

# **Using bioacoustics to assess the impacts of forest fire on primates.**

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## Using bioacoustics to assess the impacts of forest fire on primates.

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**Abstract —** The human-driven degradation of habitat has caused severe declines in biodiversity globally. Deforestation of highly diverse tropical forests directly extirpates species and increases forest vulnerability to fire. As a result, surface fires have become widespread across the Amazon, with more effective monitoring approaches needed to assess species responses and inform conservation strategies. Bioacoustic monitoring can be used to detect elusive species, providing essential information for their conservation. This study developed a bioacoustic methodology to assess the impacts of fire on forest primate species. Recordings were collected within areas of burnt and unburnt forest, and were subsequently analysed using the automated detection of vocalisations. Detection of presence was most effective for the more vocal *Alouatta discolor*, whereas *Callicebus moloch* and *Ateles marginatus* had a lack of detections. The species capable of adapting their diet had more presence in burnt forest, whereas the endangered forest specialist was only minimally found within unburnt forest. As a result, the irreplaceability of primary forest habitat is emphasised. Furthermore, recommendations for future primate bioacoustic monitoring is discussed.

**Keywords —** *Bioacoustics, forest fire impacts, primate ecology, conservation, biodiversity monitoring.*

**Word count —** 7544

## INTRODUCTION

### Forest disturbance and impacts

Global biodiversity has continued to decline as a result of anthropogenic pressures, with much concern over increasing threats to the hyper-diverse tropics (Alroy 2017). Across the tropical biome, deforestation has soared in the last few decades (Achard et al 2014). In particular, the expansion of agriculture has caused dramatic conversions of forest to pasture and crop land, with the fragmentation of original habitat (Tapia-Armijos et al. 2015). Consequently, local extinctions of many forest species are caused immediately, and over time, as forest area is reduced (Halley et al. 2016). However, direct and indirect impacts are variable between species, requiring complex conservation actions (Brodie et al. 2015). Furthermore, singularly maintaining forest cover alone does not ensure the conservation of biodiversity (Barlow et al. 2016). Conservation strategies therefore need to consider a multitude of different threats, some of which are difficult to monitor (e.g. hunting), yet have adverse effects on ecosystems (Peres et al. 2016). Likewise, species that are under threat can be elusive, requiring increasingly sophisticated methods of detection (Campos-Cerqueira & Aide 2016). Ultimately, more information on the threats to biodiversity and the subsequent responses of individual species can be used to better inform conservation strategies.

The degradation of forest habitat can have cumulative impacts due to complex ecosystem-climate interactions. During periods of drought, forest fires can ignite without anthropogenic stimulus, especially towards the end of the dry season. From such fires, forest structural recovery is often slow, and with changes to species compositions (Woods 1989). However, anthropogenic deforestation can instigate more frequent and severe fire disturbances, aggravating more permanent impacts on forest structure and ecological communities (Woods 1989). The opening up of forest canopies dries the forest interior, increasing susceptibility to combustion. Therefore increases in tree removal gives rise to a positive feedback interaction between forest loss and fire (Cochrane et al. 1999). Regardless, some types of forest are particularly more susceptible to fire, such as lowland Amazon flood forest in Brazil (De Almeida et al, 2016). This is due to canopy structural properties, including lower leaf area index, more gaps and a lower top-of-canopy (De Almeida et al, 2016). Therefore deforestation in such areas can create hotspots for fire susceptibility. In future scenarios, it is more likely that drier conditions will result in increasingly more severe fires (Malhi et al. 2008), thus raising concerns about subsequent impacts and mitigation.

Typically, forest recovery from fire is slow, with long lasting impacts on tree species composition (Chazdon 2003). Such changes to forest habitat inevitably impact the dependent forest dwelling species and communities. While some taxa eventually show signs of recovery after wildfire (Lemos da Silva et al. 2015), most have very different species compositions, even after a decade, compared to primary forest (Mestre et al. 2013, Silveira et al. 2013). For primates, immediate and short

term impacts are known to have severe effects on abundance and species composition (Barlow & Peres 2004). Different species and consumer types show heterogeneous responses due to their individual sensitivities to change (Peres et al. 2003). This can also be attributed to changes to forest resources. Abrupt and widespread increases in tree mortality are seen as a result drought and fire (Brando et al. 2014). Consequently, forest resources for many primates become depleted, with a lower abundance of fruit trees and fruit scarcity during the dry season (Barlow & Peres 2006). Species that decline in particular are large frugivores and primary forest specialists (Barlow & Peres 2006). In return, the ecosystem services that these primates provide in such disturbed areas of forest disappear too (Terborgh et al. 2008). Therefore an understanding of how primates respond to forest fire is integral to knowing how the functional community, and thus ecosystem, will change. Primates play a particularly vital role in seed dispersal and germination, consuming and defecating seeds of fruiting species (Arroyo-Rodríguez et al. 2015). Spider monkey faecal samples contained seeds from 133 different plant species (Link & Di Fiore 2006), indicating their importance in maintaining forest species diversity. Eventually forests deprived of frugivores result in the loss in tree diversity (Harrison et al. 2013). Failing to see the impacts of defaunation leads to cascading impacts and threatens tropical forest ecosystems (Wilkie et al. 2011).

### **Forest biodiversity monitoring**

Threats to biodiversity create the need for more effective and unified monitoring approaches that can reveal trends and consequences. This is especially relevant in the tropics, where declines in biodiversity remain a source of uncertainty due to the lack of widespread and high quality monitoring systems (Proença et al. 2016). Within protected areas, declines in biodiversity have been suggested to be less dire than previously thought (Beaudrot et al. 2016). However, for human impacted forests there is still a paucity of information where further monitoring is needed (Bustamante et al. 2016). Furthermore, the high species diversity, remoteness and environmental extremity of tropical forests engender difficulty in their successful monitoring. To fill the knowledge gap on human-modified forests, new initiatives have been established, such as ECOFOR in Brazil. Such projects recognise that key taxa, including dung beetles, birds, trees and large vertebrates are ideal indicators for monitoring change. These groups also occupy important roles and relationships in ecosystem functioning, thus they can be used to understand how such functions change according to impacts. The monitoring of these ecosystems can provide strong evidence for changes over time in ecological communities, assessing impacts and biodiversity loss. This evidence can be used to influence political decision makers with the authority to reduce activities that cause forest loss and degradation. Subsequent mitigating strategies can then also be reviewed and improved with further biodiversity monitoring. On the other hand, a lack of long term data can misinform decision makers, and lead to inadequate biodiversity conservation.

Whilst selecting the most important areas and taxa for monitoring is essential for conservation, so too is selecting the appropriate monitoring methods. Observer

bias found in traditional field surveys often results in the inadequate detection of all species present (Klingbeil & Willig 2015). Therefore crucial information can be overlooked. To combat this, field surveys can be intensified, ensuring all relevant ecological information is collected. However, intensive schemes are limited by their spatial area coverage (Proença et al. 2016). Furthermore, many endangered species have elusive behaviour and low population densities, thus reducing their likelihood of detection (Chadès et al. 2008). Consequently, imperfect detection remains a challenge for conservation, and needs to be met with more effective approaches. The use of automated monitoring is one solution. Camera trap and bioacoustic systems are non intrusive and can operate simultaneously in multiple locations, monitoring habitats 24h per day (Aide et al. 2013). This ensures cost effective monitoring over time, only requiring the analysis of recordings that are time stamped and permanent. Nonetheless, methodologies still need to be developed for these systems so that they can be applied to the monitoring of specific habitats and species.

### **Bioacoustic monitoring**

Bioacoustic recording systems provide a powerful tool in the monitoring of biodiversity. Most animals across different ecosystems produce species-specific sounds for a variety of communicative reasons. Therefore the automatic recording of these sounds can enable species detection without human presence. This is especially important for not interrupting species that adjust their behaviour in response to human activity (Hicks et al. 2013). Since bioacoustic methods are still being developed, there are no particular guidelines for their use in monitoring primates. While the approach has been effective for monitoring of some primate species (Kalan et al. 2015), it is debatable whether it should replace or complement traditional surveys. Nonetheless, spatial structure could remain similar to that of point-count count surveys. Within a targeted study area, multiple recording devices would likely be installed to gain a large spatial coverage. However, to avoid pseudo-replication, detection distances must be realised. The differences in vegetation density between habitats can lead to differences in sound travel (Darras et al. 2016), and thus this must be accounted for too. Sound detection tests involve using pre-recorded vocalisations to measure the furthest distance at which they can be detected. This information can also be used so that triangulation of detections can be achieved (van Kuijk et al. 2016). Once installed, devices can be programmed to collect data continuously or intermittently (e.g. 1 minute every 10 minutes), storing permanent recorded evidence. To extract vocalisation data from the recordings, species-specific algorithms can be developed that make detections based on unique sound patterns within the soundscape. Therefore the time that would be needed to listen to each recording and detect species is eliminated (Heinicke et al. 2015). Such algorithms have been developed for terrestrial groups including birds, insects, amphibians and mammals (Aide et al. 2013). To develop a pattern recognition model, a selection of training data is needed, including vocalisations from the species expected to be present. Raw sound files are converted into .wav format so that spectrograms can be viewed and analysed. Within these spectrograms, areas of high energy become regions of interest (ROI). ROIs that are identified as animal vocalisations are then

closely selected and used to train the species-specific model. This can then be applied to all the recordings to create classifications of presence or absence for that species in each recording. Often, presence can be falsely detected when vocalisations have similar ROIs to environmental variables, such as *Alouatta* species and thunderstorms (Aide et al. 2013). These would therefore need to be recognised by listening to and confirming the detections.

Through the analysis of presence and absence at sites, valuable information on populations and activity patterns can also be extracted. The presence data can be used to estimate density of a population using the frequency of vocalisations detected within a specified area (Darras et al. 2016, van Kuijk et al. 2016). Estimating abundances using bioacoustics is especially important for rare, elusive and endangered species where distributions remain uncertain (Campos-Cerqueira & Aide 2016). Furthermore, the accuracy of population estimates can be greatly improved using triangulation, with potentially important results for conservation (van Kuijk et al. 2016). For this, three devices with overlapping detection ranges are needed at each site, increasing the spatial precision at which individuals are detected. However, avoiding misleading results of occupancy must be achieved, either through a high model accuracy, or the manual validation of positive detections (Campos-Cerqueira & Aide 2016). Nonetheless, bioacoustic data has provided more accurate population estimates in comparison to human point count surveys that underestimated results (Kalan et al. 2015). Another output of bioacoustic monitoring is the behavioural information collected from vocalisations. Activity patterns during daily and seasonal cycles can be determined when vocalisations are plotted according to the timing of detection (Aide et al. 2013). By extracting information on occupancy and behavioural patterns, bioacoustics can effectively monitor the impacts of forest disturbance on these properties. Without acoustic detection, cryptic responses to forest impacts may go unnoticed, yet few studies have applied bioacoustics for this purpose (Wredge et al. 2010). Specifically, using bioacoustics as a method of monitoring the impacts of forest fire on primate species has not yet been developed.

## **Objectives and hypotheses**

This study aimed to advance our capabilities of monitoring primate species using bioacoustics. This was also used to test whether such a methodology could be applied to monitor the ecological impacts of habitat disturbance, and specifically forest fire impacts. These aims were achieved with the following objectives: i) to determine whether different primate species could be monitored by detecting their vocalisations in the Tapajos national forest; ii) measure the differences in primate vocalisation detectability and sound travel between different forest types; and iii) compare the results of acoustically detected primate vocalisations between burnt and unburnt areas of forest. The hypotheses are that: i) primate vocalisations will be successfully detected, however the ability of models to recognise vocalisations from other sounds will vary between species, with more distinct vocalisations being recognised more accurately; ii) vocalisations will be heard at greater distances within burnt areas due to a lack of vegetation muffling; and iii) more primate vocalisations

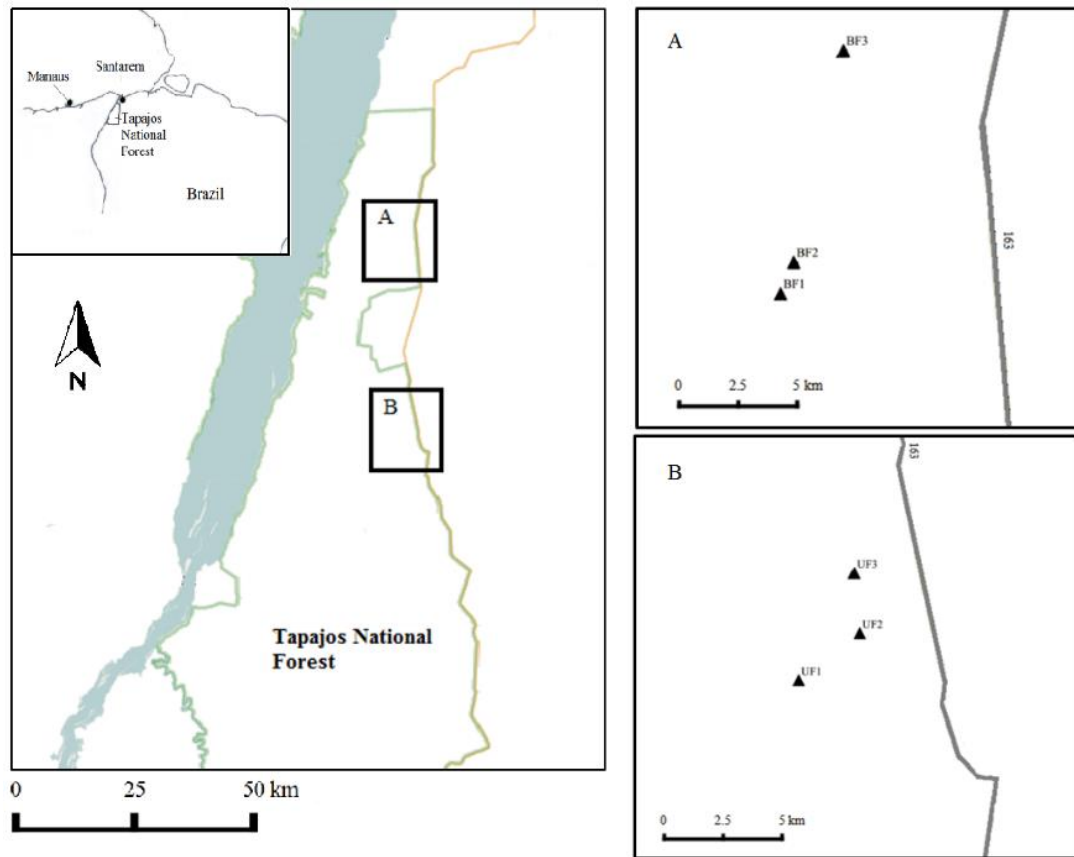
will be heard in unburnt forest areas as populations will be less impacted. Additionally in this study, the effectiveness of bioacoustic recording, detection software and research applications are reviewed. Furthermore, recommendations and future prospects for this technique are discussed.

## METHODS

### Study area

The study area was situated in the Tapajos National Forest, Para, Brazil (figure 1). The protected area of forest runs parallel along the north-western bank of the Tapajos, a clear-water southern tributary of the Amazon River. The western side of the forest is bordered by highway 163 running southwards from Santarém to Cuiabá. The habitats present include variations of Amazon lowland forest, including primary, secondary and burnt forest areas. Primate species distributions are not abundantly studied but have been documented previously (Branch 1983, Pimenta & Souza e Silva Junior 2005). In the wider Tapajos-Xingu region, 13 primate taxa were confirmed to be present with five in particular (*Cebus apella*, *Callicebus moloch*, *Chiropotes albinasus*, *Ateles marginatus* and *Alouatta belzebul discolor*) spread throughout (Pimenta & Souza e Silva Junior 2005). This study focused on three of these monkey species, the dusky titi (*Callicebus moloch*), white-cheeked spider (*Ateles marginatus*) and red-handed howler (*Alouatta belzebul discolor*). While *C. moloch* is of least conservation concern, *At. marginatus* and *A. discolor* are classified as endangered and vulnerable respectively. All three species are threatened locally by habitat loss and degradation. The Santarém-Cuiabá highway bisected the Xingu-Tapajos interfluvium, providing access for agricultural expansion. Furthermore, the highway also provides access for hunting, which *At. marginatus* is particularly susceptible to. The Tapajos national forest aims to oppose these threats and protects >545,000 ha of land.

In parts of the Tapajos National Forest, accidental surface fires have had destructive impacts in recent years. Between September 2015 and February 2016 surface fires intermittently swept across the now burnt area of this study, in the northern part of the forest. During this process, 11.7% of protected habitat was impacted. These surface fires consisted of a thin layer of fire moving across the forest floor. Slow moving animals were likely eliminated in the process and burnt vertebrate corpses were spotted on the highway border of the forest. The cause of the fire was likely to be accidental but initiated by local anthropogenic sources. Furthermore, the forest had become drier and more vulnerable to combustion towards the end of the dry season. Since this event, secondary regrowth has established, despite a high number of fire-induced tree mortalities.



**Figure 1.** Map of the areas studied within the Tapajos National Forest, showing the locations of each bioacoustic audio recorder (BAR). The Tapajos National Forest is located south of Santarém (see inset top left). Areas of burnt (A) and unburnt (B) forest were selected for device installation within predefined transects (e.g. BF1 = burnt forest transect one).

### Devices and installation

The devices used were Frontier Bioacoustic Audio Recorders. Their robust, waterproof and portable design suited their deployment in a tropical forest environment. Ten bioacoustic audio recorders (BARs) were installed in the Tapajos National Forest, five in each forest type (burnt and unburnt). 300m long transects were already established in the area as part of other ongoing studies. Each BAR was placed 30m away from the midpoint of the 300m transects perpendicularly. They were placed 8-10m high in trees that were suitable for climbing. Installing the devices away from trails and at this height was necessary to reduce their visibility to other forest users, preventing theft or vandalism.

Both areas of forest were monitored between May and June 2017. The BARs were set to record two of every ten minutes. This was twice the recording intensity of previous similar bioacoustic studies (Aide et al. 2013), but was necessary to increase temporal coverage due to the short duration of fieldwork. Recording over the 24 hours of each day also provided a comparative insight into daily activity patterns, which are not seen from studies that focus on particular parts of the day of which studied species are likely to present (van Kuijk et al. 2015). Assessing daily activity patterns can be used to understand the differences in behaviour due to habitat differences; to detect the presence and timing of other events (such as gun shots) in each area; and assess whether these events relate to primate behaviour. The original aim for recording effort was to install five BARs in each forest type and collect 20



days of data for each. However, due to unexpected errors, two of the BARs in the unburnt forest area only recorded <2 days of data. Due to the short duration of this study, the BARs were not re-installed to collect more data. To have an equal comparison of datasets, the three remaining unburnt area sets were selected for comparison with three burnt area sets. The minimum recorded time from this selection was 5.5 days, thus this length of time was used from each set. The recordings chosen subsequently for analysis commenced two hours after the device had been installed, so that human disturbance was not included, and finished 5.5 days after this.

### **Sound detection ranges**

To test the detection ranges of the BARs, we repeated sound transmission experiments for each plot. This also ensured that any sound transmission differences between transects and forest types could be accounted for. For the tests we used a Sharkk Boombox portable speaker mounted on a five metre wooden pole (figure 2). Raising the speaker to this height ensured that it was not muffled by the understorey vegetation, and reached a similar height to the installed BARs. Pre-recorded vocalisations of *A. discolor*, *C. mollarch* and *At. marginatus* were emitted at every 25m interval away from the BAR up to 350m. Vocalisations from each species were used so that differences in sound travel due to frequency could be compared. *Alouatta* species, for example, vocalise at low frequencies, a feature that promotes long range sound transmission (Mitani & Stuht 1998). Each vocalisation playback was calibrated to 90db using a decibel measurement android app (Sound Meter by Tools Dev), to standardise each vocalisation. All sound test recordings were listened to, until the emitted vocalisations for each species could not be heard. The furthest point at which each vocalisation could be heard was recorded. The spectrograms for each recording were also checked, to ensure that the sounds could no longer be identified beyond each maximum distance. This gave a maximum distance of detection (MDD) for each transect. Thus enabling comparison between sites, species and forest types.

Since the 90db playbacks were not equal to the volume of an actual primate vocalisation, the detection distances would not be accurate without further calculation. Therefore these results were multiplied by the maximum volume of each species to gain a realistic maximum detection distance. Since *Alouatta* species have the loudest vocalisations, they were a good measure of the maximum possible detection distance. However, for this reason, they would also likely cause an overlap in detection ranges between BARs, and therefore this had to be recognised to avoid duplicate detections. Thus *Alouatta* detections that were made by both BARs that were less than 4km apart were only counted as one detection.



**Figure 2.** Photograph illustrating a sound detection test. The portable speaker (A) was attached to the end of a five metre long makeshift wooden pole (B) and connected to a mobile device (C) with Bluetooth connection, so that prerecorded vocalisations could be played.

### **Automated detection**

Analysis of the recordings was mainly carried out using ARBIMON. This was used to create species-specific models using training data from recognised

vocalisations. However, before the analysis could take place, the recording data needed to be listened to manually in order to identify primate vocalisations for training data. This did not take long for more vocal species, such as *A. discolor*. However, for more elusive species such as *C. moloch* and *At. marginatus*, the process was more time consuming. To supplement the vocalisations identified for training data, previous recordings of these species were used and compared. Such recordings could be found by searching the xeno-canto<sup>1</sup> database for primate species heard in the background of recordings. Once species presence had been realised in a recording, their visual sound signature was selected in ARBIMON Visualiser. Selections had to fit closely around the signature to build accurate frequency (KHz) and duration (s) information for modelling. The inclusion of background noise in selections should be avoided to reduce the number of false positives identified when the model is applied. Training data was accompanied by a sample of recordings that had been identified as having each species present or absent, so that models could be validated during their creation. The species-specific models were applied to recording sets of each site, producing .csv files of presence and absence. If a vocalisation was identified at least once within a two minute recording, the recording was labeled as having presence for that species. If no vocalisation could be detected or not distinguished from other sounds, then the recording would be labeled absent. Model accuracy was determined by four outputs, the number of actual positives (presence) and negatives (absence), and the number of false positives and negatives (table 1). A higher accuracy was achieved through a lower number of false detections from the training data. This could be adjusted using a sensitivity threshold for detections. For this experiment, the threshold was lowered to increase the number of false positives and reduce the number of false negatives. This ensured that vocalisation detections were less likely to be missed. However, all positive detections had to be manually listened to, to distinguish between false and actual positives.

**Table 1.** Table indicating detection model accuracy. Actual results are predefined as training data, whereas predicted results indicate the detections made by the model using sample recordings for validation. The model makes an correct detection when actual presence is also predicted presence (PP) or the same for absence (AA). However false presence (AP) or false absence (PA) indicates the model made an incorrect detection for such recordings.

Actual	Predicted	
	P	A
P	24	10
A	5	46

P = Present, A = Absent

## RESULTS

### Vocalisation detections

Over 33 sampling days, *A. discolor* was identified as present 317 times. Whereas *C. moloch* and *At. Marginatus* were only identified as present 4 and 2 times respectively. The differences in vocalisations between forest types and transects for each species were compared (table 2).

**Table 2.** The number of automatically detected vocalisations of each species from the recordings collected from UF (unburnt forest) and BF (burnt forest) plots.

Transect	Number of vocalisations		
	<i>A. discolor</i>	<i>At. marginatus</i>	<i>C. moloch</i>
UF1	48	2	1
UF2	43	0	0
UF3	43	0	0
BF1	55	0	0
BF2	58	0	3
BF3	70	0	0

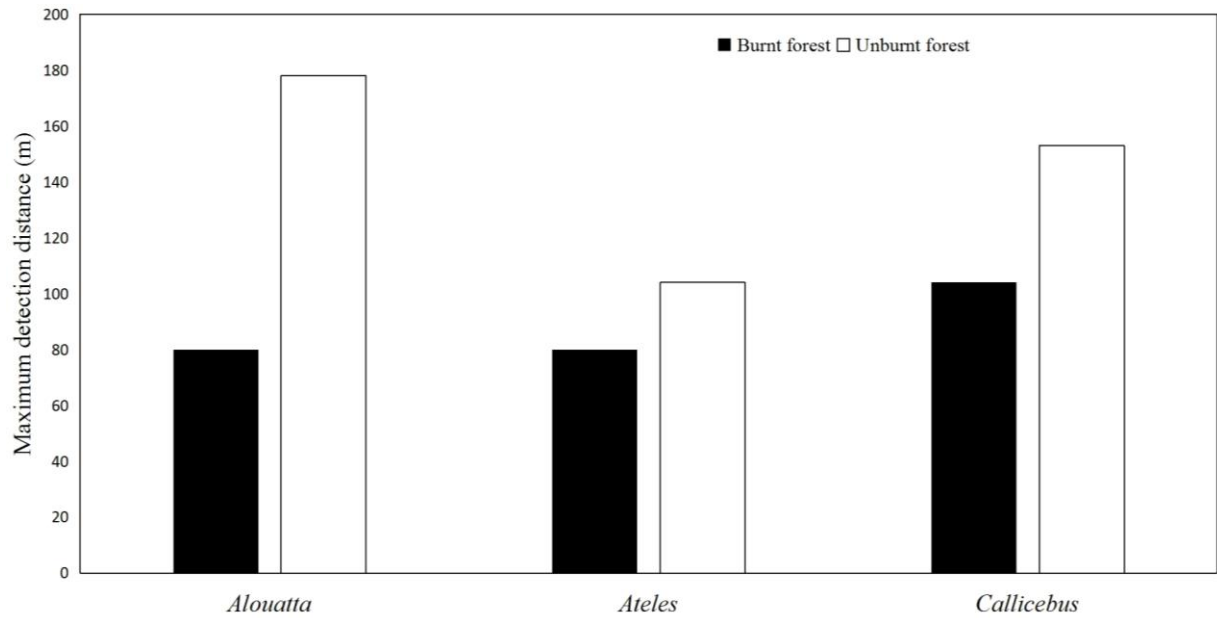
Since *A. discolor* generated the most vocalisation detections, this provided more data for behavioral activity analysis and comparison between sites. Overall, more *A. discolor* vocalisations were detected within burnt areas than unburnt, without a significant difference in the means of sites ( $P=0.079$ ). *At. marginatus* was only detected twice in unburnt forest, highlighting its sensitivity to disturbance and cryptic behaviour. *C. moloch* was present in both areas but was detected more in burnt forest. The models showed varying degrees of accuracy between species, mainly due to differences in the number of training recordings for each species. Model accuracy did not necessarily determine the number of detections but poor accuracy resulted in more false positives, thus requiring more time to exclude these from analysis. The *Alouatta* model was effective at identifying vocalisations, with an accuracy of 0.82. However, the remaining 18% of false positives identified had to be

removed by listening to each of the 849 identified recordings. False positives were commonly identified as rainfall, thunder or wind because the consistent, prolonged spectrograms at low frequencies were similar to *A. discolor* howls. Model accuracy was highest for *At. marginatus*, detecting the only two vocalisations among all recordings with 100% accuracy. Whilst testing different models for *At. marginatus*, many false positives were identified as avian vocalisations including those of *Ramphastos* spp.. For *C. moloch*, the ability of a specific model was limited as distant vocalisations were sometimes not recognised. This was likely due to excess noise interference, creating ambiguous spectrograms, despite the vocalisation being identified by the human ear. False negatives such as this indicate that more vocalisations could have been detected, either through a more accurate model or by listening to all of the data. However, there was neither enough training data for a more accurate model, or time available for listening.

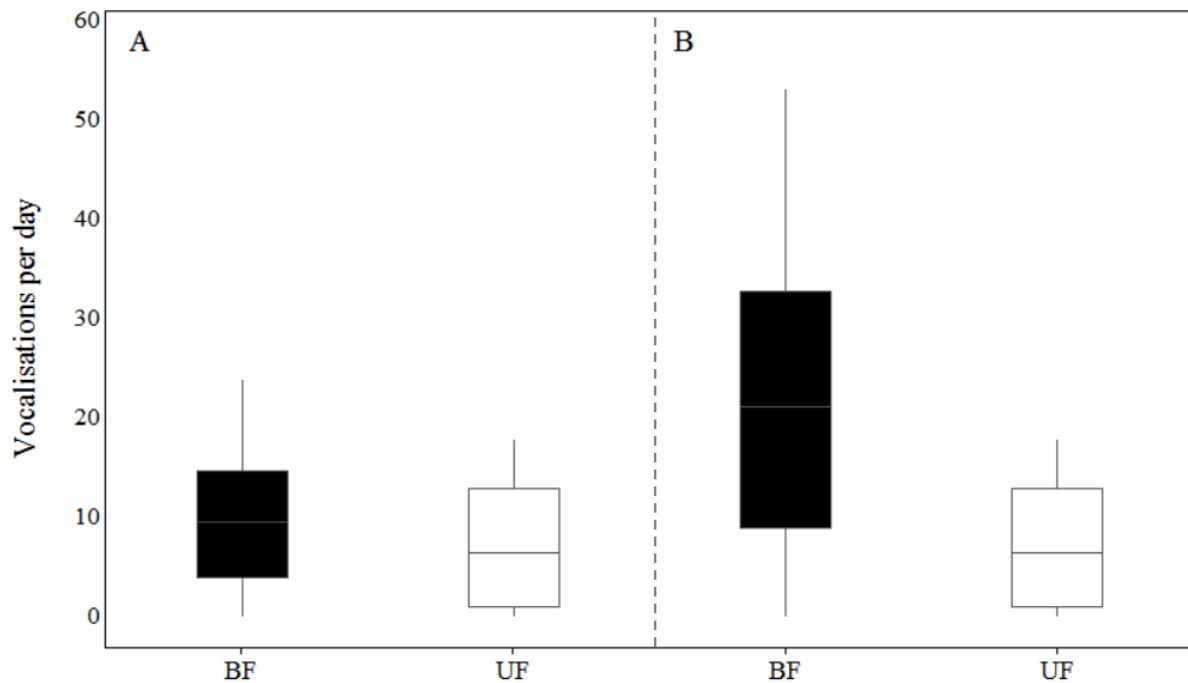
### Sound detection ranges

Sound test results revealed the differences in the maximum distance of detection (MDD) between each species and forest type (figure 3). Vocalisations could be heard at greater distances for all burnt forest plots, and were greater by 100, 20 and 50 metres for *Alouatta*, *Ateles* and *Callicebus* playbacks respectively. Differences between forest types were difficult to compare as windy conditions impaired the detection distances for the burnt forest sound tests. As a result, MDDs were much lower as the prerecorded vocalisations could not be identified at further distances, especially as wind speed increased, causing interference. This would conclusively not represent the differences in MDD between forest types, which would need to be carried out in very similar weather conditions. However, interior forest wind speed could be higher as a result of more forest gaps and sparse canopy. The BAR was particularly more likely to receive the muffling effects of greater wind speed due to its elevated placement. As wind speed could be continuously higher in burnt forests because of these factors, these sound test results may simply reflect these differences throughout the study. Therefore contrary to expectation, higher vegetation density within the unburnt forest did not directly muffle MDD. Instead it could have shielded the BAR from windy conditions and improved MDD in comparison to the sparser, burnt forest.

Using the abundance of *A. discolor* vocalisation data, the number of detections per day for burnt forest (BF) and unburnt forest (UF) were compared (figure 4). There was no significant difference in *A. discolor* vocalisations per day between BF and UF ( $P=0.308$ ,  $T=1.04$ ,  $DF=33$ ), in raw number of vocalisations per day. To account for the differences in detection between each site (figure 3), the vocalisation data for BF was multiplied using the sound test results to therefore match the MDD of unburnt forest. This produced estimates of BF vocalisations that assumably could be compared to UF. However, predicted spatial coverage depended on the accuracy of sound test results that were also influenced by weather conditions. Nonetheless, from this projection, there were significantly higher vocalisations per day in BF ( $P=0.003$ ,  $T=3.4$ ,  $DF=21$ ).



**Figure 3.** Differences in the maximum detection distance (MDD) between each species for each forest type. MDD was calculated as the average maximum distance that pre-recorded vocalisations could be heard at after they were replayed at 25 metre intervals away from the BAR during sound testing.

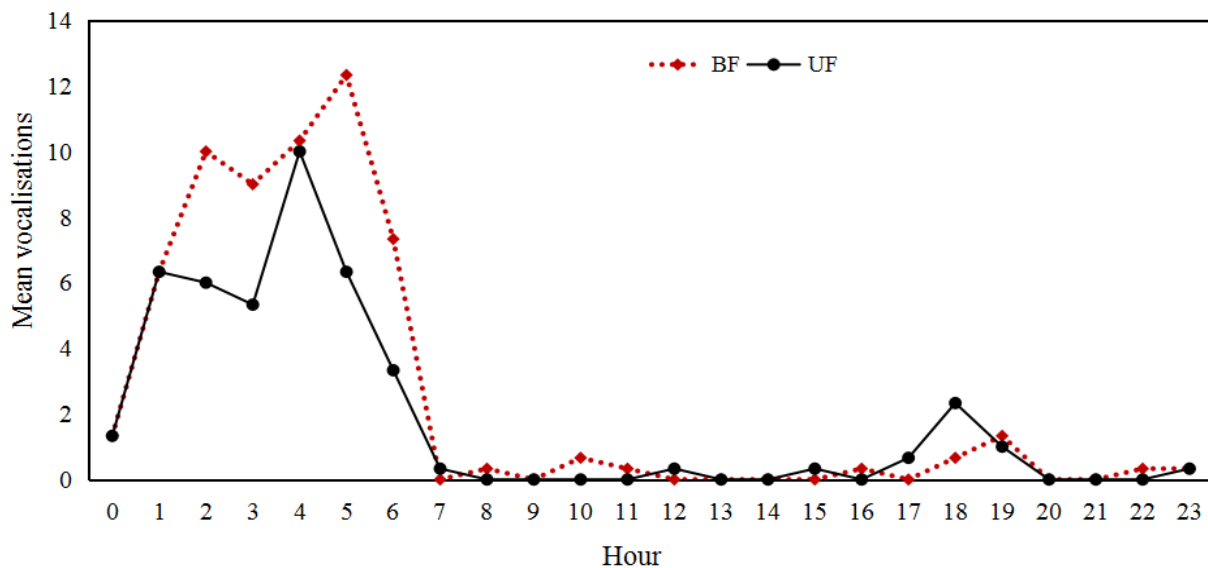


**Figure 4.** Boxplot of average number of *Alouatta discolor* vocalisation detections per day for burnt forest (BF) and unburnt forest (UF) sites. Unchanged data comparison (A) showed no significant difference between each area. However, when differences in detectability between each habitat were accounted for, projected data (B) showed significantly higher vocalisations in the burnt forest area.



## Alouatta activity

The *A. discolor* vocalisation detection results were grouped into hours of the day to view daily activity patterns and compare these between sites. This enabled the visualisation of cryptic differences in behavioural patterns (figure 5). The first six hours of the day was the most active for the species, with occasional vocalizations late afternoon or early evening. Within the first six hours, more vocalisations were detected in the burnt areas, especially at the hours of two, three and five. However, the differences over this period this were not significant ( $P=0.076$ ,  $T=1.81$ ,  $DF=37$ ).



**Figure 5.** Daily vocal activity patterns for *A. discolor*. Vocalisations detected by the species-specific model were grouped into the hours they were recorded. Although burnt forest detections were higher in total (burnt  $n=183$ , unburnt  $n=134$ ), both showed similar patterns of vocalisation throughout the day.

## DISCUSSION

Using bioacoustic methods, the monitoring of primates was achieved in the Tapajos National Forest in Para, Brazil. Bioacoustic detection data enabled the assessment of tropical forest fire impacts on primate species presence, and behaviour. It was found that, although all three species could be detected, only *A. discolor* could be effectively monitored. This was due to frequent and detectable vocal behaviour that could be used to compare detections between sites. Results indicated a larger presence of *A. discolor* within burnt forest due to a higher number of vocalisations per day and in total. On the other hand, *At. marginatus* and *C. moloch* were only minimally detected, either due to their low abundance or frequency of vocalisations. As a result, only a crude picture of presence and distribution could be gained for these species, only an expansion of sampling effort could facilitate testing their responses to fire more sufficiently. For *A. discolor*, sufficient data enabled an analysis of daily vocal activity. The results indicated that throughout both areas,

vocal behaviour was greatest during the first six hours of the day and mostly absent after this. Furthermore, vocalisations were more frequent during the first six hours for burnt forest, either suggesting higher density or altered vocal behaviour. This study recommends that bioacoustic monitoring is an effective tool in conservation, yet its purpose should be adapted according to the species and habitat under study. For highly vocal species, bioacoustic methods can be used to collect more in-depth information than presence, including abundance and behaviour. However, for elusive and rare species, a lack of bioacoustic detections can limit the depth of study. Nonetheless, detections alone indicate presence and crude distribution, facilitating direction for further study. For example, sampling could be intensified by installing more BARs in areas that previously had detections. Thus for important species under threat such as *At. marginatus*, questions of abundance and behaviour could be tackled.

### **Primate behaviour**

*At. Marginatus* had the least detections of all three species and was only detected in unburnt forest. This highlights its scarcity as an endangered species (Mittermeier et al. 2008), but also its sensitivity to disturbance. Although no population estimates have been made for this species, its current trend is decreasing due to a limited number of protected areas (Mittermeier et al. 2008). Within the Tapajos National Forest, it is also likely that the population could have been reduced by disturbances such as hunting, of which *Ateles* spp are particularly susceptible to. Furthermore, forest burning is a more extreme disturbance and thus *At. Marginatus* would not likely occupy such impacted areas. Moreover, the species was previously indicated to be very sensitive to fire and were found to be locally extinct in sites three years after wildfire (Peres et al. 2003). It is suggested that *Ateles* is more vulnerable to hunting than wildfire, although it can be difficult to assess the impacts of fire specifically when both areas would have been subject to hunting (Peres et al. 2003). Despite this, changes to habitat and particularly a decline in fruit resources following wildfire would have forced *At. marginatus* into unburnt areas of forest (Barlow & Peres 2006). This is due to their highly frugivorous diet, and that they would only resort to eating foliage if necessary during times of fruit scarcity, such as at the beginning of the dry season. As a result, their absence from burnt forests will subsequently lead to the loss of seed dispersal and the regeneration of species that rely on this (Terborgh et al. 2008). This includes more than 133 species that can be dispersed by *Ateles* (Link & Di Fiore 2006). Furthermore, lack of such dispersal for fruit trees would favour abiotically dispersed species (Terborgh et al. 2008). The ephemeral foraging behaviour of *Ateles* means that individuals and groups can move quickly through an area, and could have a lower chance of detection because of this. Further, their vocal activity patterns are less consistent, and quieter than *Alouatta*. To achieve better detectability of *Ateles* it may be necessary to complete bioacoustic studies in unburnt forests with a high population density. This would improve the repertoire of vocalisations for use as training data to create a more accomplished detection model. Application of such a model to disturbed forest would result in unabridged detections. The longer term sustainability of this species may rely on



bioacoustic monitoring of populations within protected forests such as Tapajos National Forest. However, this must be combined with efforts to reduce hunting and further fire outbreaks.

Similarly to *At. marginatus*, *C. moloch* vocalisations were not detected in abundance. However, this was unlikely to be due to the same reasons because *C. moloch* is not endangered or known to be in low abundance. More presumably, less vocalisations would indicate lower activity or a mismatch between BAR placement and *C. moloch* territories. The species was detected in both forest types suggesting a widespread distribution. More detections were found in burnt forest however. The reason for this is likely to be that *Callicebus* species can do well in burnt forests compared to other primates (Peres et al. 2003). In parallel to other forest generalists as *Cebus apella*, they are capable of subsisting in large areas of disturbed or secondary growth forests (Michalski & Peres 2005). This is enabled with dietary flexibility and the ability to adopt a highly conservative energy budget, whilst increasing foliage consumption relative to fruit pulp. Similarly to *At. Marginatus*, *C. moloch* are capable of dispersing seeds, albeit through smaller fruits and thus filling a disparate niche. Further, as *C. moloch* do not necessarily decline in secondary growth areas (Michalski & Peres 2005), their seed dispersal functioning would remain and possibly encourage the regeneration of reliant tree species. The specific vocal behaviour of *C. moloch* could have affected the number of detections. Models were trained to detect the ‘chirrup’ that comprise the distinctive *Callicebus* duets (Robinson 1979). The absence of duets would have increased the time needed for detection, in comparison to more frequently vocal species such as *Alouatta*. Furthermore, long distance detection could not be achieved through modelling, and false negatives remained undetected even after model adjustment. Both of these factors would have contributed towards a misleadingly low abundance, a limitation that would need to be overcome through more effective automated detection. The detection of *Callicebus* becomes increasingly more complex and variable due to their use of space as a territorial species (Mason 1986). One group will move to the edge of their territory in the early morning and call. However, calls depend on the responses of neighbouring animals at this boundary (Robinson 1979). Two problems arise from this: firstly, groups could appear anywhere along the edge of their territory and not necessarily within range of bioacoustic detection; secondly, lack of movement and responsive behaviour at the boundary could lead to less vocalisation (Robinson 1979), decreasing detectability. Furthermore, boundary setting results in exclusive areas with less encounters (Robinson 1979). As a result, spatial and temporal coverage should be maximised to increase the likelihood of detection for *Callicebus* species. Future studies using bioacoustics to estimate population densities would need to take account of territory ranges using mapping or triangulation (van Kuijk et al. 2015). Nonetheless, simply detecting presence within a habitat type can be achieved as demonstrated in this study.

*A. discolor* were distinctly more active in burnt forests due to higher numbers of detections. The following question from this is whether a higher number of vocalisations is due to higher abundance or simply more vocal activity. Once again,

this species has no population data, but is suggested to be decreasing and vulnerable (Boubli et al. 2008). The Tapajos National Forest is one of four key protected areas within the species' range. Still, much of the threats appear to be from forest loss and hunting, and so it is surprising to find that more detections were made within areas impacted by wildfire. The results conflict with previous *Alouatta* responses to fire where detections declined between unburnt and burnt, further declining for areas twice burnt (Barlow & Peres 2006). It was also found that primate species reliant on fruit were more resilient to fire than foliage browsers (Barlow & Peres 2006). This study instead indicates that primates capable of resorting to foliage are less sensitive to forest disturbance (Benchimol & Peres 2014). Furthermore, as facultative folivores, *Alouatta* species can adjust their dietary behaviour when fruit resources become scarce (Bicca-Marques 2003). Therefore even in disturbed areas where other primates would fail to persist, *Alouatta* species can become ubiquitous. Their ability to consume mature leaves in particular is advantageous. However, burnt areas would likely be composed of more preferable young leaves and secondary growth. An influx of *A. discolor* into the burnt forest area could be explained by this. However, as a mobile foraging species, it is also likely that they would be reliant on the close proximity of surrounding unburnt forest habitat. Movements between the two areas could enable exploitation of fresh leaf resources, whilst retreating to complementary unburnt refuges for survival. In similar fashion to *Alouatta*, generalist species of birds were more common in burnt areas and occupied multiple habitat types (Barlow et al. 2002). Thus for species with adaptable foraging behaviour, the impacts of forest fire can be overcome, using the unexploited area to advantage. Despite this, *Alouatta* population increases in burnt forest are not necessarily sustainable. In fact high populations can increase vulnerability to dependent factors such as disease and hunting (Bicca-Marques 2003).

The consistent and long distance (1-2km) vocalisation capabilities facilitated accurate detection of *Alouatta* (Drubbel & Gautier 1993), and if present they would always vocalise. The disadvantage of this is that pseudo-replication was possible, and concurrent vocalisation detections between BARs close together (<4km) had to be recorded as single detections. Nonetheless, sufficient detections facilitated the analysis in behavioral differences between areas. The low abundance of daytime vocalisations across both sites was dissimilar to previously seen *Alouatta* activity (Aide et al. 2013), with lower midday and afternoon peaks even for a wet season (Cornick & Markowitz 2002). This could have indicated an adaptive behavioural response to avoid detection by hunters during the day. However, since daytime calls are shorter in duration (Drubbel & Gautier 1993), they would have also had less chance of detection due to intermittent recording. Higher numbers of vocalisations throughout the early morning in burnt forest could have also suggested altered activity patterns. *Alouatta* increase their vocal responses to other groups over intrusion into high quality habitat (Whitehead 1989), which could include burnt forest due to due to preferably younger leaf resources. Thus aligning with this study's results of having higher detections in burnt forest. Other causes of loud vocal activity include group density and the distance to receiver (Whitehead 1989). To achieve population density estimates, triangulation of *Alouatta* vocalisations would

be easily achieved as long distance calls reduce the need for tighter spacing between BARs. On the other hand, population estimates are less crucial for viable species such as *A. discolor* compared to more threatened species. Nonetheless, changes in primate species and abundance indicate altered ecosystem functioning. As a result, *A. discolor* may play an important role in maintaining ecosystem functioning through seed dispersal within burnt areas as one of the few primates that can remain ubiquitous in such areas.

### **Bioacoustic monitoring**

The hypothesis that vocalisations could be detected at greater distances within burnt forest was not true (figure 3). It was expected that lower vegetation density would facilitate further sound travel in these areas. However, fast-growing secondary plants could have muffled sound travel, despite higher tree mortality and more gaps (Brando et al. 2014). Furthermore, as unburnt forests had more intact canopies, this would have reduced understorey growth at BAR height. Unexpectedly, more windy conditions within burnt areas, due to a more open canopy, reduced sound detection (figure 3). Thus density in terms of vocalisations per unit area could not be achieved. Furthermore, a speaker capable of playing vocalisation recordings at realistic volumes would have ensured more accurate detection ranges, as opposed to simply comparing relative sound travel. Darras et al. (2016) demonstrated the need to realise detection spaces in order to achieve bioacoustic monitoring of populations. In addition, failure to quantify such differences between sites can result in substantial bias for biodiversity estimates (Darras et al. 2016). Since previous attempts at primate bioacoustic monitoring have taken place within a single habitat type (Kalan et al. 2015), comparative tests were not necessary. Nonetheless, this study highlights the need for future bioacoustic studies to take into account environmental variables that can influence sound travel between habitats.

ARBIMON was effective at organising large sets of recordings according to site, date and time (Aide et al. 2013). The software was also capable of developing the automatic detection of species-specific sounds. However, detection accuracy varied between species due to differences in available training data. Therefore effective bioacoustic monitoring will depend on how vocal a species is, and thus how much training data can be acquired for modelling. A vocal species would possess the ability to emit loud vocalisations, combined with the behaviour to frequently repeat calls and enable presence detection. Failing to account for detectability differences between species will result in false significant differences in site occupancy (de Solla et al. 2005). For quieter primates, sampling intensity would need to be increased with more BARs to thus increase likelihood of detection. Furthermore, detecting quieter and distant vocalisations will require more advanced automated detection that can effectively discriminate smaller signals from background noise. Without this, manually listening to recordings will be necessary. Creating models that can also discriminate between vocalisations and false positives is difficult, especially in a highly noisy forest environment. Adjusting a detection threshold to eliminate false negatives meant that presences had to be confirmed manually (Campos-Cerqueira &

Aide 2016). Most false positives involved either rain, wind or thunder. These should be screened out before analysis to save time, and because no detections could be made due to completely unidentifiable spectrograms. Ultimately, developments in the automated detection capabilities will encourage the uptake and development of bioacoustic monitoring.

Advances in specialised technology enable increasingly more effective biodiversity monitoring. Systems can be established with the use of automated devices that send data through satellite. As a result, automated devices can be used to expand biodiversity monitoring across large and remote areas. Such extensive monitoring has been achieved in the marine biome (Sánchez-Gendríz & Padovese 2016). The approach has potential to be used across large areas of tropical forest with the development of standardised protocol. Such a project would initially require substantial investment in equipment, or the prioritisation of areas most in need of monitoring to reduce costs. Furthermore, the training of bioacoustic methods to operators would also be needed for successful project establishment. Nonetheless, volunteers can be motivated to use bioacoustic systems by the excitement of cutting-edge technology. In France the extensive monitoring of bats was facilitated through citizen scientists with the opportunity to use new technology (Penone et al. 2013). One main limitation of bioacoustic monitoring is the reliance on, and bias towards vocal species. Therefore it is vital that the vocal behaviour of each monitored species is recognised and accounted for. Nonetheless undetectable quiet species remain a blind spot when composing a species presence list, thus other techniques may need to be combined to complement this data. A combination with traditional methods could overcome this, or even olfactory systems. Olfactory systems can be used in similar fashion to the bioacoustic monitoring of biodiversity, but with the detection of species-specific biomolecules instead of sound (Raiteri et al 2002). The deployment of olfactory systems alongside BARs could be possible as the analysis capabilities of biomolecules have already been developed (Scott et al. 2006). Overall, studies testing the use of emerging monitoring technologies will aid methodology development for future application.

The use of bioacoustics can cultivate another dimension to biodiversity monitoring. As a result, more detailed information on biodiversity, populations and behavioural patterns can be used in understanding the current biodiversity crisis and responses to environmental change. Measured declines in biodiversity provoke the response of mitigating impacts by use of protected areas. For example in Brazil, government actions and enforcement of protected areas has reduced deforestation (Arima et al. 2014). This is also encouraged through linking forests to ecosystem services with economic benefits. However there are concerns that biodiversity may not sufficiently protected under ecosystem services schemes (Murray et al. 2015). Additionally, simply maintaining forest cover through protection does not guarantee effective biodiversity protection (Barlow et al. 2016). More subtle anthropogenic disturbances such as hunting and surface fires do not cause the complete removal of forest yet can eradicate many species (Peres et al. 2003). As a result, conservation strategies that preserve habitat but not biodiversity need to be revised. Bioacoustic

monitoring will be increasingly needed to ensure that conservation strategies protect many species that often go overlooked by traditional methods. These include endangered species such as *At. marginatus* that have been proven difficult to detect due to their low abundance and sensitivity to disturbance. Influencing public policies about the need for better protection of Amazonian forests should guide conservation strategies. Furthermore, forest vulnerability to fire, and the subsequent impacts of fire events on biodiversity (Peres et al. 2003) need to be recognised. Thus guiding measures to reduce the flammability of forests, especially those untouched and under protection such as Tapajos National Forest.

## CONCLUSIONS

The results collected in this study found little significance between primate detections in burnt and unburnt forests. However, more adaptable and generalist species *C. moloch* and *A. discolor* had more presence in burnt areas, suggesting such species could take advantage of degraded secondary growth habitat. A lack of competition for resources in an area deprived of biodiversity could have suited these facultative folivores, whilst using the surrounding unburnt areas as complementary refuges. On the other hand, *At. Marginatus* was minimally detected, and only within unburnt forests. As an endangered species, *At. marginatus* was likely to have low abundance, particularly due to its vulnerability to hunting. Nonetheless, it's detection alone indicates the value of maintaining unburnt forest to protect rare forest specialist species with important ecosystem functions.

Bioacoustic detection was effective and species-specific models successfully identified primate vocalisations within a highly noisy environment. The accuracy of detections varied between models and weather conditions such as wind and rain were often identified as false positives, requiring the manual confirmation of positive detections to remove these. Nonetheless, automated detection significantly reduced the time needed to identify vocalisations within large amounts of data. The capability for detecting presence was not always dependent on pattern recognition models, and was mainly due to differences in primate vocal activity and detectability. The more vocal species *A. discolor* was more effectively monitored than the other two primates as vocalisations were distinct, loud and regular, thus increasing detectability. As a result, presence was easier to compare between burnt and unburnt areas, with further information on vocal activity providing insight into behavioural patterns. This study therefore recommends that more vocal species are prime targets for bioacoustic monitoring, whereas quieter species would require increased sampling effort for detection.

The importance of bioacoustic monitoring was discussed with recommendations for future development. Ultimately the automated detection of species enables much more elusive species to be monitored, building a more complete picture of species present. Therefore bioacoustics will be a vital tool for understanding the impacts of environmental change on biodiversity. Such insight can provide strong evidence on the impacts of more subtle disturbances in tropical forests

such as hunting and surface fire. This evidence is essential to encouraging policies that go beyond the maintenance of forest cover and reinforce the conservation of biodiversity.

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