12/22

Soundscapes of Sumatra:

AN ANALYSIS OF THE SUSCEPTIBILITY OF VOCAL FAUNA TO ECOLOGICAL EDGE EFFECTS WITHIN A HISTORICALLY DISTURBED LOWLAND FOREST SUMATRA, INDONESIA.

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Masters by research
Bournemouth University
December 2022

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Abstract

Globally between 2000 and 2015, 128.4 million hectares of native forest were destroyed through deforestation. Although Indonesia still has one of the highest levels of biodiversity globally, it has seen an increase in deforestation due to increased land-use change. In highly degraded forest areas, edge effects in combination with microclimate variation and anthropogenic disturbance can influence the species composition of both flora and fauna and can cause losses in biodiversity and ecosystem resilience. Soundscape analysis can assess the spatial and temporal variations in biodiversity and species composition within an ecosystem using its bioacoustics.

This study modelled the variations in the edge effects of forests structure, microclimate, anthropogenic disturbance, and biodiversity within a historically logged lowland forest in Sumatra. As well as the susceptibility of the diversity of vocal fauna and two endangered gibbon species *Hylobates lar and Symphalangus syndactylus* to the variations in microclimate forest structure and anthropogenic disturbance. With the aim of understanding the impacts of edge effects in degraded forests and highlighting the key ecological drivers that can aid in the conservation of the gibbon species and general fauna within degraded lowland forests. Data were collected in at 2019 10 sites within a historically disturbed lowland forest in the Sikundur region of Sumatra. Each sites forest structure was visually surveyed and dataloggers and Open Filed Recorders were installed to collect microclimate and bioacoustics data continuously for 3 months.

The analyses showed no correlation between forest structure and distance from the forests edge but a relationship between microclimate and distance suggesting presence of the buffer effect of the forest canopy. Anthropogenic disturbance from machine hums decreased further into the forest but had no impact on the acoustic diversity or presence of gibbon species. Maximum acoustic diversity at dawn was negatively correlated with distance to forest edge, but dusk showed no trends. Similarly, no relationship was found between the distance from the forests edge and the diversity at both dawn and dusk. Interestingly, the median daily acoustic diversity showed a positive relationship to distance from forest edge. The presence of the gibbon species was not impacted by distance from the edge but the length of their stay at a site was.

The results show a distinct susceptibility of general diversity to temporal variations in microclimate that lead fauna to depend on trees with a large crown area as a shaded microhabitat and source of food. Similarly, call density of both *Hylobates lar and Symphalangus syndactylus* correlated with the presence of large canopy connectivity. This reflects the importance of large trees for resilient ecosystems and highlights the importance of conserving them in disturbed forests.

Acknowledgements

I would like to thank my supervisors Professor Amanda Korstjens, Dr Phillipa Gillingham, Associate Professor Tom Davis, and Professor Richard Stillman for all their help and guidance throughout this project. I would also like to thank Bournemouth University for their support throughout the course and project. Finally, I would like to thank Dr Helen Slater for her data collection for this project and her guidance on the research and use of R studio.

Author's declaration

I hereby declare that the data used in this research were derived from raw acoustic data collected by but by Dr Helen Slater. The data derived from those recordings were collected and analysed by myself and the research project is itself individual.

Introduction

Biodiversity loss and extinction of species are increasing globally, contributing to this planet's sixth mass extinction event (Ceballos et al., 2010; Thomas 2004). Many of the causes of biodiversity loss are related to anthropogenic stressors such as: high levels of disturbance (Barlow et al., 2016); habitat fragmentation and deforestation (Giam 2017; Maitima et al., 2009); and climatic changes (Dawson et al., 2011). Biodiversity loss has been found to double in deforested areas that have high levels of anthropogenic disturbance (Barlow et al. 2016). Globally, between 2000 and 2015, 128.4 million hectares of native forest were destroyed through deforestation (Keenan et al. 2015).

With increasing fragmentation and deforestation, edge effects in combination with climate change can influence forest animals. The edge effect first defined by Odum in 1958 is the tendency for population density and species richness of fauna to increase at the zone between two habitats (Odum in 1958). Forest fragmentation increases the number of edges and causes the subdivision of large forest areas into smaller fragments (Laurance, 2000). It has been shown to cause decreases in biodiversity through tree mortality (Alencar et al., 2004; Cochrane and Laurance 2002) and changes in the species composition of both flora and fauna (Benitez-Malvido and Martinez-Ramos 2003; Cushman 2006). This may be due to increases in predation at edges (Herrerías-Diego 2008; Sosa and Lopez de Casenave 2017), easier access for humans to the interior forest, causing increased anthropogenic disturbances such as resource extraction or poaching (Peres 2001), increased temperatures (Stangler et al., 2015) and carbon emissions through loss of larger tree species (Laurance et al., 2000).

Within intact forest there are more shaded areas than in logged forests, microhabitats created by shading from forest canopies can buffer temperature, reducing heat exposure by up to 10 °C (Scheffers et al. 2014). Many mobile animals rely on these microhabitats to maintain optimum temperature (Morecroft 2014). When forests are logged, there is a reduction in shaded areas and microclimates alter (Stangler et al., 2015) reducing the impact of the microclimate-buffering effect (Ewers and Banks-Leite 2013) and reducing the availability of cooler microhabitats (Marsh et al. 2022) that fauna rely on. This intern increases the impacts of climate change on fauna. Species with slow life histories such as primates are especially vulnerable to the impacts of rapid climate change due to a lack of time for potential genetic adaptions forcing rapid behavioural changes or causing losses to the population (Quetglas et al. 2016).

Indonesia: Although Indonesia has one of the highest levels of biodiversity globally (Koh and Ghazoul 2008) it has seen an increase in deforestation, losing over 6.02 Mha from 2000 to 2012 (Hansen et al. 2013; Margono et al., 2014). This is due to increased land-use change, for plantations (Abood et al., 2015) and production of rubber and palm oil (Koh and Wilcove 2008). The island of Sumatra has increasing numbers of rubber and palm oil plantations (Miettinen et al., 2011) and due to this has lost 2.9 million hectares of forest between 2000 and 2012 (Margono et al., 2014). Sumatra is inhabited by 22 primate species, nine of which are endemic (Roos et al., 2014). Many of these are vulnerable to anthropogenic disturbance through logging and hunting, including gibbons (Yanuar and Chivers 2010). Documented reports on the loss of forest cover and primate habitat within Sumatra (Mariati et al., 2014; Supriatna and Mariati 2014) have been ignored by the government and private sector, causing a need to raise conservation practices for vulnerable species.

Tropical Fauna

Hylobatidae: The canopy structure of tropical forest is related to the evolution of the fauna within it (Fleagle 2013), with many primate species evolving morphological and behavioural adaptations for arboreal habitation (Fleagle and Lieberman 2015). As Hylobatidae are adapted to extremely arboreal lifestyles and have a slow maturation rate of up to 9 years (Geissmann, 1991; Reichard and Barelli 2008), they are vulnerable to disturbances in forest structure (Phoonjampa et al., 2011) and thus may be affected by edge effects. Virtually all gibbons are listed as endangered on the IUCN red list, including both *Hylobates lar* and *Symphalangus syndactylus* (Brockelman and Geissmann 2020; Nijman et al., 2020).

Hylobatidae live in small territorial social groups ranging between 2 and 7 individuals and occupy annual home ranges of 15–60 hectares (Bartlett 2011; Brockelman et al., 2014). Ecological factors such as the production of important fruit species corelate to home range size (Savini et al., 2009) suggesting food availability is a key factor influencing the presence of Hylobatidae in an area. Home ranges with constant food availability are normally defended for extended periods of time (Brockelman et al., 2014) and are not usually susceptible to changes in group members if the group size remains consistent (Koda et al., 2012).

As gibbons are frugivores and particularly important seed dispersers (McConkey 2000; McConkey & Chivers 2007, McConkey 2009) the taxon has an ecological role in sustaining tree populations. However, gibbons are susceptible to changes in tree phenology (Brockelman 2011) as variations in tree phenology impacts seasonal variations in fruit availability (Suwanvecho et al., 2018). Variations in plant phenology have been linked climate change (Yang & Rudolf 2010) therefore gibbons may be impacted by microclimate variation and edge effects. Fruit sources for Hylobatidae encompass a variety of structures, such as vines, understory, and large canopy trees (Suwanvecho et al., 2018) each vary in the abundance of fruit. When foraging, groups of gibbons can travel a distance up to 3000m per day around their home range (Bartlett 2009). Due to the high mobility of gibbon species, Hylobatidae have the capability to decide to move towards a new food source several hundred meters away (Asensio et al., 2011) and the need to communicate over large distances (Asensio et al., 2014).

Primates with slow life histories, large home ranges and frugivorous diets such as members of the genera *Hylobates* and *Symphalangus* are particularly susceptible to alterations of habitat (Boyle and Smith 2010). Previous studies have investigated relationships between forest structure and the presence of H*ylobates lar* and *Symphalangus syndactylus,* but very few have investigated the impacts of variation in microclimate. Hankinson et al. (2021) found that canopy connectivity and height to the first bole were significant predictors for the continued presence of the sympatric *Hylobates lar* and *Symphalangus syndactylus* in Sikundur, a historically disturbed lowland forest in Sumatra*.* The study found the density of *Hylobates lar* and *Symphalangus syndactylus* was positively influenced by the median height of first bowl and the percentage of canopy connectivity. *Hylobates lar* group density also exhibited positive correlations to diameter at breast height between 30–100 cm, tree heights between 20–25m, and crown areas between 100–300 m^2. However, little research has been done into how variations in microclimate, forest structure or human disturbance at different increasing distances from the forest edge impact the presence of these species, and with increased deforestation it is important to understand these how these edge effects vary in degraded forests and there impacts on Hylobatidae.

Avian species: Within Indonesia avifauna is also heavily under pressure of hunting for songbirds, and songbirds have seen a decrease in numbers (Leupen et al. 2020). Within fragmented forests the predation of avian nests increases nearer the forests edge (Keyser 2002; Sosa and Lopez de Casenave 2017). This may be due to the edge being prime habitat for generalist predators (Robinson and Wilcove 1994; Zanette and Jenkins 2000) and due to the nest being more visible at the edge (Sosa and Lopez de Casenave 2017). Increased fragmentation is therefore likely contributing to the decline in avian diversity resulting from an edge effect. However, for ground nesting birds no edge effect is found (Batáry et al., 2014).

Many studies have suggested that microclimate does not impact habitat selection for forest birds (Jähnig et al., 2020; Pollock et al., 2015). However, Jähnig et al., (2020) showed microclimate having an impact on grassland ground nesting birds. These findings can be linked to insectivore understory birds due to their prey being selective of microclimate and food source being a greater driver for habitat selection than abiotic conditions (Pollock et al., 2015). Understory birds have also been found to avoid areas of high light intensity (Pollock et al., 2015). There is evidence of microclimate Impacting the habitat preference of montane forest birds on a species-specific level, with 53.3% of species being warm associated 46.7% being cool associated (Frey et al., 2016). Within forests with elevated levels of anthropogenic disturbance the species richness of frugivorous avian species has been found to not be impacted by disturbance, but the species composition has (Gomes et al., 2008). There is also evidence that larger frugivore birds are tolerant to some limited disturbance but intolerant to higher levels of disturbance, while smaller frugivore avian species often have a higher tolerance to disturbance (Gomes et al.,2008).

Avifauna are grouped into feeding guilds to determine the impacts of disturbance, the edge effect, habitat selection, community composition (Gray et al., 2007, Edwards et al., 2009) and functional diversity (Flynn et al., 2009; Hidasi-Neto et al., 2012; Edwards et al., 2014). Within secondary forests the main feeding guilds tend to be frugivores and insectivores (Azman et al., 2011). Frugivorous bird species are vulnerable to variation in the phenology of fruiting trees (Suwanvecho et al., 2018; Brockelman 2011; Dhondt 2011) and therefore may be susceptible to climatic and forest structure edge effects. There is still limited knowledge of how edge effects influence avifauna in tropical forests, where edges can be sharply defined. And with songbirds seeing a decrease in numbers (Leupen et al. 2020) research into the taxa's vulnerabilities is crucial to its conservation.

Amphibians: Amphibians are experiencing an extinction crisis (Meredith 2016) due to their multiple vulnerabilities, especially to climatic changes (Kiesecker et al., 2001, Baillie et al., 2004). As most amphibians have low vagility they are more susceptible to the increasing impacts of habitat fragmentation and destruction (deMaynadier and Hunter, 2000, Bowne and Bowers, 2004). Other issues causing their decline include climate change, environmental pollution through anthropogenic disturbance and the introduction of invasive species (Bridges and Semlitsch, 2000, Davidson et al., 2001, Stuart et al., 2004, Kiesecker et al., 2001).

Invertebrates: The prey of many of avifauna and amphibians are invertebrates (Azman et al., 2011; Le Roux et al., 2018; Attard, 2011) giving these taxa a key role on the trophic scale. Invertebrates are also responsible for most of the pollination within a forest (Taki 2017). This gives the taxon a key role in sustaining the ecosystems through plant reproduction (Ghazoul and McLeish 2002; Murren 2002; Quesada et al. 2004) but also makes them susceptible to changes in plant phenology (Ulrich et al. 2020; Warner et al., 2021). As the phenology of plants is shifting in response to climate change, shifts in phenological events such as the advancement of spring and autumn postponement are created (Gill et al. 2015; Jeong et al., 2011). This causes the phenology of many species to become unsynchronised across the trophic levels (Renner & Zohner, 2018) and may be contributing to losses in invertebrate species.

Invertebrates depend on thermoregulation and can sense variation in their ambient temperature (Ni et al., 2013). To avoid heat stress, invertebrates utilise thermal microhabitats (Pincebourde and Woods, 2012;

Woods et al., 2015) altering their distribution within an ecosystem (Caillon et al., 2014; Kührt et al., 2005). Therefore, invertebrates are susceptible to changes in microclimate caused by habitat fragmentation. However, not much research has investigated the impact of edge effects on invertebrates and the drivers to their presence. As a food source for many faunae the conservation of invertebrate species may aid in conservation of overall biodiversity.

Bioacoustics

Bioacoustics Monitoring: Bioacoustics are the acoustic signals made by living organisms, these signals are used by multiple terrestrial species including avifauna, amphibians, mammals, and invertebrates (Rogers and Kaplan 2000; Gerhardt and Huber 2002). Fauna across many orders use vocalisations for an array of purposes, such as detecting a mate (Bardeli et al., 2010) and alerting group members of predators (Ramakrishnan and Coss 2000). Ecosystems such as tropical rainforest contain some of the greatest acoustic diversity on the planet (Planque and Slabbekoorn 2008) and have therefore been the basis of many studies. The monitoring of bioacoustics has been used in ecological studies to estimate population size (Wood et al. 2019), changes to community structure and for discovering the impacts of anthropogenic disturbance on an ecosystem (Joo et al., 2011; Deichmann et al., 2017). Previous terrestrial bioacoustics studies have evaluated the impacts of hydrocarbon operations on avifauna within North America (Francis et al., 2010) and Proboscidea in Gabon (Wrege et al., 2010).

Hylobatidae vocalisations: Gibbon vocalisations are used for communication with neighbouring groups and individuals, sharing resources, territorial defence, mate defence, as well as mate attraction (Lappan & Whittaker 2009; Cowlishaw 1992). Vocalisations can travel distances of 1-2km (Fan et al., 2010) and within *Hylobates lar* these distances are further than the species' average home range, suggesting an evolutionary selection for long-range communication (Asensio et al., 2014). *Hylobates lar* favour vocalising from high trees in order to gain maximum distance to the call (Okuda et al. 2022) by avoiding interception to the soundwave by dense vegetation (Fang et al. 2003). One of the most studied gibbon vocalisations is the morning call, which is performed by most gibbon species (Geissmann 2002, Marshall and Marshall 1976). It is these morning calls that are used in studies on gibbon presence and habitat preference as they indicate a gibbon has been staying overnight in an area.

Avifauna vocalisations: Avifauna use vocal signalling for territorial defence, mate attraction, social interactions and to avoid predators (Brumm and Zollinger 2013). The majority of avian vocalisations take place at dawn and at dusk (Marín-Gómez 2021), this is most likely to fill the temporal niche of other vocal fauna at that time and because it is when birds are most active (Marín-Gómez et al., 2020). Evidence suggests that avifauna also use temporal segregation of vocalisations to avoid overlap with other bird species (Luther 2009). There are many studies that use the detection of these vocal signals to determine the abundance, species richness and composition of avian species (Acevedo et al., 2009; Agranat, 2009; Brandes, 2008; Somervuo et al., 2006). However recent studies suggest that the disruption of the avian chorus is due to of increasing levels of anthropogenic disturbance (Pijanowski et al. 2011; Marín-Gómez 2021).

Amphibian vocalisations: Some studies have used amphibian vocalisations to determine distribution and abundance of the taxon (Brandes et al., 2006; Hu et al., 2010) but there is still a need for further research (Bowne and Bowers, 2004), especially for the elusive tropical amphibians. The taxa also often have low habitat tolerance and are therefore impacted more greatly by the effects of habitat loss, derogation, and the edge effect (Semlitsch, 2000, Houlahan and Findlay, 2003).

Invertebrate vocalisations: One of the most vocal invertebrate species within a tropical forest are cicadas (*Hemiptera; Cicadidae*) (Hart et al., 2015) with the vocalization of some species having a pressure greater than 100db (Sanborn and Phillips 1995). These species have been proven to interfere with bioacoustics studies and the vocalizations of avifauna (Hart et al., 2015) as they encompass an array of frequencies portioning acoustic space (Sueur 2002) causing signal masking. It is therefore important within a bioacoustics study to account for the presence of *Cicadidae.*

Soundscape ecology

A non-invasive and relatively cost-effective way to measure biodiversity and edge effects is to use passive acoustic recordings to capture the soundscape of an area. A soundscape consists of the biological, geophysical, and anthropogenic sounds emitted from a landscape over a period of time (Pijanowski et al.,2011a). Variations in soundscapes can reflect ecosystem processes and anthropogenic activity (Pijanowski et al., 2011a). Soundscape ecology is fundamentally built up of four foundations spread across different intellects (Figure 1) (Pijanowski et al., 2011b), Spatial ecology, Psychoacoustics, Bioacoustics, and Acoustic ecology. Soundscape analysis can determine the spatial and temporal distribution of biodiversity, the richness and composition of the community, and the acoustic activity of taxonomic groups or individual species (Andrews and Dick 2021). It has also been used to analyse the impacts of human disturbance (Deichmann et al., 2017).

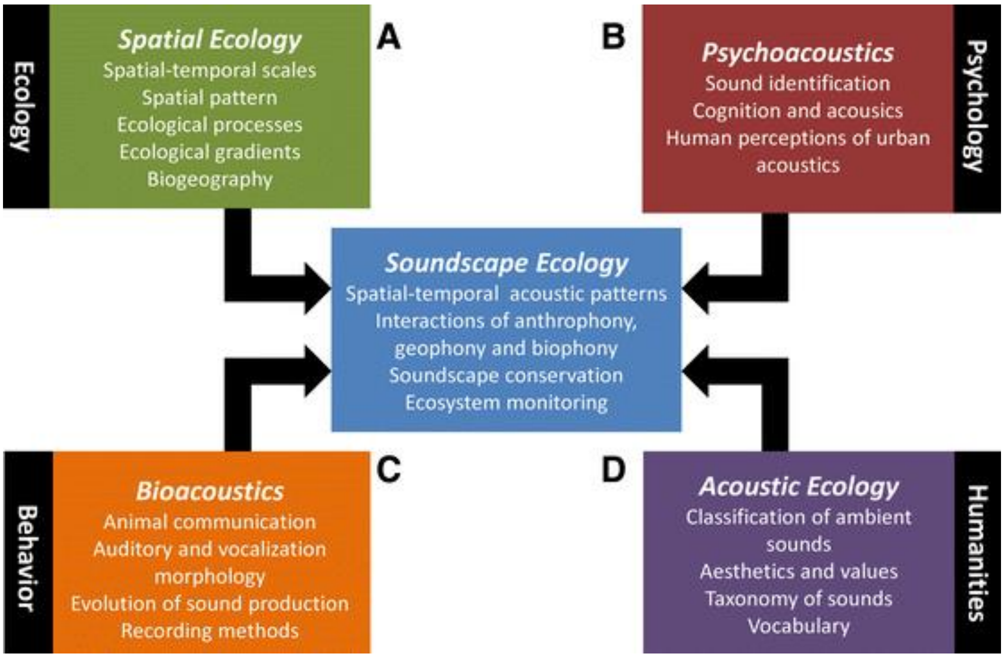

Figure 1: Intellectual foundations of soundscape ecology (Pijanowski et al., 2011b).

Acoustic partitioning and Acoustic diversity indices: To be heard vocalising, fauna partition temporal and frequency domains across a soundscape (Marín-Gómez et al., 2020; Metcalf et al., 2021). Examples of this can be seen in neotropical birds that partition their vocalisations across the frequency spectrum to avoid being masked by louder species such as cicadas (Aide et al., 2017; Hart et al., 2015). Research points to a broad negative correlation between the body size and the frequency at which fauna vocalise (Gillooly and Ophir, 2010; Seddon 2005; Wilkins et al., 2013), suggesting larger species such as mammals vocalise at lower frequencies and smaller species such as those in the order Orthoptera vocalise at higher frequencies. Figure 2, adapted from Metcalf et al. (2020), displays these temporal and frequency portions across taxa (Figure 2).

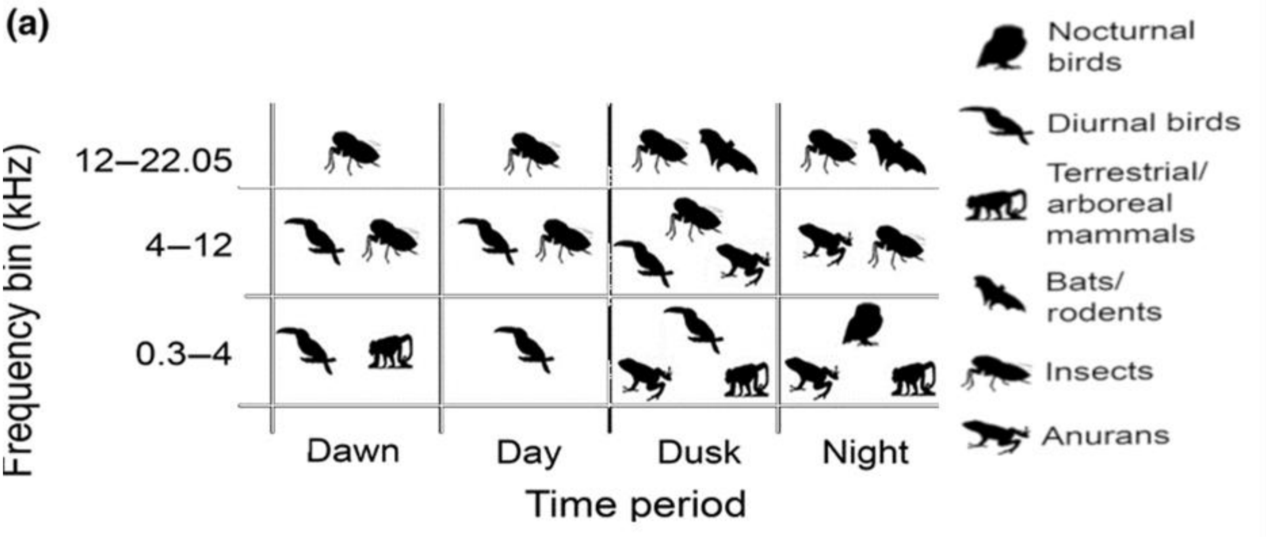

Figure 2: Adapted from Metcalf et al., 2021 a conceptual framework illustrating the expected acoustic niches of different taxonomic groups.

This acoustic niche partitioning suggests that more diverse ecosystems produce more diverse and complex soundscapes (Sueur et al., 2008). Acoustic diversity indices utilise this hypothesis by producing a quantitative measurement of acoustic complexity used to measure an ecosystem's diversity without any species identification. One of the more commonly used acoustic diversity indices is the Acoustic Complexity Index (ACI). First proposed by Pieretti et al. (2011), the index aims to portray variations in call frequency within a soundscape without being impacted by background noise. It achieves this by splitting the frequency range of the soundscape into bins and comparing the spatial and temporal variations in the amplitude and complexity (Pieretti et al., 2011). An environment with a high ACI is reflective of high diversity.

Acoustic diversity indices have been used for a range of ecological studies including monitoring long-term ecological changes (Phillips et al., 2018), calculating species richness (Eldridge et al., 2018), and impacts of disturbance on community structure (Gasc et al., 2018). However, there are limitations to some of these indices involving interference from anthropogenic and geophonic sounds as they correlate strongly with the indices and can interfere with the estimates (Towsey et al., 2014; Pieretti et al., 2015; Sánchez-Giraldo 2020). This makes measuring diversity in urban environments difficult without filtering the anthropogenic and geothermic sounds (Fairbrass et al., 2017). Metcalf et al., 2021 found acoustic indices can be most efficient when calculated at the ecologically appropriate time and frequency bins informed by an understanding of the soundscape, to reduce the impacts of signal masking from other non-target taxa (Metcalf et al., 2021). Currently, there is very little research using soundscape ecology to assess the impacts of ecological variation within disturbed forests, most studies use field observations which are far more time consuming.

Aim and objectives

Aim: To investigate the impacts of edge effects on forest structure, microclimate, anthropogenic disturbance, and diversity in degraded forests and discover key ecological drivers that are associated with high levels of diversity and the presence of *Hylobates lar, Symphalangus syndactylus.*

Objectives:

1. Analyse the variation in forest structure and microclimate at increasing distances from the forests edge.

2. Analyse the variation in soundscape ecology to determine levels of acoustic diversity (ACI), presence of *Hylobates lar and Symphalangus syndactylus* and levels of anthropogenic disturbance (machine hums) at increasing distances from the forest edge.

3. Determine the susceptibility of *Hylobates lar, Symphalangus syndactylus* and diversity of vocal fauna to the edge effects and uncover drivers associated with high levels of diversity.

Predictions:

Table 1: list of predictions and hypotheses for model outcomes based on current knowledge.

Driver	Prediction
Edge effect	No correlation between forest structure and distance from the forests edge will be found due to habitat fragmentation impacting the typical succession of a forest.
Edge effect	Microclimate will correlate with distance from the forest edge showing a decrease in daytime temperature and light intensity as distance increases, due to decreased penetration of solar radiation though the canopy.
Edge effect	The number of machine hums detected will decrease with distance from the forests edge.
Edge effect	The acoustic diversity will correlate with distance from the forest edge showing an increase in activity of vocal fauna further from the forests edge.
Edge effect	The presence of *Hylobates Lar* will increase at areas further from the forests edge.
Edge effect	The presence of *Symphalangus syndactylus* will increase at areas further from the forests edge.
Forest structure	The presence of both *Hylobates lar and Symphalangus syndactylus* will be higher at areas with high percentage of canopy connectivity and trees with a large height depth ratio (representative of more mature forest), providing good shelter from high temperatures and solar radiation, increased microhabitats, and access to resources.
Forest structure	During peak heat hours (11:00-14:00) vocal fauna will be more active in shaded areas with higher canopy connectivity and larger crown areas, due to heat stress.
Microclimate	The acoustic diversity will correlate with times and sites with lower temperatures and light intensity.
Microclimate	The presence of both *Hylobates lar and Symphalangus syndactylus* will correlate with sites of lower temperature and light intensity.
Anthropogenic disturbance	Fauna will be more vocally active in areas of less anthropogenic disturbance through machine hums.

Methods

Location: Data were collected in a historically disturbed lowland forest in the Sikundur region on the boundary of the Gunung Leuser National Park (GLNP) in the North Sumatra province of Sumatra, Indonesia. This area was logged until 2004 when it gained full legal protection under designation as the Gunung Leuser National Park (YOSL-OIC 27 2009). As the area is on the boundary of the National Park boundary, it is still subject to illegal logging and hunting of wildlife (Roth et al. 2020).

Study sites

In 2019 ten sampling points were randomly generated using ArcGIS Pro software (ESRI 2010), at increasing intervals ranging from 0m to 1600m from the forest edge running along 4 transects. Each point had a minimum of 100m between them. To allow for accessibility a location within 50m of the sample point was selected for collection of bioacoustics, microclimate, and forest structure data. The distance from each site to the forest edge was measured using satellite images and the GPS locations of each site. To standardise this, the shortest distance in a straight line, ignoring direction was measured from each site to the closest visible edge, using google earth's satellite images. Table 2 shows the GPS coordinates, distances from the forest edge for each site and the dates data were collected.

Table 2: Sample sites distance to nearest visible edge, GPS coordinates, and dates sound recording were taken within 2019. NR stands for no recordings.

Site name	Distance to nearest edge	GPS Coordinates	Dates recorded		
			August 2019	September 2019	October 2019
S1-0m	0m	98.08740666 3.978311294	14th, 15th, 31st.	1st, 2nd, 27th, 28th,	NR
S2-19m	19m	98.08522894 3.984640883	14th, 15th, 16th, 17th,	NR	3rd, 4th, 5th, 6th
S3-375m	375m	98.08201019 3.979372682	13th, 14th, 15th, 28th, 29th, 30th, 31st	24th, 25th, 26th, 27th	1st, 2nd, 3rd, 4th
S4-423m	423m	98.08001584 3.982771641	13th, 14th, 15th, 16th	26th	2nd, 3rd,4th,5th
S5-530m	530m	98.08351859 3.975575162	13th,14th,15th,16th,31st	1st,2nd,3rd,27th,28th	4th,5th,6th
S6-942m	942m	98.07699494 3.969715297	13th,14th,15th,16th	27th,28th,29th,30th	4th,5th,6th
S7-1087m	1087m	98.07644985 3.990067521	13th,14th,15th,30th	1st,2nd,26th,27th	3rd,4th
S8-1217m	1217m	98.07310324 3.978367661	13th,14th,15th,28th,29th,30th	24th	NR
S9-1497m	1497m	98.07083614 3.983991521	13th,14th,15th,16th	25th,26th,27th,28th	2nd,3rd,4th,5th
S10-1572m	1572m	98.07224441 3.989221523	13th,14th,15th,16th,30th,31st	1st,26th,27th,29th	NR

Survey methods

Audio devices: 10 of the Open Field Recorder (OFR) (Version 1.1) created by invisible flock (Invisible Flock 2021) were used for this project. Each OFR are fitted with time and location sync via GPS, a DPA 4060 microphone up to seven days battery life and is fitted with a protective cover designed to withstand tropical rainstorms. One OFR Audio recording device was attached to a tree at head height with the microphone pointing outwards at each of the 10 sample sites. All OFR devices were set to record continually until the battery ran out using a sample rate of 48khz and a bit depth of 24bitt. Due to reduced access to the sample sites and varied battery life in the OFR devices, data were collected intermittently from August through October 2019.

Forest structure: A visual survey of the forest structure in a 25x 25m plot surrounding each of the sample sites was conducted (Slater et al. Under review). Within each plot, the total number of trees with a diameter at breast height (DBH) of more than 10cm was recorded, and the following structural data were collected from each of these trees: Circumference at breast height (cm), Total height, Bole height, Crown connectivity %, Diameter at breast height (DBH) and Crown area (m2). The Height to DBH ratio (HDR) was calculated by dividing the total height by the DBH.

Dataloggers: To record the variation in microclimate HOBO UA-002–08 8 K Pendant Temperature/ Light Loggers were placed at each sample site to collect data on temperature (°C) and light intensity (lux). Each datalogger has an accuracy of±0.47 °C and was set to record the temperature and light intensity every 30 minutes. Each sensor was secured at approximately 1.5m high and placed in a well shaded area to prevent the greenhouse effect from direct sunlight (Marsh et al. 2022; Slater et al. Under review)

Data extraction

Soundscapes and ACI: A visual representation of each site's soundscape (Soundscape image) was created to visualise the frequency and temporal variation of acoustic activity at each location and help decide what frequency and time bins to use for the ACI. Using these images and Metcalf et al., 2021 frequency and time bin table (Figure 2), tables were created displaying each site's commonly occurring frequency and time bins and the taxa equating to the frequencies and time bins.

The soundscape images for each of the 10 samples sites' soundscapes were built using Arbimon RFXc's soundscape feature (RFCx Arbimon - Projects, 2022). The aggregation of each soundscape was set to time scale of hour in the day (24 hours), the frequency bin size of was set to the maximum of 84Hz and the minimum distance between peaks was set to 0Hz. This was to allow for the optimum number of species vocalisations to be captured within the image and allow for assessment of the full soundscape. An amplitude filter of 0.2 was used to ensure any vocalisations from species too far from the site were not included. All soundscapes were adjusted to control the different number of recordings at each site and each time interval. This was calculated by dividing the number of peaks in frequency at each hour by the total number of recordings at each site. The visual representations of the soundscapes were used to interpret the taxa present at each site by comparing the temporal and frequency bins that showed most activity.

These visual representations were then used to select the sample optimum frequency and temporal bins to extract the acoustic complexity indices (ACI). Plots of each sample site's ACI across 24 hours were also made using Arbimon RFXc soundscape data as well as boxplots of all the ACI data by site.

Pattern matching: The morning call vocalisations of *Hylobates lar* and *Symphalangus syndactylus* were located manually from within the audio recordings by searching through the morning time period of 6:00 to 10:00. The spectrograms of these vocalisations were then used to create a template for pattern matching (Figures 3 and 4). Using the Arbimon RFX pattern matching algorithm (RFCx Arbimon - Projects, 2022) the recordings from each sample site were searched for *Hylobates lar* and *Symphalangus syndactylus* vocalisations. As the morning calls of Hylobatidae occur between 6.00 and 10:00 am (Geissmann 2002) the pattern matching algorithm was set to search each site for morning calls between the hours of 6:00 to 09:59. All searches were set with an accuracy threshold of 0.2, an amplitude of 0.2 to avoid detection of species outside the sample site as gibbons vocalisations can travel of 1-2km (Fan et al., 2010).Each search were limited to 1 match per 60-second recording to represent presence rather than abundance as it is hard to distinguish individual calls.

All matches were then checked manually to ensure only *Hylobates lar* and *Symphalangus syndactylus* vocalisations were being logged and any unclear or faint vocalisations were removed. As gibbon call bouts can last between 10- 30 minutes (Geissmann 2000), it was difficult to separate bout events or individual gibbon calls, so only gibbon presence rather than abundance was accounted for. This was calculated using a binomial logging of presence at each site within the morning call range and a percentage of days present. To account for the large home range of the gibbons and length of stay in potential sleep spots an area a binomial of presents detected more than one consecutive day was recorded.

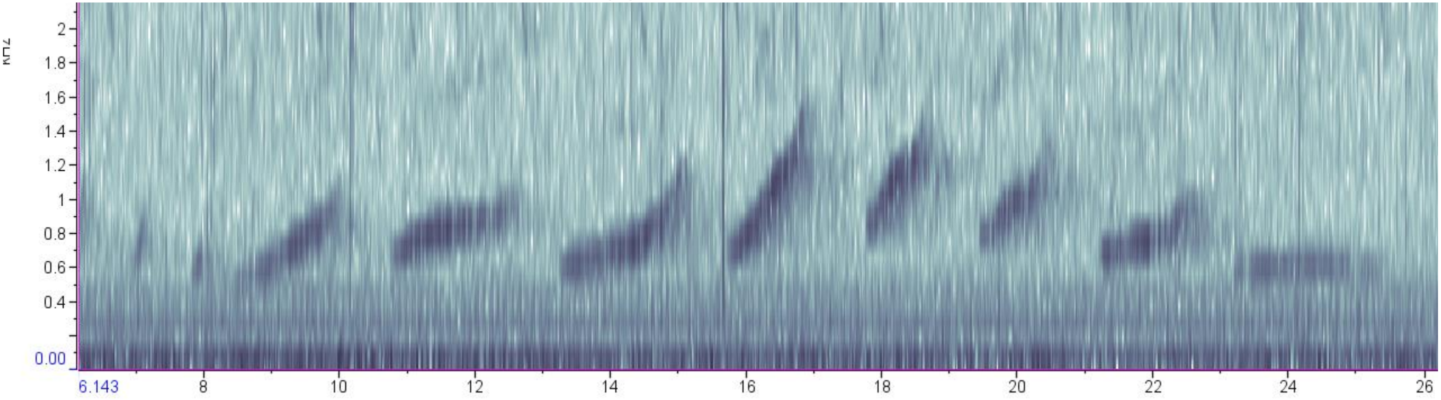

Figure 3: Spectrogram used for Hylobates lar pattern matching (20 Seconds).

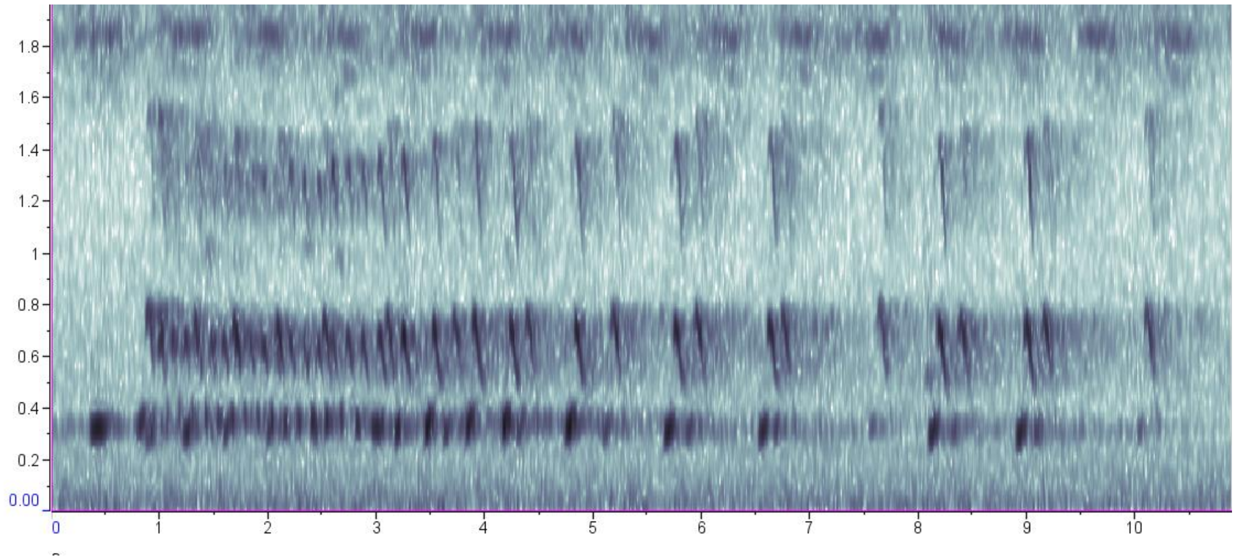

Figure 4: Spectrogram used for Symphalangus syndactylus pattern matching template (10 seconds)

Human disturbance: To quantify the level of human disturbance at each site, machine hums were used. Machine hums were clarified as any low frequency humming (0-0.5kHz) that continued without variation for longer than 3 seconds. Once some machine hums were located manually in the recordings the clearest spectrogram (Figure 5) was used to construct a template for Arbimon's pattern matching algorithm (RFCx Arbimon - Projects, 2022). A pattern matching search was then run for each of the sample sites throughout all hours of the day. Each search was set to an accuracy of 0.2, with an amplitude of 0.2 and was limited to 1 result per 60-second audio recording. As machine hum events can last between 3 seconds to over 60 seconds it was difficult to separate individual events within multiple recordings. So, a detection was recorded as the presence of at least one hum within a 60-second recording. All matches were then checked manually for false positives and any unclear or faint matches were removed. Boxplots were then plotted of the number of machine hum detections in each site.

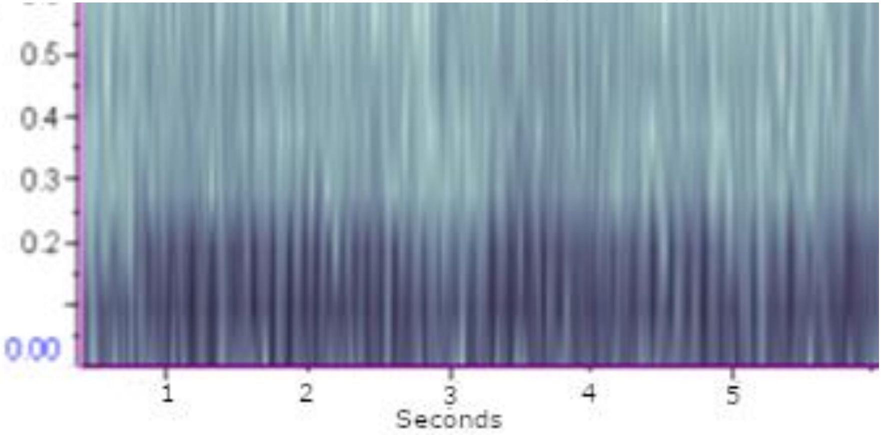

Figure 5: Spectrogram used for machine hum pattern matching template.

Microclimate: Daily maximum temperatures and light intensity were calculated for each site as well as averages for each hour on the dates matching the audio recordings. The daily range in temperature for each site was calculated by deducting the daily minimum temperature from the daily maximum. The temperature and light intensity of each site was plotted by time to display the variation by hour as well as boxplots displaying both the temperature and light intensity data by site. To standardise the same methodology as Marsh et al. (2022) was used and light intensity above 32000Lux was considered representative of direct sunlight hitting the datalogger (potentially causing a greenhouse effect) and was removed along with the corresponding temperature data.

Statistical analysis

All data were analysed using Rstudio (Rstudio Team,2022), the package ggplot2(Wickham 2016) was used for all figures and the lme4 package (Bates, et. al 2015) was used for all models.

Generalised linear models: A mixture of General linear models (GLM) and general linear mixed models (GLMM) were used to analyse the relationships between distance from the forest edge and each variable, as well as how the ecological variables (independent variables) impact both ACI and the presence of both gibbon species (dependent variables in separate models).

Transformation: All data for measurements of forest structure were standardised into cm and the rest of the data were standardised using a Z score. Any model that included both a forest structure variable and any other predictor was also transformed with a Z score.

Model selection: A correlation matrix was first run for all the variables in the dataset, looking for any variables that correlated with the dependent variables Distance, ACI, and Gibbon presence. Any correlated variables with a significant R>0.7 were never entered into the same model. Simultaneously, when choosing predictors for models, individual models were first run for each variable to look for strong relationships with the dependent variables before running models with multiple predictors. For any long models with multiple predictors the drop1 function in Rstudio(Rstudio Team, 2020) was used eliminating the predictors with AIC scores that suggested they do not explain the variation in the models.

Distance from the forest edge

Forest structure: To investigate which forest structure variables were impacted by the edge effect, the mean of each site's forest structure variables was calculated and run through a gaussian GLM with identity link function and distance as a predictor. GLM's where used with the forest structure data over GLMM as each site only had one observation for each variable, so the data set was too small to include a random variable such as day of the year. This multiple testing was necessary to uncover which variables may be varied due to distance and help inform which forest structure variables to use as predictors in the susceptibility models.

Microclimate: To check for correlations between distance from the edge and climatic variables, gaussian GLMMs with identity link function were run. The models used the daily results for max temperature, max light intensity and temperature range as dependent variables. The predictor for these models was distance and the day of the year was used as a random variable.

Machine Hums: To test the relationship between the number of machine hum detections and distance from the forest edge a GLMM was run using distance as the predictor and day of the year and number of days recordings were made at each site as random variables.

Gibbons and diversity: To assess the relationships between the presence of both primate species and distance from the forests edge binomial GLMMs (General Liner Mixed Models) were run using the daily presence of each species as the dependent variable, with 1 representing a detection on that day and 0 representing no detections. The predictor of these models was the distance, and the random variable was set as day of the year. To assess the relationship between the daily mean ACI and the distance from the forest's edge a gaussian GLMM was run with daily mean ACI as the dependent variable, distance as the predictor and day of the year as a random variable. To see if the distance from the forests edge was impacting the ACI at dawn and dusk GLMMs were run using dawn ACI and dusk ACI as variables, distance as the predictor and day of the year as a random variable.

Susceptibilities and drivers

Forest structure impact on Gibbon presence and diversity: To look for relationships between forest structure and presence of both gibbon species and to compare this project's findings with the findings of Hankinson et al. (2021), a binomial GLM was used, with 1 if the species was observed for more than one consecutive day and a 0 if no detections were made on consecutive days. Each model used one of the forest structure variables as a predictor, once run, any predictor variables that had no significant effect had its AIC checked with drop1 and then was removed. This process was repeated until each model was significantly explained by its predictors. To test forest structures impact on ACI, gaussian GLMs were run as above for each forest structure variables as a predictor and removing any that did not explain the model.

Microclimate impact on ACI and presence of gibbon species: To see whether microclimate affects the diversity of vocalising taxa at each site, gaussian GLMMs were made using the daily mean ACI from each site as the dependent variable and daily max temperature, daily light intensity and daily temperature range as predictors, with the day of the year as a random variable. To see if microclimate affects the presence of *Hylobates lar* and *Symphalangus syndactylus,* binomial GLMMs with logit link function were used with 1 representing a detection of the species on a day and 0 representing no detection that day. Each model used max temperature, Temperature range and max light intensity as predictors and the day of the year as random variables.

Results

Soundscape images showing sound variation throughout the day

The soundscape images for each site (Figures 6 to 15) visualised the frequency and temporal variation of acoustic activity at each location and helped decide what frequency and time bins to use for the ACI. The colour scale on each image is representative of the percentage of recordings the acoustic activity occurred within. Tables 3-12 display each site's commonly occurring frequency bins at dawn (6:00-10:00) day (10:00-18:00) and dusk (18:00-21:00) and the taxa equating to the frequencies and time bins was created (Tables 4 -13).

The soundscape images at all sites show a spike in acoustic activity at dusk between 18:00 and 21:00 that falls within most frequency bins between 0.3kHz and 12kHz. At each site, the greatest acoustic diversity is exhibited at night between 18:00 and 06:00 on frequency bins between 3kHz and 6kHz which S7-1087m displays throughout all hours in the 24hr period. All the sites exhibit low levels of acoustic activity at dawn 06:00 to 09:00 between 0.3kHz and 4KHz, of which S7-1087m, S9-1479m and S10-1572m display the most acoustic activity. Each site displays acoustic activity at night between the hours of 23:00 and 03:00 in the frequency bins between 8kHz and 10kHz, this is at the highest levels at sites, S5-530m, S7-1087m, S81217m, S9-1479m and S10-1572m. There are low levels of acoustic activity within the frequency bins above 12kHz during the dawn and daylight hours at all sites except S4-423m and high levels during dusk and night hours at S5-530m and S10-1572m.

Within S1-0ms soundscape there are high levels of acoustic activity at low frequencies <1kHz during daytime between the hours of 8:00 and 18:00 representative of high levels of machine hums. From these images the time ranges of dawn 6:00 to 9:00, day 10:00 to 17:00 dusk 18:00 to 21:00 and frequency bins of 0.3kHz to 12kHz were selected for the extraction of the acoustic complexity indices. This was as dusk was the most acoustically active time with at least 3 taxa vocalising, giving an ACI representative of the higher levels of biodiversity. Dawn was a less acoustically active time but still had at least 3 taxa vocalising, representing time and frequency bins with high levels of vocalisation but less signal masking from cicadas. Day was selected as a general representation of peak heat intervals displayed in the climate time data.

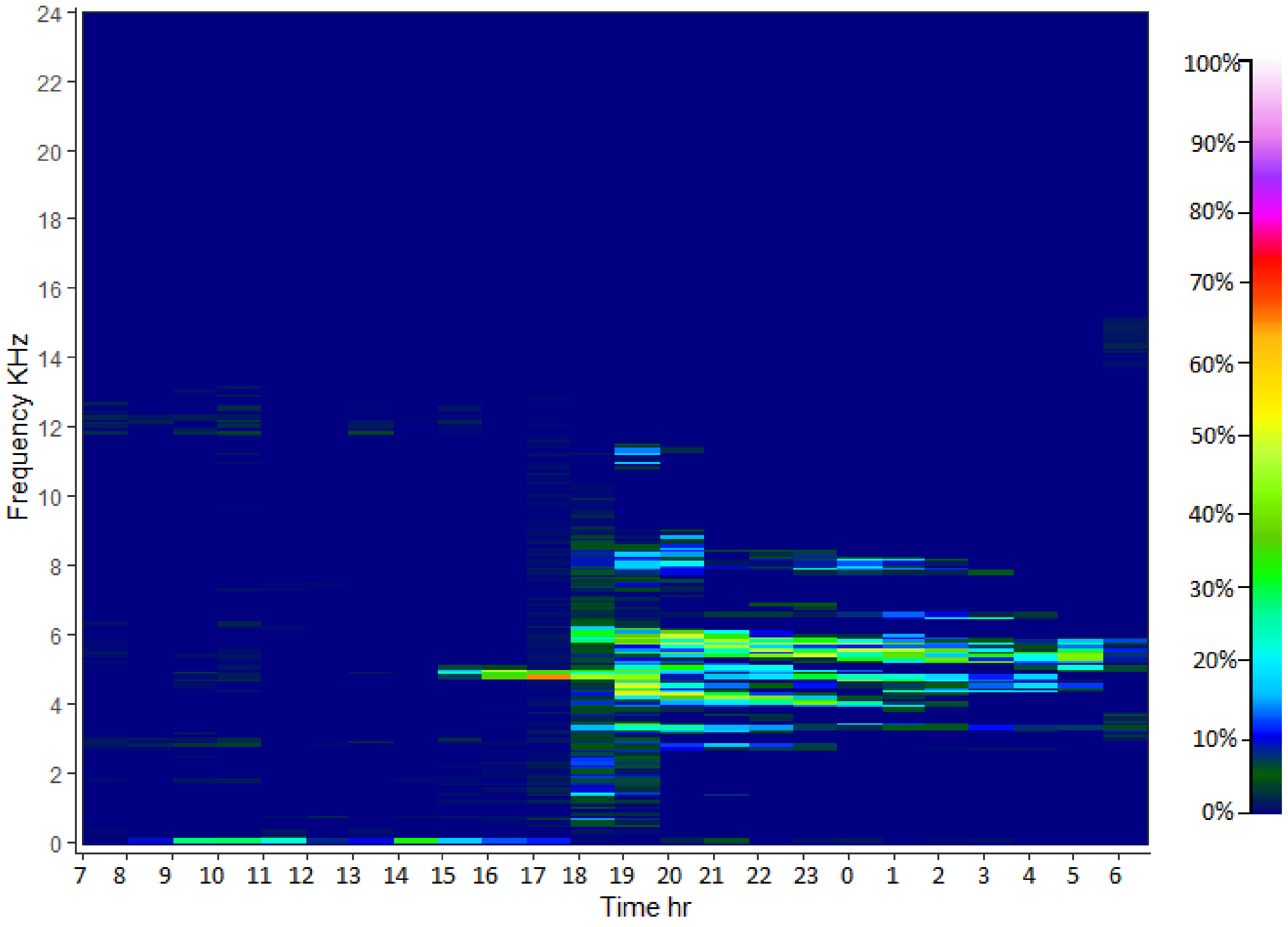

Figure 6: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S1-0m. Colour scale representative of the percentage of 60-second recordings acoustic activity occurred in.

Table 3: S1-0m commonly occurring frequency bins (<5 %) in each time Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	300-4000Hz	300-4000Hz	300-4000Hz 4000-12000Hz
TAXA	Avifauna, Arboreal, mammals	Avifauna	Amphibians, Avifauna, Arboreal mammals, Invertebrates

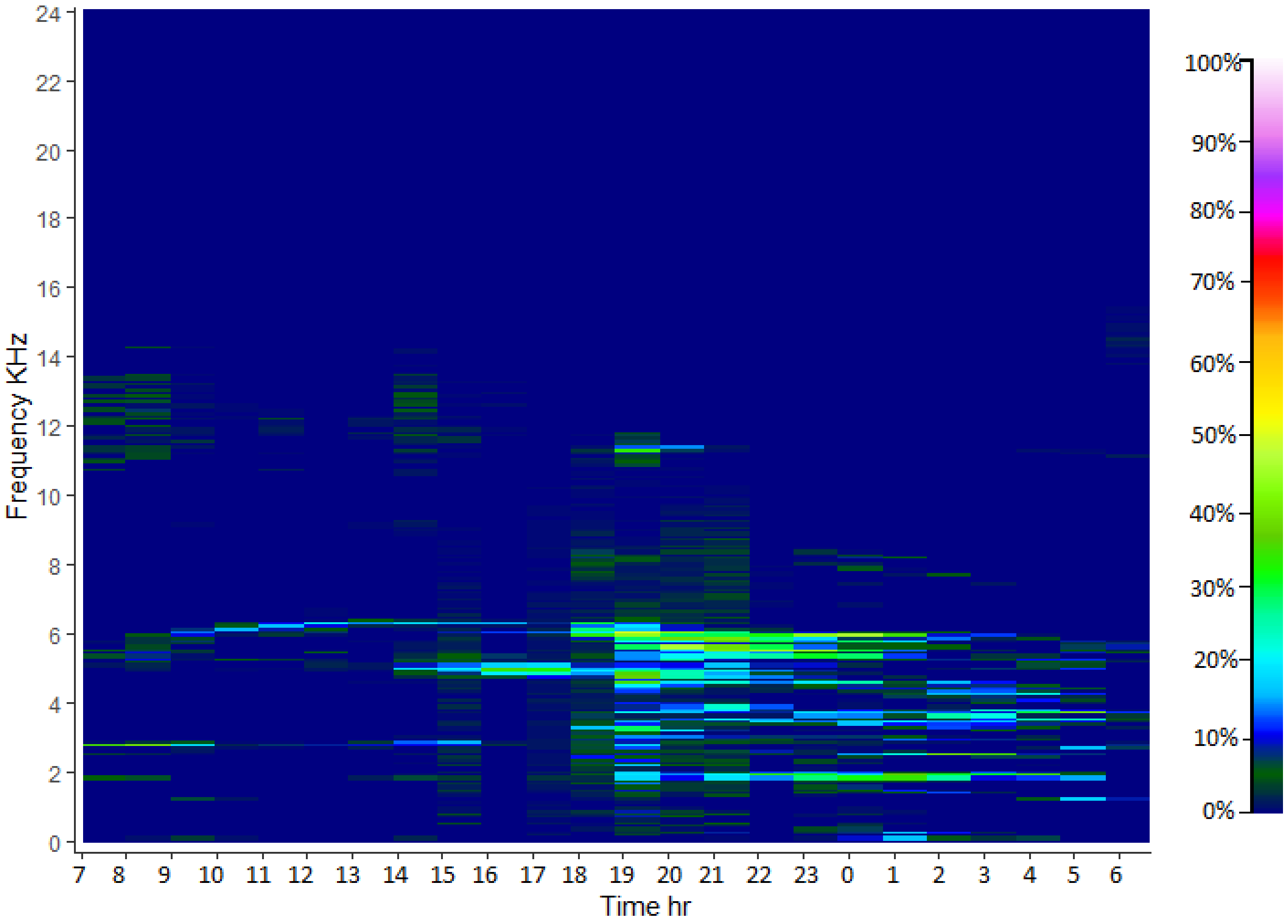

Figure 7: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S2-19m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 4: S2-19m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	12000-22050Hz 4000-6000Hz	300-4000Hz 4000-8000Hz	300-4000Hz 4000-12000Hz
TAXA	Avifauna, Invertebrates	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

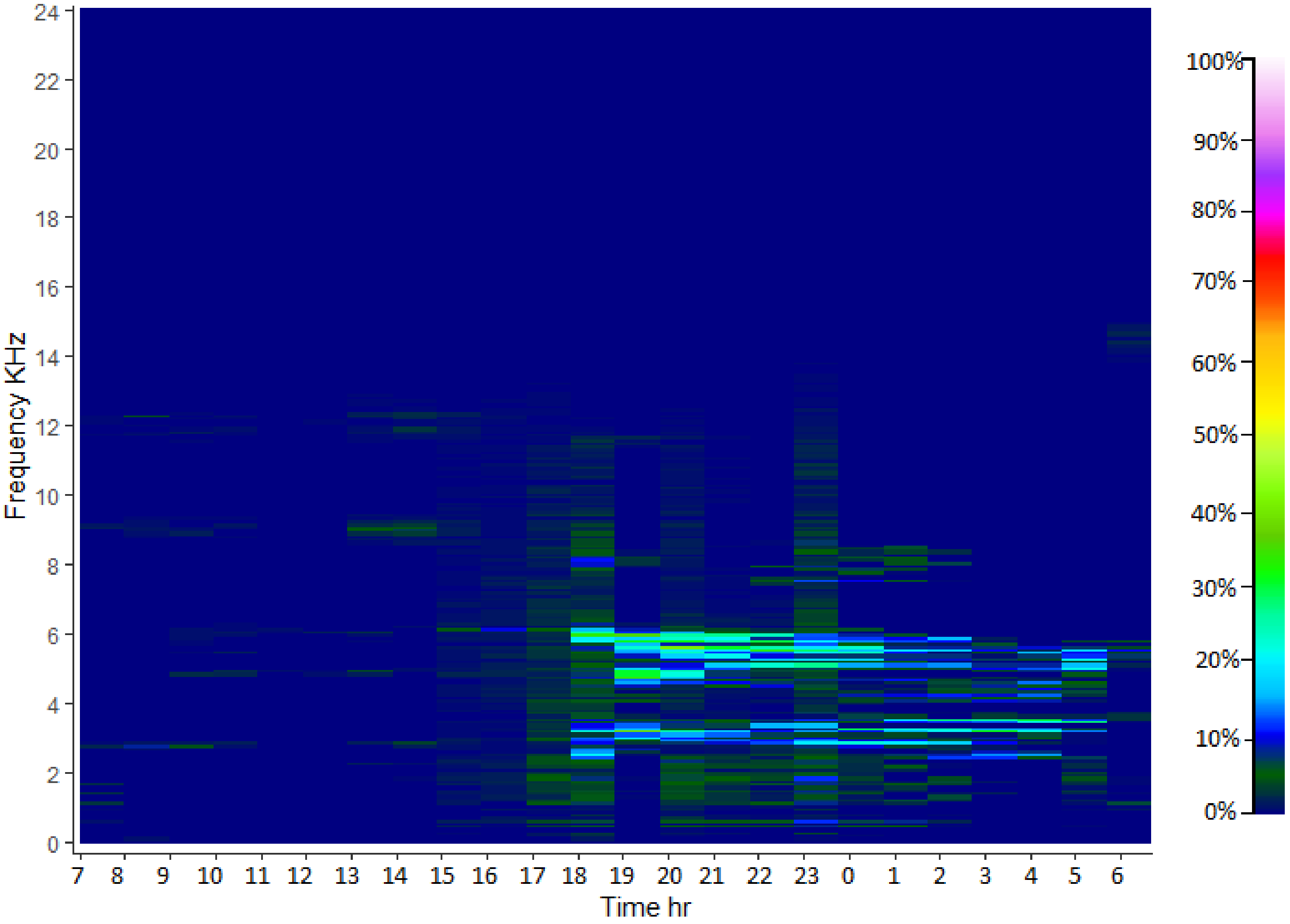

Figure 8: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S3-375m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 5: S3-375m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	3000-4000Hz	300-4000Hz 6000-8000Hz	300-4000Hz 4000-6000Hz 8000-10000Hz
RELATED TAXA	Avifauna	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

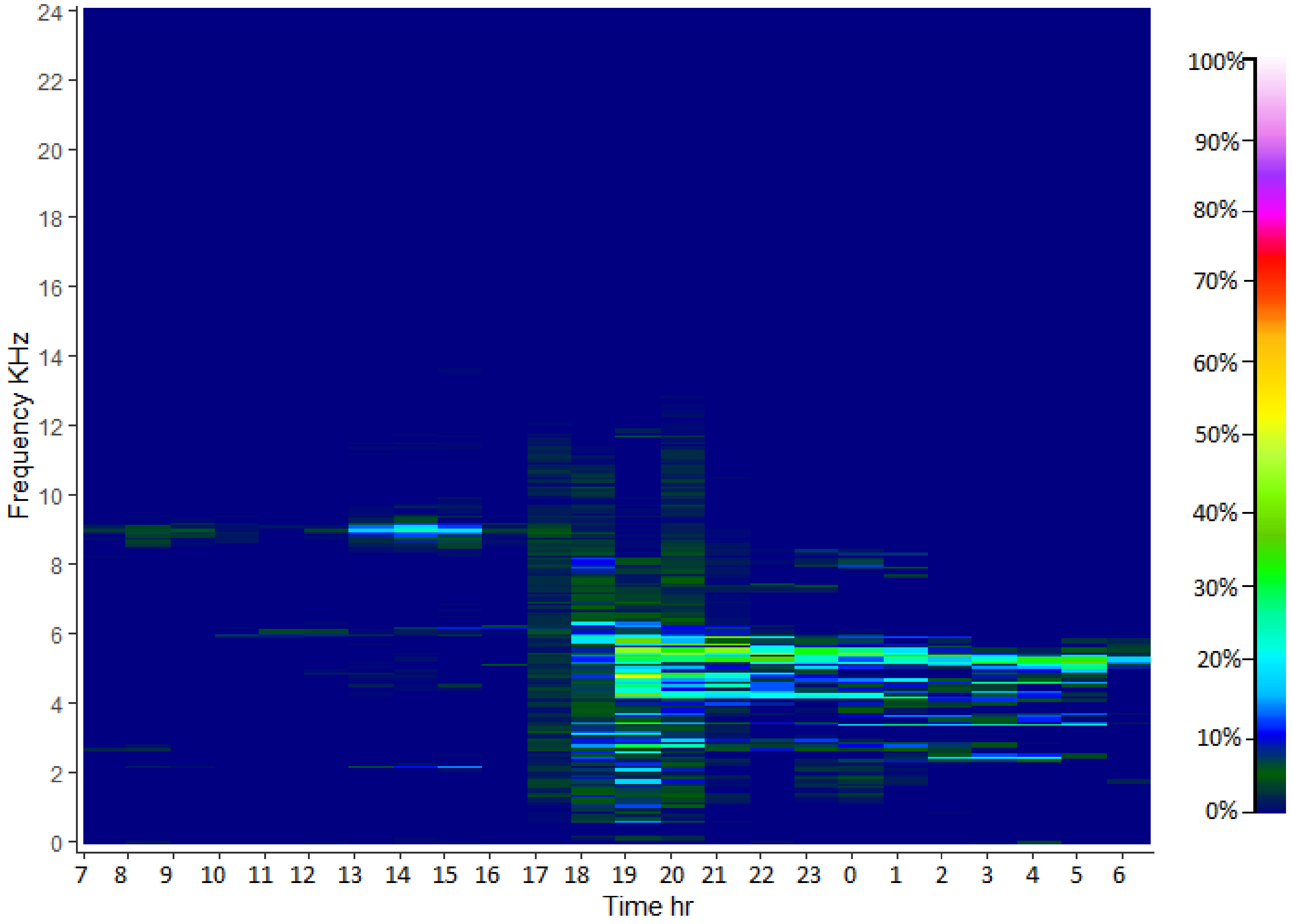

Figure 9: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S4-423m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 6: S4-423m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	None	2000-4000Hz 8000-10000Hz	300-4000Hz 4000-8000Hz
RELATED TAXA	None	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

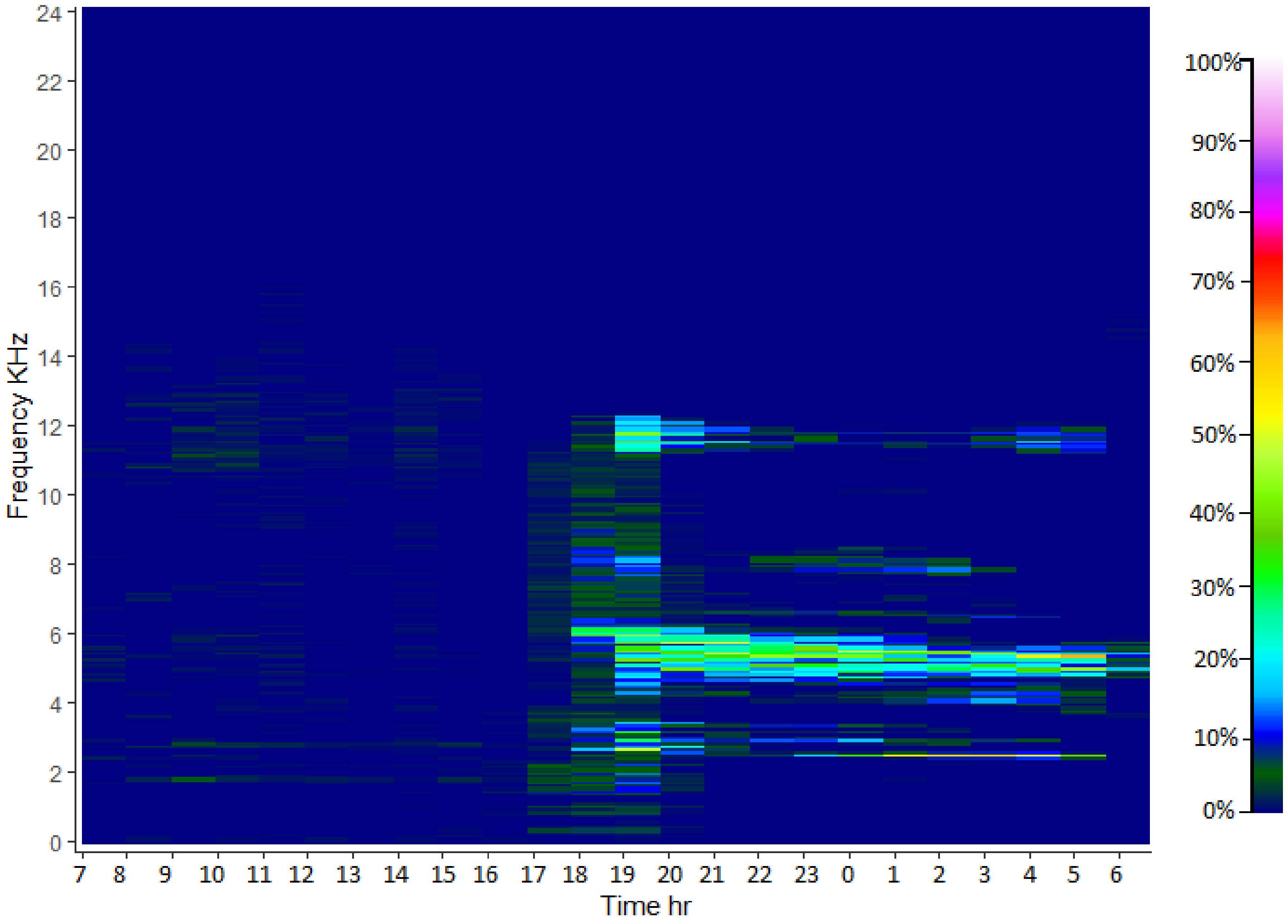

Figure 10: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S5-530m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 7: S5-530m commonly occurring frequency bins in (<5%) each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	None	300-4000Hz	300-4000Hz 4000-12000Hz
RELATED TAXA	None	Avifauna	Amphibians, Avifauna, Arboreal mammals, Invertebrates

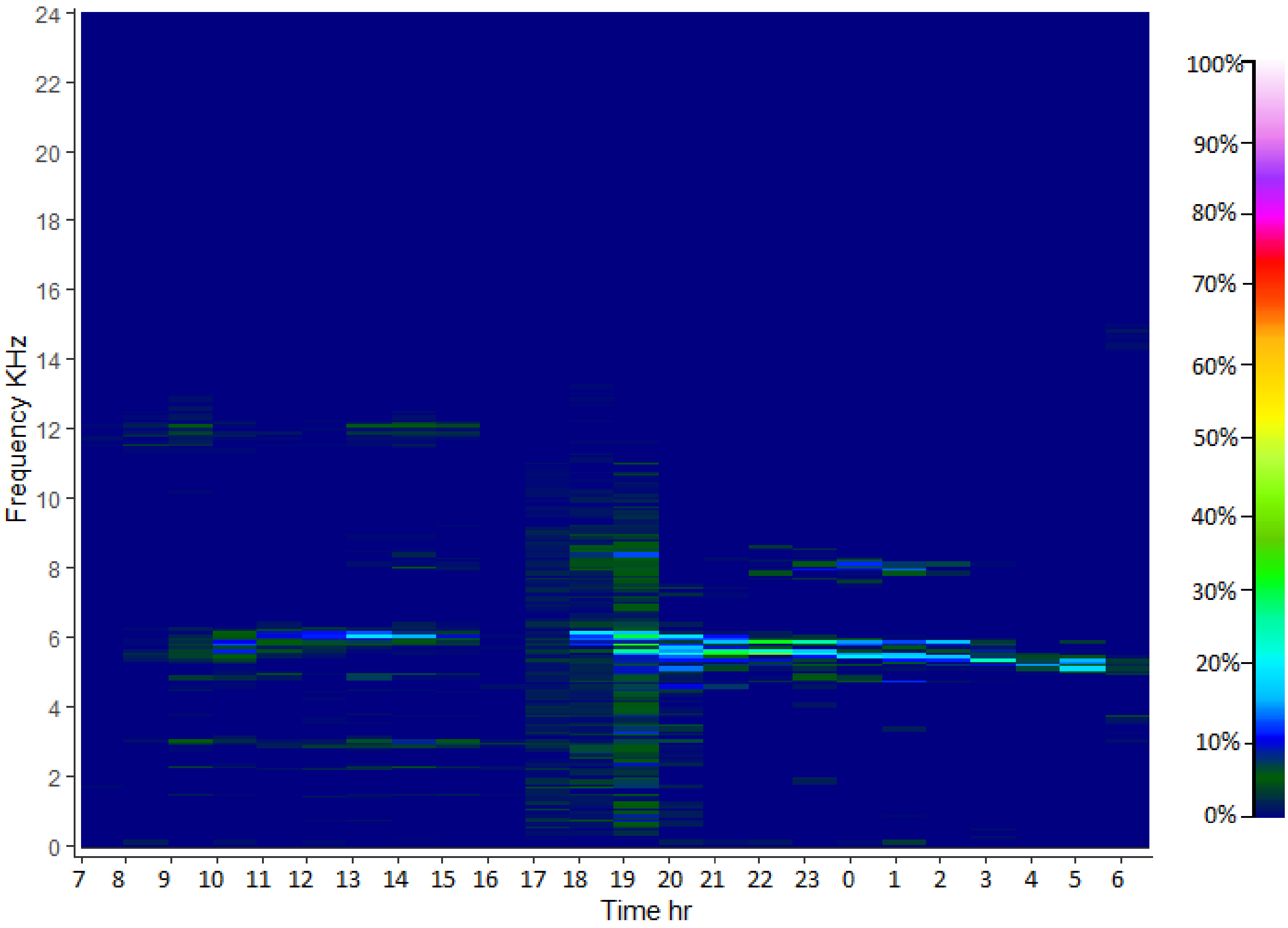

Figure 11:Soundscape displaying the acoustic activity across frequency bins of all recordings made in S6-942m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 8: S6-942m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	2000-4000Hz 4000-6000Hz	200-4000Hz 4000-6000Hz	300-4000Hz 4000-10000Hz
RELATED TAXA	Avifauna, Arboreal mammals, Invertebrates	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates,

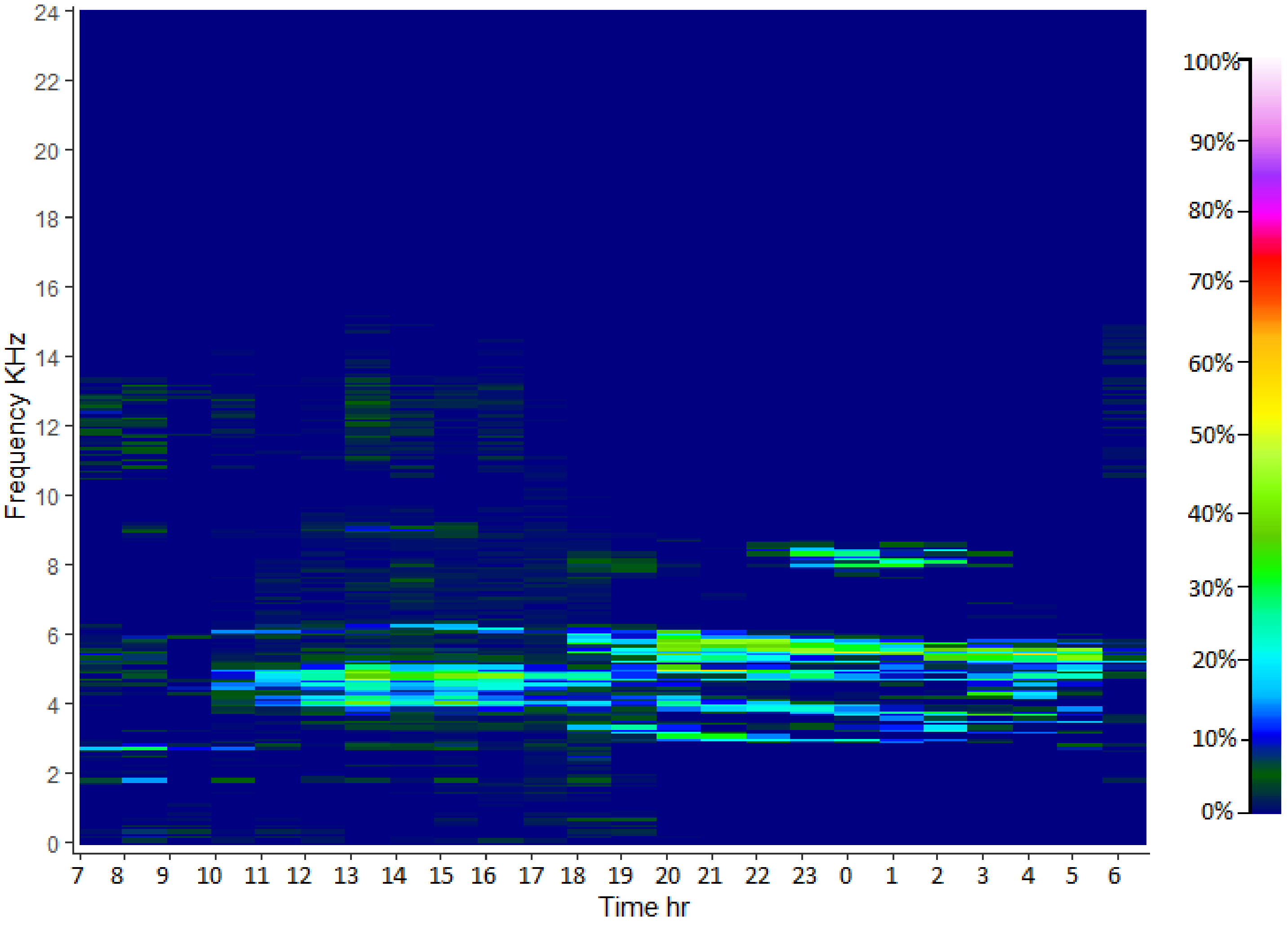

Figure 12: Soundscape displaying the acoustic activity across frequency bins of all recordings made in S7-1087m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 9: S8-1217m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	300-4000Hz 4000-6000Hz	300-4000Hz 4000-12000Hz	300-4000Hz 4000-6000Hz 8000-12000Hz
RELATED TAXA	Avifauna, Arboreal mammals, Invertebrates	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

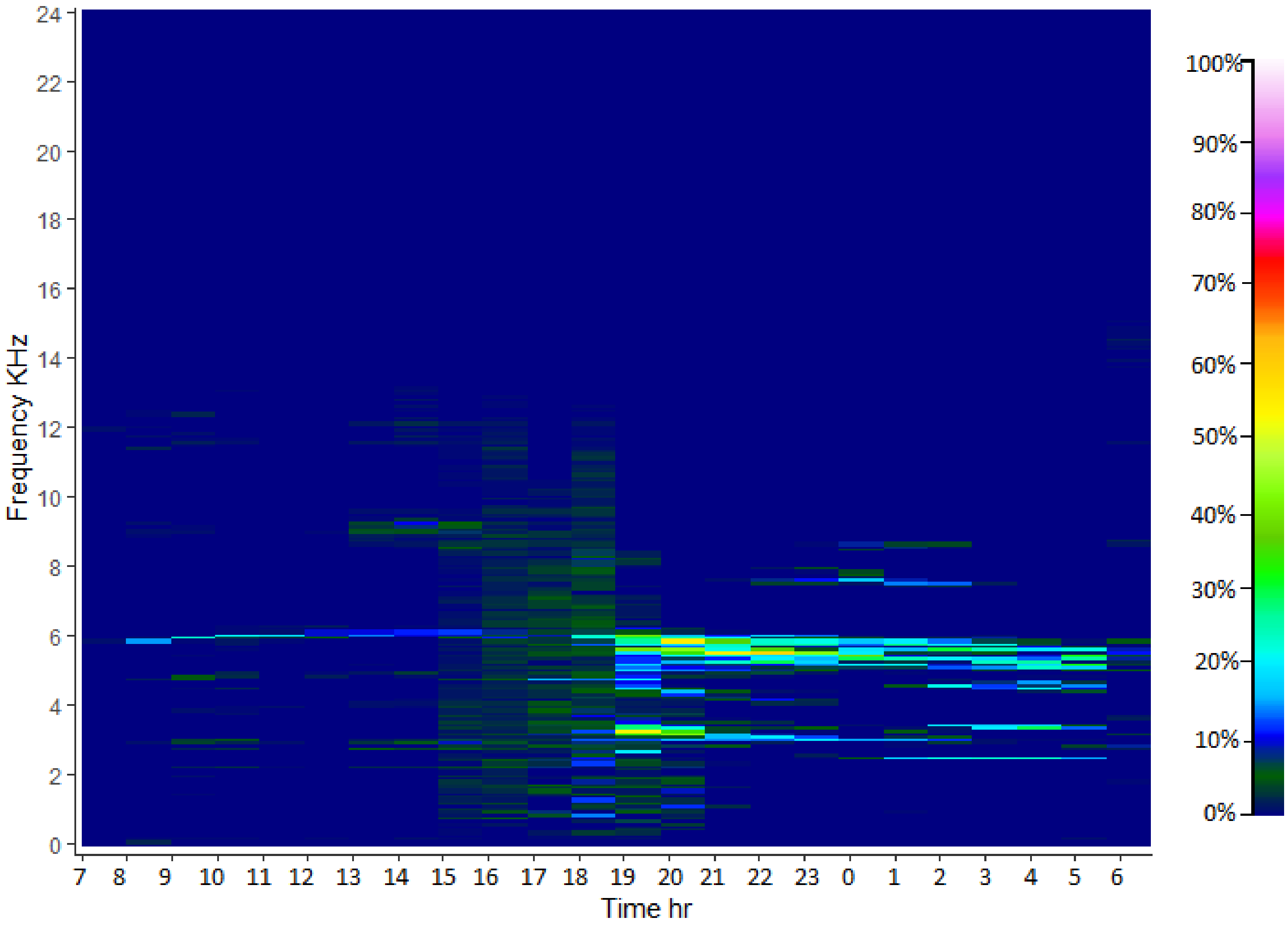

Figure 13:Soundscape displaying frequency the acoustic activity across bins of all recordings made in S8-1217m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 10: S8-1217m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	2000-4000Hz 6000-8000Hz	2000-4000Hz 6000-8000Hz	300-4000Hz 4000-12000Hz
RELATED TAXA	Avifauna, Invertebrates	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

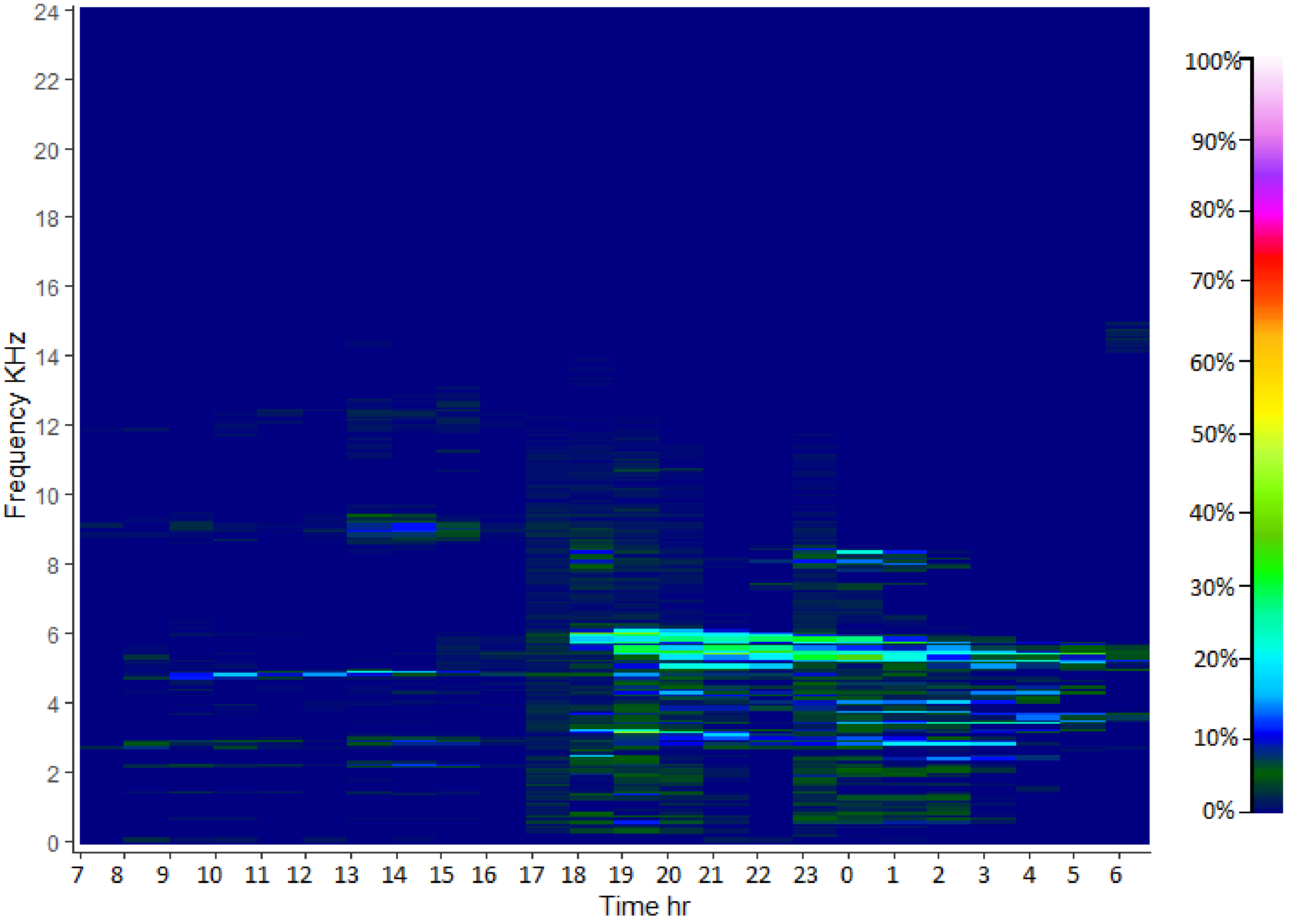

Figure 14:Soundscape displaying the acoustic activity across frequency ins of all recordings made in S9-1497m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

Table 11: S9-1497m commonly occurring frequency (<5%) bins in each time Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	3000-6000Hz	4000-6000Hz 6000-10000Hz	300-4000Hz 4000-12000Hz
RELATED TAXA	Avifauna, Invertebrates	Avifauna, Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

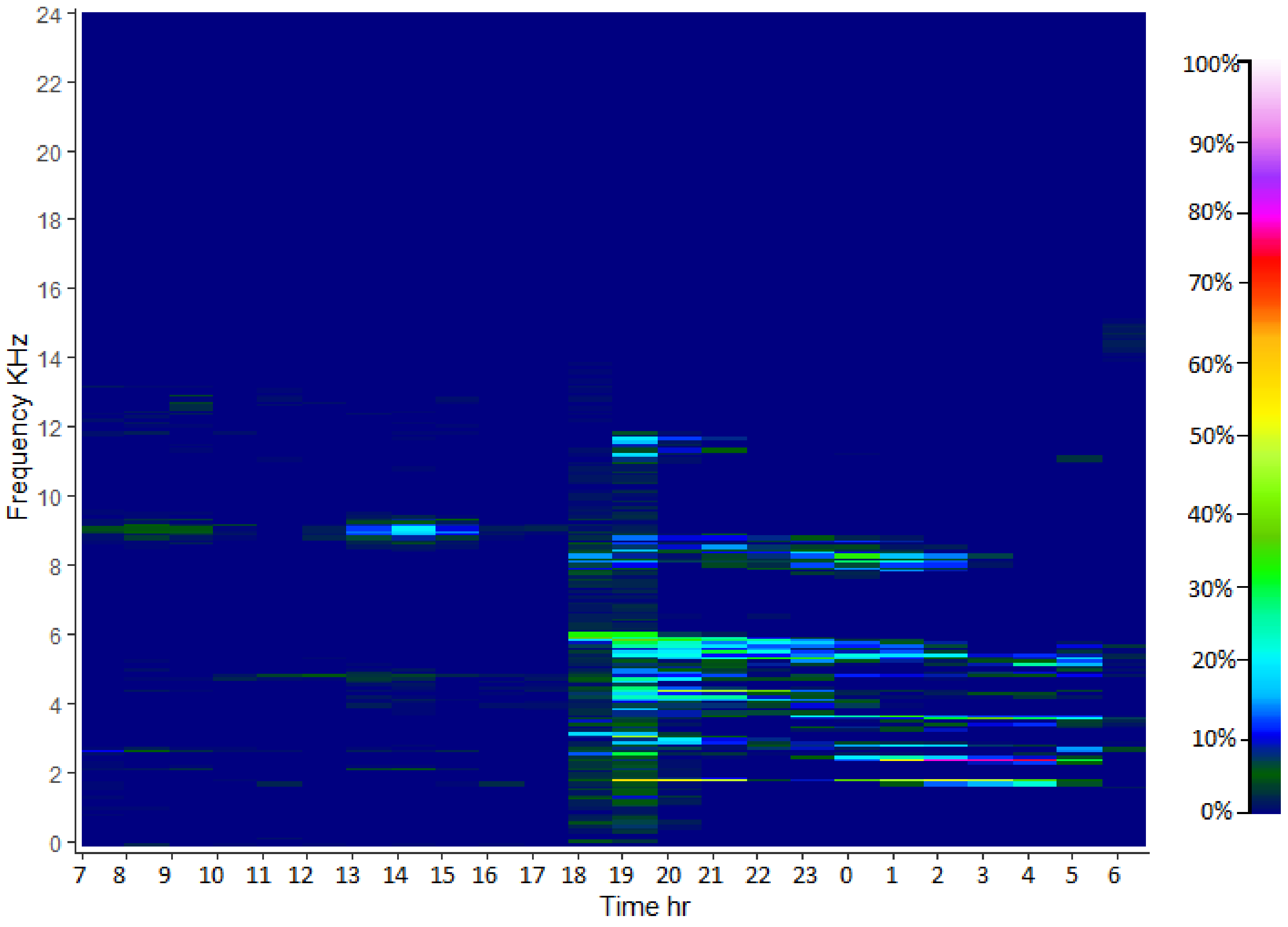

Table 12: S9-1497m commonly occurring frequency bins (<5%) in each time period Dawn (06:00-10:00) Day (10:00-18:00) and Dusk (18:00-21:00) and the taxa equated to the frequency and time bins according to Metcalf et al., 2021 frequency and time bin table.

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Figure 15: soundscape displaying the acoustic activity across frequency bins of all recordings made in S10-1572m plotted by hour. Colour scale representative of the percentage of recordings acoustic activity occurred in.

	DAWN	DAY	DUSK
COMMON FREQUENCY BINS	2000-4000Hz	8000-10000Hz	300-4000Hz 4000-12000Hz
RELATED TAXA	Avifauna	Invertebrates	Amphibians, Avifauna, Arboreal mammals, Invertebrates

Forest structure edge effect

There is no relationship between distance and number of trees with diameter >10cm at each 25x25m (Figure 16). A total of 110 trees were recorded across the 10 sample sites and of them S3-375m has the largest number of trees present 13, whereas S10-1572m has the lowest number of trees present 2. No significance was found between distance from the forests edge and the number of trees recorded at each site P = 0.425 (GLM, table 13).
No significant relationship was found between tree height and distance from the forests edge (GLM, Table 13, P = 0.335). However, there is an increase of the median tree height seen between S5-530m and S9-1479m before dropping at S10-1572m (Figure 17). S9-1497m has the largest range as well as the tallest tree (51.4m) and largest median. This is contrasted by S10-1572m which has the smallest range and interquartile range as well as the smallest median. S5-530m has the smallest tree height (5.6m). Both S4-423m and S5-530m have large interquartile ranges and are both positively skewed. There are two positively skewed outliers on both S6-924m and S7-1087m.

No relationship is displayed between distance from the forest edge and bole height (Figure 18). S9-1497m exhibits the largest bole height (33.5m), median of (13m) and range. S10-1552m exhibits a normal distribution, has the smallest median (3) and interquartile range. S5-530m has the smallest bole height and has anormal distribution. There are 2 negative outliers on S3-375 and one positive. No significance was found between distance from the forests edge and bole height (GLM, Table 14).

The DBH does not appear to correlate with distance from the forests edge (Figure 19). S2-19m has the largest interquartile range with a positive skew but S9-1497 has the largest DBH (73.21cm) and a positive skew. The highest median is exhibited by S4-423m, whereas S6-924m has the lowest median. S7-127m has the lowest inter quantile range. There are several high outliers and most of the sites have positive skewers. No significant difference was found between the distance from the forests edge and the mean DBH of trees at sample sites P = 0.134 (GLM, Table 13).

The HDR does not correlate to distance from the forests edge (Figure 20). S9-1497m has the highest HDR (95.8) as well as the highest median. S4-423m has the lowest HDR (17.7) however, S4-423m exhibits the largest range and interquartile ranges for HDR. The Smallest median is displayed in S10-1572m which also shows the smallest range. There is a negative skew in S2-19m which also has a low outlier and two higher outliers. No Correlation was found between mean HDR and distance from the forests edge in any model.

No significant correlation between distance to the forests edge and mean crown area of trees was found P = 0.31 (GLM, Table 13). However, a correlation between max crown area and distance from the forests edge can be seen (Figure 21). The highest two crown areas were recorded at S9-1497m (204.4m2) the lowest crown area was recorded at S1-0m (1.86m2). The crown areas from all sites were positively skewed except S8-1217m which exhibits a negative skew. There are many high outliers but only one low outlier of 8 within S8-1217m. The highest median can be seen within S2-19m, and the lowest median is within S7-1087m.

No significant relationship between distance from the forests edge and mean crown connectivity was found P = 0.463 (GLM, Table 13). S7-1087m exhibits the largest range in crown connectivity and S10-1572m displays the smallest (Figure 22). Both S1-0m and S7-1087m display the highest medians whilst S10-1572m displays the lowest. There are two outliers in S8-1217m one high and one low. S1-0m and S7-1087m are both negatively skewed whereas all other sites are positively skewed.

Table 13: GLMs of each sample sites mean tree height, crown area (CA), crown connectivity, dimeter at breast height (DBH) and the number with distance as the predictor.

	Theightmean			**no_trees**			**CA_mean**			**conn_mean**			**DBH_mean**		
Predictors	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*p*
(Intercept)	-0.00	0.32	1.000	0.00	0.32	1.000	-0.00	0.29	1.000	0.00	0.32	1.000	-0.00	0.30	1.000
dist	0.32	0.33	0.335	-0.27	0.34	0.425	0.49	0.31	0.112	-0.25	0.34	0.463	0.47	0.31	0.134
Observations	10			10			10			10			10		
R^2	0.104			0.074			0.240			0.063			0.219		

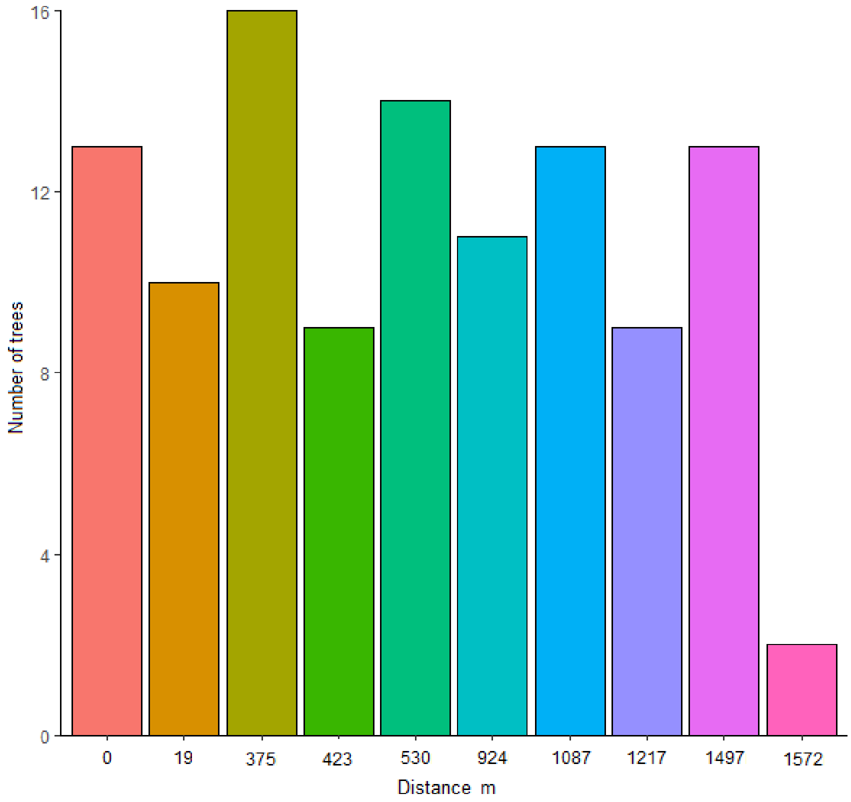

Figure 16: Bar plot of number of trees with a DBH over 20cm recorded in each 25m x 25m sample site.

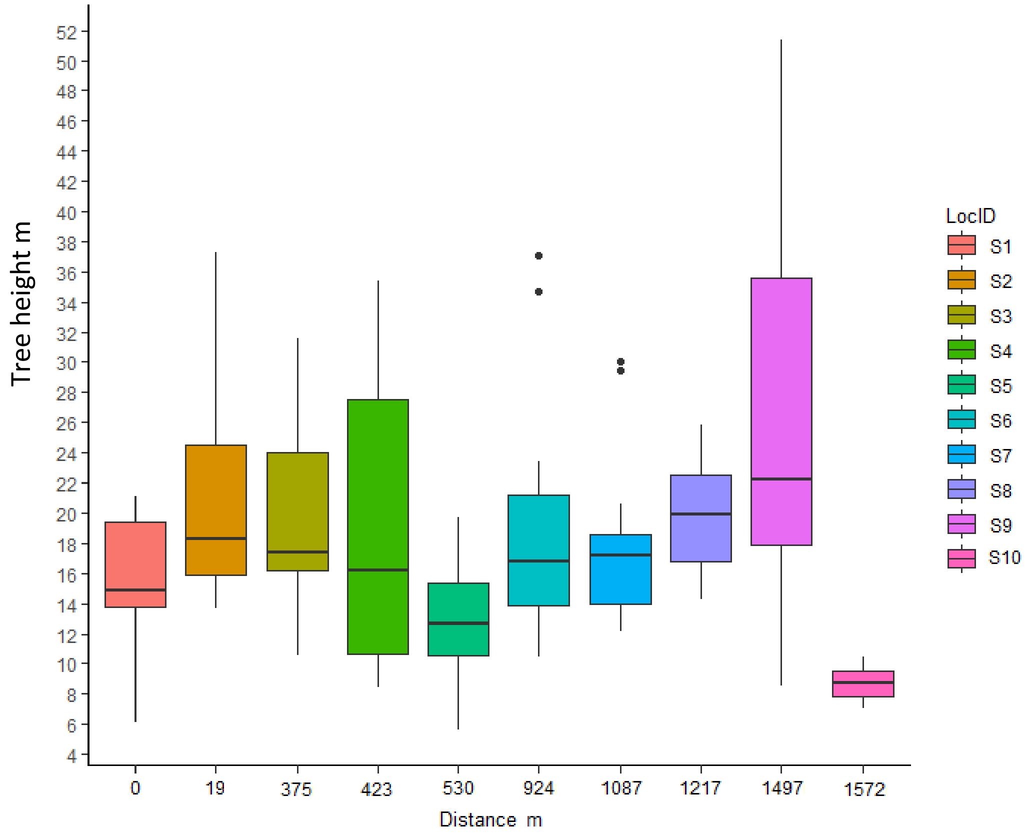

Figure 17: Boxplot of tree height at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

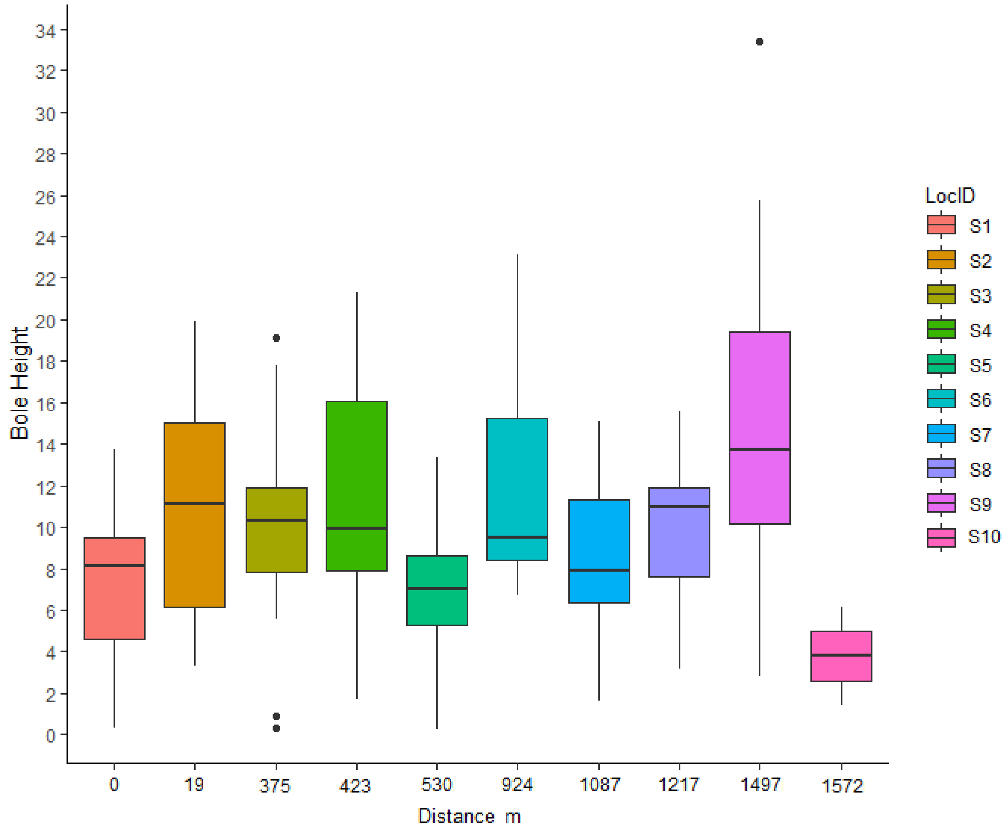

Figure 18:Boxplot displaying the bole height at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

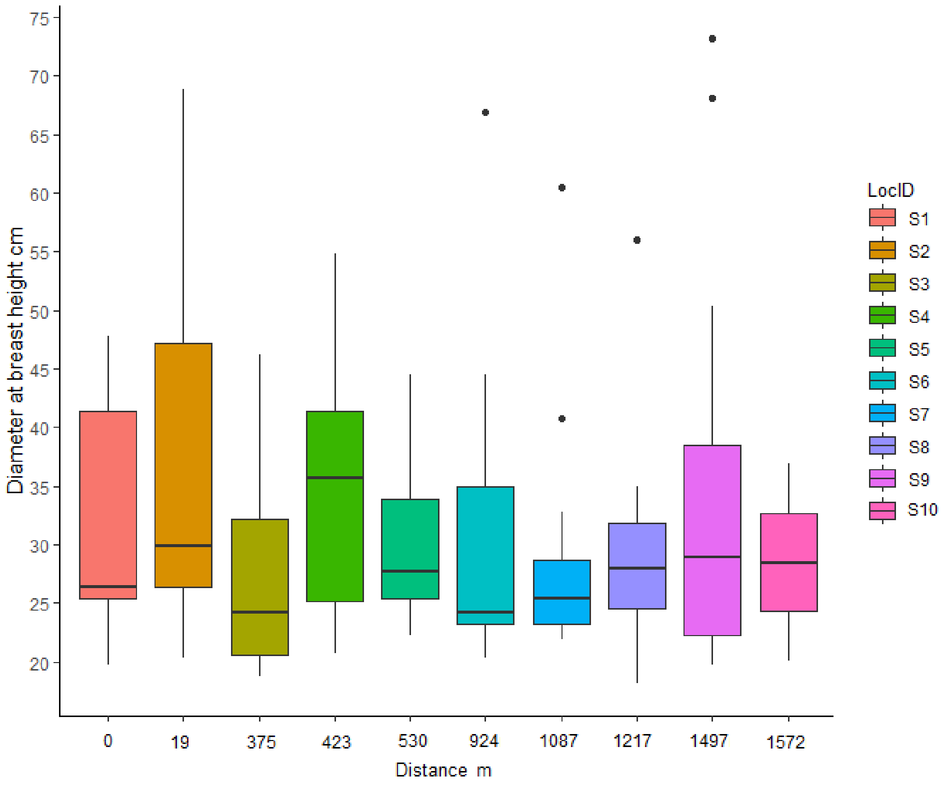

Figure 19: Boxplot displaying diameter at breast height (DBH) at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

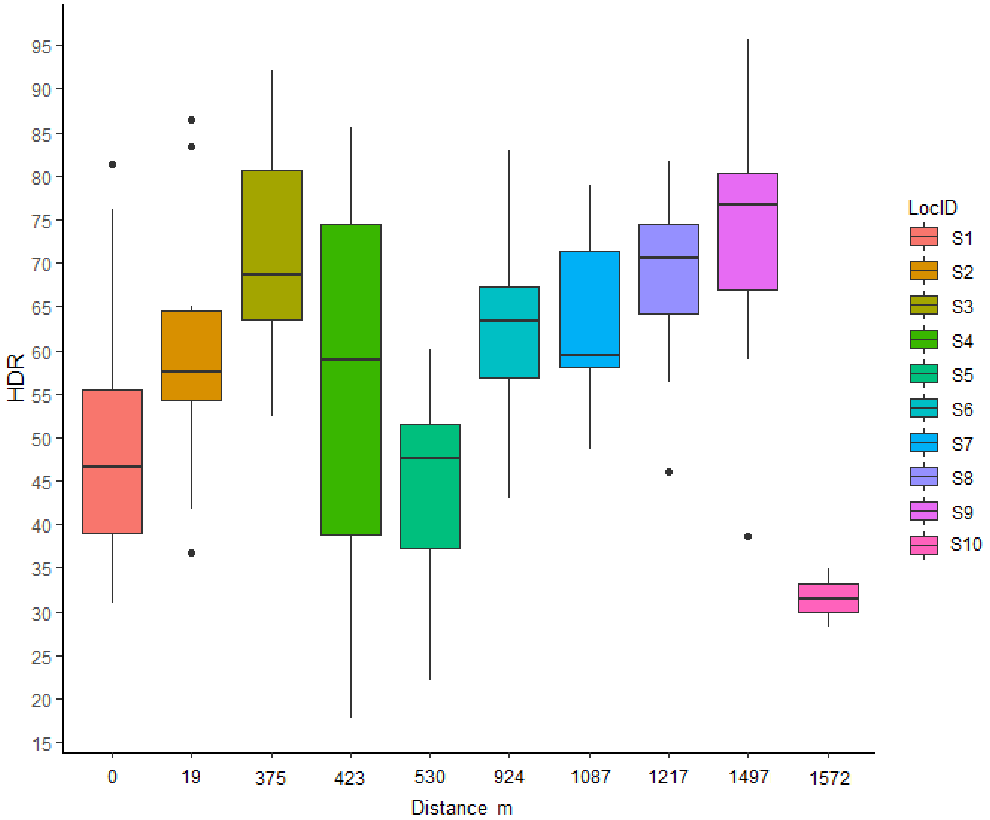

Figure 20:Boxplot displaying the height depth ratio HDR of the trees at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

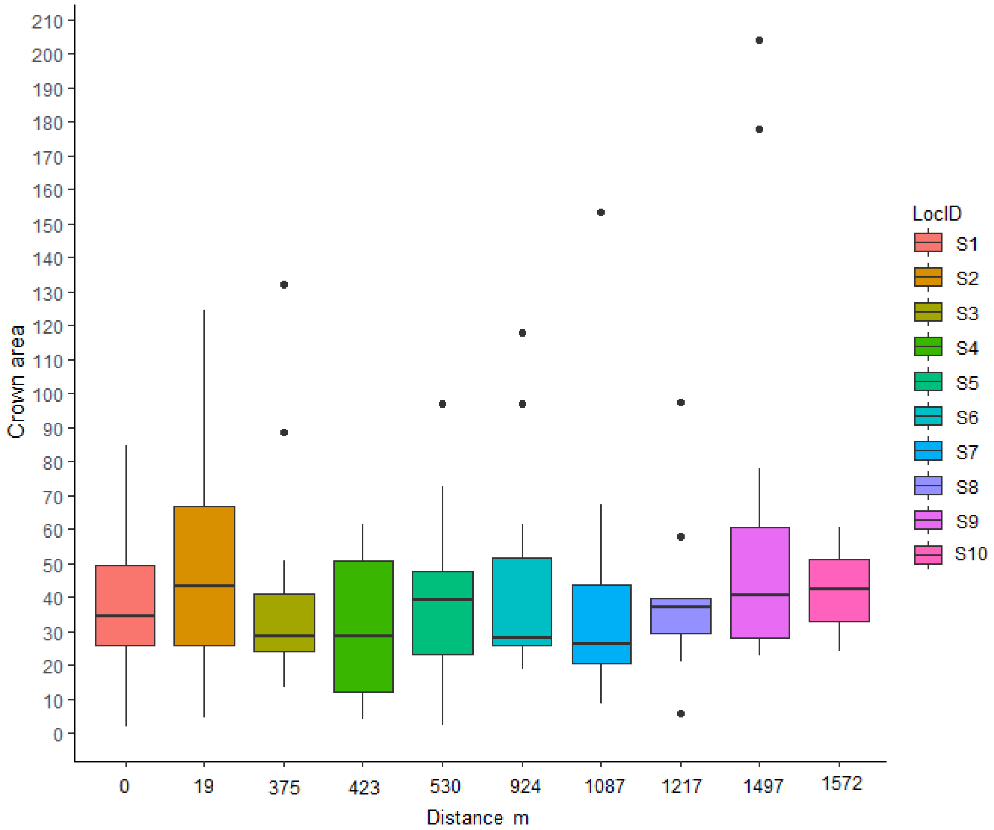

Figure 21: Boxplot displaying the crown area of the trees at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

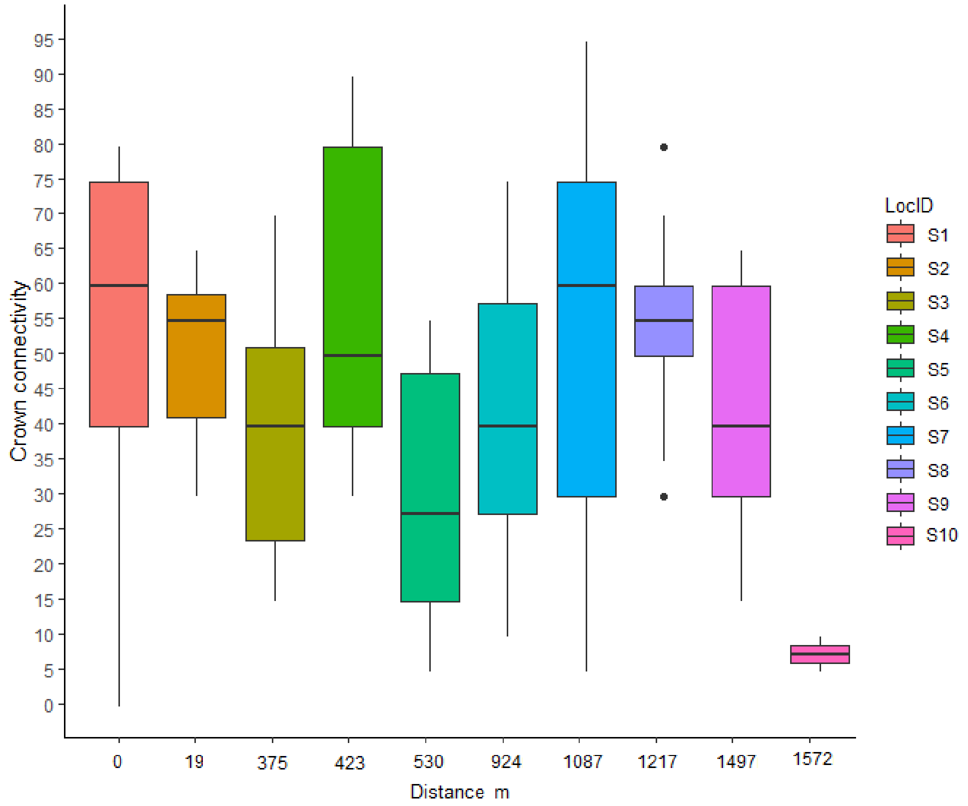

Figure 22:Boxplot displaying the crown connectivity % of the trees at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

Microclimate edge effect

The hourly light intensity peaked and dropped at the same time at each site (Figure 23), the LUX rose at 7:00, peaked between 11:00 and 13:00 and dropped back to 0 at 19:00. There was a decrease of light intensity seen at all sites after S1-0m which displayed the highest mean LUX at 12:00. All other sites exhibit a LUX less than 5000 for the majority of daylight hours 6:00 to 18:00. Within daylight peak hours 11:00 to 13:00, S3-375m displayed the lowest light intensity of 247.6LUX across all days along with S4-423m also displaying a low LUX of 355.2. The daily light intensity (Figure 24) at S1-0m had the largest interquartile range in light intensity and S3-375 had the smallest. There was not much variation between the medians of the light intensity with most sites below 3000 LUX apart from S1-0m (median=11567.4) and the lowest value at S3-375 (median= 612.02). There is a positive skew across all distances. The max light intensity decreased significantly with distance from the forest edge (Table 14, GLMM N=108 across 10 sites and 26 days of the year, P<0.001, estimate of -0.42 S.E. 0.09).

The hourly temperature increased from 07:00 peaking between 12:00 and 14:00 and decreasing at 18:00 (Figure 25). S1-0m had the highest recorded temperature of 37.5°C at 13:00 and displayed temperatures above 24°C throughout the daylight hours, while S3-375m exhibited the lowest temperature recorded of

24.4°C at 12:00. Excluding the top two sites, the other sites have a similar temperature across the daylight hours of 6:00 to 18:00 ranging from 22°C and 26°C with peaks up to 30 °C at mid-day dropping back to 22°C to 26°C in the evening. The plots for most of the sites exhibit an interquartile range between 24°C and 27°C for the daily temperature (Figure 26), whilst S1-0m displays the greatest interquartile range (24°C- 30°C). A decrease in temperature can be seen at all distances past 0m with a slight increase at S5-530m.

There is not much variation between the median temperature at each site. S1-0m displays the highest temperature median and is also positively skewed. The lowest temperature median is S3-375m which has a normal distribution. There are some outliers on S5-530m with temperatures over 34°C. The max temperature and temperature range both decreased significantly with distance from the forests edge (Table 14, GLMM N=108 across 10 sites and 26days, P <0.001, estimate -0.41 S.E. 0.0.9).

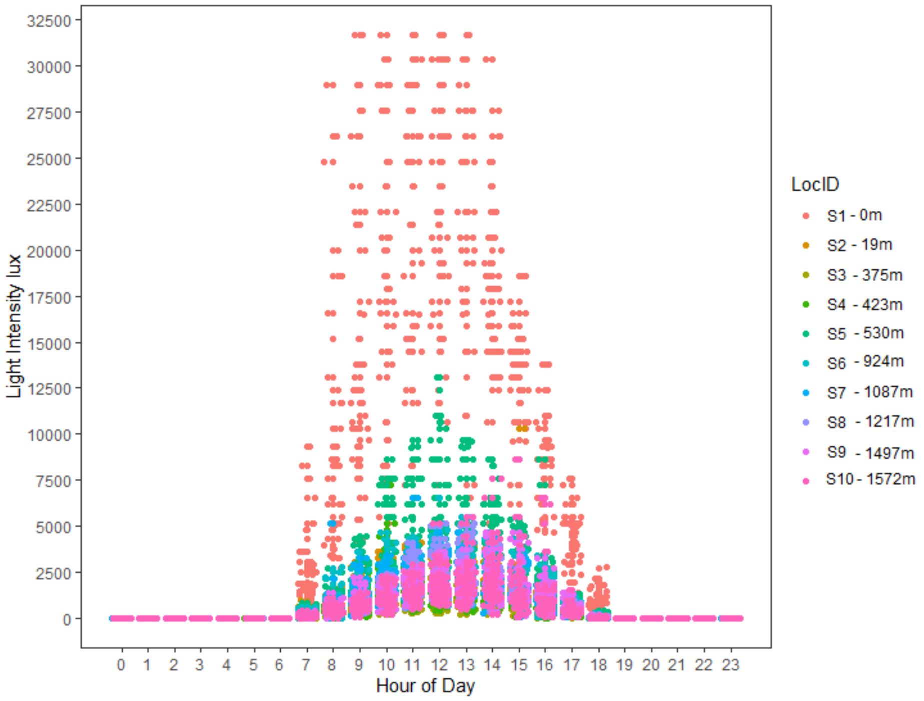

Figure 23: Light intensity (LUX) daily variation by hour of each sample site

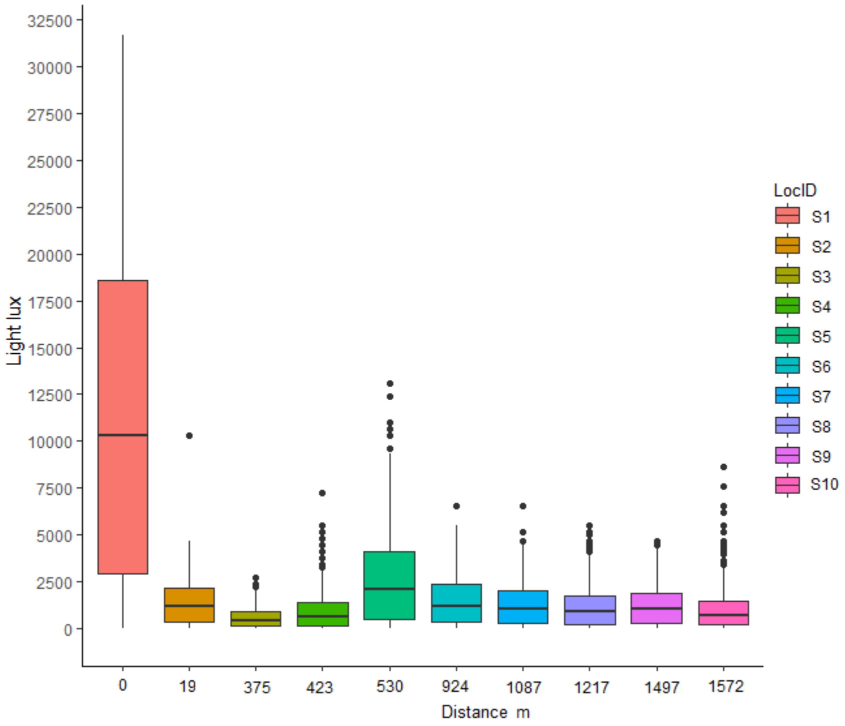

Figure 24: Boxplot of light intensity (LUX) in daylight hours 6:00 to 18:00 for each site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

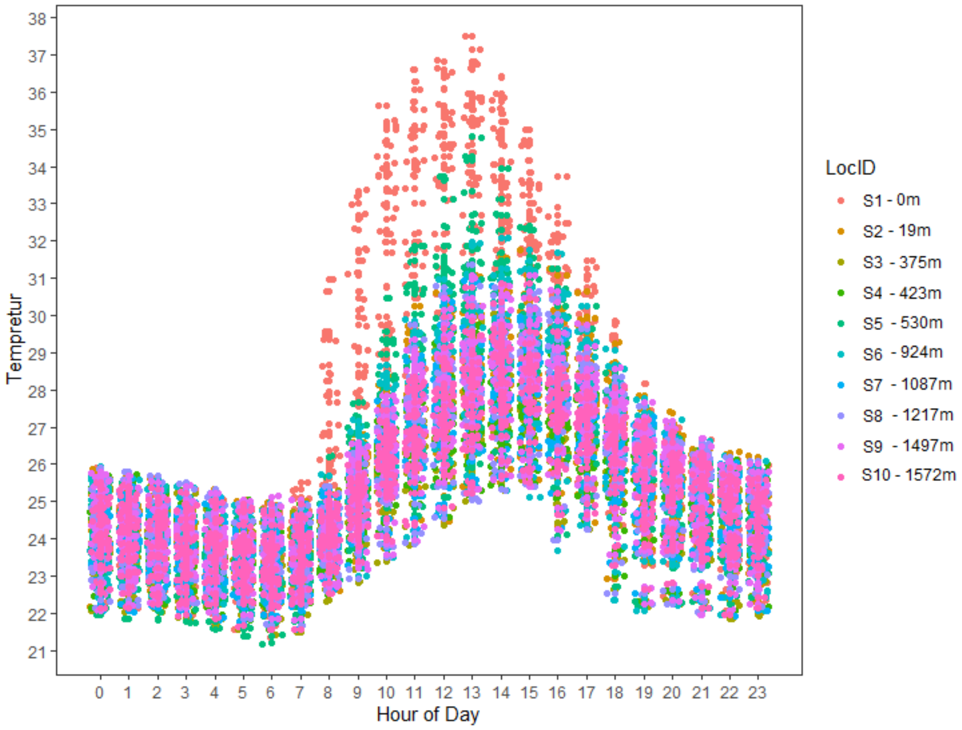

Figure 25: Temperature °C daily variation by hour of each sample site.

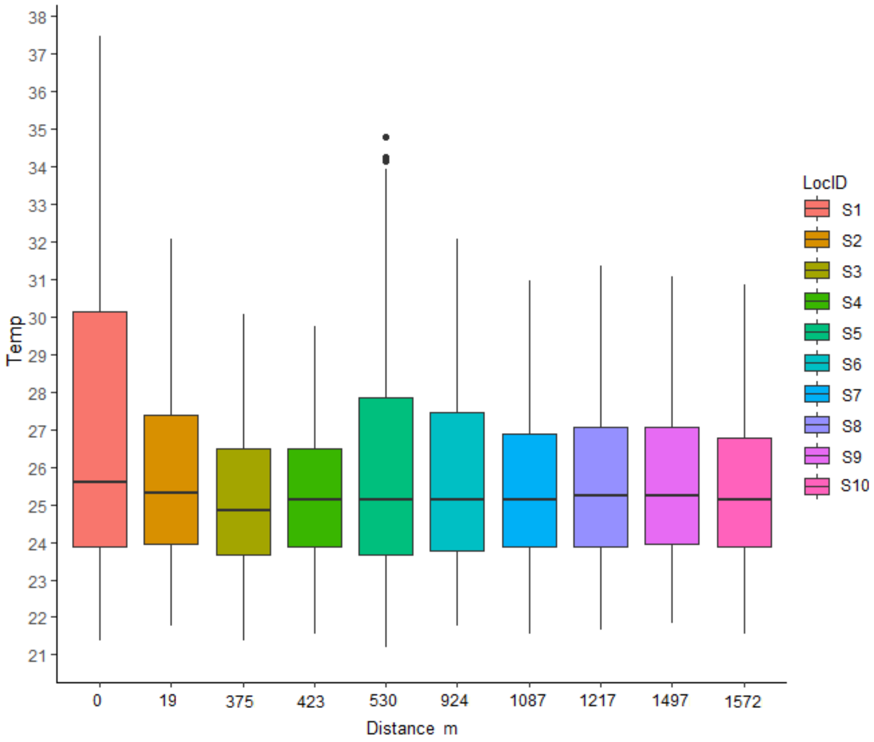

Figure 26: Box plot of temperature of each site ordered by distance from edge across all hours, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

Table 14: GLMM of climate variables temperature max temperature range and light intensity with distance as the predictor and the day of the year as a random variable.

	Temperature max			Temperature range			Light intensity max		
Predictors	*Estimates*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	-0.03	0.10	0.751	-0.04	0.10	0.689	0.00	0.09	1.000
Distance	-0.41	0.09	**<0.001**	-0.41	0.09	**<0.001**	-0.42	0.09	**<0.001**
Random Effects									
σ^2	0.79			0.78			0.83		
τ_{00}	0.05 DOY			0.06 DOY			0.00 DOY		
ICC	0.06			0.07					
N	26 DOY			26 DOY			26 DOY		
Observations	108			108			108		
Marginal R^2 / Conditional R^2	0.170 / 0.216			0.164 / 0.227			0.176 / NA		

Anthropogenic disturbance

Machine hum detections decreased as the distance from the forests edge increased (Figure 27). S1-0m exhibits the highest median and the highest interquartile range whereas S6-924 has the lowest median and S9-1497 displays the smallest interquartile range. The largest interquartile range is shown in S3-375m which has a positive skew. The number of machine hum detections decreased significantly with distance from the forests edge, when correcting for the day of the year and number of observation days per site (GLMM, Table 15) GLMM N=108 across the 10 sites and 26 days of the year, P<0.001, estimate of -0.55 and standard error of 0.11. No significant relationship was found between the machine hums and the ACI or presence of either of the gibbon species (Table 16).

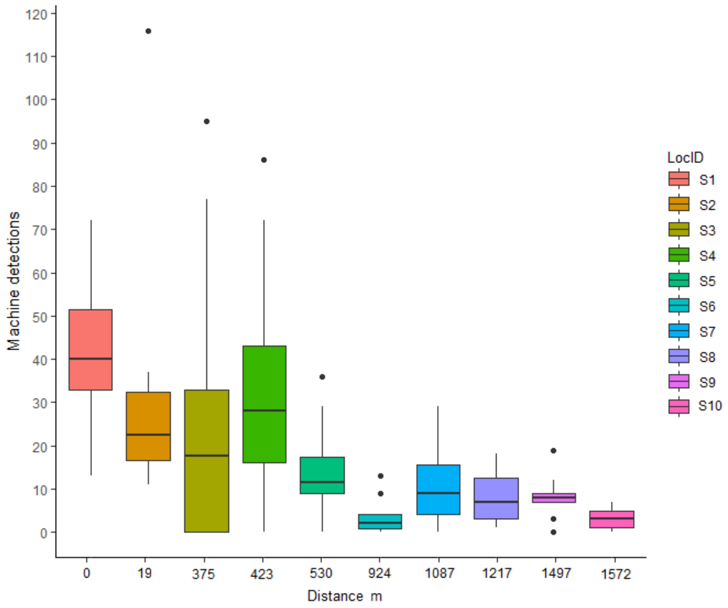

Figure 27:Boxplot displaying the number of machine hums detected at each sample site, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

Table 15: GLMM of machine Hum detections with distance as the predictor and the day of the year and number of days recordings made as random variables.

	Machine hum detections		
Predictors	*Estimates*	*std. Error*	*P*
(Intercept)	-0.00	0.13	0.970
Distance	-0.55	0.11	**<0.001**
Random Effects			
σ^2	0.66		
$\tau_{00\ \mathrm{DOY}}$	0.04		
$\tau_{00\ \mathrm{NoDays}}$	0.07		
ICC	0.14		
N_{DOY}	26		
N_{NoDays}	8		
Observations	108		
Marginal R^2 / Conditional R^2	0.283 / 0.383		

Table 16: Binomial GLMM primate presence and daily ACI mean with machine hum as the predictor and day of the year and number of days recordings were made as random variables.

	Lar Presence			Sia Prescence			ACI Daily Mean		
Predictors	*Odds Ratios*	*std. Error*	*P*	*Odds Ratios*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	0.40	0.12	**0.002**	0.20	0.06	**<0.001**	0.14	0.19	0.449
Mhum	0.98	0.22	0.916	0.99	0.26	0.980	0.08	0.08	0.325
Random Effects									
σ^2	3.29			3.29			0.62		
τ_{00}	0.43_{DOY}			0.13_{DOY}			0.69_{DOY}		
ICC	0.12			0.04			0.52		
N	26_{DOY}			26_{DOY}			26_{DOY}		
Observations	108			108			108		
Marginal R^2 / Conditional R^2	0.000 / 0.115			0.000 / 0.038			0.005 / 0.527		

Diversity edge effect

Acoustic complexity indices for each location dropped in the day light hours from 6:00 until 10:00, with a small rise until 12:00 after which it drops before beginning to rise again at 15:00 (Figure 28). S1-0m starts and finishes with the lowest ACI at 00:00 and 23:00, S9-1497m starts and finishes with the highest ACI. S5-530m has the highest ACI at 12 and S6-924m has the lowest. There are two outliers to the trend, one at s3-375m at 4:00 and the other at s9-1497m at 23:00. S6-924m has a lower median than the other sites but still follows the same pattern.

Although there is some variability in ACI (Figure 29), there is a positive relationship between distance to the edge and median ACI when correcting for day of the year and number of observation days per site (GLMM, Table 17) P=<0.001. The variation in ACI (Figure 29) is greatest at S9-1497m and smallest at S1-0m, Whilst the median is greatest at S2-19m and lowest at S1-0m and S6-924m. There are 3 outliers within S1-0m of ACI above 156.

As the ACI was high at dawn and dusk, the relationship between distance to edge and dawn and dusk ACI (Figures 30 and 31) was also investigated. At dawn ACI (Figure 30) S3-350m exhibits the highest ACI of as well as the highest medium and range S6-924m has the lowest ACI of as well as the lowest medium and range. Most of the sites show a normal distribution in ACI apart from S1-0m and S9-1497m which are positively skewed. A negative relationship was found between maximum dawn ACI and distance (Table 18) when correcting for day of the year, but no significance was found with median dawn ACI (Table 19).

At dusk, the ACI differs from the dawn with S9-1497m exhibiting the highest ACI and range. S3-375m displays the highest median and S8-1217m shows the lowest median as well as ACI (Figure 31). Seven of the sites have high outliers and all sites display a positive skew apart from S1-0m which is negatively skewed. No significance between max dusk ACI (Table 18) or median dusk ACI (Table 19) and distance was found when correcting for day of the year.

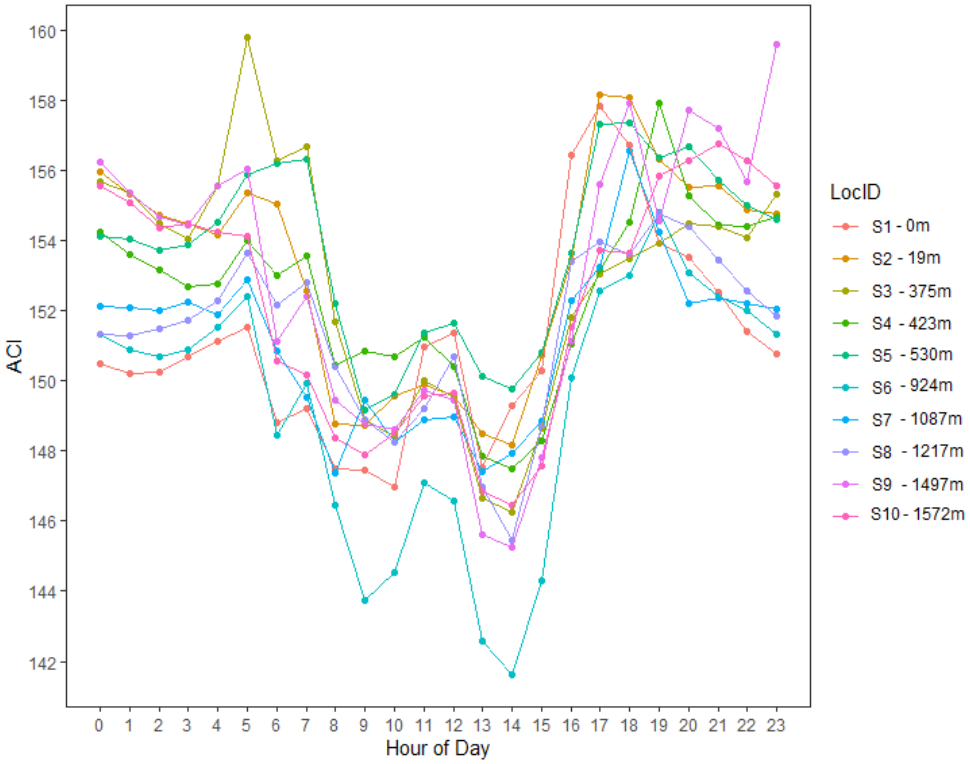

Figure 28: Acoustic complexity index (ACI) plotted by hour at each site.

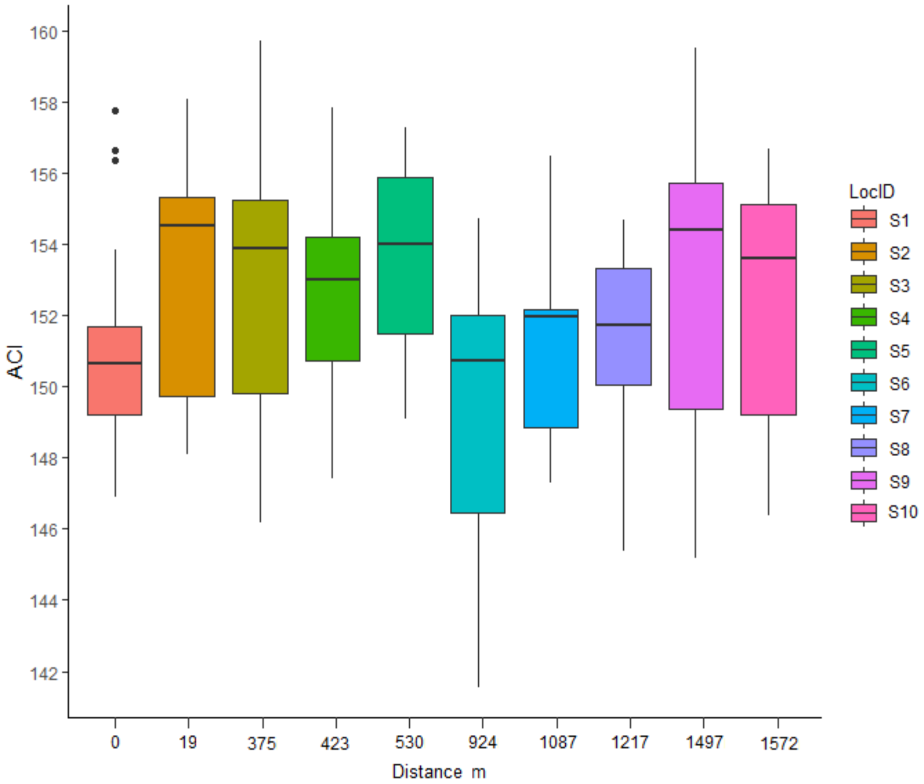

Figure 29: Boxplot of Acoustic complexity index (ACI) for each site across the 24hour period, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

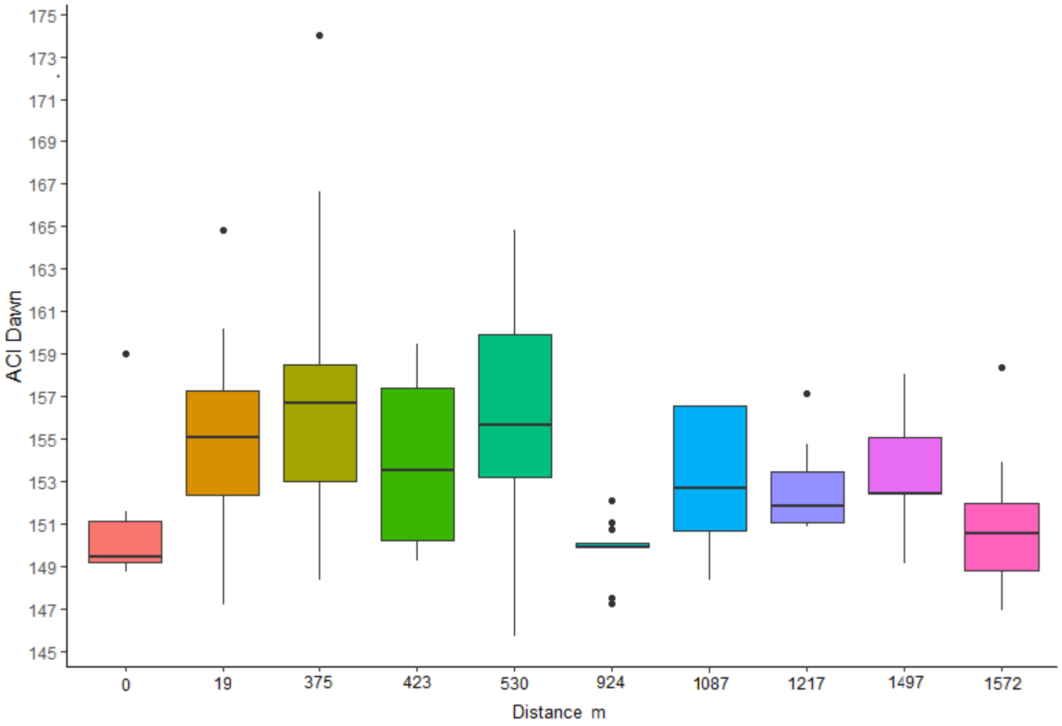

Figure 30: Boxplot of the ACI of each site at dawn (06:00 - 09:00), boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

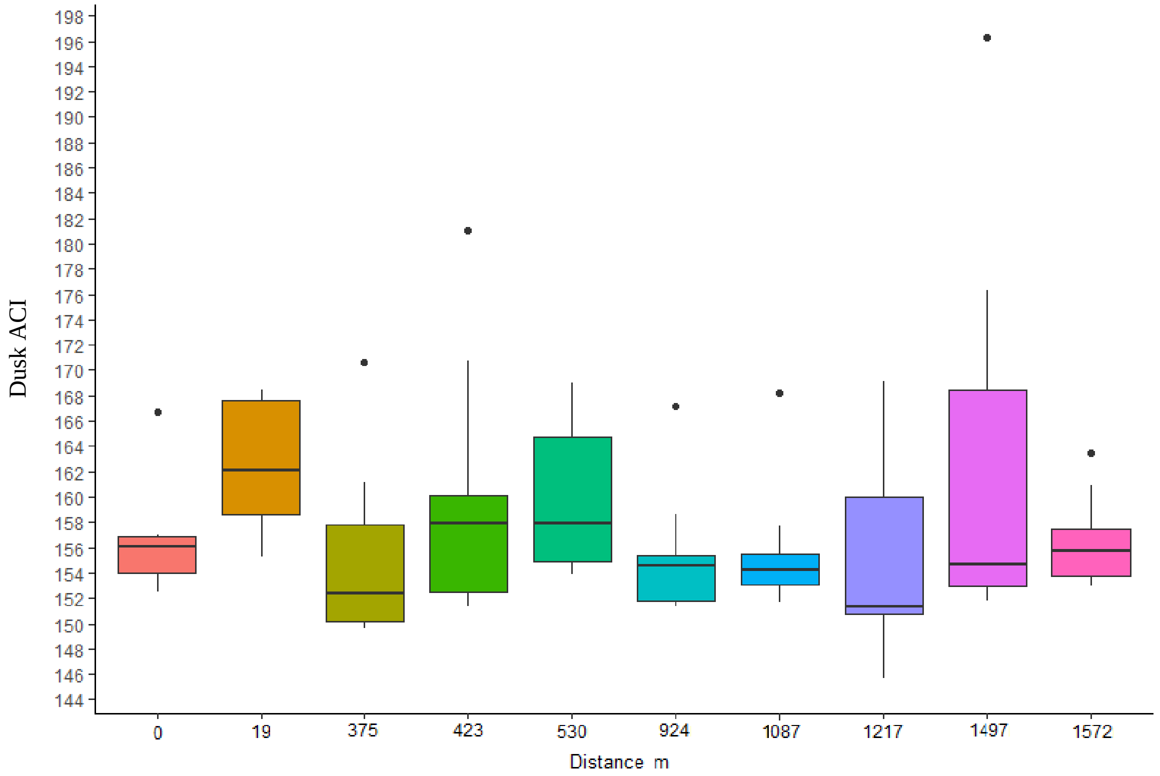

Figure 31: Boxplot ACI of each site at dusk (18:00 - 21:00), boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

Table 17: GLMM for Daily median ACI with distance as predictor and day of the as a random variable.

	ACI Daily Median		
Predictors	*Estimates*	*std. Error*	*P*
(Intercept)	0.04	0.41	0.856
Distance	0.81	0.11	**<0.001**
Random effects			
σ^2	0.3		
τ_{00}	0.79 $_{DOY}$		
	1.35 $_{NoDays}$		
ICC			
N	26 $_{DOY}$		
Observations	108		
Marginal R^2 / Conditional R^2	0.678 / NA		

Table 18: GLMM for max dawn and dusk ACI with distance as the predictor and day of the year as a random variable.

	ACI Dawn Max			ACI Dusk Max		
Predictors	*Estimates*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	0.05	0.13	0.728	-0.00	0.10	1.000
Distance	-0.25	0.08	**0.003**	-0.02	0.10	0.811
Random Effects						
σ^2	0.69					
τ_{00}	0.24 $_{DOY}$					
ICC	0.26					
N	26 $_{DOY}$					
Observations	108			108		
Marginal R^2 / Conditional R^2	0.063 / 0.303			0.001		

Table 19: GLMM of median dawn and dusk ACI with distance and the predictor and day of the year as a random variable.

	Dawn ACI Median			Dusk ACI Median		
Predictors	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*P*
(Intercept)	0.00	0.13	0.973	-0.02	0.14	0.859
Distance	-0.09	0.09	0.292	0.06	0.08	0.498
Random Effects						
σ^2	0.78			0.69		
τ_{00}	0.23 $_{DOY}$			0.28 $_{DOY}$		
ICC	0.22			0.29		
N	26 $_{DOY}$			26 $_{DOY}$		
Observations	108			108		
Marginal R^2 / Conditional R^2	0.009 / 0.231			0.003 / 0.290		

Gibbon vocalisations in relation to distance to edge

The total number of primate detections (Figure 32) exhibits a higher median across all sites for *Hylobates lar* detections (median=32) than *Symphalangus syndactylus* detections (median=16). The upper quartile in *H. lar* detections is also greater than that of *S. syndactylus* but *S. syndactylus* exhibits a larger range. *S. syndactylus* results are skewed positively whereas *H. lar* exhibits a normal distribution. Both species had days with over 50 detections observed but *S. syndactylus* was the only species to have sites with 0 detections observed.

No relationship can be seen between the *Hylobates lar* detection ratio and the distance from the forests edge (Figure 33, Table 20). *Hylobates lar* was detected at every sample site but not on every day. S5-530m exhibits the highest percentage of days detected (50%) and S2-19m has the lowest (12.5%). S7-1087m and S1-0m also both display a high percentage of days detected over 40%. The binomial GLMM found no significant relationship between the distance from the forest edge and presence of *Hylobates lar* when correcting for the day of the year and number of observation days per site P= 0.634.

There are 3 sample sites where *S. syndactulus* (Figure 34) was not detected on any days, S4-423m, S7-1087 and, S8-1217m. S5-530m displays the highest detection ratio with *S. syndactylus* being detected on 50% of the days. The second highest detection ratio is displayed in S10-1572m with *S. syndactylus* being detected 27.3% of days. No correlation is exhibited between distance from the forests edge and the presence of *S. syndactyluss.* The binomial GLMM found no significant relationship between the distance from the forests edge and presence of *S. syndactylus* when correcting for the day of the year and number of observation days per site P= 0.845.

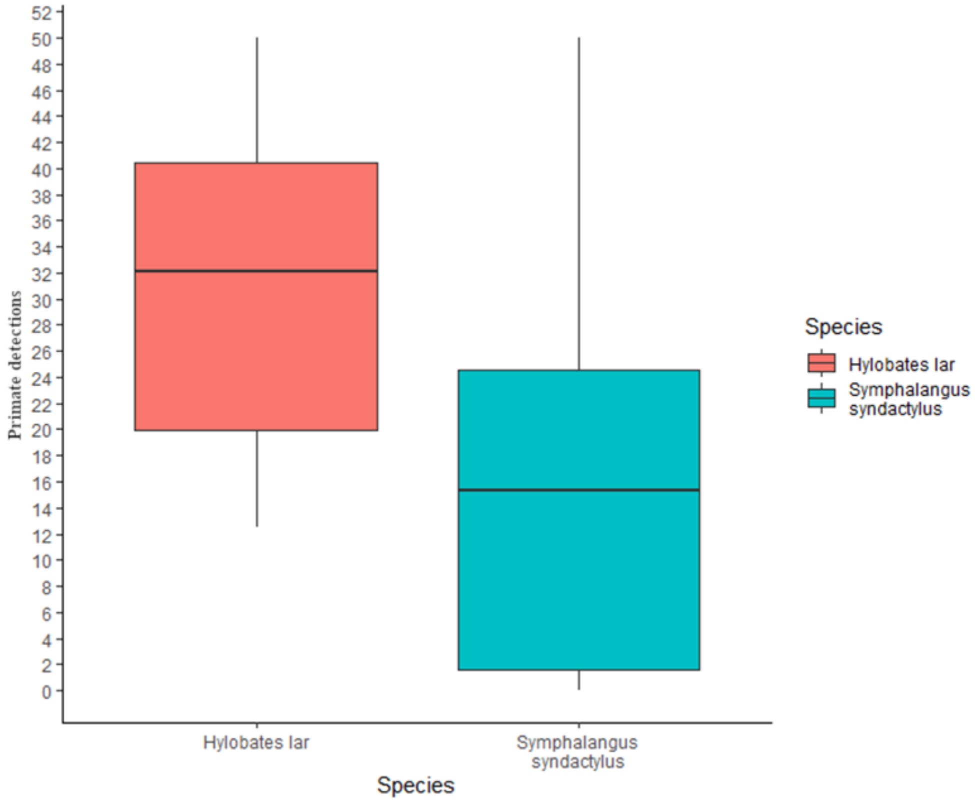

Figure 32: Boxplot comparing the total number of 1-minute recordings containing detections of Hylobates lar and Symphalangus syndactylus at all the sample sites, boxes represent quartiles whiskers as set to the 95th percentile and dots represent the outliers.

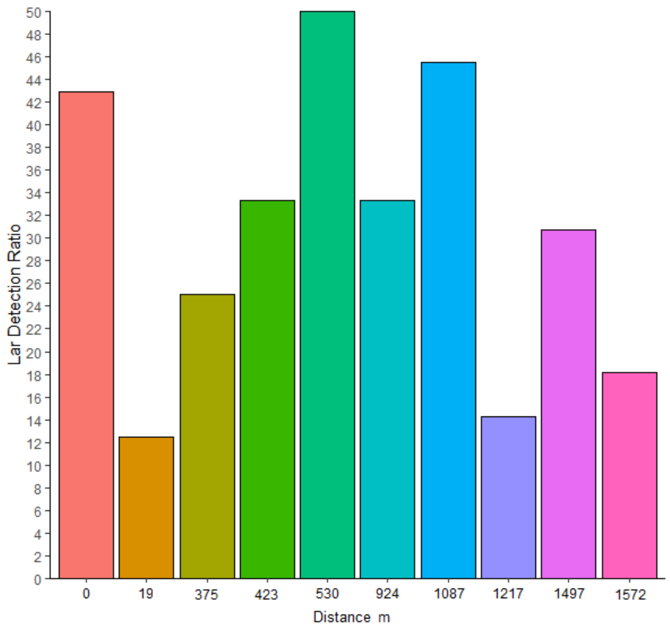

Figure 33: Bar plot displaying the percentage of days Hylobates lar vocalisations were detected in the morning call window 6:00 to 10:00 (detection ratio) for each sample site.

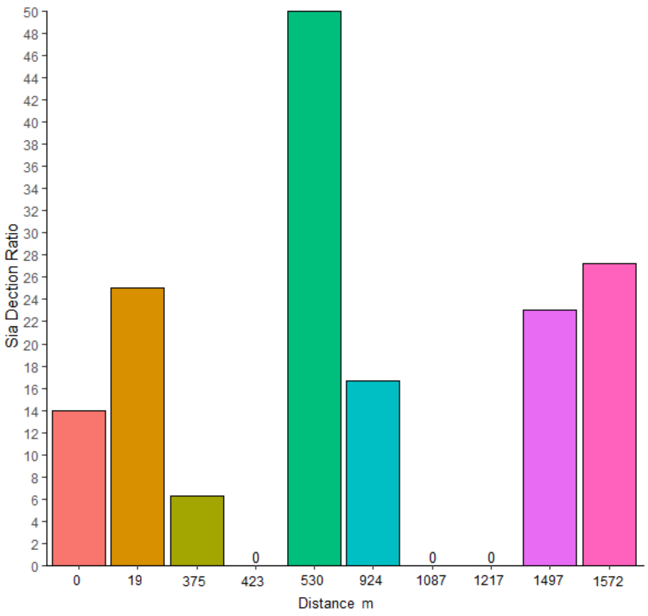

Figure 34: Bar plot displaying the percentage of days Symphalangus syndactylus vocalisations were detected in the morning call window 6:00 to 10:00 (detection ratio) for each sample site.

Table 20: Binomial GLMM of primate presence and daily ACI mean with distance as the predictor and day of the year and number of days recordings were made as random variables.

	Lar presence			Sia Presence		
Predictors	*Odds Ratios*	*std. Error*	*p*	*Odds Ratios*	*std. Error*	*p*
(Intercept)	0.40	0.12	**0.002**	0.17	0.08	**<0.001**
Distance	0.90	0.20	0.634	1.07	0.38	0.845
Random effects						
σ^2	3.29			3.29		
τ_{00}	0.45_{DOY}			0.22_{DOY}		
	0.00_{NoDays}			0.51_{NoDays}		
ICC				0.18		
N1	26_{DOY}			26_{DOY}		
	8_{NoDays}			8_{NoDays}		
Observations	108			108		
Marginal R^2 / Conditional R^2	0.003 / NA			0.001 / 0.183		

Gibbon presence and ACI in relation to forest structure

Apart from mean crown connectivity which exhibited a positive correlation to *S. syndactylus* (P = 0.002) and was close to significance with *H. lar* (P = 0.056) none of the forest structure variables showed any relationships in models using primate detections on more than one consecutive day (Table 21). Distance from forest edge was a significant predictor in the model for both primate species being detected more than one consecutive day, *Hylobates lar* P = 0.003 and *Symphalangus syndactylus* P = <0.001. The max daily ACI and the max dusk ACI are both significantly related to the max crown area in a site, however no significance was found between max dawn ACI and max crown area (Table 22). No other models found any significance between ACI and forest structure.

Table 21: Binomial GLM of primate detected more than one day consecutively with distance and mean crown connectivity as predictors.

	Lar Detection >1 Day Consecutively			Sia Detection >1 Day Consecutively		
Predictors	*Odds Ratios*	*std. Error*	*P*	*Odds Ratios*	*std. Error*	*P*
(Intercept)	1.02	0.55	0.972	0.33	0.21	0.089
Mean Crown connectivity	0.98	0.01	0.056	0.96	0.01	**0.002**
Distance	1.00	0.00	**0.003**	1.00	0.00	**<0.001**
Observations	110			110		
R^2 Tjur	0.121			0.263		

Table 22: GLMs of the ACI daily max, ACI dawn max and ACI dusk max with crown area as a predictor.

	ACI daily Max			ACI Dawn Max			ACI Dusk Max		
Predictors	*Estimates*	*std. Error*	*p*	*Estimates*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	0.00	0.27	1.000	-0.00	0.34	1.000	0.00	0.27	1.000
Crown Area max	0.61	0.28	**0.029**	0.03	0.35	0.931	0.58	0.29	**0.045**
Observations	10			10			10		
R^2	0.372			0.001			0.334		

Gibbon presence and ACI in relation to microclimate

No relationship was found between the daily max light intensity or the daily temperature range and presence of either of the primate species (GLMMs, Tables 23 and 24). However daily range in temperature were found to negatively correlate with the daily ACI mean (P=0.014, estimates of -0.19) and daily ACI mean was lower in sites with higher Daily Max light intensity (P = 0.012, estimates of -0.22). However no significant relationship was found between daily max light and daily presence of *Hylobates lar* or *Symphalangus syndactylus* (Table 23).

A negative relationship was found between the max dawn ACI and mean dawn temperature (Table 25) (P = 0.043) and distance (P = 0.001) when correcting for day of the year, however no significance was found in the dusk ACI mean (Table 26). Similarly, a negative corelation was found between the dawn ACI median Table 27), the light intensity (P = 0.017) and distance (P = <0.001) when correcting for day of the year but no significance was found in the dusk ACI (Table 28).

Table 23: Binomial GLMMs of daily primate presence and daily ACI mean with each site's daily temperature range as the predictor and day of the year and number of days recordings were made as random variables.

	Lar presence			Sia Presence			ACI Daily Median		
Predictors	*Odds Ratios*	*std. Error*	*P*	*Odds Ratios*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	0.40	0.12	**0.002**	0.17	0.08	**<0.001**	-0.08	0.29	0.796
Daily Temperature range	1.14	0.25	0.567	0.96	0.33	0.909	-0.19	0.08	**0.014**
Random Effects									
σ^2	3.29			3.29			0.47		
τ_{00}	0.44 $_{DOY}$			0.22 $_{DOY}$			0.62 $_{DOY}$		
	0.00 $_{NoDays}$			0.47 $_{NoDays}$			0.63 $_{NoDays}$		
ICC				0.17					
N	26 $_{DOY}$			26 $_{DOY}$			26 $_{DOY}$		
	8 $_{NoDays}$			8 $_{NoDays}$			8 $_{NoDays}$		
Observations	108			108			108		
Marginal R^2 / Conditional R^2	0.005 / NA			0.000 / 0.175			0.073 / NA		

Table 24: Binomial GLMMs of daily primate presence and daily ACI mean with each site's daily max light intensity as the predictor and day of the year and number of days recordings were made as random variables.

	Lar presence			Sia Presence			ACI Daily Median		
Predictors	*Odds Ratios*	*std. Error*	*P*	*Odds Ratios*	*std. Error*	*P*	*Estimates*	*std. Error*	*P*
(Intercept)	0.40	0.12	**0.002**	0.17	0.08	**<0.001**	-0.08	0.29	0.783
Daily Light Intensity max	1.14	0.24	0.534	0.91	0.33	0.793	-0.22	0.09	**0.012**
Random Effects									
σ^2	3.29			3.29			0.47		
τ_{00}	0.41_{DOY}			0.22_{DOY}			$0.\ 75_{\text{DOY}}$		
	0.00_{NoDays}			0.44_{NoDays}			0.62_{NoDays}		
ICC	0.11			0.17					
N	26_{DOY}			26_{DOY}			26_{DOY}		
	8_{NoDays}			8_{NoDays}			8_{NoDays}		
Observations	108			108			108		
Marginal R^2 / Conditional R^2	0.005 / 0.115			0.002 / 0.168			0.094 / NA		

Table 25: GLMM of max dawn ACI with mean Dawn temperature and distance as predictors and day of the year as a random variable

	ACI Dawn Max		
Predictors	*Estimates*	*std. Error*	*p*
(Intercept)	0.04	0.12	0.752
Mean Dawn Temp	-0.19	0.09	**0.043**
Distance	-0.31	0.09	**0.001**
Random Effects			
σ^2	0.71		
τ_{00} DOY	0.16		
ICC	0.18		
N DOY	26		
Observations	108		
Marginal R^2 / Conditional R^2	0.101 / 0.267		

Table 26: GLMM of max dusk ACI with mean dusk temperature and distance as predictor and day of the year as a random variable.

	ACI Dusk Max		
Predictors	*Estimates*	*std. Error*	*P*
(Intercept)	-0.01	0.15	0.971
Mean Dusk Temp	-0.05	0.10	0.644
Distance	0.03	0.08	0.737
Random Effects			
σ^2	0.65		
τ_{00} DOY	0.36		
ICC	0.36		
N DOY	26		
Observations	108		
Marginal R^2 / Conditional R^2	0.003 / 0.359		

Table 27: GLMM of max dawn ACI with mean dawn light intensity and distance as predictor and day of the year as a random variable.

	ACI Dawn Max		
Predictors	*Estimates*	*std. Error*	*p*
(Intercept)	0.03	0.12	0.777
Mean Dawn Light Intensity	-0.22	0.09	**0.017**
Distance	-0.35	0.09	**<0.001**
Random Effects			
σ^2	0.68		
τ_{00} DOY	0.18		
ICC	0.21		
N DOY	26		
Observations	108		
Marginal R^2 / Conditional R^2	0.110 / 0.296		

Table 28: GLMM of max dusk ACI with mean dusk light intensity and distance as predictor and day of the year as a random variable.

	ACI Dusk Max		
Predictors	*Estimates*	*std. Error*	*P*
(Intercept)	-0.01	0.15	0.972
Mean Dusk Light Intensity	-0.05	0.09	0.567
Distance	0.01	0.09	0.934
Random Effects			
σ^2	0.65		
τ_{00} DOY	0.36		
ICC	0.36		
N DOY	26		
Observations	108		
Marginal R^2 / Conditional R^2	0.003 / 0.363		

Predictions and findings

Table 29: Predictions tested and findings.

Driver	Prediction	Prediction supported
Edge effect	No correlation between forest structure and distance from the forests edge will be found due to the historic logging at the site impacting the typical succession of a forest.	Yes
Edge effect	Microclimate will correlate with distance from the forest edge showing a decrease in daytime temperature and light intensity as distance increases, due to decreased penetration of solar radiation though the canopy.	Yes
Edge effect	The number of machine hums detected will decrease with distance from the forests edge.	Yes
Edge effect	The acoustic diversity will correlate with distance from the forest edge showing an increase in activity of vocal fauna further from the forests edge.	Partly (Daytime)
Edge effect	The presence of *Hylobates Lar* will increase at areas further from the forests edge.	No
Edge effect	The presence of *Symphalangus syndactylus* will increase at areas further from the forests edge.	No
Forest structure	The presence of both *Hylobates lar and Symphalangus syndactylus* will be higher at areas with high percentage of canopy connectivity and trees with a large height depth ratio (representative of more mature forest), providing good shelter from high temperatures and solar radiation, increased microhabitats, and access to resources.	Party (Canopy connectivity)
Forest structure	During peak heat hours (11:00-14:00) vocal fauna will be more active in shaded areas with higher canopy connectivity and larger crown areas, due to heat stress.	Partly (crown areas)
Microclimate	The acoustic diversity will correlate with times and sites with lower temperatures and light intensity.	Yes
Microclimate	The presence of both *Hylobates lar and Symphalangus syndactylus* will correlate with sites of lower temperature and light intensity.	No
Anthropogenic disturbance	Fauna will be more vocally active in areas of less anthropogenic disturbance through machine hums.	No

Discussion

This study investigated how variations in forests structure, microclimate variables, anthropogenic disturbance and distances from the forests edge impacted the acoustic diversity of fauna and presence of two gibbon species. The analyses showed no correlation between forest structure and distance from the forests edge but a relationship between microclimate and distance suggesting presence of the buffer effect of the forest canopy. Anthropogenic disturbance from machine hums decreased further into the forest but had no impact on the acoustic diversity or presence of gibbon species. Maximum acoustic diversity at dawn was negatively correlated with distance to forest edge, but maximum dusk ACI showed no trends. Similarly, no relationship was found between the distance from the forests edge and the median ACI at both dawn and dusk. Interestingly, the median daily acoustic diversity showed a positive relationship to distance from forest edge. The presence of the gibbon species was not impacted by distance from the edge but the length of their stay at a site was.

There was a greater diversity of vocal fauna in sites with more stable microclimates and lower light intensity. These findings were reflected in the dawn ACI but not the dusk ACI suggesting the taxonomic groups that are vocal at dawn are more susceptible to changes in microclimate than those vocal at dusk. No correlations were found between the microclimate variables and either of the gibbon species suggesting they are not susceptible to changes in microclimate. However, *S. syndactylus* did show a positive relationship with the crown connectivity of trees and *Hylobates Lar* showed a similar trend, partly supporting the findings of Hankinson et al. (2022). The daily max ACI and the dusk max ACI correlated positively with crown area of trees but no other forest structure variables.

Forest structure edge effect

None of the models showed any correlation between the distance from the edge and the forest structure, suggesting other variables were more likely to influence the disturbed lowland forest than the edge effect, or sample size was not sufficient enough to detect such an effect. Previous logging in the area has contributed to changes in the forest's structure, within an undisturbed or unlogged forest it is more likely to see the edge effect's impact on forest structure (Hentz et al. 2018). Although the site is in the buffer zone of a national park and under some protection, it is likely that illegal logging is continuing at the site and impacting the forest structure into the interior to the 1.5 km that was studied in this study.

There is also abiotic variation in the sites such as lakes and rivers which will have impacted the number of trees present and thus the data collected at site. An example of this is also seen within site S10-1572m, as this site had a fallen tree. So, although S10-1572m was further away from the forests edge than any other site, it contained very few trees with a diameter larger than 10cm. This may have impacted the models and therefore not as accurately depicted the impacts of the edge effect on forest structure.

Studies with a larger sample size within the same disturbed forest such as Slater et al. 2021 found significant increase in tree height with distance from the forests edge. However, this variation was limited as no other forest structure variables such as bole height, crown area, and canopy connectivity correlated with distance from the forest edge reflecting the findings of this study. This further demonstrates the impacts of degradation of the structure of a forest (Marsh et al.2022) when compared to undisturbed forests that show an increase in density of trees further from the edge (Hentz et al. 2018).

Microclimate edge effect

A negative correlation was found between the distance from the forests edge and temperature and light intensity, this supports the idea of the buffer effect (Ewers and Banks-Leite 2013). These changes are likely linked to the variation in forest structure creating microhabitats, which affects solar radiation, the most important determinant of temperature variation in this forest (Marsh et al.2022). The variation in temperature and light intensity varies more greatly across the day than across different locations, despite the correlation between light and temperature and distance from the forest edge.

Moreover, temperature and light intensity decreased sharply by distance after S1-0m and then showed a slow decrease, this may be due to logging practises in the area (Slater et al.2021) making the forest structure variation inconsistent. Therefore, the variation in climate seen in the model may not be representative of the edge effect but selection of the locations and placement of data loggers in shaded areas. In a larger sample Slater et al. (2021) also showed decreases to both temperature and light with distance, which corresponds with the climate buffer effect (Ewers and Banks-Leite 2013).

The edge effects' impact on soundscape ecology and ACI

The models negative correlation between dawn ACI and distance from the forest edge, along with levels of invertebrate activity near the edge of the forest at dawn is representative of the invertebrates need for thermoregulation (Ni et al., 2013). It is only once the invertebrates have gathered energy and the temperatures increase near the edges invertebrates move to microhabitats to avoid heat stress (Pincebourde and Woods, 2012; Woods et al., 2015). Which corresponds with the increase of ACI during daytime hours at sites further from the forests edge. This suggests in the dawn, fauna would be more active at sites nearer the edge and more active in more shaded microhabitats as the temperature increases throughout the day.

Furthermore, during the daytime the diurnal avian species where more active in sites reflective of shaded microclimate conditions. This also corresponds with the findings of Pollock et al. (2015) who found avifauna to be sensitive to light (Pollock et al., 2015) and would therefore move further into the forest as light intensity increase. This and the insectivore diets (Pollock et al., 2015) of many bird species suggests after the dawn chorus Avifauna move further from the forests edge in search of food and shade.

However, the number avifauna vocalisations above 3kHz, could be higher than detected as they could have been masked by Cicadas vocalising at the same and higher frequencies. Cicadas are very prevalent in the forest and their calls are known to mask the vocalisations of avifauna (Metcalf et al.,2021; Aide et al., 2017; Hart et al., 2015). This means that the greater perceived reduction in avifauna calls in the more exposed areas could actually mean that they were simply not detected.

One clear outlier to the overall patterns was S7-1087m which showed high ACI across multiple frequency bins at all hours of the day representative of most taxa, it is likely that the ACI scores at dawn and dusk for the site had low ranges due to high levels of signal masking from high presence of invertebrates (Metcalf et al.,2021; Aide et al., 2017; Hart et al., 2015). S7-1087m had high levels of invertebrate and avifauna presence throughout peak heat hours in the daytime and had high canopy connectivity and crown areas reminiscent of shaded microhabitats (Marsh et al. 2022). Further displaying evidence for invertebrate's use of microhabitats to avoid heat stress (Pincebourde and Woods, 2012; Woods et al., 2015), and the sensitivity of some avifauna species to light (Pollock et al., 2015). It also infers that many of the avian species at S7-1087m are likely insectivores (Pollock et al., 2015) there to prey on the high levels of invertebrates.

The spike in acoustic activity exhibited by all sites at dusk between 18:00 and 20:00 that hits the frequency bins between 0.3kHz and 12kHz suggests the increased vocal activity of diurnal avian species, invertebrates, amphibians, and arboreal mammals at the sites (Metcalf et al.,2021). As this high-level dusk acoustic activity is demonstrated across all sites, it is reflective of the models lack of correlations between dusk ACI and distance from the forests edge as there is always a high level of acoustic activity at dusk despite location.

The soundscape images display high diversity in acoustic activity between 3kHz and 6kHz at all the sites from 18:00 until dawn, suggesting the presence of nocturnal avian species, invertebrates, amphibians, and arboreal mammals at all sites (Metcalf et al.,2021). The dawn and night ACI may be affected by signal masking from the high levels of acoustic activity across multiple taxa (Metcalf et al.,2021; Aide et al., 2017; Hart et al., 2015). This would impact the daily mean ACI and dusk ACI scores meaning a less accurate ACI was modelled against distance from the forests edge during night-time hours and may not truly reflect the impacts of the edge effect at these times.

The higher levels of acoustic activity observed at night in the frequency bins between 8kHz and 10kHz in sites S1-0m, S5-530m, S7-1087m, S81217m, S9-1479m and S10-1572m suggests there are more invertebrate and amphibian species present at these sites than any of the other sites (Metcalf et al.,2021). The presence of amphibians at these sites is most likely due to the sites being closer to a water source or due to high levels of invertebrate species present creating a food source for the amphibian species (Attard, 2011). However, due to signal masking (Metcalf et al.,2021) it is hard to determine which sites exhibited higher levels of amphibians or invertebrates. Due to amphibians having low vagility (deMaynadier and Hunter, 2000, Bowne and Bowers, 2004) and multiple vulnerabilities (Kiesecker et al., 2001, Baillie et al., 2004) it is likely that most of the soundscape is encompassed by higher levels of invertebrates than amphibians.

The high levels of acoustic activity at frequencies above 12kHz during dusk and night hours at S5-530m and S10-1572m suggests a higher presence of bats at these sites (Metcalf et al.,2021) and is most likely linked to dietary choice rather than the edge effect. However, this taxon ranges into frequencies much higher than that picked up by the FRD audio recorders so not much can be said about the presence of the taxon at other sites. Therefore, this study shows little of the edge effects impact on bats due to capturing only the lower frequency vocalisations of bats creating a small sample size. However, tropical frugivore bats species have been found to be more active in isolated trees and their abundance was found to increase with distance from a forest’s fragmentation (Galindo-González and Sosa 2003). Stressing the need to research further into the susceptibility of this taxon to fragmentation of tropical forests.

Anthropogenic disturbance edge effect

Human activity (measured in machine hums) drops with distance from the forests edge. This is reflective of other findings (Peres 2001) showing the higher levels of anthropogenic disturbances at edges. However, machine hums do not efficiently measure the presence of anthropogenic activity and as the study location is in disturbed lowland forest the presence of other anthropogenic activity is likely. Slater et al. (2022) found presence of humans at many of the sites used for both studies. This would imply that human disturbance was occurring at higher levels further into the forest than this study found and the impact of this presence on distribution of fauna needs further research. Peres et al, 2006 suggests using remote sensing to detect a variety of anthropogenic disturbances damaging to biodiversity such as forest fragmentation, mining, slash and burn agriculture.

The prominent levels of acoustic activity at frequencies of less than 1kHz at the site closest to the edge is reflective of the high levels of machine hums heard at that site and the negative correlation found between machine hums and distance from the forests edge. This is most likely due to S1-0m being closer to the road than any other site and the detected machine hums being reflective of vehicles. It may also be due to S1-0m being part covered with forest and part open farmland, as sound can travel further across open land without losing amplitude (Towsey et al., 2014). This also is reflected in research that found using ACI in urban areas unreliable due to interference from urban noise (Towsey et al., 2014; Pieretti et al., 2015; Sánchez-Giraldo 2020; Fairbrass et al., 2017), suggesting the ACI results from S1-0m may not be comparable to those at other sites due to signal masking from the urban noise from the roadside not being filtered (Fairbrass et al., 2017). However, as this paper was also measuring the impact of anthropogenic disturbance the road sounds were deliberately not filtered out.

Hylobates lar and *Symphalangus syndactylus* edge effect

No correlations were found with either *Hylobates lar* or *Symphalangus syndactylus* presence and distance from the forest's edge, suggesting neither species is impacted by the edge effect. However, both species did show a positive correlation with the edge effect when the length of their stay and crown connectivity was considered. This may be due to gibbon species having a need for connected canopies for travel due to their brachiation-type locomotion (Brockelman et al., 2014) and for foraging (McConkey 2009). In this model the R value for *Symphalangus syndactylus* was lower than that of *Hylobates lar,* this suggests the species may be less susceptible to the impacts of the edge effect than *Hylobates lar*.

A higher presence of *H. Lar* was recorded at sites near the forest edge than *S. syndactylus.* The elevated levels of visits from *H. lar* at these sites suggests the sites have resources that the species valued there more than the impacts of the edge effect. This may be due to forest structure variables such as DBH this supports the findings of Hankinson et al., (2021) that *H. lar* group density correlates with DBH between 30-100 but *S. syndactylus* did not. The high percentage of crown connectivity at these sites equates for the visits to the sites from S. *syndactylus* and fits with the findings of Hankinson et al., (2021). In sites that *Hylobates lar was* recorded a higher percentage of days than *Symphalangus syndactylus*.

The bar plots also show both primate species being present most regularly, 50% of the days, at site S5-530m. However, none of the forest structure variables at the site align with Hankinson et al., (2021) findings of what constitutes prime hylobatid habitat. Suggesting that the analyses here may not be identifying important determinants of hylobatid site selection due to the sample site size being 25m x 25m which is much smaller than the home range of gibbon species or samples sites used by Hankinson et al. (2021). Also, as no information was gathered on presence of food sources (McConkey 2009) or sleeping trees (Marsh et al. 2022) which if were present at the site may have been attracting both gibbon species. Okuda et al.

(2022) found that *H. Lar* species vocalise in areas canopy trees of higher elevations and prefer primary forest, suggesting in unfragmented forest an edge effect may exist.

Susceptibility of diversity (ACI) to variation in forest structure

The daily max ACI and the dusk max ACI correlated positively with max crown area of trees but no other forest structure variables, suggesting vocal fauna are more active at areas with trees that have large crown areas during daylight and night hours but not dawn hours. This is most likely due to the microhabitats formed in the shade that attract invertebrates avoiding heat stress during peak heat hours (Pincebourde and Woods, 2012; Woods et al., 2015). These invertebrates would then likely attract an array of fauna to prey on them, including avifauna, amphibians and primates, all of which are also vocal at dusk and during the night (Metcalf et al., 2021). There is also the possibility that trees with large crown areas are fruiting trees such as Ficus species that fruit all year round (Milton et al. 1892), these trees would also attract an array of frugivore species.

Susceptibility of *Hylobates lar* and *Symphalangus syndactylus* to variations in forest structure

Symphalangus syndactylus was more likely to be present in sites with high canopy connectivity and presence of *Hylobates lar* was very close to significance (P=0.056) in sites with high canopy connectivity, suggesting both species are susceptible to variation in canopy connectivity. This supports work in the same forest by Marsh et al. (2022) showing how siamang travel routes run through taller forest areas, and the writings of Cheyne et al (2023) suggesting gibbons spend time in high areas of the canopy (Cheyne et al. 2023; Okuda et al. 2022). Their presence in the areas of higher canopy connectivity may be to avoid heat stress due to shade created from the canopy (Hankinson et al. 2021). This may also be due to the arboreal lifestyles of the gibbons and their locomotion of branching (Fleagle and Lieberman 2015) being reliant on tree canopies being connected to gather food and move around their home ranges. This would suggest many other gibbons and other arboreal species are also dependent on these areas with highly connected canopies. If forest degradation continues, arboreal species may struggle to find food and be forced into territories of other species, causing conflict over rescores (Brockelman et al., 2014).

These findings along with the findings of Hankinson et al. (2021) suggest that the two primate species are quite susceptible to the impacts of deforestation and habitat fragmentation. Hankinson et al. (2021). also found diameter at breast height between 30–100 cm, tree heights between 20–25m, and crown areas between 100–300 m^2 to be heavily linked with the group density of *Hylobates lar* (Hankinson et al. 2021). These findings were not reflected in this study; however, this study did not investigate group density only presence of the two gibbon species. These differences may also be due to the smaller sample size of only 10 sites in a relatively smaller area of the forest used in this study or the missing dates with recordings. So, these preferences could well still be impacting the gibbon species as the forest becomes more degraded.

Susceptibility of diversity (ACI) to Microclimate

Temperature and light intensity range impacts daily ACI suggesting fauna like to stay in areas with a climate they are attuned too, with climate change increasing and temperatures predicted to bypass the 1.5ºC threshold by 2040 (IPCC 2022) more faunae will migrate north to cooler climates or will go extinct (Marsh et al. 2022). This is reflected in the plots of hourly ACI and both temperature and light intensity (figures 16, 18 and 28), in which the ACI of all sites drop during the peak times for temperature and light intensity 11-2pm, suggesting fauna are less vocal at these times and are impacted by the changes in climate. However, it is well known that invertebrates move to different microhabitats to avoid heat stress (Pincebourde and Woods, 2012; Woods et al., 2015) and many of the fauna use these invertebrates as a food source (Azman et al., 2011). So, it is unclear if all faunae are impacted by the changes in climate or whether, they are simply just following their prey. With increased fragmentation of forest there are fewer microhabitats for fauna to locate to.

The ACI microclimate models during the dusk hours and ACI levels throughout the night hours in both the soundscape images and ACI plots (figures 6-16 and 28) suggest that solar radiation throughout the day is driving the patterns observed and that nocturnal species are not impacted by the changes in microclimate as the climate is stable throughout this time. This supports research that suggests more species will move from diurnal to a nocturnal lifestyle as temperatures increase (Bonebrake et al., 2020). However, further research is needed into the monitoring of individual species' changes as temperature increases over a larger time span to prove this. Moreover, it is unclear whether future climate change may begin to impact the microclimate of night as peak temperature hours may increase overlapping into the dusk causing further phenological changes (Checa et al., 2014; Gill et al. 2015).

Susceptibility of *Hylobates lar and Symphalangus syndactylus* to variations in microclimate

No relationship was found between the presence of either of the gibbon species and the climate variables, suggesting they are not susceptible to variations in microclimate as measured in this study. This may be due to their arboreal lifestyles (Fleagle and Lieberman 2015) as they spend more time in tree cover reflected in this study's correlation between canopy connectivity and the findings of Marsh et al. (2022). In these areas of higher connectivity, the gibbons will experience less solar penetration and therefore are not as impacted by variations in microclimate (Marsh et al. 2022). The gibbons may also be using theses higher shaded areas to avoid heat stress (Cheyne et al 2023). However, it is important to stress the methodology of this study only measured temperatures at heights of 1.5 meters and therefore did not capture the temperatures of the canopy where the gibbons were present.

Furthermore, his does not mean arboreal mammals will not eventually be impacted by climate change, as the changes in climate are linked to the changes in plant phenology (Checa et al., 2014; Gill et al. 2015). This may in turn impact the food resources for both species, causing a decline in both the vulnerable species. It is also important to note that only the dawn calls of the species were logged in this study and to fully understand the impact of microclimate on the species further research into the peak temperature hours and the presence of the species should be accounted for.

Susceptibility of vocal fauna to machine hums

None of the models showed machine hums impacting the presence of *Hylobates lar*, *Symphalangus syndactylus* or general diversity (ACI). This may be as the hums of the machines occur at much lower frequency than that of the gibbons or the other vocal fauna, and therefore do not cause signal masking (Metcalf et al., 2021) or disturbance to the species. However, number of machine hums is not fully reflective of the full range of human activities that could cause disturbance. There are many papers showing the impacts of anthropogenic disturbance on wildlife (Pieretti et al., 2011). Barlow et al. 2016 found that biodiversity loss doubles in deforested areas with high levels of anthropogenic disturbances. Therefore, further research is needed into the impacts of anthropogenic disturbance on wildlife in disturbed forest areas along with indexing of anthropogenic disturbances.

Limitations and Recommendations

The study was limited by its sample size of only 10 sites only giving a distance up to 1572m into the forest. Gathering data further into the forest and increasing the number of sites at each distance may have provided more evidence towards the edge effect. The sample size was also limited by the number of successful recordings, due to a combination of the FRDs battery life and difficulty getting to all locations, meaning the FRDs could not always be retrieved before timing out. As an improvement to this it is recommended that more people are responsible for data collection to ensure maximum data are collected.

There is also the 1-2km that gibbon vocalisations can travel (Fan et al., 2010) that may impact these results, especially in open areas where sound can be detected at a higher amplitude that may have impacted the pattern matching.

The study was also limited as the methods used for data collection were not designed with this project in mind. It would have been more valuable to gather survey data on fruiting tree numbers and abiotic features such as river presence or lake presence. The locations and placement of the study sites helped encompass a wide variety of taxa but did not stick to any standardisation, for example some sites were placed next to a track which would also affect canopy structure, microclimate and human disturbance which may have impacted the results. This was partly done because the locations were also used for camera trap recordings (Slater et al. 2021). Unfortunately, it was hard to use satellite imagery to detect all the tracks in the forest, so no length from habitat fragmentation variable could be added.

There are also technical limitations such as determining the distance of fauna from the FRDs, so it cannot be certain all of fauna heard on the device resided within the 25*25m site. The use of ACI indices also has its limitations, sometimes an index can be unreflective of the soundscape due to issues such as signal masking from anthropogenic and geophonic sounds. As mentioned earlier this study made use of the anthropogenic disturbance data and knew it would not impede the study too much as it was in a non-urban area with low levels of human-caused noise but did not make use of geophonic sounds. It is therefore important in further studies using ACI that all recordings are checked first for geophonic sounds such as rain and those associated recordings are removed.

Conclusion

To conclude both the soundscape images and acoustic complexity indices (ACI) exhibited spatial and temporal variation in diversity and presence of different taxa. However, these variations were not clearly driven by the edge effect impacts on vocal fauna within the disturbed lowland forest. No clear correlation was found between the edge effect and the presence of *Hylobates lar* and *Symphalangus syndactylus* apart from length of stay. The forest structure variables varied at each site, but no edge effect was found, suggesting in this disturbed forest the forest structure is not impacted by the edge effect. Microclimate correlated negatively with the distance from the edge, suggesting an edge effect however, the timings of peak temperatures proved more impactful on location of vocal fauna than the microclimate edge effect.

Dawn ACI correlated negatively with microclimate variables, but day and dusk ACI did not, suggesting a movement of vocal fauna into the forest at peak climate intervals 11:00-14:00. Microclimate did not appear to currently be impacting *Hylobates lar* and *Symphalangus syndactylus* due to their arboreal lifestyles and correlation with areas with a large canopy connectivity. Trees with a large crown area correlated positively with ACI suggesting the creation of microhabitats for heat sensitive invertebrates that then attract insectivores into the forest at peak climate intervals. This also implies the presence of large fruiting trees such as Ficus species attracting frugivorous fauna such as *Hylobates lar and Symphalangus syndactylus.*

It is these frugivorous species as seed dispersers and invertebrates as pollinators that are key to the restoration of the degraded forests, and they are all dependent on large tree species. The degradation and logging of the forest is removing large trees and areas of large canopy connectivity. This happens most in the practice of selective logging where the larger more valuable trees are targeted. Although selective logging is often pushed as a more environmentally friendly practice it is now evident fauna depend on these large trees and areas of canopy connectivity as a source of food and for protection from heat stress. To conserve biodiversity within these degraded forests more must be done to protect the larger trees and maintain canopy connectivity, as they hold fundamental roles within ecosystem resilience.

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