendre 2009) were calculated for each species at different distances from the ROW, and at different elevations, from all survey years combined.

There are three related metrics (range 0 to 1). The indicator value index (I) shows the value of a particular species as an 'Indicator Species' for that factor level (e.g., at a particular elevation level). Species found in many habitat types tend to have low index scores, and species with a high value are useful Indicator Species for a particular condition or habitat. It is the product of two components, 'A' and 'B'. Component 'A' ('specificity') is the probability that the surveyed site belongs to the factor level given that the species has been detected (i.e., whether a species is limited to a certain category). Component 'B' ('fidelity') is the probability of finding the species in sites belonging to a factor level (i.e., a species may be found only in a certain category, but if it is rare and difficult to detect, then its usefulness as an Indicator Species is reduced).

All analyses were conducted, and output plots were produced, using a modified version of the custom-written [R] language statistical computing language script that was developed for these surveys.

DNA barcoding

We were unable to obtain additional tissues of frog species during the 2021 field survey, so DNA barcoding was not undertaken.

Results and Discussion

A species list showing the frog species detected by Audio Recorders on each transect is shown in Table 1.2. This table also summarises species detections at increasing distances from the disturbance edge.

Table 1.2. Summary of species encountered on each transect in both BAAs, using Acoustic Recorders. Species presence at each recording site is presented for both elevation and distance. The sequence of circles is increasing distance from the road (5, 70, 170, left to right), with a filled black circle indicating a detection of that species, an open circle an apparent absence. Species shaded in grey were not encountered during 2021.

	BAA 2					BAA 1					
Elevation	**1,000**	**1,000**	**1,400**	**1,400**	**1,400**	**2,200**	**2,200**	**2,200**	**2,700**	**2,700**	**2,700**
Transect	**M4**	**M5**	**M1**	**M2**	**M3**	**H1**	**H2**	**H3**	**H4**	**H5**	**H6**
PELODRYADIDAE											
Litoria iris	○○○	○○○	○○○	○○○	○○○	○○○	●○○	○○○	○○○	○○○	○○○
Litoria sp. 1 'yellow legs'	○●○	○○○	●●●	●●○	●●○	○○○	○○○	○○○	○○○	○○○	○○○
Litoria sp. cf. *becki*	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
LIMNODYNASTIDAE											
Lechriodus aganoposis	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
MICROHYLIDAE											
Asterophrys slateri	●●●	●●●	○○○	○●○	●○○	○○○	○○○	○○○	○○○	○○○	○○○
Austrochaperina fulva	●●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Austrochaperina laurae	○○○	○○○	●●●	●●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○
Callulops omnistriatus	●●●	●○●	●●●	●●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○
Callulops wilhelmanus	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	●○○	○●○	●○○
Choerophryne alainduboisi	○○●	●●●	●●●	●●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○
Choerophryne brevicrus	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	●○●	○●●
Choerophryne burtoni	○○○	○○○	○●○	○●●	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Choerophryne crucifer	○○●	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Choerophryne multisyllaba	○○○	○○○	○●●	○○●	○○●	○○○	○○○	○○○	○○○	○○○	○○○
Choerophryne murrita	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Choerophryne sp. 1 'arboreal'	○○○	○○○	○○○	○○○	○○○	●○●	●●●	●●●	○○○	○○○	○○○
Choerophryne sp. 2 'tiny'	○○○	○○○	○○○	○○○	○○○	●○●	○●○	●●●	○○○	○○○	○○○
Cophixalus cateae	○○○	○○○	○○○	○●●	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Cophixalus wempi	○○○	○○○	○○○	○○○	○●●	○○○	○○○	○○○	○○○	○○○	○○○
Cophixalus sp. 1 'musical call'	○○○	○○○	●●●	○●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○
Cophixalus brevidigitus	○○○	○○○	○●○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Cophixalus melanogenys	○○○	○○○	●●●	●●●	●●●	○○○	○○○	○○○	○○○	○○○	○○○
Cophixalus sp. 4 'loud grunter'	○○○	○○○	○○○	○○○	○○○	●●●	●●●	●●●	○○○	○○○	○○○
Copiula bisyllaba	●●○	○●●	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○
Hylophorbus richardsi	○○○	○○○	○○○	○○○	○○○	○○○	○○●	○●●	○○○	○○○	○○○
Hylophorbus sp. 1 'slow call'	○○○	○○○	●●●	●●●	●●○	○○○	○○○	○○○	○○○	○○○	○○○
Hylophorbus sp. 2 'fast call'	●●●	●●●	○●●	●●●	○●●	○○○	○○○	○○○	○○○	○○○	○○○
Liophryne schlaginhaufeni	○○●	●●●	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○	○○○

	BAA 2					BAA 1					
Elevation	1,000	1,000	1,400	1,400	1,400	2,200	2,200	2,200	2,700	2,700	2,700
Transect	M4	M5	M1	M2	M3	H1	H2	H3	H4	H5	H6
Oreophryne anamiatoi	ooo	ooo	ooo	ooo	ooo	ooo	●●●	ooo	ooo	ooo	ooo
Oreophryne flavomaculata	●●●	●●●	●●●	●●●	●●●	ooo	ooo	ooo	ooo	ooo	ooo
Oreophryne nicolasi	ooo	oo●	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Oreophryne notata	ooo	ooo	ooo	ooo	ooo	●●●	●●●	●●●	●●●	●●●	●●●
Oreophryne sp. cf. *notata*	oo●	●●●	●●●	●●●	●●●	ooo	ooo	ooo	ooo	ooo	ooo
Oreophryne oviprotector	●●●	●●●	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Oreophryne pseudunicolor	●●●	●●●	●●o	●oo	●o●	ooo	ooo	ooo	ooo	ooo	ooo
Oreophryne sp. 2 'ratchet call'	ooo	ooo	●●●	oo●	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Sphenophryne cornuta	●oo	oo●	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Xenorhina lacrimosa	●o●	●o●	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Xenorhina sp. 2 'fast call'	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Gen. nov. sp. nov.	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo	ooo
Total Richness											
2021 only	**15**	**14**	**15**	**16**	**14**	**4**	**7**	**5**	**2**	**3**	**3**
All years Acoustic only	**16**	**16**	**19**	**18**	**17**	**9**	**9**	**7**	**3**	**3**	**3**
All years Acoustic + VAES	**17**	**18**	**20**	**20**	**20**	**9**	**9**	**7**	**4**	**4**	**3**
All years by elevation	**1000-**	**18**		**1400-**	**21**		**2200-**	**10**		**2700-**	**4**

Overview of the frog fauna

A total of 35 species of frogs was documented on the permanent transects, including nine species in BAA 1 and 26 species in BAA 2 (Table 1.2). This is within the range of species diversities encountered during the 2015–2019 surveys (34–37 species; Richards & Armstrong 2017; Richards et al. 2019, 2021). Five species that were encountered during at least one of the previous surveys were not encountered during 2021. Four of these, *Litoria* sp. cf. *becki*, *Lechriodus aganoposis*, *Xenorhina* sp. 2 'fast call' and Gen. nov. sp. nov. are species that were rarely or never detected on Audio Recorders at night during previous surveys, so their absence is not surprising. The fifth species that was not detected, *Choerophryne murrita*, has been encountered at KP 107 on all previous surveys but occurs at low densities on the transects (personal observations). Attempts to detect this species should be a priority during the 2023 PMA3 survey.

No frog species were added to the total species inventory for either BAA 1 or BAA 2 during the 2021 PMA3 surveys.

As documented during previous PMA3 surveys the family Microhylidae, a group characterised by direct embryonic development, dominated the fauna in 2021. As a result of the failure to detect one species of pelodryadid treefrog (*Litoria* sp. cf. *becki*) and the Limnodynastid *Lechriodus aganoposis* during 2021 the proportion of microhylids in the fauna increased slightly to 94% from 89% in 2019 but was still within the range recorded during previous surveys (89–97%; Richards & Armstrong 2017; Richards et al. 2019, 2021).

Influence of Project infrastructure on species diversity, community composition and relative abundances

To determine whether edge effects reduce frog diversity or result in changes to community structure, we analysed acoustic recording data to determine whether there were any relationships between distance from the primary forest edge and frog species diversity (= richness) and community composition using the protocols described by Richards and Armstrong (2017) and Richards et al. (2019, 2021).

Graphical summaries of species diversity recorded in 2015, 2017, 2019 and 2021 at increasing distances from the forest edge by acoustic recordings are shown in Figures 1.2 and 1.3, and the means ± standard deviations for frog diversity at each distance from the road or ROW, elevation and a comparison between survey years are presented in Table 1.3. Statistical analysis using Generalized Linear Mixed Model (GLMM) (Table 1.4) found no significant influence of distance from infrastructure on species diversity at any site. Furthermore, the slight (non-significant) trend for increasing species diversity with increasing distance from infrastructure at KP107 that was apparent in 2017 and 2019 (Richards et al. 2019, 2021) was no longer apparent in 2021 (Figure 1.2).

GLMM analysis supports the hypothesis that, overall, the influence of distance from infrastructure on species diversity was not statistically significant during any year.

There were several statistically significant interaction terms (Table 1.4) but these are difficult to interpret because there are no consistent patterns of higher levels of species richness at one distance category at a particular elevation in either all years, or only one year. For example, in 2021 species richness was higher than in previous years at 170 m at an elevation of 1,000 m asl, and at 70 m at 1,400 m asl (Figure 1.2). A similar situation was reported by Richards et al. (2021) who concluded that "the statistically significant results are most likely the result of the level of sampling and general variation in habitats, weather patterns and frog population size over the years. They do not contribute any further understanding to the effect of linear infrastructure." Results from the 2021 survey support this contention, suggesting that these results reflect variation that is not accounted for by either year or distance.

The lack of a temporal shift in species diversity at different distances along the ROW since 2015 strongly suggests that the Project infrastructure in BAA 1 and BAA 2 is currently having no detectable impact on local frog communities. This is further supported by Non-metric Multidimensional Scaling (NMDS) analysis which have not shown any differentiation of frog communities based on distance from linear infrastructure in 2015, 2017, 2019 or 2021 (Figure 1.3).

For the first time since 2015, the 2021 survey did not add to the total diversity of species known from transects in both BAAs, suggested that the frog assemblage is almost or fully characterised within the study area. This is not surprising because only one additional species was encountered in 2017 and again in 2019, and VAES surveys were not conducted during 2021. Although it is possible that a small number of additional species will be detected during future surveys, particularly if VAES surveys are reinstated, it is likely that the accumulation of species at the PMA3 survey sites has rapidly reached an asymptote and that few additional species will be encountered in the future.

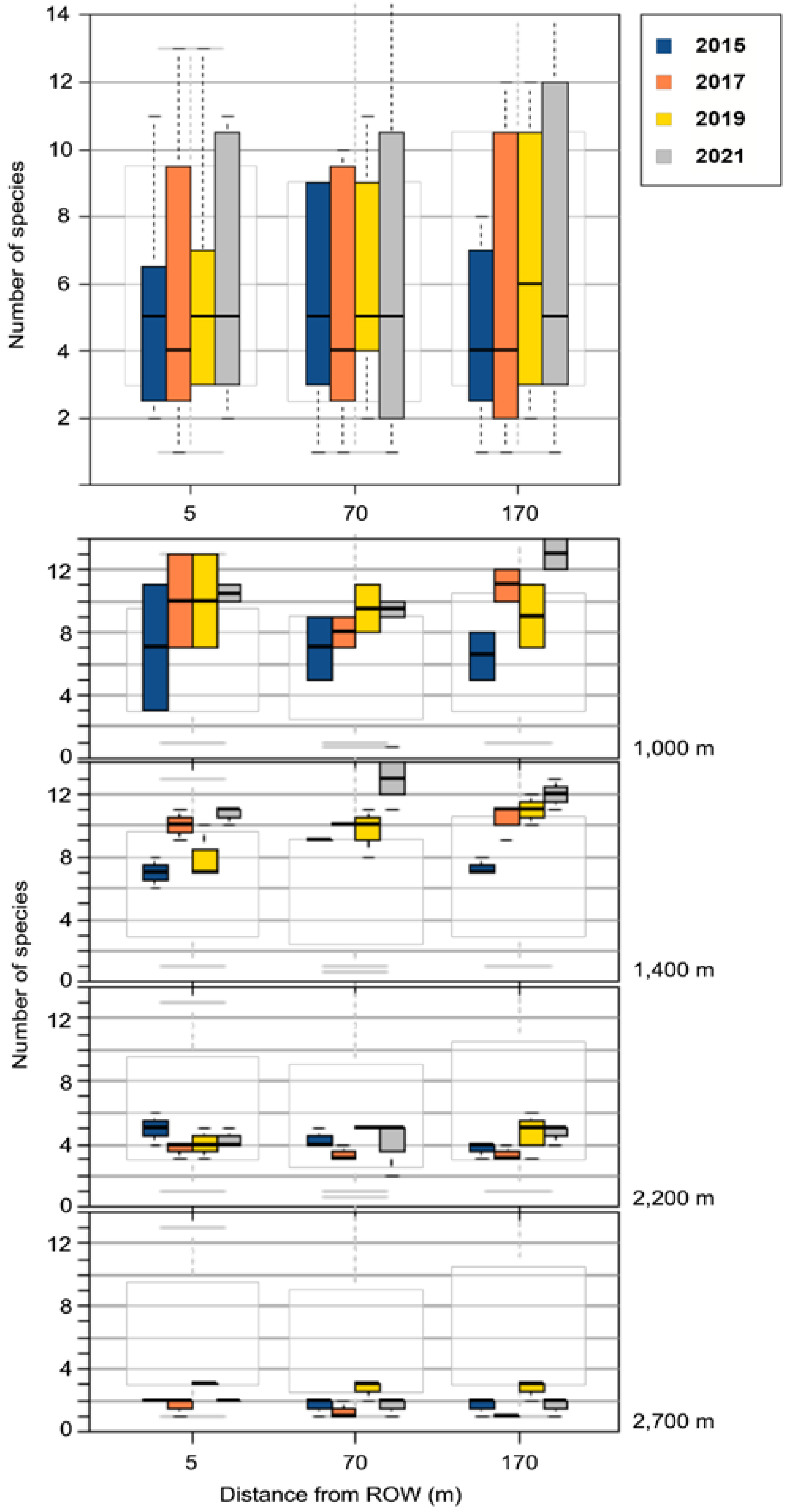


Figure 1.2. Summary of frog diversity (as number of species) at different distances from the forest edge based on data from the acoustic survey in 2015, 2017, 2019 and 2021. The uppermost graph is pooled across all distances, while those below are for each of the elevational zones (B). Boxplot components are: central bar—median; boxes—inter-quartile range, with second quartile group below median, third quartile group above median; bars—minimum and maximum values of quartile groups; circles—outliers.

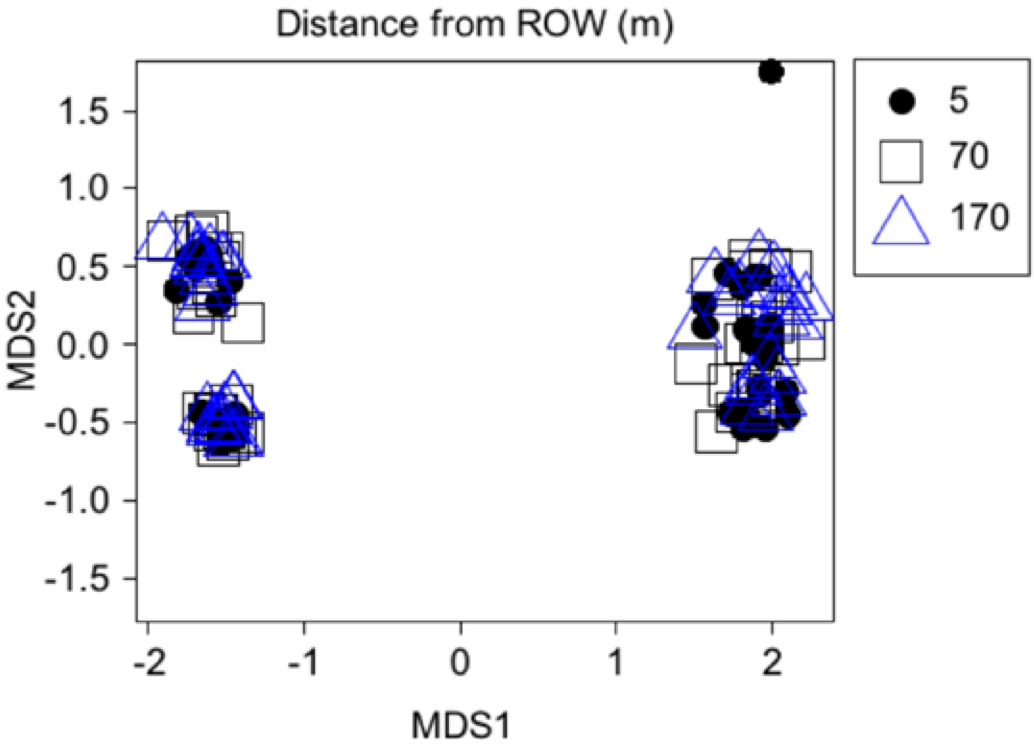


Figure 1.3. Non-metric Multi-dimensional Scaling (NMDS) ordination summarising patterns of species composition at different distances from the road or ROW based on acoustic recording datasets in 2015, 2017, 2019 and 2021.

Table 1.3. Summary of means ± standard deviation in 2021 for frog diversity at each distance from the road or ROW, elevation and a comparison between survey years.

Distance (m)	**Acoustic recordings**
5	5.9 ± 3.5
70	6.0 ± 3.8
170	6.2 ± 4.1
Elevation (m)	**Acoustic recordings**
1,000	9.2 ± 2.8
1,400	9.8 ± 2.0
2,200	4.2 ± 0.9
2,700	1.9 ± 0.7
Year	**Acoustic recordings**
2015	5.0 ± 2.8
2017	5.8 ± 4.1
2019	6.3 ± 3.4
2021	6.9 ± 4.6

Table 1.4. Summary of the tests of the Linear Mixed Effects Model (values from the Analysis of Deviance table; Type II Wald F tests with Kenward-Roger df) and post hoc pairwise comparisons to test for the influence on frog diversity of elevation and distance from the road or ROW, and year (2015, 2017, 2019 and 2021) for the acoustic recording data set (only significant pairs shown; values are elevations in metres; Significance codes: 0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1). Full model coded in [R] as: m <- lmer(total_richness ~ dist + elev + year + dist*elev + elev*year + dist*year + dist*elev*year + (1 | transect) + (1|transect.dist) + (1|transect.year), data = y).

Acoustic recordings	F	Df, Df.res	Pr(>F)	Pairwise
Distance	0.6	2, 14	0.56 NS	All NS
Elevation	43.41	3, 7	<0.0001***	1,000 > 2,200** 1,000 > 2,700*** 1,400 > 2,200*** 1,400 > 2,700***
Year	15.29	3, 21	<0.00012***	2015 < 2017* 2015 < 2019*** 2015 < 2021*** 2017 < 2021**
Distance*Elevation	2.81	6, 14	0.053 NS	—
Distance*Year	2.76	6, 42	0.023*	—
Elevation*Year	6.95	9, 21	0.0001***	—
Distance*Elevation*Year	2.1	18, 42	0.024*	—

Elevational trends in frog diversity and community composition

Elevational trends in both species diversity and composition have not changed over the course of the PMA3 program. The frog fauna in BAA 2 remains substantially more diverse than that encountered in BAA 1 on Hides Ridge, with nearly three times as many species (26 vs 9) detected there; and the general reduction in frog diversity with increasing elevation that was reported following the 2015, 2017 and 2019 surveys was observed again in 2021. The one exception was that slightly more species were detected at Arakubi than at KP 107 (18 vs. 17 species) in 2019 but this situation was reversed in 2021 (16 vs.18 species; Table 1.2). Figure 1.4 illustrates the rapidly dropping species richness with increasing elevation across the two BAAs that was documented in all years. In all four years elevation was the major factor influencing differences in the number of species on transects in both BAAs (Tables 1.3, 1.4; Figure 1.4) and GLMM statistical analysis of 2021 data demonstrated that these differences are significant, with lower elevation sites having significantly higher diversity than high elevation sites (Table 1.4). This pattern is widely repeated in the mountains of New Guinea (e.g. Richards and Dahl 2011; Tallowin et al. 2017).

The differences in composition of frog communities at different elevations are illustrated by the clear clustering of elevations in NMDS ordinations of species presence based on the acoustic recording datasets in 2015, 2017, 2019 and 2021 (Figure 1.5). NMDS ordinations emphasise the strong differentiation not only between the BAA 1 and BAA 2 frog communities but also between frog communities in each of the two elevational zones within each BAA.

Overall, the frog communities documented on transects in the BAAs are strongly influenced by elevation, with total diversity in each altitudinal band decreasing with increasing elevation (except for the similar totals at the two lowest elevations), and with high species turnover among elevation bands. This reflects each species’ particular climatic and microhabitat requirements.

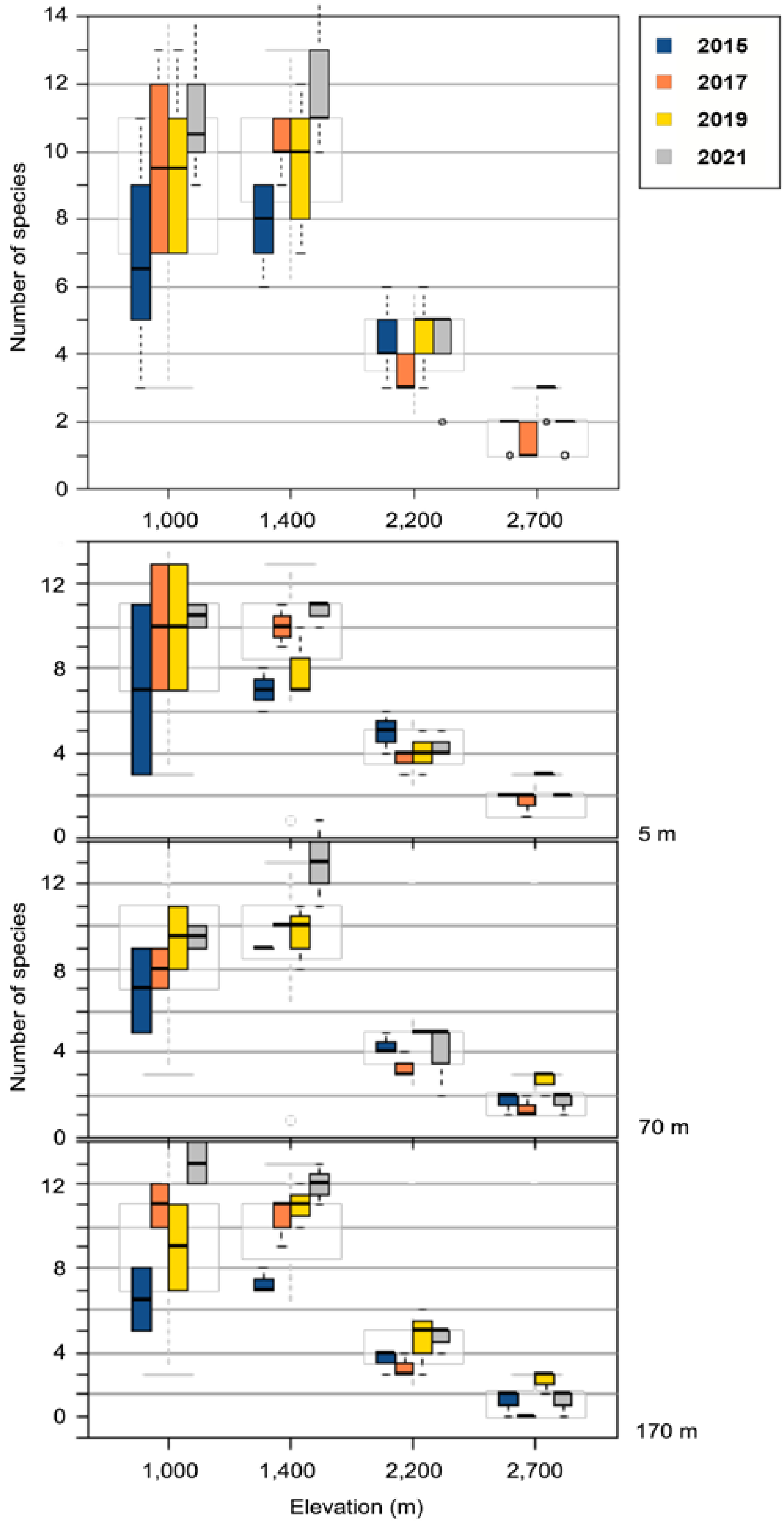


Figure 1.4. Summary of frog diversity (as number of species, or richness) at different elevations based on acoustic recordings in 2015, 2017, 2019 and 2021. For the uppermost plot data are pooled across all distances from linear infrastructure. See Figure 1.2 for explanation of boxplots.

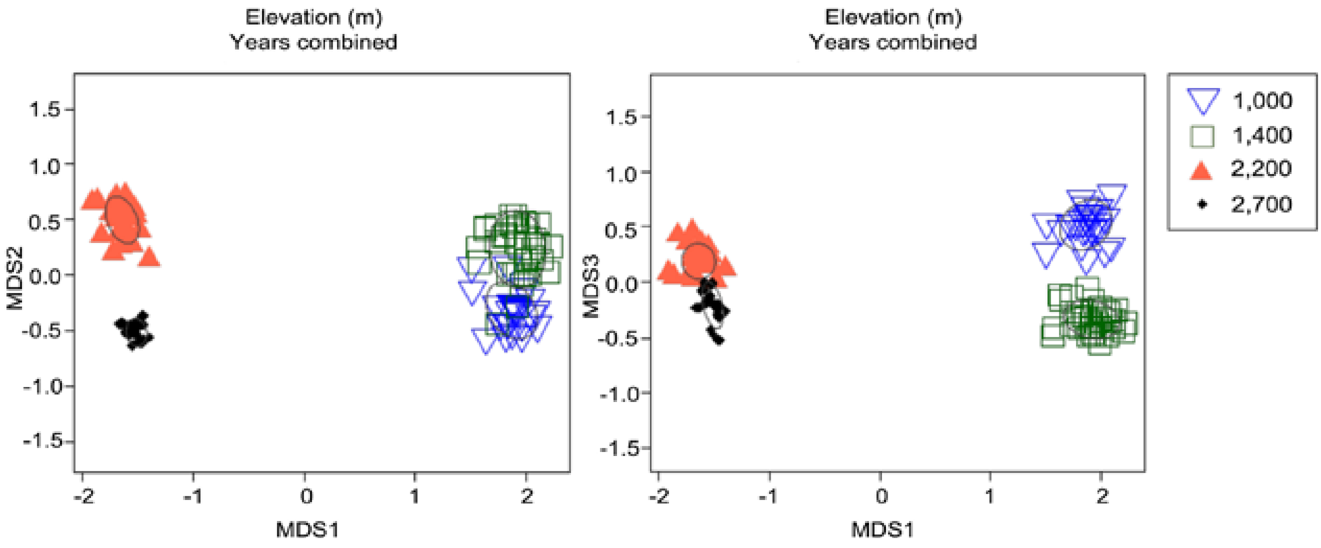


Figure 1.5. Non-metric Multi-dimensional Scaling (NMDS) ordinations summarising patterns of species composition at different elevations within the BAAs in 2015, 2017, 2019 and 2021 (data ellipses are one standard deviation).

Indicator species

The Indicator Species analysis did not identify any species of frog that was associated significantly with any of the distance categories from the ROW. This is consistent with the outcomes from the statistical test that did not detect any difference of species richness with increasing distance from the ROW. The 2021 data demonstrated that two species identified as possible 'Indicator Species' based on 2015 data, *Choerophryne burtoni* and *Liophryne sclaginhaufeni*, are unlikely to be useful indicators due to low encounter rates.

There was however, a list of frog species associated with each of the elevation categories, which is not surprising given the strong difference in species composition and diversity amongst them. These associations are obvious when presence/absence at different elevation categories is listed alongside the three metrics. Most frog species are found at only one or two elevation categories, which accounts for the high values of the A metric. The difference in the Indicator Species metric I is based largely on how common a particular species is. For example, *Oreophryne nicolasi* has only been documented at 1,000 m asl, but its value as an Indicator Species (I = 0.46) is lower than that of *O. oviprotector* (I = 0.96) because it has a much lower B metric.

Table 1.5. Indicator Species index (I; range 0 – 1), with a higher value indicating a greater value as an indicator species for that distance from the ROW or elevation; A is the specificity of the species for that particular site category, and B is the probability of encountering it.

	Total records	1,000	1,400	2,200	2,700	Group	I	A	B	P, sig
Austrochaperina fulva	18	X				1000	0.87	1	0.75	0.001***
Choerophryne crucifer	2	X				1000	0.29	1	0.08	0.032*
Copiula bisyllaba	11	X				1000	0.68	1	0.46	0.001***
Oreophryne nicolasi	5	X				1000	0.46	1	0.21	0.001***
Oreophryne oviprotector	22	X				1000	0.96	1	0.92	0.001***
Sphenophryne cornuta	11	X				1000	0.68	1	0.46	0.001***
Xenorhina lacrimosa	9	X				1000	0.61	1	0.38	0.001***
Litoria sp. 1 'yellow legs'	23	X	X							
Asterophrys slateri	25	X	X			1000	0.81	0.83	0.79	0.001***
Callulops omnistriatus	36	X	X							
Choerophryne alainduboisi	51	X	X							
Hylophorbus sp. 2 'fast call'	42	X	X							
Oreophryne flavomaculata	59	X	X							
Oreophryne pseudunicolor	33	X	X			1000	0.80	0.72	0.88	0.001***
Oreophryne sp. cf. *notata*	49	X	X							
Austrochaperina laurae	18		X			1400	0.71	1	0.50	0.001***
Choerophryne burtoni	9		X			1400	0.50	1	0.25	0.001***
Choerophryne multisyllaba	21		X			1400	0.76	1	0.58	0.001***
Choerophryne murrita	11		(x)			1400	0.55	1	0.31	0.001***
Cophixalus cateae	9		X			1400	0.50	1	0.25	0.001***
Cophixalus wempi	9		X			1400	0.50	1	0.25	0.001***
Cophixalus sp. 1 'musical call'	29		X			1400	0.90	1	0.81	0.001***
Cophixalus brevidigitus	1		X							
Cophixalus melanogenys	27		X			1400	0.87	1	0.75	0.001***
Hylophorbus sp. 1 'slow call'	13		X			1400	0.60	1	0.36	0.001***
Oreophryne sp. 2 'ratchet call'	23		X			1400	0.80	1	0.64	0.001***
Liophryne schlaginhaufeni	10	X				1000	0.65	1	0.42	0.001***
Lechriodus aganoposis	1			(x)						
Litoria iris	3			X						
Choerophryne sp. 1 'arboreal'	34			X		2200	0.97	1	0.94	0.001***
Choerophryne sp. 2 'tiny'	15			X		2200	0.65	1	0.42	0.001***
Cophixalus sp. 4 'loud grunter'	36			X		2200	1.00	1	1.00	0.001***
Hylophorbus richardsi	5			X		2200	0.37	1	0.14	0.007**
Oreophryne anamiatoi	10			X		2200	0.53	1	0.28	0.001***
Oreophryne notata	72			X	X					
Callulops wilhelmanus	16				X	2700	0.63	0.94	0.42	0.001***
Choerophryne brevicrus	27				X					
Litoria sp. cf. *becki*	—				VAES					
Xenorhina sp. 2 'fast call'	—		VAES							
Gen. nov. sp. nov.	—	VAES								
Total Richness	**37**	**17**	**20**	**9**	**4**					

The Indicator Species metrics might have greater applicability in the future if habitats start to change. Under a theoretical scenario where changes in overall patterns of diversity might be subtle and masked by sampling or natural variation, a change in the relative abundance or detectability of one particular species showing a markedly different response could become obvious. For example, if the edge habitat changed in structure, and one species of frog increased its abundance close to the ROW, it would show an increase in both the I and B metrics. In the analysis conducted here, all four survey years were combined, but in Richards et al. (2021) a separate Indicator Species metric was calculated for each year. There were no significant patterns with distance from the ROW for any year, but if overall diversity was shown to differ significantly in future surveys, a break-down by year would allow further interpretation.

Significant species and taxonomic uncertainties

Taxonomic studies since the 2019 survey have resulted in the formal descriptions of an additional three frog species occurring in BAA 2 that were undescribed at that time (*Cophixalus brevidigitus*, *C. melanogenys* and *Xenorhina lacrimosa*) (Günther and Richards 2021a,b). A summary of the taxonomic changes that have been incorporated into species tabulations in this report is presented in Appendix 1.2.

The identity of several small arboreal frogs of the genus *Oreophryne* with 'rattling' calls in BAA 2 remains uncertain. These are tree-dwelling frogs that are rarely seen calling and are difficult to capture for DNA sampling. DNA barcoding indicates that at least two species occur at KP107 (*Oreophryne flavomaculata* and 1–2 unidentified species). Additional DNA samples, preferably associated with vouchered calls, are needed to resolve this issue. It was not possible to obtain these data in 2021 due to restrictions on night work, so for the purposes of integrating our 2021 data with that from previous surveys (2015–2019), we again combined all unidentified rattling calls at KP107 and Arakubi with *O. flavomaculata* for the current analyses. However, it is likely that with the availability of additional DNA samples in future it will be demonstrated that *O. flavomaculata* does not occur at Arakubi, and it will be possible to re-analyse data from the four putative 'rattling *Oreophryne* species' at KP107 separately. To this end we recommend that VAES surveys be reinstated in 2023 to allow collection of new genetic and acoustic data that will help to resolve the issues surrounding this group of frogs.

Species of conservation significance (IUCN-Listed)

No species of frogs known from either BAA currently has an IUCN Red List conservation status higher than Least Concern. Three species described recently from the Upstream Project Area (Günther and Richards 2021a,b) have yet to be assessed by the IUCN.

Frogs at Wellpad D on Hides Ridge

Calls of *Litoria iris* were present on all of the ten 1-hour time blocks of BAR recordings over the two days that the BAR units were recording calls at Wellpad D, and > 10 egg masses were detected hanging on vegetation during the day. The Hides Ridge population of this species appears to be secure.

No calls that could be attributed to *Litoria vivissimia*, a species known only from a single adult collected at this pond in 2005 (Oliver et al. 2019) were detected on the BAR recordings.

Conclusions

1. The forests at Hides Ridge in BAA 1 and on the Agogo Range near Moro in BAA 2 continue to support a high diversity of frog species. Our results suggest that no major declines or losses have been experienced within these communities.

2. Three species of undescribed frogs that had been recorded from the two BAAs during previous surveys have been formally described since 2019.

3. Quantitative surveys of frog communities at different elevations within the two BAAs confirm differences in community composition at different elevations, and a statistically significant effect of decreasing species diversity with increasing elevation.

4. However, statistical analysis comparing the diversity of frog communities at different distances from linear infrastructure across four surveys (2015–2021) found no evidence for shifts with increasing distance from the forest edge.

5. Overall, the results from the third monitoring survey suggest that, in relation to frogs, the biodiversity values of the Upstream Project Area have been retained to date.

Recommendations

1. This survey provided quantitative data that are suitable for long-term documentation of frog communities in BAA 1 and BAA 2 and we recommend that frog monitoring be continued biennially for the duration of the PMA3 program.

2. Although the Audio Recorders produced statistically robust data during the 2021 survey, we recommend that the use of VAES transects be reinstated for the 2023 survey as long as logistical difficulties can be addressed because these transects generate valuable data on poorly known species that cannot be obtained using only Acoustic Recorders.

3. Given that there is now both an excellent understanding of frog diversity in the study area, and a large resource of acoustic recordings, there is opportunity to streamline aspects of the acoustic analysis by developing a semi-automated identification system. The application of 'deep learning' methods is likely to significantly increase efficiency for both the static sites survey and the VAES transect components of the survey, but needs resourcing to develop.

References

Anstis, M., Parker, F., Hawkes, T., Morris, I. & Richards, S. J. 2011. Direct development in some Australopapuan microhylid frogs of the genera *Austrochaperina*, *Cophixalus* and *Oreophryne* (Anura: Microhylidae) from northern Australia and Papua New Guinea. *Zootaxa* 3052: 1–50.

De Cáceres, M., & Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. *Ecology* 90: 3566–3574.

Dufrêne, M. & Legendre, P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs* 67: 345–366.

ExxonMobil 2016. PMA 3. Biodiversity Surveys Protocol (PGGP-EH-SPPRO-008002-003 Rev.1).

Günther, R. & Richards, S. J. 2021. Three new tiny species of the microhylid frog genus *Cophixalus* Boettger, 1892 from central Papua New Guinea. *Proceedings of the Biological Society of Washington* 34: 339–362.

Günther, R. & Richards, S. J. 2021. Description of six new species of *Xenorhina* Peters, 1863 from southern Papua New Guinea (Amphibia, Anura, Microhylidae). *Zoosystematics and Evolution* 97: 355–382.

Menzies, J. I. 2006. *The frogs of New Guinea and the Solomon Islands*. Pensoft, Sofia-Moscow.

Oliver, P., Richards, S. J. and Donnellan, S. C. 2019. Two new species of treefrog (Pelodrydidae: *Litoria*) from southern New Guinea elucidated by DNA barcoding. *Zootaxa* 4609: 469–484.

Richards, S. J. & Armstrong, K. 2017. Chapter 2 – Frogs. Pp 53–90 *In*: Richards, S. J. (Editor). *Biodiversity assessment of forests in the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. Port Moresby, ExxonMobil PNG.

Richards, S. J., Armstrong, K. & Dahl, C. 2019. Chapter 2 – Frogs. Pp 33–61 *In*: Richards, S. J. (Editor). *Results of the second PMA3 biodiversity monitoring survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. Port Moresby, ExxonMobil PNG.

Richards, S. J. & Dahl, C. 2011. Herpetofauna of the Strickland Basin and Muller Range, Papua New Guinea. Pp 190–197 *In*: Richards, S. J. & Gamui, B. G. (Editors). *Rapid Biological Assessments of the Nakanai Mountains and the upper Strickland Basin: surveying the biodiversity of Papua New Guinea's sublime karst environments. RAP Bulletin of Biological Assessment 60*. Conservation International, Arlington, VA.

Tallowin, O., Allison, A., Algar, A. C., Kraus, F. & Meiri, S. 2017. Papua New Guinea terrestrial-vertebrate richness: elevation matters most for all except reptiles. *Journal of Biogeography* 44: 1734–1744.

The frog team setting up a BAR recorder in preparation for two night's recording of frog calls (photo by Kalyna Taule).

Appendix 1.1. Frog recording site locations in BAA 1 on Hides Ridge and BAA 2 on the Agogo Range near Moro. Coordinates in WGS84 datum.

Elevation category	Transect	Site	Latitude	Longitude	Elevation (m asl)
Arakubi	M4	M4_005	S6.462013	E143.256616	1,017
		M4_070	S6.461926	E143.256018	1,030
		M4_170	S6.461667	E143.255006	1,041
	M5	M5_005	S6.461944	E143.250132	1,052
		M5_070	S6.462124	E143.250560	1,057
		M5_170	S6.461528	E143.251531	1,056
KP 107	M1	M1_005	S6.440230	E143.224085	1,403
		M1_070	S6.440240	E143.223590	1,398
		M1_170	S6.440079	E143.222562	1,408
	M2	M2_005	S6.440718	E143.225566	1,395
		M2_070	S6.441409	E143.225425	1,378
		M2_170	S6.442099	E143.224895	1,391
	M3	M3_005	S6.441778	E143.227103	1,379
		M3_070	S6.442142	E143.226678	1,375
		M3_170	S6.443061	E143.226314	1,392
Hides Low	H1	H1_005	S5.972520	E142.753279	2,163
		H1_070	S5.972856	E142.752890	2,155
		H1_170	S5.973729	E142.752471	2,151
	H2	H2_005	S5.969087	E142.751274	2,167
		H2_070	S5.969068	E142.750669	2,187
		H2_170	S5.969126	E142.749804	2,217
	H3	H3_005	S5.943807	E142.741784	2,289
		H3_070	S5.944572	E142.741865	2,284
		H3_170	S5.945233	E142.741622	2,322
Hides High	H4	H4_005	S5.918423	E142.695320	2,695
		H4_070	S5.919144	E142.694951	2,702
		H4_170	S5.919827	E142.694924	2,692
	H5	H5_005	S5.916343	E142.692853	2,751
		H5_070	S5.916471	E142.692311	2,749
		H5_170	S5.916749	E142.691230	2,731
	H6	H6_005	S5.913796	E142.690169	2,733
		H6_070	S5.914176	E142.689647	2,737
		H6_170	S5.914911	E142.688983	2,729

Appendix 1.2. Summary of taxonomic changes to frog names since 2015 that are incorporated into this report.

2015	2017	2019	2021
HYLIDAE	**PELODRYADIDAE**	**PELODRYADIDAE**	**PELODRYADIDAE**
Litoria iris	*Litoria iris*	*Litoria iris*	*Litoria iris*
Litoria sp. 1 'yellow-legs'	*Litoria* sp. 1 'yellow legs'	*Litoria* sp. 1 *'yellowlegs'*	*Litoria* sp. 1 'yellow legs'
not detected in 2015	not detected in 2017	*Litoria* sp. cf. *becki*	not detected in 2021
LIMNODYNASTIDAE	**LIMNODYNASTIDAE**	**LIMNODYNASTIDAE**	**LIMNODYNASTIDAE**
Lechriodus aganoposis	*Lechriodus aganoposis*	*Lechriodus aganoposis*	*Lechriodus aganoposis*
MICROHYLIDAE	**MICROHYLIDAE**	**MICROHYLIDAE**	**MICROHYLIDAE**
Metamagnusia slateri	*Asterophrys slateri*	*Asterophrys slateri*	*Asterophrys slateri*
Austrochaperina sp. 2 'long call'	*Austrochaperina* sp. 2 'long call'	*Austrochaperina fulva*	*Austrochaperina fulva*
Austrochaperina sp. 1 'short call'	*Austrochaperina laurae*	*Austrochaperina laurae*	*Austrochaperina laurae*
Callulops sp.	*Callulops omnistriatus*	*Callulops omnistriatus*	*Callulops omnistriatus*
Callulops wilhelmanus	*Callulops wilhelmanus*	*Callulops wilhelmanus*	*Callulops wilhelmanus*
Choerophryne sp. 3 'buzz call'	*Choerophryne alainduboisi*	*Choerophryne alainduboisi*	*Choerophryne alainduboisi*
Choerophryne brevicrus	*Choerophryne brevicrus*	*Choerophryne brevicrus*	*Choerophryne brevicrus*
Choerophryne burtoni	*Choerophryne burtoni*	*Choerophryne burtoni*	*Choerophryne burtoni*
Choerophryne sp. 5 'lowland clicker'	*Choerophryne crucifer*	*Choerophryne crucifer*	*Choerophryne crucifer*
Choerophryne sp. 4 'montane clicker'	*Choerophryne multisyllaba*	*Choerophryne multisyllaba*	*Choerophryne multisyllaba*
Choerophryne murrita	*Choerophryne murrita*	*Choerophryne murrita*	*Choerophryne murrita*
Choerophryne sp. 1 'arboreal'	*Choerophryne* sp. 1 'arboreal'	*Choerophryne* sp. 1 'arboreal'	*Choerophryne* sp. 1 'arboreal'
Choerophryne sp.2 'tiny'	*Choerophryne* sp. 2 'tiny'	*Choerophryne* sp. 2 'tiny'	*Choerophryne* sp. 2 'tiny'
Cophixalus sp. 5'peeping call'	*Cophixalus* sp. 5 'peeping call'	*Cophixalus cateae*	*Cophixalus cateae*
Cophixalus wempi	*Cophixalus wempi*	*Cophixalus wempi*	*Cophixalus wempi*
Cophixalus sp. 1 'musical call'	*Cophixalus* sp. 1 'musical call'	*Cophixalus* sp. 1 'musicalcall'	*Cophixalus* sp. 1 'musical call'
Cophixalus sp. 2 'tiny A'	*Cophixalus* sp. 2 'tiny'	*Cophixalus* sp. 2 'tiny A'	*Cophixalus brevidigitus*
Cophixalus sp. 3 'tiny B'	*Cophixalus* sp. 3 'tiny B'	*Cophixalus* sp. 3 'tiny B'	*Cophixalus melanogenys*
Oreophryne? sp. 5 'loud grunter'	*Cophixalus* sp. 6 'loud grunter'	*Cophixalus* sp. 5 'loud grunter'	*Cophixalus* sp. 4 'loud grunter'
*Copiula*sp. 1 '2-note call'	*Copiula* sp. '2-note call'	*Copiula bisyllaba*	*Copiula bisyllaba*
not detected in 2015	*Hylophorbus richardsi*	*Hylophorbus richardsi*	*Hylophorbus richardsi*
Hylophorbus sp. 1 'small'	*Hylophorbus* sp. 1 'slow call'	*Hylophorbus* sp. 1 'slow call'	*Hylophorbus* sp. 1 'slow call'
*Hylophorbus*sp. 2 'large'	*Hylophorbus* sp. 2 'fast call'	*Hylophorbus* sp. 2 'fast call'	*Hylophorbus* sp. 2 'fast call'
Liophryne schlaginhaufeni	*Liophryne schlaginhaufeni*	*Liophryne schlaginhaufeni*	*Liophryne schlaginhaufeni*
Oreophryne anamiatoi	*Oreophryne anamiatoi*	*Oreophryne anamiatoi*	*Oreophryne anamiatoi*
Oreophryne sp. 4 'yellow-spots'	*Oreophryne flavomaculata*	*Oreophryne flavomaculata*	*Oreophryne flavomaculata*

2015	2017	2019	2021
Cophixalus sp. 4'rasping call'	*Oreophryne* sp. 6 'rasping call'	*Oreophryne nicolasi*	*Oreophryne nicolasi*
Oreophryne notata	*Oreophryne notata*	*Oreophryne notata*	*Oreophryne notata*
not distinguished in 2015	not distinguished in 2017	*Oreophryne* sp. cf. *notata*	*Oreophryne* sp. cf. *notata*
Oreophryne oviprotector	*Oreophryne oviprotector*	*Oreophryne oviprotector*	*Oreophryne oviprotector*
Oreophryne sp. 3 'slow peeper'	*Oreophryne pseudunicolor*	*Oreophryne pseudunicolor*	*Oreophryne pseudunicolor*
Oreophryne sp. 1 'tiny'	removed	removed	removed
Oreophryne sp. 2 'ratchet call'	*Oreophryne* sp. 2 'ratchet call'	*Oreophryne* sp. 2 'ratchet call'	*Oreophryne* sp. 2 'ratchet call'
Sphenophryne cornuta	*Sphenophryne cornuta*	*Sphenophryne cornuta*	*Sphenophryne cornuta*
Xenorhina sp.	*Xenorhina* sp. 1 'slow call'	*Xenorhina* sp. 1 'slow call'	*Xenorhina lacrimosa*
not distinguished in 2015	*Xenorhina* sp. 2 'fast call'	*Xenorhina* sp. 2 'fast call'	*Xenorhina* sp. 2 'fast call'
Gen. nov. sp. nov.	Gen. nov. sp. nov.	Gen. nov. sp. nov.	Gen. nov. sp. nov.

Chapter 2 – Camera trap monitoring of terrestrial mammals and birds

Iain A. Woxvold, Banak Gamui and Salape Tulai

Salape Tulai setting up a camera trap (photo by Kalyna Taule)

Summary

Background and aims

Terrestrial (ground-dwelling) mammals and birds are suitable for monitoring because they include a variety of species that are targeted by hunters, are sensitive to forest disturbance or to invasive species impacts, or are otherwise indicative of ecosystem health. Wildlife most at risk in Papua New Guinea (PNG) include a variety of medium- to large-bodied species such as echidnas, wallabies, tree kangaroos and cassowaries, many of which are listed by the IUCN as Threatened or Near Threatened with extinction. However, monitoring terrestrial species presents a challenge as they often occur at naturally low densities and are difficult to detect due to their avoidance of humans.

Camera traps are increasingly used to monitor terrestrial wildlife populations, as they are effective at detecting rare and elusive species and they provide standardised datasets that can be used to address a variety of ecological questions. Here we present the results of repeat camera trap surveys undertaken in two Biodiversity Assessment Areas (BAAs) in upland sectors of the PNG LNG Upstream Project Area in 2017, 2019 and 2021. In each sampling year, 80 camera traps were deployed across four sites (two sites in each BAA) for a period of 89–135 days. Objectives of the camera trap monitoring study are:

1. To improve our understanding of vertebrate diversity in sampling areas.

2. To monitor trends in resident wildlife populations over time.

3. To examine habitat preferences of wildlife in relation to proximity to Project infrastructure ('edge effects').

Major results

Objective 1—To date, more than 100 vertebrate species have been documented in 17,834 independent photographic events recorded over 23,178 sampling days. Results of the 2021 survey include two species newly reported from the BAAs – Masked Ring-tailed Possum (*Pseudochirulus larvatus*) at BAA 1 and Torresian Striped Possum (*Dactylopsila trivirgata*) at BAA 2. Biennial data are provided for seven IUCN listed species, including: four Threatened species – the Eastern Long-beaked Echidna (*Zaglossus bartoni*), Pademelon (*Thylogale* sp.), Ifola (*Dendrolagus notatus*) and Goodfellow's Tree Kangaroo (*D. goodfellowi*); two Near Threatened species – New Guinea Quoll (*Dasyurus albopunctatus*) and Small Dorcopsis (*Dorcopsulus vanheurni*); and one Data Deficient species – Woolley's Three-striped Dasyure (*Myoictis leucura*). The Small Dorcopsis was the most frequently camera trapped of all species with 1,126 photographic events recorded in 2021.

Objective 2— We used three types of surrogate population estimates to monitor population trends between years (based on data availability, and in decreasing of order of preference) – occupancy estimates, activity rates and naïve occupancy rates. Goodness of fit test results precluded modelling of occupancy estimates in all but one species. Compared to 2017 estimates, significant declines in activity rates were observed for four mammal and six bird species at individual sites. In addition, notable declines in naïve occupancy (proportion of cameras detecting a species) have been recorded for three hunting-sensitive IUCN Threatened mammals at BAA 2. Following a sharp decline in records of the IUCN Vulnerable Eastern Long-beaked Echidna and Pademelon at Arakubi in 2019, neither species was again recorded there in 2021. In 2021, novel declines were also recorded for the IUCN Endangered Goodfellow's Tree Kangaroo at Arakubi (first year with no records) and for the Pademelon at KP107 (sharp decline to <10% of previous annual records). By contrast, at Hides High in BAA 1, naïve occupancy of the Eastern Long-beaked Echidna increased sharply in 2021 and that of the Ifola has risen steadily from one camera (5%) in 2017 to three cameras (16%) in 2021.

The 2021 rate of daily incursions by humans and/or dogs decreased at all sites compared with 2019 levels. At BAA 1, 2021 incursion rates were also lower than those recorded there in 2017. By contrast, at the BAA 2 sites, both the 2019 and the 2021 incursion rates were higher than those recorded there in 2017. At Arakubi, where the decline in records of IUCN listed species has been strongest, most incursions have been recorded closer to the PNG LNG pipeline ROW than to the Arakubi Quarry and its access road (2019 – 13/16 incursion days; 2021 – 12/17 incursion days), suggesting that most hunting parties are accessing the area via the pipeline route.

Objective 3—Animal activity rate was correlated with distance from infrastructure in 16 species at one or more sites. Edge avoidance patterns were demonstrated by 10 species, being strongest at BAA 2 sites for Raffray's Bandicoot (*Peroryctes raffrayana*), Small Dorcopsis, Collared Brushturkey (*Talegalla jobiensis*), New Guinea Scrubfowl (*Megapodius decollatus*), Pheasant Pigeon (*Otidiphaps nobilis*) and Russet-tailed Thrush (*Zoothera heinei*). Seven species displayed reverse-pattern edge effects, with higher rates of activity nearer to the forest edge. Reverse edge effects were most common at BAA 1 and were the only patterns observed at the Hides High site.

Conclusions and recommendations

The value of camera traps in detecting rare and elusive species was reinforced during the third full season of camera trap sampling. Deployment time is sufficient to attain a near-complete census of the resident terrestrial bird and medium- to large-bodied mammal faunas in each sampling year.

The 2017–2021 dataset provides a useful baseline against which to measure future population trends. Despite notable changes in population estimates for some species, it is too early to draw conclusions as to the ongoing status of local populations because: (1) data from additional sampling years are required to improve inferential power in light of natural fluctuations in animal populations, and; (2) sampling commenced in different months each year, so that seasonal effects may account for some of the observed changes.

The 2021 data consolidate most of the edge-response patterns observed in 2017 and 2019 and extend them to additional species. Data collected on environmental covariates in 2019 provide for strong model performance and improve our ability to make reliable inferences about behavioural responses to Project infrastructure. Edge avoidance was most clearly demonstrated at BAA 2, where some species may avoid near-edge environments due to the presence of degraded forest near infrastructure (particularly at Arakubi) and/or display an aversion to frequent human activity along roads and the pipeline ROW. The reverse-pattern edge effects commonly observed at BAA 1 are counter-intuitive for interior forest species. Causal factors are likely to be environmental rather than anthropogenic. Regardless of the cause, after three years of camera trap monitoring there is little evidence of forest edge avoidance on Hides Ridge.

We recommend that the camera trapping program continue in 2023, and in subsequent years, because it provides a reliable method for detecting changes in population estimates of multiple rare, elusive and hunting sensitive species. However, as far as practical, surveys should take place at the same time of year to control for seasonal effects. The biennial schedule should be maintained as fewer data points will extend the time before which reasonable inferences can be made about population trends.

The current sampling design – based on the clustering of 20 cameras within sites of c. 70–180 ha – is suitable for modelling edge effects but limits our ability to monitor wildlife population trends because: (1) it is not possible to draw conclusions as to the status of populations beyond the site scale, and; (2) the close spacing of many cameras constrains our ability to use the preferred occupancy modelling approach. Future datasets are likely to yield diminishing returns in terms of revealing novel edge response patterns, particularly on Hides Ridge (BAA 1). We therefore recommend that consideration be given to expanding the sampling design (using the same number of cameras) within BAAs beyond the site scale, as far as practical, to increase sampling of important local populations and to improve the scope for occupancy modelling. Details are provided regarding possible expansion scenarios.

Introduction

Tropical forest faunas include many ground-dwelling ('terrestrial') species that are vulnerable to anthropogenic change. Population declines have been observed in terrestrial mammals and birds in response to habitat fragmentation and degradation (Michalski and Peres 2007; Burivalova et al. 2014), over-hunting (Bennett and Robinson 2000; Peres and Palacios 2007) and the presence of invasive species such as feral dogs (Cassano et al. 2014; Lessa et al. 2016). Many medium- to large-bodied species are specifically targeted by hunters (Jerozolimski and Peres 2003; Sampaio et al. 2010), and those with low population densities and slow life histories are often the most susceptible to over-harvesting (Bodmer 1995; Peres and Palacios 2007) and to habitat change (Michalski and Peres 2007; Costantini et al. 2016). Although not specifically targeted by hunters, a variety of smaller terrestrial birds are also known to be sensitive to the fragmentation and degradation of tropical forest habitat (Thiollay 1997; Lambert and Collar 2002; Peh et al. 2005).

New Guinea's upland forests support a rich assemblage of endemic terrestrial mammals and birds. Many of the larger species have suffered population declines as a result of hunting and habitat loss (e.g., Johnson et al. 2004; Eldridge and Coulson 2015; Nicol 2015) and are listed as Threatened or Near Threatened on the IUCN *Red List of Threatened Species* (IUCN 2022). However, population status varies markedly at the local scale (species extirpated near some settlements may be fairly common in remote areas) and the response of various species to individual stressors remains poorly known.

Given their susceptibilities, terrestrial species serve as useful surrogate indicators of ecosystem health and connectivity (e.g., Crooks et al. 2011; Peters et al. 2015) and present suitable monitoring targets in the vicinity of anthropogenic development sites. Yet collecting sufficient data to monitor populations presents a challenge – animals are often difficult to census in tropical forest environments (Ahumada et al. 2007) and many terrestrial species are rare or elusive (O'Brien and Kinnaird 2008).

Camera traps are increasingly used as an efficient and non-invasive tool in the study of terrestrial wildlife populations (Burton et al. 2015). Able to run continuously for long periods without human intervention, they are effective at detecting rare and elusive species (e.g., Dinata et al. 2008; Beirne et al. 2017; Thomas et al. 2020) and they provide standardised datasets that can be used to address a variety of ecological questions – among their many applications, camera trap studies have been used to estimate species richness and abundance (O'Connell et al. 2011; Rovero et al. 2014; Li et al. 2018), to examine habitat preferences (Pettorelli et al. 2010; Martin et al. 2015; Zimbres et al. 2017), to monitor population trends over time (Blake et al. 2017; Beaudrot et al. 2019; O'Brien et al. 2019), and to investigate the impacts of roads and forest edges (Srbek-Araujo and Chiarello 2013; Rovero et al. 2017), feral animals (Murphy et al. 2017, 2018) and hunting and disturbance (Rao et al. 2005; Hegerl et al. 2015; Oberosler et al. 2017; Pardo et al. 2021).

Following a pilot study in 2015 (Woxvold and Aplin 2017), in 2017 a biennial camera trap monitoring program was initiated within two Biodiversity Assessment Areas (BAAs; see Report Summary) in upland sectors of the PNG LNG Upstream Project Area (Woxvold and Legra 2019a). Camera trap arrays were deployed in the vicinity of Project infrastructure to meet three main objectives:

1. To improve our understanding of vertebrate diversity present in sampling areas.

2. To monitor trends in resident wildlife populations over time.

3. To examine habitat preferences of wildlife in relation to proximity to Project infrastructure ('edge effects').

This report integrates and interprets the results of the first three camera trap monitoring surveys undertaken in 2017, 2019 and 2021.

Methods

Study sites

Camera traps were deployed at two sites within each of the two BAAs (Figures 3–4 & 6–7 of Report Summary):

- BAA 1 – at 'Hides Low' immediately northwest of Wellpad D, and at 'Hides High' between Wellpad E and Wellpad G.
- BAA 2 – at 'Arakubi' around Arakubi Quarry and east of the pipeline ROW, and at 'KP107' in the vicinity of kilometre-point 107 along the pipeline ROW.

The four sites are positioned along an elevational gradient, with camera positions spanning an overall altitudinal range of more than 1,800 m (922–2,731 m; Table 2.1). All sites are located on polygonal limestone karst terrain, with dolines and valleys separated by an anastomosing network of undulating ridgelines. The overlying vegetation is described in detail in Venter and Ona (2017). Rainfall throughout the region is continuously heavy (little seasonality: McAlpine et al. 1983), averaging approximately 4 m per year at BAA 1 and more than 4 m per year at BAA 2 (Bryan and Shearman 2008). Despite the high rainfall, no watercourses or wetlands are present at the surveyed sites due to the porous limestone substrate.

Sampling design and effort

Twenty white-flash digital camera traps (Reconyx PC850 & HP2W) were deployed at each site in each sampling year. To test for possible edge effects (Objective 3), four cameras were positioned in each of five parallel 'bands' of increasing distance from the nearest clearing (0–50 m; 50–100 m; 100–200 m; 200–300 m; 300+ m). To maximise inference potential in population monitoring (Objective 2), cameras were placed in the same position in all years wherever possible. Fifteen positions used in 2017 were relocated in 2019 due to security concerns (Hides Low – seven cameras lost), or to site damage resulting from garden construction (Arakubi – 2) or from an earthquake in 2018 (Arakubi – 1, KP107 – 4, Hides High – 1). Three of the positions changed at Hides Low were again moved in 2021 following the loss of cameras in 2019. The position of functioning camera traps (see below for summary of losses and malfunctions) at each site/year is mapped in the Report Summary. Distance to the nearest camera within sites ranged from 25–365 m. Between sites, camera arrays were separated by a minimum distance of 4.5 km at BAA 1 (Hides Low–Hides High) and 2.7 km at BAA 2 (Arakubi–KP107).

Camera traps operated 24 hours/day, were programmed to maximum detection sensitivity and to take three photographs at each trigger with the minimum amount of rest time between triggers (<2 seconds). Each camera was fixed to a tree or purpose-cut wooden pole 15–25 cm above the ground and directed along an animal trail in an area of flat or gently sloping ground. Most positions were located on ridges/spurs or on hill-slope terraces; valley floors and gullies were avoided as these were often difficult to reach in the steep terrain and in order to minimise variation in detectability associated with local topographic effects. Site disturbance was restricted to removal of low vegetation (herbs, ferns, etc.) from 2–3 m directly in front of the camera. Camera positions were unbaited and fruiting trees were avoided to minimise the influence of natural attractants. Once set, camera traps were left to operate undisturbed until collection.

Sampling periods (first deployment to last collection) ran from 10 May to 30 August 2017 (112 days), from 10 August to 2 December 2019 (114 days), and from 4 July to 27 November 2021 (146 days). Table 2.1 summarises the trapping effort at each site/year.

Of 240 cameras deployed across all years, 15 (6.3%) yielded no data due to theft (11 cameras) or malfunction soon after deployment (four cameras). Most of the remaining cameras (2017 – 60/71; 2019 – 65/75; 2021 – 54/79) operated for the full deployment period of 89–135 days (per camera). Thirty cameras (2017 – 9; 2019 – 6; 2021 – 15) yielded partial datasets (reduced functioning period: 27–123 days) due to camera malfunction, camera movement or complete obstruction of view by fallen or growing vegetation.

In addition, at 18 cameras data quality during the functioning period was compromised by: (1) mud accumulation on lens and infrared detector covers (2017 – 3; 2019 – 5; 2021 – 4); (2) partial obstruction by shifting vegetation (2021 – 2); (3) partial malfunction (2021 – 1), or; (4) poor camera placement/orientation (2021 – 3). Four of these units were affected for the entire sampling period (mud accumulation – 1; poor orientation – 3). The remaining 14 affected cameras took clear images for periods of 26–95 days. Although these cameras continued to take some photographs, animal detection and subsequent identification rates were lower during the periods of reduced performance. Data collected during periods of reduced camera performance are included in the species summary totals for each site (Objective 1). However, because changes in detectability may influence the interpretation of animal behaviour patterns, data from these periods were excluded from activity rate and occupancy model analyses (Objectives 2 & 3). The overall trapping effort across all site/years was 23,178 camera days, of which 21,934 were included in statistical models.

Table 2.1. Altitudinal range and camera trapping effort at each site.

	BAA 2		**BAA 1**		**Total**
	Arakubi	**KP107**	**Hides Low**	**Hides High**	
Elevation (m asl)	922–1,054	1,297–1,398	2,192–2,389	2,645–2,731	
No. operating cameras					
2017	19	19	13	20	71
2019	19	20	17	19	75
2021	20	20	20	19	79
Mean (range) days/camera					
2017	91.0 (35–101)	95.2 (40–101)	91.2 (76–97)	91.2 (62–98)	92.2
2019	92.0 (38–98)	95.2 (93–98)	102.1 (84–107)	94.7 (48–103)	96.0
2021	124.9 (97–131)	105.5 (27–129)	122.1 (43–135)	126.1 (72–135)	119.6
Total camera days*					
2017	1,729	1,809	1,186	1,824	6,548
2019	1,748	1,903	1,735	1,799	7,185
2021	2,498	2,109	2,442	2,396	9,445

* Number of days operating, excluding deploy and collection dates and malfunction periods, but including periods of reduced detectability.

Analysis

Data preparation

Data organisation and all analyses were performed in R (R Development Core Team 2015).

All bird and most mammal images were identified by IW. Ken Aplin contributed to the identification of rodent and bandicoot images in 2017. Kristofer Helgen (Australian Museum) and Kevin Rowe (Museums Victoria) assisted with the identification of newly recorded taxa in 2019 and 2021. Photographs of uncertain identity were excluded from analysis. Images and associated metadata were managed for analysis using the 'camtrapR' package (Niedballa et al. 2016).

For all taxa, we calculated the number of independent photographic 'events' per camera. Within-camera events were considered independent where consecutive pictures of the same species were taken more than 60 minutes apart (Burton et al. 2015). Multiple events were scored within 60-minute periods where more than one individual was seen in a single photograph or sequence.

Relative rates of hunting pressure were estimated as the proportion of days in which humans and/or dogs were photographed at each site. In order not to over-estimate forest incursions by hunting parties, events for humans and dogs were limited to one record per site/day, regardless of the number of cameras on which they appeared in a single day. Anthropogenic events leading to the loss of cameras or garden construction were not included.

Terrestrial species are the most suitable for monitoring under the current approach, since lower image rates result from incidental trapping of predominantly arboreal species. Analyses addressing Objectives 2 and 3 were therefore limited to terrestrial taxa.

Covariates

In order to examine potential edge effects (Objective 3), we assessed the relationship between animal activity rates and distance from Project infrastructure while controlling for influential environmental covariates. Five measures of distance from Project infrastructure were assessed:

- Distance from the nearest clearing (road, pipeline ROW, wellpad, quarry or sidecast) – as a continuous measure (DClr) or as categorical measures comparing activity within distance classes of <50/>50 m (LT50) and <100/>100 m (LT100).

- Distance from the nearest road – as a continuous measure (DRd) or as a categorical measure comparing activity within <100/>100 m distance classes (LT100Rd).

We measured the following environmental covariates at each camera position. Terrain measures (local relief) – measured as the difference between the highest and lowest elevations present within 20 m, 50 m and 100 m of the camera (LR20, LR50, LR100); elevation data were taken from 5 m LiDAR Digital Elevation Model (DEM). Canopy height (CnpHt) – taken as the average height of the three tallest trees located within 30 m of the camera position, as measured by a Nikon Forestry Pro laser range finder. Remaining covariates were measured within a 20 m x 20 m plot established by arranging two 20 m cords perpendicular to one another in cross-hair fashion with the centre point at the camera position. Tree density – calculated as the number of trees >10 cm (TrSm) and >30 cm (TrLge) diameter at breast height (dbh) within the plot. Understorey density – measured using a 2 m pole with alternating colours in 10 cm segments and held vertically at the four ends of each of the two cords (10 m from the camera position); densities below 1 m (UD1) and below 2 m (UD2) were calculated by counting the number of wholly visible coloured segments in each height class. Leaf litter depth (LfLit) – measured with a ruler as the distance between the leaf litter surface and underlying soil or root mass, at the nearest three 2 m intervals along each of the four cord axes emanating from the camera position (12 measures at each camera position). Proportion of the plot ground surface covered by moss (Moss), rock (Rock) and large woody debris (>10 cm diameter) (Logs) – estimated by eye and ordered categorically as: 0 (none visible), 1 (0–2%), 2 (2–5%), 3 (5–10%), 4 (10–20%), 5 (20–35%), 6 (35–50%), 7 (50–65%), 8 (65–80%), 9 (>80%).

We tested for correlations among all covariates in order to avoid collinear terms appearing together in the same models. Collinearity among numeric variable-pairs was tested using the Pearson Correlation Coefficient (PCC). Correlations between ordinal (ordered categorical) terms, and between ordinal and numerical terms, were examined using Kendall's rank correlation coefficient. Where final models included three or more covariates, we tested for higher-order multicollinearity using the 'mctest' R package.

Activity rates

Photographic rates were used: (1) to monitor population trends by assessing changes in animal activity between years (Objective 2) in cases where occupancy modelling could not be applied (see below), and; (2) to test for edge effects by assessing habitat preferences in terms of spatial proximity to Project infrastructure (Objective 3). For each species of interest, an activity index was calculated from the daily number of events per camera.

Generalised Linear Mixed Models (GLMMs) were used to model activity rates of terrestrial species across their distribution (1–4 sites) where sufficient data were available (generally ≥16–20 events per site/year; models for taxa with fewer events often fail to converge: this study; Martin et al. 2015; Oberosler et al. 2017). Site and camera position were treated as random effects.

Objective 2—To assess changes in activity between years, the dataset was contracted to those camera positions that were sampled in all years (Arakubi – 15; KP107 – 16; Hides Low – 13; Hides High – 17) and was analysed using a negative binomial distribution. Environmental covariates were not included in these models as there was assumed to be no change between years in the conditions present at each position.

Objective 3—When assessing habitat preferences and edge effects, data were used from all camera positions at which the full complement of covariates was measured (Arakubi – 19, KP107 – 20, Hides Low – 17, Hides High – 20). Models were initially compared using the dredge function in 'MuMIn' (Bartoń 2015). We used a common set of models for each species as determined by all covariate combinations excluding collinear pairs. Models were ranked using Second-order Akaike Information Criterion (AIC_c) and their associated Akaike weights (w_i). Models that differ from the best-ranked model by an AIC_c value of less than two ($\Delta AIC_c<2$) are considered equally as good as the best model, while models with $\Delta AIC_c>6$ may be readily discounted (Richards 2005; Symonds and Moussalli 2011). Model averaging was used to assess the 'relative importance' (sum of w_i) of each covariate from a candidate set of models with $\Delta AIC_c<6$. Covariate coefficient point estimates and standard errors were generated using full model averaging (Grueber et al. 2011; Symonds and Moussalli 2011). Collinear distance and terrain measures compete with one another for weight among the best-ranked models. For each species-site/BAA, models were therefore ultimately compared using only the best performing distance and terrain measures, as initially determined by model averaging under the full covariate set. A 'Year*Distance measure' interactive term was introduced to explore possible changes in edge effects across years. Final models were constructed using the most influential covariates with back-removal of non-influential terms (at $P>0.10$). Because we frequently detected overdispersion, we routinely used a quasi-Poisson model structure that included an additional observation-level random term (camera-year).

For most species present at more than one site, activity rate at each site was influenced by different covariates. Most species' activity was therefore modelled at the site level. Because the Null model was often the best-ranked model for species recorded at relatively low frequencies (10–20 events/site-year), data were then pooled across sites within BAAs for seven species in an attempt to improve model performance.

Occupancy

'Occupancy' (ψ) refers to the proportion of an area or sites that is occupied by a species. 'Naïve occupancy' is a simple measure of the proportion of cameras at each site at which a species is detected.

Model-based occupancy is a more complex analytical procedure involving the estimation of occupancy across sampled positions while accounting for imperfect detection – the possibility that a species is present but remains undetected. This is achieved by creating capture histories for each sampling unit (e.g., camera) by dividing the sample 'season' into a series of sampling 'occasions'. The probability of detection (*p*) is then estimated and used to inform the occupancy measure. By accounting for variation in *p*, occupancy estimation is often considered to be a more reliable estimate of population density than are 'relative abundance indices' (RAIs) such as activity rate models based on photographic rates (Burton et al. 2015; Sollmann 2018).

Where the data permitted, we prioritised the use of multi-season occupancy modelling (MacKenzie et al. 2006) to monitor population trends between years using the package 'unmarked' (Fiske and Chandler 2011). We used the same dataset that was applied to comparing activity rates between years (positions sampled in all years). Capture histories for each monitoring year were created by registering presence/absence data (0 or 1) over repeat sampling occasions of 1 day (Rovero and Spitale 2016).

Occupancy modelling assumes that sampling stations are independent, such that individuals cannot be photographed at more than one camera station. To satisfy this condition, we grouped cameras into 'areas' whose nearest neighbouring cameras were at least 250 m apart (4–5 areas per site, 1–8 cameras per area). We then modelled occupancy trends at the area scale in a subset of species whose home range, based on published estimates, is considered unlikely to span more than one area (based on a circular home range of c. 5 hectares (ha)), including bandicoots (Dickman 2015) and birds with body mass less than c. 100 g (e.g., Jansen 1999; Lindsell 2001; Kattan and Beltran 2002). To ensure sufficient sample size, we further restricted our assessment to the BAA level, and thus to species that were present at both sites within a BAA.

Occupancy modelling was applied to two mammal and five bird species. We used a bootstrapped goodness of fit test with 1,000 simulations to examine the data-model relationship. Acceptable goodness of fit scores (Chi-squared values of 0.1–0.9) were obtained for only one species – the Striped Bandicoot (*Microperoryctes longicauda*). Activity rate models were applied to other candidate species.

P-values were not used to test the significance of change in occupancy between years. Rather, a change in occupancy estimate was considered 'significant' where both the lower and upper bounds of the estimated 'growth' rate's 95% confidence intervals had the same sign (positive or negative).

Conventions

Taxonomic order and nomenclature follow the *Handbook of the Mammals of the World* (Wilson and Mittermeier 2015; Wilson et al. 2017) in most cases and the *IOC World Bird List* (Gill et al. 2022). An exception is made for the local subspecies of Doria's Tree Kangaroo (*Dendrolagus dorianus notatus*), which is assessed separately by the IUCN as the Ifola (*D. notatus*) and classified as Endangered (IUCN 2022). Invasive alien mammals appear after native species in tabulated lists.

Where species are referred to in the text, the scientific name appears with the English name on first mention. For species whose identity and taxonomy are certain, only the English name is used in the text thereafter. Scientific names are used in the text for species whose identity or taxonomy are not well known; for example, because photographs are insufficient to identify an animal to species level or where their relationship with closely related taxa is still under investigation. The scientific name appears with the English name in photographs, tables and appendices.

Results

Recorded diversity (Objective 1)

A total of 17,834 independent photographic events have been recorded over a sampling period of 23,178 camera days (all years). More than 100 vertebrate species have now been documented by camera trap within the BAAs, including 62 bird species, more than 37 mammal taxa and one reptile. All taxa photographed under the full sampling program are listed in Appendix 2.1 along with their conservation status and the number of independent photographic events recorded at each site. A selection of taxa photographed in 2021 is shown in Figures 2.8–2.43. The total number of species photographed is considered to be more than 100 since a number of the mammal taxa are only identifiable to genus level in the images; for example, multiple species of *Murexia*, *Paramelomys* and *Rattus* are known to occur within the study area (Aplin and Opiang 2017), and given changes in community structure with elevation, more than one species in each of these genera is believed to have been photographed.

Two mammal and three bird species were newly camera trapped in 2021 (cf. 2017 and 2019: Table 2.2). All have been recorded previously within the Kikori Basin, and each of the birds has been recorded previously on Hides Ridge (BAA 1)

(Namo 2004; Crome 2008; Aplin and Opiang 2017; Woxvold and Legra 2017). However, both of the mammal species are newly reported from the BAAs in which they were photographed – the Masked Ring-tailed Possum (*Pseudochirulus larvatus*) at BAA 1 and the Torresian Striped Possum (*Dactylopsila trivirgata*) at BAA 2. Each of the newly camera trapped species is arboreal.

Bandicoot identifications were reappraised in 2021 and the distinctive Long-nosed Echymipera (*Echymipera rufescens*) was recognised for the first time. Although a new species for the BAA 2 list (Aplin and Opiang 2017), it was camera trapped at both Arakubi and at KP107 in all sampling years (Appendix 2.1).

Appendix 2.1 includes 57 taxa with predominantly or entirely terrestrial habits. Figures 2.1–2.2 show the total number of independent photographic events taken for each terrestrial mammal and bird taxon in each sampling year (raw figures not adjusted for variation in length of sampling season).

Table 2.2. Species newly camera trapped in 2019 and the number of photographic events at each site.

Species	BAA 2		BAA 1	
	Arakubi	KP107	Hides Low	Hides High
Mammals				
Masked Ring-tailed Possum (*Pseudochirulus larvatus*)			1	
Torresian Striped Possum (*Dactylopsila trivirgata*)		1		
Birds				
Mountain Owlet-nightjar (*Aegotheles albertisi*)			1	
Honeyeater (*Ptiloprora*) sp.				1
Brown-backed Whistler (*Pachycephala modesta*)				1

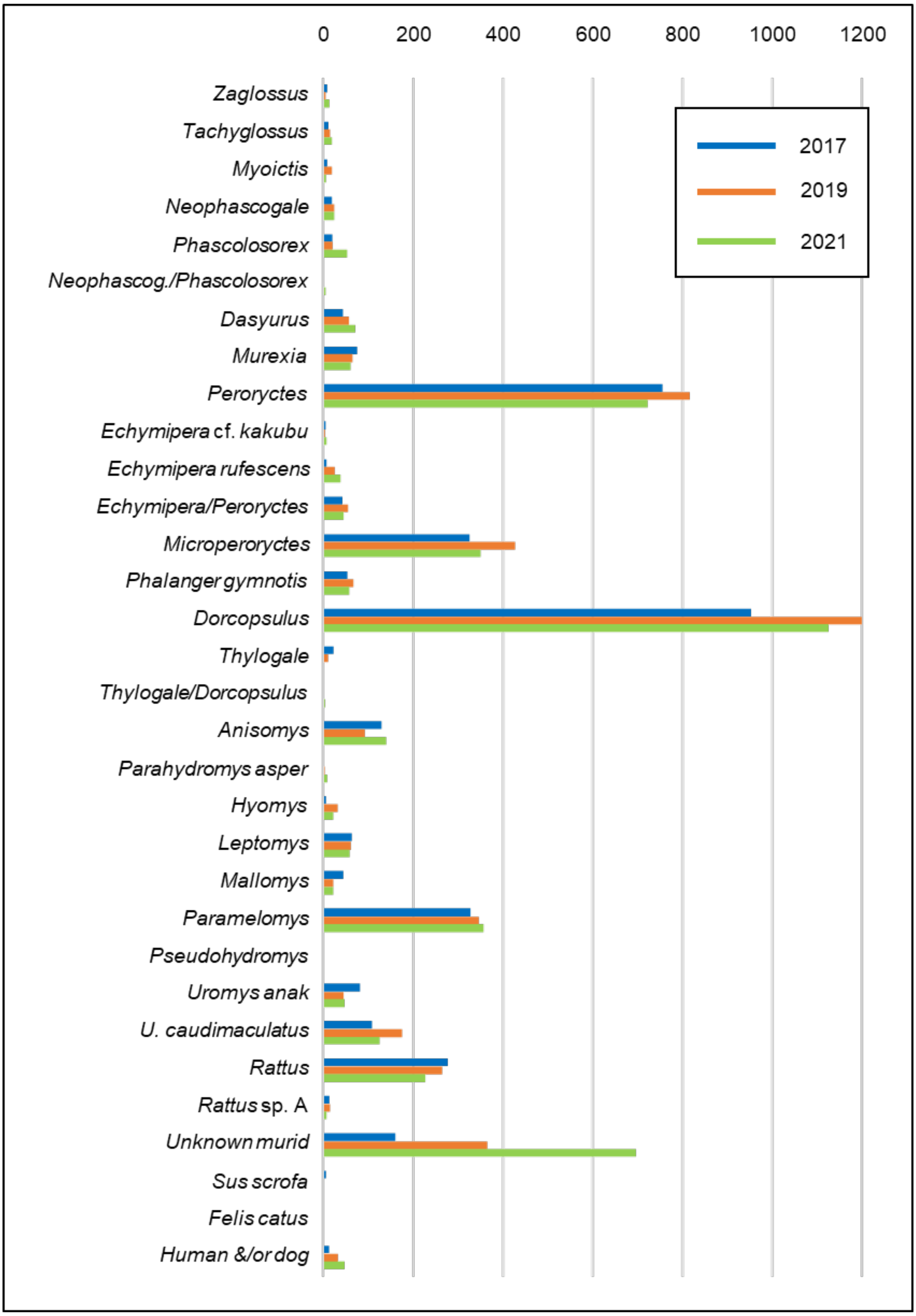


Figure 2.1. The number of independent terrestrial mammal photographic events recorded in each sampling year.

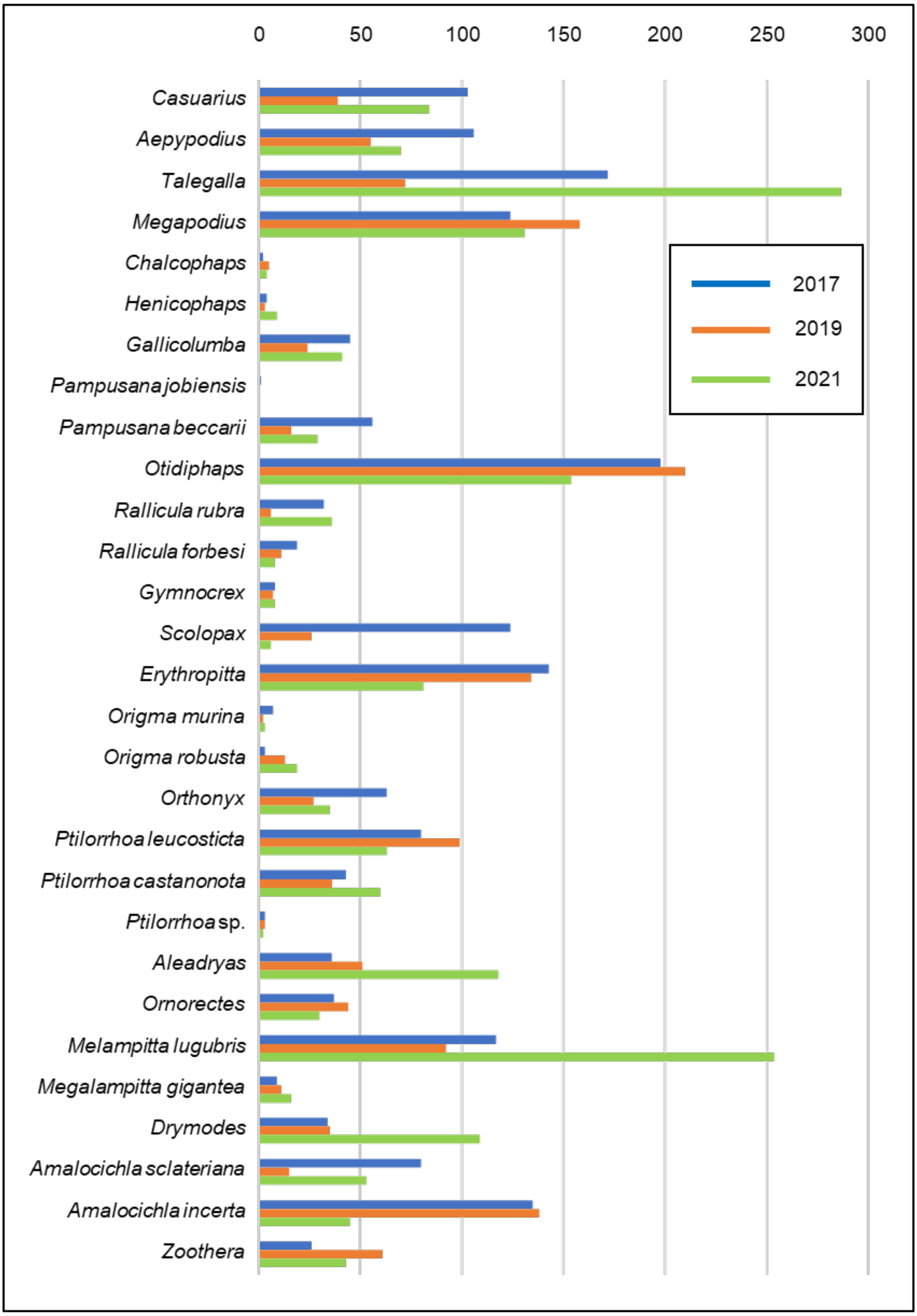


Figure 2.2. The number of independent terrestrial bird photographic events recorded in each sampling year.

Population monitoring (Objective 2)

Annual model-based occupancy estimates are shown for the Striped Bandicoot in Figure 2.3. Annual activity indices are shown for 21 species (five mammals and 16 birds) in Figure 2.4. Table 2.3 describes the direction and strength of effect in cases where the change in annual activity rate were significant (P<0.05) and near-significant (0.05<P<0.10). Annual naïve occupancy measures are shown in Figure 2.5 for five species of conservation significance for which too few data were available to generate statistical models of occupancy or activity rate. Significance testing can only be applied to species represented in Figures 2.3 and 2.4.

Observed trends varied both between species and among sites within species. Most comparisons indicate a steady state between years (Figs. 2.3–2.5).

Fifteen (15) species showed a significant change in activity rate at one or more sites during 2019–2021. In most species (14/15) the change was restricted to a single site. The activity rates of 10 species were significantly lower in 2021 than when they were first assessed in 2017 – Ground Cuscus (*Phalanger gymnotis*), Small Dorcopsis (*Dorcopsulus vanheurni*) and New Guinea Scrubfowl (*Megapodius decollatus*) at Arakubi, Collared Brushturkey (*Talegalla jobiensis*) at KP107, Striped Bandicoot (*Microperoryctes longicauda*) and Lesser Ground Robin (*Amalocichla incerta*) at Hides Low, and Raffray's Bandicoot (*Peroryctes raffrayana*), New Guinea Woodcock (*Scolopax rosenbergii*), Papuan Logrunner (*Orthonyx novaeguineae*) and Greater Ground Robin (*Amalocichla sclateriana*) at Hides High. Activity rates of three species were significantly higher in 2021 than in 2017 – Collared Brushturkey at Arakubi, Papuan Scrub Robin (*Drymodes beccarii*) at KP107 and Rufous-naped Bellbird (*Aleadryas rufinucha*) at Hides High. Significant increases in the activity rates of Pheasant Pigeon (*Otidiphaps nobilis*) and Russet-tailed Thrush (*Zoothera heinei*) at KP107 in 2019 were matched by decreases at those sites in 2021.

Data were sufficient to model activity rates for two conservation listed species. There was no notable change in the activity rates of New Guinea Quoll (*Dasyurus albopunctatus*) at any site or of Small Dorcopsis (*Dorcopsulus vanheurni*) at KP107, Hides Low or Hides High. As mentioned above, there was a significant decrease in Small Dorcopsis activity at Arakubi in 2021.

Among other conservation listed fauna, four IUCN listed species showed a marked decline in naïve occupancy rate at one or more BAA 2 sites in 2019 and/or 2021 compared with the initial 2017 dataset (Figure 2.5). The strongest declines were observed at Arakubi—after records on multiple cameras there in 2017, there have been no subsequent images of the Eastern Long-beaked Echidna (*Zaglossus bartoni*) or the Pademelon (*Thylogale* sp.), Goodfellow's Tree Kangaroo was unrecorded in 2021, and Woolley's Three-striped Dasyure (*Myoictis leucura*) was photographed on only one camera in 2021. At KP107, naïve occupancy of both the Pademelon and Woolley's Three-striped Dasyure was also much lower in 2021 than in previous years. Conversely, naïve occupancy of the Endangered Ifola has remained high at KP107, where it was recorded on 40% of cameras in both 2019 and 2021.

At BAA 1, detection rates remain low for IUCN threatened mammals at Hides Low—it is the only site at which tree kangaroos have not been camera trapped and the Eastern Long-beaked Echidna has been photographed there only once (in 2021). By contrast, at Hides High, naïve occupancy of the Eastern Long-beaked Echidna increased sharply in 2021 (from 11% to 37% of cameras) and that of the Ifola has risen steadily from one camera (5%) in 2017 to three cameras (16%) in 2021.

Figure 2.6 shows the proportion of sampling days in which humans and/or dogs were photographed at each site in each sampling year. The 2021 rate of daily incursions decreased at all sites compared with 2019 levels. At BAA 1, incursion rates have fallen steadily since sampling began in 2017. By contrast, at the BAA 2 sites, both the 2019 and the 2021 incursion rates were higher than those recorded there in 2017. At Arakubi, where the decline in records of IUCN listed species has been strongest, most incursions have been recorded closer to the PNG LNG pipeline ROW than to Arakubi Quarry and its access road (2019 – 13/16 incursion days; 2021 – 12/17 incursion days), suggesting that most hunting parties are accessing the area via the pipeline route.

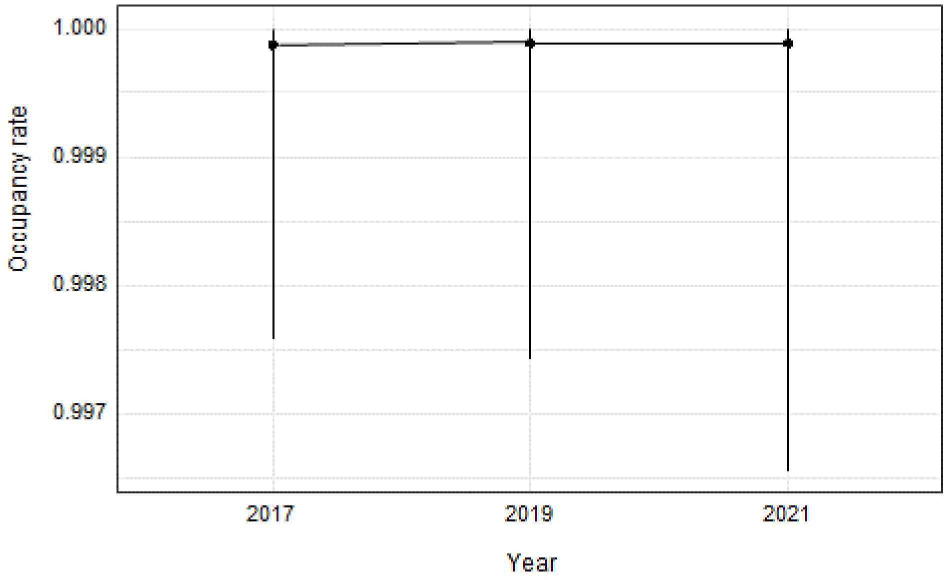


Figure 2.3. Change in occupancy rate between years for the Striped Bandicoot at BAA 2. Numbers on the Y axis show the proportion of 'areas' occupied within each BAA (see Methods).

Table 2.3. The direction and strength of effect for significant (P<0.05) and near-significant (0.05<P<0.10) changes in daily activity rate between years (see also Figure 2.4).

Species, site	Years	t value	P	Estimate (±SE)
Raffray's Bandicoot (*Peroryctes raffrayana*)				
Hides High	2017:2021	-3.38	0.041	-0.77(0.23)
Striped Bandicoot (*Microperoryctes longicauda*)				
Hides Low	2017:2021	-3.13	0.028	-0.80(0.26)
Ground Cuscus (*Phalanger gymnotis*)				
Arakubi	2017:2021	-3.40	0.025	-2.07(0.61)
Small Dorcopsis (*Dorcopsulus vanheurni*)				
Arakubi	2017:2021	-3.98	0.006	-0.94(0.24)
Arakubi	2019:2021	-4.47	<0.001	-1.05(0.24)
Collared Brushturkey (*Talegalla jobiensis*)				
Arakubi	2017:2021	3.48	0.009	1.19(0.34)
Arakubi	2019:2021	3.59	0.007	1.23(0.34)
KP107	2017:2019	4.81	<0.0001	1.76(0.37)
KP107	2017:2021	3.14	0.027	1.05(0.34)
New Guinea Scrubfowl (*Megapodius decollatus*)				
Arakubi	2017:2021	-3.86	0.006	-1.57(0.41)
Arakubi	2019:2021	-3.72	0.009	-1.52(0.41)
Pheasant Pigeon (*Otidiphaps nobilis*)				
KP107	2017:2019	3.25	0.020	0.97(0.30)
KP107	2019:2021	-4.23	<0.001	-1.31(0.31)
New Guinea Woodcock (*Scolopax rosenbergii*)				
Hides High	2017:2021	-5.15	<0.0001	-2.74(0.53)
Hides High	2019:2021	-3.29	0.005	-1.81(0.55)

Species, site	**Years**	**t value**	**P**	**Estimate (±SE)**
Papuan Pitta (*Erythropitta macklotti*)				
KP107	2017:2019	-2.77	0.073	-1.01(0.37)
KP107	2019:2021	-3.03	0.037	-1.11(0.36)
Papuan Logrunner (*Orthonyx novaeguineae*)				
Hides High	2017:2019	-4.96	<0.0001	-2.88(0.58)
Hides High	2017:2021	-5.25	<0.0001	-3.06(0.58)
Rufous-naped Bellbird (*Aleadryas rufinucha*)				
Hides High	2017:2021	3.35	0.015	1.25(0.37)
Hides High	2019:2021	3.38	0.014	1.25(0.37)
Lesser Melampitta (*Melampitta lugubris*)				
Hides High	2019:2021	2.31	0.064	0.86(0.37)
Papuan Scrub Robin (*Drymodes beccarii*)				
KP107	2017:2021	3.59	0.007	1.74(0.49)
KP107	2019:2021	2.83	0.063	1.30(0.46)
Greater Ground Robin (*Amalocichla sclateriana*)				
Hides High	2017:2019	-4.60	<0.0001	-1.65(0.36)
Hides High	2017:2021	-2.53	0.039	-0.79(0.31)
Hides High	2019:2021	2.33	0.062	0.86(0.37)
Lesser Ground Robin (*Amalocichla incerta*)				
Hides Low	2017:2021	-2.59	0.035	-1.04(0.40)
Russet-tailed thrush (*Zoothera heinei*)				
KP107	2017:2019	4.54	0.0001	1.39(0.31)
KP107	2019:2021	-2.88	0.017	-0.84(0.29)

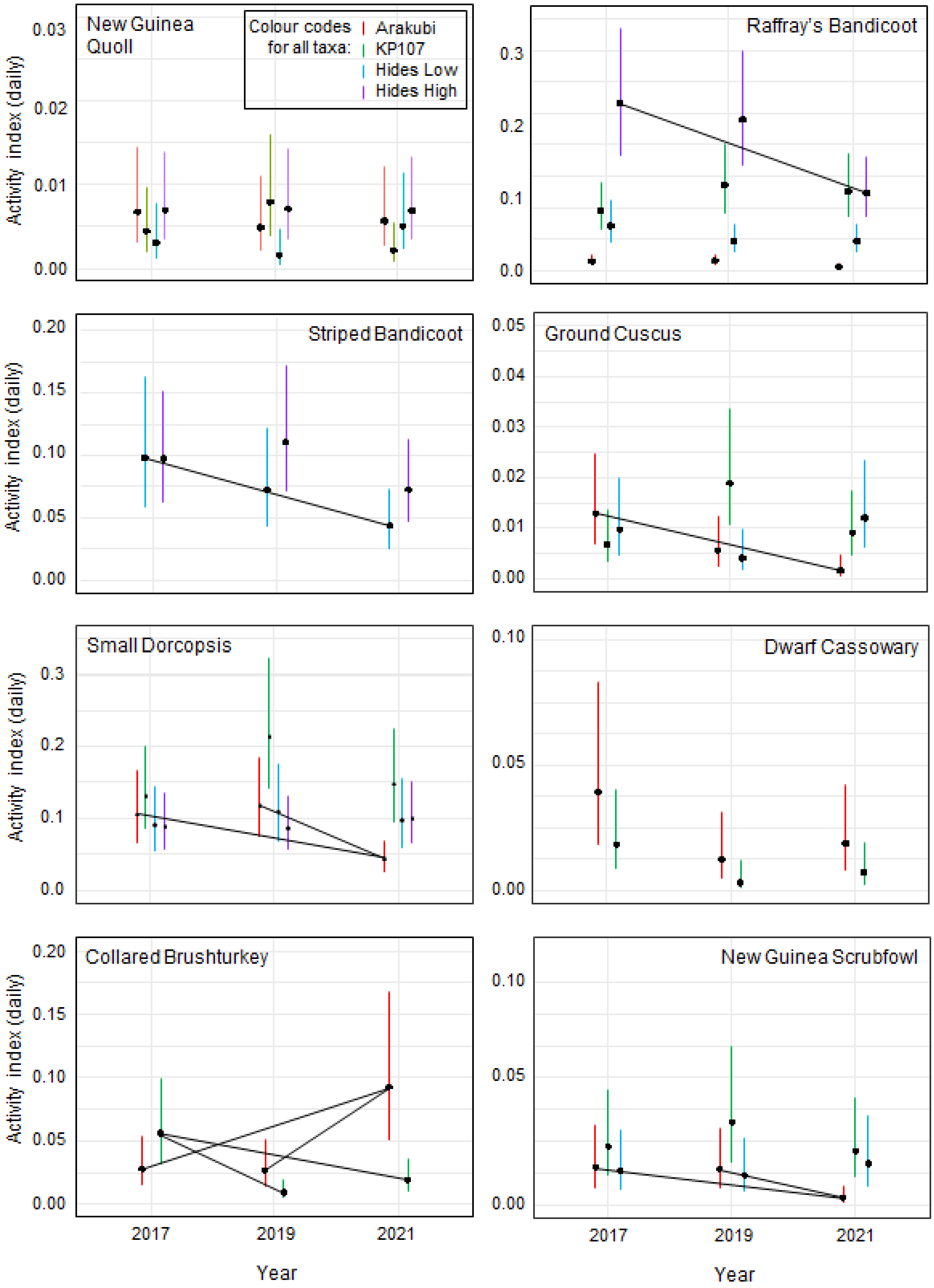


Figure 2.4. Change in animal activity rate between years. Solid lines indicate significant change in activity rate between years (P<0.05); dashed lines indicate near-significant change (0.05<P<0.1); see Table 2.3 for strength of effect.

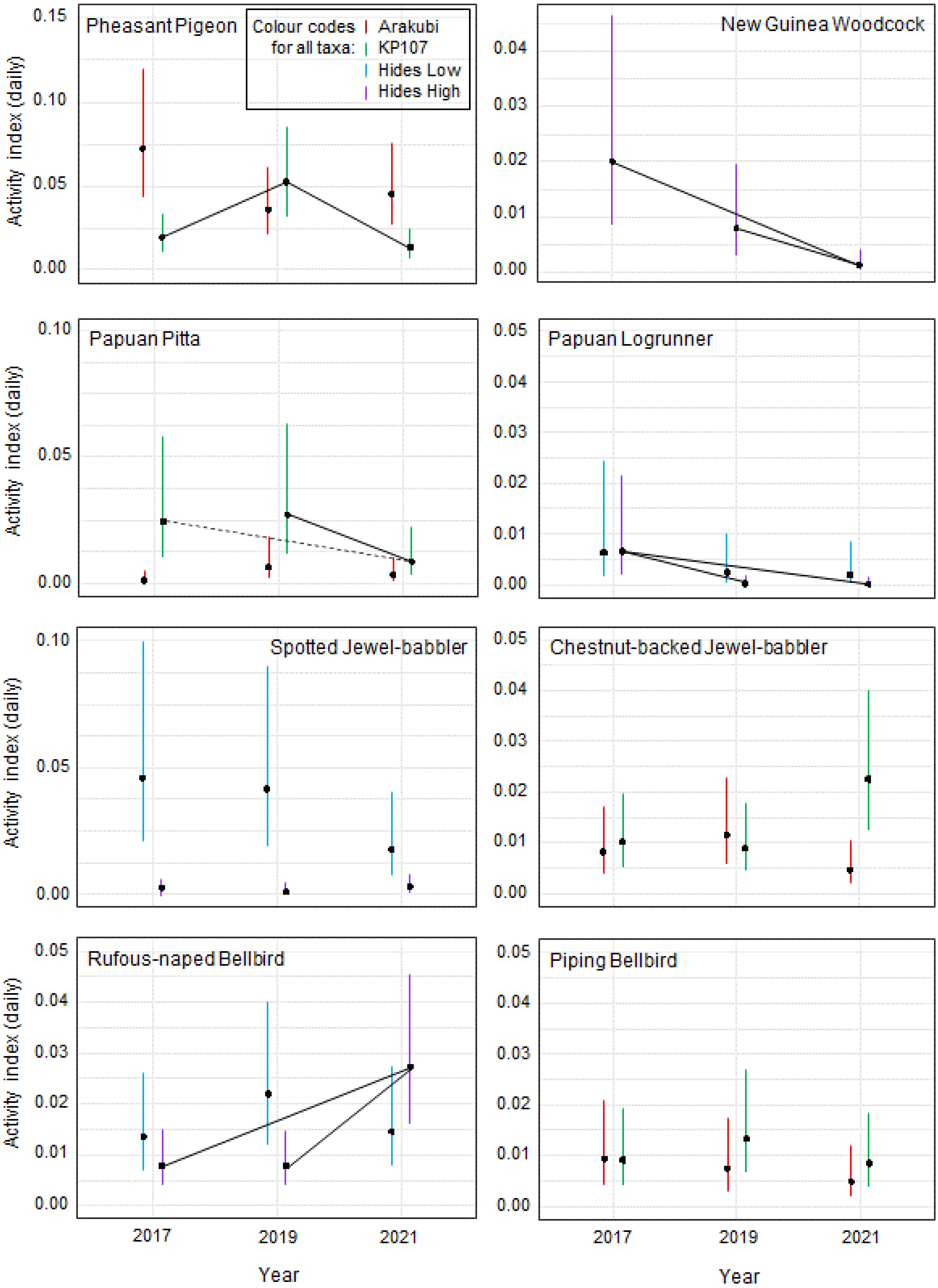


Figure 2.4. (Continued)

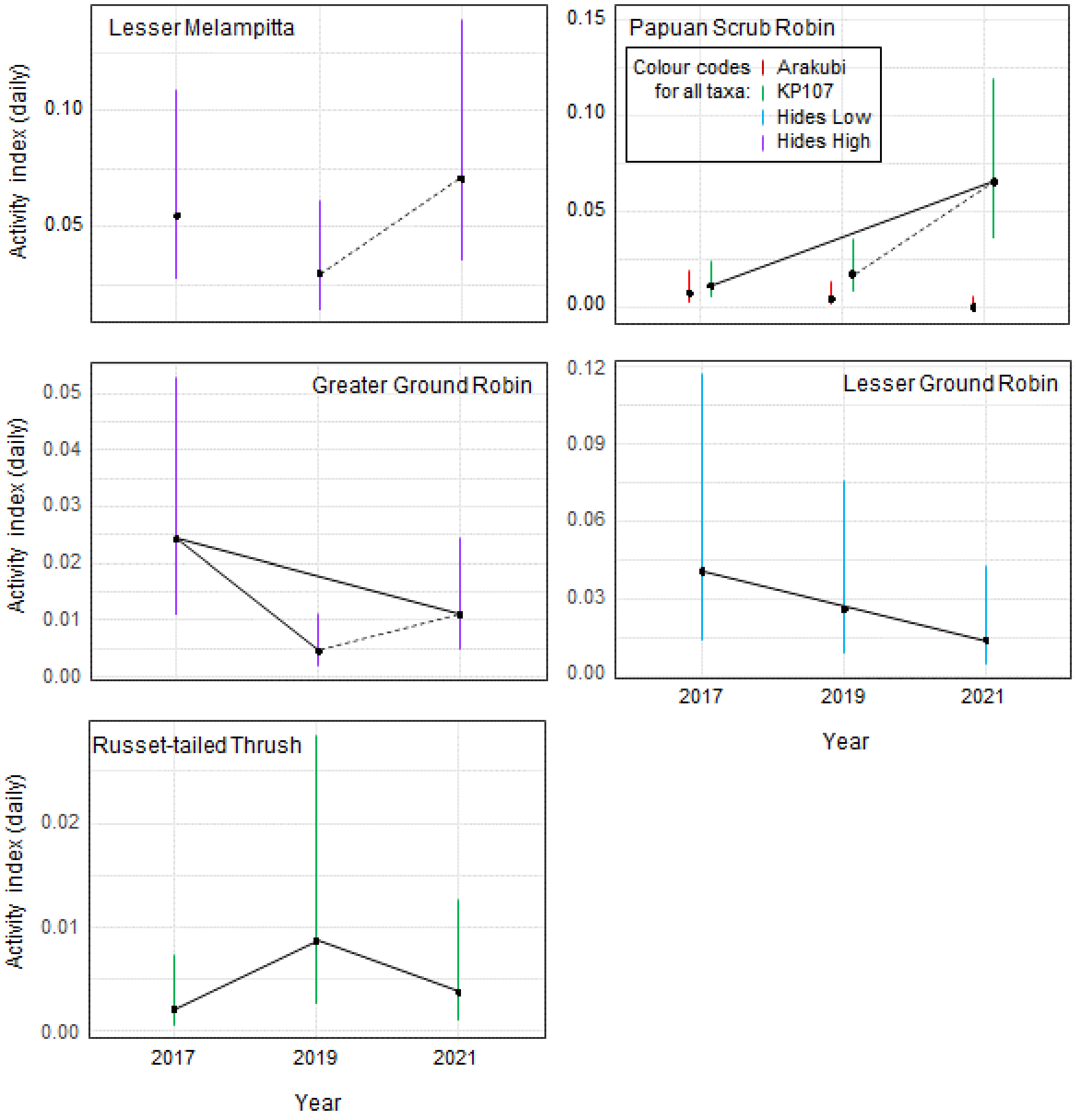


Figure 2.4. (Continued)

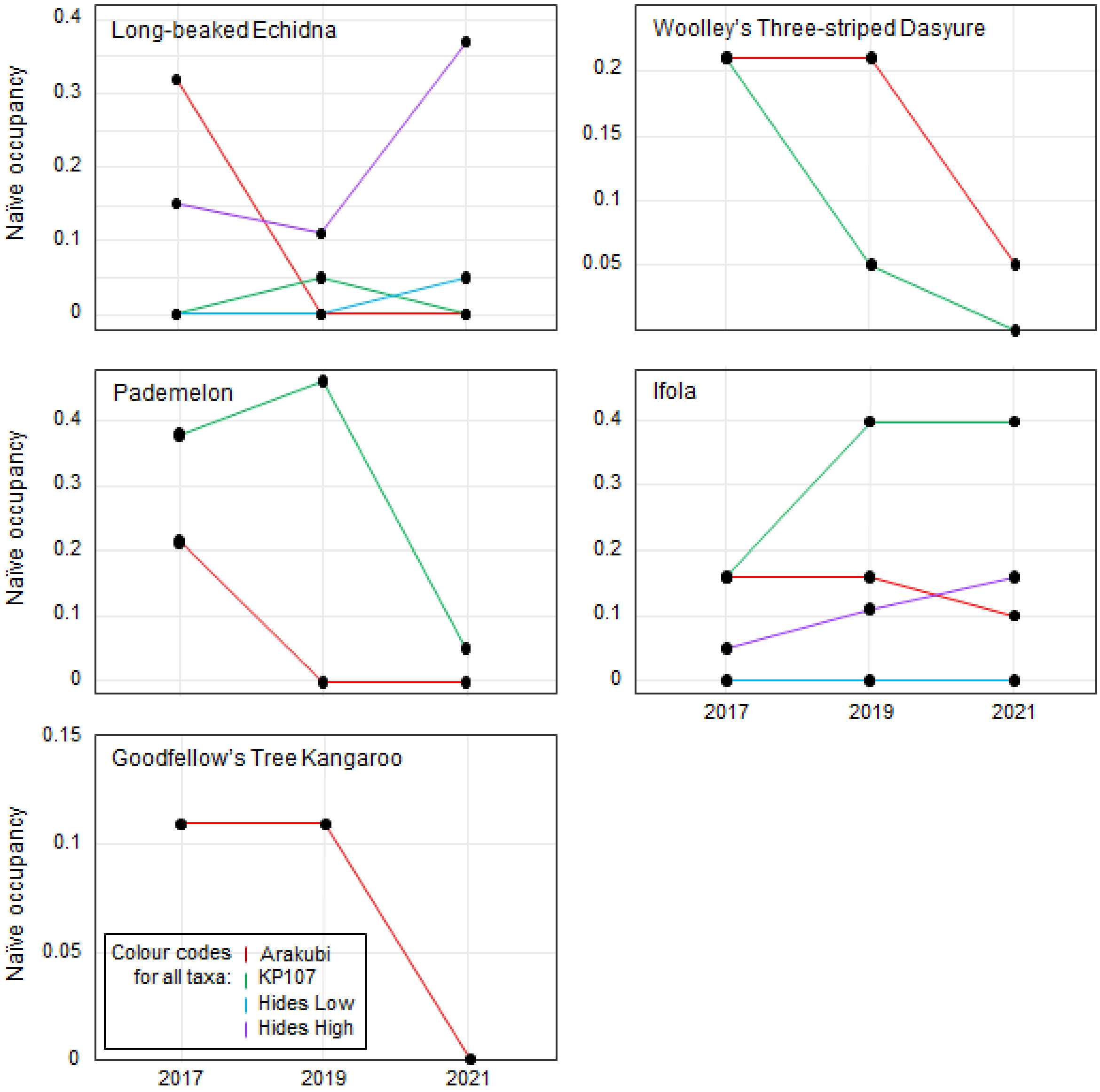


Figure 2.5. Change in naïve occupancy rate between years, for five IUCN listed species for which statistical modelling could not be performed.

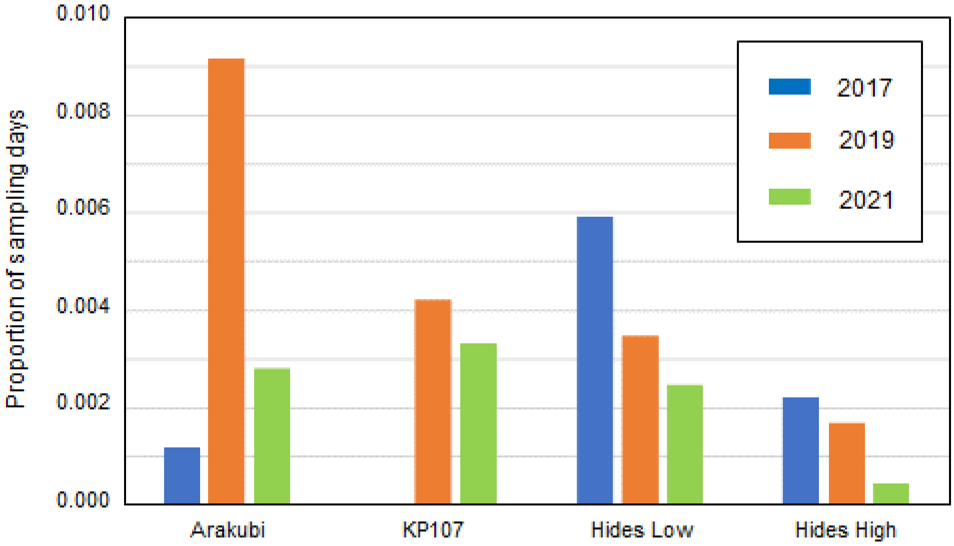


Figure 2.6. The proportion of sampling days in which humans and/or dogs were photographed.

Edge effects (Objective 3)

Multi-model comparisons were successfully run for nine terrestrial mammal and 20 terrestrial bird species (60 site- or BAA-level analyses). We did not analyse rodent data as this family (Muridae) is the focus of a separate study (Chapter 3, this report). For all taxa analysed, Appendix 2.2 shows the model-averaged relative importance of the predictor variables with an importance value >0.2, along with their coefficient point estimates and standard errors.

Focusing on edge effects, and considering only those cases where final models ranked higher than the associated null model, distance measures featured in 23 final models (at site or BAA level) constructed for 16 species (four mammals and 12 birds) (Table 2.4, Figure 2.7). In all but one of these models, distance from the nearest road or clearing was significantly correlated with activity rate ($P<0.05$); in the one exception, Dwarf Cassowary (*Casuarius bennetti*) activity showed a near-significant correlation with distance from the nearest clearing (P=0.052).

In nearly two-thirds of the final models (15/23) the observed pattern is consistent with edge avoidance, with less activity recorded nearer to Project infrastructure. Evidence for edge avoidance was strongest at BAA 2, particularly for Raffray's Bandicoot (*Peroryctes raffrayana*) at KP107, Small Dorcopsis at Arakubi and KP107, Collared Brushturkey at Arakubi and KP107, New Guinea Scrubfowl (*Megapodius decollatus*) at Arakubi, Pheasant Pigeon at Arakubi and KP107, and Russet-tailed thrush (*Zoothera heinei*) at KP107 – these cases comprise nine of the 11 models (81.8%) that were clearly better than the associated null and in which there was significantly more activity further from Project infrastructure. Similar patterns were observed for Dwarf Cassowary at Arakubi and at BAA 2 (sites pooled), although statistically these were less compelling (null model not clearly rejected at Arakubi and a non-significant trend at the BAA level; Table 2.4).

There was comparatively little evidence for edge avoidance at BAA 1. Relevant cases were observed only at Hides Low, where activity rates of Striped Bandicoot (*Microperoryctes longicauda*) and New Guinea Scrubfowl (*Megapodius decollatus*) were significantly lower within 100 m of clearings (null model rejected in both cases) (Table 2.4, Figure 2.7). Similarly, Wattled Brushturkey (*Aepypodius arfakianus*) activity at Hides Low was significantly lower within 100 m of clearings, and Small Dorcopsis showed less activity near roads, although the final models in these cases were less well supported (Wattled Brushturkey null ΔAIC_c=0.64, Small Dorcopsis null ΔAIC_c=5.90).

The remaining models presented in Table 2.4 describe a reverse-pattern edge effect, with higher rates of activity closer to Project infrastructure (Figure 2.7). This pattern was observed in five species at BAA 1, with the null model clearly rejected for Lesser Melampitta (*Melampitta lugubris*) and Greater Ground Robin at Hides High, and for Papuan Logrunner (*Orthonyx novaeguineae*) at Hides Low. Similar patterns were observed for Ground Cuscus (*Phalanger gymnotis*) at Hides Low and Rufous-naped Bellbird (*Aleadryas rufinucha*) across BAA 1, although the null models performed equally well in these cases. At BAA 2, reverse-pattern edge effects are reported in well-supported models for New Guinea Scrubfowl and Papuan Pitta (*Erythropitta macklotti*) at KP107. A similar model for Cinnamon Ground Dove (*Gallicolumba rufigula*) at Arakubi was no better than the null (ΔAIC_c=0.28).

Table 2.4. Model summaries for 16 species in which distance from clearings or roads was influential in the final model structure, and where the final model ranked higher than the null model. Model estimates and standard errors are shown for each variable. Abbreviated variables (see Methods): DClr – distance from clearing; LT50/LT100 – less/more than 50/100 m from clearing; LT100Rd – less/more than 100 m from road; LR20 – local relief at the 20 m radius scale; CnpHt – canopy height; TrSm/TrLge – density of trees >10/30 cm dbh; UD1/UD2 – understorey density below 1/2 m; LfLit – leaf litter depth; Logs/Moss – proportion of ground covered by large woody debris/moss.

Species, sites, model structure and performance	**Variable**	**t value**	**P**	**Estimate (±SE)**
Raffray's Bandicoot (*Peroryctes raffrayana*) – KP107				
LT50+Year+TrLge	LT50: >50 m	5.31	<0.00001	0.93(0.18)
R^2 = 0.366, n =55	Year (2017:2019)	2.03	0.048	0.31(0.15)
Null ΔAICc = 22.20	Year (2017:2021)	1.48	0.145	0.23(0.16)
	TrLge	2.60	0.012	0.07(0.03)
Striped Bandicoot (*Microperoryctes longicauda*) – Hides Low				
LT100+Year+TrSm	LT100: >100 m	2.14	0.038	0.49(0.23)
R^2 = 0.164, n =47	Year (2017:2019)	-0.61	>0.5	-0.16(0.27)
Null ΔAICc = 19.52	Year (2017:2021)	-2.76	0.009	-0.77(0.28)
	TrSm	4.32	<0.0001	0.06(0.01)
Ground Cuscus (*Phalanger gymnotis*) – Hides Low				
LT100Rd	LT100Rd	-2.12	0.040	-1.07(0.51)
R^2 = 0.158, n =47				
Null ΔAICc = 0.21				
Small Dorcopsis (*Dorcopsulus vanheurni*) – Arakubi				
LT50+Year+LR100+UD1+TrSm	LT50: >50 m	3.98	<0.001	1.48(0.37)
R^2 = 0.359, n =52	Year (2017:2019)	0.11	>0.9	0.03(0.26)
Null ΔAICc = 17.19	Year (2017:2021)	-2.74	0.009	-0.75(0.27)
	LR100	2.67	0.011	0.02(<0.01)
	UD1	-3.57	<0.001	-0.17(0.05)
	TrSm	4.30	<0.0001	0.10(0.03)
Small Dorcopsis (*Dorcopsulus vanheurni*) – KP107				
LT50+Year+UD1	LT50: >50 m	5.08	<0.00001	1.24(0.24)
R^2 = 0.766, n =55	Year (2017:2019)	2.24	0.030	0.50(0.22)
Null ΔAICc = 33.51	Year (2017:2021)	0.21	>0.8	0.04(0.23)
	UD1	5.53	<0.00001	0.20(0.04)
Small Dorcopsis (*Dorcopsulus vanheurni*) – Hides Low				
DRd+Logs	DRd	2.37	0.022	0.002(0.001)
R^2 = 0.185, n =47	Logs	-2.10	0.042	-0.220(0.105)
Null ΔAICc = 5.90				

Species, sites, model structure and performance	**Variable**	**t value**	**P**	**Estimate (±SE)**
Dwarf Cassowary (*Casuarius bennetti*) – Arakubi				
LT50+Year	LT50: >50 m	2.21	0.032	1.21(0.55)
R^2 = 0.151, n =52	Year (2017:2019)	-2.48	0.017	-1.03(0.41)
Null ΔAICc = 2.88	Year (2017:2021)	-1.56	0.127	-0.58(0.38)
Dwarf Cassowary (*Casuarius bennetti*) – BAA 2				
LT50+Year+Site+TrSm	LT50: >50 m	1.96	0.052	0.76(0.39)
R^2 = 0.241, n =107	Year (2017:2019)	-3.11	0.002	-0.99(0.32)
Null ΔAICc = 6.88	Year (2017:2021)	-2.64	0.010	-0.81(0.31)
	Station (KP107)	-2.81	0.006	-0.75(0.27)
	TrSm	-2.26	0.026	-0.05(0.02)
Wattled Brushturkey (*Aepypodius arfakianus*) – Hides Low				
LT100+LfLit	LT100: >100 m	3.12	0.003	1.77(0.57)
R^2 = 0.465, n =47	LfLit	3.58	<0.001	0.04(0.01)
Null ΔAICc = 0.64				
Collared Brushturkey (*Talegalla jobiensis*) – Arakubi				
DClr+Year+UD2	DClr	5.79	<0.000001	0.007(0.001)
R^2 = 0.366, n = 52	Year (2017:2019)	-0.02	>0.9	-0.005(0.325)
Null ΔAICc = 30.75	Year (2017:2021)	4.33	<0.0001	1.28(0.30)
	UD2	-1.90	0.064	-0.04(0.02)
Collared Brushturkey (*Talegalla jobiensis*) – KP107				
LT50+Year	LT50: >50 m	2.19	0.033	1.04(0.48)
R^2 = 0.224, n = 55	Year (2017:2019)	-4.02	<0.0002	-1.68(0.42)
Null ΔAICc = 11.49	Year (2017:2021)	-2.62	0.012	-1.02(0.39)
New Guinea Scrubfowl (*Megapodius decollatus*) – Arakubi				
LT100Rd+TrSm+LfLit	LT100Rd	2.69	0.0098	2.30(0.856)
R^2 = 0.0175 n = 52	TrSm	2.10	0.041	0.08(0.04)
Null ΔAICc = 6.74	LfLit	-4.00	0.0002	-0.04(0.01)
New Guinea Scrubfowl (*Megapodius decollatus*) – KP107				
DRd+UD1+Logs+LfLit	DRd	-3.01	0.004	-0.004(0.001)
R^2 = 0.101, n = 55	UD1	-2.74	0.008	-0.22(0.08)
Null ΔAICc = 6.16	Logs	2.33	0.024	0.42(0.18)
	LfLit	-2.94	0.005	-0.02(0.01)
New Guinea Scrubfowl (*Megapodius decollatus*) – Hides Low				
LT100+Moss+Moss^2	LT100: >100 m	2.90	0.006	0.80(0.28)
R^2 = 0.248, n = 47	Moss	-2.80	0.008	-1.70(0.61)
Null ΔAICc = 6.00	Moss^2	-2.12	0.040	-0.25(0.12)
Cinnamon Ground Dove (*Gallicolumba rufigula*) – Arakubi				
DRd	DRd	-2.28	0.027	-0.003(0.001)
R^2 = 0.066, n = 52				
Null ΔAICc = 0.28				

Species, sites, model structure and performance	**Variable**	**t value**	**P**	**Estimate (±SE)**
Pheasant Pigeon (*Otidiphaps nobilis*) – Arakubi				
DRd+LR20+UD2+Moss	DRd	2.81	0.007	0.003(0.001)
R^2 = 0.104, n = 52	LR20	1.87	0.068	0.04(0.02)
Null ΔAICc = 8.58	UD2	-2.23	0.031	-0.05(0.02)
	Moss	-3.96	<0.0003	-0.55(0.14)
Pheasant Pigeon (*Otidiphaps nobilis*) – KP107				
LT50+Year	LT50: >50 m	3.37	0.001	1.33(0.40)
R^2 = 0.385, n = 55	Year (2017:2019)	3.32	0.002	1.01(0.30)
Null ΔAICc = 18.33	Year (2017:2021)	-0.53	>0.6	-0.18(0.35)
Papuan Pitta (*Erythropitta macklotti*) – KP107				
LT100+LR50+UD2+Logs+Moss	LT100: >100 m	-2.59	0.013	-1.66(0.64)
R^2 = 0.193, n = 55	LR50	-3.24	0.002	-0.11(0.03)
Null ΔAICc = 8.77	UD2	-3.08	0.003	-0.15(0.05)
	Logs	-3.21	0.002	-1.03(0.32)
	Moss	2.31	0.025	0.56(0.24)
Papuan Logrunner (*Orthonyx novaeguineae*) – Hides Low				
LT50+TrLge	LT50: >50 m	-4.08	0.0002	-2.11(0.52)
R^2 = 0.231, n = 47	TrLge	-2.25	0.030	-0.27(0.12)
Null ΔAICc = 5.16				
Rufous-naped Bellbird (*Aleadryas rufinucha*) – BAA 1				
LT100Rd	LT100Rd	-2.74	0.007	-0.75(0.27)
R^2 = 0.123, n = 104				
Null ΔAICc = 2.08				
Lesser Melampitta (*Melampitta lugubris*) – Hides High				
LT100+TrLge	LT100: >100 m	-4.77	0.00001	-1.61(0.34)
R^2 = 0.163, n = 57	TrLge	-2.35	0.022	-0.16(0.07)
Null ΔAICc = 18.50				
Greater Ground Robin (*Amalocichla sclateriana*) – Hides High				
DClr+Year+TrSm+UD1	DClr	-3.29	0.0018	-0.005(0.001)
R^2 = 0.640, n = 57	Year (2017:2019)	-3.43	0.0012	-1.35(0.39)
Null ΔAICc = 16.49	Year (2017:2021)	-2.00	0.051	-0.66(0.33)
	TrSm	-4.14	0.0001	-0.07(0.02)
	UD1	3.04	0.004	0.20(0.07)
Russet-tailed thrush (*Zoothera heinei*) – KP107				
DClr+TrSm+UD1	DClr	4.92	<0.00001	0.009(0.002)
R^2 = 0.415, n = 55	TrSm	-1.95	0.056	-0.06(0.03)
Null ΔAICc = 8.71	UD1	2.60	0.012	0.22(0.09)

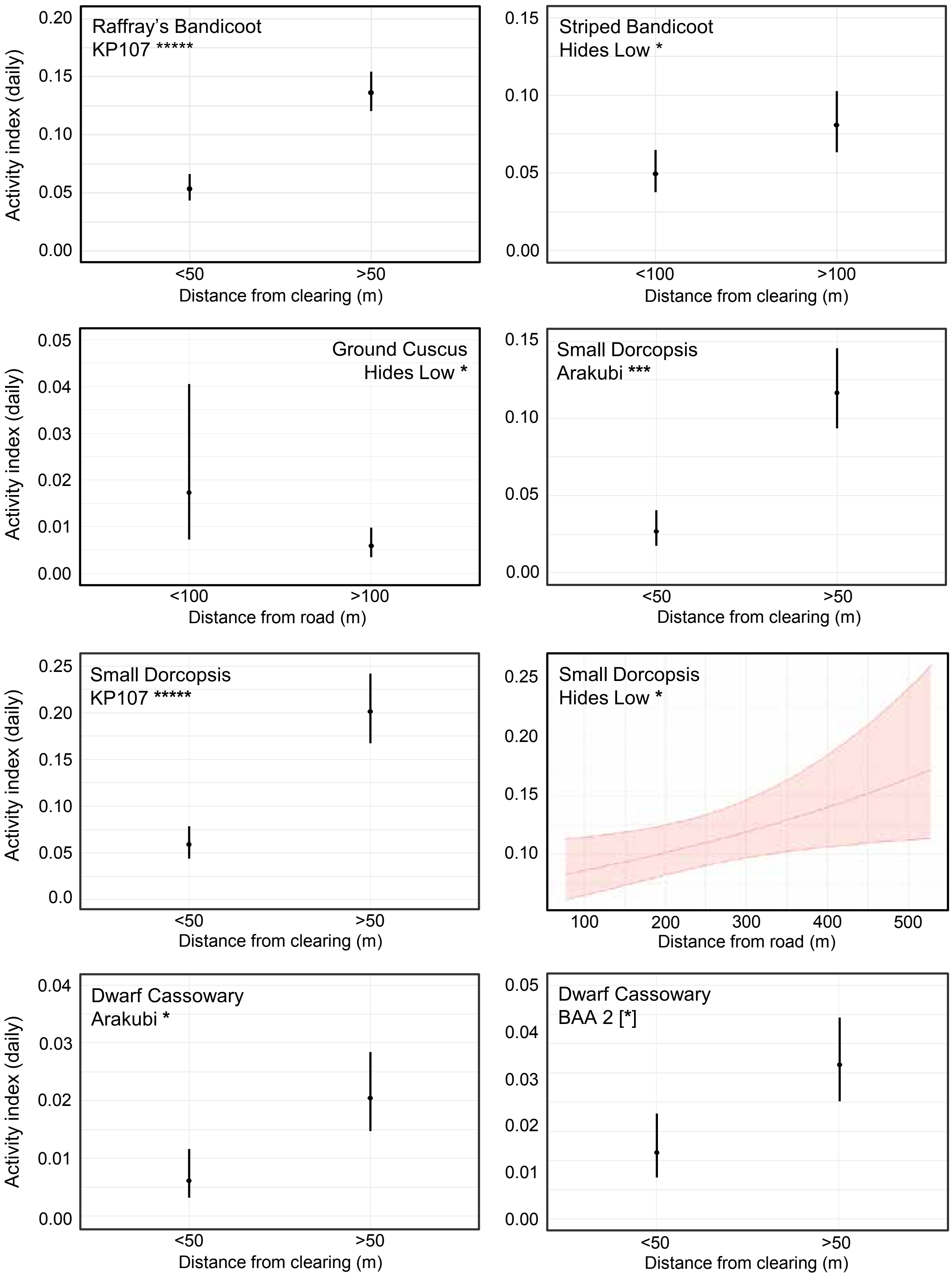


Figure 2.7. The relationship between animal activity rate and distance from Project infrastructure for species in which distance featured in the final models. Other final model terms are held constant (see Table 2.4). Significance codes: [*] =0.05<P<0.1; * =P<0.05; ** = P<0.01; *** =P<0.001; **** =P< 0.0001; ***** =P<0.00001.

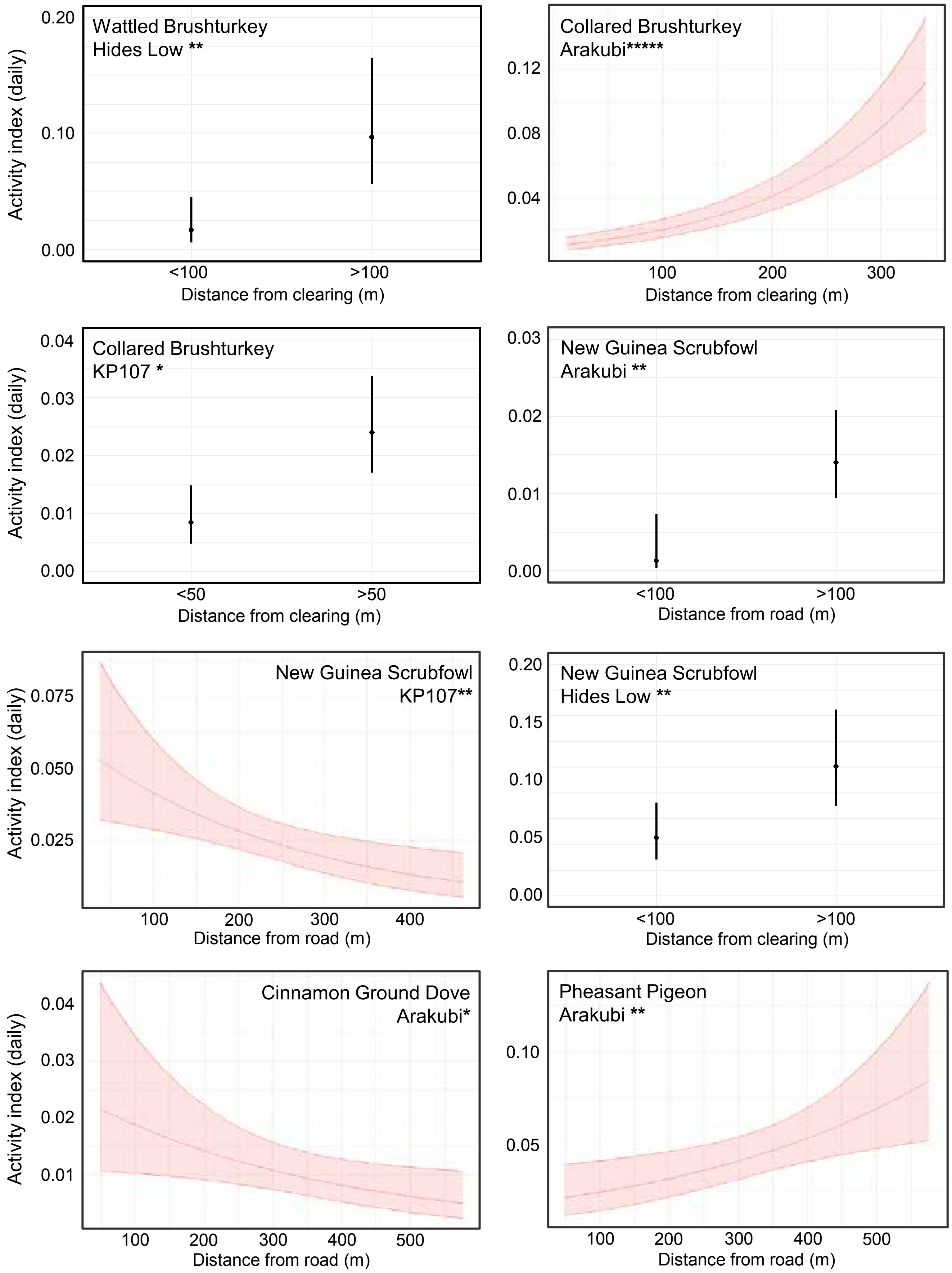


Figure 2.7. (Continued)

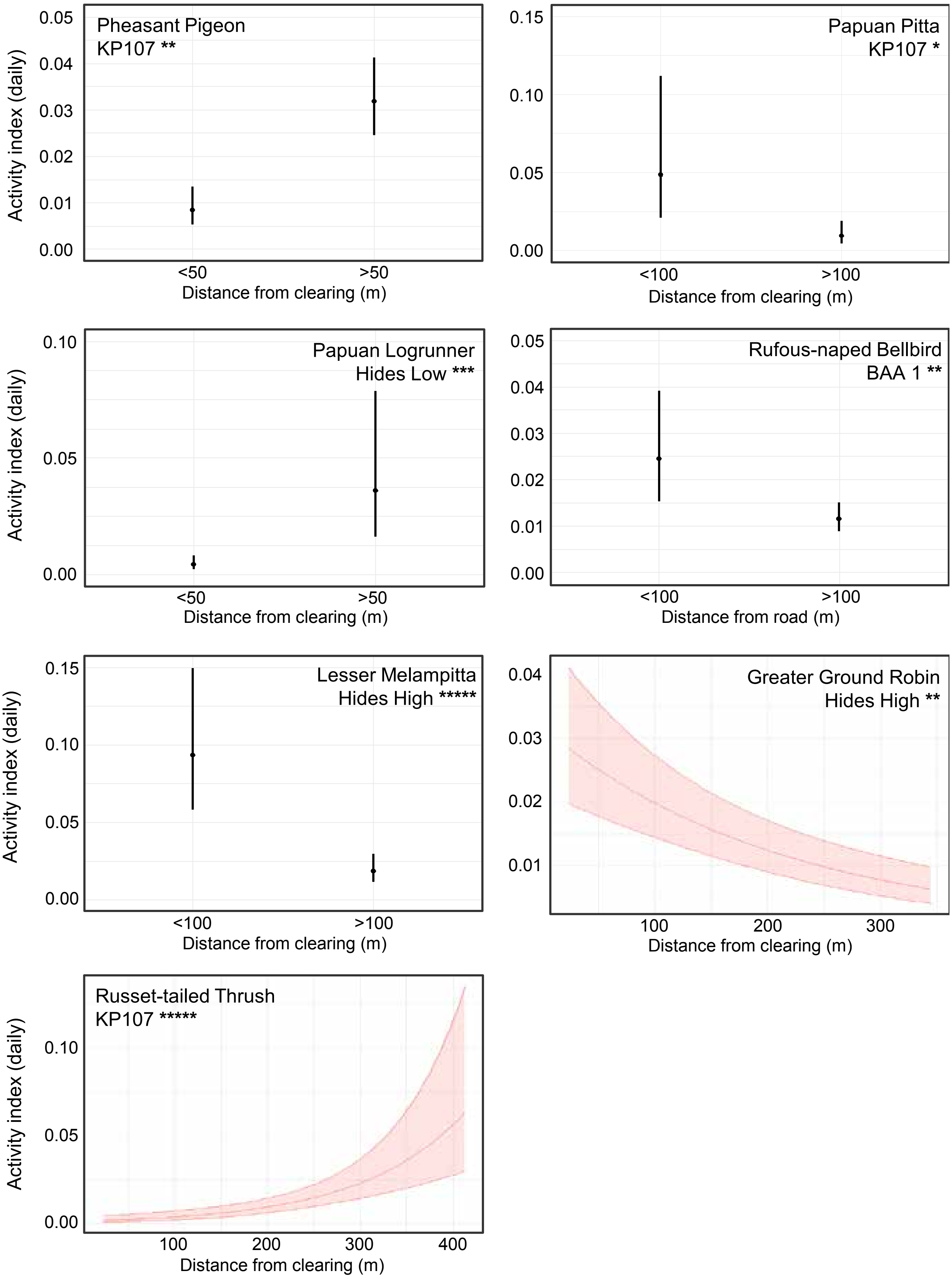


Figure 2.7. (Continued)

Discussion

Species diversity and survey completeness (Objective 1)

The PMA3 camera trapping program has recorded the vast majority of ground-dwelling bird and mammal species expected to reside or regularly occur at sampled sites. Coverage of the terrestrial bird and medium- to large-bodied mammal assemblages appears to be almost complete in each full sampling year (2017, 2019, 2021). No new terrestrial bird or mammal species were recorded in 2021, and no additional terrestrial birds are expected in subsequent years.

Most or all unrecorded terrestrial species are likely to be small murid rodents. A number of (at least partly) terrestrial, forest-dwelling murid genera are known to be regionally present but are yet to be recorded, including *Coccymys*, *Macruromys*, *Mammelomys*, *Melomys*, *Microhydromys* and *Xenuromys* (Denys et al. 2017). Other predominantly arboreal taxa such as *Abeomelomys* also occasionally come to ground. At least some of these taxa may be recognised in good quality images. Small murid rodents are covered in more detail by the small non-volant mammal monitoring study (Chapter 3, this report).

Additionally, there is almost certainly hidden diversity among individuals already photographed, with taxa such as *Murexia* and all of the small murid genera recorded in this study being rather cryptic in appearance so that trapping would be required to accurately determine the number of species represented (e.g., Chapter 3, this report).

In addition to terrestrial species, the camera trapping study continues to provide useful information on a number of arboreal mammals. Most importantly, biennial data on the IUCN Endangered Ifola and Goodfellow's Tree Kangaroo provide the only means of monitoring the local status of these rare and elusive species. A number of other species are not otherwise recorded from the BAAs, including the Masked Ring-tailed Possum (*Pseudochirulus larvatus*; BAA 1) and the Torresian Striped Possum (*Dactylopsila trivirgata*; BAA 2), both of which were recorded for the first time in 2021 (Mamu 2008; Aplin and Opiang 2017).

Population trends (Objective 2)

In cases where statistical models could be applied, a significant change in activity rate between years was observed for 15 species at 16 sites. Significant declines were reported for 10 species at 16 sites during the 2017–2021 period. In all cases where a species' activity was modelled at more than one site, a significant change recorded at one site was countered at other sites by a steady rate between years, or, in the case of the Collared Brushturkey at BAA 2, a reverse direction of significant change at another site (Table 2.3; Figure 2.4). By contrast, the decline recorded for Greater Ground Robin at Hides High is of potential significance as this is the only site within the Kikori Basin at which this rare and restricted-range species is known to occur (Woxvold and Legra 2019a,b).

Among conservation listed species, statistical models were run for two IUCN Near Threatened species – the New Guinea Quoll and Small Dorcopsis. Their activity rates were steady between years at most sites, but fell significantly for the Small Dorcopsis at Arakubi in 2021. However, the Small Dorcopsis remains the most frequently detected terrestrial vertebrate across the study area (Figure 2.1.) with records from almost all cameras (76/79; naïve occupancy=0.96) in the most recent sampling year.

By contrast, naïve occupancy fell sharply at Arakubi for three IUCN threatened species – the Long-beaked Echidna, Pademelon and Goodfellow's Tree Kangaroo (*Dendrolagus goodfellowi*) – from registers on multiple cameras in 2017 (six, four and two respectively) to none in 2021 (Figure 2.5; Appendix 2.1). Pademelon records have fallen sharply across both BAA 2 sites, from a total of 23 events in 2017 to only one in 2021. Each of these species is susceptible to hunting, particularly where dogs are used (Nicol 2015; Eldridge and Coulson 2015). In the case of the echidna, whose permanent (no seasonal migration), generally non-overlapping home ranges may exceed 50 hectares (Opiang 2009), the capture of just one or two individuals may result in its long-term absence from the sampled area (c. 70 ha). Interestingly, the decline in records of Long-beaked Echidna and Pademelon at Arakubi in 2019 corresponds with a sharp rise there in the rate of incursions reported for humans and dogs in that year (Figure 2.6). Although the rate of human/dog incursions at Arakubi fell again in 2021, it was still more than twice as high as the rate recorded in 2017, corresponding with a lack of records of Goodfellow's Tree Kangaroo for the first time. Importantly, most incursions have been recorded closer to PNG LNG infrastructure than to the Santos/OSL operated Arakubi Quarry, suggesting that most hunters and dogs are entering the Arakubi forest via Project infrastructure.

A link cannot be proved, but further sampling is warranted to understand the local status of these high value species and patterns of forest use by local residents and their dogs.

In contrast to the patterns observed at BAA 2, the rate of human and/or dog incursions recorded by camera trapping at BAA 1 has declined steadily across all sampling years (Figure 2.6). Hunting pressure is difficult to quantify. However, this decline in the measure of hunting activity at BAA 1 has coincided with an increase in the naïve occupancy of two IUCN Threatened species at Hides High – the Eastern Long-beaked Echidna and Ifola – and the first record of Eastern Long-beaked Echidna at Hides Low since camera trap monitoring began.. These trends are encouraging, although anecdotal observations over the years suggest a sustained increase in hunting activity and the harvesting of other natural resources (see Project Summary) compared to pre-construction (and pre-monitoring) rates. Acting on recomendations from the 2019 PMA3 biodiversity survey report, ExxonMobil have initiated a hunting survey in BAA 1 to collect data on hunting activity and determine target species.

While the 2017–2021 dataset provides a useful baseline against which to measure future changes, there are at least two reasons why it is too early to draw conclusions as to the ongoing status of terrestrial vertebrate populations. First, for some species, the observed changes may be a result of seasonal effects. Photographic rates reflect both animal abundance and behaviour, both of which may be influenced by seasonal factors (Burton et al. 2015) – for example: (1) the presence or departure (dispersal) of a cohort of dependent offspring; (2) changes in resource availability leading to movement into or out of the home range (landscape-level nomadism); (3) changes in foraging/movement rate within a stable home range, for example due to a change in animal density, seasonal food availability or the requirements of dependent offspring, or; (4) a localised increase in resource availability near some camera positions, for example via seed- or fruit-fall. Sampling in 2019 and 2021 was similarly timed but began some months after the 2017 start date (three and two months respectively). Changes in activity rate observed in 2019, and which held steady or only partially returned to 2017 levels in 2021, may in part be attributable to sampling at a different time of year. Seasonal patterns of movement and breeding are still poorly known for many New Guinean mammal and bird species, and in the aseasonal climate of the study area (McAlpine et al. 1983) monthly rainfall patterns can be difficult to predict. Nevertheless, to help unpack the relative influence of seasonal vs. annual changes, as far as possible future sampling should be standardised with respect to seasonal timing.

Second, there may be natural variation in abundance between years, so that longer term studies are required to make reliable inferences about population change. Although previously considered to be relatively stable, recent studies have shown that many tropical animal populations are subject to natural fluctuations (Latta et al. 2011; Blake and Loiselle 2015). For example, annual camera trap monitoring of terrestrial mammals and birds in Ecuador has revealed marked inter-year changes in detection rate and occupancy estimates for multiple species, but no evidence of a consistent change in the status of any species across the full 11-year sampling period (Blake et al. 2017). There are no available data on population dynamics for the species assessed in this study. If natural fluctuations are similar to those observed in lowland Ecuador, then a sampling period of around 20 years would be required to draw similar conclusions under the biennial PMA3 sampling program. However, if the observed changes are more stable across sampling years, then earlier inferences may be possible. Continued sampling will help to determine which pattern is relevant for which species, although the timing with which inferences can be made will depend on the sampling frequency.

Edge effects (Objective 3)

Anthropogenic infrastructure has been shown to influence the presence and behaviour of a variety of animal species (Laurance and Bierregaard 1997; Laurance et al. 2004; Leblond et al. 2013; van der Ree et al. 2015), and camera trapping has been used to demonstrate behavioural responses to infrastructure or forest edge in multiple taxa (e.g., Leblond et al. 2013; Martin et al. 2015; Oberosler et al. 2017; da Silva et al. 2018). In this study, the activity rates of 16 species (of 29 analysed) were correlated with distance from the nearest clearing or road.

Within species, there was considerable variation in the edge response patterns observed at different sites. Ten species for which a correlation was observed had their activity modelled at multiple sites. Seven of these displayed no trend at one or more sites (Raffray's Bandicoot, Striped Bandicoot, Ground Cuscus, Small Dorcopsis, Wattled Brushturkey, Cinnamon Ground Dove and Papuan Logrunner), and one species exhibited a change in the direction of effect between sites, with an edge avoidance pattern displayed at one site and a reverse-pattern edge effect at another (New Guinea Scrubfowl) (Table 2.4, Figure 2.7).

Considering patterns across sites, the direction of observed effect tended to vary between BAAs. Edge avoidance patterns were most common at BAA 2 (11/14 BAA 2 models in Table 2.4). By contrast, nearly half of the models (4/9) in which activity at BAA 1 was correlated with infrastructure distance demonstrated a reverse-pattern edge effect, with higher rates of activity nearer to the forest edge. This was the only pattern observed at the Hides High site, and only two of the five cases consistent with edge avoidance at Hides Low (Striped Bandicoot and New Guinea Scrubfowl) were statistically conclusive (requiring both a significant correlation and a definitive rejection of the null model at $\Delta AIC_c \geq 6$). This shift in the direction of effect across BAAs was also observed in previous sampling years (Woxvold and Legra 2019a; Woxvold et al. 2020).

Reverse-pattern edge effects are counter-intuitive for interior forest species. In such cases, we think it unlikely that Project roads and clearings have 'improved' the quality of near-edge habitat. Instead, the causal factors are likely to be environmental rather than anthropogenic. Candidate parameters include those correlated with infrastructure distance at BAA 1, as well as unmodelled factors. Three examples were discussed in Woxvold et al. (2020) – terrain effects (steepness), understorey density and predator avoidance; overall, there is little evidence that pre-considered parameters are responsible for near-edge habitat preferences observed at BAA 1.

Regardless of the cause, after three years of camera trap monitoring there is little evidence that terrestrial mammal or bird species avoid forest edge on Hides Ridge. We did not place cameras within 5–10 m of clearings, and it is possible that some animals do avoid these near-edge habitats; for example, along the pipeline ROW at Hides Low where increased sunlight has led to the development of dense tangles of climbing bamboo (*Nastus productus*). However, for most species investigated in this study, these narrow strips of habitat represent only a small proportion of their home range requirement – most occupy home ranges larger than two hectares, and for animals larger than 0.5 kg, including all terrestrial IUCN Threatened and Near Threatened species examined here, the home range is typically in the order of 10s or 100s of hectares. Changes restricted to very near-edge environments are likely to have a minimal impact on such species, and we have therefore elected to investigate habitat effects in the range of 20 to >300 m from the forest edge. While it is possible that edge effects do impact terrestrial vertebrates at the measured scale, the responses on Hides Ridge are evidently negligible for most species (other than Striped Bandicoot and New Guinea Scrubfowl at Hides Low), and in some cases appear to have been reversed by the influence of one or more natural (non-anthropogenic) environmental factors.

Edge avoidance was the predominant pattern at BAA 2. Reduced activity near BAA 2 infrastructure was first observed in 2017 for Small Dorcopsis, Raffray's Bandicoot and Collared Brushturkey (Woxvold and Legra 2019a), and additionally in 2019 for Dwarf Cassowary, Pheasant Pigeon and Russet-tailed Thrush (Woxvold et al. 2020). The 2021 data consolidate these results and extend them to New Guinea Scrubfowl at Arakubi. Observed effects are particularly strong for Collared Brushturkey and New Guinea Scrubfowl at Arakubi, Raffray's Bandicoot and Russet-tailed Thrush at KP107, and Small Dorcopsis and Pheasant Pigeon at Arakubi and KP107.

Given the likelihood that unmodelled heterogeneity in environmental factors has led to spurious edge response patterns at BAA 1, it is possible that some of the patterns observed at BAA 2 are also environmentally driven. In the absence of pre-construction data, it is difficult to untangle the effects of infrastructure distance and environmental parameters. Nevertheless, there are a number of reasons why some species may avoid near-edge environments at these sites.

First, forest near infrastructure at the Arakubi site is in many places heavily disturbed. This was most evident at a small number of camera positions along the quarry road. This may partly explain apparent edge avoidance by Small Dorcopsis, Collared Brushturkey, New Guinea Scrubfowl and Pheasant Pigeon at Arakubi. However, forest condition was generally good near roads and the pipeline ROW at KP107, where edge avoidance was observed for five species.

Second, species may avoid near-edge habitat due to anthropogenic disturbance. Human disturbance is very difficult to quantify (da Silva et al. 2018), and distance from settlements is sometimes used as a proxy measure (e.g., Oberosler et al. 2017). Across the study area, settlement distance increases with elevation so that the BAA 2 sites are positioned closest to the nearest village (mean straight-line distances: Arakubi – 1.7 km, KP107 – 2.7 km, Hides Low – 3.6 km, Hides High – 5.7 km). Consistent with this, the daily rate of incursions by humans and/or dogs was highest at the BAA 2 sites in the last two sampling years (Figure 2.6). However, incursion rates at these sites were the lowest recorded across the study area in 2017, and in 2021 were similar to those recorded at Hides Low (BAA 1). Moreover, these incursions were recorded on camera traps inside forest environments, and humans and dogs readily travel much further into the forest interior than the distances covered by our camera arrays. Edge effects observed at the measured scale may instead reflect avoidance of frequent human activity along roads and the pipeline ROW.

Sampling design and analytical approach

We have discussed the suitability of our statistical approach to assessing edge effects (Objective 3) in an earlier report (Woxvold and Legra 2019a) and have continued to use the same approach here.

Similarly, in a previous report (Woxvold et al. 2020) we have discussed our tiered statistical approach to monitoring population change between years (Objective 2); in decreasing of order preference, and based on data availability and species-specific life history traits (home range size, see Methods), we have modelled occupancy estimates, activity rates, and calculated naïve occupancy rates.

Unfortunately, goodness of fit test results in the 2017–2021 dataset precluded modelling of occupancy estimates for all but one species (Striped Bandicoot at BAA 1). Consequently, we reverted to the use of activity rate models for all species with sufficient data to run statistical models. A number of studies have demonstrated that changes in activity rate do sometimes accurately reflect changes in abundance (Rovero and Marshall 2009; Kuprewicz 2013; Palmer et al. 2018; Broadley et al. 2019). Moreover, with repeat sampling from a standardised design, and where analysis is restricted to comparing activity rates within species and within sites, as done in this study, changes in activity rate are generally considered to provide a reliable indication of the direction of change (increase, decrease) in a species' local population density (O'Brien 2011). This approach is thus considered a suitable next-best option in cases where occupancy modelling cannot be applied.

Naïve occupancy measures are reported for conservation-priority species with too few data to run statistical models (of occupancy or activity rate). This was applied to four IUCN threatened mammal species – the Endangered Ifola and Goodfellow's Tree Kangaroo, and the Vulnerable Long-beaked Echidna and Pademelon – and to the IUCN Data

Deficient Woolley's Three-striped Dasyure. Because of imperfect detection ($p<1$), naïve occupancy estimates tend to underestimate true occupancy levels. However, naïve and model-based occupancy estimates are often strongly correlated (Rovero et al. 2014; Hegerl et al. 2015; Blake et al. 2017; Neilson et al. 2018). Naïve estimates therefore present a useful and best-available surrogate for assessing population trends in those priority species for which too few data are available to produce a model-based estimate.

The current sampling design is suitable for activity rate modelling of habitat preferences, such as the approach used in this study to assess potential edge effects. It is also practical in terms of the monitoring program's resource and logistic constraints, and as such has worked well in conjunction with the other PMA3 study disciplines. However, the current camera arrays limit our ability to monitor wildlife population trends in two ways.

First, the clustering of cameras within sites of c. 70–180 ha makes it difficult to draw conclusions as to the overall status of some local populations. For many medium- to large-bodied animals, only a small number of individuals, in some cases perhaps only one or two (for example, Long-beaked Echidna at Arakubi), may hold territories at each site. Relevant taxa include most terrestrial IUCN listed species recorded in our study. For these species, the localised loss of a small number of individuals – for example due to hunting – may lead to a dramatic decline in recorded observations but a limited ability to assess population trends at a more meaningful scale.

Second, we are constrained in our ability to use the preferred occupancy modelling approach. The relatively close spacing of cameras within sites violates a key assumption of occupancy modelling – that of independence – by allowing individuals of most species to be photographed at more than one camera position. Because of this, we have modelled occupancy among groups of cameras whose units are no less than 250 m apart, reducing the effective per-site sample size from 20 cameras deployed to 4–5 camera groups. Moreover, in order to improve sample size, the occupancy approach is necessarily restricted to a handful of smaller species whose home ranges are unlikely to span more than one camera group, but whose overall population spans both sites within a BAA. Relevant species include a small number of potentially sensitive terrestrial insectivorous birds, but exclude most ground-dwelling non-rodent taxa camera trapped in the study area, including all IUCN listed species.

After three years of edge effect monitoring, improvements in pattern resolution may be expected to diminish in future sampling seasons. Given the proven ability of camera traps to detect the presence of rare and elusive priority monitoring targets, it is therefore timely to consider how camera arrays might be adjusted to maximise return-for-effort and to provide useful data for long-term population studies. An occupancy-based sampling design would involve cameras positioned at 'independent' stations at least 500–1,000 m apart (e.g. Blake et al. 2017; Ehlers Smith 2017; Murphy et al. 2017), thereby sampling more animal territories across a broader area. The Upstream Project Area presents a number of challenges to establishing such arrays, including areas of difficult terrain, land ownership issues and logistic constraints. However, the Hides Ridge landscape presents a good opportunity to apply an occupancy-based design, since: (1) the area supports multiple priority monitoring target species; (2) it already accommodates the camera trap monitoring study, and; (3) a Project-controlled road provides access to more than 14 km of continuous forest above 2,000 m asl. Moreover, we do not expect further sampling with the current arrays to overturn the reverse-pattern edge effects observed on Hides Ridge.

Opportunities to expand the sampling area at BAA 2 are more limited, due in part to higher local human population densities and the overlapping interests of other oil and gas operators. However, any expansion in the number of independent camera groupings would be beneficial, and it may be possible to achieve this in areas along the pipeline ROW north of KP107, and to the southeast between Arakubi and KP107.

Conclusions

The following changes were made to sampling arrays in 2021:

- Three camera positions at Hides Low were relocated due to security concerns following the loss of cameras in 2019.

- Three full seasons of camera trap sampling have improved our knowledge of vertebrate diversity within the study area and across the broader PNG LNG Upstream Project Area. Results of the 2021 sampling year include two species newly reported from BAAs – Masked Ring-tailed Possum and Torresian Striped Possum.

- The biennial deployment period of 89–135 days is sufficient to attain a near-complete census of the resident terrestrial bird and medium- to large-bodied mammal faunas in each sampling year. Biennial data are provided for a suite of priority monitoring targets, including seven IUCN listed mammal species – Eastern Long-beaked Echidna, Woolley's Three-striped Dasyure, New Guinea Quoll, Small Dorcopsis, Pademelon, Ifola and Goodfellow's Tree Kangaroo.

- Occupancy estimates could not be obtained for most species. Consequently, we modelled activity rates to assess population change in all species with sufficient data to run statistical models.

- Compared to the 2017 results, activity rates declined significantly in 2021 in four mammals and six bird species at individual sites. Naïve occupancy measures fell sharply for four IUCN listed mammals at BAA 2 – the Eastern Long-beaked Echidna and Goodfellow's Tree Kangaroo at Arakubi, and Woolley's Three-striped Dasyure and the Pademelon at Arakubi and KP107. An increase in population estimate was recorded for a number of taxa, including two IUCN threatened mammals – the Eastern Long-beaked Echidna at Hides High and the Ifola at KP107 and Hides High. It is too early to draw conclusions as to the ongoing status of local populations; continued sampling, standardised for seasonal effects, is required to make reliable inferences about population trends.

- The rate of incursions by humans and/or dogs decreased at all sites in 2021 compared to 2019 levels. At BAA 1, incursion rates have fallen steadily since sampling began in 2017. By contrast, at the BAA 2 sites, both the 2019 and the 2021 incursion rates were higher than those recorded there in 2017. At Arakubi, where the decline in records of IUCN listed species has been strongest, most incursions have been recorded closer to the PNG LNG pipeline ROW than to Arakubi Quarry and its access road, suggesting that most hunting parties are accessing the area via the pipeline route.

- Animal activity rate was correlated with distance from infrastructure in 16 species at 23 sites or BAAs. Final models were consistent with edge avoidance by 10 species at 16 sites/BAAs. Evidence for edge avoidance was strongest at BAA 2, particularly for New Guinea Scrubfowl at Arakubi, Raffray's Bandicoot and Russet-tailed Thrush at KP107, and Small Dorcopsis, Collared Brushturkey and Pheasant Pigeon at Arakubi and KP107. Some species may avoid near-edge environments at these sites due to the presence of degraded forest near infrastructure (particularly at Arakubi) and/or an aversion to frequent human activity along roads and the pipeline ROW.

- Seven species displayed reverse-pattern edge effects at individual sites, with higher rates of activity nearer to the forest edge. These were most commonly observed at BAA 1 (4/9 final models) and were the only patterns observed at the Hides High site. Reverse-pattern edge effects are likely due to unmodelled heterogeneity in environmental factors.

Recommendations

- We recommend that the camera trapping program continue in 2023 and in subsequent survey years. The biennial schedule should be maintained as fewer data points will extend the time required to make reasonable inferences about population trends. As far as practical, each monitoring survey should take place at the same time of year to control for seasonal effects.

- We recommend that consideration be given to expanding the camera trap sampling design within BAA 1 beyond the site scale, using the same number of cameras placed further apart along the length of Hides Ridge. There is little evidence of edge avoidance by terrestrial birds or mammals on Hides Ridge, and this change in design will improve our ability to assess population change in a variety of priority medium to large-bodied hunting-sensitive species. If accepted, a revised design including a set of proposed camera positions will be prepared prior to survey. Additional field assistants may be required to deploy cameras in new positions and to collect environmental covariate data in the time available.

- Sampling design should remain unchanged at BAA 2, given: (1) the strong evidence for edge effects at Arakubi and KP107, and; (2) the importance of continuing to monitor the only known population of the new-to-science jewel-babbler (*Ptilorrhoa* sp.) discovered at KP107.

- The R script used to analyse data in this study was developed in consultation with a biostatistician from the Arthur Rylah Institute (Department of Environment, Water, Land and Planning, government of Victoria). Analysis of future datasets will require expansion of the occupancy modelling procedure to incorporate data from additional years and the influence of environmental covariates (here applied only to activity rate modelling). We recommend funds continue to be made available for time with the ARI biostatistician to help expand the analysis protocol. The time required will be less than that used in previous years.

References

Ahumada, J. A., Hurtado, J. & Lizcano, D. 2013. Monitoring the status and trends of tropical forest terrestrial vertebrate communities from camera trap data: a tool for conservation. *PLoS One* 8:e73707.

Aplin, K. & Opiang, M. 2017. Non-volant mammals (rodents and marsupials). Pp. 141–208 *In*: Richards, S. (Editor) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Armstrong, K.N., Kale, E., Okena, D., Yalang, J. & Billy, J. (in press) Small non-volant mammals (Rodents) *In*: Richards, S. (Editor) *Results of the fourth PMA3 biodiversity monitoring survey of the PNG LNG Upstream Project Area, 2–27 July 2021*. ExxonMobil PNG Limited, Port Moresby

Bartoń, K. 2015. MuMIn: Multi-model inference. Available at https://cran.r-project.org/web/packages/MuMIn/index.html

Beaudrot, L., Ahumada, J., O'Brien, T. G. & Jansen, P. A. 2019. Detecting tropical wildlife declines through camera-trap monitoring: an evaluation of the Tropical Ecology Assessment and Monitoring protocol. *Oryx* 53: 126–129.

Beirne, C., Pillco-Huarcaya, R., Serrano-Rojas, S. J. & Whitworth, A. 2017. Terrestrial camera traps: essential tool for the detection and future monitoring of the Critically Endangered Sira Curassow *Pauxi koepckeae*. *Endangered Species Research* 32: 145–152.

Bennett, E. L. & Robinson, J. G. 2000. *Hunting of wildlife in tropical forests: Implications for biodiversity and forest peoples.* World Bank Environment Department, Paper no. 76. The World Bank, Washington, D. C.

Blake, J. G. & Loiselle, B. A. 2015. Enigmatic declines in bird numbers in lowland forest of eastern Ecuador may be a consequence of climate change. *PeerJ* 3:e1177.

Blake, J. G., Mosquera, D., Loiselle, B. A., Swing, K. & Romo, D. 2017. Long-term variation in abundance of terrestrial mammals and birds in eastern Ecuador as measured by photographic rates and occupancy estimates. *Journal of Mammalogy*, 98: 1168–1178.

Bodmer, R. E. 1995. Managing Amazonian wildlife: Biological correlates of game choice by detribalized hunters. *Ecological Applications* 5: 872–877.

Broadley, K., Burton, A. C., Avgar, T. & Boutin, S. 2019. Density-dependent space use affects interpretation of camera trap detection rates. *Ecology and Evolution* 9: 14031–14041.

Bryan, J. E. & Shearman, P. L. (Comps) 2008. *Papua New Guinea Resource Information System Handbook*. 3rd edition. University of Papua New Guinea, Port Moresby.

Burivalova, Z., Sekercioğlu, Ç. H. & Koh, L. P. 2014. Thresholds of logging intensity to maintain tropical forest biodiversity. *Current Biology* 24: 1893–1898.

Burton, A. C., Neilson, E., Moreira, D., Ladle, A., Steenweg, r., Fisher, J. T., Bayne, E. & Boutin, S. 2015. Wildlife camera trapping: a review and recommendations for linking surveys to ecological processes. *Journal of Applied Ecology* 52: 675–685.

Cassano, C. R., Barlow, J. & Pardini, R. 2014. Forest loss or management intensification? Identifying causes of mammal decline in cacao agroforests. *Biological Conservation* 169: 14–22.

Costantini, D., Edwards, D. P. & Simons, M. J. P. 2016. Life after logging in tropical forests of Borneo: A meta-analysis. *Biological Conservation* 196: 182–188.

Crome, F. H. J. (Editor) 2008. Papua New Guinea Liquefied Natural Gas Project: Terrestrial biodiversity analysis of the upstream project area. Appendix 1 in: *The PNG LNG Project Environmental Impact Statement*. Prepared by Coffey Natural Systems for Esso Highlands Limited.

Crooks, K. R., Burdett, C. L., Theobald, D. M., Rondinini, C. & Boitani, L. 2011. Global patterns of fragmentation and connectivity of mammalian carnivore habitat. *Philosophical Transactions of the Royal Society of London* Series B 366: 2642–2651.

da Silva, M. X., Paviolo, A., Tambosi, L. R. & Pardini, R. 2018. Effectiveness of Protected Areas for biodiversity conservation: mammal occupancy patterns in the Iguaçu National Park, Brazil. *Journal for Nature Conservation* 41: 51–61.

Denys, C., Taylor, P. J. & Aplin, K. P. 2017. Family Muridae (true mice and rats, gerbils and relatives). Pp. 536–884 *In*: Wilson, D. E., Lacher, T. E. & Mittermeier, R. A. (Editors) *Handbook of the mammals of the world*. Vol. 7. Rodents II. Lynx Edicions, Barcelona.

Dickman, C. R. 2015. Family Peramelidae (bandicoots and echymiperas). Pp. 362–398 *In*: Wilson, D. E. & Mittermeier, R. A. (Editors) *Handbook of the mammals of the world*. Vol. 5. Monotremes and marsupials. Lynx Edicions, Barcelona.

Dinata,Y., Nugroho, A., Haidir, I. A. & Linkie, M. 2008. Camera trapping rare and threatened avifauna in west-central Sumatra. Bird Conservation International (2008) 18:30–37.

Ehlers Smith, D. A., Ehlers Smith, Y. C., Ramesh, T. & Downs, C. T. 2017. Camera-trap data elucidate habitat requirements and conservation threats to an endangered forest specialist, the Spotted Ground Thrush (*Zoothera guttata*). *Forest Ecology and Management* 400: 523–530.

Eldridge, M. D. B. & Coulson, G. M. 2015. Family Macropodidae (kangaroos and wallabies). Pp. 630–735 *In*: Wilson, D. E. & Mittermeier, R. A. (Editors) *Handbook of the mammals of the world*. Vol. 5. Monotremes and marsupials. Lynx Edicions, Barcelona.

Fiske, I. & Chandler, R. 2011. Unmarked: an R package for fitting hierarchical models of wildlife occurrence and abundance. *Journal of Statistical Software* 43: 1–23.

Gill, F., Donsker, D. & Rasmussen, P. (Editors). 2022. IOC World Bird List (v 12.1). Doi 10.14344/IOC.ML.12.1. http://www.worldbirdnames.org/

Grueber, C. E., Nakagawa, S., Lws, R. J. & Jamieson, I. G. 2011. Multimodel inference in ecology and evolution: challenges and solutions. *Journal of Evolutionary Biology* 24: 699–711.

Hegerl, C., Burgess, N. D., Nielsen, M. R., Martin, E., Ciolli, M. & Rovero, F. 2015. Using camera trap data to assess the impact of bushmeat hunting on forest mammals in Tanzania. *Oryx*, available on CJO2015. doi:10.1017/S0030605315000836.

IUCN 2022. *IUCN Red List of Threatened Species*. Version 2022.1. Available at: www.iucnredlist.org

Jansen, A. 1999. Home ranges and group-territoriality in Chowchillas *Orthonyx spaldingii*. *Emu* 99: 280–290.

Jerozolimski, A. & Peres, C. A. 2003. Bringing home the biggest bacon: a cross-site analysis of the structure of hunter-kill profiles in Neotropical forests. *Biological Conservation* 111: 415–425.

Johnson, A., Bino, R. & Igag, P. 2004. A preliminary evaluation of the sustainability of cassowary (Aves: Casuariidae) capture and trade in Papua New Guinea. *Animal Conservation* 7: 129–137.

Kattan, G. H. & Beltran, J. W. 2002. Rarity in antpittas: territory size and population density of five *Grallaria* spp. in a regenerating habitat mosaic in the Andes of Colombia. *Bird Conservation International* 12: 231−240.

Kuprewicz, E. K. 2013. Mammal abundances and seed traits control the seed dispersal and predation roles of terrestrial mammals in a Costa Rican forest. *Biotropica* 45: 333–342.

Lambert, F. R. & Collar, N. J. 2002. The future for Sundaic lowland forest birds: long-term effects of commercial logging and fragmentation. *Forktail* 18: 127–146.

Latta, S. C., Tinoco, B. A., Astudillo, P. X. & Graham, C. H. 2011. Patterns and magnitude of temporal change in avian communities in the Ecuadorian Andes. *Condor* 113: 24–40.

Laurance, S. G. W., Stouffer, P. C. & Laurance, W. F. 2004. Effects of road clearing on movement patterns of understory rainforest birds of central Amazonia. *Conservation Biology* 18: 1099–1109.

Laurance, W. F. & Bierregaard, R. O. Jr. 1997. *Tropical forest remnants. Ecology, management, and conservation of fragmented communities*. University of Chicago Press, Chicago.

Leblond, M., Dussault, C. & Ouellet, J. P. 2013. Avoidance of roads by large herbivores and its relation to disturbance intensity. *Journal of Zoology* 289: 32–40.

Lessa, I., Guimarães, T. C. S., Bergallo, H. G., Cunha, A. & Vieira, E. M. 2016. Domestic dogs in protected areas: a threat to Brazilian mammals? *Natureza & Conservação* 14: 1–34.

Li, X., Bleisch, W. V. & Jiang, X. 2018. Using large spatial scale camera trap data and hierarchical occupancy models to evaluate species richness and occupancy of rare and elusive wildlife communities in southwest China. *Diversity and Distributions* 24: 1560–1572.

Lindsell, J. A. 2001. Th ranging behaviour of a tropical forest terrestrial insectivore: the Scaly-breasted Illadopsis. *Ostrich* Supplement No. 15: 92–97.

MacKenzie, D. I., Nichols, J. D., Royle, J. A., Pollock, K. H., Bailey, L. L. & Hines, J. E. 2006. *Occupancy estimation and modeling: inferring patterns and dynamics of species occurrence*. Academic Press, Burlington, Massachusetts.

Mamu, T. 2008. Environmental Impact Study of Terrestrial Mammals for the PNG LNG Project, Papua New Guinea. Annex 4 in: Crome, F. H. J. et al. 2008. Terrestrial biodiversity analysis of the upstream project area. Papua New Guinea Liquefied Natural Gas Project. Appendix 1 in: *The PNG LNG Project Environmental Impact Statement*. Prepared by Coffey Natural Systems for Esso Highlands Limited.

Martin, E. H., Cavada, N., Ndibalema, V. G. & Rovero, F. 2015. Modelling fine-scale habitat associations of medium-to-large forest mammals in the Udzungwa Mountains of Tanzania using camera trapping. *Tropical Zoology* 28: 137–151.

McAlpine, J., Keig, G. & Falls, R. 1983. *Climate of Papua New Guinea*. Australian National University, Canberra.

Michalski, F. & Peres, C. A. 2007. Disturbance-Mediated Mammal Persistence and Abundance-Area Relationships in Amazonian Forest Fragments. *Conservation Biology* 21: 1626–1640.

Murphy, A. J., Farris, Z. J., Karpanty, S., Kelly, M. J., Miles, K. A., Ratelolahy, F., Rahariniaina, R. P. & Golden, C. D. 2017. Using camera traps to examine distribution and occupancy trends of ground-dwelling rainforest birds in north-eastern Madagascar. *Bird Conservation International*, 1-14. doi:10.1017/S0959270917000107

Murphy, A. J., Kelly, M. J., Karpanty, S. M., Andrianjakarivelo, V. & Farris, Z. J. 2018. Using camera traps to investigate spatial co-occurrence between exotic predators and native prey species: a case study from northeastern Madagascar. *Journal of Zoology* 307: 264–273.

Namo, R. 2004. *Annotated checklist of mammals of Kikori Basin*. WWF KICDP Area Report.

Neilson, E. W., Avgar, T., Burton, A. C., Broadley, K. & Boutin, S. 2018. Animal movement affects interpretation of occupancy models from camera-trap surveys of unmarked animals. *Ecosphere* 9: e02092.

Nicol, S. C. 2015. Tachyglossidae (echidnas). Pp. 34–56 *In*: Wilson, D. E. & Mittermeier, R. A. (Editors) *Handbook of the mammals of the world*. Vol. 5. Monotremes and marsupials. Lynx Edicions, Barcelona.

Niedballa, J., Sollmann, R., Courtiol, A. & Wilting, A. 2016. camtrapR: an R package for efficient camera trap data management. *Methods in Ecology and Evolution* 7: 1457–1462.

Oberosler, V., Groff, C., Iemma, A., Pdrini, P. & Rovero, F. 2017. The influence of human disturbance on occupancy and activity patterns of mammals in the Italian Alps from systematic camera trapping. *Mammalian Biology* 87: 50–61.

O'Brien, T. G. 2011. Abundance, density and relative abundance: a conceptual framework. Pp. 71–96 *In*: O'Connell, A. F., Nichols, J. D. & Karanth, K. U. (Editors) *Camera traps in animal ecology: methods and analyses.* Springer, New York.

O'Brien, T. G., Ahumada, J., Akampurila, E., Beaudrot, L., Boekee, K., Brncic, T., Hickey, J., Jansen, P. A., Kayijamahe, C., Moore, J., Badru, M., Mulindahabi, F., Ndoundou-Hockemba, M., Niyigaba, P., Nyiratuza, M., Opepa, C. K., Rovero, F., Uzabaho, E. & Strindberg, S. 2019. Camera trapping reveals trends in forest duiker populations in African National Parks. *Remote Sensing in Ecology and* Conservation. https://doi.org/10.1002/rse2.132

O'Brien, T. G. & Kinnaird, M. F. 2008. A picture is worth a thousand words: the application of camera trapping to the study of birds. *Bird Conservation International* 18: S144–S162.

O'Connell, A. F., Nichols, J. D. & Karanth, K. U. (Editors) 2011.*Camera traps in animal ecology: methods and analyses*. Springer, New York.

Opiang, M. D. 2009. Home ranges, movement, and den use in Long-beaked Echidnas, *Zaglossus bartoni*, from Papua New Guinea. *Journal of Mammalogy* 90: 340–346.

Palmer, M. S., Swanson, A., Kosmala, M., Arnold, T. & Packer, C. 2018. Evaluating relative abundance indices for terrestrial herbivores from large-scale camera trap surveys. *African Journal of Ecology* 56: 791–803.

Pardo, L. E., Edwards, W., Campbell, M. J., Gómez-Valencia, B., Reuben Clements, G. & Laurance, W. F. 2021. Effects of oil palm and human presence on activity patterns of terrestrial mammals in the Colombian Llanos. *Mammalian Biology* 101: 775–789.

Peh, K. S.-H., de Jong, J., Sodhi, N. S., Lim, S. L.-H., & Yap, C. A.-M. 2005. Lowland rainforest avifauna and human disturbance: persistence of primary forest birds in selectively logged forests and mixed-rural habitats of southern Peninsular Malaysia. *Biological Conservation* 123: 489–505.

Peres, C. A. & Palacios, E. 2007. Basin-wide effects of game harvest on vertebrate population densities in Amazonian forests: Implications for animal-mediated seed dispersal. *Biotropica* 39: 304–315.

Peters, W., Hebblewhite, M., Cavedon, M., Pedrotti, L., Mustoni, A., Zibordi, F. & Cagnacci, F. 2015. Resource selection and connectivity reveal conservation challenges for reintroduced brown bears in the Alps. *Biological Conservation* 186: 123–133.

Pettorelli, N., Lobora, A. L., Msuha, M. J., Foley, C. & Durant, S. M. 2010. Carnivore biodiversity in Tanzania: revealing the distribution patterns of secretive mammals using camera traps. *Animal Conservation* 13: 131–139.

R Development Core Team, 2015. *R: a Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria http://www.R-project.org.

Rao, M., Myint, T., Zaw, T. & Htun, S. 2005 Hunting patterns in tropical forests adjoining the Hkakaborazi National Park, north Myanmar. *Oryx* 39: 292–300.

Richards, S. A. 2005. Testing ecological theory using the information-theoretic approach: examples and cautionary results. *Ecology* 86: 2,805–2,814.

Rovero, F. & Marshall, A. R. 2009. Camera trapping photographic rate as an index of density in forest ungulates. *Journal of Applied Ecology* 46: 1,011–1,017.

Rovero, F., Martin, E., Rosa, M., Ahumada, J. A. & Spitale, D. 2014. Estimating species richness and modelling habitat preferences of tropical forest mammals from camera trap data. *PLoS one* 9: e103300.

Rovero, F., Owen, N., Jones, T., Canteri, E., Iemma, A. & Tattoni, C. 2017. Camera trapping surveys of forest mammal communities in the Eastern Arc Mountains reveal generalized habitat and human disturbance responses. *Biodiversity and Conservation* 26:1103–1119.

Rovero, F. & Spitale, D. 2016.Species-level occupancy analysis. Pp. 68–94 *in* Rovero, F. & Zimmermann, F. (eds). *Camera trapping for wildlife research*. Pelagic Publishing, Exeter, UK.

Sampaio, R., Lima, A. P., Magnusson, W. E. & Peres, C. A. 2010. Long-term persistence of midsized to large-bodied mammals in Amazonian landscapes under varying contexts of forest cover. *Biodiversity and Conservation* 19: 2421–2439.

Sollmann, R. 2018. A gentle introduction to camera-trap data analysis. *African Journal of Ecology* 56: 740–749.

Srbek-Araujo, A. C. & Chiarello, A. G. 2013.Influence of camera-trap sampling design onmammal species capture rates and community structures in southeastern Brazil. *Biota Neotropica* 13: 51–62.

Symonds, M. R. E. & Moussalli, A. 2011. A brief guide to model selection, multimodel inference and model averaging in behavioural ecology using Akaike's information criterion. *Behavioural Ecology and Sociobiology* 65: 13–21.

Thiollay, J.-M. 1997. Disturbance, selective logging and bird diversity: a Neotropical forest study. *Biodiversity and Conservation* 6: 1155–1173.

Thomas, M. L., Baker, L., Beattie, J. R. & Baker, A. M. 2020. Determining the efficacy of camera traps, live capture traps, and detection dogs for locating cryptic small mammal species. *Ecology and Evolution* 10: 1054–1068.

van der Ree, R., Smith, D. J. & Grilo, C. (eds) 2015. *Handbook of road ecology*. John Wiley & Sons, Oxford, UK.

Venter, S. & Ona, A. 2017. Vegetation. Pp. 1–52 *In*: Richards, S. (ed.) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Wilson, D. E. & Mittermeier, R. A. 2015. *Handbook of the Mammals of the World.* Vol. 5. Monotremes and Marsupials. Lynx Edicions, Barcelona.

Wilson, D. E., Lacher, T. E. Jr & Mittermeier, R. A. 2017. *Handbook of the Mammals of the World.* Vol. 7. Rodents II. Lynx Edicions, Barcelona.

Woxvold, I. A. & Aplin, K. 2017. Camera trap monitoring of terrestrial birds and mammals: a pilot study. Pp. 121–139 *In*: Richards, S. J. (ed.) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Woxvold, I. A. & Legra, L. 2017. Birds. Pp. 91–120 *In*: Richards, S. J. (ed.) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Woxvold, I. A. & Legra, L. 2019a. Chapter 3. Camera trap monitoring of terrestrial mammals and birds. Pp. 63–120 *In*: Richards, S. J. (ed.) *Results of the second PMA3 biodiversity monitoring survey of the PNG LNG Upstream Project Area, 10–31 May 2017.* ExxonMobil PNG Limited, Port Moresby.

Woxvold, I. A. & Legra, L. 2019b. A population of Greater Ground Robin *Amalocichla sclateriana* (Petroicidae) from central New Guinea. *Bulletin of the British Ornithologists' Club* 139: 85–87.

Woxvold, I. A., Tulai, S., Yama, S. & Mani, A. 2020. Chapter 2. Camera trap monitoring of terrestrial mammals and birds. Pp. 53–112 *In*: Richards, S. J. (ed.) *Results of the third PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 8 August–2 September 2019*. ExxonMobil PNG Limited, Port Moresby.

Zimbres, B., Peres, C. A. & Machado, R. B. 2017. Terrestrial mammal responses to habitat structure and quality of remnant riparian forests in an Amazonian cattle-ranching landscape. *Biological Conservation* 206: 283–292.

Plate 1

Figure 2.8. Eastern Long-beaked Echidna (*Zaglossus bartoni*).

Figure 2.9. Short-beaked Echidna (*Tachyglossus aculeatus*).

Figure 2.10. Speckled Dasyure (*Neophascogale lorentzii*).

Figure 2.11. New Guinea Quoll (*Dasyurus albopunctatus*).

Figure 2.12. *Murexia* sp.

Figure 2.13. Raffray's Bandicoot (*Peroryctes raffrayana*).

Plate 2

Figure 2.14. Common Echymipera (*Echymipera* cf. *kalubu*).

Figure 2.15. Long-nosed Echymipera (*Echymipera rufescens*).

Figure 2.16. Striped Bandicoot (*Microperoryctes longicauda*).

Figure 2.17. Masked Ring-tailed Possum (*Pseudochirulus larvatus*).

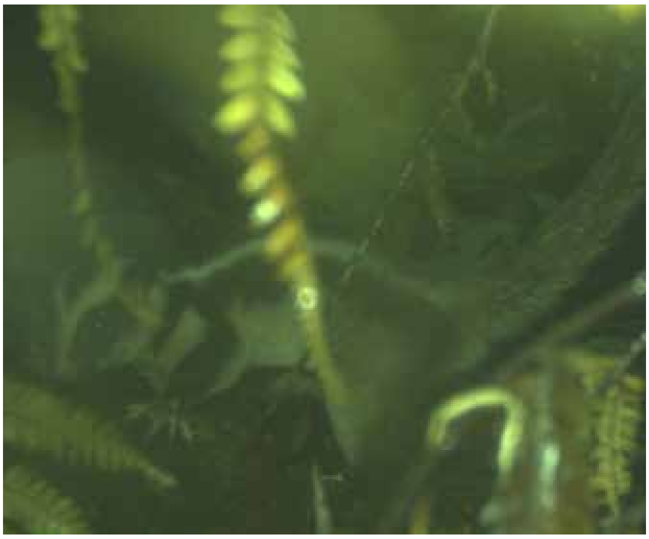


Figure 2.18. Torresian Striped Possum (*Dactylopsila trivirgata*).

Figure 2.19. Small Dorcopsis (*Dorcopsulus vanheurni*).

Plate 3

Figure 2.20. Ifola (*Dendrolagus* [*dorianus*] *notatus*).

Figure 2.21. A woolly giant rat (*Mallomys* sp.).

Figure 2.22. Black-tailed Giant Rat (*Uromys anak*).

Figure 2.23. Feral Pig (Sus *scrofa* x *celebensis*).

Figure 2.24. Domestic Cat (*Felis catus*).

Figure 2.25. Domestic Dog *(Canis familiaris*).

Plate 4

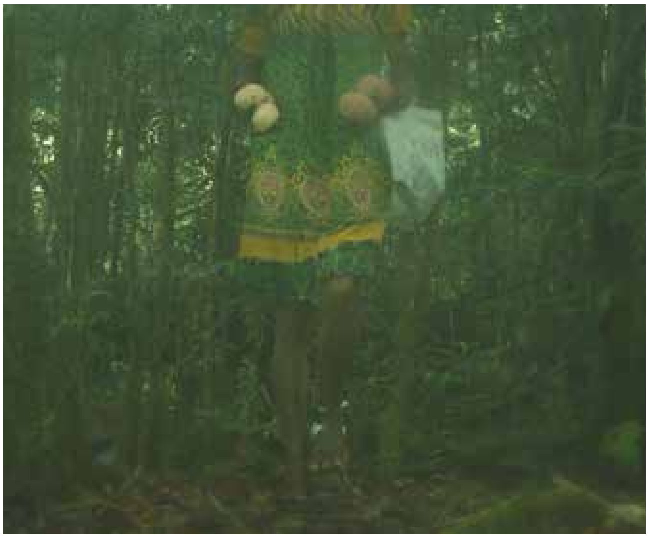


Figure 2.26. Local resident with megapode eggs at Arakubi.

Figure 2.27. Dwarf Cassowary (*Casuarius bennetti*).

Figure 2.28. Wattled Brushturkey (*Aepypodius arfakiensis*).

Figure 2.29. Collared Brushturkey (*Talegalla jobiensis*).

Figure 2.30. New Guinea Scrubfowl (*Megapodius decollatus).*

Figure 2.31. New Guinea Bronzewing (*Henicophaps albifrons*).

Plate 5

Figure 2.32. Cinnamon Ground Dove (*Gallicolumba rufigula*).

Figure 2.33. Bronze Ground Dove (*Pampusana beccarii*).

Figure 2.34. Pheasant Pigeon (*Otidiphaps nobilis*).

Figure 2.35. Forbes's Forest Rail (*Rallicula forbesi*).

Figure 2.36. Papuan Logrunner *(Orthonyx novaeguineae*).

Figure 2.37. Chestnut-backed Jewel-babbler (*Ptilorrhoa castanonota*).

Plate 6

Figure 2.38. Piping Bellbird (*Ornorectes cristatus*).

Figure 2.39. Lesser Melampitta (*Melampitta lugubris*).

Figure 2.40. Greater Melampitta (*Megalampitta gigantea*).

Figure 2.41. Papuan Scrub Robin (*Drymodes beccarii*).

Figure 2.42. Greater Ground Robin (*Amalocichla sclateriana*).

Figure 2.43. Russet-tailed Thrush (*Zoothera heinei*).

Appendix 2.1.

Species recorded on camera traps in 2017, 2019 and 2021, their conservation status (Status) and the number of independent photographic events (No. events) at each site. Species with predominantly or entirely terrestrial habits are indicated by (T) after the scientific name. Conservation status indicates those species listed in the IUCN Red List of Threatened Species (IUCN 2022) as Endangered (EN), Vulnerable (VU), Near Threatened (NT) or Data Deficient (DD) and those Protected (P) under the *PNG Fauna (Protection and Control) Act* 1966. Species not given an IUCN category are either Least Concern (most species) or Not Evaluated.

English Name	**Scientific Name**	**Status**	**No. events**				
			Arakubi	**KP107**	**Hides Low**	**Hides High**	**Total**
Mammals							
Eastern Long-beaked Echidna	*Zaglossus bartoni* (T)	VU, P	6/–/–	–/1/–	–/–/1	3/4/13	9/5/14
Short-beaked Echidna	*Tachyglossus aculeatus* (T)		10/13/15	2/2/4			12/15/19
Woolley’s Three-striped Dasyure	*Myoictis leucura* (T)	DD	4/18/6	5/1/–			9/19/6
Speckled Dasyure	*Neophascogale lorentzii* (T)				9/3/12	10/22/12	19/25/24
Narrow-striped Dasyure	*Phascolosorex dorsalis* (T)				18/19/51	2/2/2	20/21/53
Unknown diurnal dasyurid	*Neophascogale/ Phascolosorex* (T)				–/1/5		–/1/5
New Guinea Quoll	*Dasyurus albopunctatus* (T)	NT	13/10/17	9/22/6	6/8/21	16/17/27	44/57/71
Multiple *Murexia* spp.	*Murexia* spp. (T)		19/30/17	18/11/8	25/23/22	14/1/14	76/65/61
Raffray’s Bandicoot	*Peroryctes raffrayana* (T)		31/31/15	167/262/236	98/79/119	460/444/353	756/816/723
Common echymipera	*Echymipera* cf. *kalubu* (T)		4/3/7	1/1/0			5/4/7
Long-nosed Echymipera	*Echymipera rufescens (T)*		6/11/32	1/15/6			7/26/38
Striped Bandicoot	*Microperoryctes longicauda* (T)			7/7/3	147/158/139	172/262/209	326/427/351
Unidentified bandicoot	*Peroryctes/ Echymipera* sp. (T)		21/32/28	14/9/7	3/6/1	5/8/10	43/55/46
Ground Cuscus	*Phalanger gymnotis* (T)		21/16/8	14/38/19	14/7/29	5/6/2	54/67/58
Mountain Cuscus	*Phalanger carmelitae*					–/3/–	–/3/–

English Name	Scientific Name	Status	No. events				
			Arakubi	KP107	Hides Low	Hides High	Total
Masked Ring-tailed Possum	*Pseudochirulus larvatus*				–/–/1		–/–/1
A Ring-tailed Possum	*Pseudochirops* sp.				1/5/22	4/1/1	5/6/23
Long-fingered Striped Possum	*Dactylopsila palpator*				1/1/4	3/5/7	4/6/11
Torresian Striped Possum	*Dactylopsila trivirgata*			–/–/1			–/–/1
Small Dorcopsis	*Dorcopsulus vanheurni* (T)	NT	279/255/158	337/559/342	128/220/368	210/166/258	954/1,200/1,126
A pademelon	*Thylogale* sp. (T)	VU	7/–/–	16/11/1			23/11/1
Dorcopsis/ pademelon	*Dorcopsulus/ Thylogale* sp. (T)	NT/ VU	1/–/1	1/1/3			2/1/4
Ifola	*Dendrolagus [dorianus] notatus*	EN, P	3/5/2	3/9/12		2/4/4	8/18/18
Goodfellow's Tree Kangaroo	*Dendrolagus goodfellowi*	EN, P	4/2/–				4/2/–
Uneven-toothed Rat	*Anisomys imitator* (T)			49/33/29	14/31/44	67/29/67	130/93/140
Loria's Tree Mouse	*Pogonomys* cf. *loriae*			–/1/–	–/–/1		–/1/1
New Guinea Waterside Rat	*Parahydromys asper* (T)			–/–/1	–/–/2	–/3/6	–/3/9
A white-eared giant rat	*Hyomys* sp. (T)				2/1/5	4/31/17	6/32/22
Elegant Water Rat	*Leptomys elegans* (T)		16/38/3	48/24/56			64/62/59
A woolly giant rat	*Mallomys* sp. (T)			5/2/2	9/12/16	31/8/4	45/22/22
Multiple *Paramelomys* species	*Paramelomys* spp. (T)		108/88/62	81/80/135	90/137/175	49/42/120	328/347/357
Shrew Mouse	*Pseudohydromys* sp(p). (T)				–/1/1	–/–/1	–/1/2
Black-tailed Giant Rat	*Uromys anak (T)*		4/1/–	10/12/5	26/13/8	42/20/35	82/46/48
White-tailed Giant Rat	*Uromys caudimaculatus* (T)		82/120/63	27/54/63	–/1/–		109/175/126
Multiple small *Rattus* species	*Rattus* spp. (T)		21/–/–	2/2/4	44/78/100	210/185/124	277/265/228

English Name	**Scientific Name**	**Status**	**No. events**				
			Arakubi	**KP107**	**Hides Low**	**Hides High**	**Total**
A distinctive large *Rattus*	*Rattus* sp. A (T)		14/15/8				14/15/8
Unidentified murids			26/70/47	35/81/74	48/97/211	52/118/365	161/366/697
Feral Pig	*Sus scrofa* x *celebensis* (T)		2/–/2	4/–/–			6/–/2
Domestic Cat	*Felis catus* (T)		–/1/–	–/–/2	1/–/–		1/1/2
Domestic Dog	*Canis familiaris* (T)		–/19/22	–/10/7	3/8/5	3/6/1	6/43/35
Human	*Homo sapiens* (T)		2/–/5	–/–/2	6/–/1	–/1/–	8/1/8
Human with dog	(T)		–/–/5	–/1/–	2/–/–	2/–/–	4/1/5
Total incursion days, human &/or dog	(T)		2/16/17	–/8/7	7/6/6	4/3/1	13/33/31
Birds							
Dwarf Cassowary	*Casuarius bennetti* (T)		70/23/51	29/15/14	3/1/19	1/–/–	103/39/84
Wattled Brushturkey	*Aepypodius arfakianus* (T)		2/3/10	24/42/11	3/10/49	77/–/–	106/55/70
Collared Brushturkey	*Talegalla jobiensis* (T)		59/51/239	113/20/48	–/1/–		172/72/287
New Guinea Scrubfowl	*Megapodius decollatus* (T)		41/43/23	58/69/50	25/44/58	–/2/–	124/158/131
Mountain Owlet-nightjar	*Aegotheles albertisi*				–/–/1		–/–/1
(Terborgh's) Barred Owlet-nightjar	*Aegotheles bennetti terborghi*			–/2/–			–/2/–
Stephan's Emerald Dove	*Chalcophaps stephani* (T)		2/5/4				2/5/4
New Guinea Bronzewing	*Henicophaps albifrons* (T)		1/1/2	3/1/6	–/1/1		4/3/9
Cinnamon Ground Dove	*Gallicolumba rufigula* (T)		38/15/27	7/9/14			45/24/41
White-breasted Ground Dove	*Pampusana jobiensis* (T)				1/–/–		1/–/–
Bronze Ground Dove	*Pampusana beccarii* (T)				10/7/15	46/9/14	56/16/29

English Name	Scientific Name	Status	No. events				
			Arakubi	KP107	Hides Low	Hides High	Total
Pheasant Pigeon	*Otidiphaps nobilis* (T)		155/88/114	40/119/36	3/3/4		198/210/154
Papuan Mountain Pigeon	*Gymnophaps albertisii*					1/–/–	1/–/–
Chestnut Forest Rail	*Rallicula rubra* (T)					32/6/36	32/6/36
Forbes's Forest Rail	*Rallicula forbesi* (T)			5/1/1	14/10/7		19/11/8
Bare-eyed Rail	*Gymnocrex plumbeiventris* (T)			8/7/8		–/1/–	8/8/8
New Guinea Woodcock	*Scolopax rosenbergii* (T)				–/–/2	124/26/4	124/26/6
Papuan Eagle	*Harpyopsis novaeguineae*	VU, P	1/–/–				1/–/–
Grey-headed Goshawk	*Accipiter poliocephalus*		1/–/–				1/–/–
Shovel-billed Kookaburra	*Clytoceyx rex*				3/–/1	–/3/–	3/3/1
Dusky Lory	*Pseudeos fuscata*			–/4/–			–/4/–
Papuan Pitta	*Erythropitta macklotii* (T)		4/13/10	139/121/71			143/134/81
Ochre-breasted Catbird	*Ailuroedus stonii*		3/1/2	–/1/1			3/2/3
Black-capped Catbird	*Ailuroedus melanocephalus*			6/–/–			6/–/–
Archbold's Bowerbird	*Archboldia papuensis*	NT				1/–/1	1/–/1
MacGregor's Bowerbird	*Amblyornis macgregoriae*				–/1/5	2/–/–	2/1/5
Honeyeater sp.	*Ptiloprora* sp.					–/–/1	–/–/1
Long-billed Honeyeater	*Melilestes megarhynchus*		1/–/–	–/–/1			1/–/1
Rusty Mouse-warbler	*Origma murina* (T)		7/2/3				7/2/3
Mountain Mouse-warbler	*Origma robusta* (T)				1/2/2	2/11/17	3/13/19
Large Scrubwren	*Sericornis nouhuysi*					1/–/–	1/–/–

English Name	Scientific Name	Status	No. events				
			Arakubi	KP107	Hides Low	Hides High	Total
Papuan Logrunner	*Orthonyx novaeguineae* (T)				35/25/33	28/2/2	63/27/35
Crested Satinbird	*Cnemophilus macgregorii*					3/–/4	3/–/4
Fan-tailed Berrypecker	*Melanocharis versteri*				–/1/1	–/–/1	–/1/2
Spotted Jewel-babbler	*Ptilorrhoa leucosticta* (T)				62/96/48	18/3/15	80/99/63
Chestnut-backed Jewel-babbler	*Ptilorrhoa castanonota* (T)		22/18/9	21/18/51			43/36/60
Jewel-babbler sp.	*Ptilorrhoa* sp. (T)			3/3/2			3/3/2
Black Butcherbird	*Melloria quoyi*		–/2/–	–/–/1			–/2/1
Rufous-naped Bellbird	*Aleadryas rufinucha* (T)				19/35/49	17/16/69	36/51/118
Piping Bellbird	*Ornorectes cristatus* (T)		17/18/11	20/26/19			37/44/30
Brown-backed Whistler	*Pachycephala modesta*					–/–/1	–/–/1
Sclater's Whistler	*Pachycephala soror*				1/–/–		1/–/–
Variable Shrikethrush	*Colluricincla fortis*		–/1/–				–/1/–
Hooded Pitohui	*Pitohui dichrous*			1/–/–			1/–/–
Sooty Thicket Fantail	*Rhipidura threnothorax*		–/1/–				–/1/–
Black Fantail	*Rhipidura atra*				1/6/4		1/6/4
Dimorphic Fantail	*Rhipidura brachyrhyncha*				–/1/–		–/1/–
Lesser Melampitta	*Melampitta lugubris* (T)				1/–/5	116/92/249	117/92/254
Greater Melampitta	*Megalampitta gigantea* (T)		2/1/7	7/10/9			9/11/16
Queen Carola's Parotia	*Parotia carolae*	P		4/–/1			4/–/1
Brown Sicklebill	*Epimachus meyeri*	P			3/3/12	18/4/1	21/7/13

English Name	Scientific Name	Status	No. events				
			Arakubi	KP107	Hides Low	Hides High	Total
Magnificent Bird-of-paradise	*Diphyllodes magnificus*	P		–/2/–			–/2/–
White-winged Robin	*Peneothello sigillata*					–/2/7	–/2/7
Slaty Robin	*Peneothello cyanus*				1/13/3		1/13/3
Black-throated Robin	*Plesiodryas albonotata*				–/–/1	–/1/1	–/1/2
Ashy Robin	*Heteromyias albispecularis*				19/4/49	16/36/88	35/40/137
Papuan Scrub Robin	*Drymodes beccarii* (T)		16/7/4	18/28/105			34/35/109
White-eyed Robin	*Pachycephalopsis poliosoma*			3/–/5			3/–/5
Greater Ground Robin	*Amalocichla sclateriana* (T)					80/15/53	80/15/53
Lesser Ground Robin	*Amalocichla incerta* (T)				135/138/45		135/138/45
Russet-tailed Thrush	*Zoothera heinei* (T)		3/11/3	23/50/40			26/61/43
Blue-faced(/Papuan) Parrotfinch	*Erythrura trichroa(/ papuana)*					–/2/–	–/2/–
Reptiles							
Monitor species	*Varanus indicus*-group		7/10/7	1/–/4			8/10/11

Appendix 2.2.

Model-averaged coefficient point estimates, standard errors (SE), *P*-values and the relative importance of each of the most influential covariates present in the ΔAICc<6 candidate set for each species-site/BAA. The number of models in which each variable appears, and (in brackets) the total number of models in the candidate set, is also shown. Results are presented for all covariates with a relative importance value >0.2, or for the three most important covariates in cases where fewer than three have a relative importance value >0.2. Cases where the estimate is larger than the standard error are shown in bold. Abbreviated variables: DClr – distance from clearing; LT50/LT100 – less/more than 50/100 m from clearing; DRd – distance from road; LT100Rd – less/more than 100 m from road; LR20/LR50/LR100 – local relief at the 20, 50 or 100 m radius scales; CnpHt – canopy height; TrSm/TrLge – density of trees > 10/30 cm dbh; UD1/UD2 – understorey density below 1/2 m; LfLit – leaf litter depth; Moss/Rock/Logs – proportion of the ground surface covered by moss, rock or large woody debris. Quadratic terms are denoted by the symbol ^ (e.g. Rock^2).

Taxa/sites/covariates	Estimate	SE	P	Rel. Imp.	No. models
Short-beaked Echidna (Tachyglossus aculeatus) – Arakubi					
Moss	0.050	0.140	0.724	0.156	11(63)
DRd	0.000	0.001	0.862	0.148	10
UD2	-0.001	0.013	0.928	0.130	10
Speckled Dasyure (*Neophscogale lorentzii*) – Hides High					
Rock	-0.042	0.154	0.790	0.293	80(238)
TrSm	-0.005	0.013	0.709	0.259	57
Moss	-0.085	0.229	0.713	0.252	63
Narrow-striped Dasyure (*Phascolosorex dorsalis*) – Hides Low					
LT50	0.290	0.530	0.588	0.330	34(84)
Year (2017:2019)	-0.090	0.285	0.755	0.199	20
Year (2017:2021)	-0.192	0.450	0.671		
Logs	-0.023	0.144	0.877	0.153	17
New Guinea Quoll (*Dasyurus albopunctatus*) – Arakubi					
Rock	0.016	0.058	0.793	0.164	9(55)
Moss	0.022	0.088	0.804	0.153	10
CnpHt	-0.012	0.034	0.714	0.142	9
New Guinea Quoll (*Dasyurus albopunctatus*) – KP107					
UD2	0.003	0.015	0.864	0.166	12(60)
LT50	0.083	0.296	0.781	0.159	10
CnpHt	0.007	0.027	0.798	0.140	12
New Guinea Quoll (*Dasyurus albopunctatus*) – Hides Low					
Moss	0.111	0.287	0.699	0.285	15(53)
Rock	0.338	1.332	0.803	0.236	13
LR100	0.009	0.021	0.668	0.179	6
New Guinea Quoll (*Dasyurus albopunctatus*) – Hides High					
CnpHt	0.033	0.053	0.539	0.427	51(121)
UD1	0.012	0.045	0.791	0.194	30
Rock	0.016	0.078	0.836	0.180	28

Taxa/sites/covariates	**Estimate**	**SE**	**P**	**Rel. Imp.**	**No. models**
New Guinea Quoll (*Dasyurus albopunctatus*) – BAA 1					
LR50	**0.045**	**0.018**	**0.014**	**0.926**	**60(71)**
LT50	0.047	0.212	0.828	0.207	21
UD1	0.047	0.212	0.828	0.191	19
Raffray's Bandicoot (*Peroryctes raffrayana*) – Arakubi					
Year (2017:2019)	**-0.034**	**0.267**	**0.902**	**0.657**	**48(90)**
Year (2017:2021)	**-0.632**	**0.546**	**0.250**		
CnpHt	0.055	0.062	0.382	0.590	48
Rock	-0.172	0.218	0.430	0.415	28
DRd	0.001	0.002	0.562	0.320	25
LR50	-0.022	0.037	0.554	0.287	28
Moss	0.039	0.199	0.845	0.208	19
Raffray's Bandicoot (*Peroryctes raffrayana*) – KP107					
LT50	**0.925**	**0.200**	**0.000**	**1.000**	**95(95)**
TrLge	0.040	0.040	0.330	0.573	35
UD2	0.005	0.009	0.580	0.336	30
Logs	-0.054	0.137	0.695	0.287	33
CnpHt	0.005	0.015	0.714	0.224	23
Year (2017:2019)	0.069	0.167	0.679	0.211	24
Year (2017:2021)	0.058	0.166	0.729		
Raffray's Bandicoot (*Peroryctes raffrayana*) – Hides Low					
Moss	-0.359	0.522	0.495	0.497	53(116)
TrSm	0.018	0.022	0.421	0.478	38
Moss^2	-0.070	0.116	0.548	0.353	33
LT100Rd	0.089	0.263	0.739	0.240	38
Raffray's Bandicoot (*Peroryctes raffrayana*) – Hides High					
Year (2017:2019)	0.015	0.187	0.939	**1.000**	**28(28)**
Year (2017:2021)	**-0.616**	**0.191**	**0.002**		
LfLit	**0.025**	**0.016**	**0.122**	**0.833**	19
Rock^2	-0.020	0.026	0.444	0.545	13
LR50	-0.015	0.017	0.354	0.532	10
CnpHt	-0.023	0.034	0.498	0.374	14
Striped Bandicoot (*Microperoryctes longicauda*) – Hides Low					
Year (2017:2019)	-0.199	0.272	0.477	**0.939**	**48(57)**
Year (2017:2021)	**-0.780**	**0.341**	**0.025**		
CnpHt	-0.041	0.054	0.446	0.401	22
LT100	0.158	0.256	0.541	0.378	21
LR20	0.028	0.041	0.500	0.331	20
TrSm	0.016	0.027	0.560	0.268	15
Moss	-0.128	0.325	0.695	0.266	13
UD2	0.006	0.016	0.736	0.200	13

Taxa/sites/covariates	Estimate	SE	P	Rel. Imp.	No. models
Striped Bandicoot (*Microperoryctes longicauda*) – Hides High					
TrLge	**-0.101**	**0.048**	**0.038**	**0.938**	**67(81)**
CnpHt	**0.080**	**0.042**	**0.056**	**0.909**	**66**
Year (2017:2019)	0.127	0.221	0.570	0.394	26
Year (2017:2021)	-0.078	0.184	0.679		
LR100	0.006	0.011	0.586	0.335	32
Rock	0.032	0.073	0.668	0.266	17
Ground Cuscus (*Phalanger gymnotis*) – Arakubi					
TrLge	0.057	0.097	0.558	0.314	36(102)
LT100Rd	-0.051	0.281	0.859	0.161	19
Moss	-0.030	0.120	0.803	0.158	19
Ground Cuscus (*Phalanger gymnotis*) – KP107					
CnpHt	0.029	0.041	0.485	0.454	77(182)
Year (2017:2019)	0.398	0.529	0.454	0.413	77
Year (2017:2021)	0.081	0.278	0.775		
Logs	0.052	0.191	0.787	0.187	36
Ground Cuscus (*Phalanger gymnotis*) – Hides Low					
LT100Rd	-0.479	0.639	0.458	0.464	45(91)
Logs	-0.054	0.193	0.784	0.181	20
UD1	0.001	0.054	0.991	0.136	18
Small Dorcopsis (*Dorcopsulus vanheurni*) – Arakubi					
TrSm	**0.101**	**0.029**	**0.001**	**1.000**	**12(12)**
UD1	**-0.151**	**0.066**	**0.025**	**0.929**	**9**
LT50	**1.031**	**0.648**	**0.115**	**0.789**	**8**
Year (2017:2019)	-0.013	0.205	0.950	0.497	5
Year (2017:2021)	-0.384	0.439	0.384		
LR100	0.008	0.012	0.497	0.377	4
CnpHt	0.019	0.040	0.632	0.211	4
Small Dorcopsis (*Dorcopsulus vanheurni*) – KP107					
LT50	**1.263**	**0.262**	**0.000**	**1.000**	**36(36)**
UD1	**0.181**	**0.069**	**0.009**	**0.904**	**27**
Year (2017:2019)	0.277	0.301	0.361	0.552	12
Year (2017:2021)	0.027	0.175	0.880		
CnpHt	0.006	0.016	0.717	0.209	9
Small Dorcopsis (*Dorcopsulus vanheurni*) – Hides Low					
Logs	-0.192	0.205	0.357	0.710	64(100)
DRd	**0.001**	**0.001**	**0.313**	**0.618**	**57**
Rock	0.066	0.384	0.867	0.217	26

Taxa/sites/covariates	Estimate	SE	P	Rel. Imp.	No. models
Small Dorcopsis (*Dorcopsulus vanheurni*) – Hides High					
Logs	**-0.389**	**0.258**	**0.133**	**0.750**	**116(168)**
UD1	0.027	0.047	0.573	0.367	56
TrSm	-0.004	0.008	0.588	0.348	51
CnpHt	0.011	0.031	0.737	0.252	46
UD2	0.004	0.014	0.751	0.225	43
DRd	0.000	0.001	0.739	0.220	39
Logs^2	0.044	0.127	0.730	0.205	38
Pademelon (*Thylogale* sp.) – KP107					
LfLit	-0.002	0.006	0.799	0.166	8(51)
TrSm	-0.005	0.016	0.774	0.162	9
Rock	-0.008	0.052	0.875	0.148	9
Dwarf Cassowary (*Casuarius bennetti*) – Arakubi					
Year (2017:2019)	-0.611	0.622	0.329	0.565	115(242)
Year (2017:2021)	-0.339	0.413	0.417		
CnpHt	0.037	0.052	0.481	0.420	91
LR100	-0.007	0.012	0.570	0.363	73
LT50	0.423	0.665	0.528	0.359	81
TrSm	-0.012	0.028	0.670	0.276	78
Dwarf Cassowary (*Casuarius bennetti*) – KP107					
TrSm	-0.018	0.031	0.564	0.334	50(131)
LT50	0.149	0.409	0.720	0.270	37
LR50	0.007	0.019	0.706	0.268	35
Moss	0.040	0.197	0.841	0.206	34
Dwarf Cassowary (*Casuarius bennetti*) – BAA2					
LT50	**0.635**	**0.514**	**0.219**	**0.759**	**94(148)**
Year (2017:2019)	**-0.656**	**0.553**	**0.237**	**0.645**	**85**
Year (2017:2021)	**-0.536**	**0.473**	**0.259**		
TrSm	-0.024	0.029	0.403	0.493	58
Station	-0.341	0.452	0.452	0.471	66
Wattled Brushturkey (*Aepypodius arfakiensis*) – KP107					
LR100	0.017	0.025	0.488	0.489	63(149)
Year (2017:2019)	0.122	0.272	0.659	0.371	49
Year (2017:2021)	-0.343	0.526	0.517		
LT100	-0.060	0.218	0.788	0.238	44
Wattled Brushturkey (*Aepypodius arfakiensis*) – Hides Low					
LT100	0.871	1.009	0.390	0.476	34(80)
LfLit	0.014	0.021	0.502	0.343	20
Moss	-0.086	0.269	0.753	0.215	20

Taxa/sites/covariates	Estimate	SE	P	Rel. Imp.	No. models
Collared Brushturkey (*Talegalla jobiensis*) – Arakubi					
Year (2017:2019)	0.024	0.326	0.943	**1.000**	**42(42)**
Year (2017:2021)	**1.268**	**0.298**	**0.000**		
DClr	**0.006**	**0.002**	**0.011**	**0.899**	29
Moss	0.084	0.148	0.577	0.368	11
UD2	-0.008	0.019	0.662	0.218	7
Collared Brushturkey (*Talegalla jobiensis*) – KP107					
Year (2017:2019)	**-1.649**	**0.411**	**0.000**	**1.000**	**127(127)**
Year (2017:2021)	**-1.061**	**0.385**	**0.007**		
Moss	-0.081	0.344	0.815	0.539	52
LR20	-0.022	0.036	0.548	0.349	36
TrLge	0.034	0.067	0.622	0.280	29
LT50	0.216	0.462	0.642	0.249	36
TrSm	-0.005	0.014	0.710	0.209	28
New Guinea Scrubfowl (*Megapodius decollatus*) – Arakubi					
LfLit	**-0.032**	**0.021**	**0.141**	**0.746**	**43(67)**
LT100Rd	1.283	1.293	0.324	0.603	26
TrSm	0.033	0.046	0.474	0.454	22
Logs	0.988	2.337	0.674	0.274	25
LR50	-0.027	0.052	0.596	0.241	22
Logs^2	0.188	0.430	0.664	0.203	19
New Guinea Scrubfowl (*Megapodius decollatus*) – KP107					
DRd	**-0.003**	**0.002**	**0.211**	**0.769**	**27(39)**
LfLit	**-0.019**	**0.016**	**0.232**	**0.745**	**30**
Logs	0.164	0.452	0.720	0.608	19
UD1	-0.129	0.142	0.367	0.531	17
Rock	0.067	0.124	0.592	0.327	14
LR20	-0.015	0.032	0.637	0.217	6
New Guinea Scrubfowl (*Megapodius decollatus*) – Hides Low					
LT100	**0.807**	**0.362**	**0.029**	**0.947**	**43(48)**
Moss	**-0.808**	**0.657**	**0.223**	**0.926**	**40**
Moss^2	-0.068	0.127	0.593	0.285	11
Cinnamon Ground Dove (*Gallicolumba rufigula*) – Arakubi					
DRd	-0.001	0.002	0.527	0.350	46(137)
LR100	0.008	0.015	0.577	0.311	41
TrLge	0.029	0.063	0.649	0.237	35
LfLit	0.003	0.007	0.707	0.236	37
Cinnamon Ground Dove (*Gallicolumba rufigula*) – KP107					
CnpHt	0.014	0.045	0.766	0.228	8(33)
LT100	-0.005	0.255	0.985	0.170	8
TrLge	0.015	0.063	0.811	0.160	6

Taxa/sites/covariates	**Estimate**	**SE**	**P**	**Rel. Imp.**	**No. models**
Bronze Ground Dove (Pampusana beccarii) – BAA 1					
Year (2017:2019)	**-1.185**	**0.431**	**0.006**	**0.966**	**71(76)**
Year (2017:2021)	**-1.020**	**0.388**	**0.009**		
LR50	**0.020**	**0.019**	**0.300**	**0.631**	**43**
LfLit	-0.002	0.006	0.716	0.198	12
Pheasant Pigeon (*Otidiphaps nobilis*) – Arakubi					
nMoss	**-0.551**	**0.269**	**0.043**	**1.000**	**47(47)**
DRd	**0.002**	**0.001**	**0.110**	**0.859**	**32**
UD2	-0.025	0.033	0.459	0.419	12
LR20	0.019	0.029	0.510	0.383	14
UD1	-0.030	0.058	0.609	0.261	17
Pheasant Pigeon (*Otidiphaps nobilis*) – KP107					
Year (2017:2019)	**1.000**	**0.305**	**0.001**	**1.000**	**36(36)**
Year (2017:2021)	-0.196	0.350	0.584		
LT50	**1.343**	**0.410**	**0.001**	**1.000**	36
TrLge	-0.015	0.038	0.701	0.248	7
Chestnut Forest Rail (*Rallicula rubra*) – Hides High					
TrSm	-0.012	0.023	0.604	0.298	70(231)
Rock	-0.003	0.316	0.991	0.285	75
UD2	0.019	0.042	0.652	0.256	59
LT50	0.168	0.453	0.713	0.225	62
Logs	-0.059	0.190	0.757	0.213	57
New Guinea Woodcock (*Scolopax rosenbergii*) – Hides High					
LR100	-0.016	0.034	0.634	0.264	75(242)
Logs	0.013	0.265	0.963	0.262	59
UD2	0.022	0.048	0.650	0.253	63
CnpHt	0.022	0.063	0.739	0.247	59
Year (2017:2019)	-0.124	0.370	0.742	0.225	65
Papuan Pitta (*Erythropitta macklotii*) – KP107					
Logs	**-1.727**	**1.595**	**0.284**	**0.971**	**40(43)**
LR50	-0.031	0.041	0.455	0.504	16
UD2	-0.039	0.061	0.520	0.395	12
LT100	-0.562	0.876	0.523	0.392	14
Moss	0.152	0.344	0.662	0.346	15
Logs^2	-0.138	0.291	0.639	0.295	16
Papuan Logrunner (*Orthonyx novaeguineae*) – Hides Low					
LT50	**-2.089**	**0.548**	**0.000**	**1.000**	**54(54)**
TrLge	-0.098	0.152	0.523	0.355	19
Logs	0.056	0.331	0.867	0.198	14
UD2	-0.007	0.031	0.820	0.157	11

Taxa/sites/covariates	**Estimate**	**SE**	**P**	**Rel. Imp.**	**No. models**
Papuan Logrunner (*Orthonyx novaeguineae*) – Hides High					
TrSm	-0.003	0.014	0.851	0.208	9(40)
Rock	-0.023	0.127	0.857	0.189	9
UD1	0.015	0.068	0.829	0.152	8
Sotted Jewel-babbler (*Ptilorrhoa leucosticta*) – Hides Low					
Year (2017:2019)	-0.198	0.347	0.578	**0.691**	**47(77)**
Year (2017:2021)	**-0.666**	**0.556**	**0.236**		
CnpHt	0.058	0.058	0.320	0.546	34
DClr	-0.001	0.001	0.603	0.344	31
TrSm	-0.014	0.027	0.599	0.247	20
Sotted Jewel-babbler (*Ptilorrhoa leucosticta*) – Hides High					
LT100Rd	-0.172	0.500	0.735	0.216	8(34)
UD2	-0.011	0.036	0.773	0.176	7
TrLge	0.007	0.058	0.911	0.170	7
Chestnut-backed Jewel-babbler (*Ptilorrhoa castanonota*) – KP107					
LfLit	-0.012	0.014	0.410	0.560	123(262)
Rock	-0.066	0.124	0.597	0.417	117
TrLge	-0.037	0.072	0.615	0.366	98
DRd	0.001	0.001	0.622	0.319	87
TrSm	-0.004	0.015	0.772	0.206	61
Chestnut-backed Jewel-babbler (*Ptilorrhoa castanonota*) – BAA 2					
LfLit	**-0.015**	**0.009**	**0.105**	**0.869**	**117(156)**
Station	0.149	0.263	0.572	0.439	61
DClr	0.000	0.001	0.638	0.328	49
TrSm	-0.004	0.011	0.758	0.202	36
Rufous-naped Bellbird (*Aleadryas rufinucha*) – BAA 1					
LT100Rd	-0.380	0.429	0.376	0.501	51(114)
LR20	-0.024	0.032	0.467	0.398	50
Rock	0.025	0.121	0.839	0.257	30
Piping Bellbird (*Ornorectes cristatus*) – BAA 2					
DRd	0.000	0.001	0.722	0.220	33(133)
Rock	-0.014	0.044	0.759	0.212	29
Logs	-0.020	0.280	0.945	0.199	36
Lesser Melampitta (*Melampitta lugubris*) – Hides High					
TrLge	-0.076	0.088	0.392	0.547	45(98)
LT100	-0.887	0.902	0.326	0.517	49
Rock	-0.679	0.772	0.381	0.483	49
Rock^2	-0.216	0.236	0.363	0.483	49
LfLit	0.008	0.021	0.693	0.208	19

Taxa/sites/covariates	**Estimate**	**SE**	**P**	**Rel. Imp.**	**No. models**
Papuan Scrub Robin (*Drymodes beccarii*) – KP107					
Year (2017:2019)	0.391	0.576	0.507	**0.796**	**99(136)**
Year (2017:2021)	**1.271**	**0.824**	**0.126**		
Rock	-0.177	0.211	0.404	0.472	55
UD1	0.030	0.070	0.669	0.251	39
TrSm	0.007	0.020	0.728	0.250	30
LR20	-0.025	0.051	0.624	0.224	26
Greater Ground Robin (*Amalocichla sclateriana*) – Hides High					
DClr	**-0.005**	**0.002**	**0.006**	**1.000**	**12(12)**
TrSm	**-0.070**	**0.019**	**0.000**	**1.000**	**12(12)**
UD1	**0.180**	**0.097**	**0.066**	**0.876**	**9**
Year (2017:2019)	**-0.913**	**0.716**	**0.205**	**0.673**	**4**
Year (2017:2021)	**-0.450**	**0.419**	**0.288**		
Lesser Ground Robin (*Amalocichla incerta*) – Hides Low					
Year (2017:2019)	-0.349	0.500	0.497	**0.972**	**23(24)**
Year (2017:2021)	**-1.383**	**0.571**	**0.018**		
CnpHt	**0.156**	**0.083**	**0.060**	**0.823**	**17**
LT100	0.089	0.272	0.748	0.222	7
Russet-tailed Thrush (*Zoothera heinei*) – KP107					
DClr	**0.008**	**0.003**	**0.006**	**0.957**	**109(122)**
TrSm	-0.037	0.043	0.391	0.534	49
TrLge	-0.036	0.080	0.655	0.280	35
UD1	0.059	0.107	0.583	0.280	25
UD2	0.024	0.047	0.608	0.245	26

Chapter 3 – Small non-volant mammals (Rodents)

Kyle N. Armstrong, Enock Kale, Daniel Okena,
Joachim Yalang and Jasper Billy

A *Rattus* sp. cf. *niobe* captured in BAA 1 (photo by D. Okena)

Summary

Background and aims

The aim of this study was to document and interpret observed changes in rodent species diversity and abundance adjacent to linear project infrastructure (the pipeline ROW, accompanying access roads, Arakubi quarry) to provide informed advice about potential project-related impacts on forest quality. The 2021 survey followed the same format as that undertaken in 2017 and 2019, using the same nine trapping transects that were established in 2015 at two elevations above 2,000 m asl on Hides Ridge (BAA 1) and at two elevations below 1,500 m asl on the Agogo Range near Moro (BAA 2).

This report focuses on small to medium-sized rodents in two main groups: the murine tribe Hydromyini (dominated by *Paramelomys* in this study area) and the tribe Rattini (dominated by *Rattus* in this study area). Larger species of rodent and marsupial were documented in a separate study using camera traps, and those results are presented by Woxvold et al. (Chapter 2, this volume).

Major results

The total number of novel (non-recapture) captures of small rodents on the 2021 survey was greater than on previous surveys: 2015: 133 individuals; 2017: 53 individuals; 2019: 134 individuals—compared with 2021: 202 individuals (total 263 captures, including recaptures). No rodent species new to the study area were detected during the 2021 trapping program, and most undectected species known to be in the area from camera trapping are too large for box traps or unlikely to be encountered on the ground because of their preferred habitat in trees.

Despite the highest capture rate of the 6-year study to date, statistical tests indicated that Species Richness and the total number of captures were not significantly different amongst the categories defining distance from the ROW, elevation and survey year. The overall patterns in the number of captures are driven by four relatively common species: *Paramelomys* sp. cf. *mollis* AC*, Paramelomys* sp. cf. *rubex* A, *Rattus* sp. cf. *niobe* B and *Rattus* sp. cf. *niobe* D, which together made up 84.5% of captures in 2021.

No invasive species of rodent was detected in 2021 despite the high capture rate. This is significant because an increasing population of invasive rodent species is likely to be the initial and most conspicuous indication of an edge effect that has the potential to affect the native rodent assemblage, but clearly this has not yet happened.

Conclusions and recommendations

The most important result of the rodent monitoring to date is the continued lack of a significant shift in Species Richness, abundance or species composition at increasing distances from the ROW, in either BAA, and in any survey year, indicating that there has been no detectable impact of edge effects from the ROW on these taxa.

Native rodents have so far been resilient to the removal of adjacent forest for the pipeline and access road because intact habitat extends right to the forest edge. However, if these forest edge habitats degrade in future then we may yet detect changes in the rodent community, including the increased presence of invasive rodents.

The trapping effort appears to have encountered most of the terrestrial mammal species in BAA 1 and BAA 2 that can be trapped with medium-sized box traps. Some additional uncommon taxa were detected by camera trapping during the past four PMA3 surveys and earlier baseline work, and these may be encountered during future PMA3 rodent surveys.

Genetics-based identification has been the foundation of reliable comparisons between sites, survey years and investigators in this study, and the remarkable results (that have included the discovery of at least two new species not seen elsewhere) are indicative of an under-estimated level of rodent diversity across New Guinea.

The lack of encounters of invasive rat species in 2021 contrasts with results from previous surveys, and previous conclusions that *Rattus exulans* is probably well-established at the forest edges on the Agogo Range. We conclude that invasive rodents are unlikely to be increasing significantly in number and influence at this stage, but there is still potential for them to do so, especially on Hides Ridge because of the predicted occurrence of *Rattus rattus* in nearby source areas.

We recommend that the live-trapping rodent study continue because the results from four surveys over six years demonstrate the value of collecting long term data to understand natural variation in species presence and abundance. Any departures from baseline will be more evident from the understanding of rodent diversity and relative abundance established over the first four surveys. Consideration should also be given to a rapid assessment of the presence of *Rattus rattus* and *R. exulans* in inhabited areas around the HGCP to provide information about their abundance and whether they represent a significant source population for invasions along the access road and pipeline along Hides Ridge. Lastly, consideration should be given to making an assessment of the vegetation community structure around sampling transects, ideally with drone-based multi-spectral imaging of the tree canopy, so that future studies have a quantitative basis for making interpretations of and associations with habitat condition.

Introduction

Predictions made as part of environmental impact assessments for developments include consideration of patterns and processes occurring at the landscape scale. Ideally, these will be based on a foundation of resolved taxonomy, an understanding of species biology, extent of occurrence and elevational range, and whether certain species have special sensitivities to anthropogenic factors. The initial responses of wildlife to the encroachment of infrastructure footprints in pristine forest habitats can be informative context for impact assessments. But understanding such responses might take several years because they can be subtle and masked by natural variation and sampling limitations.

Developments can interrupt ecological communities through the creation of an infrastructure footprint, and in the longer term alter the state and functioning of the adjacent habitat. The composition and dynamics of remaining biotas are strongly influenced by these 'edge effects' (Laurance 1997). Edges can be regarded as zones of transition and characterised in some situations by increased levels of biodiversity, but they can also be subject to undesirable outcomes (Harris 1988). Disturbed areas and forest edge habitats often have increased species richness relative to undisturbed forest because of the influx of species from other more open habitats, and because of the increased resources that may be present near edges (e.g., Mortellitti and Boitani 2005). By contrast, some species of the forest interior may decline if the amount of edge habitat in forest remnants dominates by proportional area (e.g., Puettker et al. 2008).

Specific knowledge about the responses of ecosystem components to disturbances in Papua New Guinea (PNG) is deficient in the published literature, and predictions about the effects of developments are typically made in a general sense using knowledge derived from the same broad ecosystem type elsewhere in the world. Having the opportunity to observe changes to PNG wildlife assemblages through post-construction monitoring can therefore be valuable for making predictions in similar habitats elsewhere. The present study examines the rodent assemblage adjacent to linear infrastructure in rainforest communities at different elevations in Southern Highlands and Hela Provinces of PNG. The study area contains intact forest that has not been subjected to confounding factors associated with dissection and a multitude of other activities over periods spanning hundreds of years.

An accurate understanding of species diversity and methods of identification is still lacking for Papua New Guinea rodents, despite the impression that authoritative books might give (e.g., Flannery 1995). Most of the literature on Papua New Guinean rodents focusses on their discovery by Europeans, their taxonomy and distribution, and their role as disease vectors, with relatively few studies of ecology. The diet of some species has been examined (McPhee 1988; Jackson and Woolley 1994; Elliott and Vernes 2020), and there have also been studies on home range (Dwyer 1978; Berry et al. 1987), and survival rate (Kale et al. 2012). Given that there is still a high level of cryptic diversity in the PNG rodent fauna (Aplin and Opiang 2017; Armstrong et al. 2019, 2021), there is a risk that ecological studies might base species-specific conclusions on an incorrectly named taxon, or on more than one taxon. The present study sought to avoid errors with species identification by using genetic markers as the basis for identification. By determining species richness and composition based on genetically defined groups, we can confidently compare taxa across sites and years, and avoid potential errors derived from having different investigators with different levels of experience involved in the study each year.

During the 2015 PMA3 rodent survey, cytochrome-*b* mitochondrial DNA markers were used to define these groups, providing a reasonable level of discrimination of 'putative species' and revealing discrete genetic groups at different elevations in the apparently widespread species *Paramelomys 'rubex'* and *Rattus 'niobe'*. It also helped to assign species names to animals that could not be identified in the field. However, while the mitochondrial DNA markers helped to confirm field identifications and contributed to the construction of a working list of taxa present in the study area, some of the relationships amongst samples from the PMA3 study and other geographical regions remained unresolved, and hinted at undocumented cryptic diversity that prevented the accurate application of names to some samples.

In the 2017 survey, we introduced a newer DNA sequencing approach ('DArTseq') to avoid some of the limitations described above and allow future investigators to allocate consistent names to taxa across survey sites and survey years. This technique provides information from thousands of positions across an entire genome, and therefore has better discriminating power than a single mitochondrial gene (Armstrong et al. 2019). Each novel capture (i.e., excluding recaptured individuals) from the 2015 (original samples re-sequenced), and 2017–2021 PMA3 surveys has been sequenced by the DArTseq approach. This has allowed our ecological comparisons to be based on unambiguous identifications of native rodents. Furthermore, detection of incursions by invasive pest species such as *Rattus exulans* and *Rattus rattus* has become relatively straightforward with genetic markers because they are genetically well-separated from all native New Guinea rodents (e.g., Armstrong et al. 2019). Invasive species are a key source of potential decline in native rodent diversity related to an edge effect, either through competition or exposure to novel pathogens.

The first three PMA3 surveys of 2015–2019 found no overlap between ground-dwelling rodent assemblages in BAA 1 (transects at 2,200 m asl and 2,700 m asl on Hides Ridge) and BAA 2 (transects at 1,000 m asl and 1,400 m asl on the Agogo Range; Armstrong et al. 2019, 2021). There were no significant shifts in species richness or capture frequency on transects at increasing distances from the ROW or its access road in any year (Armstrong et al. 2019, 2021). Given that these rodents are short-lived, biennial surveys are unlikely to encounter the same individuals so the results suggest the stability of local populations. The results to date suggest that removal of forest for the ROW has had no detectable effect on native rodent populations.

Aim

The overall aim of this study is to document and interpret observed changes in small, non-volant mammal species diversity and abundance to provide informed advice about potential project-related impacts. To achieve this aim, the study has five specific objectives (modified from Aplin and Opiang 2017):

1. Monitor the impact of linear ROW infrastructure on non-volant rodent communities, as a more general indicator of the influence of the ROW of the adjacent forest habitat.

2. Document the diversity and abundances of small, non-volant mammals within the two BAAs.

3. Identify species of conservation significance (including new or undescribed species) within each of the BAAs and, where practicable, determine their special sensitivities.

4. Monitor the status of exotic mammal species in each of the BAAs.

5. Assess the usefulness of non-volant mammal communities in each of the BAAs more broadly as potential indicators of change in habitat quality.

Methods

Overview of methods in 2021

The 2021 PMA3 survey was conducted in the same way as the 2017 and 2019 surveys (Armstrong et al. 2019, 2021), which departed in several respects from the initial survey in 2015 (Aplin and Opiang 2017)—by splitting the effort for camera trapping into a separate study (see Woxvold et al., this volume) and using a different genetic marker system for confirming identifications. Three main sets of methods are described for the 2021 survey: those pertaining to field sampling, those for making robust identifications using genetic markers, and those used in the ecological analysis of the trapping data.

Field sampling

The July 2021 survey used the same nine trapping transects that were established in 2015 by Aplin and Opiang (2017) and operated in 2017 and 2019 (Armstrong et al. 2019, 2021): five transects on Hides Ridge (BAA 1; elevation categories 2,200 m asl and 2,700 m asl; nights of 4–12 July 2021); one transect adjacent to Arakubi quarry (BAA 2; 1,000 m asl; 17–24 July 2021), and three transects at KP107 (BAA 2; 1,400 m asl; 17–24 July 2021). Metal tags attached to trees identified trap positions (these were replaced where necessary) along each transect. Box traps (medium-sized: 37 x 10 x 10 cm; and large-sized: 15 x 15 x 46 cm) were then placed along the transect in a T-shaped configuration (Figure 3.1). Each transect was sampled for six (BAA 1) or seven (BAA 2) nights (Table 3.1), and the total number of trapping nights was 2,396 (one trap-night = one trap set for one night).

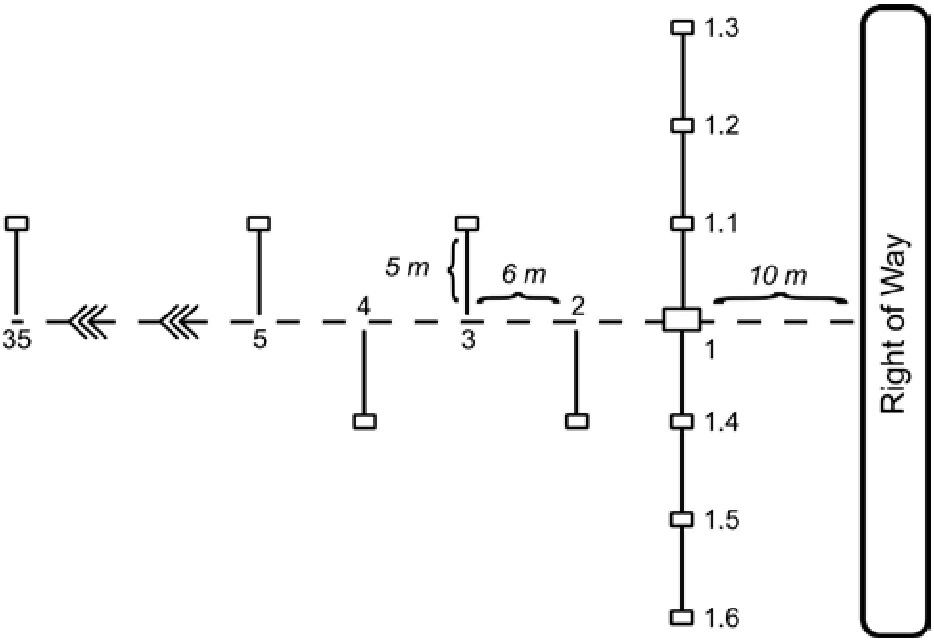


Figure 3.1. Standard design of a 210 m transect trap-line. Large rectangle = large-sized box trap; small rectangle = medium-sized box trap. The central access path is represented by the dashed line. Trap positions were located at 6 m intervals along the transect access path. Traps were placed on alternate sides of the access path, and no more than 5 m to the right or left. Large-sized box traps were placed at positions 1, 11, 21 and 31. Medium-sized box traps were placed at all other positions. An additional six medium-sized box traps were placed along the forest edge parallel to the ROW to provide additional sampling of the most heavily impacted 'edge' habitat, and together with position 2 comprises its own distance category in statistical analyses (Appendix 3.1).

Table 3.1. Summary of box trap deployments in July 2021 (total number of traps is given as the number of medium-sized traps + number of large-sized traps).

Elevation (m asl)	Transect	No. traps	Open date	Close date	Total nights	Total trap nights	Elevation totals
1,000	M4	47 (43+4)	17/07/2021	23/07/2021	7	329	329
1,400	M1	39 (35+4)	17/07/2021	23/07/2021	7	273	
1,400	M2	39 (35+4)	18/07/2021	24/07/2021	7	273	
1,400	M3	39 (35+4)	18/07/2021	24/07/2021	7	273	819
2,200	H1	43 (39+4)	6/07/2021	11/07/2021	6	258	
2,200	H2	40 (36+4)	7/07/2021	12/07/2021	6	240	
2,200	H3	41 (37+4)	4/07/2021	9/07/2021	6	246	744
2,700	H5	43 (39+4)	4/07/2021	9/07/2021	6	258	
2,700	H6	41 (37+4)	6/07/2021	11/07/2021	6	246	504
						Overall	2,396

Trapping results depend heavily on trap condition and placement, and the status of bait. Given that the investigators were different in 2015, it was important that the trapping lines were run with an equivalent level of diligence (see Aplin and Opiang 2017: their Appendix 5.2). Of particular importance was ensuring that trigger sensitivity of the floor treadle was sufficient for successful operation, and that bait (local sweet potato) was always present, which required daily attention. Cleaned traps were placed off the main transect by c. 5 metres as per the transect design and arranged against habitat features such as fallen logs where rodents had a greater chance of encountering them. The shiny trap surface was covered with leaf litter. In 2019 and 2021, the sweet potato bait was supplemented with ripe banana slices.

Captured animals ('novel' or first-time captures) were processed on-site and then released at the point of capture. No animals were intentionally sacrificed as voucher specimens, but inadvertent trap deaths were collected as museum specimens and submitted to the South Australian Museum (Appendix 3.2). To assist with field identifications, each individual was sexed, weighed, measured (lengths of the head-body, tail and pes) and assessed for age (based on body mass and proportions) and reproductive condition. The tip of the tail (<0.5 cm) was removed with clean scissors and placed into 95% ethanol for later genetic analysis. Barcoded vials (with a human-readable 'MEL-number') were used to minimise the likelihood of sample mix-ups. Animals recaptured on subsequent trapping nights were recognised on the basis of their freshly snipped tail tips and re-released near the point of capture.

Genomics-based identification

To ensure that comparisons across transects, elevations and survey years were made based on accurate species identifications, we built further on the accumulating genetic resource of genome-scale genetic markers by submitting a non-lethal biopsy tissue sample from every captured individual for DNA sequencing. This comparative genetic framework is still the most comprehensive ever created for New Guinea native murid rodents.

A 'reduced representation' genome sequencing approach, which generates many thousands of single variable sites (Single Nucleotide Polymorphisms: SNPs) from random locations across the entire chromosome area (the 'genome') was used for DNA sequencing. The specific DNA sequencing method chosen is called 'DArTseq' (Kilian et al. 2003, 2012; Grewe et al. 2015), which is the commercial equivalent of another widely-used technique called 'RADseq' (restriction site-associated DNA sequencing; Peterson et al. 2012). In this technique the entire DNA content in a sample is cut randomly with two enzymes, the resulting fragments ligated with indexed adapters, and the indexed fragments are then sequenced to 75 base pairs in length on an Illumina DNA sequencing platform. All biopsy tissues were put into

96-well plates and sent to a commercial service for library-making and DNA fragment sequencing (Diversity Arrays Technology Pty Ltd, University of Canberra).

A custom-written [R] language analysis script using the package *dartR* (version 1.9.6; Gruber et al. 2018) was used to tidy and filter the genotype matrix supplied after bioinformatic processing conducted by Diversity Arrays Technology. Individuals and loci that had an excess of missing data were removed. Individuals for which no genotype was available were allocated the field identification. One of the simplest ways to illustrate the results is to produce an ordination plot derived from Pearson Principal Component Analysis (PCA) of the genotypes. The PCA plot shows a pattern where individual samples cluster together in terms of their overall relative genetic similarity at the several thousand SNP loci. These clusters represent discrete gene pools, which are interpreted to represent distinct species (because species do not share gene pools, unless reproductive isolation is not quite complete). The geographic origin of samples for each captured individual was coded in PCA plots so that a species list could be created for each transect. Names of genetic clusters were guided by the groundwork established by Aplin and Opiang (2017) as well as information associated with context samples that was available from the Australian Biological Tissue Collection database (South Australian Museum).

Ecological analysis

Basic trapping results including the total number of novel and recaptured individuals for each species on each transect, as well as 'trapping success' (proportion of captures per trap-night on each transect) were calculated. The overall percentage of recaptures was also calculated (from the number of recaptures divided by the number of released individuals, both novel and recapture, and excluding the small number of inadvertent trap deaths).

The trapping results matrix used in statistical analyses consisted of elevation, transect name, trap position, capture date, field identification, capture type (novel/recapture), plus the survey year. Following genetic analysis, the 'SNP-based identification' was added. Data from 2015–2019 were also appended to the 2021 results. Raw data are available in the appendices of Aplin and Opiang (2017), and Armstrong et al. (2019, 2021); and data from 2021 are presented in Appendices 3.3 and 3.4. Statistical analyses and plotting were conducted on novel captures only. All statistical tests and plots were generated in a custom-written [R] language (R Core Development Team 2021) statistical computing script, which contains a record of all manipulations of the cleaned raw dataset, and all analyses and plot instructions for transparency (script available for future surveys).

Trapping data were pooled into six distance categories, each of which included eight trapping positions (see notes in Appendix 3.1). The categories are defined as representing the immediate edge of the forest adjacent to the ROW ('0–20 metres': one large box trap at position 1, a medium-sized box trap at position 2, and the six medium-sized traps extending in parallel with the ROW either side of position 1) (see Figure 3.1); and up to five distance categories of around 50 metres ('20–70 metres': positions 3–10; '70–120 metres': positions 11–18; '120–170 metres': positions 19–26; '170–220 metres': positions 27–35; '220–370 metres': positions 36–60). Note that Aplin and Opiang (2017) established transect M4 at Arakubi quarry to be longer than all other transects, and this has been copied in 2017–2021. Abundance values in each category were adjusted according to survey effort (number of trap nights per transect) in statistical tests.

Generalised Linear Mixed Models were used to determine whether there were differences in four main dependent variables: 'Species Richness' (total number of species) and 'abundance' for total number of captures from all rodent species; total captures of all Hydromyini rodents combined (genera *Hydromys*, *Leptomys, Lorentzimys, Paramelomys, Pogonomys, Uromys*), and total captures of all Rattini rodents combined (genus: *Rattus*); with three fixed factors: increasing distance from the ROW in the distance categories defined above (Appendix 3.1), elevation category, and survey year; with transect as a random factor. The dependent variables were $\log_{10}$-transformed before testing. Recapture rates were too low to include this parameter in the models.

Species composition (the relative mix of each species at different sites) was examined by calculating Bray-Curtis Dissimilarity values for sites defined by the five distance categories and four elevations, over the four survey years combined. A Non-metric Multidimensional Scaling ordination was then used to summarise the patterns of similarity in a two-dimensional ordination plot.

Usage of scientific names

We followed the scheme established by Aplin and Opiang (2017) for naming undescribed or taxonomically ambiguous taxa. We use 'sp. cf.' to refer to individuals that resemble certain species, but where the identification cannot be confirmed. When there is more than one such taxon, they are given a sequential letter code. The codes used follow Aplin and Opiang (2017) to allow correspondence between the different mitochondrial and genome-scale genetic datasets, but there were some additional codes used in surveys after 2015 that are summarised and discussed in Appendices 3.5 and 3.6. For the purpose of this report, the words 'taxon' and 'taxa' are interchangeable with 'species'—distinct genetic groups as revealed by the DArTseq are treated as distinct species.

Results

Species identification

The identification framework derived from genetic markers generated during the 2015–2019 surveys and from context samples outside the study area permitted unambiguous association of specimens encountered in 2021 to established genetic groupings. First, the dataset was checked for the two invasive species of *Rattus* using a dataset that contained all rodents. No individuals of either invasive species were captured in 2021 (Figure 3.2).

Second, each 2021 sample was plotted over the context samples and samples from previous years that together comprise the reference identification framework (this framework presented in Figures 3.3 and 3.4 without the 2021 samples). Separate PCA analyses were performed for the genera *Paramelomys* and *Rattus*, with the allocation of 2021 samples to genus level based on the field identifications (see also Aplin et al. 2018). Morphologically-based identifications in all survey years, including 2021, suggested that '*Paramelomys rubex'* and '*Rattus niobe*' are found across the elevational range of the study area (1,000–2,700 m asl). The DArTseq SNP-based genetic identification method (and mitochondrial data; Appendices 3.5 and 3.6) showed unambiguously that there are three elevational forms in '*Paramelomys rubex*', and two in '*Rattus niobe*', which could represent distinct species.

Third, patterns of species occurrence were examined across elevations (by BAA) and across survey years for the 2021 samples (Figures 3.5 and 3.6).

A comparison of field identifications and SNP-based identifications is presented in Appendix 3.7. From an inspection of this list, it is clear that some species present a challenge to identification, with 85.5% of captures being correctly identified in the field:

- Identification of *Paramelomys* sp. cf. *mollis* AC was correct in 17 instances, and 20 individuals from four other species (*Paramelomys* sp. cf. *lorentzii* C, *Paramelomys platyops*, *Paramelomys* sp. cf. *rubex* A, *Paramelomys* sp. cf. *rubex* F) were incorrectly attributed to *Paramelomys* sp. cf. *mollis* AC. An additional two individuals of *Paramelomys* sp. cf. *mollis* AC were incorrectly identified as *Paramelomys* sp. cf. *rubex*.

- Some field identifications of *Paramelomys platyops* were marked as *Paramelomys* sp. cf. *rubex*.

- One individual of *Rattus steini* was incorrectly identified as *Rattus* sp. cf. *niobe*.

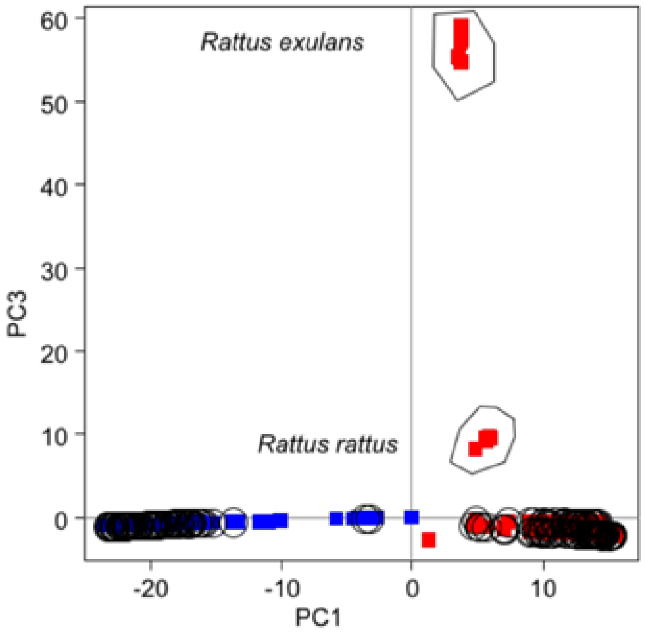


Figure 3.2. Principle Components Analysis of native New Guinea rodent species showing the separation of the two invasive *Rattus* from New Guinea native rodents (blue squares: samples of Hydromyini species; red squares: samples of Rattini; black circles: samples from 2021; large polygons: invasive species clusters).

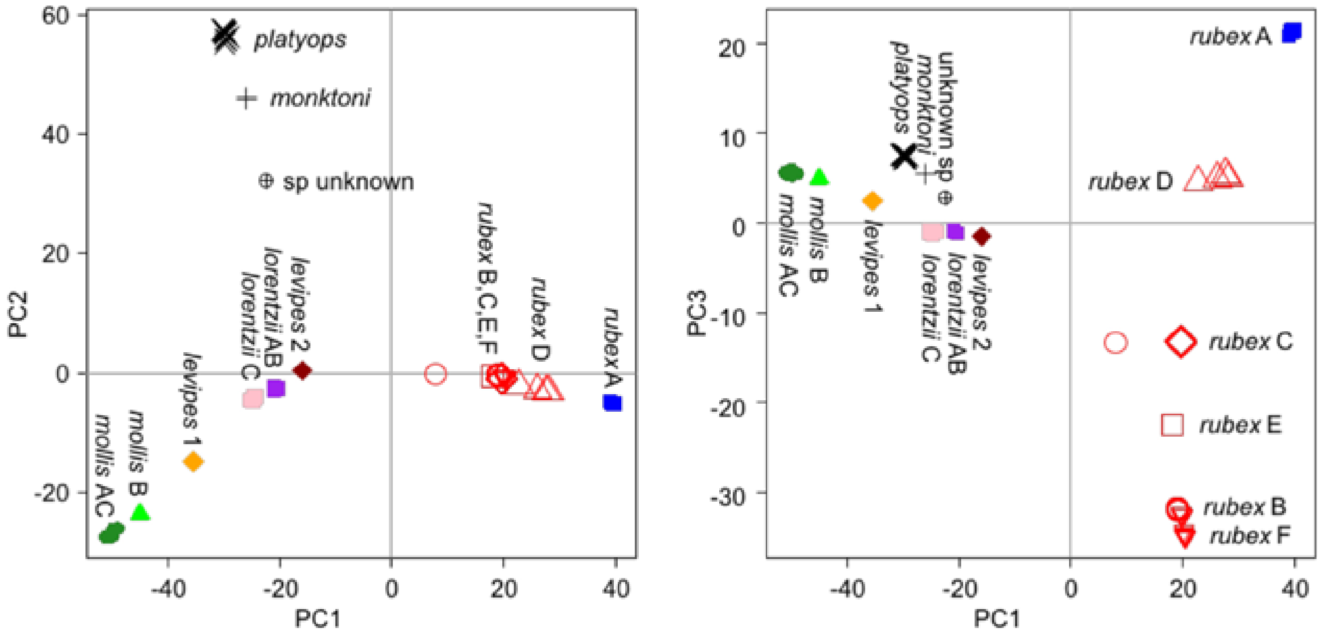


Figure 3.3. Principle Components Analysis of New Guinea *Paramelomys* species in three dimensions, showing some of the main genetic clusters that correspond to species and species complexes, their relative degree of separation from one another, and illustrating the diversity within the '*rubex*' group (see also Appendix 3.5).

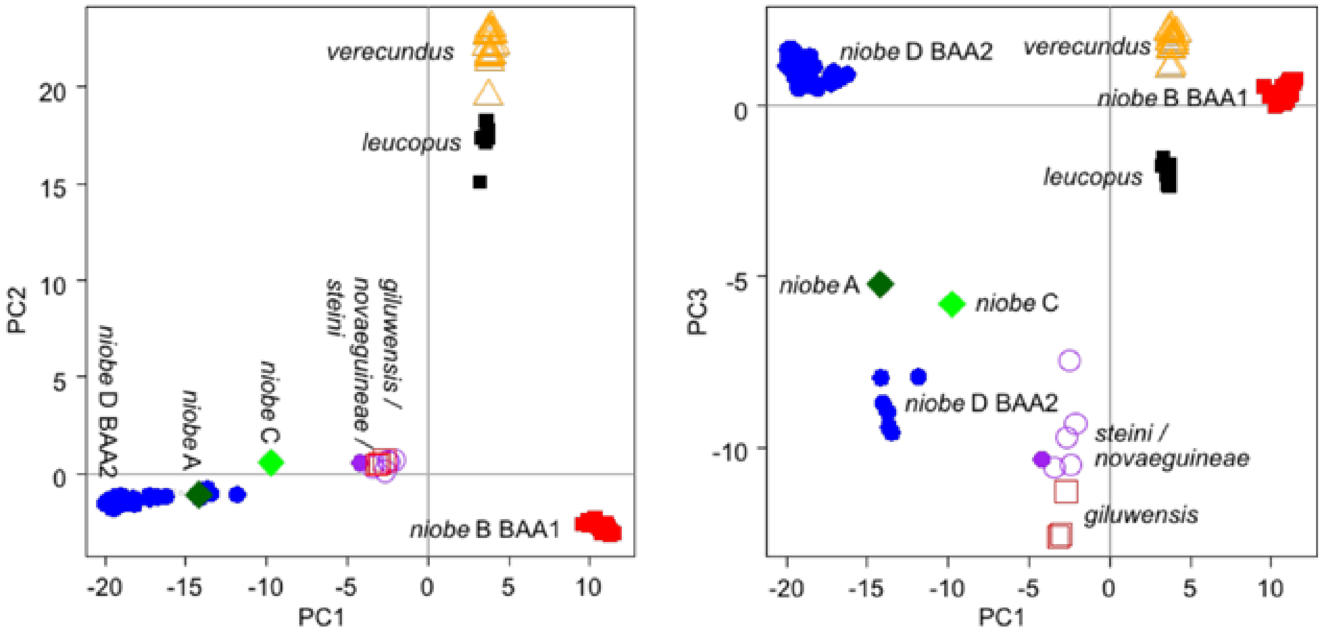


Figure 3.4. Principle Components Analysis of New Guinea *Rattus* species in three dimensions, showing some of the main genetic clusters that correspond to species and species complexes, their relative degree of separation from one another, and illustrating the diversity within the 'niobe' group (see also Appendix 3.6).

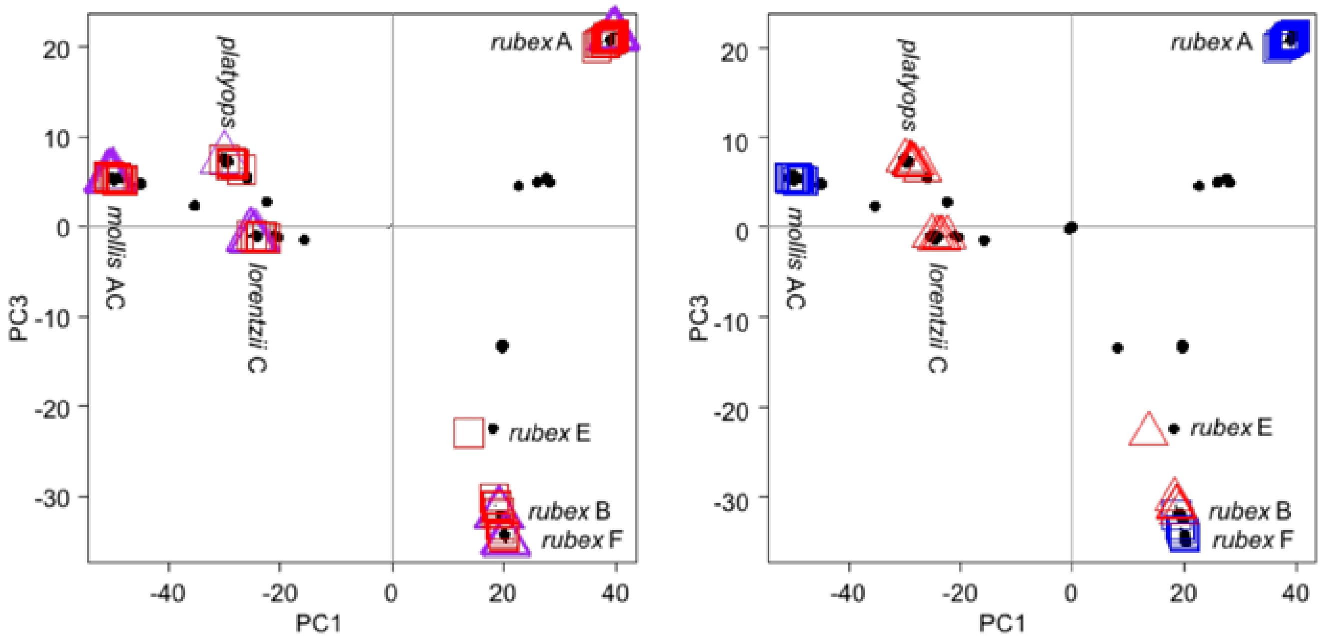


Figure 3.5. Principle Components Analysis of New Guinea *Paramelomys* species (PC1 versus PC3). **Left:** Identifications made in 2021 (red squares) versus all other survey years combined (purple triangles) and context samples from outside the BAA study areas (black dots); **Right:** Identifications made in 2021 from Hides Ridge (BAA 1; blue squares), and from the Agogo Range (BAA 2; red triangles).

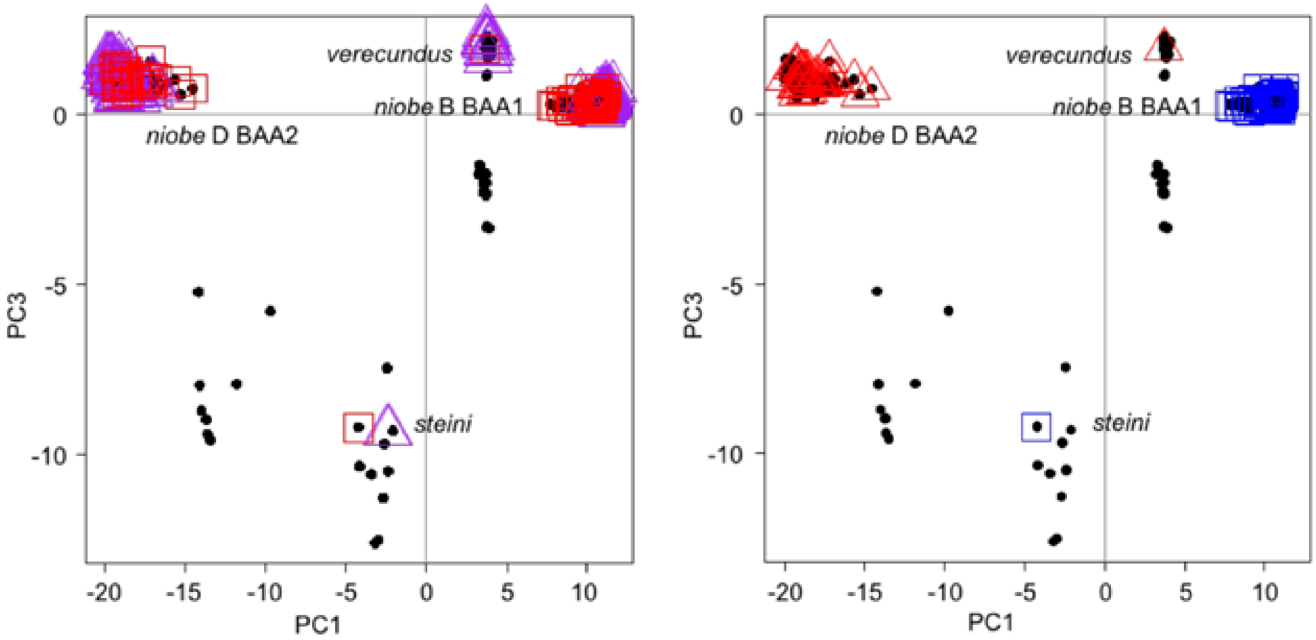


Figure 3.6. Principle Components Analysis of New Guinea *Rattus* species (PC1 versus PC3). **Left:** Identifications made in 2021 (red squares) versus all other survey years combined (purple triangles) and context samples from outside the BAA study areas (black dots); **Right:** Identifications made in 2021 from Hides Ridge (BAA 1; blue squares), and from the Agogo Range (BAA 2; red triangles).

Trapping results summary

A total of 17 small to medium-sized rodent taxa have been trapped on the nine transects in both BAAs during the course of the 2015–2021 PMA3 surveys. Eleven of these were captured in 2021 (Table 3.2). In 2021, five rodent taxa were trapped in BAA 2 (with 10 taxa now recorded since 2015), and six taxa were trapped at the higher elevations in BAA 1 (with nine taxa now recorded since 2015). There are two taxa that have been recorded in both BAAs over the study program, both being captured in 2021 at 2,200 m, and lower elevations prior to 2021: *Pogonomys macrourus* (1,400 m and 2,200 m) and *Rattus steini* (1,000 m and 2,200 m). These two species are known from a broad area outside the PMA3 study areas (Flannery 1995), and they may also contain undescribed species-level diversity. The remaining taxa are not shared between the two BAAs.

The total number of novel captures of rodents in 2021 was 200 individuals (57 captures from BAA 1 and 143 captures from BAA 2; plus two Common Spiny Bandicoots *Echymipera kalubu*, and two rodents for which no trap data was associated), with an additional 61 recapture events (Table 3.3). The total exceeds that from all previous surveys: 133 individuals in 2015; 53 individuals in 2017; 134 individuals in 2019. The recapture rate was 23.5% (204 novel captures, 61 recaptures, minus six individuals that were vouchered rather than released; Appendix 3.2).

Trapping success overall was 10.9%, and higher in BAA 1 (14.3%) than BAA 2 (7.2%; Table 3.3). In the survey of 2019, the potential for capture was reduced slightly in BAA 1 because of interference of traps by dogs (loss of 59 trap nights; total 2,207 trap nights versus 2,396 in 2021; Table 3.3). Such levels of interference were not encountered in 2021. Trapping success ranged from 3.6% at transect M4 to 18.7% at transect H3. The transects at 2,200 m had both the highest capture rate (92 novel rodent individuals, 28 recaptures) and Species Richness (six species) compared to other elevations. Of interest, the two transects at 2,700 had only two species, but an equivalent number of novel captures to the three transects at 1,400 m, suggesting that overall BAA 1 is a more productive environment for this part of the rodent assemblage that can be encountered using box traps.

Four species accounted for 84.5% of novel captures: *Paramelomys* sp. cf. *mollis* AC (19 individuals; 9.5%), *Paramelomys* sp. cf. *rubex* A (31 individuals; 15.5%), *Rattus* sp. cf. *niobe* B (83 individuals; 41.5%), and *Rattus* sp. cf. *niobe* D (36 individuals; 18%; Table 3.4).

Non-Metric Multidimensional Scaling ordination showed clear differences in species composition amongst elevations, but not with categories of distance from the ROW (all survey years combined; Figure 3.7). The community at 2,700 m asl overlaps with that from at 2,200 m asl, mainly because it contains a smaller subset of species. By contrast, sites from the two elevations in BAA 2 cluster separately from each other, as well as from the BAA 1 transects. There is partial overlap of species distributions at 1,000 m and 1,400 m, with four species shared and six species not shared between them (Table 3.2). There is almost no overlap between BAA 1 and BAA 2 in terms of species composition—only rare encounters of *Pogonomys macrourus* and *Rattus steini*. This pattern diminishes when rodent species that are encountered with camera traps are taken into consideration (Appendix 3.8).

Casual inspection of tabulated captures at different distance categories does not reveal any obvious pattern (Table 3.5; Figure 3.7). No native rodent species was associated unambiguously with either the forest edge, or the forest interior. Statistical analyses were undertaken to examine the patterns amongst distance, elevation and survey year in more detail.

Table 3.2. Summary of captures at each elevation, in each survey year (in each cell the survey years (2015, 2017, 2019, 2021) are represented in order from left to right as one of two symbols: open circle is an absence of observations, closed circle is at least one observation in that survey year; the last column is the total number of novel captures across all elevations [m asl], for each year, in order).

	1,000 m	1,400 m	2,200 m	2,700 m	Novel captures
HYDROMYINI					
Hydromys chrysogaster	○○○○	○○○○	○○●○	○○○○	0,0,1,0
Leptomys elegans	○○○○	○●○○	○○○○	○○○○	0,1,0,0
Lorentzimys nouhuysi	○○○○	○○○○	○○●○	○○○○	0,0,1,0
Paramelomys sp. cf. *lorentzii* C	○○○○	●●●●	○○○○	○○○○	5,1,4,5
Paramelomys sp. cf. *mollis* AC	○○○○	○○○○	●●●●	●○○○	6,2,6,19
Paramelomys platyops	●○○●	○●○○	○○○○	○○○○	1,1,0,9
Paramelomys sp. cf. *rubex* A	○○○○	○○○○	●●●●	●●●●	11,12,15,31
Paramelomys sp. cf. *rubex* B	●○○○	●●●●	○○○○	○○○○	14,3,1,6
Paramelomys sp. cf. *rubex* F	○○○○	○○○○	○●●●	○○○○	0,4,9,8
Pogonomys macrourus	○○○○	○○●○	○○○●	○○○○	0,0,1,1
Uromys caudimaculatus	○○○○	●○○○	○○○○	○○○○	1,0,0,0
RATTINI					
Rattus exulans	○○●○	●○●○	○○○○	○○○○	5,0,4,0
Rattus sp. cf. *niobe* B	○○○○	○○○○	●●●●	●●●●	42,11,63,83
Rattus sp. cf. *niobe* D	○●○○	●●●●	○○○○	○○○○	38,11,29,36
Rattus rattus	○○○○	○○○○	○●○○	○○○○	0,1,0,0
Rattus steini	○●○○	○○○○	○○○●	○○○○	0,1,0,1
Rattus verecundus	●●●●	○○○○	○○○○	○○○○	10,5,1,1
Total novel captures by box trap					**540**
Total Richness each elevation over all years	**6**	**9**	**9**	**3**	***17 spp. total***

Table 3.3. Summary of rodent captures in 2021 by transect, BAA and elevation.

	Trap nights	Trap success %	Captures	Recaptures	Species Richness
BAA 1					
H1	258	13.2	28	6	6
H2	240	16.7	30	10	4
H3	246	18.7	34	12	3
H5	258	12.8	29	4	2
H6	246	10.2	22	3	2
BAA 2					
M1	273	11.4	22	9	3
M2	273	7.3	12	8	2
M3	273	7.3	13	7	2
M4	329	3.6	10	2	2
BAA					
BAA 1	1,248	14.3	143	35	6
BAA 2	1,148	7.2	57	26	5
Elevation					
1,000 m	329	3.6	10	2	2
1,400 m	819	8.7	47	24	3
2,200 m	744	16.1	92	28	6
2,700 m	504	11.5	51	7	2
Total	**2,396**	**10.9**	**200**	**61**	**11**

Table 3.4. Summary of novel rodent captures in 2021 by species across elevations (m asl) and BAA.

	1,000	1,400	2,200	2,700	BAA 1	BAA 2	Total
HYDROMYINI							
Hydromys chrysogaster	0	0	0	0	0	0	0
Leptomys elegans	0	0	0	0	0	0	0
Lorentzimys nouhuysi	0	0	0	0	0	0	0
Paramelomys sp. cf. *lorentzii* C	0	5	0	0	0	5	5
Paramelomys sp. cf. *mollis* AC	0	0	19	0	19	0	19
Paramelomys platyops	9	0	0	0	0	9	9
Paramelomys sp. cf. *rubex* A	0	0	13	18	31	0	31
Paramelomys sp. cf. *rubex* B	0	6	0	0	0	6	6
Paramelomys sp. cf. *rubex* F	0	0	8	0	8	0	8
Pogonomys macrourus	0	0	1	0	1	0	1
Uromys caudimaculatus	0	0	0	0	0	0	0
RATTINI							
Rattus exulans	0	0	0	0	0	0	0
Rattus sp. cf. *niobe* B	0	0	50	33	83	0	83
Rattus sp. cf. *niobe* D	0	36	0	0	0	36	36
Rattus rattus	0	0	0	0	0	0	0
Rattus steini	0	0	1	0	1	0	1
Rattus verecundus	1	0	0	0	0	1	1

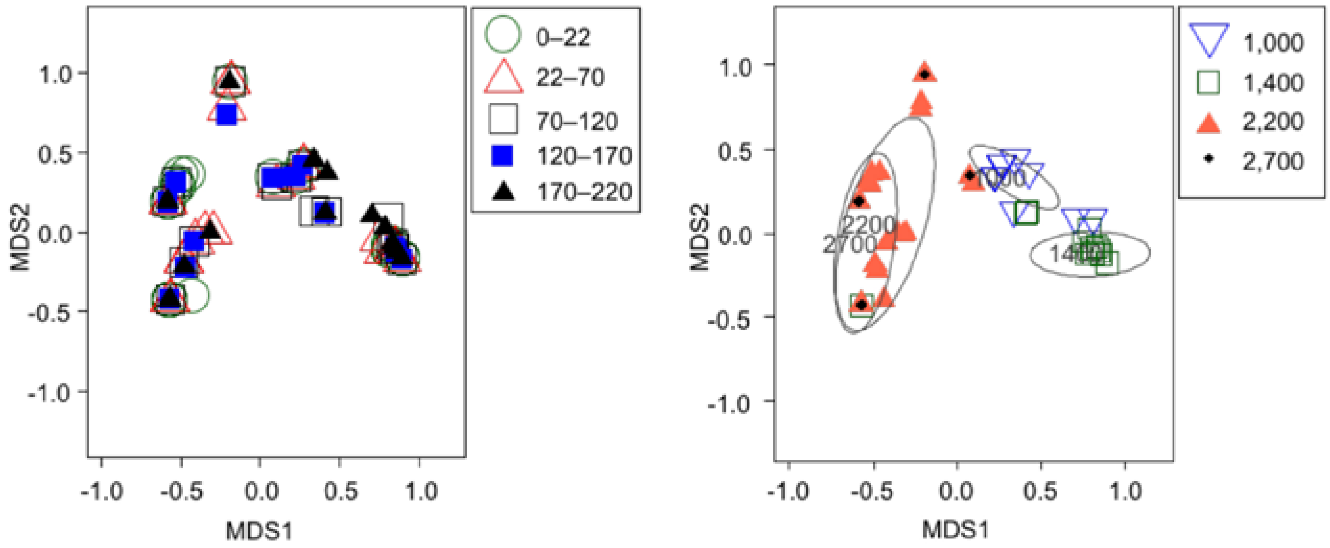


Figure 3.7. Multidimensional Scaling (NMDS) ordinations summarising patterns of species composition (as derived from species lists at each distance category), at increasing distance from the ROW **(left),** and at different elevations **(right),** over all survey years combined.

Table 3.5. Trapping summary from the 2021 survey (symbols in each cell represent the trapping result from each distance category along the transect, beginning at 0 metres from the left [0–20, 20–70, 70–120, 120–170, 170–220 m for all transects]; open circle is an absence of captures; closed circle is at least one capture in that distance category).

Elevation (m asl)	**1,000**	**1,400**			
Transect	**M4**	**M1**	**M2**	**M3**	
HYDROMYINI					
Hydromys chrysogaster	○○○○○	○○○○○	○○○○○	○○○○○	
Leptomys elegans	○○○○○	○○○○○	○○○○○	○○○○○	
Lorentzimys nouhuysi	○○○○○	○○○○○	○○○○○	○○○○○	
Paramelomys sp. cf. *lorentzii* C	○○○○○	○●○●○	○●○○○	○○○●●	
Paramelomys sp. cf. *mollis* AC	○○○○○	○○○○○	○○○○○	○○○○○	
Paramelomys platyops	●●●●●	○○○○○	○○○○○	○○○○○	
Paramelomys sp. cf. *rubex* A	○○○○○	○○○○○	○○○○○	○○○○○	
Paramelomys sp. cf. *rubex* B	○○○○○	○○○○●	○○○○○	○○○○○	
Paramelomys sp. cf. *rubex* F	○○○○○	○○○○○	○○○○○	○○○○○	
Pogonomys macrourus	○○○○○	○○○○○	○○○○○	○○○○○	
Uromys caudimaculatus	○○○○○	○○○○○	○○○○○	○○○○○	
RATTINI					
Rattus exulans	○○○○○	○○○○○	○○○○○	○○○○○	
Rattus sp. cf. *niobe* B	○○○○○	○○○○○	○○○○○	○○○○○	
Rattus sp. cf. *niobe* D	○○○○○	●●●○●	●●○○●	●●○●●	
Rattus rattus	○○○○○	○○○○○	○○○○○	○○○○○	
Rattus steini	○○○○○	○○○○○	○○○○○	○○○○○	
Rattus verecundus	○○○○●	○○○○○	○○○○○	○○○○○	

Elevation (m asl)	**2,200**			**2,700**	
Transect	**H1**	**H2**	**H3**	**H5**	**H6**
HYDROMYINI					
Hydromys chrysogaster	ooooo	ooooo	ooooo	ooooo	ooooo
Leptomys elegans	ooooo	ooooo	ooooo	ooooo	ooooo
Lorentzimys nouhuysi	ooooo	ooooo	ooooo	ooooo	ooooo
Paramelomys sp. cf. *lorentzii* C	ooooo	ooooo	ooooo	ooooo	ooooo
Paramelomys sp. cf. *mollis* AC	●●o●●	o●●oo	●●●oo	ooooo	ooooo
Paramelomys platyops	ooooo	ooooo	ooooo	ooooo	ooooo
Paramelomys sp. cf. *rubex* A	●oooo	●oooo	●●●●●	o●●●●	●●●●●
Paramelomys sp. cf. *rubex* B	ooooo	ooooo	ooooo	ooooo	ooooo
Paramelomys sp. cf. *rubex* F	●●o●●	●●ooo	ooooo	ooooo	ooooo
Pogonomys macrourus	oooo●	ooooo	ooooo	ooooo	ooooo
Uromys caudimaculatus	ooooo	ooooo	ooooo	ooooo	ooooo
RATTINI					
Rattus exulans	ooooo	ooooo	ooooo	ooooo	ooooo
Rattus sp. cf. *niobe* B	●●●●●	●●●●●	●o●●●	●●●●●	●●●●●
Rattus sp. cf. *niobe* D	ooooo	ooooo	ooooo	ooooo	ooooo
Rattus rattus	ooooo	ooooo	ooooo	ooooo	ooooo
Rattus steini	o●ooo	ooooo	ooooo	ooooo	ooooo
Rattus verecundus	ooooo	ooooo	ooooo	ooooo	ooooo

Statistical analyses

Generalised Linear Mixed Models were created to determine if there were statistically significant differences in rodent Species Richness and abundance amongst survey years, at different distances from the ROW, and with elevation. Separate tests were performed on four dependent variables: total Species Richness, total number of captures, total captures of Hydromyini species, and total captures of Rattini species. There was a small but significant level of difference in Species Richness for categories of distance from the ROW, and difference among survey years for total captures, Hydromyini captures and Rattini captures (Appendix 3.9).

Contrasting with this statistical outcome, an inspection of the means and standard deviations for these four variables reveals how similar these values were across elevation and distance categories (Table 3.6). Further investigation of the source of the significant F-tests using pairwise comparisons did not produce any significant combinations of factor levels for any of the four sets of comparisons. The inconsistency of the statistical outcomes does not suggest a strong effect for any of the three factors. Such inconsistency is typically the result of high levels of variation amongst factor levels, though the dependent variables were transformed first. This variation is evident in boxplots of means (Figures 3.8–3.11), with conspicuously higher capture rates in BAA 1 that might be the basis for the inconsistent statistical outcomes.

Despite the variation in rodent abundance detected across survey years, there has been no detectable change in the overall diversity or population sizes of the rodent assemblage that is clearly attributable to the influence of the ROW.

Table 3.6. Mean ± standard deviation for all distance, elevation and survey year categories.

	Category	Total Species Richness	Total captures	Total captures Hydromyini	Total captures Rattini
Distance	0–20	1.8 ± 1.0	3.9 ± 2.6	1.1 ± 1.6	2.7 ± 1.8
from ROW (m)	20–70	1.9 ± 1.0	3.4 ± 2.1	1.3 ± 1.0	2.1 ± 1.7
	70–120	1.4 ± 0.7	2.9 ± 2.5	0.9 ± 1.5	2.0 ± 1.8
	120–170	1.6 ± 0.6	3.1 ± 1.8	1.2 ± 1.3	1.9 ± 1.6
	170–220	1.9 ± 0.8	4.3 ± 2.6	1.6 ± 1.6	2.7 ± 1.8
Elevation (m asl)	1,000	1.4 ± 0.7	2.1 ± 1.4	0.7 ± 1.2	1.4 ± 1.5
	1,400	1.7 ± 0.9	3.5 ± 1.9	0.9 ± 1.3	2.6 ± 1.5
	2,200	1.9 ± 1.0	3.8 ± 2.8	1.7 ± 1.6	2.1 ± 1.9
	2,700	1.5 ± 0.5	3.6 ± 2.5	1.1 ± 1.2	2.6 ± 1.8
Year	2015	1.7 ± 0.9	3.4 ± 2.2	1.0 ± 1.4	2.4 ± 1.6
	2017	1.5 ± 0.7	2.1 ± 1.3	1.0 ± 1.1	1.2 ± 1.1
	2019	1.7 ± 0.8	3.3 ± 2.0	0.9 ± 1.1	2.4 ± 1.6
	2021	1.9 ± 0.9	4.6 ± 2.8	1.8 ± 1.7	2.8 ± 2.1

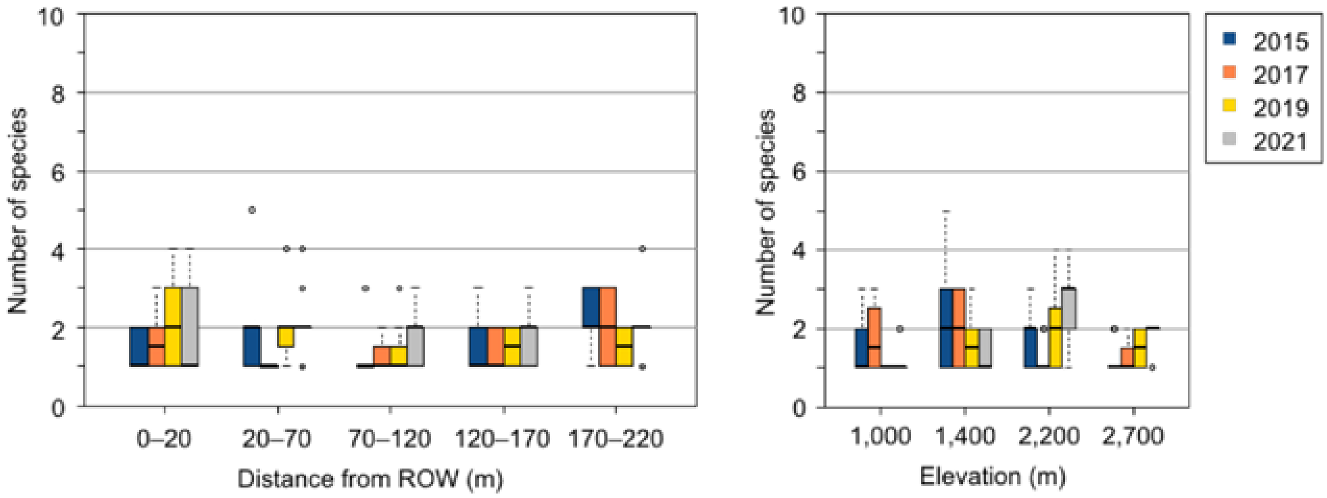


Figure 3.8. Summary of the patterns of Species Richness with increasing distance from the ROW, and with increasing elevation. All sites have been combined for each of the two factors, but segregated by year. [Boxplot components: central bar—median; boxes—inter-quartile range, with second quartile group below median, third quartile group above median; bars—minimum and maximum values; circles—statistical outliers].

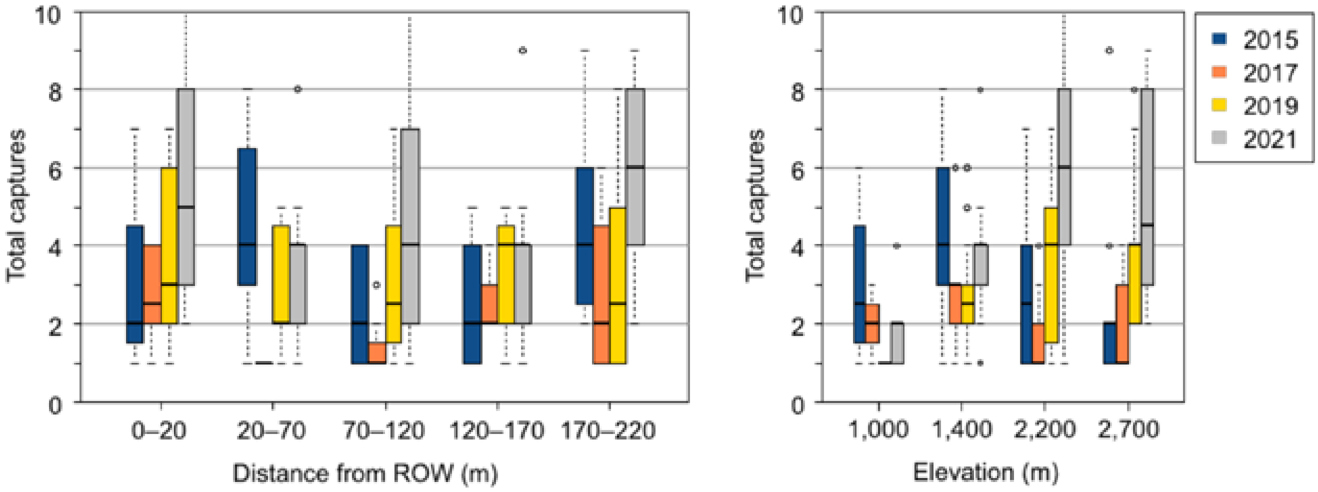


Figure 3.9. Summary plots of the pattern of the total number of captures for all rodent species at increasing distance from the ROW, and with increasing elevation.

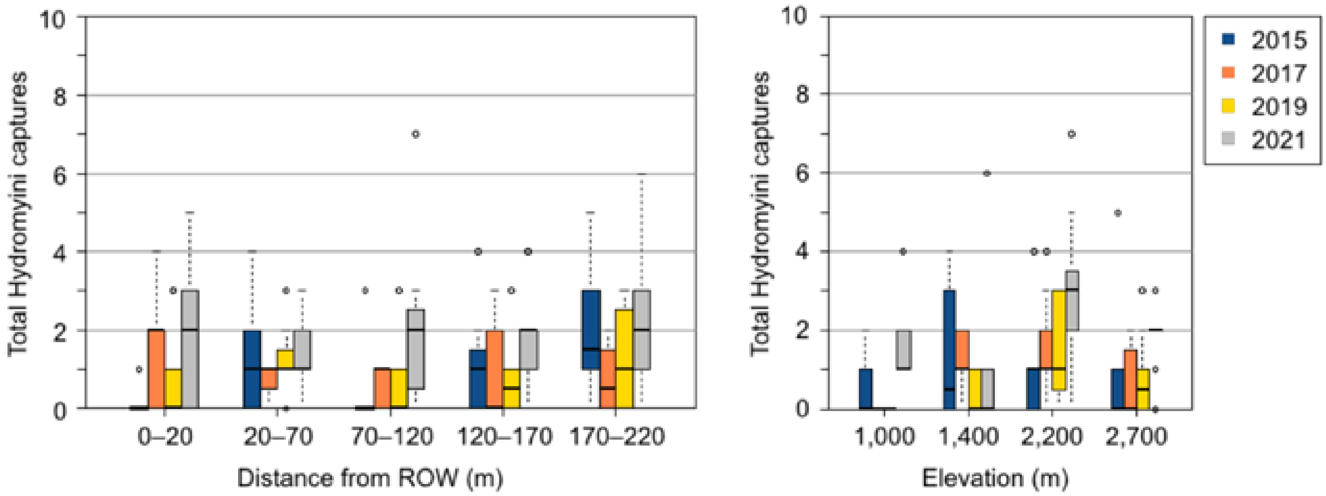


Figure 3.10. Summary plots of the pattern of the total number of captures for Hydromyini rodent species at increasing distance from the ROW, and with increasing elevation.

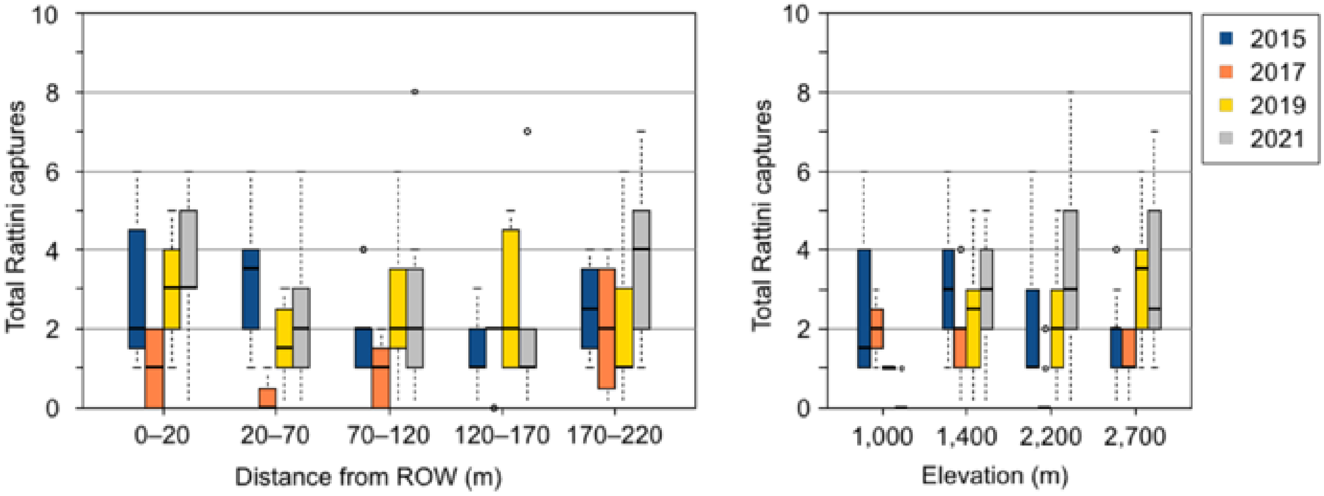


Figure 3.11. Summary plots of the pattern of the total number of captures for Rattini rodent species at increasing distance from the ROW, and with increasing elevation.

Discussion

Patterns of diversity over time

There has been relatively little change detected in the rodent community adjacent to the ROW throughout the duration of the 6-year PMA3 rodent study, though the 2021 results were remarkable for the greater number of captures. No species were detected for the first time in 2021, suggesting that there are few species left to encounter with medium-sized box traps, and that both BAAs are well-documented. The camera trapping does indicate that other rodent species are present in both BAAs (Appendix 3.8), but these are either too large or do not spend sufficient time on the ground where they would encounter box traps.

In previous reports (Armstrong et al. 2019, 2021), we stated that the difference in species composition between sampling years shows that deterministic ecosystems such as closed rainforest in PNG can be variable, and that populations of small mammals sampled two years apart can also show signs of demographic fluctuation. Further, the timing of surveys could be somehow related to capture rates across surveys. While effort has been made to keep the surveys at the same time of the year, they have varied across four months: (10 June–8 July 2015; 11–30 May 2017; 1–31 August 2019; 4–24 July 2021). The highest captures of small ground-dwelling rodents have been made between June and August, with fewer captures in May. Relatively little is known about seasonal breeding cycles of rodents in the highlands of PNG, or the relationship of population size and the seasonal availability of food.

Potential confounding factors here include diligence with maintaining trap condition (which has been a key consideration on all surveys), the presence and interference of traps by dogs in some years, and particularly changes in the type of bait used. Plantain banana was used to supplement the sweet potato in 2019 and 2021, which likely led to greater overall capture rates in those two years. While it is important to maintain the same sampling protocols across survey years, it was also considered important to maximise the capture rate so that a more accurate perspective of abundance might be gained, and also so that statistical tests have increased power. Variation amongst years is likely to be a combination of various factors such as differences in survey timing and any related natural factors, changes in bait type and trap interference on some surveys. Continued monitoring into the future should give further support to an understanding of levels of rodent diversity that are not related to edge effects.

Elevational community dominance and replacement

In terms of overall abundance, and as observed across all survey years, the rodent community in BAA 1 is dominated by *Rattus* sp. cf. *niobe* B, and to a lesser extent by *Paramelomys* sp. cf. *rubex* A; and the community in BAA 2 is dominated by *Rattus* sp. cf. *niobe* D. In 2021, there was also a conspicuously high proportion of *Paramelomys* sp. cf. *mollis* AC in BAA 1. Such patterns of dominance can be considered further across elevations (Table 3.2), and it is evident that between one and four species dominate each elevational range. Most species trapped by box traps are ground-dwelling and of approximately the same size, which prompts a suggestion that species replacement might occur for ecologically similar species with increasing elevation: *Rattus verecundus* that is the most common species of its size at 1,000 m is replaced by *Paramelomys* sp. cf. *lorentzii* C, *Rattus* sp. cf. *niobe* B and *Paramelomys* sp. cf. *rubex* B at 1,400 m; and then by *Rattus* sp. cf. *niobe* B, *Paramelomys* sp. cf. *mollis* AC, *Paramelomys* sp. cf. *rubex* A and *Paramelomys* sp. cf. *rubex* F at 2,200 m. By 2,700 m in elevation, only *Paramelomys* sp. cf. *rubex* A and *Rattus* sp. cf. *niobe* B persist. The remaining nine species encountered throughout the study area are likely present at lower density, or else less likely to be trapped because of a different foraging microhabitat.

Edge effects

This study did not detect any shifts in native rodent diversity with increasing distance from the ROW, the associated roads and the Arakubi quarry, suggesting that any effects that might be operating on the forest edge are still having no major impacts on this local fauna. Although no quantitative or qualitative methods have been used to characterise the vegetation along the transects, with the exception of some localised changes (tree falls in the middle of transect H1 and at the beginning of transect H2; the first five metres of forest cover removed from M2 since 2015; of the first 30 m of transect H4 damaged by an earthquake-related treefall in 2019), the forest structure does not appear to have changed significantly across the four survey periods (E. Kale and K.N. Armstrong pers. obs.). To date, the lack of change in fauna at the forest edge appears to reflect a significant lack of change of forest structure adjacent to the ROW and roads.

The creation of linear infrastructure in both BAA's has increased access for local people, and we documented evidence of hunting along the access road on Hides Ridge and on transects during the 2015–2019 surveys (Aplin and Opiang 2017; Woxvold and Legra 2019; Woxvold et al. 2021). Although small rodents are of less interest to hunters than larger mammals, dogs that accompany them, or that roam freely, probably kill smaller mammal species when they are encountered. To date, it appears unlikely that incursions by people and dogs are causing significant changes in the rodent communities.

Invasive species

The most likely and conspicuous indication of an edge effect detectable in native rodent communities will be the presence of invasive species. The indication of most relevance are incursions of exotic rodent species and their subsequent establishment. Invasive species have the potential to affect resident native species in several ways: by actively excluding them from their territories, out-competing them for resources, and introducing novel pathogens to naïve populations. Studies of pathogens and zoonotic diseases carried by invasive rodents abound in the scientific literature, and the risk to humans of rodent-borne disease is a common theme. However, there is also a growing recognition that wildlife diseases may contribute to population declines of native species (e.g., Begon 2006; Wolf and Edge 2006). The risk of disease transfer from invasive *Rattus rattus* to native species of rodents has been highlighted (Aplin and Singleton 2006; Aplin and Opiang 2017), but not yet demonstrated.

Despite the relatively high overall capture rate in 2021, no individuals of an invasive rat species were detected. There were five captures of *Rattus exulans* at KP107 in 2015, one capture of *Rattus rattus* on Hides Ridge in 2017, and a total of four captures of *Rattus exulans* at Arakubi quarry (1 individual) and KP107 (3 individuals) in 2019 (Aplin and Opiang 2017; Armstrong et al. 2019, 2021). *Rattus exulans* appears to be reasonably well established at KP107, perhaps because of the relative proximity of other industrial facilities. Likewise, *Rattus exulans* might also have been established for some time along the nearby well-used road and associated habitation close to Arakubi quarry. Some level of local population fluctuation or movement could account for yearly detection differences. By contrast, the detection of *Rattus rattus* in BAA 1 is likely to derive from an incursion via linear infrastructure and sourced from the well-settled areas at the base of Hides Ridge. Thus, their continued or increased level of presence will indicate an edge effect.

Invasive rodents often gain access to new areas of forest along infrastructure corridors, and so we would expect to have higher capture rates at the forest edge. Given that all captures during this study were relatively close to the ROW, this is suggestive of an edge effect. In previous surveys, seven of the nine *Rattus exulans* were captured in one of the box traps placed running parallel with the edge of the forest (Figure 3.1). Only two individuals were trapped more than 20 m into the forest: one at each of the 34 m and 82 m positions. The *Rattus exulans* captured at 82 m into the forest was on transect M4 (BAA 2, Arakubi quarry). The first c. 150 m of this transect is located in regrowth forest along an old access track. The only capture of *Rattus rattus* to date was made in BAA 1 at the second trapping position of transect H2, only 16 m from the forest edge.

It is significant that captures of invasive species have not increased during the course of this seven-year study. This suggests that incursions from source areas are uncommon, and that populations have not yet expanded significantly in the study areas. But preventing the incursion of invasive species on Hides Ridge may not be possible in the long-term. At present, the distance from human occupation that is likely to be a source for commensal rodents is relatively large but might gradually decrease with continued encroachment. Continuing the active program of baiting and pest rodent control at the Hides Gas Conditioning Plant and camp will be important for preventing the establishment of a source population that could extend up the spineline over time. However, *Rattus rattus* is almost certainly established in the surrounding rural areas and villages because it is a successful and pervasive commensal species (e.g. Kale et al. 2018a). Thus, if more people begin to live along the spineline, as was first observed in 2019, commensal rodents are likely to follow.

Inventory completeness

Trapping with box traps has provided excellent capture rates over the surveys since 2015, especially with the reconsideration of the bait used. However, box trapping is not an efficient method for detecting all species of PNG rodents, and there are several additional species that have been detected in the PMA3 study area with camera traps (Woxvold and Aplin 2017; Woxvold and Legra 2019; Woxvold et al. 2021; Woxvold et al. this volume), from remains found in five individual owl pellets collected from a cave on Hides Ridge at an elevation of c. 2,065 m asl in 2011 (Aplin and Opiang 2017), from road kills and casual observations made during the PMA3 field surveys, and from the environmental impact assessment work conducted before the construction of the pipeline ROW (summarised in Aplin and Opiang 2017). There are 11 additional species of Muridae identified in these sources (Appendix 3.8). Thus, trapping with box traps has detected 60.7% of species in the Muridae that are known from the area based on these studies. Of particular note, the camera trapping results show that six species are found in both BAAs (*Anisomys imitator, Mallomys* sp.*, Parahydromys asper, Pogonomys* sp. cf. *loriae, Uromys anak, Uromys caudimaculatus*), which indicates that the composition of Muridae at different elevations, and between Hides Ridge and the Agogo Range, is not mutually exclusive.

The potential for encountering additional species in the box trapping effort will likely be dependent on several factors: their ecology (whether they forage on the ground or in trees), their body size, and their general abundance. Scansorial species of *Chiruromys*, *Melomys* and *Pogonomys* have been detected by camera traps and identified in owl pellet accumulations (Appendix 3.8). Only three individuals of a *Pogonomys* species (possibly *P. macrourus* or *P. loriae*) have been captured so far, probably because most of their time is spent foraging in trees. Most species not yet trapped are larger-bodied animals that cannot enter the medium-sized box traps used on the study transects. However, each transect has four large-sized box traps, which would accommodate species the size of *Hyomys*, *Mallomys* and *Uromys*, and it is unclear why no rodents have been captured in these larger traps. There is also the possibility that rarer species such as *Protochromys fellowsi*, and species of *Parahydromys* and *Pseudohydromys* might be captured using box traps. While there has been four PMA3 surveys to date, greater levels of trapping effort that include more sites over a wider area might be required to trap these species because of the patchiness of their local occurrences. Fortunately, most species of rodent not documented by trapping have morphological features that allow them to be identified in camera trap images, which underlines the importance of the camera trapping effort for contributing information on this component of the Muridae. By contrast, the smaller rodent species that enter medium-sized box traps are rarely identifiable to species level in camera trap photos (see Woxvold and Legra 2019; Woxvold et al. 2021; Woxvold et al. this volume; also Kale et al. 2018a,b,c).

Making better field identifications

The taxonomy of New Guinea mammals remains incomplete, which has implications for any biological survey on the island, and for the confidence in identifications made by experienced field biologists and taxonomic specialists alike. A high level of experience with PNG rodents can bring an excellent rate of successful morphological-based identification in the field (Aplin and Opiang 2017), but morphologically cryptic species that can be diagnosed only with genetic markers will still confound results, as these authors found with the detection of various forms within *Paramelomys 'rubex'* and *Rattus 'niobe'*. The application of genetics-based identification in the present study has demonstrated the value of including advanced, but cost-effective and practicable methods to ensure the consistency of identifications among sites, years and investigators.

Having now genotyped four sets of captures, and established which genetic groups are present, it was suggested following the 2019 survey that there would be value in considering whether there are sets of morphological characters that can be used to make these same identifications in the field. The limited information on external morphology available from captures in each of the three surveys was combined with information on body size and mammary formula summarised in the individual accounts of Flannery (1995) (data not shown) and Aplin et al. (2018). Based on this compilation, there appeared to be several issues that currently prevent the derivation of a field-based identification system: the range of external measurements overlaps for many species; morphological measurements from subadults can be lead to misidentification; fur colour can be variable; information on diagnostic characters (for example, possibly useful features of tail scalation, or those of the skull and teeth) is unavailable or not of practical use for the PMA3 study; and the ability of investigators to make consistent and accurate identifications will vary among people with different levels of experience. Given the importance of detecting even rare occurrences of invasive species, it does not seem either cost effective or practical to replace the genetics-based identification established over the past four surveys.

In the 2021 survey, there was a reasonable rate of correct identification made in the field (85.5%), but some species appeared to present a persistent problem for identification (Appendix 3.7). Most notable is *Paramelomys* sp. cf. *mollis* AC. Given that the same lead investigators have been making identifications during the last three surveys (E. Kale, D. Okena), which was also informed by the groundwork of Aplin and Opiang (2017) and Armstrong and Aplin (2017), familiarity with the local assemblage is obviously key to making consistent identifications. A higher rate of correct identification will only be possible if there is opportunity for follow-up examinations that include comparison of captured vouchers and museum specimens from a broader range of localities. Taxonomic work is also needed so that the species-level distinctness of the putative new species discovered by the PMA3 study can be better understood.

Conclusions

The 2021 survey has contributed to each of the five specific objectives of the study:

1. *Monitor the impact of linear ROW infrastructure on non-volant mammal communities, as a more general indicator of the influence of the ROW of the adjacent forest habitat.*

The most relevant result documented to date is the continued lack of a significant shift in Species Richness, abundance or species composition at increasing distances from the ROW, in either BAA, and in any survey year, indicating that there has been no detectable impact of edge effects from the ROW on these taxa.

2. Document the diversity and abundances of small, non-volant mammals within the two BAAs.

The trapping effort appears to have encountered almost all species that can be trapped with medium-sized box traps, but camera trapping results from the past four surveys and earlier baseline work has identified the presence of additional taxa.

A greater appreciation of apparent population fluctuations in the four most common rodents (*Paramelomys* sp. cf. *mollis* AC, *Paramelomys* sp. cf. *rubex* A, *Rattus* sp. cf. *niobe* B, *Rattus* sp. cf. *niobe* D) was gained from comparing the number of captures across the four surveys. Temporal fluctuations in local density as part of natural processes will need to be considered as context when making interpretations about the possible effects of linear infrastructure, as well as potentially confounding factors such as changing the bait type, and the timing of surveys.

3. Identify species of conservation significance (including new or undescribed species) within each of the BAAs and, where practicable, determine their special sensitivities.

The genetics-based identification methods used in in all four surveys has helped to identify several likely undescribed species within *Paramelomys 'lorentzii'*, *Paramelomys 'mollis'*, *Paramelomys 'rubex'*, and *Rattus 'niobe'*. Further consideration of the *Pogonomys* species present is also warranted. Our results demonstrate that some apparently common and widespread species are complexes of genetically distinct populations with presumably smaller distributions. Species with smaller distributions have a greater chance of being affected significantly by developments.

Understanding the special sensitivities of native rodent species, and their interactions with commensal predators such as dogs, competitors such as *Rattus rattus*, and novel pathogens that might spread from invasive rodent species, requires further study.

4. Monitor the status of exotic mammal species in each of the BAAs.

No invasive species of rodent was detected in 2021, even given the much greater capture rate of individual rodents compared to previous years. This is important because an increasing population of invasive rodent species is likely to be the initial and most conspicuous indication of an edge effect that has the potential to affect the native rodent assemblage.

5. Assess the usefulness of non-volant mammal communities in each of the BAAs more broadly as potential indicators of change in habitat quality.

Native rodents have so far been resilient to the removal of adjacent forest for the pipeline and access road because intact habitat extends right to the forest edge. However, if these forest edge habitats degrade in future then we may yet detect changes in the rodent community, including the increased presence of invasive rodents.

Recommendations

- It is recommended that the live-trapping rodent component of the study continue, because the results from four monitoring surveys are demonstrating the value of collecting long term data to understand natural variation in presence and abundance. Given the key overall aim of being able to detect change over the long term, any departures form baseline will be more evident from the understanding of rodent diversity and relative abundance established over the first four surveys.

- The recommendations of Aplin and Opiang (2017; internal version that includes the recommendations) should be revisited, with some revision based on our refined understanding of the rodent assemblage in the BAAs:

    - Consideration should be given to a rapid assessment of the presence of *Rattus rattus* and *R. exulans* in inhabited areas around the HGCP to provide data on how common these species are, and whether these populations are a potential source for invasions along the access road and pipeline on Hides Ridge. Records from contractors involved in pest control are limited and extending box trapping effort and other survey methods to areas around HGCP would be the most effective approach.

    - Continue to build upon the genetic work that has been initiated here because morphologically diagnostic characters for most rodent species in the study area are inadequate for consistently accurate field identifications. An ideal situation would be to support a Papua New Guinean postgraduate student working on rodent taxonomy with advanced methods and access to museum specimens in Australian and other relevant institutions internationally.

    - Consideration should be given to making an assessment of the vegetation community structure around sampling transects using a rapid assessment measure, so that future studies have a quantitative basis for making interprtations and associations with habitat conditions.

References

Aplin, K. P. & Singleton, G. R. 2003. Balancing rodent management and small mammal conservation in agricultural landscapes: challenges for the present and the future. Pp. 80–88 *In*: G. R. Singleton, L. A. Hinds, Krebs, C. J. & Spratt, D. M. (Editors) *Rats, mice and people: rodent biology and management*. ACIAR Monograph No. 96.

Aplin K. P. & Opiang, M. 2017. Chapter 5 – Non-volant mammals (rodents and marsupials). Pp. 141–208 *In*: Richards, S. J. (Editor) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Aplin K., Armstrong, K. & Woxvold, I. 2018. Non-volant mammals. Pp. 118–143 *In*: Richards, S. J. (Editor) Identification guide to flora and fauna of Hides Ridge and the Agogo range (Moro), Papua New Guinea. ExxonMobil PNG Limited, Port Moresby.

Armstrong, K. N. & Aplin K. P. 2017. Chapter 7 – Enhancing biological monitoring with genetic information. Pp. 255–269 *In*: Richards, S. J. (Editor) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Armstrong, K. N., Kale, E. & Amick, P. 2019. Chapter 4 – Small non-volant mammals (Rodents). Pp. 121–151 *In*: Richards, S. J. (Editor) *Results of the second PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. ExxonMobil PNG Limited, Port Moresby.

Armstrong, K. N., Kale, E., Okena, D. & Dahl, G. 2021. Chapter 3 – Small non-volant mammals (Rodents). pp. 113–145 *In*: Richards, S. J. (Editor). *Results of the third PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 8 August–2 September 2019*. ExxonMobil PNG Limited, Port Moresby.

Begon, M. 2006. Disease: health effects on humans, population effects on rodents. Pp. 10–19 *In*: Singleton, G. R., Hinds, L. A., Krebs, C. J. and Spratt, D. M. (Editors) 2003. Rats, mice and people: rodent biology and management. ACIAR Monograph No. 96, 564 pp.

Berry, A. J., Anderson, T. J. C., Amos, J. N. & Cook, J. M. 1987. Spool-and-line tracking of giant rats in New Guinea. *Journal of Zoology* 213: 299–303.

Dwyer, P. D. 1978. A study of *Rattus exulans* (Peale) (Rodentia: Muridae) in the New Guinea highlands. *Wildlife Research* 5: 221–248.

Elliott, T. F. & Vernes, K. 2020. Notes on the diets of four rodent species from Goodenough Island. *Australian Mammalogy*. https://doi.org/10.1071/AM20022

Flannery, T. F. 1995. *The mammals of New Guinea*. Revised edition. Reed Books, Chatswood, NSW.

Grewe, P. M., Feutry, P., Hill, P. L., Gunasekera, R. M., Schaefer, K. M., Itano, D. G., Fuller, D. W., Foster, S. D. & Davies, C. R. 2015. Evidence of discrete yellowfin tuna (*Thunnus albacares*) populations demands rethink of management for this globally important resource. *Scientific Reports* 5: 16916.

Gruber, B., Unmack, P. J., Berry, O. F. & Georges, A. 2018. dartR: An r package to facilitate analysis of SNP data generated from reduced representation genome sequencing. Molecular Ecology Resources 18: 691–699. URL: https://doi.org/10.1111/1755-0998.12745

Harris, L. D. 1988. Edge effects and conservation of biotic diversity. *Conservation Biology* 2: 330–332.

Jackson, K. L. & Woolley, P. A. 1993. The diet of five species of New Guinean rodents. *Science in New Guinea* 19: 77–86.

Kale, E., Whitmore, N., Mack, A. L. & Wright, D. D. 2012. Survival rates of *Rattus verecundus* and *Paramelomys platyops* in a murid rich tropical rainforest of Papua New Guinea. *Pacific Conservation Biology* 18: 26–32.

Kale, E., Armstrong, K. N. & Amick, P. 2018a. Chapter 1.6. Mammals of the Wau Creek proposed Wildlife Management Area, Gulf Province, Papua New Guinea. pp. 125–151 *In*: Richards, S. J. (Editor) *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kale, E., Armstrong, K. N. & Amick, P. 2018b. Chapter 2.6. Mammals of the Uro Creek catchment, Gulf Province, Papua New Guinea. Pp. 245–270 *In*: Richards, S. J. (Editor) *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kale, E., Armstrong, K. N. & Amick, P. 2018c. Chapter 3.6. Mammals of the Lake Kutubu Wildlife Management Area, Southern Highlands Province, Papua New Guinea. Pp. 353–378 *In*: Richards, S. J. (Editor). *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kilian A., Huttner E., Wenzl P. E., Jaccoud D., Carling J, Caig V., Evers M., Heller-Uszynska K. A., Cayla C., Patarapuwadol S. & Xia, L. 2003. The fast and the cheap: SNP and DArT-based whole genome profiling for crop improvement. Pp. 27–31 *in*: Pro*ceedings of the international congress in the wake of the double helix: from the green revolution to the gene revolution*.

Kilian, A., Wenzl, P., Huttner, E., Carling, J., Xia, L., Blois, H., Caig, V., Heller-Uszynska, K., Jaccoud, D., Hopper, C. & Aschenbrenner-Kilian, M., 2012. Diversity arrays technology: a generic genome profiling technology on open platforms. Pp. 67–89 *in Data production and analysis in population genomics*. Humana Press, Totowa, NJ.

Laurance, W. F. 1997. Responses of mammals to rainforest fragmentation in tropical Queensland: a review and synthesis. *Wildlife Research* 24: 603–612.

McPhee, E. C. 1988. Ecology and diet of some rodents from the Lower Montane Region of Papua New-Guinea. *Wildlife Research* 15: 91–102.

Mortelliti, A. & Boitani, L. 2006. Patterns of rodent species diversity and abundance in a Kenyan relict tropical rainforest. pp. 365–380 *In*: *Forest Diversity and Management*. Springer, Dordrecht.

Peterson, B. K., Weber, J. N., Kay, E. H., Fisher, H. S. & Hoekstra, H. E. 2012. Double digest RADseq: an inexpensive method for de novo SNP discovery and genotyping in model and non-model species. *PloS one* 7(5): p.e37135.

Puettker, T., Meyer–Lucht, Y. & Sommer, S. 2008. Fragmentation effects on population density of three rodent species in secondary Atlantic Rainforest, Brazil. *Studies on Neotropical Fauna and Environment* 43: 11–18.

R Core Team 2021. *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. URL: https://www.R-project.org/

Wolf, J. O. & Edge, W. D. 2003. A retrospective analysis of a vole population decline in western Oregon, USA. Pp. 47–50 *In*: Singleton, G. R., Hinds, L. A., Krebs, C. J. & Spratt, D. M. (Editors) *Rats, mice and people: rodent biology and management*. ACIAR Monograph No. 96.

Woxvold, I. & Aplin, K. 2017. Chapter 4 – Camera trap monitoring of terrestrial mammals and birds: A pilot study. Pp. 121–139. *In*: Richards, S. J. (Editor) *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Woxvold, I. A. & Legra, L. 2019. Chapter 3 – Camera trap monitoring of terrestrial mammals and birds. Pp. 63–120. *In*: Richards, S. J. (Editor) *Results of the second PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. ExxonMobil PNG Limited, Port Moresby.

Woxvold, I. A., Tulai, S., Yama, S., Mani, A. 2021. Chapter 2 – Camera trap monitoring of terrestrial mammals and birds. pp. 53–111 *In*: Richards, S. J. (Editor). *Results of the third PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 8 August–2 September 2019*. ExxonMobil PNG Limited, Port Moresby.

Plate 1

Figure 3.12. Uneven-toothed Rat (*Anisomys imitator*).

Figure 3.13. A white-eared giant rat (*Hyomys* sp.).

Figure 3.14. Elegant Water Rat (*Leptomys elegans*).

Figure 3.15. *Paramelomys* sp.

Figure 3.16. White-tailed Giant Rat (*Uromys caudimaculatus*).

Figure 3.17. Enock Kale and Daniel Okena weighing a captured rodent prior to release.

Appendix 3.1. Summary of how the traps across each transect were pooled for statistical analyses.

Trap number	Distance from ROW (m)	Stated interval	Analysis Bin	Number of traps
1	10	0–20	1	8
2	16			
3	22	20–70	2	8
4	28			
5	34			
6	40			
7	46			
8	52			
9	58			
10	64			
11	70	70–120	3	8
12	76			
13	82			
14	88			
15	94			
16	100			
17	106			
18	112			
19	118	120–170	4	8
20	124			
21	130			
22	136			
23	142			
24	148			
25	154			
26	160			
27	166	170–220	5	9
28	172			
29	178			
30	184			
31	190			
32	196			
33	202			
34	208			
35	214			
36–60	220–364	220–370	6	13–25

Notes

1. Stated interval is given for ease of comprehending the categories in the plots; see actual distances and trap numbers for more detail.

2. Aplin and Opiang (2017) recorded the origin of captures in the traps at the beginning of the transect that followed the ROW as 2.1–2.6 (see examples in their Appendix 5.4), even though the transect diagram in their Figure 5.1 labelled these as 1.1–1.6. The 2017 – 2021 survey trap positions were numbered according to the scheme presented in Figure 3.1. There is no implication for the statistical analyses because the same number of traps is contained in the first bin for all survey years.

3. For all transects except M4, and excluding the six traps at the begonning parallel to the ROW, the total number of traps run along the transect perpendicular to the ROW by Aplin and Opiang (2015) was 40 (total length 240 m). In the two subsequent surveys, the maximum number of traps was 34–37. For statistical analyses, the maximum length of transects has been standardised at 35 traps, and captures made at trap numbers greater than this have excluded.

4. Aplin and Opiang (2015) were not able to sample transect M5 that was used for the frog and bat recordings because of its distance from vehicular access. Instead, transect M4 was extended from 240 m to 336 m. This was continued in 2017 – 2021, but to a maximum of 48 trap positions rather than 60. For plotting purposes only, all trap positions for transect M4 between 36–60 were combined into a single category, and this category (‘bin 6’) was not used in statistical analyses.

Appendix 3.2. Summary of whole specimen vouchers taken on the survey, which have been deposited in the South Australian Museum.

Specimen and tissue number	Verified SNP-based identification	Transect	Trap Station	Sex
MEL00801	*Paramelomys* sp. cf. *rubex* A	H5	13	M
MEL00815	*Paramelomys* sp. cf. *mollis* AC	H3	1.5	F
MEL00828	*Rattus* sp. cf. *niobe* B	H5	4	M
MEL00899	*Pogonomys macrourus*	H1	34	F
MEL01508	*Echymipera kalubu*	M1	1.4	M
MEL01512	*Rattus* sp. cf. *niobe* D	M3	10	M

Appendix 3.3. All novel captures made in 2021 (Tran: transect; Trap: trap number along transect; data in second column follows from the bottom of the first).

MEL	Tran	Trap
Echymipera kalubu		
MEL01508	M1	1.4
MEL01515	M1	
Paramelomys platyops		
MEL01504	M4	22
MEL01514	M4	25
MEL01580	M4	22
MEL01581	M4	5
MEL01598	M4	22
MEL01603	M4	1.5
MEL01606	M4	2
MEL01607	M4	15
MEL01608	M4	32
Paramelomys sp. cf. *lorentzii* C		
MEL01511	M3	19
MEL01517	M1	9
MEL01572	M1	20
MEL01596	M2	7
MEL01602	M3	27
Paramelomys sp. cf. *mollis* AC		
MEL00814	H3	1.5
MEL00815	H3	1.5
MEL00816	H3	15
MEL00833	H3	15
MEL00850	H1	22
MEL00861	H3	1.4
MEL00862	H3	15
MEL00867	H2	13
MEL00880	H3	3.0
MEL00889	H2	14
MEL00894	H2	12
MEL01526	H3	1.1
MEL01545	H2	8
MEL01552	H1	6
MEL01553	H1	1.2
MEL01554	H1	1.6
MEL01558	H2	8
MEL01569	H1	34
MEL01570	H1	1.5
Paramelomys sp. cf. *rubex* A		
Rattus sp. cf. *niobe* B		
MEL00836	H6	1
MEL00837	H6	2
MEL00838	H5	2
MEL00839	H5	4
MEL00840	H5	13
MEL00842	H5	25
MEL00843	H3	1.3
MEL00844	H3	2
MEL00845	H3	33
MEL00846	H3	35
MEL00847	H3	34
MEL00848	H1	23
MEL00849	H1	36
MEL00851	H1	17
MEL00853	H1	1.5
MEL00855	H6	7
MEL00856	H6	18
MEL00858	H5	1.1
MEL00859	H5	13
MEL00860	H5	23
MEL00863	H2	1.1
MEL00864	H2	14
MEL00865	H2	1.4
MEL00868	H1	10
MEL00869	H2	17
MEL00871	H6	1.6
MEL00872	H6	34
MEL00876	H1	1.5
MEL00877	H6	26
MEL00878	H5	13
MEL00879	H5	25
MEL00881	H5	30
MEL00882	H3	1.1
MEL00884	H3	26
MEL00885	H3	19
MEL00886	H3	33
MEL00887	H2	1.1
MEL00888	H2	1.3
MEL00890	H2	13

MEL	Tran	Trap
MEL00801	H5	13
MEL00802	H5	14
MEL00805	H5	26
MEL00808	H5	32
MEL00810	H5	3
MEL00817	H3	17
MEL00818	H3	18
MEL00819	H3	20
MEL00820	H3	26
MEL00822	H3	35
MEL00825	H5	29
MEL00832	H3	17
MEL00834	H3	7
MEL00841	H5	23
MEL00854	H6	1.4
MEL00857	H6	28
MEL00883	H3	18
MEL01524	H6	16
MEL01525	H5	3
MEL01527	H3	1.4
MEL01528	H3	30
MEL01534	H2	2
MEL01543	H1	1.5
MEL01544	H2	1.4
MEL01559	H6	3
MEL01560	H6	1.1
MEL01561	H6	18
MEL01564	H6	29
MEL01565	H6	29
MEL01566	H6	25
MEL01567	H6	25
Paramelomys sp. cf. *rubex* B		
MEL01506	M1	28
MEL01516	M1	28
MEL01519	M1	29
MEL01571	M1	29
MEL01582	M1	28
MEL01594	M1	28
Paramelomys sp. cf. *rubex* F		
MEL00852	H1	8
MEL00866	H2	4
MEL00870	H1	19

MEL	Tran	Trap
MEL00891	H2	32
MEL00892	H2	23
MEL00893	H2	4
MEL00895	H2	33
MEL00896	H2	1.4
MEL00897	H1	8
MEL00898	H1	10
MEL01522	H6	29
MEL01523	H6	26
MEL01529	H1	36
MEL01530	H1	10
MEL01531	H1	1.2
MEL01535	H2	16
MEL01536	H2	18
MEL01537	H2	32
MEL01541	H1	8
MEL01546	H2	12
MEL01547	H2	32
MEL01550	H1	27
MEL01555	H2	12
MEL01556	H2	4
MEL01557	H2	23
MEL01562	H6	35
MEL01563	H6	35
MEL01568	H2	12
MEL0857	H6	28
no MEL	H3	3
Rattus sp. cf. *niobe* D		
MEL01501	M3	4
MEL01502	M3	10
MEL01503	M3	2
MEL01507	M1	18
MEL01509	M2	33
MEL01510	M2	35
MEL01512	M3	10
MEL01513	M3	
MEL01518	M2	24
MEL01520	M1	4
MEL01573	M1	17
MEL01574	M1	4
MEL01575	M1	1.3
MEL01576	M1	1.6

MEL	Tran	Trap
MEL00875	H1	22
MEL01533	H2	1.2
MEL01542	H1	1.6
MEL01549	H1	36
MEL01551	H1	23
Pogonomys macrourus		
MEL00899	H1	34
MEL01548		
Rattus sp. cf. *niobe* B		
MEL00803	H5	20
MEL00804	H5	23
MEL00806	H5	27
MEL00807	H5	30
MEL00809	H5	34
MEL00811	H5	37
MEL00812	H3	2
MEL00813	H3	1.3
MEL00821	H3	33
MEL00823	H5	34
MEL00824	H5	30
MEL00826	H5	25
MEL00827	H5	20
MEL00828	H5	4
MEL00829		
MEL00830	H3	18
MEL00831	H3	18
MEL00835	H3	1.3

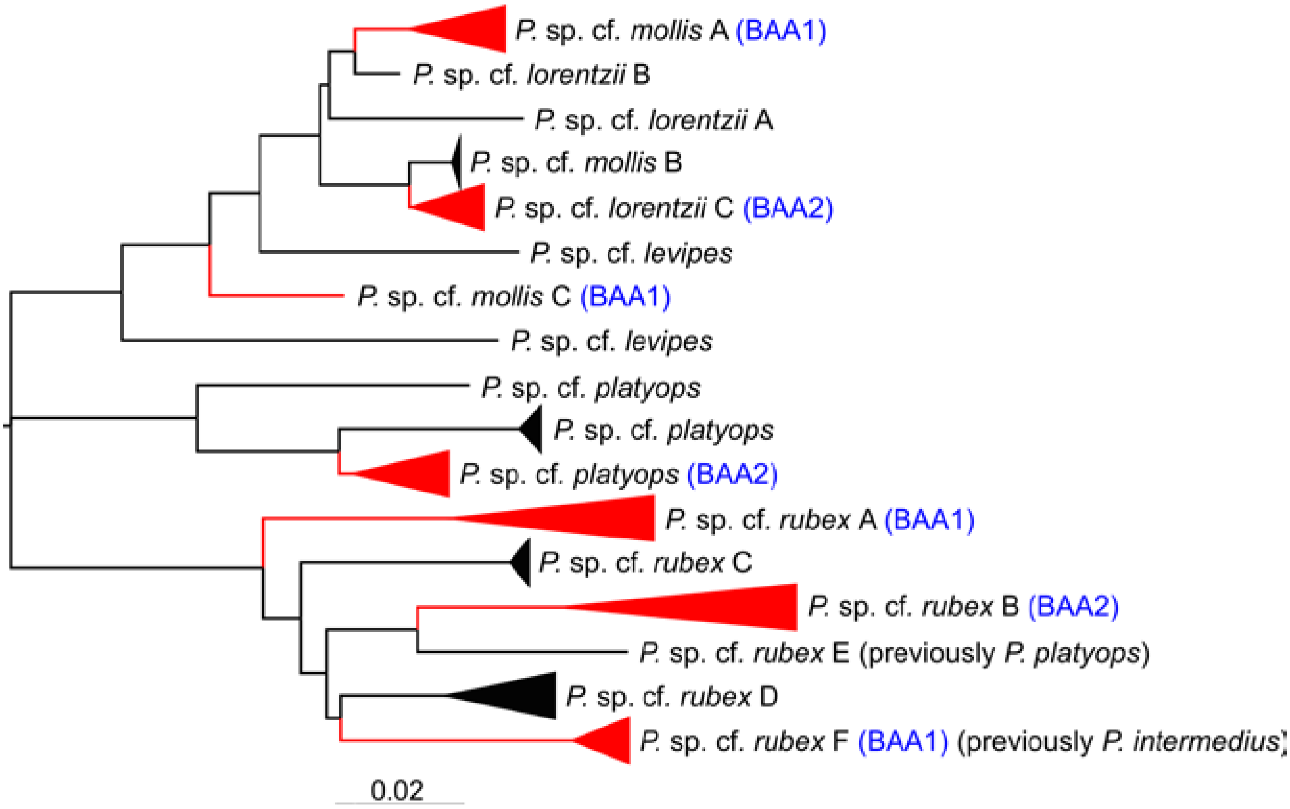
MEL	Tran	Trap
MEL01577	M2	35
MEL01578	M2	1.3
MEL01579	M2	6
MEL01583	M1	17
MEL01584	M1	13
MEL01585	M1	1.6
MEL01586	M1	1.3
MEL01587	M2	35
MEL01588	M2	7
MEL01589	M2	1.2
MEL01590	M2	1.2
MEL01591	M3	22
MEL01592	M3	1.2
MEL01593	M1	29
MEL01595	M1	6
MEL01597	M2	8
MEL01599	M1	29
MEL01600	M3	1.2
MEL01601	M3	34
MEL01604	M3	1.5
MEL01605	M3	34
no MEL	M1	1.5
Rattus steini		
MEL01532	H1	9
Rattus verecundus		
MEL01505	M4	28

Appendix 3.4. All recaptures made in 2021 (Tran: transect; Trap: trap number along transect).

Identification validated	Tran	Trap
Paramelomys sp. cf. *mollis* AC	H2	5
Paramelomys sp. cf. *mollis* AC	H2	18
Paramelomys sp. cf. *mollis* AC	H3	12
Paramelomys sp. cf. *mollis* AC	H3	17
Paramelomys sp. cf. *rubex* A	H1	1.3
Paramelomys sp. cf. *rubex* A	H1	27
Paramelomys sp. cf. *rubex* A	H3	17
Paramelomys sp. cf. *rubex* A	H5	19
Paramelomys sp. cf. *rubex* A	H5	29
Paramelomys sp. cf. *rubex* A	H5	30
Paramelomys sp. cf. *rubex* B	M1	29
Rattus sp. cf. *niobe* B	H1	2
Rattus sp. cf. *niobe* B	H1	20
Rattus sp. cf. *niobe* B	H1	20
Rattus sp. cf. *niobe* B	H1	23
Rattus sp. cf. *niobe* B	H2	1.1
Rattus sp. cf. *niobe* B	H2	1.3
Rattus sp. cf. *niobe* B	H2	1.4
Rattus sp. cf. *niobe* B	H2	4
Rattus sp. cf. *niobe* B	H2	5
Rattus sp. cf. *niobe* B	H2	30
Rattus sp. cf. *niobe* B	H2	33
Rattus sp. cf. *niobe* B	H2	
Rattus sp. cf. *niobe* B	H3	1.3
Rattus sp. cf. *niobe* B	H3	1.6
Rattus sp. cf. *niobe* B	H3	12
Rattus sp. cf. *niobe* B	H3	33
Rattus sp. cf. *niobe* B	H3	33
Rattus sp. cf. *niobe* B	H3	34
Rattus sp. cf. *niobe* B	H3	34

Identification validated	Tran	Trap
Rattus sp. cf. *niobe* B	H3	34
Rattus sp. cf. *niobe* B	H3	35
Rattus sp. cf. *niobe* B	H5	37
Rattus sp. cf. *niobe* B	H6	1.1
Rattus sp. cf. *niobe* B	H6	1.3
Rattus sp. cf. *niobe* B	H6	5
Rattus sp. cf. *niobe* D	M1	1.2
Rattus sp. cf. *niobe* D	M1	4
Rattus sp. cf. *niobe* D	M1	7
Rattus sp. cf. *niobe* D	M1	13
Rattus sp. cf. *niobe* D	M1	16
Rattus sp. cf. *niobe* D	M1	18
Rattus sp. cf. *niobe* D	M1	23
Rattus sp. cf. *niobe* D	M1	28
Rattus sp. cf. *niobe* D	M2	3
Rattus sp. cf. *niobe* D	M2	3
Rattus sp. cf. *niobe* D	M2	3
Rattus sp. cf. *niobe* D	M2	8
Rattus sp. cf. *niobe* D	M2	8
Rattus sp. cf. *niobe* D	M2	8
Rattus sp. cf. *niobe* D	M2	24
Rattus sp. cf. *niobe* D	M2	35
Rattus sp. cf. *niobe* D	M3	1.3
Rattus sp. cf. *niobe* D	M3	1.5
Rattus sp. cf. *niobe* D	M3	4
Rattus sp. cf. *niobe* D	M3	10
Rattus sp. cf. *niobe* D	M3	10
Rattus sp. cf. *niobe* D	M3	10
Rattus sp. cf. *niobe* D	M3	10
Rattus sp. cf. *niobe* D	M4	39
Rattus verecundus	M4	41

Appendix 3.5. Revision and correction of *Paramelomys* taxon designations based on the mitochondrial DNA tree from the 2015 survey (with major clades collapsed), and the DArTseq Principle Components Analyses of 2017 and 2019. Red colour indicates that at least one sample collected in 2015 from either BAA 1 or BAA 2 was present (modified from Aplin and Opiang 2017: their Figure A5.3.4).



Notes

1. *'P. mollis'* is split into three groups: A, B and C. All individuals affiliated with '*P. mollis'* that were captured on all three surveys group together using genome-scale DNA markers. In 2015, five captures were of *P.* sp. cf. *mollis* A and one capture of *P.* sp. cf. *mollis* C from Hides Ridge. The mitochondrial haplotypes of '*P. mollis'* captured in 2017–2021 are unknown because the cytochrome-*b* gene was not sequenced from individuals captured in those years. The designation '*P*. sp. cf. *mollis* AD' was incorrectly used in Armstrong et al. (2019)—the correct designation is *P*. sp. cf. *mollis* AC.

2. Letter designations were not applied by Aplin and Opiang (2017) to the three clades affiliated with '*P. lorentzii*', nor were they applied by Armstrong et al. (2019), though reference was made to *P. lorentzii* '1' in the Results section of that report for animals captured in the study area. All captures from the four surveys to date have been designated in the present report as *P.* sp. cf. *lorentzii* C.

3. The major clade containing individuals affiliated with *P. intermedius*, *P. platyops*, and *P. rubex* is assigned in the present report in its entirety to '*P. rubex*'. In 2017, four individuals were captured from BAA 1 that grouped with context samples (from Bobole and Pn'yang) that comprise the '*P. intermedius'* mitochondrial clade in the tree above. A further nine individuals were captured in 2019 from BAA 1 that grouped with these samples, and they have been renamed in Armstrong et al. (2021) and the present report as *P*. sp. cf. *rubex* F.

4. Representatives of only one mitochondrial DNA clade of '*P. platyops'* have been recorded in the study area to date, and the membership of this clade extends across a wide area of Papua New Guinea.

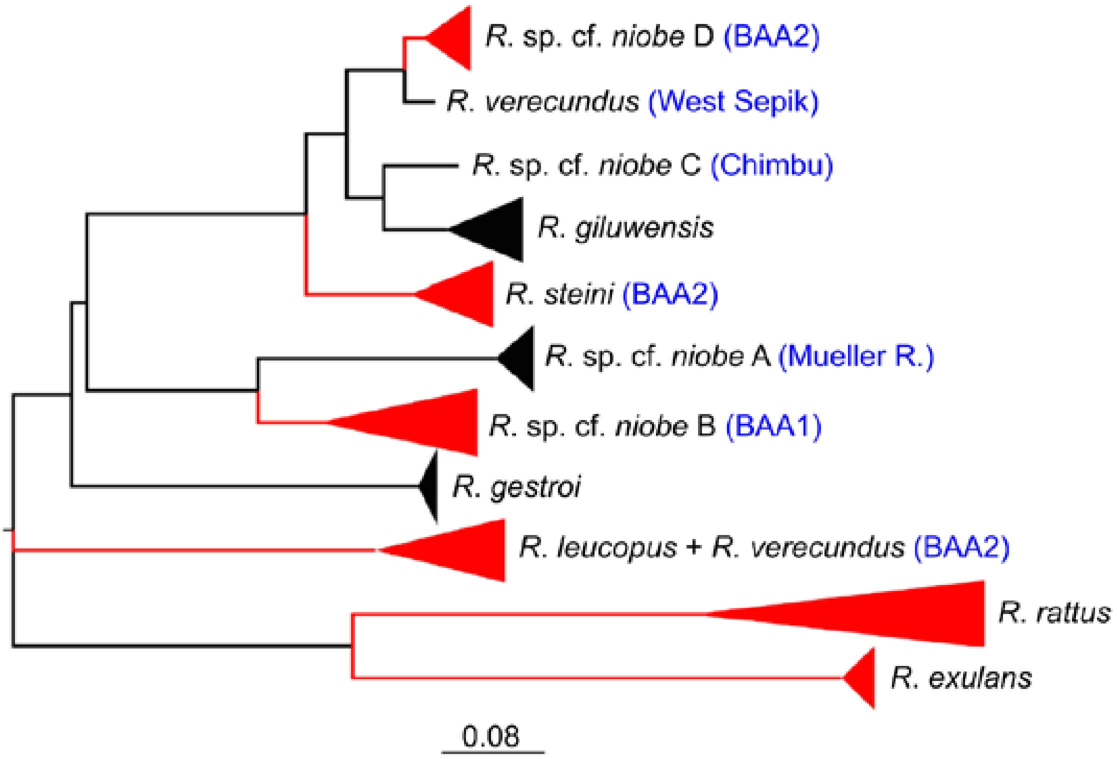
Appendix 3.6. Allocation of names in *Rattus* based on mitochondrial DNA markers, as presented by Aplin and Opiang (2017: fig. A5.3.2), showing that the nomenclature is not consistent with phylogenetic pattern using this marker type.

This mitochondrial marker system was not used in subsequent years because of the potential for mismatching external appearance, mitochondrial genotype and genome-scale genotype using DArTseq markers. The case of two individuals captured in 2015 illustrates the greater utility (combined with its cost effectiveness) of the DArTseq markers over the other two options. Two individuals captured in BAA 2 had the external appearance of *R.* sp. cf. *niobe* D, but the mitochondrial genotypes of other species.

One of these (ABTC141286) had both the mitochondrial and DArTseq genotypes of *R. verecundus* ('spiny'; the form more closely related to the *R. leucopus* group). Either genetic marker would have given an accurate identification in this case.

The second individual (ABTC141260) had the mitochondrial genotype of *R. steini*, but a DArTseq genotype of *R.* sp. cf. *niobe* D that it resembled morphologically. Given such a demonstrated possibility for mitochondrial capture, where the mitochondrial genome of one species replaces that of another, a genetic marker based on the nuclear genome was considered a more reliable tool for confirming field-based identifications.

The genetic distance is relatively large between native New Guinea *Rattus* species and the invasive species *R. exulans* and *R. rattus*. This makes distinguishing them relatively straightforward using genetics, but their appearance in the field can prompt a suggestion of a novel native taxon, which is a realistic possibility in PNG.



Appendix 3.7. Summary of correspondence between the genetics-based identification and the initial morphological-based field identification, for all 2021 novel captures sequenced successfully.

Genetics-based ID	Field ID	Count
Correct		
Echymipera kalubu	*Echymipera kalubu*	2
Paramelomys sp. cf. *mollis* AC	*Paramelomys* sp. cf. *mollis*	17
Paramelomys sp. cf. *rubex* A	*Paramelomys* sp. cf. *rubex*	24
Paramelomys sp. cf. *rubex* B	*Paramelomys* sp. cf. *rubex*	6
Paramelomys sp. cf. *rubex* F	*Paramelomys* sp. cf. *rubex*	5
Pogonomys macrourus	*Pogonomys macrourus*	2
Rattus sp. cf. *niobe* B	*Rattus* sp. cf. *niobe*	69
Rattus sp. cf. *niobe* D	*Rattus* sp. cf. *niobe*	33
Rattus verecundus	*Rattus verecundus*	1
		Correct ID total 159 (85.5%)
Incorrect		
Paramelomys sp. cf. *lorentzii* C	*Paramelomys* sp. cf. *mollis*	5
Paramelomys sp. cf. *mollis* AC	*Paramelomys* sp. cf. *rubex*	2
Paramelomys platyops	*Paramelomys* sp. cf. *mollis*	5
Paramelomys platyops	*Paramelomys* sp. cf. *rubex*	4
Paramelomys sp. cf. *rubex* A	*Paramelomys* sp. cf. *mollis*	7
Paramelomys sp. cf. *rubex* F	*Paramelomys* sp. cf. *mollis*	3
Rattus steini	*Rattus* sp. cf. *niobe*	1
		Incorrect ID total 27 (14.5%)

Appendix 3.8. Summary of species of Muridae that have been recorded from the PMA3 study area at different elevations [m asl] with camera traps (Woxvold and Aplin 2017; Woxvold and Legra 2019; Woxvold et al. 2021; Woxvold et al. this volume); plus 'Additional species': from remains found in five individual owl pellets collected from a cave on Hides Ridge at an elevation of c. 2,065 m asl in 2011 (Aplin and Opiang 2017), from road kills and casual observations made during the PMA3 field surveys, and from the environmental impact assessment work conducted before the construction of the pipeline ROW (summarised in Aplin and Opiang 2017). For camera traps, in each cell the survey years (2015, 2017, 2019, 2021) are presented in order from left to right as one of two symbols: open circle is an absence of observations, closed circle is at least one observation in that survey year; * denotes not yet encountered with box trapping; NYS: not yet seen on camera traps or in box traps.

Camera traps	**1,000 m**	**1,400 m**	**2,200 m**	**2,700 m**
Camera traps	**1,000 m**	**1,400 m**	**2,200 m**	**2,700 m**
Camera traps	**1,000 m**	**1,400 m**	**2,200 m**	**2,700 m**
**Anisomys imitator*	oooo	●●●●	●●●●	o●●●
**Hyomys* sp.	oooo	oooo	●●●●	●●●●
Leptomys elegans	o●●●	●●●●	oooo	oooo
**Mallomys* sp.	●ooo	o●●●	o●●●	o●●●
**Parahydromys asper*	oooo	●oo●	ooo●	oo●●
Paramelomys spp.	●●●●	●●●●	●●●●	●●●●
Pogonomys sp. cf. *loriae*	oooo	oo●o	ooo●	oooo
**Pseudohydromys* sp.	oooo	oooo	oo●●	ooo●
**Uromys anak*	o●●o	o●●●	o●●●	o●●●
Uromys caudimaculatus	●●●●	●●●●	oo●o	oooo
Rattus spp.	●●●●	●●●●	●●●●	●●●●
Additional Species Richness	**2**	**4**	**6**	**6**
Additional species NYS				
Abeomelomys sevia				
Chiruromys vates				
Melomys sp. cf. *dollmani*				
Melomys sp. cf. *rufescens*				
Protochromys sp. cf. *fellowsi*				

Appendix 3.9. Summary of statistical outcomes.

Total Species Richness					
	F	**Df**	**Df.res**	**Pr(>F)**	**Tukey's pairwise**
Distance category	3.76	4	16.42	**0.024***	None indicated
Elevation	2.26	3	4.52	0.21	—
Year	0.85	3	13.17	0.49	—
Distance:Elevation	1.54	12	18.39	0.19	—
Elevation:Year	2.42	9	14.66	**0.06**	—
Distance:Year	0.67	12	43.33	0.77	—
Dist:Elev:Year	0.89	29	42.68	0.62	—

Total Captures					
	F	**Df**	**Df.res**	**Pr(>F)**	**Tukey's pairwise**
Distance category	1.79	4	17.17	0.18	—
Elevation	2.12	3	4.60	0.22	—
Year	7.72	3	12.81	**0.003****	2017 < 2021: P = 0.195
Distance:Elevation	0.94	12	18.97	0.52	—
Elevation:Year	2.79	9	14.67	**0.039***	—
Distance:Year	1.50	12	43.38	0.16	—
Dist:Elev:Year	0.88	29	42.83	0.63	—

Total Captures Hydromyini					
	F	**Df**	**Df.res**	**Pr(>F)**	**Tukey's pairwise**
Distance category	1.04	4	17.34	0.42	—
Elevation	2.02	3	4.71	0.24	—
Year	3.99	3	12.76	**0.033***	2019 < 2021: P = 0.66
Distance:Elevation	1.00	12	19.00	0.48	—
Elevation:Year	1.43	9	14.36	0.26	—
Distance:Year	0.76	12	42.53	0.69	—
Dist:Elev:Year	0.84	29	42.07	0.68	—

Total Captures Rattini					
	F	**Df**	**Df.res**	**Pr(>F)**	**Tukey's pairwise**
Distance category	2.03	4	16.63	0.14	—
Elevation	2.14	3	4.53	0.22	—
Year	5.56	3	12.99	**0.011***	2017 < 2021: P = 0.149
Distance:Elevation	0.80	12	18.63	0.65	—
Elevation:Year	2.31	9	14.84	0.07	—
Distance:Year	1.23	12	43.91	0.29	—
Dist:Elev:Year	0.85	29	43.22	0.67	—

Chapter 4 – Bats

Kyle N. Armstrong, Elizah Nagombi, Steven Sau and Frank Phillip

Rhinolophus euryotis

Summary

Background and aims

The bat component of the PMA3 monitoring study seeks to determine whether ExxonMobil PNG Limited's pipeline ROW and Project roads are causing changes in the adjacent bat communities. These narrow linear disturbance zones represent 'canyon-like' breaks in broadscale extents of closed forest and could represent either habitat loss or foraging area gain for different species with different ecological requirements.

The July 2021 survey used the same acoustics-based survey approach to detect echolocating bat species, and the same 66 recording sites that were established in 2015 on Hides Ridge (BAA 1) and adjacent to Arakubi Quarry and KP 107 (BAA 2) on the Agogo Range near Moro.

The primary aim of the 2021 survey was to determine if there had been a significant change in the diversity and composition of bat communities at increasing distance from the ROW and at different elevations since 2015.

Major results

A total of 21 species was detected in the acoustic recordings, equivalent to previous surveys (2015: 21 spp.; 2017: 23 spp.; 2019: 19 spp.). No trapping was conducted in 2021. Based on combined captures and acoustic recordings from the 2015–2021 surveys, a total of 27 bat species has been documented in the PMA3 study area.

No new species for the study area were encountered in 2021, but a growing understanding of the variation in the call types of some species that emit low frequency echolocation helped to refine the species list. This led to the removal of the Greater Northern Free-tailed Bat *Chaerephon jobensis* from the overall list of species present, and the reallocation of some past records of *C. jobensis* and the Bare-rumped Sheath-tailed Bat *Saccolaimus saccolaimus* to an unidentified (but previously described) species of free-tailed bat *Otomops* sp.

No species classified in a threatened category by the IUCN have yet been recorded, but both species of free-tailed bat *Otomops* sp. that could be present are listed as Data Deficient. In addition, there are two putative new species (*Kerivoula* sp., *Nyctophilus* sp.) that were captured on Hides ridge at c. 2,700 m in 2017. Taxonomic studies are underway, but more than one individual is required for a species description. A third putative new species remains known only from single echolocation traces (unique call type *172 sCF*) at transect M5 in 2015, and at KP 87 adjacent to Lake Kutubu in 2017.

The bat communities within each of the BAAs have not changed significantly over the 6-year monitoring period—the communities in 2021 are equivalent to those documented in 2015 and subsequent years. The compilation of data from standardised surveys over four surveys has revealed some clear ecological patterns, and significantly increased the known elevational range for several species.

Some bat species have benefited from the opening of the canopy. There is a clear difference in Species Richness and other measurements (Phylogenetic Diversity, Functional Diversity) between the open area at the start of the transect (0–20 m), and the remaining recording positions that extend back through the forest (20–220 m). This difference between 'open' and 'closed' habitats is due to either the exclusive presence, or a bias in the relative number of detections, of species that have echolocation call types associated with foraging in Edge habitats. The level of detection of species that prefer the forest interior has remained similar, and they are found up to the forest edge. Thus, the linear infrastructure has advantaged some species forage in the forest (9 Edge species), but has had no detectable effect on the remainder (3 Open species and 13 Clutter species).

Entwined with the pattern related to forest cover change are natural environmental factors and the associated adaptations of bats that determine which species are present—a diminishing Species Richness and changing species composition with increasing elevation, and variation in abundance related to local factors such as the availability of nearby rocky outcrops for cave-roosting species. Some bat species are higher elevation specialists that replace species not adapted to cooler habitats where food resources are limited.

Conclusions and recommendations

The combined results from the 2015–2021 surveys suggest that forest adjacent to the ROW has so far retained its value for diverse and intact communities of bats at different elevations. The acoustic bat monitoring component should be continued in future surveys because of a demonstrated ability to detect bat responses to open areas and the forest edge.

There have been no detectable changes in the diversity of bat communities within BAA 1 and BAA 2 since 2015. Likewise, no major structural changes that would influence the bat assemblage appear to have occurred at the edge of transects during the monitoring period. However, measurement of a suitable set of habitat covariates (e.g., from drone-mounted multi-spectral imaging) at each transect as a basis for interpretation of results in future years is currently lacking.

The most obvious pattern detected in the distribution of bat species is the large elevational difference in Species Richness between BAA 1 (9 species) and BAA 2 (22 species in total over four surveys). Two species have been identified as high elevations specialists.

This study has not detected any gradual changes in the bat community with increasing distance from the forest edge. There have been minimal impacts of edge effects, which was evident mainly from the relatively high usage by species that forage in the forest edge flight space. However, some species have responded positively to the opening of canopy along the ROW and access roads. Further consideration could be given to quantifying activity levels in certain frequency bands, and of particular species identified in the Indicator Species analysis as Edge and Clutter species, including from past recordings.

Further effort could be given on future surveys to capturing species of bat that need to be distinguished from others with similar echolocation call types (detecting *Pipistrellus* that have calls similar to *Miniopterus* species), that need specimens to complete identification and taxonomic assessments (the source of the *172 sCF* call type; *Kerivoula* sp.; *Nyctophilus* sp.), and to confirm the identity and to collect quality reference calls to allow greater confidence in acoustics-based identifications for some species (*Austronomus kuboriensis*, *Otomops* sp.). Acoustics lures will likely increase capture probability of all these species (Hill et al. 2015).

Introduction

Background

The initial responses of wildlife to the encroachment of infrastructure footprints in pristine forest habitats can be informative context for predictions in future environmental impact assessments. Such responses might take several years to document because they can be subtle and masked by natural variation and sampling limitations. Specific knowledge about the responses of ecosystem components to disturbances in Papua New Guinea (PNG) is deficient in the published literature, and predictions about the effects of developments are typcially made in a general sense using knowledge derived from the same broad ecosystem type elsewhere in the world. Having the opportunity to observe changes to PNG wildlife assemblages through post-construction monitoring can therefore be valuable.

PNG retains around 70% of its natural forest cover (Shearman and Bryan 2015), with broad expanses relatively undissected by linear infrastructure corridors for roads, power transimission and pipelines. As a consequence, the effects of linear habitat disturbance on PNG fauna have not been well documented. Studies within broad areas of intact habitat have the potential to be informative about the responses of animals to a single type of initial perturbation because the effects are not confounded by decades or centuries of other types of disturbance on the same habitat patches. It is within this context that the PMA3 monitoring program considers both the short- and longer-term effects of a linear infrastructure corridor on closed lower- and mid-montane forest ecosystems. The study sites are located along an elevation gradient that spans two adjacent mountain ranges located in the Southern Highlands and Hela Provinces (to the west and east of Komo), and the program periodically (every two years) measures the diversity and composition of selected major vertebrate groups, including bat communities. The PMA3 study represents a unique long-term effort to examine the response of bat communities to linear gaps in broad areas of pristine tropical forest ecosystems.

The first PMA3 survey in 2015 documented 21 bat species (revised total) and found a clear pattern of increasing bat diversity and changing species composition with decreasing elevation in the BAA project areas (Armstrong 2017). That study detected a marginally significant higher diversity of bat species assemblages immediately adjacent to the open areas of the PNG LNG pipeline ROW and access road, but only at the lowest elevation (1,000 m asl.). Several species appeared to benefit from having increased access to open foraging areas and vegetation edges at this elevation. Repeated sampling in 2017 and 2019 (Armstrong et al. 2019; Armstrong et al. 2021a) provided further support for these patterns and documented an additional five species in the study areas.

Edge effects of linear infrastructure corridors on bats

Edge effects have been studied for decades (Harris 1988), but there are few long-term studies of bat communities occupying forest edge habitats. Most short-term studies derive from temperate habitats in Europe where landscapes have been subject to modification for hundreds of years, where the landscape is a mosaic of different landuses, and where patches of vegetated habitat may bear little resemblance to the condition before widespread anthropogenic influence. In this landscape, roads increase connectivity for people but can reduce it dramatically for the populations of animals remaining in dissected landscapes. The remaining natural habitats are then encroached upon by factors that further reduce habitat quality and biodiversity beyond the actual carriageways (Trombulak and Frissel 2000; Spellerberg 2002; Coffin 2007; Fahrig and Rytwinski 2009).

Upon first consideration, the effects of the construction of linear infrastructure on bat species in PNG forests might seem trivial. Bats have the capacity to fly over narrow 'canyon-like' breaks in natural habitat, and the ability to fly also gives bats the potential to respond relatively quickly to changes in their habitat by changing where they forage. However, bats are known to be affected by road construction, sometimes in positive ways, but in many negative ways as well. Road construction creates open habitats, exposing bats to a greater level of real or perceived threat from 'predators' (including vehicles), reduces habitat connectivity, and can introduce high levels of artificial illumination, noise from traffic and wind intrusion into habitats (Kuijper et al. 2008; Schaub et al. 2008; Stone et al. 2009, 2012; Zurcher et al. 2010). Bat species that forage in dense vegetation within forest habitats and rely on passive listening for prey capture tend to be affected to a greater extent by roads, but even bats that forage in the open and are attracted by insect accumulations at lights have decreased levels of activity overall closer to roads (Blake et al. 1994; Kerth and Melber 2009; Berthinussen and Altringham 2011).

Bats as indicators of biodiversity value

Bats can be a good indicator group for the long-term monitoring of biodiversity values and habitat quality for a wide variety of environmental disturbance types (Jones et al. 2009). In the context of forest ecosystems, changes in the abundance (or commonness/rarity) of echolocating insectivorous bats may reflect changes in insect prey biomass. The structure of forest habitats also has considerable influence on bat diversity. Extinction risk is greatest for the many specialised bat species that forage within expanses of intact closed forest (Jones et al. 2003), because their flight morphology and echolocation signal type constrain them to this habitat. When forests are reduced in size, are transected by roads, or thinned, these forest interior specialists typically decline, and generalist species that forage in more open habitats become more common (Kingston 2013).

When surveying for bats by detecting their signature echolocation calls, not only is the efficiency of survey effort and the probability of species detection maximised (reviewed in Armstrong 2017), but the shape of call signals provides information on the diversity of bat ecological niches. This allows an appreciation of ecosystem complexity beyond the simplistic view given by a species list. When forest structure and cover changes, the availability of 'flight spaces' for bat species changes, and the relative proportion of species with certain wing shapes and echolocation signal types that allow them to exploit open, edge or closed flight spaces may also change.

Flight spaces are defined by how far the bats fly from vegetation. Because bat species use different echolocation signal types, they vary in their ability to distinguish acoustic echoes of prey items from those derived from background 'clutter' (typically vegetation) that they need to avoid while in flight (Denzinger and Schnitzler 2013). Echoes from clutter or other surfaces such as water bodies have increasing influence on the sensory perception of bats with decreasing distance to the background. There are three main flight space types, and usage of them can be inferred from the echolocation signal type:

- **Open**: uncluttered space, where clutter echoes are undetectable or clearly distinct from prey echoes. Such flight spaces include open clearings and air space well above the forest canopy or rivers. Used by bat species emitting relatively low frequency, high power, and narrowband calls with a 'characteristic frequency' below 30 kHz.

- **Edge**: background cluttered space, where prey echoes follow closely but do not overlap with clutter echoes. Such flight spaces include the edges of forest, large gaps within forest, open spaces between different vegetation layers (e.g., canopy, subcanopy or understorey), and open space immediately above water and the forest canopy. Used by bat species emitting 'chirp' calls or quasi-constant frequency calls with a 'characteristic frequency' between 30–70 kHz.

- **Clutter** ("narrow" in Denzinger and Schnitzler 2013): highly cluttered space, where prey echoes are intermingled with those from background clutter. Such flight spaces include dense understory or canopy vegetation, and low over the ground. Used by bat species in Australasia emitting low power, short duration, broadband calls; or short, medium, or long constant frequency calls anywhere between 30 to 170 kHz.

In the present study, the ROW has increased the availability of Open and Edge habitats to bat species having echolocation signals and wing morphologies that are suitable for foraging in these flight spaces. These species might therefore be at an advantage, but the effects on interior specialists are harder to predict given that the linear infrastructure is not associated with additional disturbances such as from heavy traffic at night, or artifical lighting. For interior specialists, the detection of an edge effect could be more relevant, and more so in the longer term if the forest edge is observed to change in structure (level of canopy cover, sub-canopy vegetation density).

Aims of the 2021 PMA3 bat study

This study addresses the primary question: "Is there an ongoing level of habitat change following linear infrastructure construction that is reflected in changes to bat communities?".

Specific aims of this fourth survey in the program were to:

1. Document the diversity of bats in the PMA3 study areas using the same recording sites as in 2015–2019;

2. Determine if bat communities have responded significantly to the construction of the ROW by assessing whether two specific measures of bat diversity, Species Richness and Phylogenetic Diversity, vary with increasing distance from the linear infrastructure corridor; and

3. Quantify bat diversity through several additional measures (Phylogenetic Diversity, Functional Diversity, Indicator Species analysis) that provide additional perspectives on the potential differences of bat communities at different distances from the ROW, elevations, and across biennial surveys.

Methods

Sampling design

This long-term monitoring study depends on the standardisation of sampling effort, equipment type and site placements. The same number of recordings were taken from the permanent transects established and used in the biennial surveys of 2015–2019. Field sampling was undertaken between 4 and 24 July, in approximately the same season as previous surveys but closest to the 2015 survey (June–July 2015; May 2017; August 2019).

Permanent transects are located within two narrow elevational ranges in each Biodiversity Assessment Area (BAA; Table 4.1): approximately 2,200 m asl and 2,700 m asl in BAA 1 on Hides Ridge in Hela Province; and approximately 1,000 m asl (Arakubi Quarry) and 1,400 m asl (KP 107) in BAA 2 on the Agogo Range near Moro in Southern Highlands Province.

Bat detectors were deployed at each of the 66 permanent acoustic recording sites along 11 transects in BAA 1 (transects H1–H6; total 36 recording nights over eight sampling nights, 4–12 July 2021) and BAA 2 (M1–M5; total 30 recording nights over eight sampling nights, 17–24 July 2021) (Table 4.1). A total of 66 full-night recordings was collected in 2021. Diligent monitoring of nightly data accumulation led to the identification of some minor equipment issues, and affected sites were resampled towards the end of each of the two sampling periods. Results from problematic recordings were not included.

The bat detectors were spaced along each transect at 50 m intervals, and given the high attenuation rate of ultrasonic calls, are assumed to be acoustically independent, so that an individual bat can only be detected by a single recorder at any given moment. The first detector on each transect was oriented to receive signals from the open area over the ROW (distance '0 m'). The remaining bat detectors (distances of 20–220 m) represented a potentially decreasing edge effect.

Recordings were made in high quality full spectrum format with Titley Scientific Anabat Swift bat detectors. In previous years, Pettersson Elektronik D500X bat detectors were used. Both bat detector types allow automated recordings, have similar detection range and the recording quality is equivalent (500 kHz sampling rate, 16 bit resolution). Microphones on a 1 m extension cable were placed in a funnel to keep out rain and set 2 m above the ground, and the detector body was covered with a waterproof bag (Figure 4.1).

The constraints and considerations relevant to the sampling design, acoustic surveys for bats and other aspects of the PMA3 monitoring program are discussed in Armstrong (2017).

Table 4.1. Summary of the experimental design and bat recording site placements. Factors include 'distance from the ROW' (6 treatment levels, total 66 replicate reciording positions) and 'elevation' (4 treatment levels, total 11 replicate transects). GPS coordinates are listed in Armstrong (2017).

Area	Elevation	Replicate	Distance from ROW (m)						Total
	(m asl)	(m asl)	0	20	70	120	170	220	
BAA 1	'2,700 m'	H4—2,700 m (2,681–2,696 m)	1	1	1	1	1	1	
		H5—2,750 m (2,726–2,756 m)	1	1	1	1	1	1	
		H6—2,730 m (2,725–2,736 m)	1	1	1	1	1	1	
	'2,200 m'	H1—2,150 m (2,148–2,163 m)	1	1	1	1	1	1	
		H2—2,200 m (2,171–2,229 m)	1	1	1	1	1	1	
		H3—2,300 m (2,296–2,327 m)	1	1	1	1	1	1	**36**
BAA 2	'1,400 m'	M1—1,400 m (1,397–1,405 m)	1	1	1	1	1	1	
	(KP107)	M2—1,380 m (1,315–1,397 m)	1	1	1	1	1	1	
		M3—1,380 m (1,369–1,389 m)	1	1	1	1	1	1	
	'1,000 m'	M4—1,030 m (995–1,041 m)	1	1	1	1	1	1	
	(Arakubi)	M5—1,050 m (1,051–1,073 m)	1	1	1	1	1	1	**30**

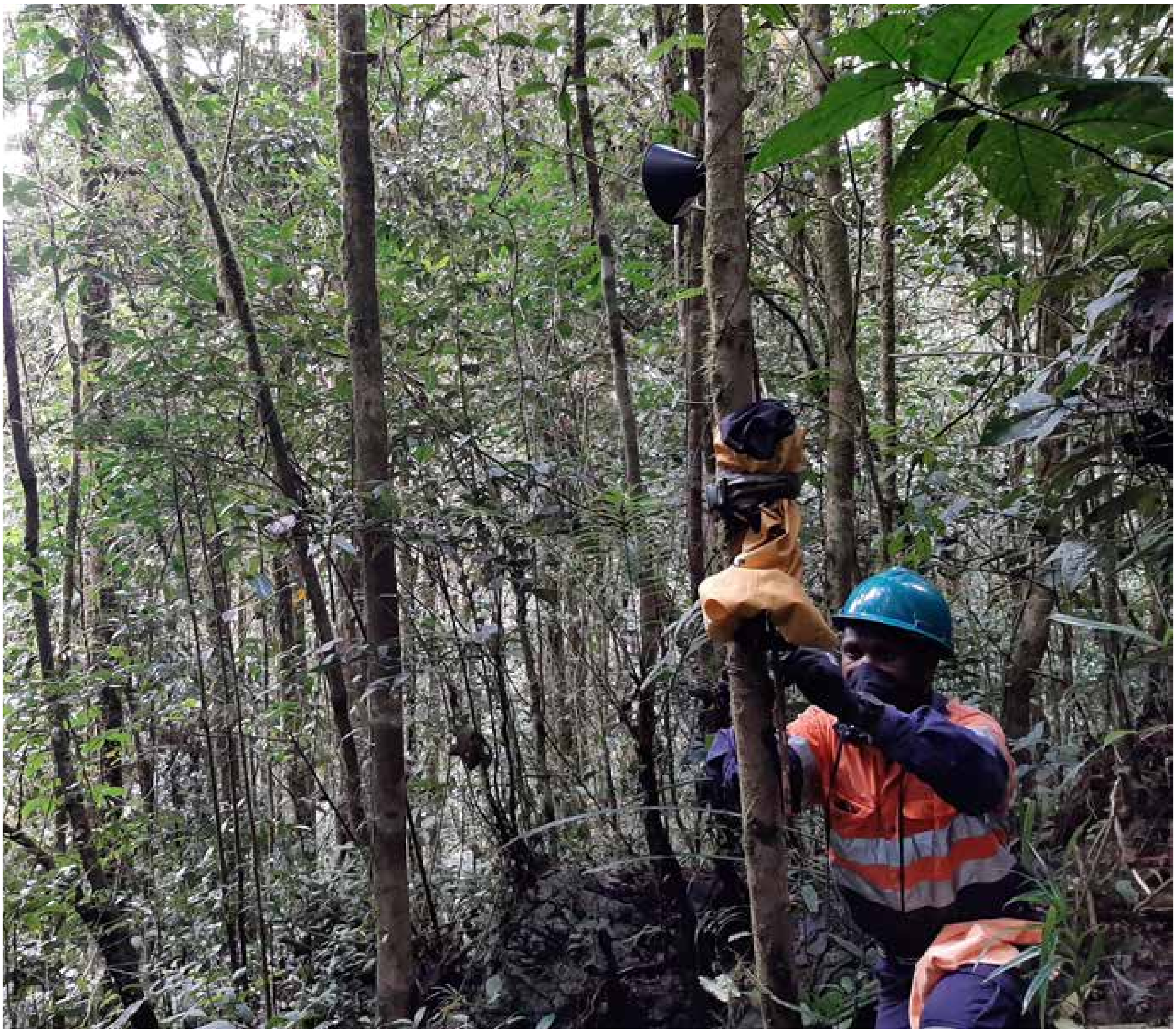


Figure 4.1. Deployment of the Anabat Swift bat detectors, with the microphone in a funnel.

Captures

During the 2015 and 2017 surveys effort was made to capture bats to help associate echolocation calls with species, with the assistance of DNA barcoding in some cases. It also confirmed the likelihood of species new to science (Armstrong et al. 2021a). No trapping was conducted in 2019 or 2021.

Processing of acoustic signals

A customised, multi-step acoustic processing procedure that can filter large bat echolocation recording datasets from Papua New Guinea (Armstrong et al. 2014, 2015a,b, 2021b,c) was applied to the recordings made on the survey. Processing first involved the recognition of bat echolocation 'call types', followed by a separate step of allocating a species identification to each of these types. The 'call types' are defined based on a standardised naming scheme that has been used in many published and unpublished surveys across Papua New Guinea and Wallacea in recent years (Armstrong and Aplin 2011, 2014; Armstrong et al. 2014, 2015a,b; Kale et al. 2018a,b,c; illustrated in Armstrong 2017). This two-step approach, along with the provision of illustrated examples of identified call types, provides a greater level of transparency that allows for future verification of call identifications, retrospective correction of species names on the basis of updated information or a better understanding of the variation produced by each species, and a comparison of diversity across sites and studies that is independent of taxonomic allocations and associated misidentifications (e.g., see Specialised Zoological 2017).

Data analysis

A comprehensive description of the data analysis is presented in Armstrong (2017) and a brief overview is provided here. Note that the term 'diversity' is used in this chapter in a general sense rather than as a specific measure. The diversity of bats encountered on the survey was summarised and compared among different distances from the ROW, elevations, and among survey years in terms of the number of species ('Species Richness'), as well as the breadth of their evolutionary relationships and ecological roles ('Phylogenetic Diversity', 'Functional Diversity', respectively). Species composition was also compared amongst independent factors, with Indicator Species' indices used to understand the basis for patterns. A brief explanation of each of these six specific measures is provided below.

1. Species Richness is the simplest measure of diversity and is a tally of the number of species at each recording site. A small proportion of echolocation call types recorded could have been derived from more than one species (e.g. some calls associated with medium- and small-sized *Miniopterus* could have also been derived from a species of *Pipistrellus* or *Nyctophilus*; *Kerivoula muscina* and *Murina florium* could not be distinguished reliably from each other), but in this study Species Richness as derived from the number of call types is assumed to be a reasonably accurate representation of the number of species. Species Richness was compared statistically amongst sites by fitting a Generalised Linear Mixed Model (GLMM) to a site-by-species matrix. Prior to analysis, a check was made to determine which distribution best fit data, and data were log transformed.

2. Phylogenetic Diversity (Faith 1992) is an overall measure of evolutionary diversity among the species present at a recording site, and considers both the number of species, as well as the degree of genetic distance among them. To better understand this measure, consider that sites with five species from five different families will have higher Phylogenetic Diversity, and thus higher value in terms of diversity, than sites with five species from the same family. The metric is calculated from a genetic distance matrix and phylogenetic tree that was created from mitochondrial DNA barcode sequences (cytochrome-*b*) generated for the 2015 study (Armstrong 2017). The genetic matrix and phylogenetic tree were updated to include additional species documented since the 2015 survey, some of them being represented by a DNA sequence from a closely related congener if unavailable, as sourced from Genbank. Phylogenetic Diversity (PD) was compared statistically amongst sites by fitting a Generalised Linear Mixed Model to a site-by-species matrix of PD values.

3. Relative Abundance was calculated to provide an indication of how common each species was, given that true abundance cannot be estimated from recordings of echolocation. This is simply the proportion of recording sites with detections of each species (e.g., a value of 0.6 indicates the species was detected at 6 out of 10 recording sites). Proportional representation for defined distances from the ROW and at each elevation was calculated using presence/absence data in the site-by-species matrix.

4. Functional Diversity (Petchey and Gaston 2002) is a measure of diversity that incorporates information on the range of 'functional types' (representing distinct ecological niches) present within bat communities. More complex ecosystems typically show both a greater range of functional types and a greater level of redundancy (more species with similar ecological roles). Functional Diversity is calculated from estimates of Relative Abundance as well as a categorisation of several aspects of the biology of each species (their 'ecological traits', such as wing shape type, echolocation signal shape, foraging habitat, prey capture strategy, flight space, roost type; summarised in Armstrong 2017).

5. Indicator Species indices (Dufrêne and Legendre 1997; De Cáceres and Legendre 2009) were calculated for each species at different distances from the ROW, and at different elevations from survey years combined. There are three related metrics calculated from the site-by-species matrix that highlights the association of certain species with particular habitats. Species found in many habitat types tend to have low scores. The measure allows comment on which species may be negatively affected by opening the forest canopy when building linear infrastructure, or that may take advantage of newly created open flight spaces and forest edges. The indicator value index (I) shows the value of a particular species as an 'Indicator Species' for that factor level (e.g., elevation level 1,000 m). It is the product of two components, called 'A' and 'B'. Component 'A' ('specificity') is the probability that the surveyed site belongs to the factor level given that the species has been detected. Component 'B' ('fidelity') is the probability of finding the species in sites belonging to a factor level.

6. Species composition is not a discrete metric, but recording sites can be compared in terms of the combinations of species detected. Differences among recording sites are most efficiently summarised in a two-dimensional ordination plot. This involves calculating Bray-Curtis Dissimilarity from the presence/absence data matrix, and then performing a Non-metric Multidimensional Scaling ordination. Discrete clustering of sites within factor levels would indicate patterns according to species composition amongst them.

All analyses were conducted using a custom-written [R] statistical computing language (R Core Team 2021) script, which takes in a standard site-by-species matrix and contains a record of all manipulations of the matrix and analytical steps. The script created for analysing the data from previous surveys was modified to allow for the incorporation of data from the fourth sampling year.

Results

Acoustic detections

A total of 21 echolocation call types was recognised from the recordings (Figure 4.2). Each call type represents one species in almost every case. A full list of species encountered to date over the four surveys shows that both the overall number and the species detected are relatively similar (Table 4.2).

From a simple inspection of the tabulated presence/absence data at each recording site in 2021 (Tables 4.3 and 4.4), it is clear that species composition differs between the two BAAs, mostly because Species Richness is substantially higher at the lower elevations in BAA 2, where 19 species were recorded (c.f. eight species in BAA 1). There was a complete absence of species in the families Emballonuridae and Hipposideridae, and only one species of Rhinolophidae in BAA 1.

By contrast the assemblages in BAA 2 had representation from all bat families documented in the study, but with fewer records of Vespertilionidae.

No previously undetected species were documented in the BAAs for the first time in 2021, but a growing understanding of the variation in call types of some species that emit low frequency echolocation calls helped to refine the species list. This led to the removal of the Greater Northern Free-tailed Bat *Chaerephon jobensis* from the overall list of species present, and the reallocation of some past records of *C. jobensis* and the Bare-rumped Sheath-tailed Bat *Saccolaimus saccolaimus* to the unidentified species of free-tailed bat *Otomops* sp. A single call sequence of *S. saccolaimus* is still retained as a record from the 2017 survey because it contains the correct harmonic profile for this species (Armstrong et al. 2021b). This refinement in the species list was possible because identifications are first made based on call type rather than species.

The name of the the unidentified bent-winged bat *Miniopterus* sp. 1 'large' was established as *Miniopterus tristis grandis* using genome-scale DNA barcoding (see Armstrong et al. 2021a; based on taxonomic work by Wiantoro 2020). Other taxonomic clarifications for woolly bats (*Kerivoula* spp.) and long-eared bats (*Nyctophilus* spp.) established in Armstrong et al. (2021a) have also been used to update the species list.

Most species were detected at relatively low overall rates. The most commonly recorded species were the Greater Melanesian Bent-winged Bat *Miniopterus tristis grandis* and the unidentified bent-winged bat *Miniopterus* sp. 3 'small'. These had relatively high and even detection rates across all distances of the transects and were the most common species at 2,200 m and 2,700 m elevation, respectively (Tables 4.4 and 4.5). The New Guinea Free-tailed Bat *Austronomus kuboriensis* was also relatively common at 2,700 m, which is notable given that at this elevation the bat assemblage is relatively depauperate and occurs at lower density (Table 4.5). This species is likely a high elevation specialist.

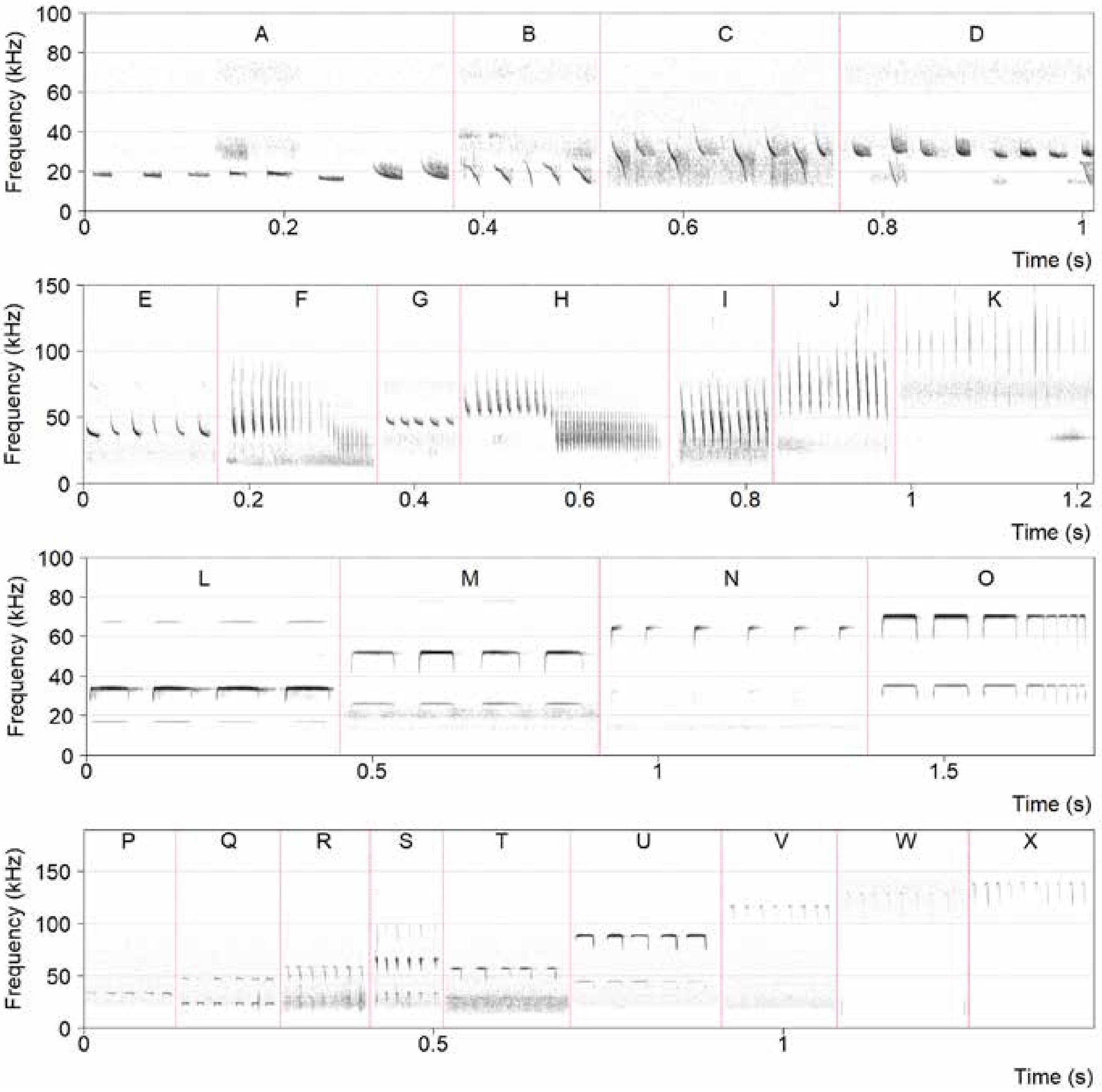


Figure 4.2. Representative sequence portions of the 21 call types recognised from the acoustic recordings in 2021, grouped by main body type of the call (time between pulses is compressed; scale of x and y axes vary).

A,**B**: *13 cFM Austronomus kuboriensis*;

C: *25 sFM Otomops* sp.;

D: *30 cFM Philetor brachypterus*;

E,**F**: *38 st.cFM Miniopterus tristis grandis*;

G: *45 st.cFM Miniopterus* sp. 2 'medium';

H: *53 st.cFM Miniopterus* sp. 3 'small';

I: *35 bFM Myotis moluccarum*;

J: *50 bFM Nyctophilus* sp.;

K: *80 bFM Kerivoula muscina / Kerivoula* sp. / *Murina florium*;

L: *33 lCF Rhinolophus* sp. cf. *robertsi*;

M: *52 lCF Rhinolophus euryotis*;

N: *65 lCF Rhinolophus megaphyllus*;

O: *70 lCF Rhinolophus mcintyrei*;

P: *35 i.fFM.d Emballonura dianae*;

Q: *45 i.fFM.d Emballonura raffrayana*;

R: *52 i.fFM.d Emballonura furax*;

S: *65 i.fFM.d Mosia nigrescens*;

T: *58 mCF Hipposideros diadema*;

U: *88 mCF Hipposideros wollastoni*;

V: *120 sCF Aselliscus tricuspidatus*;

W: *125 sCF Hipposideros maggietaylorae*;

X: *140 sCF Hipposideros cervinus*.

Table 4.2. Summary of species detection over the four survey periods (C: captured; E: echolocation calls recorded).

Scientific name	**Call type**	**Main call type**	**Flight space**	**2015**	**2017**	**2019**	**2021**
PTEROPODIDAE—2	—	—	—				
Syctonycteris australis	—	—	—	C	C	—	—
Syctonycteris sp. cf. *australis*	—	—	—	C	C	—	—
EMBALLONURIDAE—5							
Emballonura dianae	*35 i.fFM.d*	*fFM*	Edge	E	E	E	E
Emballonura furax	*52 i.fFM.d*	*fFM*	Edge	E			E
Emballonura raffrayana	*45 i.fFM.d*	*fFM*	Edge	E	E	E	E
Mosia nigrescens	*65 i.fFM.d*	*fFM*	Edge	E	E	E	E
Saccolaimus saccolaimus	*25 cFM*	*sFM*	Open		E		
HIPPOSIDERIDAE—6							
Aselliscus tricuspidatus	*120 sCF*	*sCF*	Clutter	CE	CE	E	E
Hipposideros cervinus	*140 sCF*	*sCF*	Clutter	CE	CE		E
Hipposideros diadema	*58 mCF*	*mCF*	Edge	E	E	E	E
Hipposideros maggietaylorae	*125 sCF*	*sCF*	Clutter			E	E
Hipposideros wollastoni	*88 mCF*	*mCF*	Clutter	E	E	E	E
Hipposideros sp. cf. *ater*	*172 sCF*	*sCF*	Clutter	E			
RHINOLOPHIDAE—4							
Rhinolophus euryotis	*52 lCF*	*lCF*	Clutter	E	CE	E	E
Rhinolophus mcintyrei	*70 lCF*	*lCF*	Clutter	E	CE	E	E
Rhinolophus megaphyllus	*65 lCF*	*lCF*	Clutter	CE	E	E	E
Rhinolophus sp. cf. *robertsi*	*33 lCF*	*lCF*	Clutter	E	E	E	E
MINIOPTERIDAE—3							
Miniopterus tristis grandis	*38 st.cFM*	*cFM*	Edge	CE	CE	E	E
Miniopterus sp. 2 'medium'	*45 st.cFM*	*cFM*	Edge	E	E	E	E
Miniopterus sp. 3 'small'	*53 st.cFM*	*cFM*	Edge	E	CE	E	E
VESPERTILIONIDAE—5							
Kerivoula sp.	*80 bFM*	*bFM*	Clutter		C		
Kerivoula muscina / *Kerivoula* sp. / *Murina florium*	*80 bFM*	*bFM*	Clutter	E	E	E	E
Nyctophilus microdon	*30 bFM*	*bFM*	Clutter		C		
Nyctophilus sp.	*50 bFM*	*bFM*	Clutter	E	CE	E	E
Philetor brachypterus	*30 cFM*	*cFM*	Edge	E	E	E	E
MOLOSSIDAE—2							
Austronomus kuboriensis	*13 cFM*	*cFM*	Open			E	E
Otomops sp.	*25 sFM*	*sFM*	Open		E	E	E
Total 27 species				**21**	**23**	**19**	**21**

Table 4.3. Summary of species/call type detections at each sampling position in BAA 1 on Hides Ridge in 2021. The sequence of circles is increasing distance from the road (0, 20, 70, 120, 170 and 220 m, left to right), with a filled black circle indicating a detection of that species, an open circle an apparent absence.

Scientific name	**Elevation**	**2,200 m**			**2,700 m**		
	Transect	**H1**	**H2**	**H3**	**H4**	**H5**	**H6**
	Call type						
EMBALLONURIDAE	**Sheath-tailed bats**						
Emballonura dianae	*35 i.fFM.d*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Emballonura furax	*52 i.fFM.d*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Emballonura raffrayana	*45 i.fFM.d*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Mosia nigrescens	*65 i.fFM.d*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Saccolaimus saccolaimus	*25 cFM*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
HIPPOSIDERIDAE	**Leaf-nosed bats**						
Aselliscus tricuspidatus	*120 sCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Hipposideros cervinus	*140 sCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Hipposideros diadema	*58 mCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Hipposideros maggietaylorae	*125 sCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Hipposideros wollastoni	*88 mCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Hipposideros sp. cf. *ater*	*172 sCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
RHINOLOPHIDAE	**Horseshoe bats**						
Rhinolophus euryotis	*52 lCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Rhinolophus mcintyrei	*70 lCF*	ooo●●o	●ooooo	oooooo	oooooo	oooooo	oooooo
Rhinolophus megaphyllus	*65 lCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Rhinolophus sp. cf. *robertsi*	*33 lCF*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
MINIOPTERIDAE	**Bent-winged bats**						
Miniopterus tristis grandis	*38 st.cFM*	●●●o●●	●●o●●●	●●●●●●	●●oooo	●oooo●	●●oooo
Miniopterus sp. 2 'medium'	*45 st.cFM*	oooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Miniopterus sp. 3 'small'	*53 st.cFM*	●o●ooo	oooooo	oooooo	●●●●●●	●●●o●●	●●●●●●
VESPERTILIONIDAE	**Vesper bats**						
Kerivoula muscina / *Kerivoula* sp. */ Murina florium*	*80 bFM*	oooooo	oooooo	ooooo●	oooooo	●ooooo	oo●ooo
Nyctophilus sp.	*50 bFM*	oooooo	oo●ooo	oooooo	oooooo	ooo●oo	oooooo
Philetor brachypterus	*30 cFM*	oooooo	oooooo	●ooooo	oooooo	oooooo	o●oooo
MOLOSSIDAE	**Free-tailed bats**						
Austronomus kuboriensis	*13 cFM*	oooooo	●ooooo	oooooo	oooooo	oooo●o	●●●●oo
Otomops sp.	*25 sFM*	●ooooo	oooooo	oooooo	oooooo	oooooo	oooooo
Total Species Richness		**4**	**4**	**3**	**2**	**5**	**5**

Table 4.4. Summary of species/call type detections at each sampling position in BAA 2 on the Agogo Range near Moro in 2021. Symbols as for Table 4.3.

	Elevation	**1,000 m**		**1,400 m**		
	Transect	**M4**	**M5**	**M1**	**M2**	**M3**
Scientific name	**Call type**					
EMBALLONURIDAE	**Sheath-tailed bats**					
Emballonura dianae	*35 i.fFM.d*	ooooo●	oooooo	oooooo	oooooo	oooooo
Emballonura furax	*52 i.fFM.d*	oooooo	●ooooo	oooooo	oooooo	oooooo
Emballonura raffrayana	*45 i.fFM.d*	●ooooo	●ooooo	oooooo	●●oooo	●ooooo
Mosia nigrescens	*65 i.fFM.d*	●●●●o●	●●oo●●	oooooo	oooooo	oooooo
Saccolaimus saccolaimus	*25 cFM*	oooooo	oooooo	oooooo	oooooo	oooooo
HIPPOSIDERIDAE	**Leaf-nosed bats**					
Aselliscus tricuspidatus	*120 sCF*	●●●o●o	oo●oo●	oo●●oo	ooo●oo	oooooo
Hipposideros cervinus	*140 sCF*	o●oooo	●o●●oo	o●o●oo	oo●o●o	oooooo
Hipposideros diadema	*58 mCF*	oooooo	oooo●o	oooooo	oooooo	oooooo
Hipposideros maggietaylorae	*125 sCF*	oooooo	ooooo●	oooooo	oooooo	oooooo
Hipposideros wollastoni	*88 mCF*	●●●●oo	●●o●●●	ooo●oo	o●●o●●	ooooo●
Hipposideros sp. cf. *ater*	*172 sCF*	oooooo	oooooo	oooooo	oooooo	oooooo
RHINOLOPHIDAE	**Horseshoe bats**					
Rhinolophus euryotis	*52 lCF*	●●●●●●	o●●●●●	ooo●oo	ooooo●	oooooo
Rhinolophus mcintyrei	*70 lCF*	oo●o●o	●oo●●●	oooooo	oooooo	oooo●o
Rhinolophus megaphyllus	*65 lCF*	●oo●●o	●●oooo	oooooo	oooooo	●ooooo
Rhinolophus sp. cf. *robertsi*	*33 lCF*	●●●o●●	oooo●o	oooooo	●●●oo●	●ooo●o
MINIOPTERIDAE	**Bent-winged bats**					
Miniopterus tristis grandis	*38 st.cFM*	●ooooo	●●oo●●	●●oooo	●●●oo●	●ooooo
Miniopterus sp. 2 'medium'	*45 st.cFM*	oooooo	oooooo	oooooo	oooooo	●ooooo
Miniopterus sp. 3 'small'	*53 st.cFM*	●o●●oo	●●●●●●	o●●●●●	●●●oo●	●●●ooo
VESPERTILIONIDAE	**Vesper bats**					
Kerivoula muscina / *Kerivoula* sp. / *Murina florium*	80 bFM	oooo●o	oooooo	oooooo	oooooo	oooooo
Nyctophilus sp.	50 bFM	oooooo	oooooo	oooooo	oooooo	oooooo
Philetor brachypterus	30 cFM	oooooo	oooooo	oooooo	oooooo	oooooo
MOLOSSIDAE	**Free-tailed bats**					
Austronomus kuboriensis	*13 cFM*	●ooooo	oooooo	oooooo	oooooo	●ooooo
Otomops sp.	*25 sFM*	●ooooo	●ooo●o	oooooo	●ooooo	●ooooo
Total Species Richness		**15**	**15**	**6**	**9**	**10**

Table 4.5. Relative Abundance of each bat species at recording sites for each distance (m) from the ROW along all transects combined, and over all survey years (each column sums to 1).

Species	Call type	0	20	70	120	170	220
EMBALLONURIDAE							
Emballonura dianae	*35 i.fFM.d*	0.04	0.01	0	0	0	0.01
Emballonura furax	*52 i.fFM.d*	0.02	0	0	0	0	0
Emballonura raffrayana	*45 i.fFM.d*	0.09	0.03	0.01	0.01	0	0.01
Mosia nigrescens	*65 i.fFM.d*	0.05	0.07	0.06	0.04	0.03	0.07
Saccolaimus saccolaimus	*25 cFM*	0	0.01	0	0	0	0
HIPPOSIDERIDAE							
Aselliscus tricuspidatus	*120 sCF*	0.02	0.07	0.10	0.05	0.07	0.05
Hipposideros cervinus	*140 sCF*	0.02	0.03	0.04	0.04	0.04	0.03
Hipposideros diadema	*58 mCF*	0.01	0	0.01	0	0.01	0.01
Hipposideros maggietaylorae	*125 sCF*	0	0	0	0	0.01	0.01
Hipposideros wollastoni	*88 mCF*	0.04	0.08	0.09	0.13	0.09	0.10
Hipposideros sp. cf. *ater*	*172 sCF*	0	0.01	0	0	0	0
RHINOLOPHIDAE							
Rhinolophus euryotis	*52 lCF*	0.02	0.08	0.08	0.20	0.12	0.09
Rhinolophus mcintyrei	*70 lCF*	0.04	0.06	0.06	0.08	0.09	0.02
Rhinolophus megaphyllus	*65 lCF*	0.03	0.04	0.02	0.03	0.01	0
Rhinolophus sp. cf. *robertsi*	*33 lCF*	0.04	0.03	0.04	0.03	0.04	0.05
MINIOPTERIDAE							
Miniopterus tristis grandis	*38 st.cFM*	0.23	0.20	0.10	0.09	0.15	0.21
Miniopterus sp. 2 'medium'	*45 st.cFM*	0.04	0	0	0	0	0
Miniopterus sp. 3 'small'	*53 st.cFM*	0.19	0.18	0.27	0.20	0.24	0.29
VESPERTILIONIDAE							
Kerivoula muscina / *Kerivoula* sp. / *Murina florium*	*80 bFM*	0.02	0.02	0.04	0.07	0.01	0.03
Nyctophilus sp.	*50 bFM*	0.01	0.03	0.05	0.01	0.04	0.01
Philetor brachypterus	*30 cFM*	0.03	0.02	0	0	0	0
MOLOSSIDAE							
Austronomus kuboriensis	*13 cFM*	0.02	0.02	0.02	0.01	0.01	0
Otomops sp.	*25 sFM*	0.06	0.01	0	0	0.01	0

Table 4.6. Relative Abundance of each bat species for each elevation (m) along all transects combined, and over all survey years (each column sums to 1).

Species	**Call type**	**1,000**	**1,400**	**2,200**	**2,700**
EMBALLONURIDAE					
Emballonura dianae	*35 i.fFM.d*	0.02	0.02	0	0
Emballonura furax	*52 i.fFM.d*	0.01	0.01	0	0
Emballonura raffrayana	*45 i.fFM.d*	0.04	0.08	0	0
Mosia nigrescens	*65 i.fFM.d*	0.12	0.01	0	0.01
Saccolaimus saccolaimus	*25 cFM*	0.00	0	0	0
HIPPOSIDERIDAE					
Asellíscus tricuspidatus	*120 sCF*	0.09	0.07	0	0
Hipposideros cervinus	*140 sCF*	0.04	0.06	0	0
Hipposideros diadema	*58 mCF*	0.01	0.01	0	0
Hipposideros maggietaylorae	*125 sCF*	0.01	0	0	0
Hipposideros wollastoni	*88 mCF*	0.12	0.11	0	0
Hipposideros sp. cf. *ater*	*172 sCF*	0.00	0	0	0
RHINOLOPHIDAE					
Rhinolophus euryotis	*52 lCF*	0.12	0.12	0	0
Rhinolophus mcintyrei	*70 lCF*	0.07	0.01	0.16	0
Rhinolophus megaphyllus	*65 lCF*	0.05	0.01	0	0
Rhinolophus sp. cf. *robertsi*	*33 lCF*	0.05	0.06	0	0
MINIOPTERIDAE					
Miniopterus tristis grandis	*38 st.cFM*	0.06	0.14	0.50	0.20
Miniopterus sp. 2 'medium'	*45 st.cFM*	0.01	0.03	0	0
Miniopterus sp. 3 'small'	*53 st.cFM*	0.10	0.21	0.18	0.58
VESPERTILIONIDAE					
Kerivoula muscina / Kerivoula sp. */ Murina florium*	*80 bFM*	0.02	0.02	0.03	0.06
Nyctophilus sp.	*50 bFM*	0	0	0.08	0.07
Philetor brachypterus	*30 cFM*	0.01	0.01	0.04	0.01
MOLOSSIDAE					
Austronomus kuboriensis	*13 cFM*	0.00	0.01	0.01	0.07
Otomops sp.	*25 sFM*	0.03	0.03	0.01	0

Species Richness

Species Richness was compared among different distances from the ROW, different elevations, and among the four survey years. A significant difference was observed among all treatment levels for all three factors; and there were no significant interaction terms amongst the three factors (Table 4.7). Pairwise comparisons showed that the significant difference amongst years was due to the slightly lower tally of species in 2019 (Table 4.2). However, the Species Richness in 2021 was the same as in the original 2015 survey, so clearly the statistical difference is derived from natural variation, and possibly because of the limited number of site replicates. Thus, there is no strong or clear indication of a change in the use of the forest edge habitats by bats in the period 2015–2021.

Pairwise comparisons revealed that Species Richness was greatest at the beginning of transects (the open areas at the edge of the forest, and to a lesser degree the first 20 metres inside; Table 4.7; Figure 4.3). Species Richness was both lower inside the forest canopy and similar along the remainder of thetransects (20–220 m). Species Richness was also significantly greater at the lowest elevation at Arakubi Quarry (1,000 m asl). Of surprise though, transects at KP 107 (1,400 m asl) had statistically equivalent mean Species Richness as those in BAA 1, which is likely to reflect the level of variation over survey years and amongst transects at this elevation.

Decomposing the overall plots by both distance and elevation reveals that the number of species detected at forest edges was higher only at the lowest elevation (Figure 4.4). On Hides Ridge, the number of species at different points along the transects was relatively similar. The variation amongst survey years is particularly evident in these plots, and this may be masking or eliminatings some underlying patterns.

Given that the start of transects had significantly greater Species Richness in BAA 2, it was relevant to examine whether there was a similar pattern of difference in Species Richness for just those bat species classified as using 'Edge' habitats (foraging in open areas but close to vegetation boundaries where the microphones at '0 m' were positioned; see flight space designations for each species in Table 4.2). This was indeed the case, with a greater number of Edge species detected at a distance of '0 m', as well as a greater prevalence of these species at lower elevation sites. The statistical outcomes for Edge species are almost identical to those for overall Species Richness (Table 4.7). An inspection of the means for Edge species shows them to be lower compared to overall Species Richness simply because the latter includes bat species favouring the Clutter flight space of the forest interior (Table 4.8).

When the proportion of Edge species relative to that of Open and Clutter species was plotted for increasing distance from the ROW and elevation, it was clear that there was greater relative representation of Edge species at the forest edge (Figure 4.5). However, the pattern varied over the length of the transect, with Edge species representation being almost as high at the end of the transect furthest into the forest. There was also a clear pattern of increasing Edge species representation with increasing elevation, which is evident when the two BAAs are compared (Figure 4.5).

To explore the patterns in Species Richness even further to gain a better appreciation of what elements of the bat assemblage were contributing most to the results, it was also relevant to examine whether one of the main echolocation call types (*cFM*) that is associated with edge habitats varied with the three factors in the same way as total Species Richness. This was indeed the case (Table 4.7), with the greatest number of *cFM* bat species recorded at distance '0 m' (Table 4.8), but there was no pattern for elevation.

Table 4.7. Summary of the tests of the Generalised Linear Mixed Model and Tukey's post hoc pairwise comparisons to test for the influence on bat diversity (dependent variable 'Species Richness') of the factors 'Distance' from the ROW, 'Elevation', and survey 'Year' (values from the Analysis of Deviance table; Type II Wald F tests with Kenward-Roger df; only significant pairwise tests are shown; Significance codes: '*' <0.05, '**' <0.01, '***' <0.001; the model fit used was: lmer(total_richness ~ dist + elev + year + dist*elev + elev*year + dist*year + dist*elev*year + (1|transect) + (1|transect.dist) + (1|transect.year), data = y). Total Species Richness was replaced with the number of Edge species, *cFM* species and Phylogenetic Diversity in the successive analyses.

Species Richness	**F**	**df**	**P**	**Pairwise**
Distance	13.7	5	<0.001***	0 > 20** 0 > 70*** 0 > 120*** 0 > 170*** 0 > 220*** 20 > 120*
Elevation	34.1	3	<0.001***	1,000 > 1,400*** 1,000 > 2,200*** 1,000 > 2,700***
Year	8.2	3	<0.001***	2019 < 2017** 2019 < 2021***
Distance*Elevation	3.9	15	<0.001***	—
Distance*Year	0.8	15	0.66 NS	—
Elevation*Year	2.4	9	0.04*	—
Distance*Elevation*Year	1.3	45	0.15 NS	—
Species Richness—Edge species	**F**	**df**	**P**	**Pairwise**
Distance	36.8	5	<0.001***	0 > 20*** 0 > 70*** 0 > 120*** 0 > 170*** 0 > 220*** 20 > 120** 20 > 170*
Elevation	12.6	3	0.003**	1,000 > 1,400** 1,000 > 2,200** 1,000 > 2,700**
Year	5.3	3	0.007**	2019 < 2017** 2019 < 2021*
Species Richness—*cFM* species	**F**	**df**	**P**	**Pairwise**
Distance	22.1	5	<0.001***	0 > 20*** 0 > 70*** 0 > 120*** 0 > 170*** 0 > 220*** 20 > 120** 20 > 170*
Elevation	2.7	3	0.43 NS	—
Year	0.87	3	0.83 NS	—
Phylogenetic Diversity	**F**	**df**	**P**	**Pairwise**
Distance	20.0	5	0.001**	0 > 120** 0 > 170**
Elevation	25.3	3	<0.001***	1,000 > 1,400* 1,000 > 2,200*** 1,000 > 2,700***
Year	3.0	3	0.39 NS	—
Distance*Elevation	69.4	15	0.049*	—
Distance*Year	1.2	15	0.24 NS	—
Elevation*Year	2.6	9	0.33 NS	—
Distance*Elevation*Year	103.6	42	0.39 NS	—

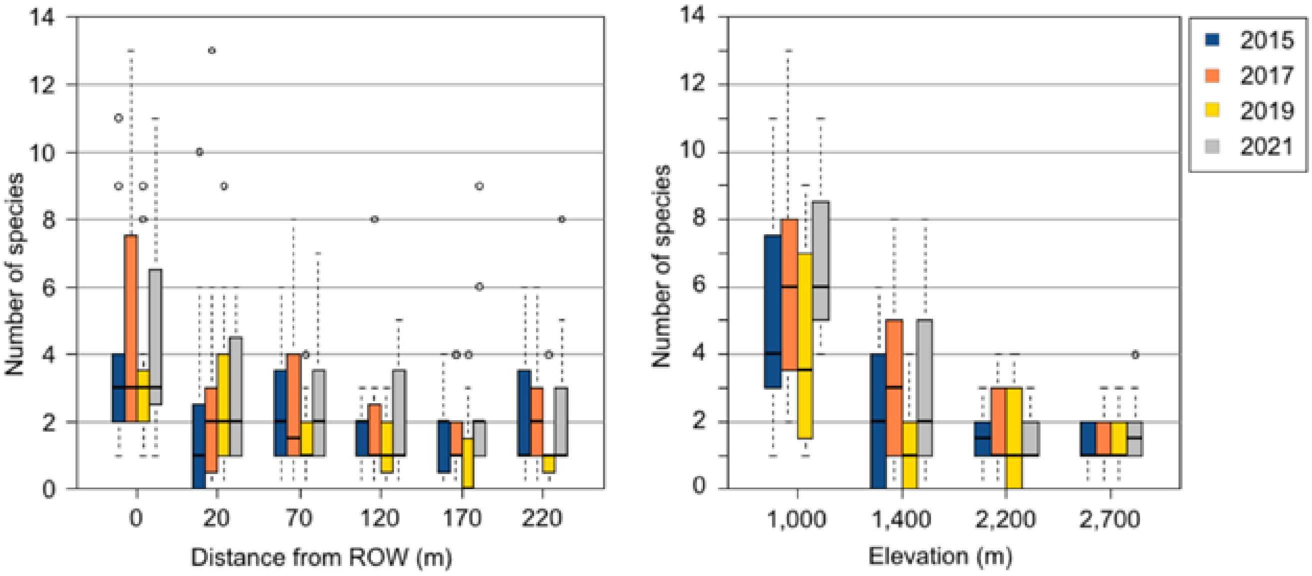


Figure 4.3. Summary of the patterns of Species Richness with increasing distance from the ROW, and with increasing elevation. All sites have been combined for each of these two factors, but segregated by year. [Boxplot components: central bar—median; boxes—inter-quartile range, with second quartile group below median, third quartile group above median; bars—minimum and maximum values; circles—statistical outliers].

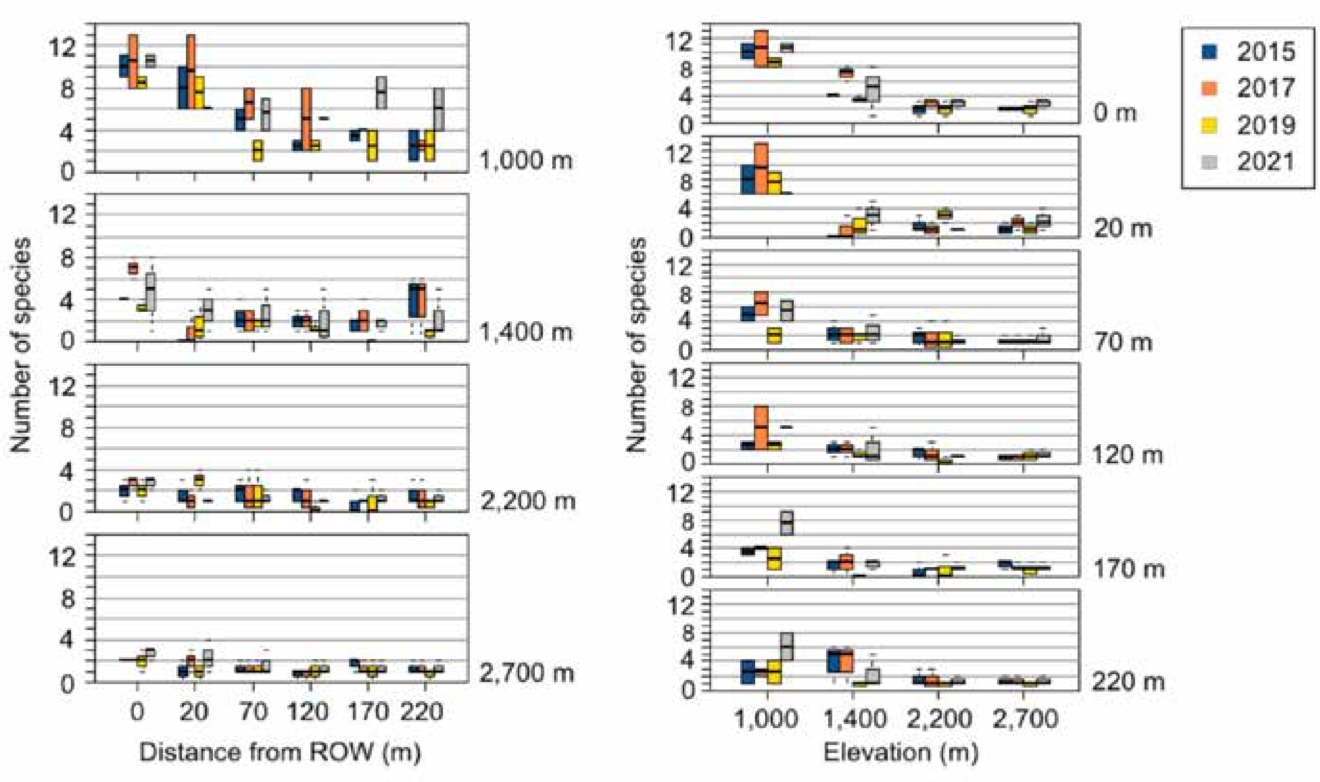


Figure 4.4. Summary of the patterns of Species Richness with increasing distance from the ROW at each elevation, and with different elevation levels for each distance from the ROW.

Table 4.8. Summary of means ± standard deviation for various dependent variables (total Species Richness, Species Richness of all Edge species and Species Richness of all species with a cFM call type (see Table 4.2 for allocation of each species to these categories) at each distance from the ROW, elevation and survey year, plus the metrics of Phylogenetic Diversity and Functional Diversity. Values in bold are significantly and consistently higher than the others based on pairwise comparisons (see Table 4.7).

Distance (m)	Species Richness	Edge species	*cFM* species	Phylogenetic Diversity	Functional Diversity
0	**4.3 ± 3.2**	**3.0 ± 1.7**	**2.5 ± 1.0**	**0.46 ± 0.31**	1.31
20	2.7 ± 2.9	1.4 ± 1.3	1.2 ± 1.0	0.34 ± 0.32	1.03
70	2.3 ± 1.8	1.0 ± 0.7	0.9 ± 0.6	0.34 ± 0.19	1.17
120	1.7 ± 1.6	0.6 ± 0.6	0.5 ± 0.6	0.22 ± 0.18	0.06
170	1.7 ± 1.8	0.7 ± 0.8	0.7 ± 0.7	0.20 ± 0.19	0.39
220	2.0 ± 1.9	1.2 ± 0.9	1.0 ± 0.7	0.30 ± 0.23	0.89
Elevation (m)					
1,000	**5.7 ± 3.2**	**2.2 ± 2.0**	1.2 ± 1.3	**0.60 ± 0.28**	1.44
1,400	2.3 ± 2.1	1.2 ± 1.5	1.0 ± 1.2	0.28 ± 0.23	1.05
2,200	1.5 ± 1.1	1.0 ± 0.7	1.1 ± 0.8	0.21 ± 0.17	1.19
2,700	1.4 ± 0.8	1.2 ± 0.7	1.2 ± 0.7	0.19 ± 0.13	0.91
Year					
2015	2.3 ± 2.3	1.3 ± 1.3	0.9 ± 0.9	0.29 ± 0.24	—
2017	2.8 ± 2.9	1.6 ± 1.6	1.2 ± 1.1	0.34 ± 0.31	—
2019	**1.9 ± 2.0**	**1.0 ± 1.2**	0.9 ± 1.0	0.25 ± 0.24	—
2021	2.8 ± 2.5	1.4 ± 1.1	1.4 ± 1.0	0.42 ± 0.24	—

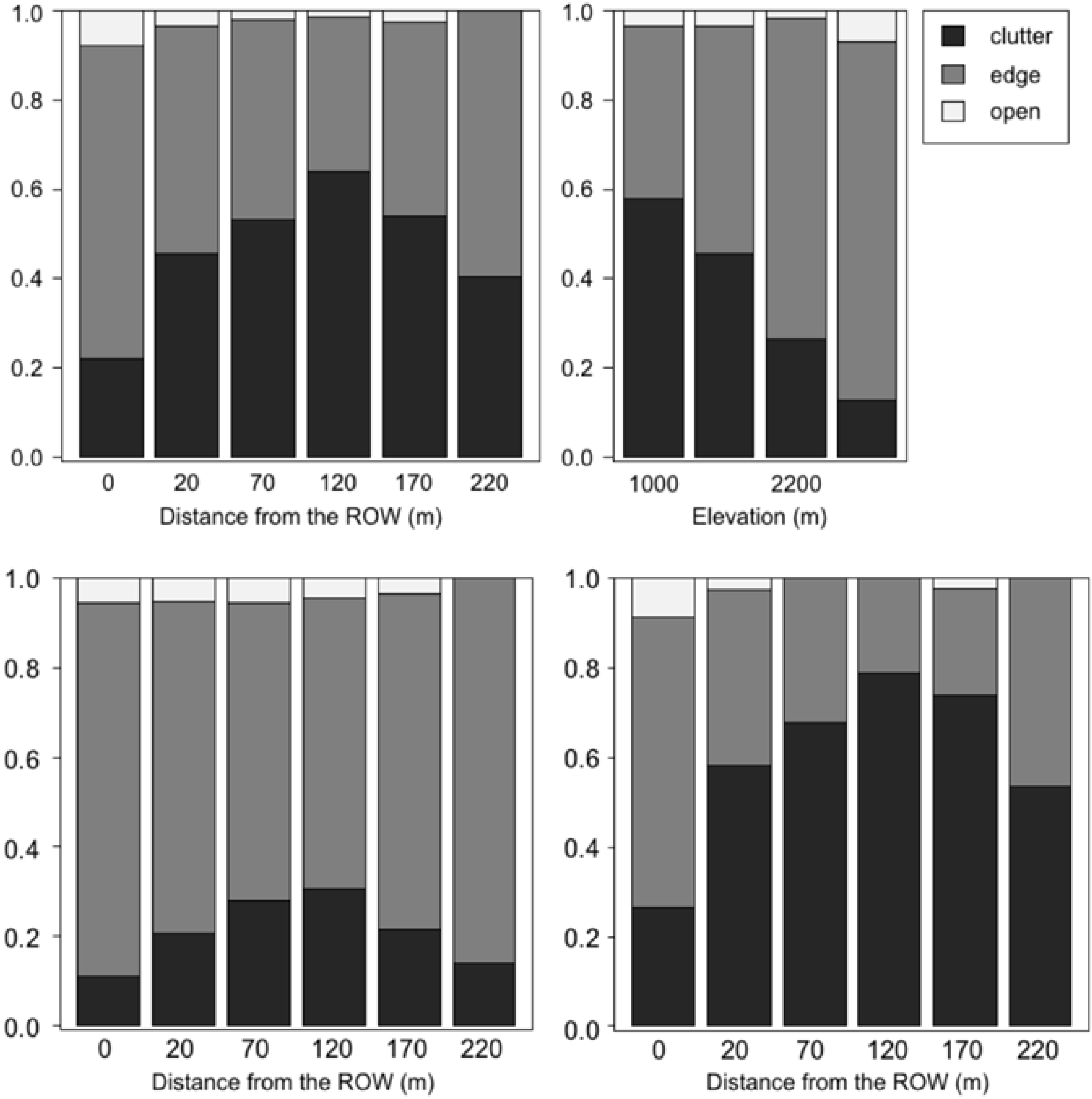


Figure 4.5. Summary plots of the proportion of bats occupying three different flight spaces at increasing distance from the ROW, and at increasing elevation for all sites combined (top); and with increasing distance from the ROW in BAA 1 (bottom left) and BAA 2 (bottom right).

Phylogenetic Diversity

Additional comparisons were undertaken to compare Phylogenetic Diversity among different distances from the ROW, different elevations, and across the four survey years to understand if there were any biases in terms of bat genera or families that were associated with the factor levels. There was a similar, but weaker overall pattern compared to Species Richness (Tables 4.7 and 4.8). Phylogenetic Diversity was slightly greater in the open areas at the start of transects and at the lowest elevation of 1,000 m (Figure 4.6). The latter is particular evident from an inspection of the raw results in Tables 4.3 and 4.4, which show the almost complete absence of species in the Emballonuridae, Hipposideridae and Rhinolophidae in BAA 1. There has been no significant change in Phylogenetic Diversity across survey years (Table 4.8).

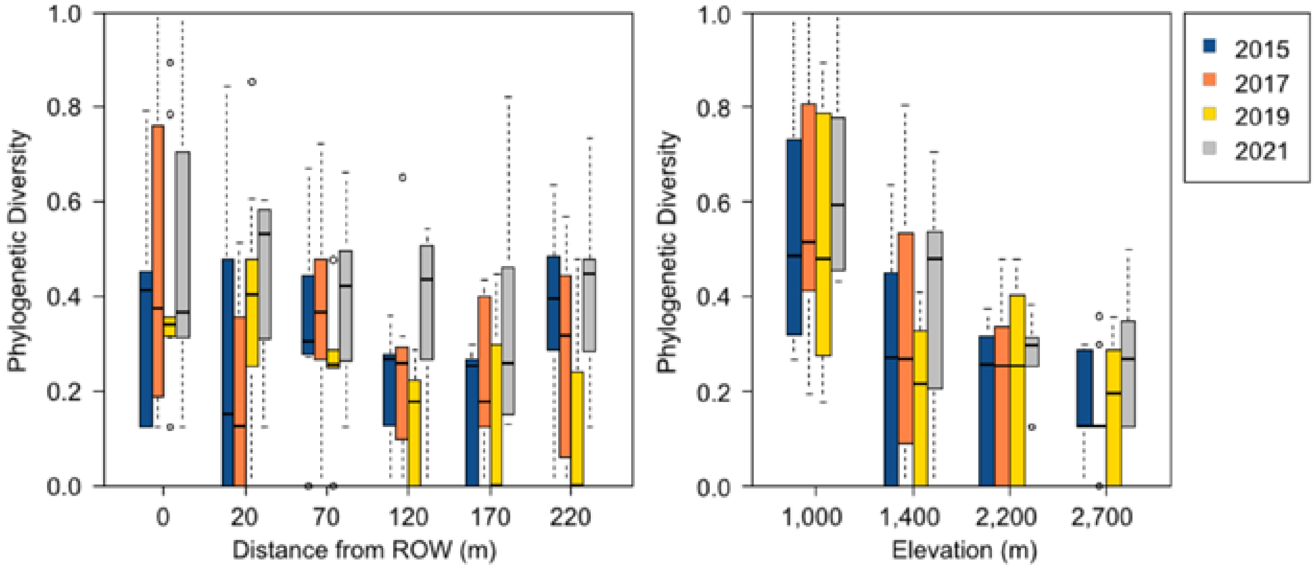


Figure 4.6. Summary plots of the pattern of Phylogenetic Diversity at increasing distance from the ROW and at increasing elevation.

Functional Diversity

Values of Petchey and Gaston's (2002) Functional Diversity, or the diversity of bat ecological niches, showed a similar trend across most survey years, with some variation in 2021 at distance 170 m and at 2,200 m elevation (Figure 4.7). Such variation in this and other metrics is expected given that sampling is based on relatively few replicate sites and transects at each distance and elevation. However, overall, these patterns are consistent with the patterns from Species Richness and Phylogenetic Diversity, illustrating that bat diversity is greatest at lower elevations, and that more species can be found exploiting resources at the forest edge.

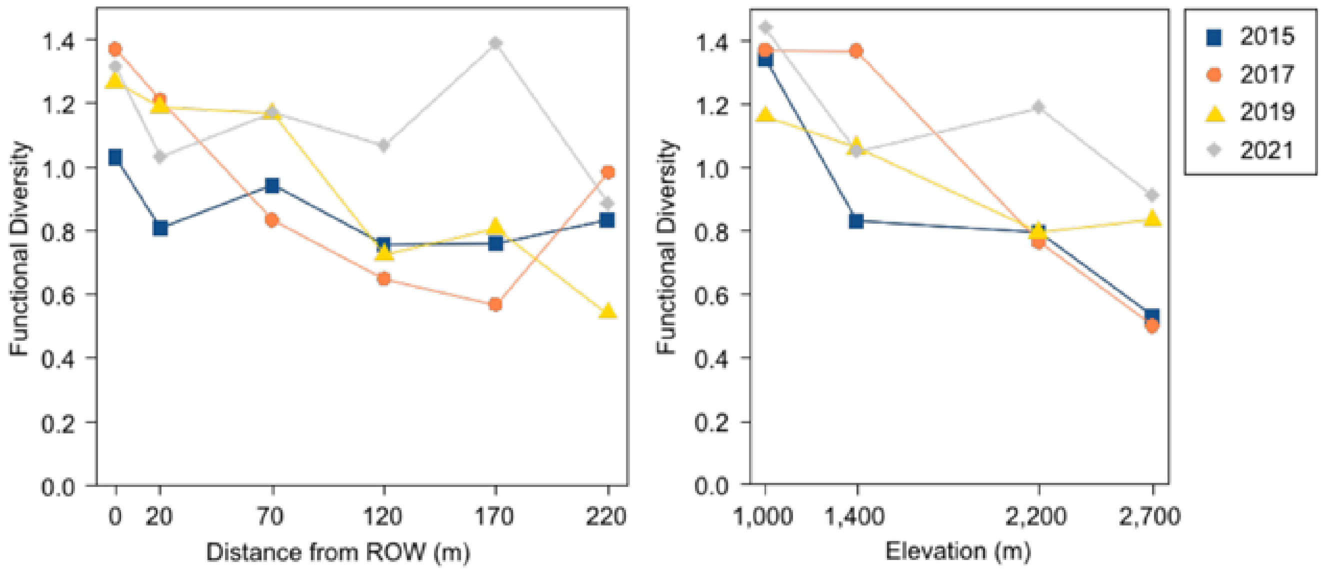


Figure 4.7. Summary plots of Functional Diversity at increasing distance from the ROW, and at increasing elevation.

Species composition

Analyses thus far have shown clearly that Species Richness is greatest at the start of transects at lower elevations in BAA 2. This greater level of diversity is attributable to species that can exploit forest edges and open habitats (Edge species; to a lesser extent those emitting the *cFM* call type). While it is obvious that smaller values of Species Richness will lead to an altered species composition at sites because some species will be missing, it is also relevant to explore whether the species at higher elevations and within the forest interior are subsets of the larger assemblages, or whether there is species replacement.

Non-metric Multidimensional Scaling (NMDS) ordination plots showed that there was no obvious discrete clustering according to distance from the ROW, except for sites at 0 m that showed a tendency to have a smaller level of variation (red dots in Figure 4.8). They had overall similarity to a low proportion of sites at other distance values, but there was less variation amongst replicate transects, which reflects a consistency in the list of species using these more open spaces. Some of those species are identified using the Indicator Species metrics (see next section *Species-level patterns*).

The higher elevations differed slightly in species representation (Figure 4.8), with the two higher elevation communities 2,200 m and 2,700 m being most different from each other, and also from the lower elevation communities at 1,000 m and 1,400 m.

The overall pattern is suggestive of both a subsetting of the broader assemblage at higher elevations (low rates of encounter or the complete absence of families Emballonuridae, Hipposideridae, Rhinolophidae in BAA 1), and some species replacement (more Vespertionidae in BAA 1 and swapping of species-level representation in the Molossidae; see Tables 4.3 and 4.4).

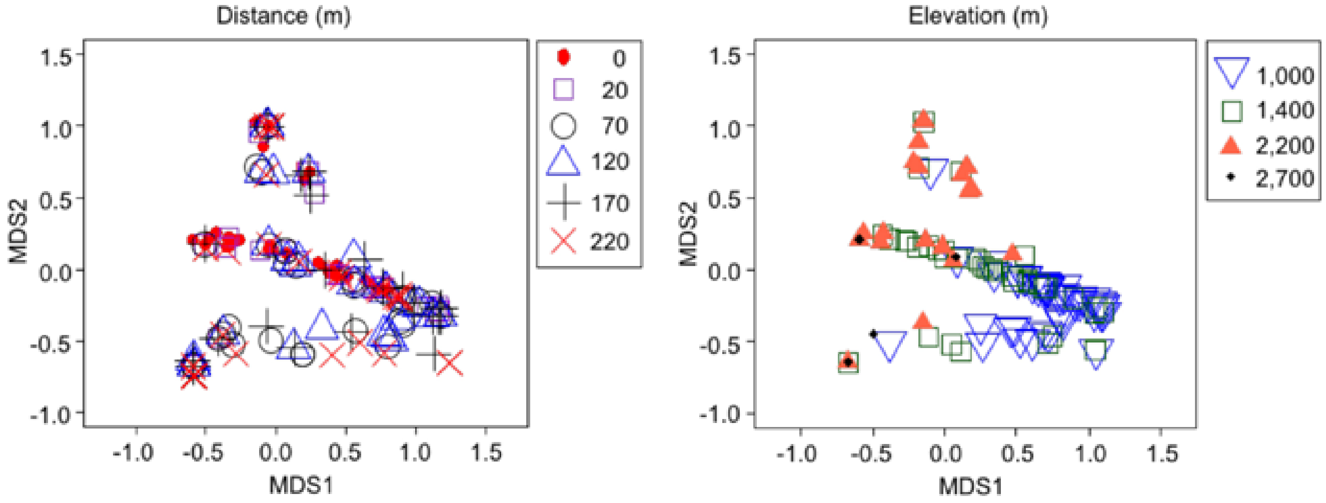


Figure 4.8. Non-metric Multidimensional Scaling (NMDS) ordinations summarising patterns of species composition (as derived from species lists at each recording site) at increasing distance from the ROW, and at different elevations, for all survey years combined.

Species-level patterns

Overall patterns can change when component species sensitive to changes in their habitat respond either positively or negatively to gaps in forest cover. It is important to understand which species are the most sensitive, or exploitative; and which are ecological generalists or more specialised.

An examination of Relative Abundance revealed the relative preponderance of two species of bent-winged bat overall, and at particular elevations (see section *Acoustic detections*; Tables 4.5 and 4.6), and statistical tests revealed that Edge species and *cFM* call emitters contributed to the overall patterns (Tables 4.7 and 4.8, Figure 4.5).

Indicator metrics help to highlight which species make a disproportionate contribution to overall patterns (Table 4.9). Three small species of *Emballonura* that emit *i.fFM.d* call types were significantly positively associated with forest edges (distance values of 0 m), as was the medium-sized *Miniopterus* (call type *45 st.cFM*), and the unidentified species of *Otomops*. No other species were significantly associated with particular distances along the transects. Two other small emballonurids, three species of Hipposideridae and one species of Rhinolophidae were associated with the lowest elevation of 1,000 m asl at Arakubi Quarry. The occurrence of the New Guinea Free-tailed Bat *Austronomus kuboriensis* only at high elevations (2,700 m asl) was also captured by the Indicator Species values, though this species is certainly not limited to such an extreme elevation.

Table 4.9. Indicator Species index (I; range 0–1), with a higher value showing its greater value as an indicator species for that distance from the ROW or elevation; A is the specificity of the species for that particular site category, and B is the probability of encountering it. For example, *H. maggietaylorae* has only ever been recorded at 1,000 m elevation (A = 1.0), but the probability of encountering it is low (B = 0.04), so overall its value as an Indicator species is relatively low (I = 0.21).

Indicator species	**Distance**	**I**	**A**	**B**	**P, sig**
Emballonura raffrayana	0 m	0.53	0.68	0.41	0.001***
Emballonura dianae	0 m	0.34	0.74	0.16	0.001***
Emballonura furax	0 m	0.26	1.00	0.07	0.031*
Miniopterus sp. 2 'medium'	0 m	0.43	1.00	0.18	0.001***
Otomops sp.	0 m	0.45	0.81	0.25	0.001***
	Elevation	**I**	**A**	**B**	**P, sig**
Austronomus kuboriensis	2,700 m	0.66	0.11	0.27	0.042*
Aselliscus tricuspidatus	1,000 m	0.59	0.71	0.49	0.001***
Emballonura dianae	1,000 m	0.30	0.70	0.13	0.006**
Hipposideros diadema	1,000 m	0.27	0.83	0.09	0.011*
Hipposideros maggietaylorae	1,000 m	0.21	1.00	0.04	0.04*
Mosia nigrescens	1,000 m	0.81	0.95	0.68	0.001***
Rhinolophus megaphyllus	1,000 m	0.53	0.94	0.30	0.001***

Discussion

Long-term trends over the PMA3 study

This component of the PMA3 monitoring study seeks to determine whether there are ongoing impacts from linear infrastructure construction on adjacent forest habitats that is reflected in changes to bat communities. Thus, the primary factor in the analyses is survey year, and whether a decline in bat diversity is observed to occur overall, at the forest edge, or at particular elevations.

There was a statistically significant difference in Species Richness and Phylogenetic Diversity amongst survey years, but pairwise comparisons showed that this derived from the lower number of overall detections in 2019 (Table 4.7). Rather than being indicative of a decline due to degrading habitat value adjacent to the edge of the ROW, access roads and Arakubi Quarry, the pattern is more consistent with variation derived from a combination of both natural variation and sampling effects (the level of site and transect replication). Variation amongst years for different factor levels of distance from the ROW and elevation is particularly evident in Figure 4.4. Likewise, the diversity of ecological niches, as measured by the Functional Diversity metric, varied somewhat over the four years, but there was no clear trend (Figure 4.7). Therefore this, there is no strong pattern that suggests ongoing habitat degradation beyond the initial creation of the infrastructure footprint.

The design of the PMA3 monitoring program does not incorporate a long-term quantitative assessment of the vegetation community at the sampling sites, so it is not possible to directly correlate bat diversity with specific habitat features or condition. However, field observations over the four surveys (K.N. Armstrong, personal observations) suggest that there have been few changes to forest structure at sampling locations since 2015. For example, the forest edge has not shifted at the start of most transects, the trees with transect markings were present, and the canopy coverage appeared unchanged. Exceptions include the removal of one tree from the beginning of transects H2 and M2, and of one tree within the middle of transect M4 between 2015 and 2017 (Richards et al. 2019). The treefalls at H2 and M2 moved the edge of the forest back around five metres, and opened a clearing around 20 metres in diameter on M4. Treefalls resulting from the major earthquake of March 2018 also opened the tree canopy on H4. The relatively small spatial scale of such tree loss is not expected to have a significant effect on bat activity. Thus, in the absence of an obvious and developing environmental gradient along transects, it is not surprising that bat diversity remains at similar levels since 2015.

A long-term perspective of changes in bat communities adjacent to the ROW must also include consideration of the original community of bats in relatively undisturbed forest. This is difficult to assess for the Agogo Range where roads and quarries have been in place for an extended period, and there was probably some degree of change in the foraging habits of the bat community in the approximately four years between the development of the ROW on Hides Ridge and the beginning of the PMA3 study in 2015. The original bat survey conducted for the PNG LNG environmental impact assessment by G. Richards (2005, 2008) represents the only available pre-construction baseline information for comparison. A re-analysis of the data collected from sites 'Hides 3' and Benaria provided an updated perspective on the identification of call types, because of recent progress in understanding their species attributions (Specialised Zoological 2017). Direct comparisons with this earlier work are constrained because the baseline sampling sites are different from the PMA3 long-term study sites, but it was possible to assess whether the same species were present in both studies.

A simple comparison of the bat species inventory between pre-construction and the combined PMA3 surveys shows that significantly more species have been identified in the four most recent surveys (25 species versus at least 13 species; Armstrong et al. 2021a). There were at least two species detected in 2005 that have not yet been encountered on the PMA3 surveys. Casual recordings made adjacent to Lake Kutubu near the team accommodation encountered the Maluku Myotis *Myotis moluccarum* (call type *35 bFM*; Figure 4.2) but it is not expected to occur in the PMA3 study sites because specific habitat requirements of this species are not present (streams with flowing water). Up to three species of *Pipistrellus* might be present, but their calls are similar to those of *Miniopterus* spp., and observations made to date suggest that only the latter is present.

Edge effects

This study identified a binary difference in Species Richness and Phylogenetic Diversity with increasing distance from the forest edge—the forest edge (0–20 m) has a greater level of diversity and usage compared to the remainder of the transect (20–220 m). This pattern is only statistically significant at the lowest elevation of 1,000 m, although there is a trend at 1,400 m (Tables 4.3 and 4.4; Figure 4.8). This pattern is also evident with higher levels of Functional Diversity at the start of transects at the lower elevations (Figure 4.7). There was a significantly greater proportion of species that forage in Edge habitats at a distance of 0 m in BAA 2, and the proportion of species that forage in Clutter was greater at all distances within the forest interior (Figure 4.5; see flight space designations for each species in Table 4.2).

The Indicator Species indices (Table 4.9) helped to highlight which species might be responsible for the pattern of higher diversity at the forest edge. There is a clear association of several Edge species with the open spaces at a transect distance of 0 m, specifically small species of *Emballonura* that emit *i.fFM.d* call types, the medium sized *Miniopterus* species (call type *45 st.cFM*) and the unidentified species of *Otomops* sp. The latter flies over the tree canopy rather than close to vegetation edges and was therefore recorded best in the open areas.

Surprisingly, based on the Indicator Species metrics and significamnce testing, forest interior species (e.g., species of *Hipposideros*, *Rhinolophus*, *Kerivoula* and *Nyctophilus*; Table 4.2) were not significantly associated with the parts of the transect under tree canopy. This is because these specialists are venturing out to the forest edges even if they do not forage completely in the open. However, some of these species were present at the lowest elevation of 1,000 m at Arakubi Quarry, possibly because of nearby rocky outcrop that provides roosts.

By contrast, most species in BAA 1 were classified as Edge species (Figure 4.5). While these species are using the ROW (especially *Miniopterus* sp. 1 'large'), an examination of Table 4.5 shows that they are also using the forest (the canopy, or the interior, or both). The Clutter species *Kerivoula muscina* / *Kerivoula* sp. / *Murina florium* and *Nyctophilus* sp. were almost always recorded inside the forest (Table 4.5).

Patterns across elevations

The most evident patterns associated with changes in elevation are substantial reductions in both Species Richness and Phylogenetic Diversity with increasing elevation (Figure 4.6). The latter trend is reflected in the absence or near-absence of certain bat families in each BAA. Transects at 2,200 m and above had less than half the Species Richness of transects at 1,400 m and below (BAA 1: 9 species; BAA 2: 22 species; Appendix 4.1). In BAA 1, there were no records of any species of Hipposideridae, only a single detection of a species in the Emballonuridae (*Mosia nigrescens* at 2,700 m in 2015) and only one of four species of Rhinolophidae (*Rhinolophus mcintyrei* detected at 2,200 m in all survey years). The reduction in Functional Diversity values with elevation (Figure 4.7) is attributable to these taxonomic and corresponding ecological absences, as are the differences in species composition with elevation (Figure 4.8). There was also a bias in the number of detections of Vespertilionidae in BAA 1, especially for the call type *80 bFM* that is attributed to more than one candidate species (*Kerivoula muscina* / *Kerivoula* sp. / *Murina florium*; Appendix 4.1). The unidentified species of free-tailed bat *Otomops* sp. (Molossidae) was detected more frequently in BAA 2; it seems to be replaced by the New Guinea Free-tailed Bat *Austronomus kuboriensis* in BAA 1 (Appendix 4.1).

The distribution patterns of free-tailed bats (Molossidae) illustrate niche replacement, with the higher elevation specialist *A. kuboriensis* likely having physiological adaptations for these cooler, resource-limited habitats. Higher elevations are associated with lower mean temperatures that affect both the abundance and activity of insect prey, and the energy budgets of small bats that are reliant on certain levels of prey capture for their survival, and physiological mechanisms to conserve heat and energy during resting periods (discussed further in Armstrong et al. 2019). To live in a less productive (in terms of insect biomass), higher elevation environment, bats need to be able to enter torpor to maintain energy reserves when prey biomass is insufficient and tempertaures are too cold. Alternatively, they need to have a different prey capture strategy, extended nightly foraging range, and/or physiology that allows activity in the cooler conditions. The unidentified long-eared bat *Nyctophilus* sp. (probably the Small-toothed Long-eared Bat *N. microdon* that was captured at transect H3 in 2017 and is known from moderate elevations of between 1,900 m asl and 2,200 m asl; Bonaccorso 1998) is another high elevation specialist that replaces the Papuan Long-eared Bat *N. microtis*, at elevations above about 1,450 m asl (Bonaccorso 1998).

There appear to be no high-altitude specialists in the Emballonuridae, Hipposideridae and Rhinolophidae. The summary accounts in Bonaccorso (1998) for each of the species in these families that has been detected in BAA 2 during the PMA3 study indicate upper elevational distribution limits below 1,800 m asl, though the distribution of Wollaston's Leaf-nosed Bat *Hipposideros wollastoni* is known to extend up to c. 2,000 m asl (Armstrong and Aplin 2021). However, the PMA3 study has extended the known upper elevational limits (as stated in the authoritative guide of Bonaccorso (1998)) for several species to: 1,400 m asl for Temminck's Leaf-nosed Bat *Aselliscus tricuspidatus* (previously 600 m); 1,000 m asl for Maggie Taylor's Leaf-nosed Bat *Hipposideros maggietaylorae* (previously 300 m); 2,700 m asl for the Lesser Sheath-tailed Bat *Mosia nigrescens*; (previously 1,600 m); and 2,200 m asl for *Rhinolophus mcintyrei* (previously 1,600 m). The record of *Mosia nigrogriseus* at transect H6 in 2015 (Armstrong 2017) is surprising and the recording has been re-checked and the observation confirmed. The detection of *Rhinolophus mcintyrei* at 2,200 m asl on transects H1 and H2 in BAA 1 across survey years possibly represents the upper limit of this species and its persistence there probably reflects the occurrence of nearby rocky roosting habitat. The abundance of cave-roosting bats at 1,000 m is likely a result of nearby rocky outcrops that probably serve as diurnal roost habitat. Records such as these serve to illustrate that the elevation limits of bat species in PNG remain incompletely documented and can be influenced locally by the availability of suitable roosting habitat including rocky outcrop.

Bat species of conservation significance

No species in an IUCN threatened category has been detected on any of the PMA3 surveys conducted to date.

The unidentified species of free-tailed bat is either *Otomops papuanus* or *O. secundus*, both of which are assessed as Data Deficient (Armstrong 2021a,b). These could be reasonably common in suitable habitat and over a much wider area than the known records suggest, but they are seldom encountered because they fly high over tree canopy.

Three species that are potentially new to science have been recorded in the study area. Two of these were captured at transect H6 at 2,700 m on Hides Ridge in 2017 and both differ in their external morphology to described species. One was a species of woolly bat (*Kerivoula* sp.), and the other was a species of long-eared bat (*Nyctophilus* sp). Both are known only from single specimens but genetic sequencing demonstrated some degree of difference from morphologically similar species captured elsewhere (Armstrong et al. 2021a). These species are good candidates for taxonomic work but additional indviduals need to be captured for adequate comparisons with existing material.

In 2015, an echolocation call type (*172 sCF*) never encountered previously in PNG was recorded on transect M5 in BAA 2 (Armstrong 2017), and it was detected again in 2017 adjacent to Lake Kutubu at KP 87 (Kale et al. 2018c). If this bat can be captured on a future survey, genetic and morphological analysis can be used to confirm its identity and whether it is indeed a species new to science.

Conclusions

1. The combined results from the 2015–2021 surveys suggest that the forest adjacent to the ROW has so far retained its value for a diverse and intact communities of bats at different elevations.

2. There have been no detectable changes in the diversity of bat communities within BAA 1 and BAA 2 since 2015.

3. The most obvious pattern detected in the distribution of bat species is the large elevational difference in Species Richness between BAA 2 (22 species in total over four surveys) and BAA 1 (9 species). Fourteen species have been found only in BAA 2 (8 spp. are shared), and one species has been detected acoustically only in BAA 1.

4. The PMA3 bat study has not detected any gradual changes in the bat community with increasing distance from the forest edge, suggesting minimal impacts of edge effects. However, it has identified that the forest edge (0–20 m) has a significantly higher level of usage compared to the remainder of the transect (20–220 m). The forest edge is exploited to a greater degree by species adapted to foraging in Edge habitats, and thus these species have responded positively to the opening of canopy in the ROW and access roads.

Recommendations

1. The acoustic bat monitoring component should be continued in future surveys because of a demonstrated ability for detecting responses of bats to open areas and the forest edge. Increasing the number of sites at the forest edge would help with statistical power, and allow a better comparison of open versus closed foraging habitats.

2. While abundance data is not available from bat detector recordings, relative levels of activity can be determined by summing the number of echolocation pulses in certain call type categories, which would give a more

detailed understanding about how much the open areas at the forest edge might be used compared to beneath the tree canopy. Relative levels of habitat usage might be better revealed by quantifying activity levels for certain echolocation call types (e.g., those from Edge species that use open areas), and of particular species identified in the Indicator Species analysis as Edge and Clutter species, including from past recordings.

3. Further effort could be given on future surveys to capturing species of bat that need to be distinguished from others with similar echolocation call types (detecting *Pipistrellus* that have calls similar to *Miniopterus* species), that need specimens to complete identification and taxonomic assessments (the source of the *172 sCF* call type; *Kerivoula* sp.; *Nyctophilus* sp.), and to confirm the identity and to collect quality reference calls to allow greater confidence in acoustics-based identifications for some species (*Austronomus kuboriensis*, *Otomops* sp.). Acoustics lures will likely increase capture probability of all these species (Hill et al. 2015).

4. We recommend that measurement of a suitable set of habitat covariates (e.g., from drone-mounted multi-spectral imaging) be undertaken at each transect as a basis for interpretation of monitoring results in future years.

References

Armstrong, K. N. 2017. Chapter 6 – Bats. Pp. 209–254 *In*: Richards, S. J. (Editor). *Biodiversity assessment of the PNG LNG Upstream Project Area, Southern Highlands and Hela Provinces, Papua New Guinea*. ExxonMobil PNG Limited. Port Morseby.

Armstrong, K. N. 2021a. *Otomops papuensis* (amended version of 2020 assessment). The IUCN Red List of Threatened Species 2021: e.T15649A209523988. https://dx.doi.org/10.2305/IUCN.UK.2021-3.RLTS.T15649A209523988.en. Accessed on 1 June 2022.

Armstrong, K. N. 2021b. *Otomops secundus* (amended version of 2020 assessment). The IUCN Red List of Threatened Species 2021: e.T15650A209524157. https://dx.doi.org/10.2305/IUCN.UK.2021-3.RLTS.T15650A209524157.en. Accessed on 1 June 2022.

Armstrong, K. N. & Aplin, K. P. 2011. Bats of the Muller Range, Papua New Guinea. Chapter 19, Pp. 222–234 *In*: Richards, S. J. & Gamui, B. G. (Editors). *Rapid Biological Assessments of the Nakanai Mountains and the upper Strickland Basin: surveying the biodiversity of Papua New Guinea's sublime karst environments*. RAP Bulletin of Biological Assessment 60. Conservation International, Arlington, USA.

Armstrong, K. N. & Aplin, K. P. 2014. Chapter 7. A survey of bats (Chiroptera) in the Baiyer River Wildlife Sanctuary, Western Highlands Province, Papua New Guinea. Pp. 111–133 *In*: Richards, S. J. (Editor). *A rapid biodiversity assessment of the Baiyer River region, Western Highlands Province, Papua New Guinea*. A report to the Mul Baiyer Lumusa District Administration, Papua New Guinea.

Armstrong, K. N. & Aplin, K. 2021. *Hipposideros wollastoni* (amended version of 2017 assessment). The IUCN Red List of Threatened Species 2021: e.T10166A209537699. https://dx.doi.org/10.2305/IUCN.UK.2021-3.RLTS.T10166A209537699.en. Accessed on 1 June 2022.

Armstrong, K. N., Novera, J. & Aplin, K. P. 2014. Survey for bats in the P'nyang Range, Western Province, Papua New Guinea. Field survey and acoustic analysis. PNG LNG Expansion Project. Unpublished report by Specialised Zoological, Ken Aplin Fauna Studies Pty Ltd and the Papua New Guinea Institute for Biological Research for Coffey Environments Pty Ltd and ExxonMobil, 21 April 2014.

Armstrong, K. N., Novera, J. & Aplin, K. P. 2015a. Acoustic survey of the echolocating bats of Manus Island and Mussau Island, Papua New Guinea. Pp. 69–85 *In*: Whitmore, N. (Editor). *A Rapid Biodiversity Survey of Papua New Guinea's Manus and Mussau Islands* Wildlife Conservation Society Papua New Guinea Program. Goroka, Papua New Guinea.

Armstrong, K. N., Aplin, K. P. & Lamaris, J. S. 2015b. Chapter 10. Bats. Pp. 166–180 *In*: Richards, S. J & Whitmore, N. (Editors). *A rapid biodiversity assessment of Papua New Guinea's Hindenburg Wall region*. Wildlife Conservation Society Papua New Guinea Program. Goroka, Papua New Guinea.

Armstrong, K. N., Amick, P. & Kale, E. 2019. Chapter 5 – Bats. Pp. 153–192 *In*: Richards, S. J. (Editor). *Results of the second PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. ExxonMobil PNG Limited. Port Moresby.

Armstrong, K. N., Mani, A. & Nagombi, E. 2021a. Chapter 4 – Bats. Pp. 147–181 *In*: Richards, S. J. (Editor). *Results of the third PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 8 August–2 September 2019*. ExxonMobil PNG Limited, Port Moresby.

Armstrong, K N, Broken-Brow, J., Hoye, G., Ford, G., Thomas, M. & Corben, C. 2021b. Effective detection and identification of sheath-tailed bats of Australian forests and woodlands. *Australian Journal of Zoology* 68: 346–363.

Armstrong, K. N., Clarke, S., Linke, A., Scanlon, A., Roetman, P., Hitch, A. T. & Donnellan, S. C. 2021c. Citizen science implements the first intensive acoustics-based survey of insectivorous bat species across the Murray-Darling Basin of South Australia. *Australian Journal of Zoology* 68: 364–381.

Berthinussen, A. & Altringham, J. 2012. The effect of a major road on bat activity and diversity. *Journal of Applied Ecology* 49: 82–89.

Blake, D., Hutson, A. M., Racey, P. A., Rydell, J., & Speakman, J. R. 1994. Use of lamplit roads by foraging bats in southern England. *Journal of Zoology* 234: 453–462.

Bonaccorso, F. J. 1998. *Bats of Papua New Guinea*. Conservation International Tropical Field Guide Series. Conservation International, Washington, D.C.

Coffin, A. W. 2007. From roadkill to road ecology: A review of the ecological effects of roads. *Journal of Transport Geography* 15: 396–406.

De Cáceres, M. & Legendre, P. 2009. Associations between species and groups of sites: indices and statistical inference. *Ecology* 90: 3566–3574.

Denzinger, A. & Schnitzler, H.-U. 2013. Bat guilds, a concept to classify the highly diverse foraging and echolocation behaviors of microchiropteran bats. *Frontiers in Physiology* 4 (Article 164): 1–15.

Dufrêne, M. & Legendre, P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. *Ecological Monographs* 67: 345–366.

Fahrig. L. & Rytwinski, T. 2009. Effects of roads on animal abundance: an empirical review and synthesis. *Ecology and Society* 14: http://www.ecologyandsociety.org/vol14/iss1/art21/

Faith, D. P. 1992. Conservation evaluation and phylogenetic diversity. *Biological Conservation* 61: 1–10.

Harris, L. D. 1988. Edge effects and conservation of biotic diversity. *Conservation Biology* 2: 330–332.

Hill, D. A., Armstrong, K. N. & Barden, P. A. 2015. Preliminary assessment suggests that acoustic lures can increase capture rates of Australian echolocating bats. *Australian Mammalogy* 37: 104–106.

Jones, K. E., Purvis, A. & Gittleman, J. L. 2003. Biological correlates of extinction risk in bats. *The American Naturalist* 161: 601–614.

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

Kale, E., Armstrong, K. N. & Amick, P. 2018a. Chapter 1.6. Mammals of the Wau Creek proposed Wildlife Management Area, Gulf Province, Papua New Guinea. Pp. 125–151 *In*: Richards, S. J. (Editor). *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kale, E., Armstrong, K. N. & Amick, P. 2018b. Chapter 2.6. Mammals of the Uro Creek catchment, Gulf Province, Papua New Guinea. Pp. 245–270 *In*: Richards, S. J. (Editor). *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kale, E., Armstrong, K. N. & Amick, P. 2018c. Chapter 3.6. Mammals of the Lake Kutubu Wildlife Management Area, Southern Highlands Province, Papua New Guinea. Pp. 353–378 *In*: Richards, S. J. (Editor). *Rapid biological assessments of Wau Creek, Uro Creek and Lake Kutubu: documenting biodiversity values to promote forest conservation in the Kikori River basin, Papua New Guinea*. ExxonMobil PNG Limited, Port Moresby.

Kerth, G. & Melber, M. 2009. Species-specific barrier effects of a motorway on the habitat use of two threatened forest-living bat species. *Biological Conservation* 142: 270–279.

Kingston, T. 2013. Response of bat diversity to forest disturbance in Southeast Asia: Insights from long-term research in Malaysia. Pp. 169–185 *In*: Adams, R. & Pedersen, S. (Editors). *Bat evolution, ecology, and conservation*. Springer, New York.

Kuijper, D. P. J., Schut, J., van Dullemen, D., Toorman, H., Goossens, N., Ouwehand, J. & Limpens, H. J. G. A. 2008. Experimental evidence of light disturbance along the commuting routes of pond bats (*Myotis dasycneme*). *Lutra* 51: 37–49.

Petchey, O. L. & Gaston, K. J. 2002. Functional diversity (FD), species richness and community composition. *Ecology Letters* 5: 402–411.

R Core Team 2021. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL: https://www.R-project.org/

Richards, G. C. 2005. The PNG gas project: a study of bat faunal biodiversity and an assessment of potential impacts. Prepared by Greg Richards and Associates Pty Ltd for Enesar Consulting Pty Ltd, July 2005. Included as *Annex 05. Biodiversity survey results: Bats at Hides, Nogoli and Benaria in 2005* in the PNG LNG Project Environmental Impact Statement Part II. Existing Environment, prepared by Coffey Natural Systems Pty Ltd for Esso Highlands Ltd, January 2005.

Richards, G. C. 2009. The PNG liquefied natural gas project: a study of bat faunal biodiversity and an assessment of potential impacts. Prepared by Greg Richards and Associates Pty Ltd for Coffey Natural Systems Pty Ltd, July 2008. Included as *Annex 06. Biodiversity survey results: Bats at Juha North, Juha South, Baia River, South Karius and Deviation Camp in 2008* in the PNG LNG Project Environmental Impact Statement Part II. Existing Environment, prepared by Coffey Natural Systems Pty Ltd for Esso Highlands Ltd, January 2009.

Richards, S. J., Armstrong, K. N., & Dahl, C. 2019. Chapter 2 – Frogs. Pp. 33–61 *In*: Richards, S. J. (Editor). *Results of the second PMA3 Biodiversity Monitoring Survey of the PNG LNG Upstream Project Area, 10–31 May 2017*. ExxonMobil PNG Limited. Port Moresby.

Schaub, A., Ostwald, J. & Siemers, B. J. 2008. Foraging bats avoid noise. *Journal of Experimental Biology* 211: 3174–3180.

Shearman, P. & Bryan, J. (Editors). 2015. *The state of the forests of Papua New Guinea 2014: Measuring change over the period 2002–2014*. University of Papua New Guinea, Papua New Guinea.

Specialised Zoological (2017). A re-analysis of bat echolocation recordings from the PNG LNG Environmental Impact Statement bat study. Acoustic analysis. Unpublished report by Specialised Zoological for ExxonMobil PNG LNG, 6 February 2017, Job number SZ425.

Spellerberg, I. 2002. Ecological effects of roads and traffic: a literature review. *Global Ecology and Biogeography* 7: 317–333.

Stone, E. L., Jones, G. & Harris, S. 2009. Street lighting disturbs commuting bats. *Current Biology* 19: 1123–1127.

Stone, E. L., Jones, G. & Harris, S. 2012. Conserving energy at a cost to biodiversity? Impacts of LED lighting on bats. *Global Change Biology* 18: 2458–2465.

Wiantoro, S. 2020. Cryptic diversity and rapid radiation of Indo-Australasian bent-winged bats (*Miniopterus* Bonaparte, 1837). PhD thesis, The University of Adelaide.

Trombulak, S. C. & Frissell, C. A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conservation Biology* 14: 18–30.

Zurcher, A. A., Sparks, D. W. & Bennett, V. J. 2010. Why the bat did not cross the road? *Acta Chiropterologica* 12: 337–340.

The bat team programming an Anabat recorder in preparation for an evening of bat detection (photo by Kalyna Taule).

Appendix 4.1. Total number of times each species was detected at a recording site over the four surveys, summarised by elevation (m), BAA and overall.

Species	Call type	1,000	1,400	2,200	2,700	BAA 2	BAA 1	Total detections
EMBALLONURIDAE								
Emballonura dianae	*35 i.fFM.d*	6	3	0	0	9	0	9
Emballonura furax	*52 i.fFM.d*	2	1	0	0	3	0	3
Emballonura raffrayana	*45 i.fFM.d*	12	13	0	0	25	0	25
Mosia nigrescens	*65 i.fFM.d*	32	1	0	1	33	1	34
Saccolaimus saccolaimus	*25 cFM*	1	0	0	0	1	0	1
HIPPOSIDERIDAE								
Aselliscus tricuspidatus	*120 sCF*	23	11	0	0	34	0	34
Hipposideros cervinus	*140 sCF*	11	10	0	0	21	0	21
Hipposideros diadema	*58 mCF*	4	1	0	0	5	0	5
Hipposideros maggietaylorae	*125 sCF*	2	0	0	0	2	0	2
Hipposideros wollastoni	*88 mCF*	33	19	0	0	52	0	52
Hipposideros sp. cf. *ater*	*172 sCF*	1	0	0	0	1	0	1
RHINOLOPHIDAE								
Rhinolophus euryotis	*52 lCF*	33	20	0	0	53	0	53
Rhinolophus mcintyrei	*70 lCF*	18	1	17	0	19	17	36
Rhinolophus megaphyllus	*65 lCF*	14	1	0	0	15	0	15
Rhinolophus sp. cf. *robertsi*	*33 lCF*	14	10	0	0	24	0	24
MINIOPTERIDAE								
Miniopterus tristis grandis	*38 st.cFM*	16	24	53	21	40	74	114
Miniopterus sp. 2 'medium'	*45 st.cFM*	3	5	0	0	8	0	8
Miniopterus sp. 3 'small'	*53 st.cFM*	27	35	19	60	62	79	141
VESPERTILIONIDAE								
Kerivoula muscina / *Kerivoula* sp. / *Murina florium*	*80 bFM*	5	4	3	6	9	9	18
Nyctophilus sp.	*50 bFM*	0	0	8	7	0	15	15
Philetor brachypterus	*30 cFM*	2	1	4	1	3	5	8
MOLOSSIDAE								
Austronomus kuboriensis	*13 cFM*	1	1	1	7	2	8	10
Otomops sp.	*25 sFM*	7	5	1	0	12	1	13
Total sampling nights		48	72	72	72	120	144	264
Total detections (all 264 sites and nights)		267	166	106	103	433	209	1284
Total Species Richness		22	19	8	7	22	9	23