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Understanding and managing impacts to biodiversity from roads and pipelines in the Beetaloo Sub-region

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Citation

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Executive Summary

A major environmental impact from development of an onshore gas industry in the Beetaloo Sub-region will be the construction of new, and extension of existing, components of linear transport infrastructure (roads, tracks, railways, pipeline corridors), and the subsequent use of roads by heavy-vehicle traffic. Measures need to be developed to mitigate risks from this linear infrastructure in the Beetaloo. However, very little is known about the impacts to biodiversity of roads and other infrastructure, and of habitat fragmentation in northern Australian savannas. The current project addressed these issues in the Beetaloo.

The project had four objectives. First, to use data from remote sensing to map connectivity (of vegetation) within the study area in order to identify connectivity corridors and 'at risk' linkages. Second, to assess mortality of animals on roads that differ in width, traffic volumes and degree of disturbance in order to predict how impacts may increase with road widening, increased traffic volumes and an increase in extent of the road network. Third, to examine how the occurrence of species of plants and vertebrate animals is influenced by the size and degree of spatial isolation of patches of habitat. Fourth, to develop appropriate measures of mitigating risk from the development of roads and pipelines, for consideration by the regulator for possible use in gas field design and policy settings.

Vegetation fragmentation and connectivity were assessed using information obtained from high-resolution (10 m) surfaces modelled using Sentinel 2A/2B composite images from the dry season of 2021 (July-September). The study focussed on vegetation with a woody cover fraction of > 20 % and a height of > 2 m. Pixel-based methods for assessing fragmentation and connectivity were applied to four regions each of 50 km by 50 km. The study showed that indicators of fragmentation (FAD; Foreground Area Density) and connectivity (MSPA; Morphological Spatial Pattern Analysis), although designed for forest ecosystems, can also be applied to savanna and associated vegetation in northern Australia. Once the development footprint of future onshore gas development in the Beetaloo is understood, the connectivity mapping undertaken in this module can be used to assess potential impacts. Specifically, the location of proposed well pads, new roads, tracks, and pipelines can be superimposed on the connectivity map for a region to enable planners to identify the intersection of linear infrastructure with high value areas in the landscape including core habitat, bridges, and branches among other landscape elements.

Road mortality of vertebrates was assessed on three types of road (primary highway, secondary highway, secondary road) each of which had two replicate transects 50 km in length. Transects were driven at a set speed of 50 km/hour and each individual vertebrate killed or injured in a collision with a vehicle was identified. Road transects were driven during three sampling periods: early dry season (June 2022), late dry season (November 2022) and the wet season (February 2023). A total of 411 dead or injured native animals from 44 species was observed including mammals, frogs, reptiles, and birds. Body mass of roadkill species ranged from 90 kg (male red kangaroos) to 7 g (small birds). Six species of macropods (kangaroos and wallabies) dominated the sample numerically making up 70% of all roadkill. The number of macropods killed was high in the early dry and late dry seasons but declined during the wet season. Reptiles and birds made up a higher proportion of road kills in the wet season than at other times of year. Frogs were only recorded in the wet season.

The study found that the secondary roads sampled had a low rate of wildlife roadkill as a consequence of a low traffic volume although speed limits were often 100 km/hour. Traffic volumes on several roads, especially the Stuart and Carpentaria Highways, are predicted to increase markedly during the construction and operation phases of an onshore gas industry. This increase in traffic will result in an increase in wildlife—vehicle collisions and the resulting mortality of wildlife.

The project did not detect any areas (hotspots) where there was a concentration of wildlife mortality following collisions with vehicles. With the exception of macropods, no species appeared particularly susceptible to collisions with vehicles. The following six species listed as threatened in the Northern Territory under the *Territory Parks and Wildlife Conservation Act* were recorded as roadkill in this study (during structured surveys and incidentally): Australian bustard, bush stone-curlew, northern nailtail wallaby, spectacled hare-wallaby, yellow-spotted monitor, and common brushtail possum. The incidence of roadkill of each of these species was low except for the northern nailtail wallaby (N = 14 individuals).

We concluded that mitigation measures such as wildlife crossing structures (overpasses, underpasses, wildlife bridges) are unlikely to be necessary in the study area. The reasons for this conclusion are a lack of evidence of hotspots of road mortality and the lack of a target species, or group of species, that is likely to benefit sufficiently at the local population level from the presence of crossing structures to justify the costs. Potential mitigation measures include road signs, vehicle speed limits, fencing roads to prevent access for wildlife, and habitat modification in the form of removing vegetation from the sides of the major highways during the wet season.

The assessment of patch mosaics focussed on bullwaddy (*Macropteranthes kekwickii*) and lancewood (*Acacia shirleyi*) communities that potentially support the highest concentrations of species 'at risk' from linear corridors. Sampling was stratified across three patch categories: large, medium, and small. Floristic composition and vegetation structure differed according to patch size. Small patches typically supported thicker stands of bullwaddy and were more productive with the highest concentrations of species, including resprouter species and rainforest trees, shrubs, and vines. Consistent with the findings for vegetation, vertebrate assemblages also differed according to patch size with small patches having a higher species richness (average of 27 species) than medium (average of 19 species) and large (average of 18 species) patches.

The results of the patch mosaics module of the project suggest that the species-rich small bullwaddy stands may be persisting as low flammability islands in the landscape matrix of highly flammable savanna. In contrast, lancewood is currently absent from portions of the landscape where fire is frequent because, compared to bullwaddy, it is fire-prone, and it is very sensitive to fire interval due to its long juvenile period and low resistance to fire in adult trees. The small patches of bullwaddy may therefore be a sign that fire is presently too frequent in this landscape and lancewood stands are being lost. Compared to fire, there was little evidence of cattle and weed impacts on bullwaddy and lancewood, although there was some suggestion that cattle posed a higher risk to bullwaddy than to lancewood.

1 Introduction

The development of an onshore gas industry in the Beetaloo Sub-basin presents risks relating to the potential loss of terrestrial biodiversity, ecosystem function and landscape amenity. One major form of environmental impact from such a development will be the construction of new, and extension of existing, components of linear transport infrastructure (roads, tracks, railways, pipeline corridors) and from the subsequent use of roads by heavy-vehicle traffic. *The Final Report of the Inquiry into Hydraulic Fracturing in the Northern Territory* (2018) (page 200) concluded that with no further mitigation the overall assessment of risk to biodiversity from road and pipeline corridors and fragmentation of habitat would be ‘medium’, and unacceptable.

Prior to developing measures to mitigate these risks, two key components of understanding are needed. First, information needs to be obtained on the likely effects on biodiversity of habitat fragmentation, and how roads and other linear transport infrastructure may function as both a barrier to movement and cause of mortality. Second, the location, intensity, and scale of development of linear transport infrastructure needs to be determined. The current report describes research that was designed to address the first issue. It seeks to provide understanding of the likely effects on biodiversity of roads and pipelines.

As identified in *The Final Report of the Inquiry into Hydraulic Fracturing*, the effects on biodiversity of fragmentation of habitat and linear transport infrastructure have not been investigated in the savannas of northern Australia. The biodiversity of the northern savannas is unique and the response of its key components to linear transport infrastructure may differ from that demonstrated in other regions of the world. Previous work on fragmentation in eucalypt-dominated woodlands has been undertaken near Darwin on a limited spatial scale (Rankmore and Price 2004, Rankmore 2006). Nothing is known regarding the impacts of habitat fragmentation and edge effects in the Beetaloo Sub-basin.

The research project described in the current report was undertaken to examine how the extension of linear transport infrastructure (mainly roads and pipelines) in the Beetaloo Sub-basin during the development of an onshore gas industry may impact biodiversity. Given the absence of any information on the effects of fragmentation on the fauna and flora of the Beetaloo Sub-basin, the project undertook a program of work that involved gathering of scientific knowledge to apply to the design of the network of roads and pipelines. To this end, this project partially addresses the following recommendations of the NT Hydraulic Fracturing Inquiry: 8.7, 8.9, 8.10 and 8.11.

1.1 Study area

The project took place within the Beetaloo Sub-basin as defined by the Northern Territory government in 2020 (see Figure 1). The focal study area (i.e., the area within which all fieldwork was to be undertaken) was based on the likely gas development area, current as of 22 August 2020. The focal study area was modified in early 2022 based on three factors: (a) discussions with gas company (Origin and Santos) representatives during February 2022 regarding changes in the likely development area, (b) information on road development and traffic volumes (KPMG 2019, Bruce et al. 2021), and (c) access to pastoral properties to carry out field work. The focal study area is predicted to be heavily impacted by the extension of linear transport infrastructure if onshore gas development goes ahead in the Beetaloo Sub-basin.

The study area is described in detail in several major reports (Huddlestone-Holmes et al. 2020, Department of Environment, Parks and Water Security 2022). Briefly, the study area forms part of the Sturt Plateau Bioregion, a predominantly flat erosional plain of shallow infertile soil and dominated by savanna woodlands of mixed eucalypts and bloodwoods with a perennial grass understorey, and open woodlands on clay soil floodplains. In this part of the bioregion, forest, and woodland patches of lancewood and bullwaddy are an additional prominent landscape component. The lancewood canopy dominant, *Acacia shirleyi*, is a single-stemmed tree to 15 m that grows in shallow gravelly or skeletal sandy soils on sandstone or laterite, often forming dense stands, but also occurring in closed forests, low open forests, or mixed savanna woodlands. The bullwaddy dominant, *Macropteranthes kekwickii*, is a tree or shrub that can form dense impenetrable thickets and is also associated with lateritic soils (Parks and Wildlife Commission of the Northern Territory 2005). These canopy species often co-occur but their relative abundance is highly variable across their range (Northern Territory Government unpublished data). Woodland with high bullwaddy cover supports high species richness relative to woodland/forest dominated by lancewood. Thick bullwaddy patches (‘thickets’) are further known to support a high richness of rainforest allied taxa, predominantly climbing vines, trees, and shrubs.

Average annual rainfall at the Daly Waters airfield (1939 to 2022) is 675.9 mm. The vast majority of that rainfall (85% on average) falls during the four months from December to March. If November is added to the wet season, 92% of rainfall falls between November and March. The rainfall during each month of the study period (June 2022 to February 2023) was above average for all but two months (Figure 2).

1.2 Study Objectives

The project had four objectives. First, to use data from remote sensing to map connectivity (of vegetation) within the study area. The purpose of this work was to identify connectivity corridors. Second, to assess road mortality of animals on roads that differ in width, traffic volumes and degree of disturbance. The aim of this component was to understand existing impacts in order to predict how impacts may increase with road widening, increased traffic volumes and an increase in extent of the road network. Third, to examine how the occurrence of species of plants and vertebrate animals was influenced by the size and degree of spatial isolation of patches of habitat (referred to as ‘patch mosaics’). Last, to develop appropriate measures to mitigate risk from the development of roads and pipelines.

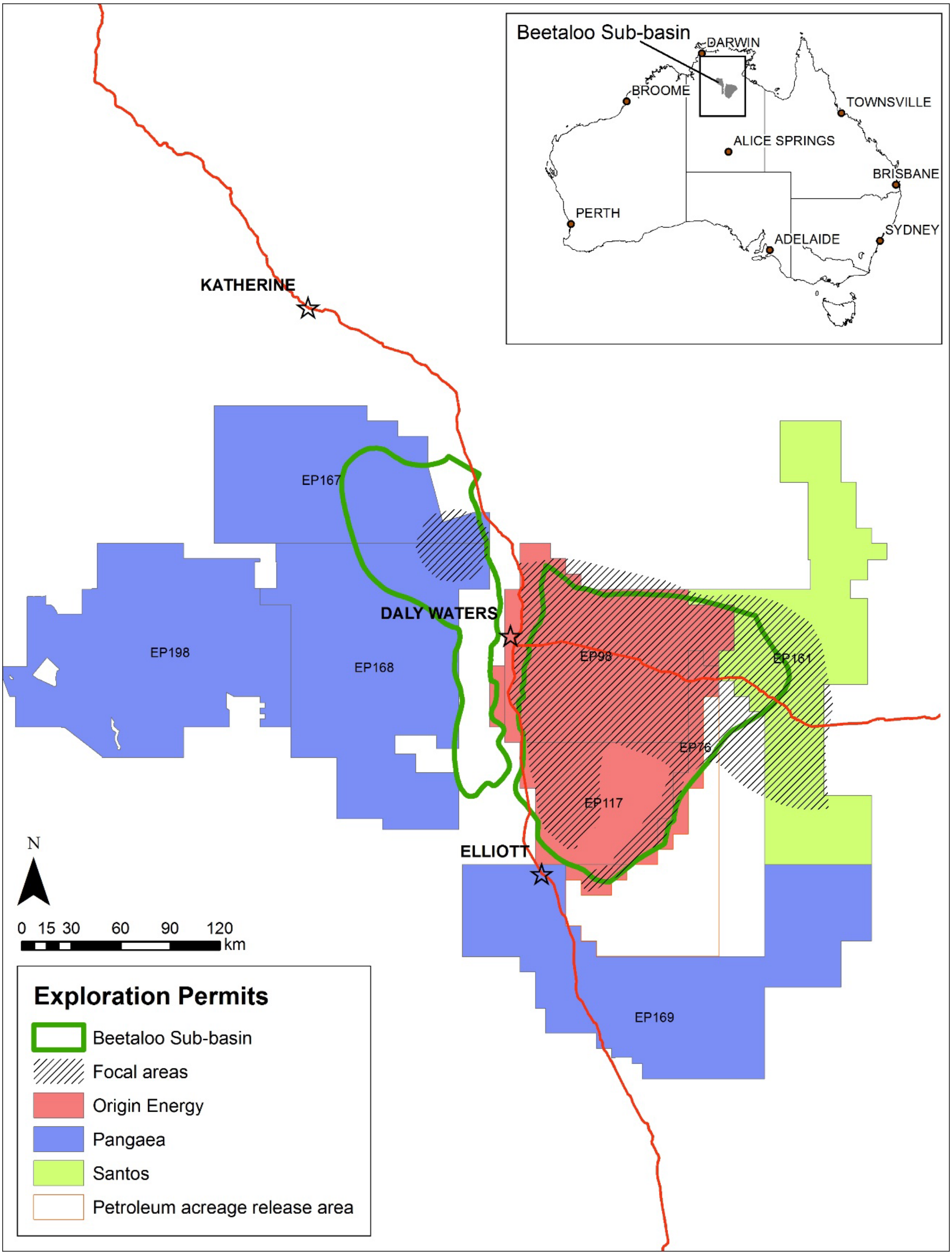


Figure 1 Map of the Beetaloo study area as of 2020

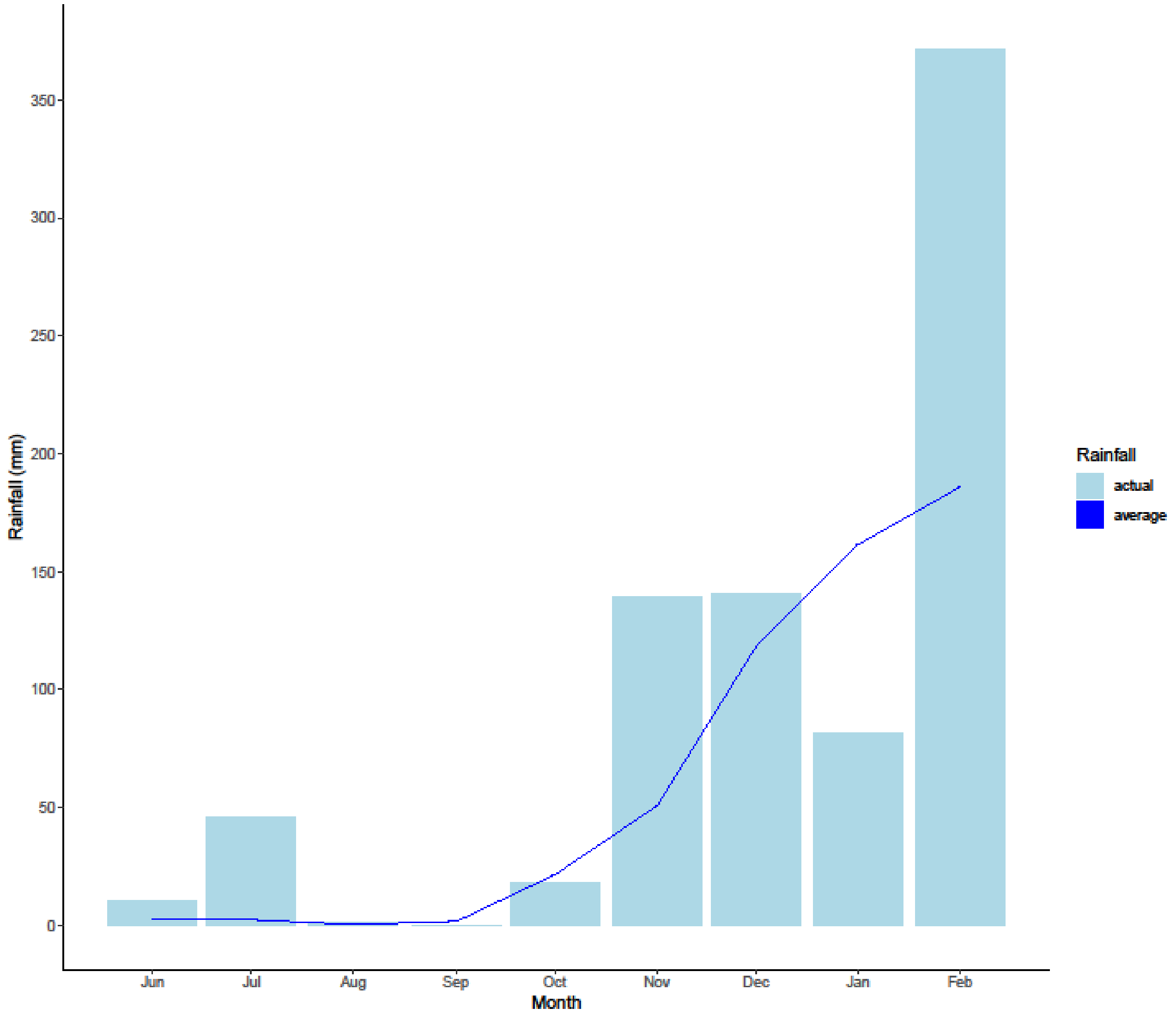


Figure 2 Rainfall at Daly Waters airport during the study period and the average annual rainfall for each month

2 General Methods

Each of the project components was unique and there was no overlap in the methods so there are no general methods *per se*.

Field research was carried out from June 2022 to February 2023. Mapping of fragmentation and connectivity was based on satellite imagery from the 2021 dry period spanning July to September.

Field research was carried out under permit from the Northern Territory Government (Permit to interfere with wildlife number: 71093). Ethics approval was granted from CSIRO's Wildlife, Livestock and Laboratory Animals Ethics Committee (CWLLA AEC) Authority number 2022-13.

3 Mapping of connectivity

3.1 Introduction

Fragmentation and connectivity are important topics in conservation and landscape ecology. Habitat fragmentation can lead to changes in the persistence of species. The response of individual species to habitat fragmentation can be difficult to predict because fragmentation is an aggregate process that involves both a decline in the area of habitat and alteration of its spatial configuration (Yeager et al. 2016, Pavey et al. 2021). When considering changes in spatial configuration, a range of attributes have been shown to be important including patch size (the area of a fragment of potential habitat), patch isolation, number of patches, matrix (the areas between the remaining fragments) quality and edge characteristics.

The connectivity of landscapes provides a measure of the impact of fragmentation. Connectivity describes the degree to which habitats are connected, enabling the movement of species and flow of ecological processes across landscapes.

Multiple approaches have been developed to measure and describe habitat fragmentation. Most approaches are designed to operate at a landscape or patch scale; an example is the FRAGSTATS software program (McGarigal et al. 2015). More recent approaches have sought to develop metrics that harness the potential of earth observation science to measure fragmentation at large scales. The Foreground Area Density (FAD) is designed to provide a pixel-level measure of fragmentation. It describes the density of a foreground class of interest and was developed to track changes in forest fragmentation in the United States (Riitters and Wickham 2012).

Similar to fragmentation, there are multiple methods and metrics for describing connectivity and most common approaches are designed to operate at landscape or patch scales. More recent approaches aligned with opportunities to harness larger datasets include the Morphological Spatial Pattern Analysis (MSPA), a pixel-level classification technique, and the Minimum Planar Graph (MPG), a flexible patch-level method based in graph theory.

This section explores the potential of applying these approaches to measure fragmentation and connectivity in the Beetaloo Sub-region. It uses the FAD and MSPA methods for measurement of fragmentation and connectivity, respectively.

3.2 Methods

3.2.1 General methods

Vegetation fragmentation and connectivity were assessed using information obtained from high-resolution (10 m) surfaces modelled using Sentinel 2A/2B composite images. The images used were from the dry season, covering the period from July to September of 2021.

For the purposes of this study the focus was on woody vegetation cover. This was defined as vegetation with a woody cover fraction of > 20 % and a height of > 2 m.

The mapping of fragmentation and connectivity is most effective for planning purposes if a development scenario is already in place. In contrast, during the period when this work was undertaken the development area for onshore gas in the Beetaloo was unknown. The Beetaloo Sub-region and beyond consisted of a series of exploration permits (Figure 1). As a consequence, for the purposes of this module of the project four regions each of 50 km by 50 km were demarcated across the Beetaloo and the relevant metrics were calculated for each of these regions (Figure 3).

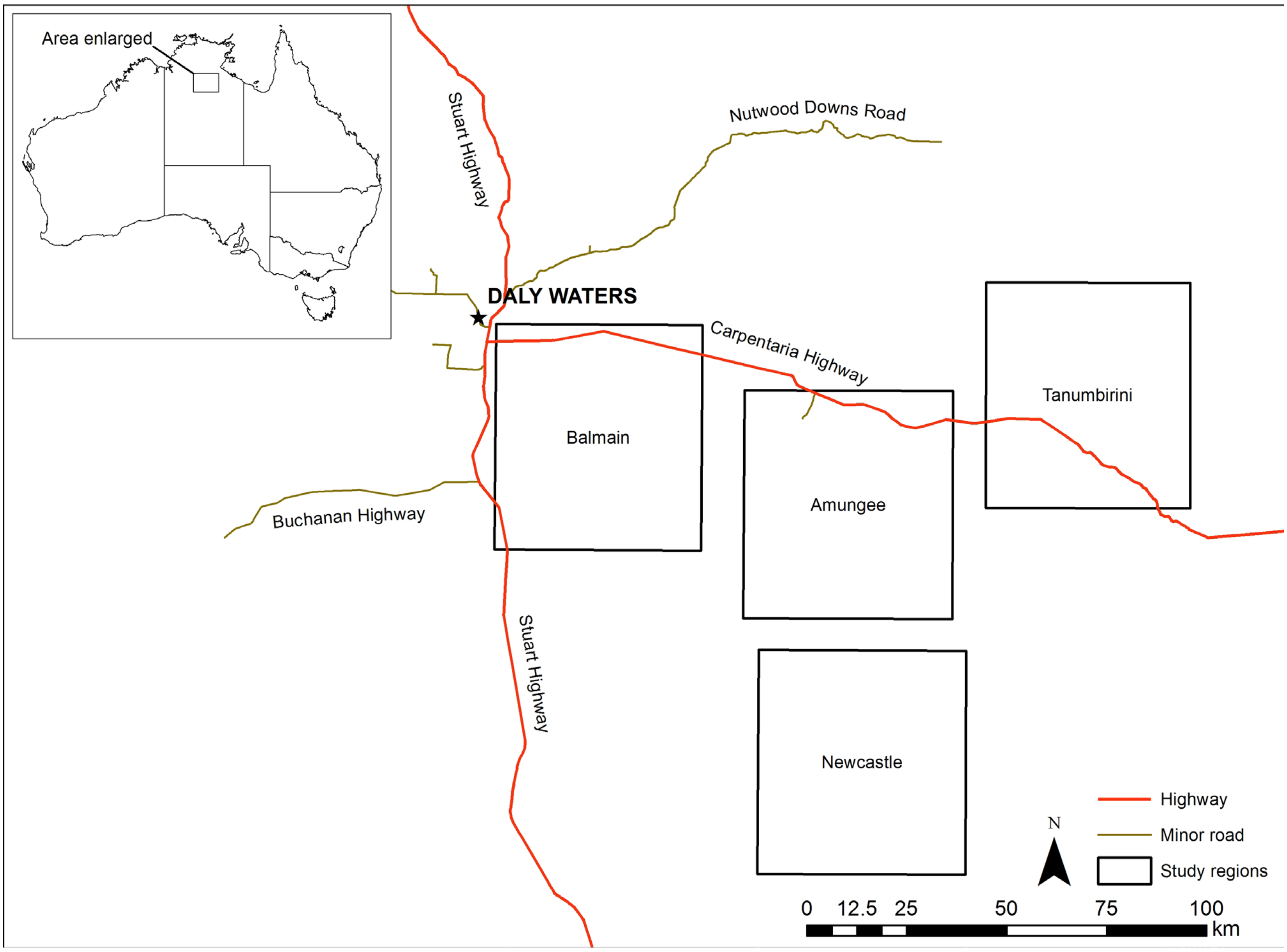


Figure 3 Map of the Beetaloo Sub-basin showing the location of the four areas assessed for fragmentation and connectivity

3.2.2 Fragmentation: Foreground Area Density (FAD)

The FAD describes the density of a foreground class of interest. It was developed to track changes in forest fragmentation in the United States (Riitters and Wickham 2012) but can be applied to any binary pattern. In this study 'forest' as used in the descriptions of Riitters and Wickham (2012) is replaced with 'woody vegetation'. The FAD is calculated as the proportion of foreground pixels (e.g., pixels associated with woody vegetation cover) that occur within a moving window across an image. It differs from a simple moving average in that background pixels (e.g., pixels not associated with woody vegetation cover) are not evaluated and are assigned a score of 0. The FAD is defined as the proportion of all land cover pixels within a fixed-area neighbourhood that are forest (woody vegetation). If woody vegetation is not fragmented in the vicinity of a given woody vegetation pixel, then by definition FAD equals 1.0 for a neighbourhood which contains that woody vegetation pixel. Alternatively, if woody vegetation is fragmented in the vicinity, then the value of FAD is less than 1.0 in proportion to the degree of fragmentation (i.e., number of pixels that are not woody vegetation) within the neighbourhood. Therefore, FAD is a simple metric of fragmentation as a contextual variable associated with a given forest/woody vegetation pixel (Riitters and Wickham 2012).

To account for the scale dependency of fragmentation, Riitters and Wickham (2012) developed the multi-scale FAD. Here the FAD process is typically applied across several window sizes to enable analysis of fragmentation across multiple scales. The multi-scale FAD allows local fragmentation to be quantified in the context of regional patterns. For example, small fragments of woody vegetation will score more highly where in proximity to larger contiguous patches. Conversely, the interior cores of large contiguous patches of woody vegetation will score more highly than the those of small to moderately sized patches of woody vegetation.

The value of FAD associated with a given forest/woody vegetation pixel will increase or decrease with neighbourhood size in proportion to changes in the degree of fragmentation at different spatial scales. A smaller neighbourhood is more sensitive to fragmentation that varies at a higher spatial frequency, while a larger neighbourhood is more sensitive to fragmentation that varies at a lower spatial frequency (Riitters and Wickham 2012). In this analysis, we calculated FAD using square moving windows with a width of 7, 13, 27, 81, and 243 pixels. The final multi-scale values are calculated as the mean across each observation scale.

The analyses presented below were conducted with the software GuidosToolBox (Vogt and Riitters 2017). The software separates the FAD into 6 key density classes for visualisation purposes (Figure 4).

The multi-scale FAD is a pixel-level metric, and therefore varies across the whole study region regardless of patch membership. Note that the density classes have been applied for visualisation purposes, but the raw imagery provides discrete integers between 0 and 100 for the FAD.

FAD 6-class	Color	FAD range
1-Rare		FAD < 10%
2-Patchy		10% ≤ FAD < 40%
3-Transitional		40% ≤ FAD < 60%
4-Dominant		60% ≤ FAD < 90%
5-Interior		90% ≤ FAD < 100%
6-Intact		FAD = 100%

Figure 4 Description of density classes for the Foreground Area Density (FAD) method

3.2.3 Connectivity: Morphological Spatial Pattern Analysis (MSPA)

To assess connectivity, we used MSPA. The MSPA provides a general approach to characterising structure in binary patterns (Vogt et al. 2007, Soille and Vogt 2009). It has been proposed for several applications, such as landscape analyses of habitat connectivity, segmenting circuit board designs, and classifying structural components in medical imagery. The MSPA uses mathematical morphometry techniques such as erosion, geodesic dilation, reconstruction by dilation, and anchored skeletonisation to classify binary patterns into discrete classes. These classes include cores, edges, and bridges, that can be used to assist in identifying important habitat patches, patch characteristics, and components of the landscape that connect core habitats to one another. Each of the classes that are typically segmented using the MSPA are tabulated below (Table 1).

Graph-based approaches to modelling landscape connectivity were also explored during this module; however, the results are not presented in this report. Graph-based approaches such as Minimum Planar Graphs (MPG) are potentially usefully because they provide a mechanism to investigate the structure and ecological function of habitat patches (Galpern et al. 2011, Galpern et al. 2012, Kupfer 2012).

Table 1 Pixel classification used for the Morphological Spatial Pattern Analysis (MSPA) resulting in eight distinct classes

Class	Description
Background	Background.
Branch	Connected at one end to edge, perforation, bridge, or loop.
Bridge	Connected to different core areas.
Core	Interior area excluding perimeter.
Edge	External object perimeter.
Islet	Disjoint and too small to contain core.
Loop	Connected to the same core area.
Perforation	Internal object perimeter.

The analyses presented below were conducted with the software GuidosToolBox (Vogt and Riitters 2017), using the MSPA module. An edge width of three pixels was selected to balance core and bridge classification, while being least as wide as the resolution of the most important spectral bands in the dense woody vegetation cover model (Sentinel 2A/B SWIR2/3, 20 m).

The MSPA provides a pixel-level classification of an image. High-resolution dense woody vegetation cover imagery allows for flexibility in the definition of edge widths that would otherwise be limited with coarser products.

GuidosToolbox provides additional options for post-processing and extracting different network components identified as part of MSPA. Each of the nodes (i.e., cores) and links (e.g., bridges that connect cores) can be identified and assigned unique identifiers. Note that this process will include edge pixels that contribute to links, ensuring that there are contiguous connections between cores.

3.3 Results

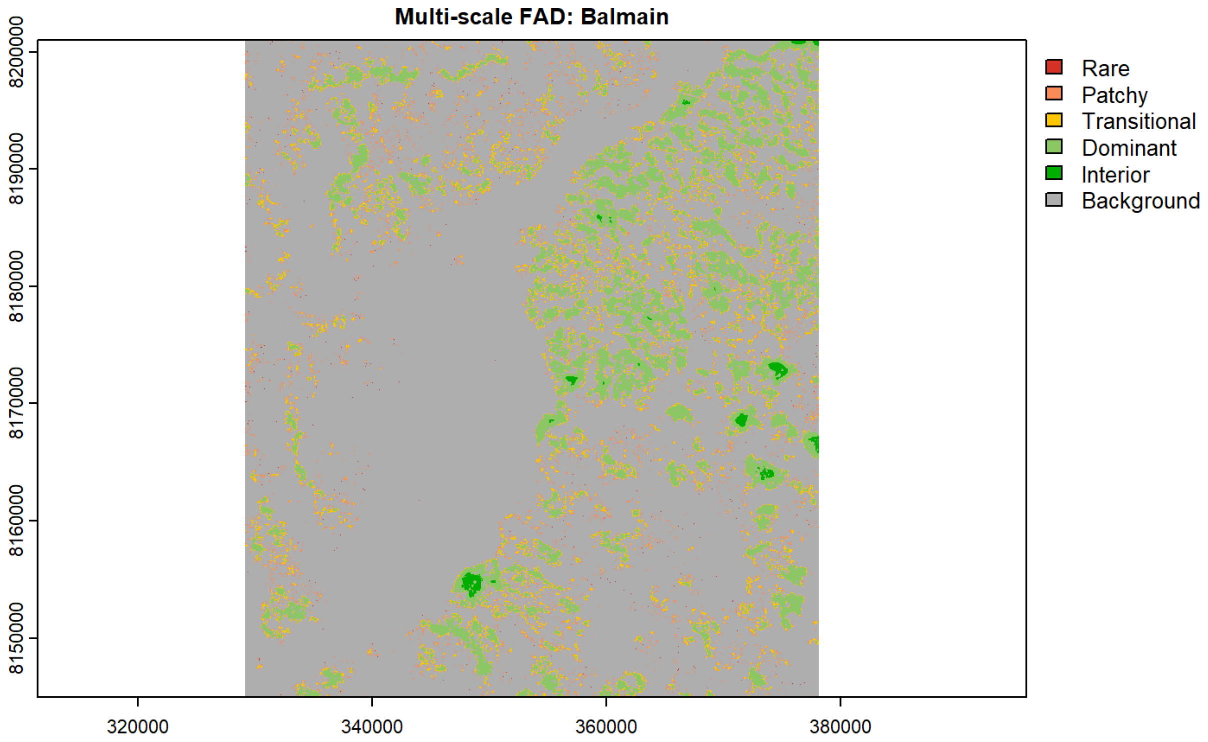
3.3.1 Fragmentation

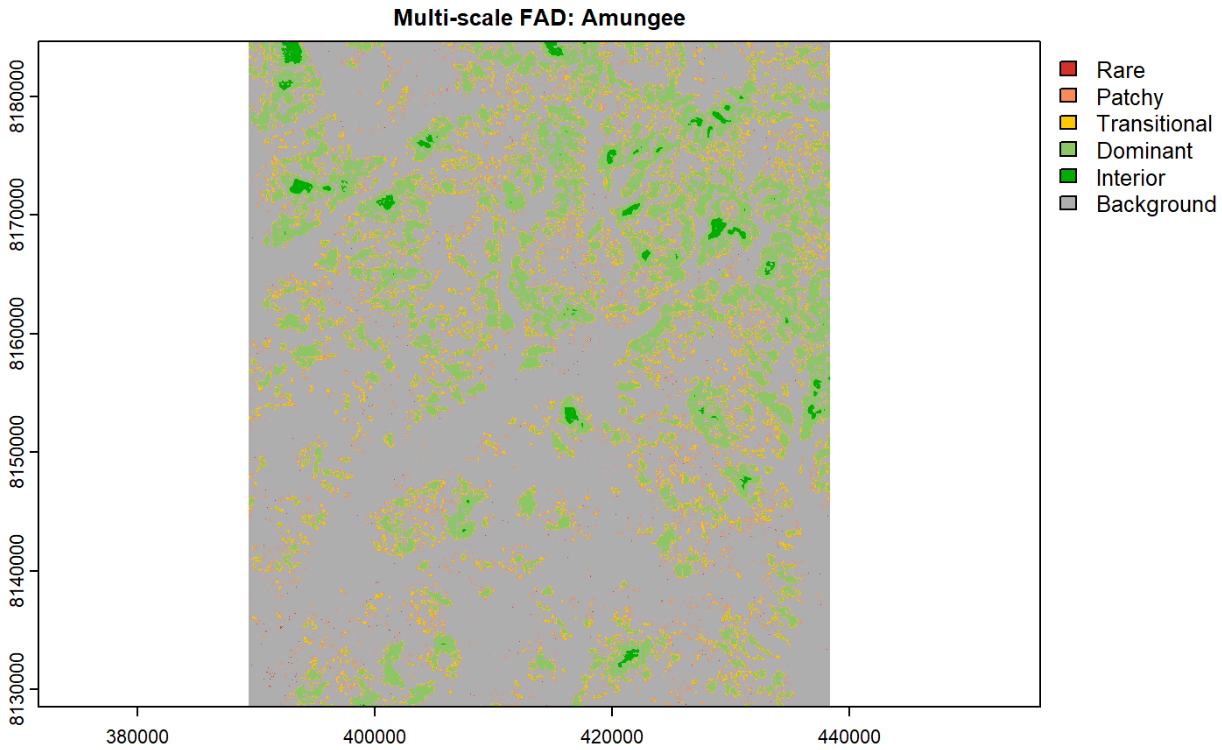
The four regions of the Beetaloo varied in their structure according to the FAD analysis (Figure 5). The majority of the area across all four regions was classified as Background meaning that the land did not support woody vegetation cover as defined in this study (i.e., > 20 % woody cover fraction and > 2 m height). Tanumbirini had the highest percentage of Background at 93.4%. It was lowest for the Amungee region (76.25%). No regions had any of the Intact class (i.e., areas where the FAD is 100%).

With the Background class removed, over 50% of the remaining land area was classed as Dominant for the Balmain, Amungee and Newcastle regions (Table 2). Tanumbirini region differed from the other three in having almost equal areas of Dominant, Transitional and Patchy.

Table 2 Comparison of the relative area (%) of FAD classes across four regions of the Beetaloo following exclusion of the Background class

Class	Balmain	Amungee	Tanumbirini	Newcastle
Rare	0.35	0.20	1.12	0.24
Patchy	14.90	10.97	27.50	6.75
Transitional	26.10	24.12	32.22	14.74
Dominant	57.02	61.90	38.13	64.16
Interior	1.63	2.81	1.03	14.10


Multi-scale FAD: Balmain
Rare
Patchy
Transitional
Dominant
Interior
Background
8200000
8190000
8180000
8170000
8160000
8150000
320000
340000
360000
380000


Multi-scale FAD: Amungee
Rare
Patchy
Transitional
Dominant
Interior
Background
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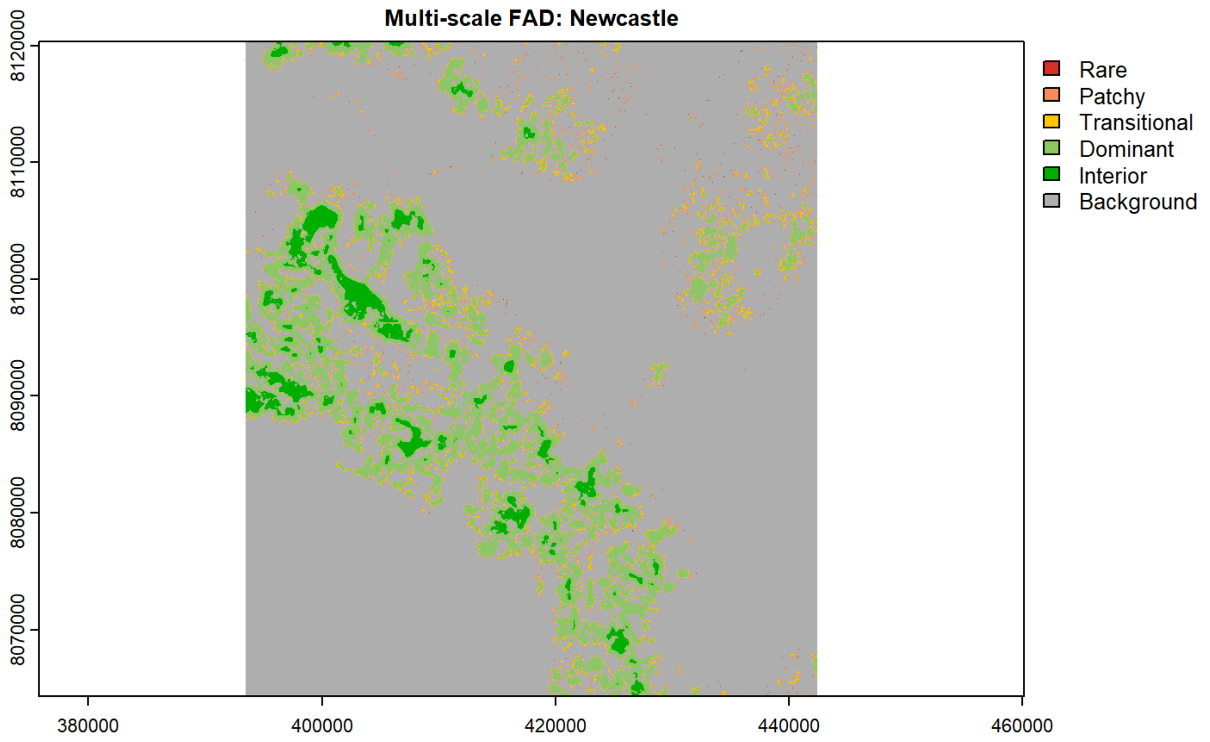
8170000
8160000
8150000
8140000
8130000
380000
400000
420000
440000



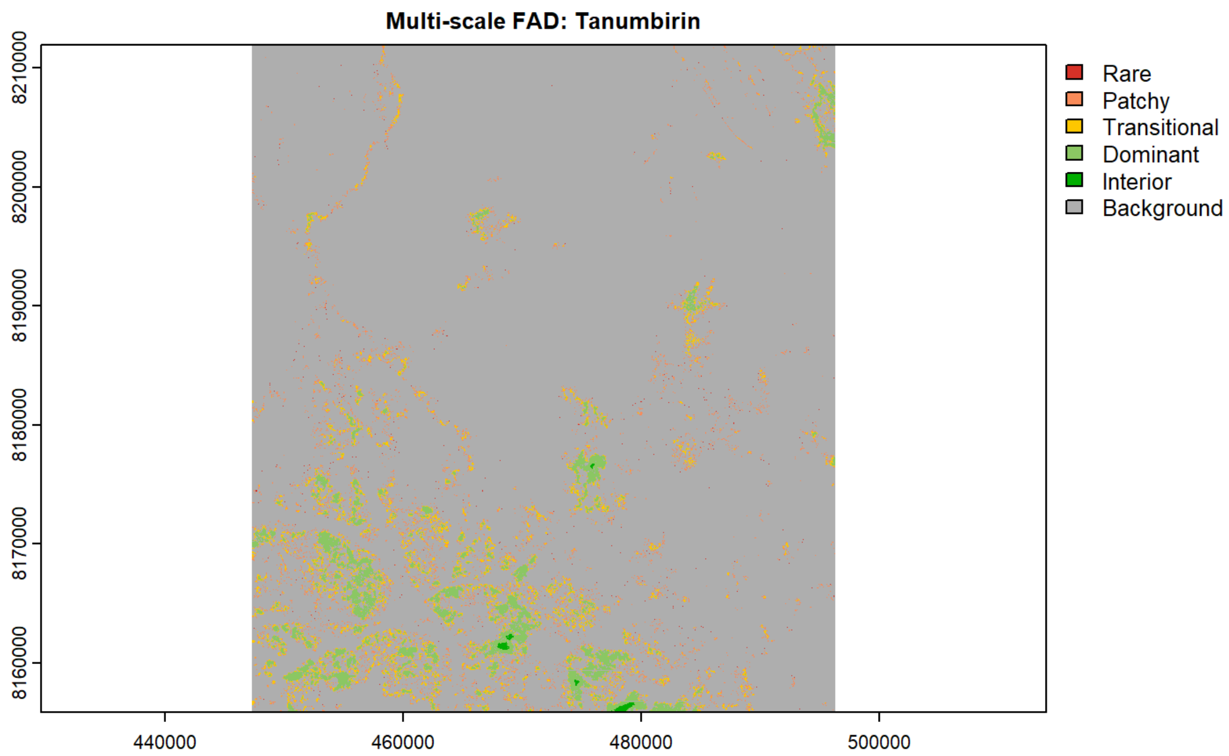


Figure 5 Maps of the FAD analysis for four regions of the Beetaloo; Balmain, Amungee, Tanumbirini and Newcastle

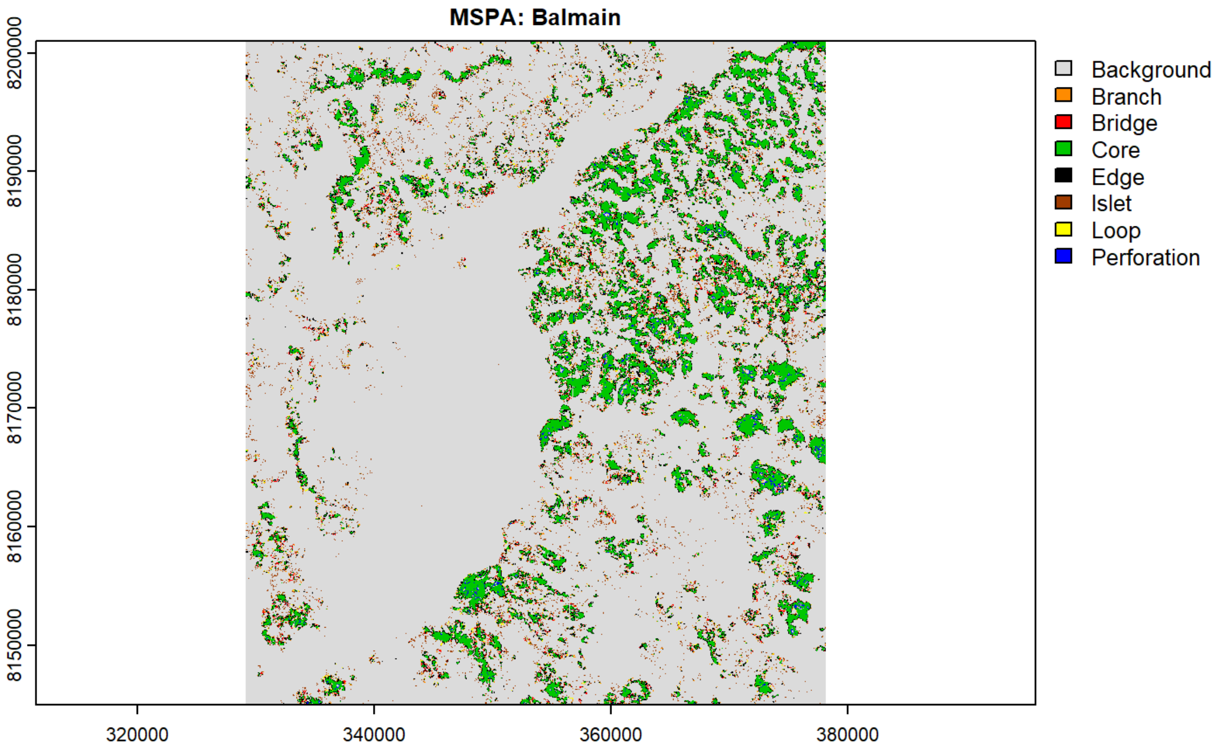
3.3.2 Connectivity

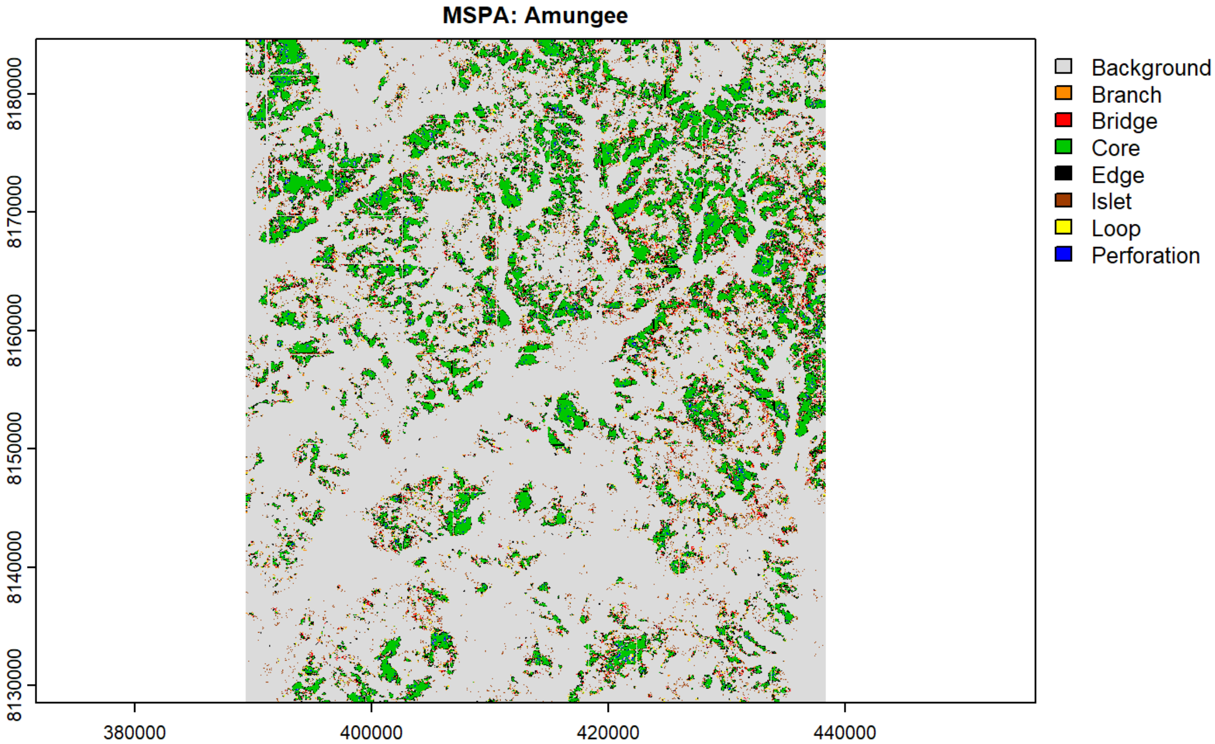
The pattern of connectivity of woody vegetation based on the MSPA analysis for each of the four regions of the Beetaloo is mapped in Figure 6. The scale of these maps is too large to show the details of connectivity that are available at a finer scale, therefore, additional maps are provided for a subset of the Balmain and Newcastle regions (Figure 7).

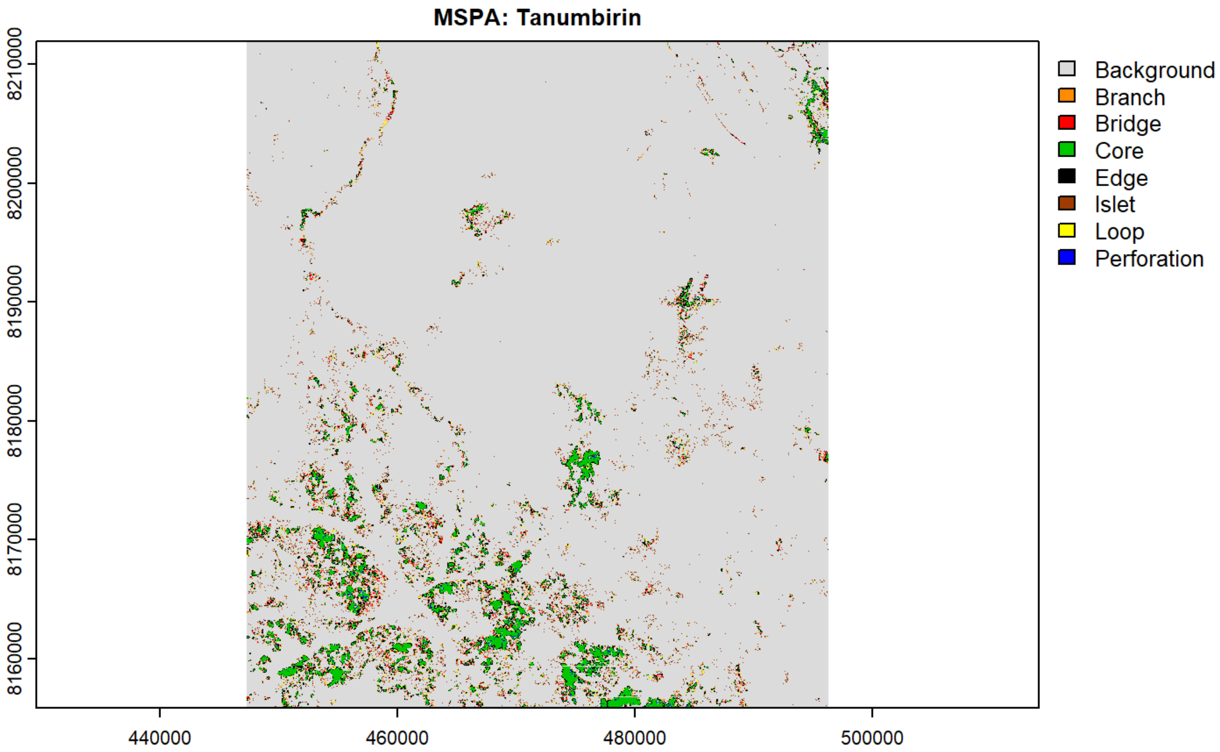
The majority of the area across all four regions was classified as Background and was excluded from subsequent assessment. With the Background class removed, Core was the dominant class for the Balmain, Amungee and Newcastle regions (Table 3). The Tanumbirini region again differed from the other three regions. It had a marginally higher proportion of the Edge class. The amount of the Core class was lowest for the Tanumbirini region (30.27%) and highest for the Newcastle region (58.16%).

Table 3 Comparison of the relative area (%) of MSPA classes across four regions of the Beetaloo following exclusion of the Background class

Class	Balmain	Amungee	Tanumbirini	Newcastle
Branch	8.14	7.58	11.28	5.11
Bridge	4.46	5.35	7.48	4.11
Core	45.08	46.04	30.27	58.16
Edge	30.02	30.49	30.89	22.44
Islet	7.97	6.14	16.05	4.49
Loop	2.56	2.84	3.25	2.93
Perforation	1.77	1.56	0.78	2.76


MSPA: Balmain
8200000
8190000
8180000
8170000
8160000
8150000
320000
340000
360000
380000
Background
Branch
Bridge
Core
Edge
Islet
Loop
Perforation


MSPA: Amungee
8180000
8170000
8160000
8150000
8140000
8130000
380000
400000
420000
440000
Background
Branch
Bridge
Core
Edge
Islet
Loop
Perforation



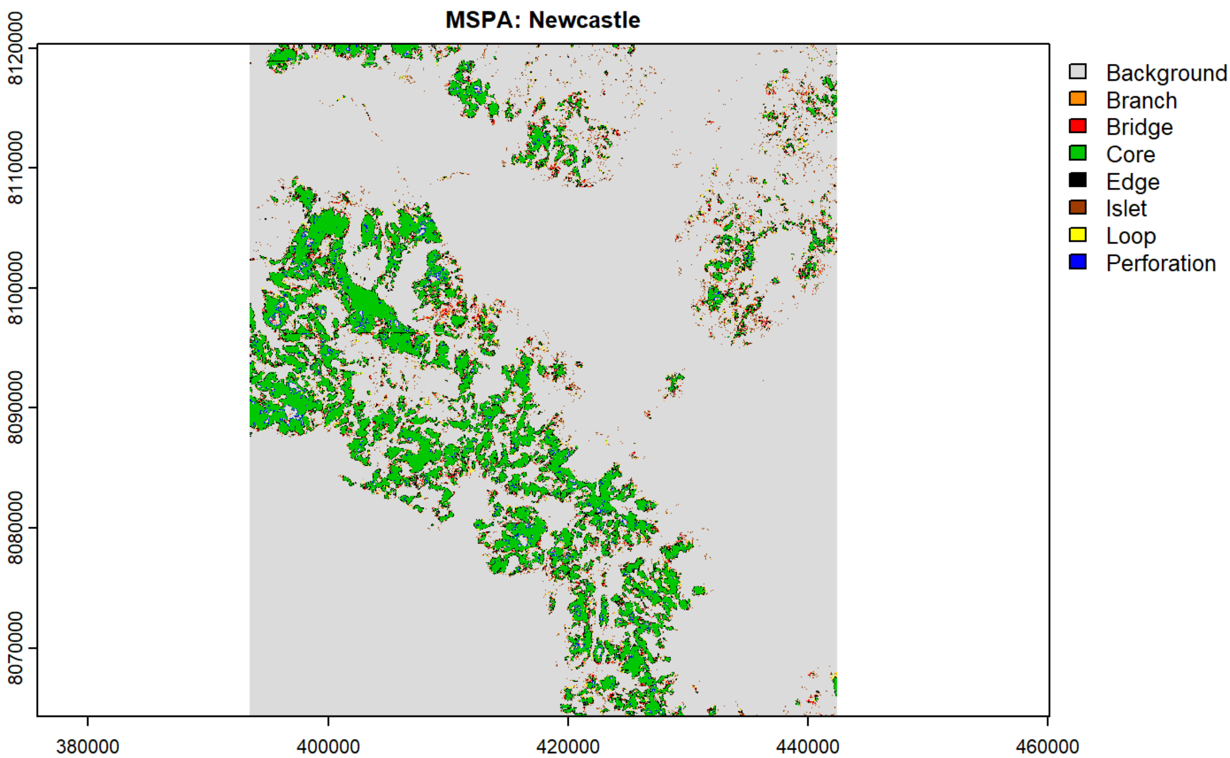


Figure 6 Morphological Spatial Pattern Analysis (MSPA) of a binary dense woody vegetation cover classification for four regions of the Beetaloo (from upper to lower): Balmain, Amungee, Tanumbirini and Newcastle

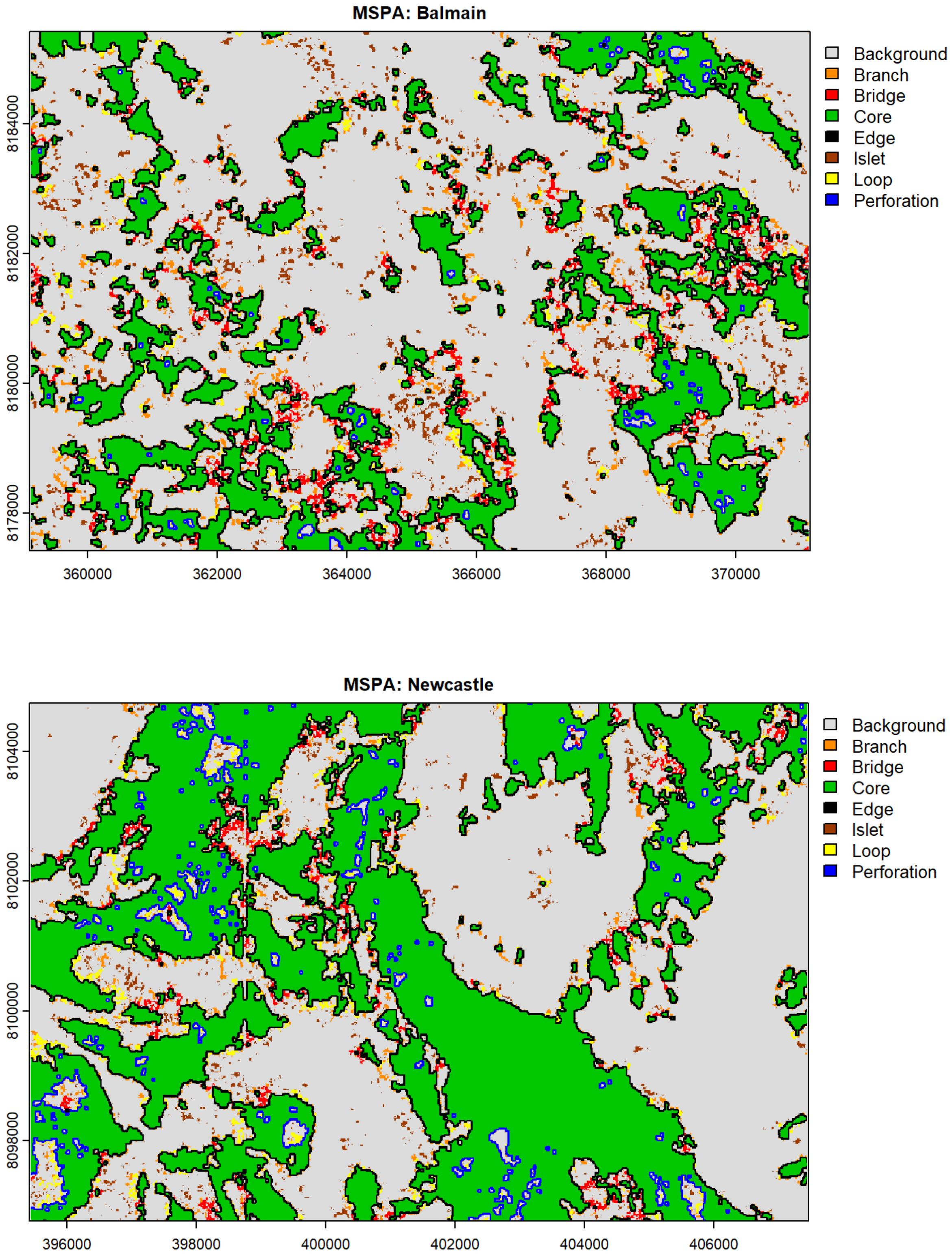


Figure 7 Morphological Spatial Pattern Analysis (MSPA) of a binary dense woody vegetation cover classification across a subset of the Balmain (upper) and Newcastle (lower) regions of the Beetaloo

3.4 Discussion

The current study has shown that indicators of fragmentation and connectivity, although designed to be applied to forest ecosystems, can also be applied to savanna and associated vegetation in northern Australia. Given the growing requirement to measure and monitor fragmentation and connectivity globally this is an important extension of the application of these indicators.

The methodology of this study involved defining woody vegetation as that with a woody cover fraction of > 20 % and a height of > 2 m. This definition included most non-grassland vegetation assemblages in the Beetaloo Sub-basin. It included 17 of the 20 broad vegetation types mapped in the Strategic Regional Environmental and Baseline Assessment (SREBA) for the Beetaloo Sub-region (Department of Environment, Parks and Water Security 2022, Figure 6-2). The vegetation types from the SREBA not covered under this definition are tussock grassland, ephemeral wetland, and lignum shrubland.

The current study has examined connectivity in a structural way. Specifically, it has interpreted the distribution of vegetation as a binary pattern and has then used mathematical morphometry techniques such as erosion and geodesic dilation to classify the binary pattern into discrete classes that represent the relationship of landscape areas to core habitat. At present this understanding has not been informed by data on movement patterns of wildlife. It is acknowledged that a structural connection does not imply a functional connection (Voigt et al. 2007). However, in the absence of functional information the structural approach used here is promising for monitoring change in connectivity over time. If required, the mapping provided using the approaches trialled here can be matched with information on the movement patterns of target species within the study area to assess the overlap between the two.

Once the development footprint of any future onshore gas development in the Beetaloo is understood, the connectivity mapping undertaken in this module can be used to assess potential impacts. For example, the proposed location of well pads, new roads, tracks, and pipelines can be superimposed on maps similar to those in Figure 7. This process will enable planners to identify the intersection of linear infrastructure with high value areas in the landscape including core habitat, bridges, and branches among other. Such areas can then be excluded from development.

Although excluded from woody vegetation as defined in this study, tussock grassland, ephemeral wetland, and lignum shrubland are all important vegetation associations in the Beetaloo Sub-basin. Each one is also potentially impacted by the effects of fragmentation. This situation should not be overlooked when considering fragmentation and connectivity in the study area.

4 Road mortality

4.1 Introduction

Roads, tracks, railways, pipeline corridors and other linear infrastructure are a significant cause of fragmentation of landscapes. Such linear infrastructure can create barriers to animal movement and can be a source of mortality of wildlife as a result of collisions with vehicles (van der Ree et al. 2015). In addition, linear infrastructure can facilitate human access to landscapes and can promote the spread of introduced animals and invasive plants.

Road ecology is an issue of emerging importance in conservation science. The prevalence of wildlife—vehicle collisions has been studied widely across the globe in recent years, however, it remains relatively understudied in Australia. In particular, there has been little relevant research undertaken in northern Australia.

The research covered in this section examined the issue of wildlife—vehicle collisions and the resulting mortality of wildlife in the Beetaloo Sub-basin. The rationale for the research was that an understanding was needed of the species that are currently killed on roads and in what numbers to determine whether any species or group of species is vulnerable to road mortality. This information would be a baseline against which projections could be made as to the increase in mortality of wildlife with enlargement of the road network (road widening, increased traffic volumes and increased extent of the road network). The research quantified the number of wildlife—vehicle collisions and the identity of the species killed on roads of differing width, traffic volumes and disturbance levels. The design was stratified across three levels of road usage and was replicated at each level. It is the first study of this type in the region.

4.2 Methods

4.2.1 Study area

The study was undertaken within the Beetaloo Sub-basin (Figure 7). It was focussed on an area centred on Daly Waters that had roads of varying size and traffic volumes. The modelling study of Bruce et al. (2021), that projected changes in road freight movements during the construction and operation phases of an onshore gas industry in the Beetaloo, was used to include roads that were expected to experience increases in traffic volume. As an example, the road freight density (number of trailers) along the Carpentaria Highway is projected to increase by 8,294 trailers during the second year of construction from a baseline of 17,166 (Bruce et al. 2021).

4.2.2 Survey methods – wildlife mortality

The design was stratified across three levels of road usage as follows:

- Primary highway (Stuart Highway),
- Secondary highway (Carpentaria Highway),

- Secondary road, currently unsealed (Buchanan Highway, Nutwood Downs Road).

Each of the three types of road had two replicate transects (Figure 8). Each of the six transects was 50 km in length. During a survey, the transect was driven at a set speed of 50 km/hour. Each individual vertebrate that had been killed or injured in a collision with a vehicle detected during a transect was identified to the lowest taxonomic level possible, marked and its location along the road recorded using GPS. Photographs were also taken of each road-killed animal. The identification of an individual animal to species became more difficult as it was progressively flattened and disintegrated by vehicular traffic or picked apart by scavengers.

Road transects were driven during three sampling periods that were chosen to capture variation in species occurrence with season. The seasons sampled were the early dry season (June 2022), late dry season (November 2022) and the wet season (February 2023). During each sampling session each transect was driven two to four times. At least 24 hours separated consecutive sampling events of an individual transect.

4.2.3 Survey methods – wildlife movement

The occurrence of live wildlife crossing roads was recorded while carrying out the driving transects and during other activities. In June 2022 focal animal sampling was undertaken along the Carpentaria Highway. Using this approach, if an animal was seen on the road, the vehicle was stopped nearby and two observers watched and recorded all species that were moving in the vicinity. This approach was particularly useful with mixed feeding flocks of birds.

An observation was considered to be a successful crossing if an individual moved from one side of the road to the other without being struck by a vehicle and if it did so while being vulnerable to vehicle strike. In other words, birds that flew across roads well above vehicle height were not recorded.

4.3 Results

4.3.1 Wildlife species recorded as roadkill

During the study we recorded a total of 411 killed or injured (as a result of collision with vehicles) native vertebrates (Table 4). The list included a total of 44 identified species spread among mammals (6 species), frogs (2 species), reptiles (17 species plus an individual from 1 additional genus that could not be identified to species), and birds (20 species plus an individual from 1 additional genus that could not be identified to species). In addition to native species, two introduced species, cane toad (*Rhinella marina*) and domestic cattle (*Bos taurus*), were also recorded.

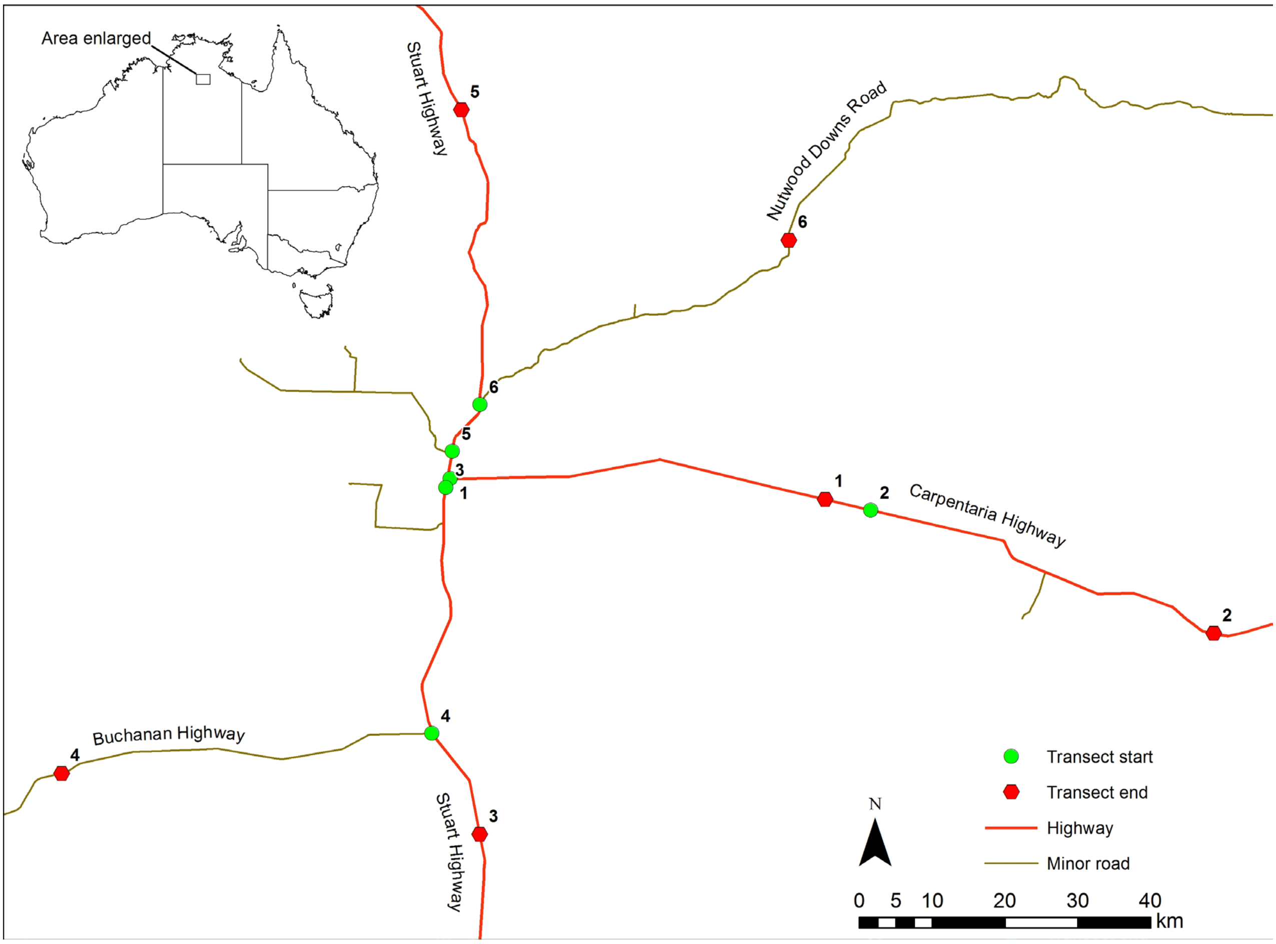


Figure 8 Map of the Beetaloo Sub-region showing the location of the six transects driven during the structured surveys of road mortality

Although 44 native species were located during the surveys, the majority of road-killed native vertebrates were macropods i.e., kangaroos and wallabies (N = 288 individuals). The species most often recorded as road kill were agile wallaby, northern nailtail wallaby and common wallaroo (Table 4). A total of 66 birds were recorded during the surveys with the commonest being the diamond dove (N = 18 individuals) followed by wedge-tailed eagle (N = 9 individuals). Reptiles contributed 41 individuals to the count and frogs 16 individuals.

A number of additional species were recorded as roadkill that were not observed during the structured surveys (i.e., the timed drives along the six 50 km transects). These species were noted during other research activities. The species are:

- Common brushtail possum, *Trichosurus vulpecula*
- Lesser long-eared bat, *Nyctophilus geoffroyi*
- Tawny frogmouth, *Podargus strigoides*
- Grey-crowned babbler, *Pomatostomus temporalis*.

Table 4 List and count of all vertebrate species (native and introduced) recorded as killed or injured on roads during the structured surveys along six 50 km transects in the Beetaloo Sub-region 2022-2023

Scientific name	Common name	Number
Frogs		
	Unidentified frog	1
Litoria caerulea	Australian green tree frog	5
Cyclorana australis	Giant frog	10
Rhinella marina[INTRODUCED]	Cane toad	1
Reptiles		
Chelodina canni	Cann's snake-necked turtle	1
Acanthophis rugosus	Rough-scaled death adder	1
Pseudonaja textilis	Eastern brown snake	2
Pseudonaja nuchalis	Northern brown snake	2
Pseudechis sp.	Unidentified black snake	2
Pseudechis weigeli	Pygmy mulga snake	1
Brachyuropis roperi	Northern shovel-nosed snake	2
Aspidites melanocephalus	Black-headed python	3
Antaresia childreni	Children's python	2
Liasis olivaceus	Olive python	4
	Unidentified agamid sp.	1
Amphibolurus centralis	Centralian lashtail	5
Diporiphora sp.		1
Pygopus nigriceps	Northern hooded scaly-foot	2
Ctenotus inornatus	Bar-shouldered ctenotus	1
Ctenotus robustus	Eastern striped skink	5

Tiliqua multifasciata	Centralian blue-tongued skink	1
Tiliqua scincoides intermedia	Northern blue-tongued skink	3
Varanus acanthurus	Spiny-tailed monitor	1
Varanus panoptes	Yellow-spotted monitor	1
Birds		
Ardeotis australis	Australian bustard	1
Corturnix sp.	Quail sp.	1
Turnix sp.	Button-quail sp.	1
Turnix castanotus	Chestnut-backed buttonquail	1
Burhinus grallarius	Bush stone-curlew	2
Aquila audax	Wedge-tailed eagle	9
Hamirostra melanosternon	Black-breasted kite	2
	Unidentified kite sp.	2
Milvus migrans	Black kite	2
Haliastur sphenurus	Whistling kite	5
Falco berigora	Brown falcon	1
Tyto javanica	Eastern barn owl	2
Eurostopodus argus	Spotted nightjar	1
Eolophus roseicapillus	Galah	1
Geopelia cuneata	Diamond dove	18
Geopelia striata	Peaceful dove	3
Phaps chalcoptera	Common bronzewing	2
Merops ornatus	Rainbow bee-eater	2
Malurus melanocephalus	Red-backed fairy-wren	1
Conopophila rufogularis	Rufous-throated honeyeater	2

Artamus cinereus	Black-faced woodswallow	4
Corvus orru	Torresian crow	1
Taeniopygia bichenovii	Double-barred finch	1
	Unidentified bird sp.	1
Mammals		
	Unidentified macropod	171
Lagorchestes conspicillatus	Spectacled hare-wallaby	2
Macropus agilis	Agile wallaby	65
Macropus antilopinus	Antilopine wallaroo	6
Macropus robustus	Common wallaroo	27
Macropus rufus	Red kangaroo	3
Onychogalea unguifera	Northern nailtail wallaby	14
Bos spp.[INTRODUCED]	Cattle	5

4.3.2 Seasonality in wildlife mortality on roads

The number of native vertebrates recorded as roadkill during the transects was similar in the early dry (N = 117) and wet seasons (N = 113). It was noticeably higher in the late dry season (N = 181).

Although the six species of macropods were the dominant group of wildlife recorded as roadkill during the surveys, the number of macropods killed varied across the three seasons. Macropods were the dominant group in the early dry and late dry seasons but were not frequently recorded during the wet season (Figure 9). Similar to macropods, birds of prey also showed a seasonal trend. Birds of prey including wedge-tailed eagle, whistling kite, and black kite were frequently seen feeding at carcasses, particularly macropod carcasses. These birds were killed following collisions with vehicles in the early and late dry seasons but not in the wet season (Figure 9).

Reptiles and birds other than birds of prey, made up a higher proportion of road kills in the wet season than at other times of year. Frogs were only recorded in the wet season.

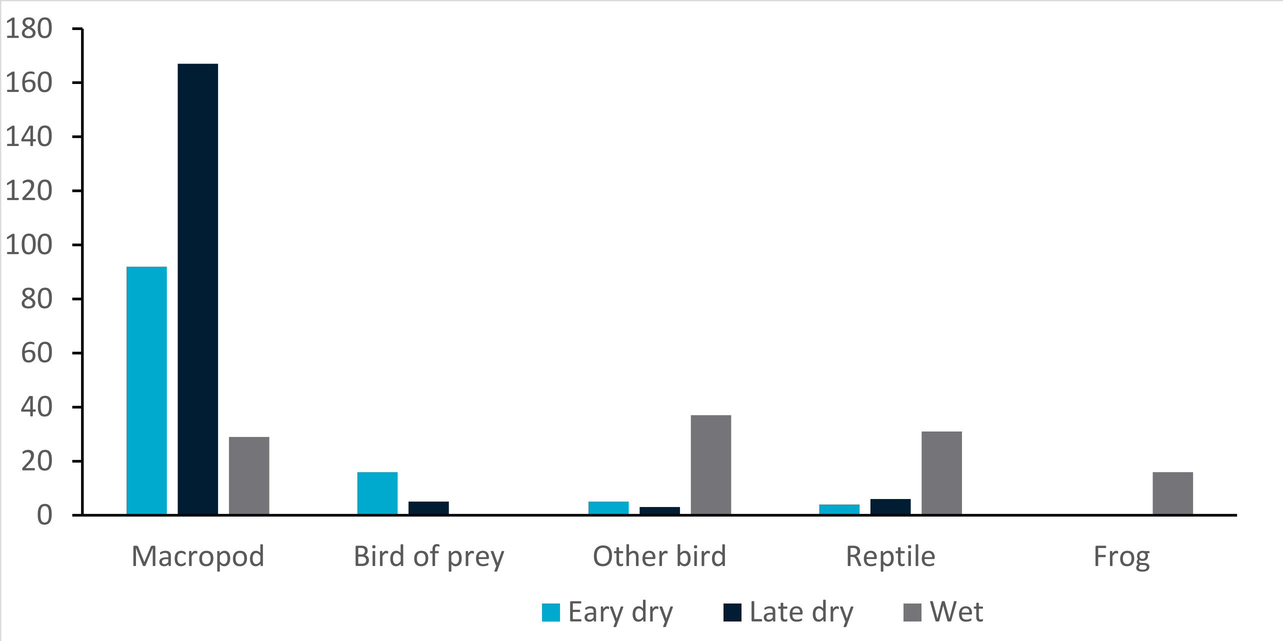


Figure 9 Variation in occurrence of wildlife groups as roadkill across seasons based on structured surveys in the Beetaloo Sub-basin 2022-2023

4.3.3 Variation in wildlife mortality across road type

The number of road-killed animals recorded during the structured surveys varied across the six transects (Table 5). The primary and secondary highways both had a large number of animals killed with the highest number on Carpentaria Highway transect #2 followed by the South Stuart Highway transect. Roadworks were taking place on Carpentaria Highway transect #1 during the late dry season and wet season sampling periods and there were speed limits in place along most of the transect (set at 60 km/h). As a consequence, the incidence of wildlife—vehicle collisions during these two sampling periods was much lower than in the early dry season (Figure 10). Therefore, the amount of roadkill in Table 5 for Carpentaria Highway 1 is likely to be misleading.

The strongest contrast in terms of the number of roadkilled wildlife across road type was between the highways and the secondary roads. Both secondary roads are unpaved and the number of wildlife—vehicle collisions was low (Table 5).

The pattern of a high rate of roadkill on highways and a low rate of roadkill on secondary roads was consistent across the three seasons sampled (Figure 10). The Carpentaria Highway transect #2 had the highest number of roadkilled animals in the early dry and late dry seasons; however, the number declined in the wet season (Figure 10). Carpentaria Highway #2 had a high proportion of macropods among its roadkilled wildlife and so it showed a decline in roadkill abundance in the wet season. In contrast, most roadkill of frogs and reptiles was along the two Stuart Highway transects and these showed an increase in the wet season compared to the early dry season (Figure 10).

Table 5 Summary of number of vertebrates recorded as roadkill on each of the six transects (sorted according to road class) during the structured surveys in the Beetaloo Sub-region 2022-2023

Road class	Transect name and number (as in Figure 8)	Number of injured or killed vertebrates recorded	Number of transects driven
Primary highway	North Stuart Highway (#5)	89	10
Primary highway	South Stuart Highway (#3)	124	10
Secondary highway	Carpentaria Highway 1 (#1)	52	10
Secondary highway	Carpentaria Highway 2 (#2)	132	10
Secondary road	Buchanan Highway (#4)	9	8
Secondary road	Nutwood Downs Road (#6)	5	8

4.3.4 Spatial distribution of roadkill

The spatial distribution of roadkill across the three seasons (Figure 11) suggests that there are no roadkill hotspots i.e., areas with a high concentration of wildlife—vehicle collisions (Figure 11). Rather the pattern is of a relatively even distribution of roadkilled vertebrates along the transects on the Stuart and Carpentaria Highways.

The spatial distribution of roadkill across the three seasons also reinforces the contrast between the four transects along highways and the two along secondary roads in terms of the number of roadkilled animals. In particular, both Stuart Highway transects and the eastern-most Carpentaria Highway transect had roadkill spread almost the entire length of the 50 km transect.

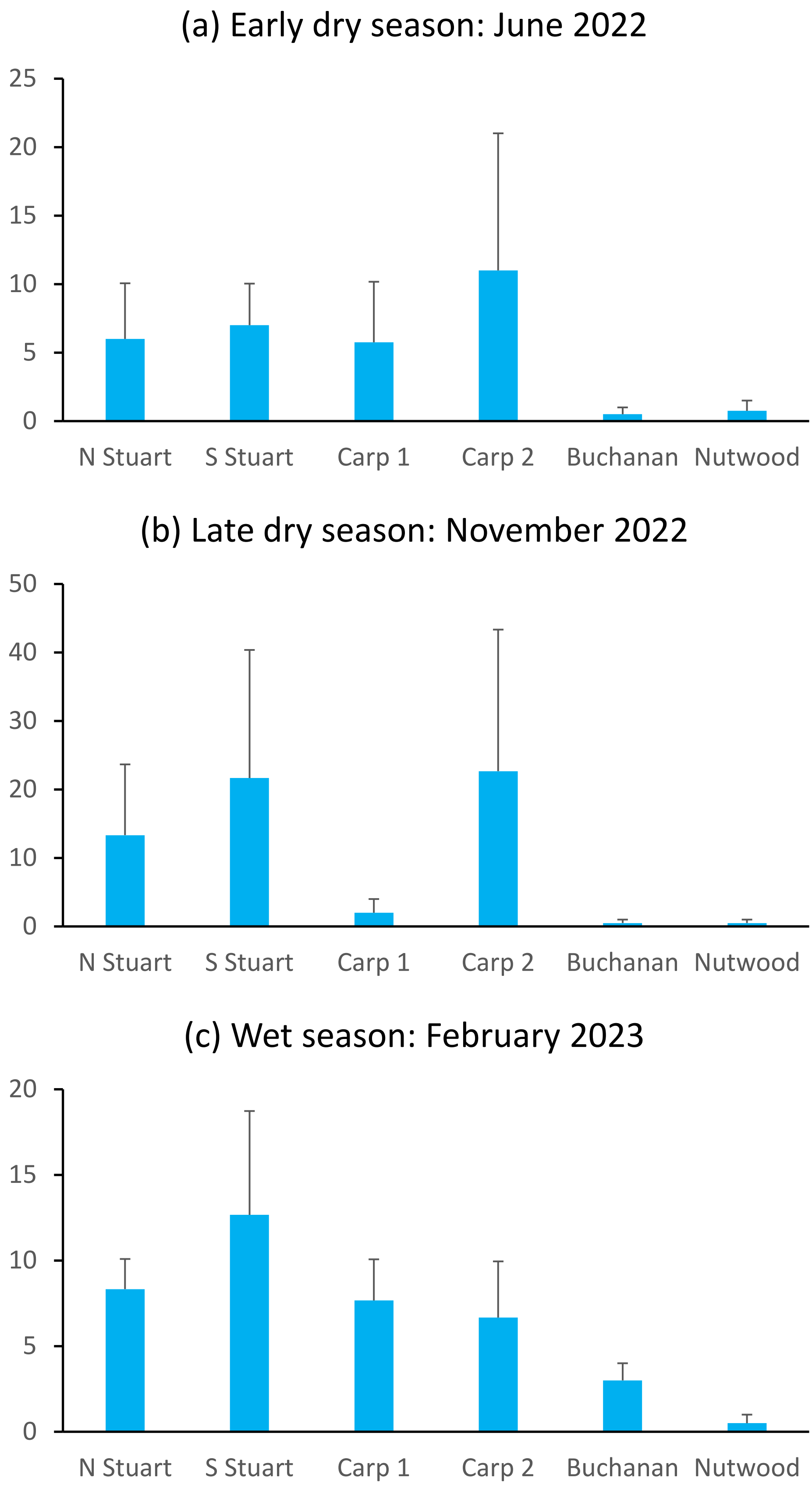


Figure 10 The number of roadkilled vertebrates + standard error recorded on six transects in the Beetaloo Sub-region in (a) early dry, (b) late dry and (c) wet seasons

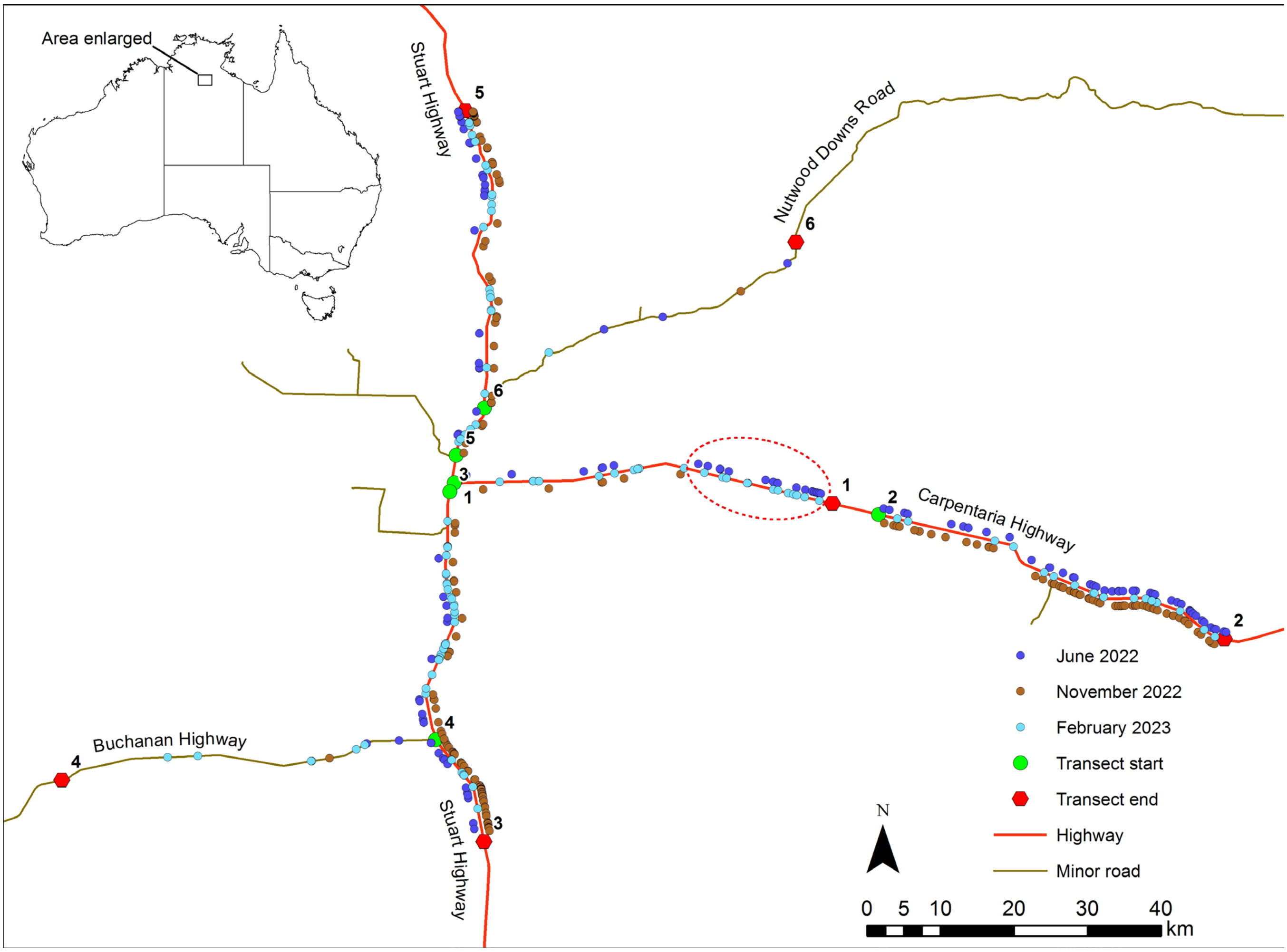


Figure 11 The geographic distribution of roadkilled vertebrates across the six transects in the Beetaloo Sub-region colour coded by sampling season. The red stippled area along the Carpentaria Highway shows the location of roadworks in the last two sampling periods

4.3.5 Wildlife movements

During the project a large number of observations were made by the study team of wildlife crossing roads (Table 6). These observations together with the identity of species recorded as roadkill (Table 4) suggest that most, if not all, species of vertebrates in the study area attempt (and often succeed) to cross roads of any size and traffic volume. As an example, the Stuart Highway was crossed successfully by species ranging in size from agile wallaby and black-headed python to brown quail, red-backed fairy-wren and eastern striped skink.

Table 6 Summary of observations of successful road crossing by vertebrates in the Beetaloo Sub-region 2022—2023

Scientific name	Common name	Road crossed		
		Stuart Highway	Carpentaria Highway	Secondary roads
Litoria caerulea	Australian green tree frog	×		
Cyclorana australis	Giant frog	×		
Acanthophis rugosus	Rough-scaled death adder		×	
Pseudonaja nuchalis	Northern brown snake			×
Pseudechis australis	Mulga snake		×	
Demansia papuensis	Papuan whipsnake	×		
Aspidites melanocephalus	Black-headed python	×		×
Amphibolurus centralis	Centralian lashtail	×	×	×
Chlamydosaurus kingii	Frilled lizard	×	×	×
Ctenotus robustus	Eastern striped skink	×		
Tiliqua scincoides intermedia	Northern blue-tongued skink	×	×	×
Varanus acanthurus	Spiny-tailed monitor			×
Varanus gouldii	Sand goanna			×
Varanus panoptes	Yellow-spotted monitor	×	×	×
Varanus tristis	Black-headed monitor		×	
Birds				
Antigone rubicunda	Brolga		×	
Ephippiorhynchus asiaticus	Black-necked stork		×	
Coturnix ypsilophora	Brown quail	×	×	
Elseyornis melanops	Black-fronted dotterel		×	
Vanellus miles	Masked lapwing	×		

Aquila audax	Wedge-tailed eagle	×	×	
Milvus migrans	Black kite	×	×	
Haliastur sphenurus	Whistling kite	×	×	
Falco berigora	Brown falcon	×	×	×
Eurostopodus argus	Spotted nightjar		×	
Eolophus roseicapillus	Galah	×	×	×
Geopelia cuneata	Diamond dove	×	×	×
Geopelia striata	Peaceful dove	×	×	×
Geopelia humeralis	Bar-shouldered dove	×	×	×
Phaps chalcoptera	Common bronzewing	×	×	×
Ocyphaps lophotes	Crested pigeon	×		
Cacomantis pallidus	Pallid cuckoo	×		
Centropus phasianinus	Pheasant coucal	×	×	×
Todiramphus pyrrhopygius	Red-backed kingfisher		×	
Todiramphus sanctus	Sacred kingfisher	×	×	×
Dacelo leachii	Blue-winged kingfisher		×	×
Merops ornatus	Rainbow bee-eater	×	×	×
Malurus melanocephalus	Red-backed fairy-wren	×	×	
Lichmera indistincta	Brown honeyeater	×	×	×
Conopophila rufogularis	Rufous-throated honeyeater	×	×	×
Manorina flavigula	Yellow-throated miner	×		
Pomatostomus temporalis	Grey-crowned babbler	×	×	×
Lalage tricolor	White-winged triller	×	×	×
Coracina novaehollandiae	Black-faced cuckoo-shrike	×	×	×
Struthidea cinerea	Apostlebird	×		

Gymnorhina tibicen	Australian magpie		×	
Artamus cinereus	Black-faced woodswallow	×	×	×
Artamus personatus	Masked woodswallow		×	
Corvus orru	Torresian crow	×	×	×
Chlamydera nuchalis	Great bowerbird	×	×	×
Grallina cyanoleuca	Magpie-lark	×	×	×
Rhipidura leucophrys	Willie wagtail	×	×	×
Melanodryas cucullata	Hooded robin		×	
Colluricincla harmonica	Grey shrike-thrush		×	
Oreoica gutturalis	Crested bellbird		×	
Poephila personata	Masked finch	×	×	×
Poephila acuticauda	Long-tailed finch	×	×	×
Mammals				
Tachyglossus aculeatus	Echidna		×	
Lagorchestes conspicillatus	Spectacled hare-wallaby		×	
Macropus agilis	Agile wallaby	×		
Macropus antilopinus	Antilopine wallaroo			×
Onychogalea unguifera	Northern nailtail wallaby			×
Saccolaimus flaviventris	Yellow-bellied sheathtail-bat	×	×	
Nytophilus geoffroyi	Lesser long-eared bat		×	

4.4 Discussion

4.4.1 General Discussion

The research in this module on road mortality of vertebrates in the Beetaloo Sub-region recorded a relatively high number of dead or injured native animals (N = 411) across the structured surveys compared to previous studies in eastern Australia (e.g., Taylor and Goldingay 2004, Matthews 2020). A wide range of species (N = 44) was killed following collisions with vehicles including mammals, frogs, reptiles, and birds. The size of animals killed ranged from male red kangaroos (with body mass of about 90 kg) to small birds such as the double-barred finch (body mass of about 7 g). Despite the species richness of the roadkill sample, six species of macropods (kangaroos and wallabies) dominated the sample numerically making up 70% of all roadkill.

The number of macropods killed varied markedly across the three seasons. Macropods were the dominant group in the early dry and late dry seasons but were not frequently recorded during the wet season (Figure 9). A similar pattern was shown for the birds of prey that fed at macropod carcasses on roads and were regularly secondary roadkill in the early dry and late dry seasons (Figure 9). This strong seasonal pattern reflects changes in the availability of food for macropods with variation in rainfall. In particular, macropods did not congregate along roadsides during the wet season as grass and other food plants were widely available across the landscape.

Reptiles, and other birds (i.e., non-birds of prey) made up a much higher proportion of road kills in the wet season than at other times of year. Frogs were only recorded in the wet season. This change reflected the presence of temporary sources of standing water beside roads (Figure 12) in the wet season, in areas that are typically devoid of water through the remainder of the year. Such temporary wetlands provide breeding habitat for frogs and also for aquatic reptiles including the freshwater turtle, *Chelodina canni*. Some of these individuals disperse across the landscape in search of suitable habitat and may cross roads at such times. Also in the wet season, seeding grass was present in abundance beside all roads including the Stuart and Carpentaria Highways often growing up to the edge of the bitumen (Figure 13). The grass provided food for granivorous birds such as quail and finches as well as habitat for invertebrates and their predators including skinks and birds such as fairy-wrens. Grass cover reduces significantly as the dry season progresses with a resultant decline in small vertebrate activity close to the roads.

An important finding from this study is that the secondary roads sampled had a low rate of wildlife roadkill. These roads were unpaved although they did have high speed limits in place (up to 100 km/h). The primary explanation for the difference in number of roadkilled animals across road types is the volume of traffic. Baseline annual road freight density (number of trailers) provided by Bruce et al. (2021, see Figures 5 and 8 of their report) for the portions of the roads included in our survey transects are as follows:

- Stuart Highway, north of Daly Waters: 55,164 trailers,
- Stuart Highway, south of Daly Waters: 38,118 trailers,
- Carpentaria Highway: 17,166 trailers,
- Buchanan Highway: 578 trailers,
- Nutwood Downs Road: 240 trailers.

It follows from this result that increased traffic volumes on any of these roads, as is predicted for the Stuart and Carpentaria Highways during the construction and operation phases of an onshore gas industry (Bruce et al. 2021), will result in an increase in wildlife—vehicle collisions and the resulting mortality of wildlife. Focussing on the Carpentaria Highway, the baseline annual traffic volume is 17,166 trailers; however, this is expected to increase by the following amounts:

- 5,970 trailers during construction year 1,
- 8,294 trailers during construction year 2,
- 1,440 during each year of operation years 1 to 5, and
- 3,012 trailers during each year of peak operation (~years 6 to 20).

The expectation from this study is that there will be an increase in the amount of roadkill along the Carpentaria Highway during the life of an onshore gas industry in the Beetaloo Sub-region.

This study did not detect any areas where there was a concentration of wildlife mortality following collisions with vehicles. The mapping of the 411 individuals road-killed did not detect any roadkill hotspots, although some roads had a higher rate of roadkill than others (Figure 11). Such hotspot locations typically occur where geographic features such as rivers or steep hills channel the movement of animals. Given the nature of the terrain that is dominated by the Sturt Plateau, a low elevation (<200 m ASL) undulating plain that does not have dramatic geological features (e.g., cliffs, hills, rivers, lakes), the lack of hotspots in the study area is not surprising.

The two species most often recorded as roadkill during the study were macropods: agile wallaby, 65 individuals, and common wallaroo, 27 individuals. With the exception of macropods, no species appeared particularly susceptible to collisions with vehicles. We recorded a total of 18 diamond doves killed by vehicles during the study but this species was very abundant at times and was often present beside roads. The following species that are listed as threatened in the Northern Territory under the *Territory Parks and Wildlife Conservation Act* were recorded as roadkill in this study (during structured surveys and incidentally):

- Australian bustard,
- Bush stone-curlew,
- Northern nailtail wallaby,
- Spectacled hare-wallaby,
- Yellow-spotted monitor,
- Common brushtail possum.

The incidence of roadkill of each of these species was low except for the northern nailtail wallaby (N = 14 individuals).

Figure 12 Temporary wetland beside the Carpentaria highway occupied by frogs during the wet season, February 2023

Figure 13 The Carpentaria Highway during the wet season (February 2023) showing the presence of grass up to the edge of the bitumen

4.4.2 Mitigation measures

Ideally, developing mitigation options for an onshore gas industry in the Beetaloo Sub-basin (or any other region) requires an understanding of the development scenario for that project. The development scenario includes information on the location and extension of roads, and construction of tracks, pipeline corridors and well pads. This information can be used to identify 'at risk' habitats and to plan approaches to ameliorate impacts. At present (May 2023), this information is not available for the Beetaloo. As a consequence, the approach taken here is to consider approaches to mitigate impacts from existing roads on the expectation that impacts will be similar (although larger in magnitude) should development of an onshore gas industry occur.

Wildlife crossing structures

The primary approach to mitigate road mortality of wildlife is to design and construct wildlife crossing structures (also referred to as animal passage structures). Overall, these structures can be used to maintain, or restore, connectivity for wildlife across the landscape (Smith et al. 2015). Various designs are available and the approach used depends on the ecology of the target species. Bridges and similar canopy structures are available for arboreal species (possums, gliders) whereas underpasses and overpasses function for terrestrial animals. Although the data collected in this study provide only preliminary findings, it is sufficient to indicate that wildlife crossing structures are unlikely to be necessary in the study area. The reasons for this conclusion are a lack of evidence of hotspots of road mortality and the lack of a target species, or group of species, that is likely to benefit sufficiently at the local population level from the presence of crossing structures to justify the costs.

The only areas that may benefit from the construction of wildlife crossing structures are major drainage lines that cross the Stuart and Carpentaria Highways. Examples of such sites are where Balmain Lagoon crosses the Stuart Highway south of Daly Waters and the channel of Newcastle Creek across the Stuart Highway north of Elliott. Large numbers of frogs and reptiles are likely to suffer mortality from vehicles in these areas during the wet season. In these cases, underpasses can facilitate movement away from the road surface. A suitable option is likely to be to retrofit or modify existing road structures that were originally designed for other purposes (Smith et al. 2015) to facilitate frog and reptile movement under roads. The most appropriate structures for such retrofitting are drainage culverts. These culverts are widely used by wildlife and the effectiveness could be increased by adding designs such as bolt-on metal shelves (see Fig. 10.1 of Smith et al. 2015).

A range of non-crossing approaches are also available to mitigate the impacts of roads. These include altering vehicle traffic patterns or driving behaviour, and placing structures (barrier fences, walls) on the roadside. These approaches are discussed below.

Road signs

The placement of relevant signs warning of wildlife crossing the road can be used to increase awareness among motorists. Such signs are inexpensive and relatively easy to place. The signs function to reduce the rate of collisions and the severity of collisions that do occur; however, they do not reduce the barrier effects of roads (Huijser et al. 2015). Road signs are used most widely for large mammals and birds and, in Australia, are most often seen in areas where collisions with macropods occur. The effectiveness of road signs in reducing wildlife mortality in the long term has not been studied.

Signage will only be effective if it results in motorists changing their driving behaviour to avoid collisions with wildlife. A number of design features are considered likely to increase the effectiveness of wildlife warning signs on roads. These include the following: (a) novel designs, (b) selective placement in locations where they will have the most impact on reducing wildlife collisions if drivers modify their behaviour, (c) specific wording that contains relevant information, and (d) temporary placement at the times of year when they are most useful (Jackson et al. 2015).

Road signs are not currently used within the study area. This is somewhat surprising especially where the Carpentaria Highway cuts through the Bullwaddy Conservation Reserve.

Vehicle speed limits

Vehicle speed is generally considered to be a factor in collision rate with wildlife; higher speeds are often correlated with higher numbers of collisions (Jackson et al. 2015). A vehicle moving at a lower speed than normal can reduce the probability of collision with wildlife because: (a) it can increase the detection of wildlife on the road or on road verges, (b) it gives an individual animal increased time to respond and avoid a vehicle, and (c) it gives the driver extra time to change course and avoid an animal.

Vehicle speed limits to avoid vehicle wildlife collisions are not currently used within the study area. This is even the case within the Bullwaddy Conservation Reserve.

Fencing

Fencing (also referred to as barrier fencing or exclusion fencing) is an approach to reduce wildlife—vehicle collisions that comes from livestock management where fences are used to prevent farm animals from straying on to roads. The main purpose of fencing is to prevent animals from accessing a road and, potentially also, to funnel animals to a crossing structure (van Der Ree et al. 2015). In the latter case the fencing is referred to as funnel fencing. Fencing is widely used across the globe. It was first developed to prevent large mammals from accessing roads and causing serious traffic incidents in collisions with vehicles; however, it is now used for a range of wildlife of differing size. Fencing works most effectively if it is placed in hotspot locations of movement that involve road crossing. In such situations the length of fencing can be minimised while ensuring reductions in collisions.

Fencing is most optimally used in combination with crossing structures such as overpasses or underpasses. In the absence of crossing structures barrier fences prevent animal movements and can result in fragmentation and loss of connectivity (Jackson et al. 2015). Reduced connectivity can result in population crashes and eventual local extinction (van der Ree et al. 2015). The end point of barrier fences can also be hotspots of wildlife—vehicle collisions as animals move across the road in concentrated numbers.

Fencing may be of some value in limited areas of the Beetaloo Sub-region. As mentioned previously for crossing structures, drainage lines that cross the Stuart and Carpentaria Highways could benefit from fencing. Also, along these highways during the wet season numerous species are attracted to water that pools on the edge of roads. If any of these are shown to be areas where large numbers of individuals are roadkilled then fencing should be considered.

Habitat modification

Habitat modification is a potential mitigation option for species of wildlife that are attracted to the edge of roads because of the availability of resources such as food (e.g., fresh herbage in run-on areas) and breeding sites (e.g., reptiles using uncompacted soil for nesting). If the presence of these animals close to roads results in their being involved in collisions with vehicles, then the benefits from the available resources may be outweighed by the negative effects on survivorship (Jackson et al. 2015). The process of habitat modification involves changing the habitat conditions along the sides of roads to remove the resources that attract wildlife. Management of roadside vegetation is one mechanism to achieve this goal.

Within the study area, numerous species are attracted to water that pools on the edge of roads and to fresh herbage on roadsides. Foremost among these species are macropods (kangaroos and wallabies). The high incidence of macropod roadkill is a reality across most of eastern Australia (Matthews 2020). In the Beetaloo Sub-region, a number of the macropod species recorded as roadkill are either threatened under Northern Territory legislation (spectacled hare-wallaby, northern nailtail wallaby) or potentially undergoing local declines (antilopine wallaroo).

Habitat modification to prevent wildlife—vehicle collisions in the Beetaloo could be as simple as periodically clearing vegetation from the edges of the Stuart and Carpentaria highways during the wet season. There are insufficient data from this study to suggest more specific locations for habitat modification; however, this is an issue that should be revisited during the planning phase of an onshore gas industry.

Flight diversion structures

A proportion of the wildlife killed on roads during the current study was birds (10.95% of 411 individuals). Despite being able to fly, birds are still vulnerable to collisions with vehicles (Kociolek et al. 2015). The other group of flying vertebrates, bats, are also vulnerable to collisions with vehicles (Abbott et al. 2015). Millions of birds and bats are killed on roads each year.

Flight diversion structures are structural elements built to encourage birds to fly above or below the traffic on roads. These work best for birds that have fast and direct flight (Kociolek et al. 2015). Examples include poles that produce an illusion of a solid barrier, flags, wider posts, fencing with flagging attached, and solid walls beside roads. Bat species are also known to use underpasses.

There are insufficient data from this study to support the use of flight diversion structures along roads in the Beetaloo.

5 Patch dynamics

5.1 Introduction

Fragmentation of habitat is a major threat to biodiversity worldwide. It is a process that involves both habitat loss and a change in the configuration of the remaining habitat. Habitat fragmentation can lead to changes in the persistence of species. Fragmentation often alters the microenvironment at the edge of the fragment and can result in increased light levels, higher daytime temperatures, higher wind speeds, and lower humidity. Each of these edge effects can have a significant impact on the vitality and composition of the species in the fragment (Primack and Morrison 2013). Species that are sensitive to humidity such as amphibians, some insects, and herbaceous plants may be eliminated from fragments in response to these edge effects.

The Beetaloo Sub-basin contains a wide range of vegetation communities (Department of Environment, Parks and Water Security 2022) including savanna woodlands, grassland and several sensitive vegetation communities (i.e., lancewood and bullwaddy woodland, open forest and thickets). Lancewood and bullwaddy vegetation support species of plants and animals that are sensitive to disturbance. These communities potentially support the highest concentrations of species 'at risk' from linear corridors and are the vegetation communities most exposed to the disruption of important ecological processes and to threats including weed invasion and fire suppression. They are ecologically productive and may represent seasonal refuges for wildlife. These communities rely on large patch sizes to provide resilience to wildfire and, thus, are considered to be vulnerable to the effects of fragmentation.

The focus of this module of the project was the documentation of the occurrence and relative abundance of vascular plant and vertebrate species in lancewood and bullwaddy patches of differing area and degree of spatial isolation. To do this, full floristic inventory and vegetation structural sampling was undertaken in 24 survey sites that were sampled to address the vertebrate component of the module. A range of analytical techniques were applied to the dataset to test for floristic and functional and structural trait differences in the replicated samples of small, medium, and large patches.

5.2 Methods

5.2.1 Study sites

The study was located in the Beetaloo Sub-basin, approximately 40 km east of Daly Waters, south of the Carpentaria Highway on Hayfield-Shenandoah Station (Figure 14). The 24 sites were situated along two closely positioned transects, one running predominantly north-south (Sites 1-12) and the other running predominantly east-west (Sites 13-24) (Figure 14).

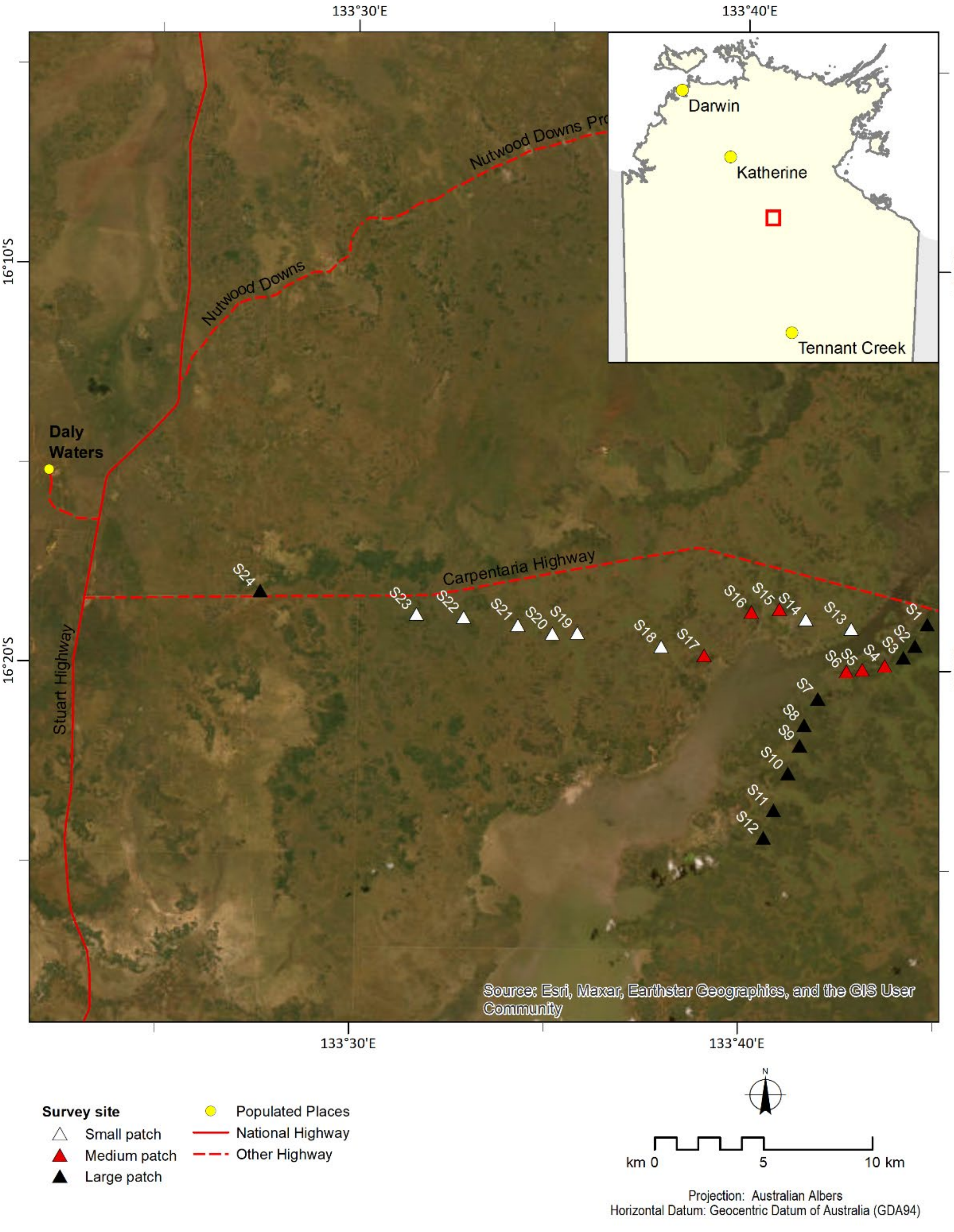


Figure 14 Location of the study area east of Daly Waters in the Beetaloo Sub-basin and positioning of sites placed in small (grey triangles), medium (red triangles) and large (black triangles) patches of bullwaddy/lancewood woodland

Fire frequency in the study area ranged from very low to moderately high (zero times to once every two years) over the preceding two decades (Figure 15). Lancewood and bullwaddy woodland patches are both less fire prone and less fire tolerant than the surrounding savanna matrix. Lancewood (*Acacia shirleyi*) itself has very high fire vulnerability. Adult trees can withstand low-severity fire, but they will be killed by fire in the event of complete canopy scorch. This species is also vulnerable to fire due to the long primary juvenile period (time to maturity from seed) relative to average fire-return intervals (Russell-Smith et al. 2010). Due to this sensitivity, lancewood vegetation only persists in this landscape in areas of relatively low fire frequency, where it then suppresses ground-fuel with high canopy and litter cover. The bullwaddy dominant (*Macropteranthes kekwickii*) differs in that it, along with many of the rainforest elements that characterise this vegetation type, has the capacity to resprout after fire by virtue of fire-protected stem-buds (Schubert et al. 2016). The extent to which these species can withstand short-interval, hot dry-season fire is undocumented, but in all likelihood, their fire tolerance is low compared to that of trees and shrubs in neighbouring savanna vegetation. Moreover, fire avoidance due to exceptionally low ground-fuel loads appears key to its persistence in the otherwise highly fire-prone landscape in which it occurs.

Within the study area (Figure 15), the south-east experiences high fire frequency (every two years, bright yellow shading); however, field observation confirmed that fire does not always carry into the large bullwaddy/lancewood stands along the adjacent north-south transect (and close inspection reveals that the location of the study sites mainly coincide with blue (low fire frequency, e.g., zero times in two decades) or pale-yellow (moderate fire frequency, e.g., every five years) shading. The large area of blue shading to the left of this transect represents the low fire frequency of the adjacent coolibah clay floodplain. The east—west transect sites are shown to experience relatively high fire frequency. Field observation confirmed the high rates of fire incidence at patch boundaries and in areas of *A*. *shirleyi* in particular. Bullwaddy fire exposure was found to be relatively rare, but the fire mapping scale was too course to detect the observed difference.

5.2.2 Study design

Sampling was stratified across three patch sizes: ten large patch sites; six medium patch sites; and eight small patch sites (Figure 14). The 24 sites were situated along two closely positioned transects, one running predominantly north—south (Sites 1-12) and the other running predominantly east—west (Sites 13-24) (Figure 14). The north—south transect was dominated by large patch sites (nine sites) with the remaining three sites being in medium patches. The east—west transect included all of the eight small patches along with one large patch and three medium patches. The positioning of the sites was strongly determined by the availability of patches in each size class. While the allocation of small patch sites to both transects would have been preferable, there was no option to do this due to the absence of small patches along the north—south transect.

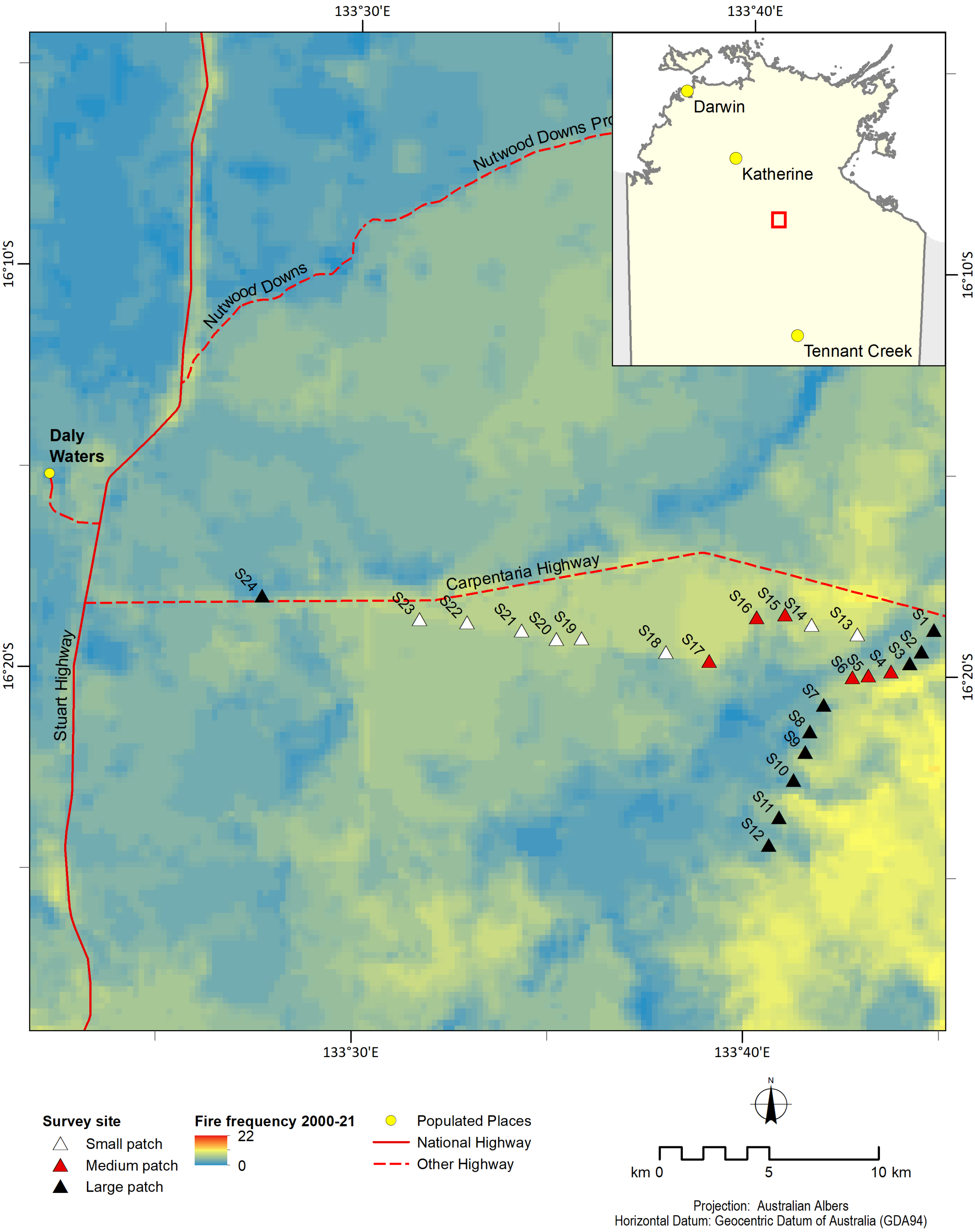


Figure 15 Fire frequency history in the study area. High fire frequency (e.g., every two years) is bright yellow shading; moderate fire frequency (e.g., every five years) is pale yellow; low fire frequency (e.g., zero times in two decades) is bright blue shading

5.2.3 Survey methods – plants

Flora surveys were conducted from 13 to 18 December 2022. Survey timing did not coincide with ideal sampling conditions (i.e., in the late—wet/early—dry season when ephemeral species are present). Due to this seasonal limitation, there was a strong perennial flora bias in the sample set and the annual/short-lived component of the flora is effectively unaccounted for in this study.

Sampling methodology followed Brocklehurst *et al*. (2007), which complies with the National Vegetation Information System (NVIS) standards for the collection of floristic and structural attributes to support vegetation classification and mapping (NVIS Technical Working Group 2017). At each site, a 50 m x 50 m plot was established within which every plant species and its percentage cover was recorded. Deciduous trees still had their canopy foliage retained at the time of survey which influenced the decision to estimate and record projected foliage cover for all species.

Maximum, minimum, and average height of each stratum and substratum present in the community (up to six but more typically three of Upper 1, Upper 2, Mid 1, Mid 2, Ground 1, Ground 2) was visually estimated. The dominant growth-form of each stratum/substratum was recorded and total percent cover of each stratum within the plot was visually estimated and recorded.

A set of environmental covariates was scored for each survey site including:

- landform pattern,
- landform element,
- site slope and aspect,
- soil surface (0—10 cm) colour,
- texture,
- pH,
- estimated percent gravel in sieved soil,
- effervescence of carbonate in fine earth,
- site drainage, and
- surface microrelief.

Soil characteristics were determined using the field texture and field pH and $CaCO_3$ methods of McDonald and Isbell (2009).

Visual estimates of groundcover were made based on lifeform or abiotic attribute (bare earth, litter, gravel/rock, perennial tussock grasses, hummock grasses, forbs, cryptogamic crust).

A record of disturbance type (if present) and intensity was also made for each plot. This included estimated time-since-fire and scorch height, cattle impacts, feral horse impacts, dieback, native herbivore impacts, erosion, and pig wallows and rooting.

5.2.4 Survey methods – vertebrate animals

Fauna surveys were carried out in 2022 from 20 September to 1 October, and from 13 to 23 October. The surveys focussed on reptiles, birds and bats. Sampling was carried out at each survey site for four days and nights. Within each survey site a 100 x 100 m area was marked out and sampling was carried out within that area.

Bird censuses involved timed (20 minute) active searches within the 100 x 100 m marked area caried out by a single observer. Each site was censused for birds five times both in the morning and afternoon. The number of individuals of each species was recorded for each count, avoiding counting the same individuals more than once.

Reptile surveys involved timed (20 minute) active searches within the 100 x 100 m marked area carried out by a single observer. Each site was censused for reptiles twice during the day and twice at night.

Insectivorous bats were trapped using two bank harp traps (Faunatech, Bairnsdale, Victoria). Harp traps were placed across likely bat flyways within each survey area; however, in contrast to the bird and reptile surveys, the location of harp traps was not necessarily restricted to the 100 x 100 m marked area. A total of four trap nights (two traps open for two nights) was completed for each survey site. Traps were checked multiple times during the night and also first thing in the morning.

The sites were not visited in the same order every day, to avoid systematic bias.

5.2.5 Analysis - plants

Floristic and environmental patterns were examined by testing the relationship between vegetation composition and patch size, along with a set of measured and derived environmental variables (Table 7). This was carried out using the PRIMER 7 software package with PERMANOVA+ add-on (Anderson *et al.* 2008, Clarke and Gorley 2015).

A Bray-Curtis resemblance matrix was constructed among the survey sites on the basis of plant species cover abundance data. The data were square-root transformed to down-weight the influence of the more abundant species.

PCO (principal coordinates analysis) was used to explore floristic gradients across the 24 sites. This technique is a projection of the points perpendicularly onto axes that minimise residual variation in Euclidean space. As with non-metric multidimensional scaling (MDS), only the first two (or three) axes are drawn in PCO ordination space.

Pearson simple linear correlations of 'intrinsic' vegetation variables were superimposed as vector overlays onto the PCO ordination plot. Vegetation intrinsic attributes included: total species richness (S) and richness of focal species groups (rainforest species and resprouter species); summed cover values (N); species diversity (*H*'(loge)); and the measured vegetation structural characteristics.

The PERMANOVA routine was used to test the hypothesis of no difference in community structure (defined by the Bray-Curtis measure on the square-root transformed data) between the patch-size groups. The categorical variable (i.e., patch size) was treated as a fixed whole factor and the default model selection options of partial sums of squares and the method of unrestricted permutation of raw data were chosen, with 9999 permutations being run. Pair-wise *t-test* comparisons were run among all pairs of levels where significant main factor effects were detected.

Distance-based linear modelling (*DISTLM*) was used to model the relationship between the multivariate vegetation data cloud (the response variable), as described by the resemblance matrix and the set of environmental predictor variables. The environmental variables that were used in the initial analysis and the range of their values are included in Table 7. A second *DISTLM* analysis examined the relationship between the multivariate vegetation data and the vegetation-based environmental data (structural and floristic).

The data for the variables fire frequency and time-since-fire were derived from Northern Australia and Rangelands Fire Information website (https://firenorth.org.au/nafi3/). *P*-values for testing the null hypothesis of no relationship (for individual variables alone, and conditional on other variables) were obtained using permutation methods. A parsimonious model was built using the Forward selection procedure and the Adjusted R^2 criterion (which discards extraneous variables). Preliminary diagnosis (using the Draftsman Plot tool) assessed for multi-collinearity among predictor variables and for heavily skewed distributions or extreme outliers. Collinear variables were removed and data were transformed as required. The focal variable, *patch size*, was fitted last to the model to test its relationship with the vegetation resemblance matrix once the environmental covariates had already been considered (referred to as conditional effects modelling). The dbRDA routine was used to perform an ordination of the fitted values from the best solution model. The dbRDA is constrained to find linear combinations of the predictor variables which explain the greatest variation in the data cloud. To characterise the dbRDA axes, the strength and direction of the relationship between individual variables and the axes was determined through calculating the multiple partial correlations between each of the variables and the dbRDA axis scores. Vector overlays for each predictor variable allow for visualisation of these relationships. These vectors can be interpreted as the (conditional) effect of a given variable on the construction of the constrained ordination picture – the longer the vector, the bigger the effect of that variable in the construction of the dbRDA axes. As with the PCO analysis, Pearson simple linear correlations of intrinsic vegetation variables were superimposed as vector overlays onto the dbRDA ordination plot.

Having gained support for a floristic basis to the *a priori* patch-size groups from the PCO and *DISTLM* results, there was then interest in characterising differences among the groups. To do this, we undertook discriminant analysis using canonical analysis of principle coordinates (CAP). The purpose of CAP is to find axes through the multivariate cloud of points that are the best at discriminating among predefined groups. The CAP routine chooses an appropriate number of principal coordinate analysis (PCO) axes to use for the discriminant analysis – this reflects the requirement to include as much of the original variability in the data cloud as possible (at least 60%), while excluding the PCO axes that do nothing to discriminate the groups. The misclassification error (the proportion of sites that were misclassified) is estimated using a leave-one-out procedure. In short, the CAP analysis was used to assess how distinct the patch groups are from one another in multivariate space (Anderson et al. 2008). The model was first run using the three patch-size groups and then using the binary contrast of small versus medium-large patch-size.

Next, indicator species of the patch-size categories were identified to further characterise the patch-size groups in terms of species and life-history and disturbance trait dominance patterns. This was done in an attempt to identify the species that may be sensitive to patch-size and which could therefore be used for condition monitoring. To do this, we used the SIMPER routine on the floristic data to determine diagnostic species for between-group Bray-Curtis dissimilarity. For this, all species contributing up to 70% (the highest-order contributors) of the between-group dissimilarity were included.

Table 7 Environmental variables used in the Distance-based linear model (DISTLM) of floristic-environmental gradients in the Beetaloo Sub-region.

Environmental variable	Range of values for the variable
Patch size	large=3; medium=2; small=1
Longitude	133.46-133.75ºE
Latitude	16.41-16.30 ºS
Approximate % clay content	1=10-20% (light sandy clay loam); 2=20-30% (sandy clay loam); 3=30-35% (fine sandy clay loam)
Soil surface pH	5.5-7.5
Gravel in soil	1-20%
Ground surface cover	(1-15% stone; 0-20% rock, 75-99% bare)
Slope	0-1º
Fire Frequency (no. times burnt 2000-2022) (from NAFI)	0-9 times
Time-since-fire (from NAFI)	1-22 years
Fire history (field estimated)	1= recent fire (c. 1-2 years); 2= long unburnt (> 2 years)
Cattle impact severity	0=no visible impact; 1 = 1-20% of site impacted; 2=21-40% of site impacted

5.2.6 Analysis – vertebrate animals

The analysis methods for fauna closely followed those for plants detailed in section 5.2.5. Specifically:

- Faunal and environmental patterns were examined by testing the relationship between faunal composition and patch size, along with a set of measured and derived environmental variables.
- A Bray-Curtis resemblance matrix was constructed among the survey sites on the basis of vertebrate species abundance data. The data were square-root transformed to down-weight the influence of the more abundant species.
- PCO (principal coordinates analysis) was used to explore faunal gradients across the 24 sites.
- The PERMANOVA routine was used to test the hypothesis of no difference in community structure (defined by the Bray-Curtis measure on the square-root transformed data) between the patch-size groups.
- Indicator species of the patch-size categories were identified using the SIMPER routine on the vertebrate data to determine diagnostic species for between-group Bray-Curtis dissimilarity. For this, all species contributing up to 70% (the highest-order contributors) of the between-group dissimilarity were included.
- Distance-based linear modelling (*DISTLM*) was used to model the relationship between the multivariate faunal data cloud (the response variable), as described by the resemblance matrix, and the set of environmental predictor variables.

5.3 Results - plants

5.3.1 Species richness

In total, 94 plant taxa were recorded from the survey. Average plot richness across all sites was 29 ± 7.5 and the range was 12—49 species. Richness for small, medium, and large patches was 34.9 ± 6.4 (range 20, 29—49), 22.5 ± 5.9 (range 17, 12—29) and 28.1 ± 5.6 (range 15, 20—35).

Canopy and mid-layer species (61 species) dominated the sample with 20 tree species, four shrub/tree species, 23 shrub species and 14 vine species having been recorded. The ground layer (33 species) was represented by one sedge species, one fern, 13 tussock grass species and 14 forb and four forb/shrub species.

5.3.2 Floristic patterns

In the PCO, the percentage of the total variation inherent in the resemblance matrix that was explained by the first two axes was relatively low (35%) indicating that the projection may not have captured the salient patterns in the full data cloud. Nevertheless, the floristic patterns were most easily accounted for by differences in patch size (Figure 16). Specifically, there was evidence of relatively high compositional uniformity among small patch sites, all of which were positively correlated with Axis 2 of the ordination. By contrast, medium and large patches lacked cohesion, with sites from each group being intermixed and widely dispersed along Axis 1 and Axis 2 of the ordination. Notably, one medium patch site (site 16) and one large patch site (site 24) were positioned within the cluster of small patch sites, implying a moderate level of floristic overlap between small patches and a subset of larger patch sizes.

There was high variability in the projected foliage cover of the two canopy dominants, *A*. *shirleyi* and *M*. *kekwickii*, with three large patch sites (sites 1, 2 and 3) having very low values of each; the large patch sites 9 and 11 having very low *M*. *kekwickii* cover; and with the medium patch sites 4 and 5 having relatively low cover of *A*. *shirleyi* (Figure 17). Notably, the medium and large patch sites (16 and 24) that were grouped with small patch sites both had relatively high cover of the canopy dominants.

Ground (G2) and mid (M1 and M2) strata cover was negatively associated with *M*. *kekwickii* and *A*. *shirleyi* cover (and positively with sites 1, 2 and 3) (Figure 17). Upper stratum cover (U1) showed the opposite relationship, being particularly associated with high *M*. *kekwickii* cover. Upper substratum cover (U2) was positively associated with a subset of medium and large patch sites and with *A*. *shirleyi* cover along Axis 1. Litter cover was positively associated with small patch sites along Axis 2.

Mid substratum (M2) height, Upper stratum (U1) height and Upper substratum (U2) height showed the same relationships as their cover values (Figure 18). Species diversity (*H*’(loge); total species richness (S); richness of rainforest species and resprouter species; richness of trees, shrub/trees, shrubs and vines, and total cover (N) were positively correlated with a subset of small patch sites along Axes 1 and 2. Sedge and Tussock grass richness showed the opposite response. Forb and fern richness was positively associated with low *A*. *shirleyi* and *M*. *kekwickii* cover (and with sites 1, 2 and 3) (Figure 19).

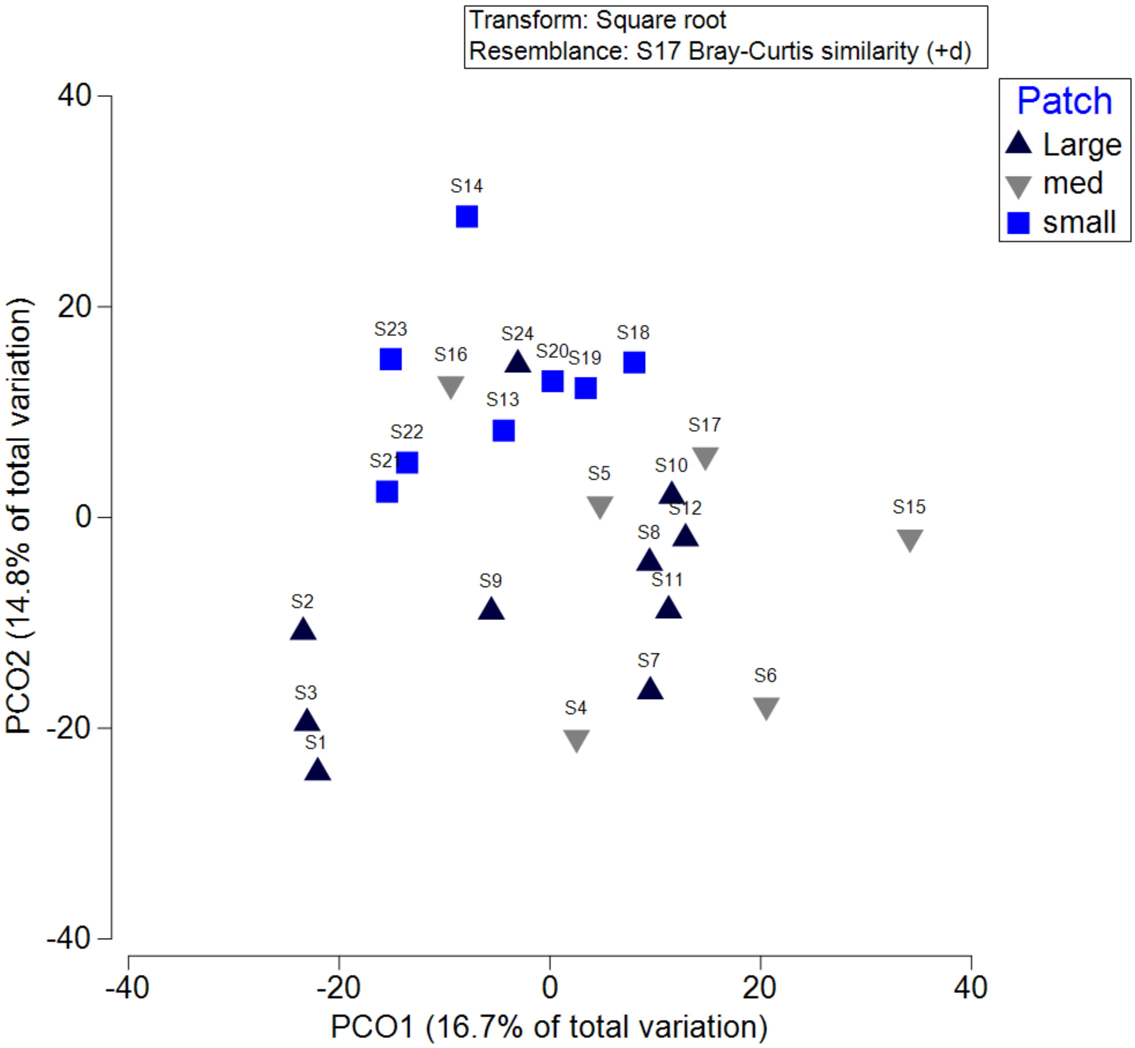


Figure 16 PCO ordination of 24 sample sites (S1-24) of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station. Large patch (black triangle) and medium patch (grey triangle) sites are intermixed, while small patch sites (blue square) form a cohesive group

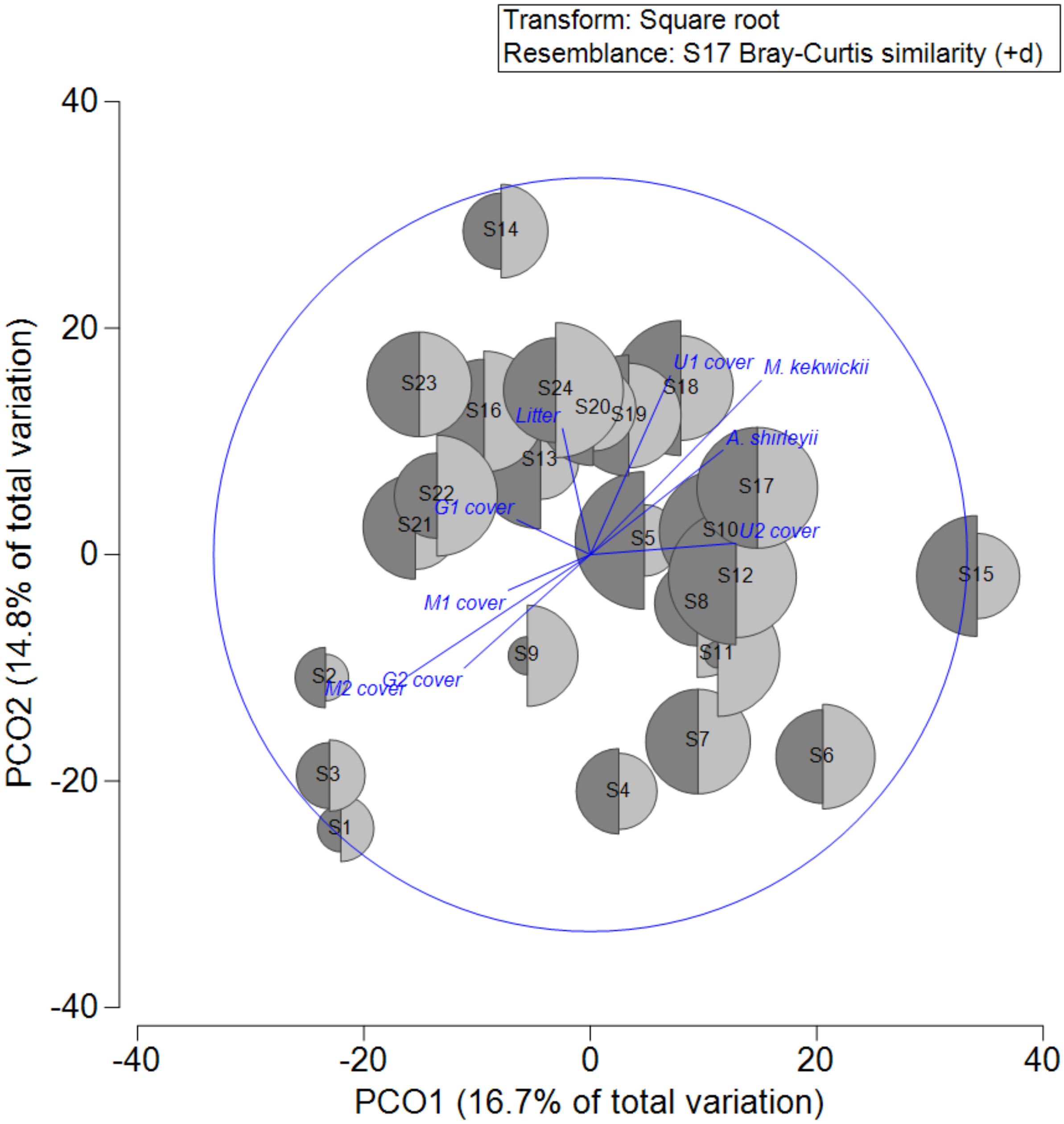


Figure 17 PCO ordination of 24 sample sites of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between site structural attributes and the positioning of the sites in ordination space. In the figure, bubble plots are used to illustrate the relative cover of the canopy dominants *A. shirleyi* (light grey) and *M. kekwickii* (dark grey) at each site. U1 = upper stratum; U2 = upper substratum; M2 = mid substratum; G1 = ground stratum. Litter = ground litter cover; *M. kekwickii* is cover of the bullwaddy habitat dominant; *A. shirleyi* = cover of the lancewood dominant

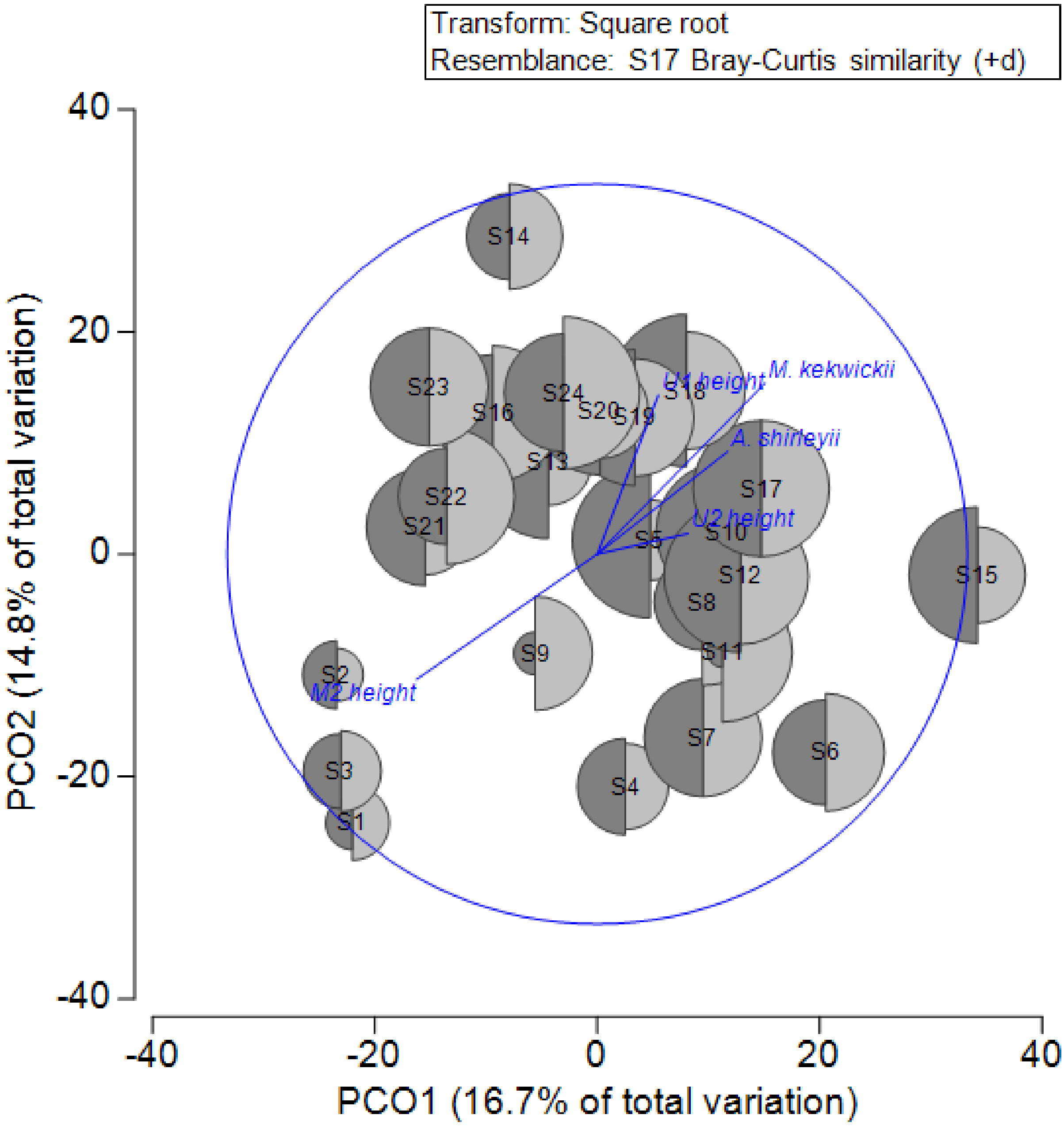


Figure 18 PCO ordination of 24 sample sites of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between site structural attributes and the positioning of the sites in ordination space. In the figure, Bubble plots are used to illustrate the relative cover of the canopy dominants *A. shirleyi* (light grey) and *M. kekwickii* (dark grey) at each site. U1 = upper stratum; U2 = upper substratum; M2 = mid substratum; *M. kekwickii* is cover of the bullwaddy habitat dominant; *A. shirleyi* = cover of the lancewood dominant

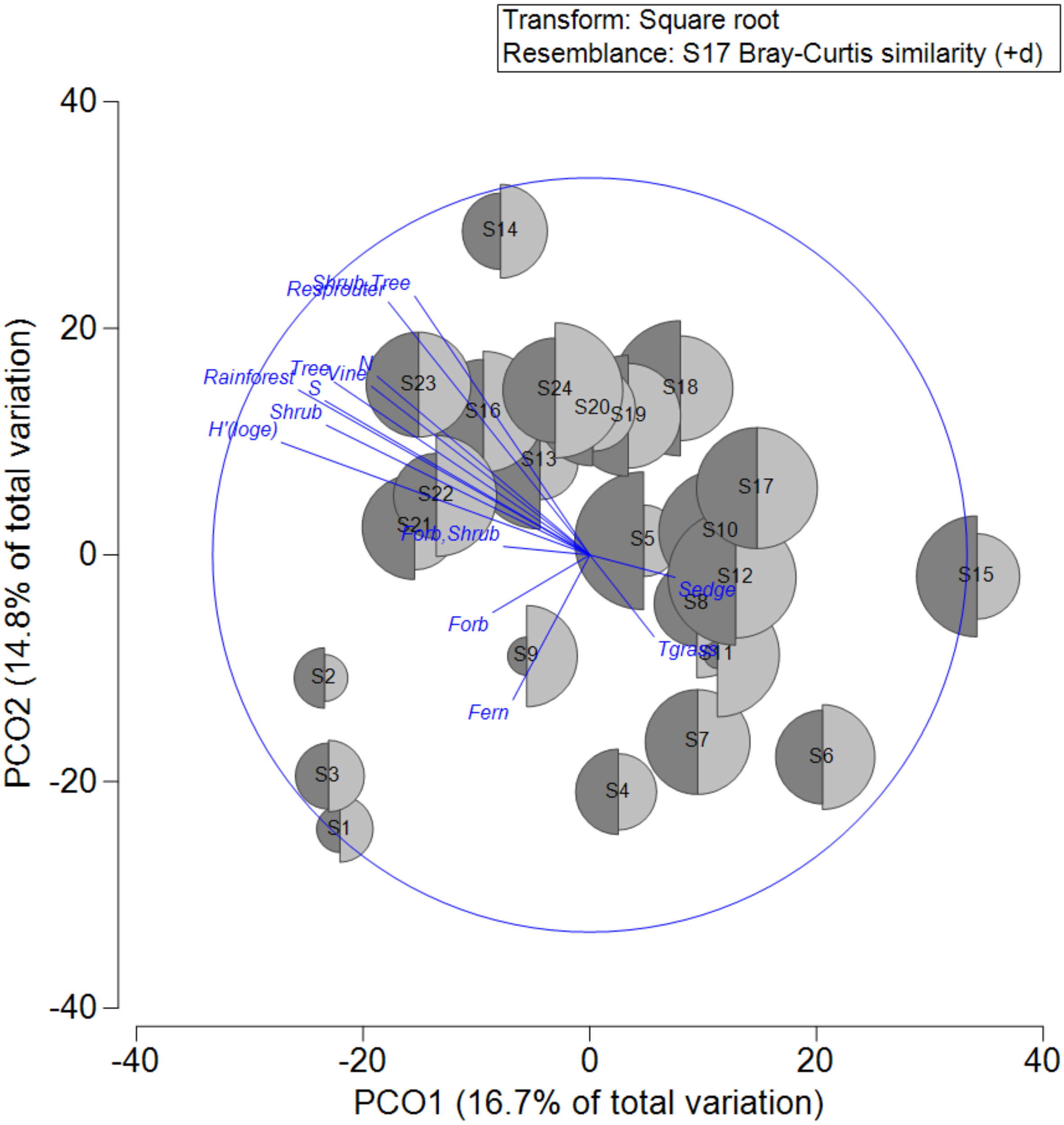


Figure 19 PCO ordination of 24 sample sites of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between site diversity indices and growth form group/focal group richness and the positioning of the sites in ordination space. Bubble plots are used to illustrate the relative cover of the canopy dominants *A. shirleyi* (light grey) and *M. kekwickii* (dark grey) at each site. H'(loge) = plant species diversity; S = species richness (number of species per plot); N = total species cover value for each plot; Tgrass = tussock grass richness; Rainforest = rainforest-allied species; Introduced = non-native species; Resprouter = fire-resprouter richness

PERMANOVA confirmed that vegetation composition varied according to patch size (Pseudo-$F_{2, 21}$ = 1.9808, P(perm) = 0.0013). Pair-wise tests showed that small patches differed from medium (t = 0.0074) and large patches (t = 0.0011), but that medium and large patches were compositionally similar (t = 0.1141).

The results of the CAP analysis were consistent with those of PERMANOVA in that they provided support for the floristic distinctiveness of small patches. Specifically, with sites first assigned to small-, medium- or large patch categories, there was an overall misclassification error of 33.3% (8/24), with 70%, 33.3% and 87.5% of large, medium, and small patches having the correct *a priori* classification (m = 7; P = 0.0045). Next, with sites classified *a priori* as small or medium-large patches, the misclassification error was reduced to 12.5%, with 87.5% of small and medium-large patch sites being correctly classified. Here, two sites (sites 16 and 24) were misclassified as small patches and one small site (site 21) was misclassified as a large patch.

Based on the PERMANOVA and CAP results, medium and large patches were merged into a single group for SIMPER analysis. Thirty-one species contributed to 70% dissimilarity between small and medium-large patches and, of these, the majority (23 species) were more prevalent in small patches (Table 8). Species that were highly characteristic of small patches were four tree species (including the bullwaddy canopy dominant *M. kekwickii*); two shrub/tree species (*Santalum lanceolatum* and *Ventilago viminalis*); six shrub species (foremost *Carissa lanceolata* and *Abutilon fraseri* subsp. *fraseri*); four vines (foremost *Secamone elliptica* and *Capparis lasiantha*); one sedge (*Scleria brownii*); and four tussock grass species and two forb/shrub species. The lancewood dominant, *A. shirleyi*, was highly characteristic of medium-large patches, along with the tree *Terminalia volucris*; two shrub species (*Flueggea virosa* subsp. *melanthesoides*, *Margaritaria dubium-traceyi*); one fern, one forb/shrub and two tussock grass species (Table 8). Thus, medium-large patches had on average, higher *A. shirleyi* canopy cover and fewer characteristic species, and especially low representation of trees, shrubs, and vines.

Differences between small and medium-large patches were also evident in relation to life-history trait dominance. Specifically, of the 23 species that characterised small patches, ten (43%) were rainforest species and 16 (70%) were resprouter/facultative-resprouter species. By comparison, of the eight characteristic species of medium-large patches, 50% were rainforest species and 50% were resprouter/facultative-resprouter species. Thus, both the smaller and medium-large patches supported a mixture of rainforest and non-rainforest species; but the former supported more (numerically and proportionately) fire resilient resprouter species.

Table 8 Results of SIMPER analysis of between-group Bray-Curtis dissimilarity of plant cover abundance data on 24 plots. Bold font indicates species with higher abundance in small patches and that contribute to 50% between-group dissimilarity

Species	Medium-large patch Av. Abund	Small patch Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Macropteranthes kekwickii	2.93	4.06	2.46	1.32	4.77	4.77
Sida sp. *excedentifolia*	1.29	0.13	1.93	1.2	3.74	8.51
Acacia shirleyi	3.78	3.59	1.62	1.51	3.14	11.65
Terminalia volucris	1.74	1.37	1.54	1.36	2.99	14.64
Abutilon fraseri **subsp.** ***fraseri***	0.43	1.27	1.5	1.84	2.91	17.55
Secamone elliptica	0.72	1.46	1.4	1.25	2.71	20.27
Scleria brownii	0.76	0.81	1.39	1.14	2.69	22.95
Sida **sp.** ***Suplejack Station***	0.67	0.86	1.35	1.23	2.62	25.57
Chrysopogon fallax	0.03	0.91	1.35	1.01	2.61	28.19
Santalum lanceolatum	0.3	1	1.31	1.25	2.53	30.72
Ventilago viminalis	0.03	0.8	1.26	0.93	2.44	33.16
Eucalyptus leucophloia **subsp.** ***euroa***	0.44	0.51	1.22	0.76	2.36	35.51
Carissa lanceolata	0.41	0.72	1.18	1.09	2.29	37.81
Ehretia saligna	0.72	0.88	1.17	1.32	2.26	40.07
Margaritaria dubium-traceyi	0.65	0.22	1.11	0.84	2.15	42.21
Capparis lasiantha	1.26	1.57	1.03	1.18	2	44.22
Melhania oblongifolia	0.56	0.57	1.03	1.05	2	46.22
Cheilanthes sieberi subsp. *pseudovellea*	1.39	1.34	1	1.11	1.95	48.16
Abrus precatorius **subsp.** ***precatorius***	0.46	0.62	0.99	1.31	1.92	50.08
Paspalidium sp.	0.2	0.59	0.99	1.02	1.92	52
Capparis umbonata	0.37	0.71	0.95	1.45	1.84	53.84
Enneapogon lindleyanus	0.6	0.84	0.92	1.15	1.79	55.64
Aristida calycina var. calycina	0.77	0.5	0.9	1.15	1.75	57.39
Digitaria brownii	0.5	0.55	0.9	1.17	1.75	59.14
Premna acuminata	0.26	0.63	0.88	1.2	1.7	60.85
Hibiscus sp.	0.27	0.47	0.86	0.86	1.67	62.51
Capparis sepiaria	0.13	0.51	0.82	0.87	1.58	64.1
Digitaria sp.	0.51	0.3	0.81	1.14	1.56	65.66
Hypoestes floribunda var. *angustifolia*	0	0.5	0.78	0.88	1.52	67.18
Galactia tenuiflora	0.44	0.2	0.76	0.94	1.48	68.65
Hybanthus aurantiacus	0	0.49	0.76	1.18	1.47	70.13

5.3.3 Species—environmental relationships (constrained ordination)

The marginal tests performed by the Distance-based linear modelling (*DISTLM*) confirmed that four of the environmental variables had a significant relationship with the species-derived multivariate data cloud when considered alone: patch size (small, medium, or large) (P = 0.0013); latitude (P = 0.0095); longitude (P = 0.0126), and fire frequency (P = 0.0492).

The best model produced by the Forward selection procedure included the four significant variables and an additional two variables (% gravel in soil and time-since-fire) and the total variation explained by these variables was 34.7%. The variable patch size explained the highest amount of the variation in the community structure (9.3 %), followed by time-since-fire (6.6%), latitude (5.4%), % gravel (5.1%), fire frequency (4.3%) and longitude (4.1%). A relatively high amount of the fitted model variation was captured by the first two dbRDA axes (56.3%). Cattle impacts and soil texture and pH were not significant and they were not included in the best solution model.

Species composition was highly correlated with patch size along Axis 1, with a gradient occurring from large patches on the left, medium patches in the middle and small patches on the right (Figure 20). Notably, one large patch site (S24) from the east—west transect showed strong affinity with the small patch sites along this Axis. Also, along Axis 1, small patch sites were strongly associated with fire frequency (FF) and whereas large patch sites were associated with infrequent fire. Floristic composition was influenced by % gravel, latitude, and time-since-fire along Axis 2 but there was no relationship between patch size and these environmental variables.

The cover of *M*. *kekwickii* was positively associated with small patch size and high fire frequency along Axis 1 while *A*. *shirleyi* cover was very strongly correlated with % gravel and with lower latitude (especially sites 10, 11 and 12 along the north-south gradient) (Figure 21). Cover of both dominants was closely associated with long time-since-fire, as was upper stratum (U1) cover and height and ground stratum (G1) cover. Mid stratum (M1) height was associated with small patches and with *M*. *kekwickii* cover along Axis 1. Mid substratum (U2) cover and height was associated with recent fire. Litter cover was strongly correlated with *A*. *shirleyi* cover along Axis 2.

Species diversity, total species cover, total species richness, richness of trees, shrub/trees, shrubs and vines and resprouter and rainforest species were positively correlated with small patch size and high fire frequency along Axis 1, while the groups tussock grass, forb and fern showed the opposite relationship. Sedge richness was associated with *A. shirleyi* cover along Axis 2. Richness of introduced plant species was related to small patch size and fire frequency (Figure 22).

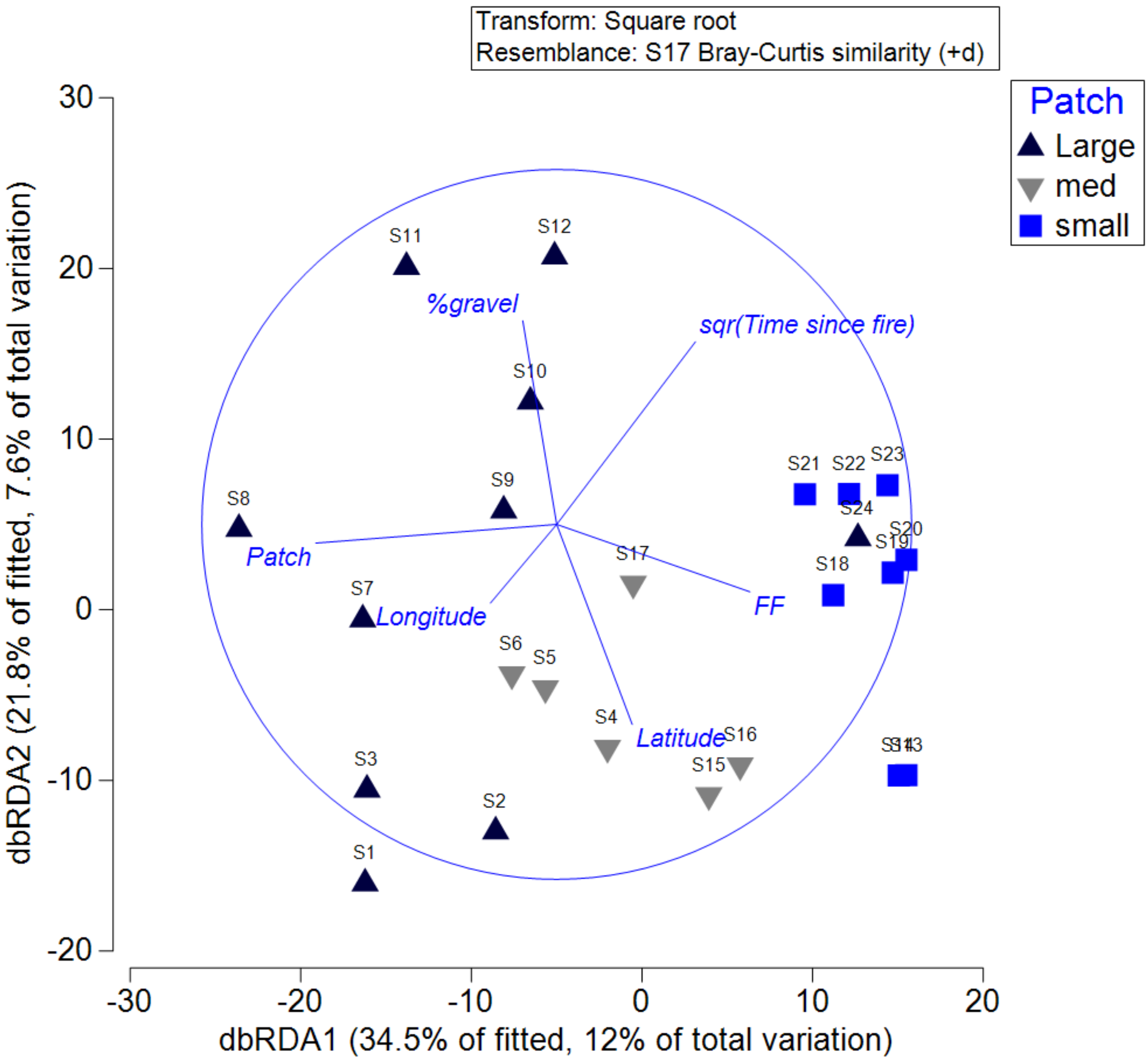


Figure 20 dbRDA ordination of 24 vegetation sample sites (S1 to S24) of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station. The positioning of the sites in this ordination is constrained by modelled environmental variables. Species composition was highly correlated with patch size and fire frequency (FF) along Axis 1. Floristic composition was influenced by %gravel, latitude, and time-since-fire (square-root transformed) along Axis 2

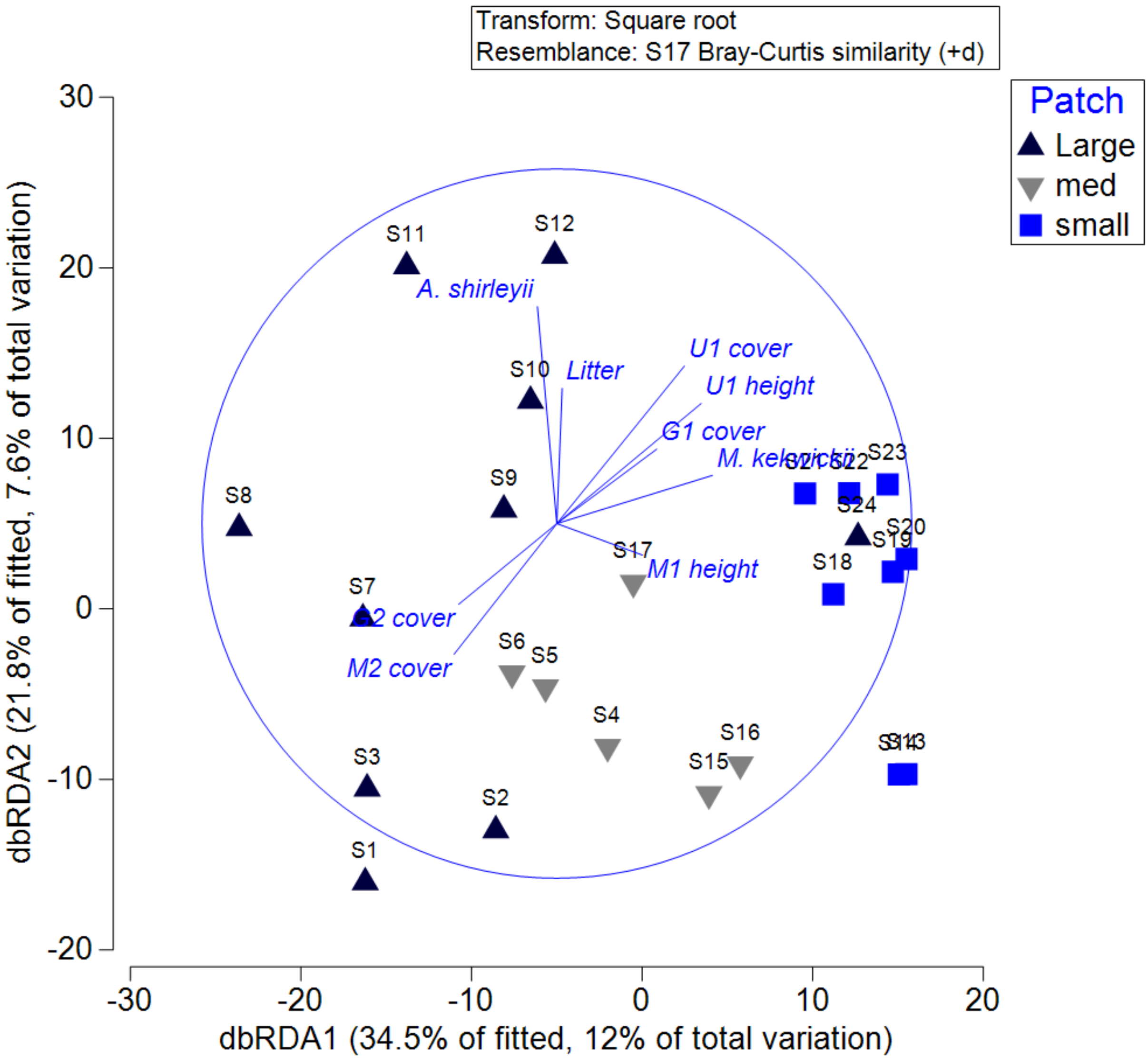


Figure 21 dbRDA ordination of 24 vegetation sample sites (S1 to S24) of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between site structural attributes and the positioning of sites in ordination space. The positioning of the sites in this ordination is constrained by modelled environmental variables. U1 = upper stratum; U2 = upper substratum; M2 = mid substratum; G1 = ground stratum. Litter = ground litter cover; *M. kekwickii* is cover of the bullwaddy habitat dominant; *A. shirleyi* = cover of the lancewood dominant

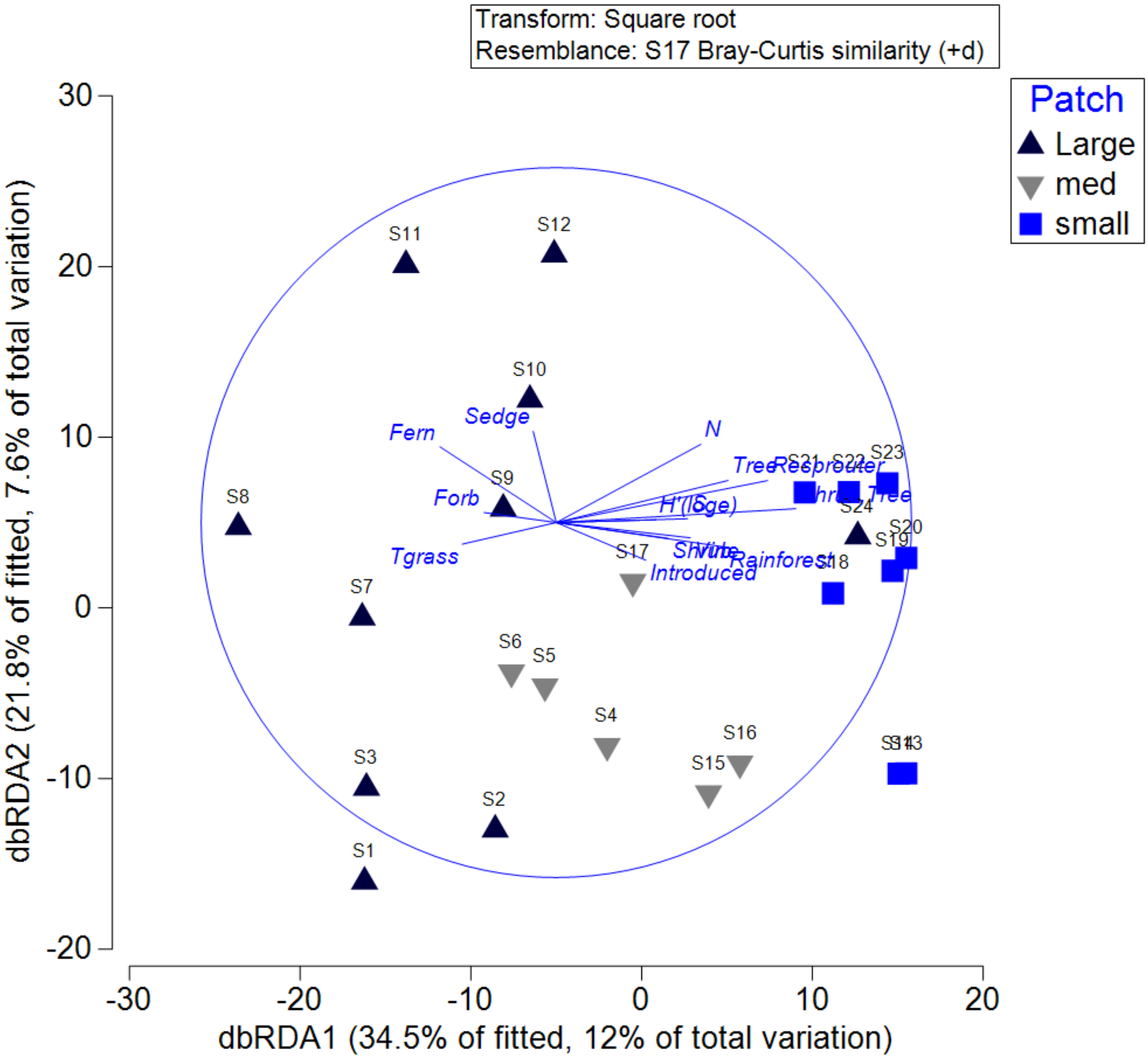


Figure 22 dbRDA ordination of 24 vegetation sample sites (S1 to S24) of bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between site diversity indices and growth form group/focal group richness and the positioning of the sites in ordination space. The positioning of the sites in this ordination is constrained by modelled environmental variables. H'(loge) = plant species diversity; S = species richness (number of species per plot); N = total species cover value for each plot; Tgrass = tussock grass richness; Rainforest = rainforest-allied species; Introduced = non-native species; Resprouter = fire-resprouter richness

5.4 Results – vertebrates

5.4.1 Species richness

In total, 85 species of vertebrates were recorded during the surveys. This consisted of six species of mammal, one frog, 19 reptiles and 59 species of birds.

Average vertebrate richness across the 24 sites was 21.33 species ± 1.20 SE with a range of 13—33 species. The average richness across patch size was:

- Large patches: 18 species ± 1.17 SE (range of 13—23 species),
- Medium patches: 19 species ± 2.08 SE (range of 15—29 species), and
- Small patches: 27 species ± 1.60 SE (range of 20—33 species).

5.4.2 Vertebrate animal patterns

In the PCO, the percentage of the total variation inherent in the resemblance matrix that was explained by the first two axes was relatively low (33.2%) indicating that the projection may not have captured the salient patterns in the full data cloud. Nevertheless, the patterns in the vertebrate fauna were most easily accounted for by differences in patch size (Figure 23). Specifically, similar to the plant compositional data (Figure 16) there was evidence of high compositional uniformity among small patch sites, all of which were positively correlated with Axis 1 of the ordination. By contrast, medium and large patches lacked cohesion, with sites from each group being intermixed and widely dispersed along Axis 1 and Axis 2 of the ordination. Only one medium patch site (site 17) was positioned within the cluster of small patch sites.

Plant species diversity (H'(loge), total plant species richness (S) and total plant cover (N) were all positively correlated with vertebrate composition in the small patch sites (and medium patch site 17) along Axes 1 and 2 (Figure 24).

PERMANOVA confirmed that vertebrate composition varied according to patch size (Pseudo-$F_{2, 21}$ = 1.8961, P(perm) = 0.0028). Pair-wise tests showed that small patches differed from large patches (t = 0.0001) and also differed marginally from medium patches (t = 0.0679). Medium and large patches were compositionally similar (t = 0.8942).

The results of the CAP analysis were consistent with those of PERMANOVA in that they provided support for the distinctiveness of small patches in terms of vertebrate composition. Specifically, with sites first assigned to small-, medium- or large patch categories, there was an overall misclassification error of 33.33% (8/24), with 50%, 50% and 100% of large, medium, and small patches, respectively, having the correct *a priori* classification (m = 7; *P* = 0.002). Next, with sites classified *a priori* as small or medium-large patches, the misclassification error was reduced to 4.167%, with 95.83% of small and medium-large patch sites being correctly classified. Here, there was a single misclassification with the medium-large site being misclassified as a small patch.

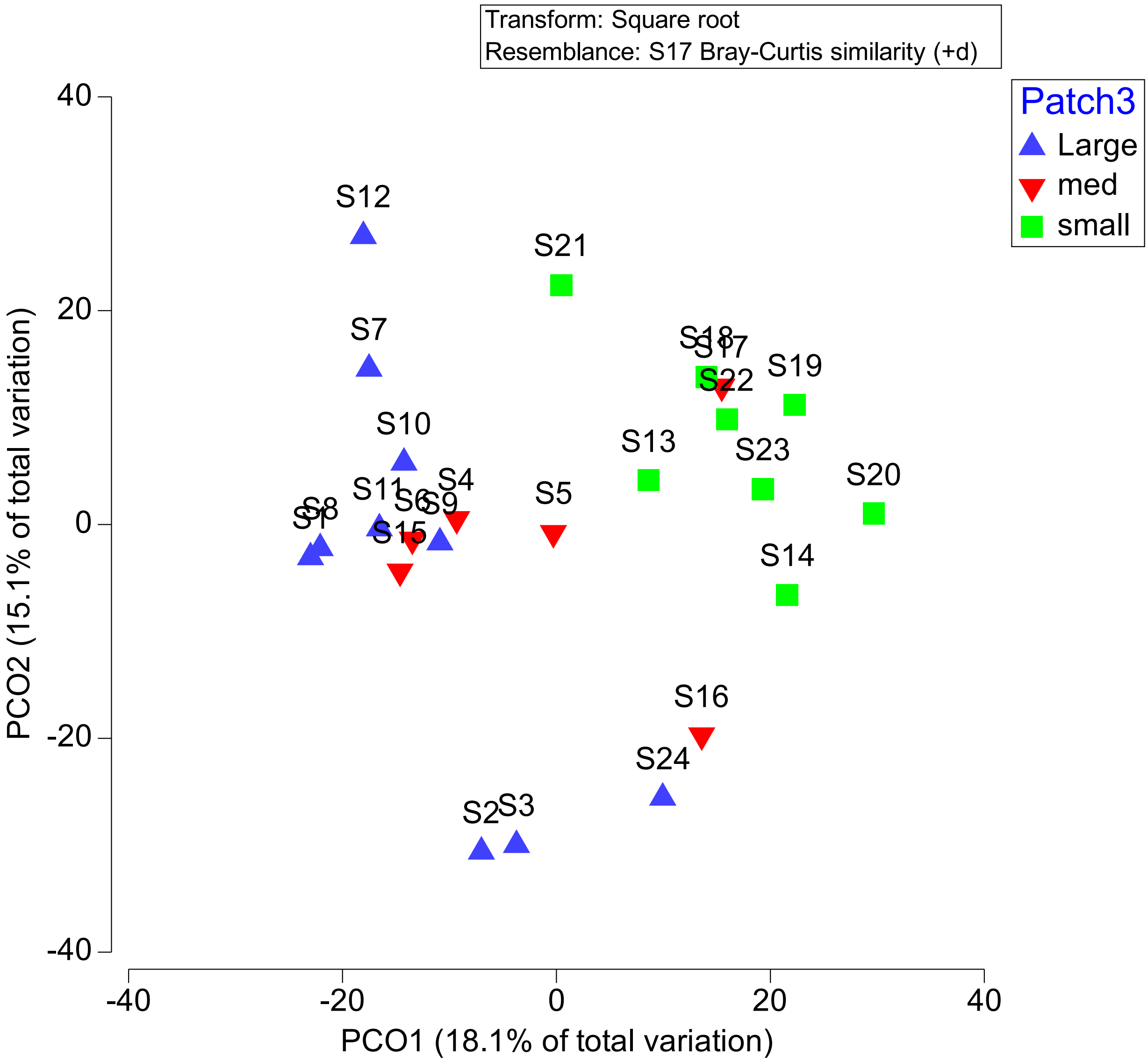


Figure 23 PCO ordination of 24 vertebrate sample sites (S1-24) in bullwaddy/lancewood woodland on Hayfield-Shenandoah Station

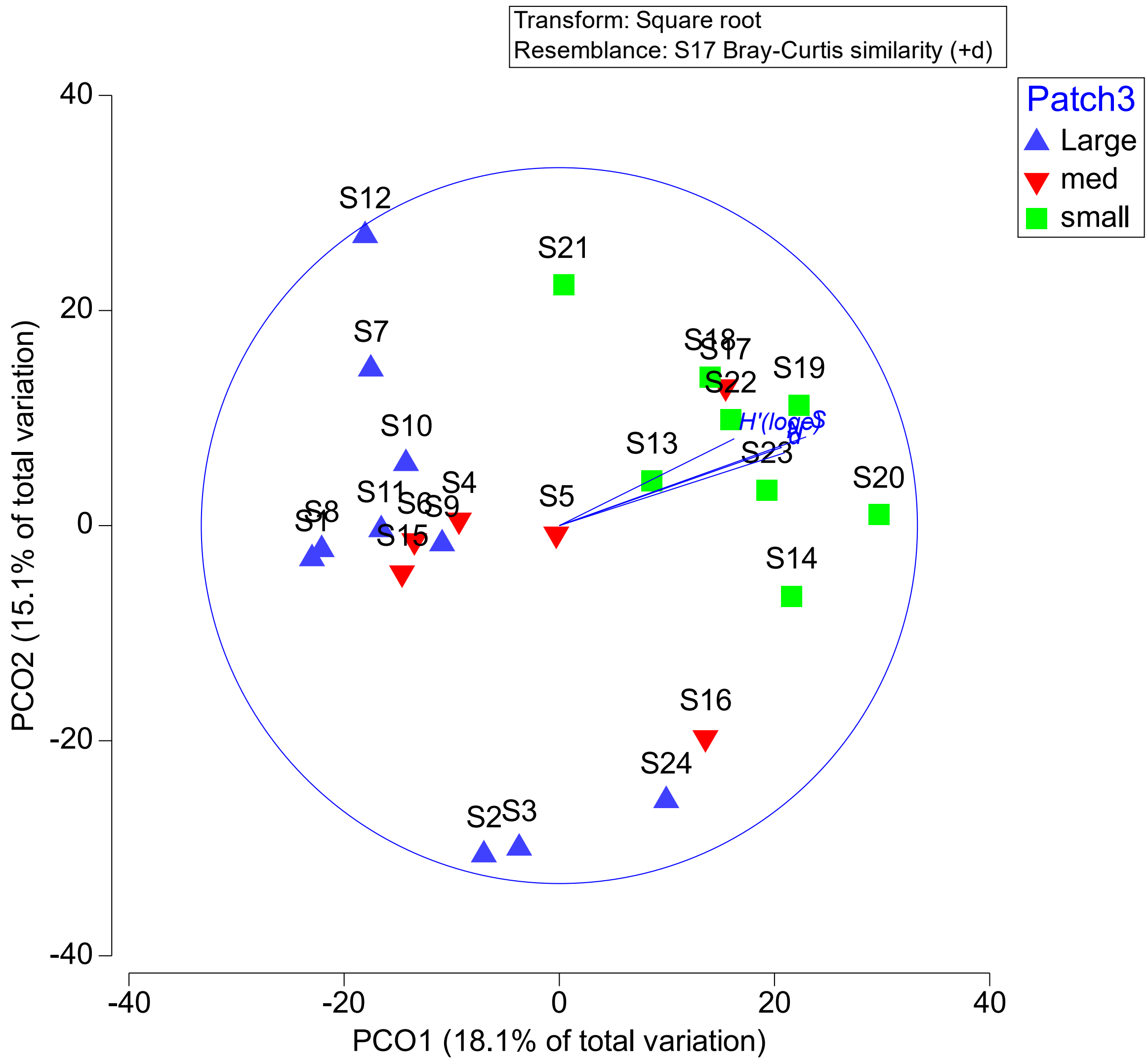


Figure 24 PCO ordination of 24 vertebrate sample sites (S1-24) in bullwaddy/lancewood woodland on Hayfield-Shenandoah Station showing correlations (vector lines) between plant species diversity and richness and the positioning of sites in ordination space. H'(loge) = plant species diversity; S = species richness (number of species per plot); N = total species cover value for each plot

Based on the PERMANOVA and CAP results, medium and large patches were merged into a single group (medium-large) for SIMPER analysis. Twenty-five vertebrate species contributed to 70% dissimilarity between small and medium-large patches, and of these, the majority (16 species) were more prevalent in small patches (Table 9). Species that were highly characteristic of small patches were five species of gecko (*Amalosia rhombifer*, *Gehyra gemina*, *Heteronotia binoei*, *Strophurus ciliaris*, *Lucasium immaculatum*), one skink (*Lerista labialis*), nine birds (eight passerines and the Australian owlet-nightjar, *Aegotheles cristatus*), and the stripe-faced dunnart (*Sminthopsis macroura*). Three skinks (*Carlia amax*, *Menetia maini*, *Lerista orientalis*) and the dragon (*Amphibolurus centralis*) were highly characteristic of medium-large patches, along with four species of passerine birds (*Melanodryas cucullata*, *Lichenostomus virescens*, *Smicrornis brevirostris*, *Daphoenositta chrysoptera*) and the bat, *Nyctophilus geoffroyi*. Only one of these 25 species was recorded in one patch size and not the other. This species was *Lerista labialis* (Table 9).

Table 9 Results of SIMPER analysis of between-group Bray-Curtis dissimilarity of vertebrate animal abundance data on 24 plots. Bold font indicates species with higher abundance in small patches and that contribute to 50% between-group dissimilarity

Species	Medium-large patch Av. Abund	Small patch Av. Abund	Av. Diss	Diss/SD	Contrib%	Cum.%
Malurus assimilis	1.85	3.25	3.04	1.37	5.31	5.31
Pomatostomus temporalis	1.70	1.89	2.50	1.21	4.38	9.69
Struthidea cinerea	0.60	1.14	2.49	0.76	4.35	14.04
Amalosia rhombifer	0.88	1.76	2.17	1.37	3.79	17.83
Pachycephala rufiventris	1.36	2.48	2.12	1.40	3.71	21.53
Carlia amax	1.40	0.38	2.08	1.59	3.64	25.17
Rhipidura dryas	0.21	1.25	2.05	1.36	3.58	28.75
Gehyra gemina	0.21	1.15	1.86	1.39	3.25	32.00
Melanodryas cucullata	1.03	1.00	1.78	1.24	3.12	35.12
Menetia maini	1.15	0.82	1.68	1.14	2.93	38.05
Heteronotia binoei	0.11	0.85	1.56	1.56	2.73	40.78
Amphibolurus centralis	0.86	0.73	1.42	1.18	2.49	43.27
Sminthopsis macroura	0.23	0.85	1.37	1.45	2.40	45.66
Lichenostomus virescens	1.80	1.56	1.33	1.12	2.32	47.99
Colluricincla harmonica	0.43	0.89	1.33	1.25	2.32	50.31
Lerista orientalis	0.78	0.73	1.32	1.11	2.30	52.61
Nyctophilus geoffroyi	0.68	0.18	1.27	0.88	2.23	54.83
Lucasium immaculatum	0.21	0.60	1.19	1.03	2.08	56.91
Ptilonorhynchus nuchalis	0.06	0.68	1.18	1.21	2.05	58.96
Strophurus ciliaris	1.27	1.47	1.16	1.09	2.03	60.99

Smicrornis brevirostris	0.57	0.34	1.13	0.91	1.98	62.97
Rhipidura leucophrys	1.30	1.75	1.05	1.34	1.83	64.80
Lerista labialis	0.00	0.63	1.03	0.72	1.80	66.60
Daphoenositta chrysoptera	0.49	0.13	0.99	0.57	1.73	68.33
Aegotheles cristatus	0.30	0.47	0.98	0.89	1.72	70.05

5.4.3 Species—environmental relationships (constrained ordination)

The marginal tests performed by the Distance-based linear modelling (*DISTLM*) confirmed that four of the environmental variables had a significant relationship with the vertebrate species-derived multivariate data cloud when considered alone: patch size (small, medium, or large) (P = 0.0001); latitude (P = 0.0011); longitude (P = 0.0016), and fire frequency (P = 0.0032). Soil surface pH was close to significant (P = 0.0596).

The best model produced by the Forward selection procedure included the five variables given above and an additional variable (cattle impact severity). The total variation explained by these variables was 43.43%. The variable patch size explained the highest amount of the variation in the community structure (13.6 %), followed by latitude (9.19%), soil surface pH (5.56%), longitude (5.56%), fire frequency (5.36%) and cattle impact severity (4.14%). A relatively high amount of the fitted model variation was captured by the first two dbRDA axes (61.86%).

As with the plant data, vertebrate species composition was highly correlated with patch size along Axis 1, with a gradient occurring from large patches on the left, medium patches in the middle and small patches on the right (Figure 25). Also, along Axis 1, small patch sites were strongly associated with fire frequency (FF) and *vice versa* for large patch sites. Vertebrate composition was influenced by latitude, soil surface pH, and cattle impact severity along Axis 2.

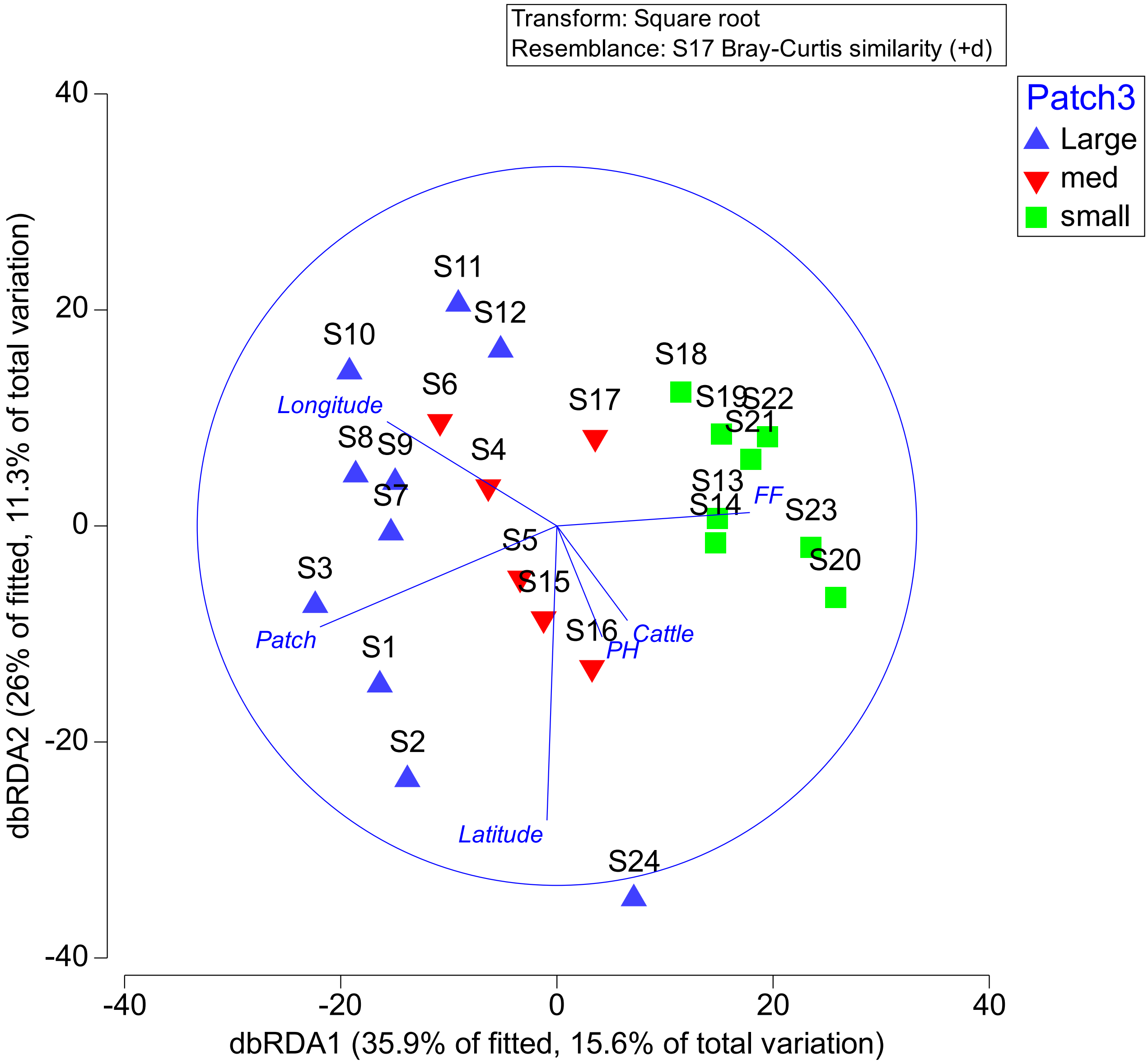


Figure 25 dbRDA1 ordination of 24 vertebrate sample sites (S1-24) in bullwaddy/lancewood woodland on Hayfield-Shenandoah Station. The positioning of sites in the ordination is constrained by modelled environmental variables

5.5 Discussion

5.5.1 Patch size and vegetation patterns

The range of analyses presented here support the proposition that for the study area, floristic composition and vegetation structure differed according to patch size. Specifically, the PCO analysis revealed that on average, small patches were more productive than medium-large patches, the former being characterised by higher total species richness; higher richness of trees, tree/shrubs, shrubs, and vines; higher richness of rainforest species and resprouter species, as well as by higher litter cover. Structural differentiation was also apparent, with mid-stratum and upper-stratum height being positively associated with small patches. Moreover, the PCO analysis revealed that higher *M. kekwickii* cover was weakly associated with the small patches (and thus with higher species richness etc.) and was the opposite for *A. shirleyi*. The floristic distinctiveness of small patches and the relative uniformity of medium and large patches implied by the PCO analysis was confirmed by the accompanying PERMANOVA and CAP analyses.

In spite of the relatively small sample size, the floristic patterning reported here is closely consistent with previously documented richness and structural differences between bullwaddy and lancewood vegetation types (Northern Territory Government unpublished data). Thus, two major findings arise from this study: 1) that community-scale partitioning was apparent in relation to woodland patch size, and 2) that small woodland patches characteristically supported the thicker stands of bullwaddy and the highest concentrations of species, including resprouter species and rainforest trees, shrubs, and vines.

5.5.2 Patch size and vertebrate animal patterns

Consistent with the findings for vegetation patterns, the analyses carried out on vertebrate wildlife showed that the composition and structure of vertebrate assemblages differed according to patch size. Smaller patches had a higher richness of vertebrates than medium and large patches. Whereas average species richness in large patches was 18 species (range of 13—23 species) and in medium patches 19 species (range of 15—29 species), average vertebrate species richness in small patches was 27 species (range of 20—33 species).

The vegetation analysis characterised small patches as having thicker stands of bullwaddy, higher total plant species richness, higher richness of trees, tree/shrubs, shrubs and vines, higher richness of rainforest species, as well as higher litter cover. The characteristic vertebrate species of small patch sites identified by SIMPER analysis have habitat requirements that align with these compositional and structural attributes. As an example, among reptiles, five species of gecko (*Amalosia rhombifer*, *Gehyra gemina*, *Heteronotia binoei*, *Strophurus ciliaris*, *Lucasium immaculatum*) were characteristic of small patches whereas no geckoes were characteristic of medium-large patches. Among birds, a number of species with a preference for denser cover and/or with rainforest affinities had higher abundance in small patches. These species include the variegated fairy-wren (*Malurus assimilis*), Arafura fantail (*Rhipidura dryas*) and grey shrike-thrush (*Colluricincla harmonica*).

5.5.3 Species—environment relationships

The constrained linear modelling (DISTLM and dbRDA) confirmed patch size as the most important environmental correlate of floristic gradients in the study area. Likewise, patch size was the most important environmental correlate of vertebrate fauna gradients across the study sites.

As in the PCO, small patch size in this analysis was positively associated with overall species richness, richness of trees, shrub/trees, shrubs and vines and resprouter and rainforest species. This congruence between the different types of analysis (unconstrained and constrained ordination) allowed for high confidence in the implied relationships. Notably too, in the RDA, the cover abundance of *M. kekwickii* was more strongly correlated with small patches. An acknowledged limitation of this study was the spatial configuration of the different patch size samples – viz. there was no opportunity to sample small patches along the north—south transect. This issue was partially addressed by including site location in the analysis. Importantly, while site latitude and longitude were included in the best model, neither variable correlated with patch size, and the amount of variability explained by each was minimal compared to that of patch size. This result allows for confidence in the implied vegetation—patch size relationship; however, increased sampling across the broader region is required to enable generalisation beyond the immediate study area.

Patch size itself is both an attribute of vegetation and an established surrogate for fire and weed incursion risk – small patches have a high perimeter-to-area ratio and this creates more opportunities for negative 'edge' effects. In this study, the dbRDA depicted a comparatively strong relationship between floristic gradients, patch size and fire frequency. Specifically, the small patches that were often characterised by bullwaddy thickets with relatively low *A. shirleyi* cover, were shown to have a history of frequent fire exposure. While the result aligned with the expectation for higher fire frequency in small versus larger patches, it is otherwise hard to reconcile in relation to bulllwaddy *per se*, given the expected sensitivity of this vegetation type to short-interval fire. Importantly too, the derived fire data did not align with field observation which instead suggested that *M. kekwickii* stands very often avoided fire, even in cases where fire carried into immediately adjacent *A. shirleyi* stands (Figure 15). In view of this, the relationship depicted in the dbRDA is interpreted as an artefact of the course-scale fire-scar mapping layer that was used to derive the site data. As such, the fire frequency scores used here more likely pertain to the surrounding flammable savanna matrix within which the small bullwaddy patches were embedded. Potentially therefore, in the region of the east—west transect (where the small patches were concentrated), high frequency landscape fire acts to limit *A. shirleyi* canopy development, along with opportunities for woodland patch expansion. This issue, and the impact of fire more generally on woodland patch size and composition, requires resolution through detailed fire-vegetation analysis of the wider region of the study area.

Time-since-fire had relatively low overall explanatory power in the environmental model; however, this variable was instructive in relation to gradients of canopy cover in both of the woodland dominants. For *A. shirleyi*, this result was consistent with field observation of canopy loss and death in many individuals from recent fire. The effect was most apparent at Site 1 which was strongly dissociated from *A. shirleyi* cover along Axis 2 of the RDA. The result is important because it is consistent with published work that shows that frequent high severity fire strongly disadvantages *A. shirleyi* (Russell-Smith et al. 2010). The results thereby confirm the vulnerability of *A. shirleyi* to frequent hot late-season fire in the study area and more widely in the Beetaloo Sub-region. It strongly suggests a need for improved fire management across the region. As discussed, *M. kekwickii* appears to have a propensity to avoid fire and it is a known resprouter. Fire impacts on this species were correspondingly less apparent from field observation.

Only two introduced plant species were recorded from the survey, *Bidens bipinnata* and *Stylosanthes viscosa*. Neither species occurred in very high abundance (i.e., exceeding 3% cover) at any sampling site. Notably, their occurrence was shown to be most closely associated with the small patches which is consistent with the 'edge effects' expectation. However, neither species is a significant environmental weed in that they are not known to change fire regimes or strongly outcompete native plant species. In this regard, all surveyed stands were considered to be in relatively good condition. Similarly, no impacts of feral animals were recorded.

5.5.4 Risks to biodiversity values from landscape fragmentation

Bullwaddy (Figure 26) has inherent conservation value as a Northern Territory endemic vegetation type. Its occurrence alongside contrasting eucalypt savanna increases landscape scale habitat diversity, and other ecological values stem from its dense, shady canopy and its low flammability which creates pyrodiversity in an otherwise highly flammable landscape. Bullwaddy is recognised as refuge for climate-sensitive and fire-sensitive species, most particularly, a suite of rainforest plant species. Despite the small sample size used in this present study, strong support was found for a positive relationship between bullwaddy canopy development and plant species and functional trait diversity.

Lancewood stands are recognised as having comparatively low fauna habitat value and relatively low plant species richness – the latter attribute also being confirmed by this present study. Their value primarily lies at the landscape scale, in that like bullwaddy, they created a diverse mosaic of vegetation community types in an otherwise uniform landscape. There is a long-standing recognition of the requirement for better fire management for the retention of stand structure (canopy maintenance) and the ecological functionality of this prominent vegetation type (Russell-Smith et al. 2017). This study provides additional support for a requirement for improved fire management in relation to lancewood.

Under the Northern Territory Planning Scheme, clearing of any native vegetation in the Northern Territory requires consent and demonstrated adherence to good land management principles, particularly in relation to protecting soil, water and biodiversity values (https://nt.gov.au/property/land-clearing). For bullwaddy and lancewood, there is poor knowledge around sustainable thresholds for change in relation to land clearing and fragmentation. In this context there is a generalised requirement for the minimisation of habitat degradation, fragmentation and loss associated with any clearing of these habitats. Risks persist especially in relation to the potential for increased fire and weed incursion into bullwaddy and lancewood patches if they are reduced in size through the creation of road networks. With habitat fragmentation, an increased rate of fire incursion of bullwaddy and lancewood could initiate a positive fire-feedback loop, whereby these communities become increasingly fire prone and unsuitable as habitat for disturbance sensitive/shade intolerant species. The present study indicates that reduced canopy cover of the dominant species through fire would likely result in local declines in species richness. In this context, there are potentially multiple biodiversity benefits that could arise from improved landscape fire management in the region. Specifically, strategic early dry season burning that reduces the frequency and spatial extent of unplanned damaging late dry season wildfire is known as being especially beneficial for fire-sensitive woodland vegetation (Russell-Smith et al. 2017).

Road development can result in the spread of introduced plant species into previously intact habitat. Novel plant species can alter vegetation community structure by outcompeting and displacing native plants. Introduced plants can also significantly increase fuel loads, resulting in more intense fire events (Setterfield et al. 2010). As high-disturbance linear networks, roads and tracks become the source of invasives into newly disturbed habitat. *The Final Report of the Scientific Inquiry into Hydraulic Fracturing* made specific recommendations in relation to weed mitigation and management in the context of onshore gas exploration and production. This present study, and the SREBA (Department of Environment, Parks and Water Security 2022) established that weed impacts are presently minimal in bullwaddy and lancewood woodland, but that the former potentially has higher susceptibility.

Figure 26 Bullwaddy, *Macropteranthes kekwickii*, in flower within Bullwaddy Conservation Reserve, February 2023

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