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INGRID MARIA DENOBILE DA ROCHA

IMPACTO DO RUÍDO ANTRÓPICO NA DIVERSIDADE DE AVES

JOÃO PESSOA
2022

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Tese apresentada ao Programa de Pós-Graduação em Ciências Biológicas (Zoologia), da Universidade Federal da Paraíba, como requisito parcial para a obtenção do título de Doutor em Ciências Biológicas.

Orientador: Gustavo Henrique Calazans Vieira
Segundo Orientador: Carlos Barros de Araújo

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Ata da 150ª Apresentação e Banca de Defesa de Doutorado de Ingrid Maria Denóbile da Rocha

Ao(s) quinze dias do mês de fevereiro de dois mil e vinte e dois, às 09:00 horas, no(a) Ambiente Virtual, da Universidade Federal da Paraíba, reuniram-se, em caráter de solenidade pública, membros da banca examinadora para avaliar a tese de doutorado de **Ingrid Maria Denóbile da Rocha**, candidato(a) ao grau de Doutor(a) em Ciências Biológicas. A banca examinadora foi composta pelos seguintes membros: **Dr. Carlos Barros de Araújo (Orientador - UNILA/PR); Dra. Sandra Maria Hartz (UFRGS/RS); Dr. Ricardo Bomfim Machado (UNB/DF); Dr. Benjamin Timothy Phalan (Parque das Aves - PR); Dr. Luiz Carlos Serramo Lopez (UFPB/PB).** Compareceram à solenidade, além do(a) candidato(a) e membros da banca examinadora, alunos e professores do PPGCB. Dando início à sessão, a coordenação fez a abertura dos trabalhos, apresentando o(a) discente e os membros da banca. Foi passada a palavra ao(à) orientador(a), para que assumisse a posição de presidente da sessão. A partir de então, o(a) presidente, após declarar o objeto da solenidade, concedeu a palavra a **Ingrid Maria Denóbile da Rocha**, para que dissertasse, oral e sucintamente, a respeito de seu trabalho intitulado **"Impacto do ruído antrópico na diversidade de aves".** Passando então a discorrer sobre o aludido tema, dentro do prazo legal, o(a) candidato(a) foi a seguir arguido(a) pelos examinadores na forma regimental. Em seguida, passou a Comissão, em caráter secreto, a proceder à avaliação e julgamento do trabalho, concluindo por atribuir-lhe o conceito **Aprovada**. Perante o resultado proclamado, os documentos da banca foram preparados para trâmites seguintes. Encerrados os trabalhos, nada mais havendo a tratar, eu, orientador(a), como presidente, lavrei a presente ata que, lida e aprovada, assino juntamente com os demais membros da banca examinadora.

João Pessoa, 15/02/2022.

Orientador(a)

Ben Phala
Examinador(a)

Examinador(a)

Examinador(a)

Sandra Maria Hartz
Examinador(a)

Ingrid Maria Denóbile da Rocha
(discente ciente do resultado)

(Em modo de webconferência, as assinaturas digitalizadas são certificadas pelo presidente da banca)

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RESUMO

A constante expansão humana cria condições acústicas sem precedentes, podendo prejudicar a comunicação acústica das espécies através da sobreposição de sinais, em um processo denominado mascaramento acústico. O mascaramento é especialmente prejudicial para as aves, pois dependem da comunicação acústica para realizar suas interações sociais, como por exemplo, cuidado parental, alerta e percepção de predadores, defesa de território e atração de parceiras. Desta forma, o ruído pode interferir com o sucesso reprodutivo e sobrevivência das espécies. Enquanto o ruído antrópico se torna onipresente, ele pode estar mudando a composição de espécies das comunidades naturais, filtrando aquelas que usam as mesmas frequências. Verificamos se as espécies potencialmente mais afetadas pelo ruído possuem características semelhantes, em termos de uso de frequência, massa corporal e tamanho de território. Avaliamos se o ruído de mineração influencia a composição das espécies de acordo com as características espectrais de seus cantos, bem como se afeta a diversidade taxonômica e filogenética das aves da Floresta Nacional de Carajás, Pará. Foram monitorados a diversidade de aves e o ruído em cinco áreas, de 2015 a 2019, sendo três sob a influência do ruído de mineração e duas sem ruído. Cada área possuí cinco transectos de monitoramento de avifauna e cada transecto possui cinco pontos de monitoramento de ruído. Nossos resultados indicam que as espécies maiores são potencialmente mais afetadas, pois usam frequências graves (que são ocupadas pelo ruído antrópico) para comunicação a longa distância, seja por motivos de tamanho de território ou dinâmica espacial de bando. As áreas permeadas pelo ruído são compostas de espécies que utilizam frequências mais agudas, o que pode indicar que o ruído está agindo como um filtro ambiental, selecionando as espécies de acordo com as características espectrais de seus cantos. O ruído teve um efeito negativo sobre a diversidade taxonômica e filogenética, mas positivo na diversidade filogenética média por espécie. A vegetação também influenciou a diversidade, influenciando especialmente a riqueza de espécies. Os resultados mostram que o ruído de mineração pode atuar como uma força seletiva influenciando a diversidade de aves. O ruído é uma fonte invisível de degradação do habitat que pode ter implicações na montagem de comunidades e funcionamento do ecossistema.

Palavras-chave: degradação, poluição sonora, mineração, comunicação acústica, Floresta Amazônica

ABSTRACT

The constant human expansion creates unprecedented acoustic conditions that can impair acoustic communication by overlapping signals. This process, called acoustic masking, is especially harmful to birds that rely on acoustic communication to interact socially, such as parental care, warning and awareness of predators, territory defense, and mate attraction. In this way, noise can interfere with the reproductive success and survival of the species. While anthropogenic noise becomes ubiquitous, it may be changing the species composition of natural communities, filtering out those species that use the same frequencies. We examined whether species most likely affected by noise have similar frequency use in the acoustic spectrum, body mass and territory size. We evaluated whether mining noise influences the species composition according to the spectral features of their songs and if noise influences the taxonomic and phylogenetic diversity of birds in the Carajás National Forest, Pará. Bird diversity and noise were monitored in five areas, from 2015 to 2019, three under the influence of mining noise and two without noise. In each area, samplings were performed in five monitoring transects, which in turn, had five noise monitoring points each. Our results indicate that the potentially most affected species are large, use lower frequencies (occupied by noise), and require long-distance communication, either for reasons of territorial size or spatial flock dynamics. The areas permeated by noise are composed of species that mostly use high-frequency songs, which may indicate that noise acts as an environmental filter, selecting the species according to the spectral features of their songs. The noise negatively affected the taxonomic and phylogenetic diversities but positive on average phylogenetic diversity per species. Vegetation also influenced diversity, especially species richness. The results show that mining noise can be a selective force shaping bird diversity. Noise is an invisible source of habitat degradation that can affect community assembling and ecosystem functioning.

Keywords: degradation, noise pollution, mining, acoustic communication, Amazon Forest

LISTA DE FIGURAS

CAPÍTULO I

CAPÍTULO II

CAPÍTULO III

LISTA DE TABELAS

CAPÍTULO I

CAPÍTULO II

CAPÍTULO III

SUMÁRIO

INTRODUÇÃO GERAL

A revolução Industrial trouxe mudanças extraordinárias e temerosas ao nosso planeta. A população humana disparou de 1.5 para 6 bilhões, e as maneiras pelas quais os humanos afetam o meio ambiente se diversificaram em complexidade, magnitude e frequência (Goudie, 2018). Os efeitos de uma rápida expansão humana têm como resultado um crescimento no consumo de insumos em detrimento de grandes danos ambientais (Myers & Kent, 2003), e para além das mudanças estruturais no ambiente, como desmatamentos e queimadas, as atividades humanas estão permeadas de ruído não-natural. Mesmo que áreas urbanas e industriais sejam as mais ruidosas, o impacto do ruído está se tornando cada vez mais comum em áreas naturais (Ortega, 2012), e os efeitos reportados vão desde leves a graves (Slabbekoorn & Ripmeester, 2008).

A exposição prolongada a níveis elevados de ruído é um dos riscos mais comuns à saúde pública nos países industrializados, podendo resultar em níveis elevados de estresse, distúrbios do sono e danos auditivos, além de uma série de efeitos para a saúde mental (Levak et al., 2008; Pepper et al., 2003). Mudanças no uso do habitat, nos padrões de atividade, aumento de estresse, diminuição da resposta imunológica, redução do sucesso reprodutivo, aumento do risco de predação e até danos na audição, se o ruído for suficientemente alto, são alguns dos efeitos negativos na vida selvagem (Pater et al., 2009). E mesmo que a maior parte dos estudos sobre os efeitos do ruído tenha sido conduzido em habitats terrestres, os ambientes aquáticos também são impactados (Leduc et al., 2021), visto que a poluição sonora se propaga mais rápida na água e atenua menos por unidade de distância da fonte ruidosa (Kight & Swaddle, 2011).

Em aves, a poluição sonora pode interferir em fenômenos biológicos importantes, podendo dificultar o cuidado parental (Leonard & Horn, 2012), a fuga de predadores (Quinn et al., 2006), a defesa territorial (Slabbekoorn & Ripmeester, 2008) e a atração de parceiros

(Brumm, 2004), reduzindo o sucesso reprodutivo (Blickley & Patricelli, 2010). O ruído pode aumentar o estresse enquanto diminui a resposta imunológica das espécies (Blickley et al., 2012; Kight & Swaddle, 2011). Sabe-se que o ruído pode provocar alterações em características do canto das aves (Brumm & Slabbekoorn, 2005), e até mesmo a evasão de espécies de locais ruidosos (Forman, 2000). É provável que a poluição sonora tenha impactos diversos e complexos sobre a vida selvagem, pois pode influenciar vários sistemas biológicos direta e indiretamente (Kight & Swaddle, 2011).

O ruído é um fenômeno global, com potencial de afetar a vida selvagem em todos os continentes e habitats (Blickley & Patricelli, 2010), e ainda assim é um dos impactos antrópicos menos estudados. Existe uma miríade de formas pelas quais a poluição sonora pode afetar o comportamento e aptidão das espécies, bem como influenciar populações e comunidades (Ortega, 2012), sendo o desaparecimento de espécies de habitats outrora ideais é o dano mais alarmante causado pelo ruído. Como consequência, a diminuição na diversidade e alteração da composição de espécies (Francis et al., 2011; McClure et al., 2013), pode ter implicações para o funcionamento do ecossistema

A maioria das fontes de ruído emite energia acústica em frequências graves, mas não o ruído antrópico não se limita em ocupar apenas as frequências graves. Existem ruídos industriais, como o produzido pela extração de gás, que pode atingir até 5kHz (Ortega & Francis, 2012), provavelmente afetando um maior número de espécies. É improvável que ocorra uma diminuição na expansão humana, se tornando cada vez mais importante entender as consequências dos estressores antrópicos na vida selvagem, como o ruído. Os sons e a comunicação sonora sempre fizeram parte do meio ambiente, e o ruído antrópico pode interferir na integridade dos ecossistemas naturais, agindo como uma fonte invisível de degradação do habitat (Ware et al., 2015).

OBJETIVO GERAL

O objetivo desse trabalho é avaliar o impacto do ruído antrópico na diversidade de aves, especialmente o impacto do ruído de mineração na diversidade de aves amazônicas da Floresta Nacional dos Carajás, Pará.

OBJETIVOS ESPECÍFICOS

1. Utilizando relações alométricas entre uso de frequências, tamanho de território e massa corporal, identificar as possíveis espécies mais afetadas pelo ruído antrópico (Capítulo I).
2. A partir do uso do espaço acústico pelas espécies, examinar alterações na composição de espécies entre áreas com ruído natural e áreas permeadas por ruído de mineração (Capítulo II).
3. Por fim, considerando índices acústicos em conjunto com medidas de habitat, avaliar a influência de tais fatores na diversidade taxonômica e filogenética das aves (Capítulo III).

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REFERENCIAL TEÓRICO

A redução na detectabilidade de um sinal pela interferência causada por um ruído é chamada de mascaramento, processo este que reduz a eficiência na transmissão do sinal (Lohr et al., 2003). Mas o que é um sinal e o que é um ruído? Um sinal acústico é formado a partir de uma combinação variada de frequência, intensidade e tempo, e que a partir dessa variação codifica informações sobre a espécie ou sobre o ambiente (Bradbury & Vehrenkamp, 1998). O ruído, por sua vez, é qualquer som indesejável que interfira na transmissão do sinal. Se definirmos o sinal como a vocalização de uma espécie, todos os demais sons presentes naquele ambiente onde o sinal se propaga é um ruído mascarante em potencial, seja ele de origem geológica (ex. vento), biológica (outras espécies, outros indivíduos) ou antrópica (energia acústica produzida pelo homem e seu maquinário).

Considerando que ruído acústico é um componente naturalmente presente no ambiente, em termos de sons geológicos e biológicos, a comunicação acústica evoluiu no sentindo de reduzir o mascaramento, aumentando a eficiência de transmissão do sinal por meio da partição do espaço acústico (Villanueva-rivera, 2014). Dessa forma, os sistemas biológicos podem estar em um delicado equilíbrio, fruto da competição pelo espaço acústico (Torres et al., 2020). No entanto, o ruído de origem antrópica altera os perfis de ruído natural, e ao ocupar largas porções do espaço acústico, compromete a transmissão do sinal das espécies que utilizam a mesma banda de frequência (Brumm, 2013). O ruído antrópico surge como um competidor imbatível, afetando a disponibilidade de frequências e comprometendo a aptidão dos indivíduos que utilizam as mesmas frequências (Francis & R, 2013).

A comunicação acústica é amplamente distribuída na vida selvagem, sendo observadas em invertebrados, como grilos e cigarras, e em vertebrados, como peixes, anfíbios, mamíferos e aves (Bradbury & Vehrenkamp, 1998; EY & FISCHER, 2009; Maccagnan, 2008; Schmidt et al., 2013; Taylor & Popper, 2011; Wells, 1978). Em aves, um sistema de comunicação eficiente é fundamental para diversos aspectos biológicos essenciais à sobrevivência e sucesso reprodutivo, como cuidado parental, alerta de predadores, seleção sexual, defesa e manutenção de territórios (Kumar, 2003).

Os efeitos do ruído antrópico na comunicação acústica das aves podem ser avaliados a partir de três escalas biológicas, respostas de caráter individual, populacional e de comunidades. Estudos em nível individual analisam as respostas comportamentais para reduzir o mascaramento, e tem sido observado mudanças no uso de frequências (Wood & Yezerinac, 2006), na amplitude do sinal (Brumm, 2004), na taxa de emissão de notas (Potvin et al., 2011) e duração do canto (Francis et al., 2011a), como também, evitação de horários ruidosos e evasão de áreas ruidosas (McClure et al., 2014; Bergen & Abs, 1997). Algumas espécies têm grande plasticidade vocal, como em Oscines, enquanto outras não (Brumm, 2013). Entretanto, estudos recentes vêm questionado a metodologia utilizada para avaliar plasticidade em função do ruído, pois além do próprio ruído provocar erros nas análises, o observador pode ser conduzido a erros (Brumm et al., 2017; Zollinger et al., 2012).

Em nível populacional, é possível avaliar como o ruído afeta o espaço ativo de comunicação (Lohr et al., 2003), ou seja, como o ruído afeta as interações entre os indivíduos (Francis et al., 2009). A comunicação acústica é um fenômeno populacional, ela serve como ferramenta para as interações sociais, e qualquer interferência no processo de comunicação pode potencialmente afetar as populações de várias formas. Por exemplo, a redução no alcance dos sinais pode interferir na aptidão e sobrevivência dos indivíduos de uma espécie (ou de várias), já que atividades essenciais que ocorrem em nível populacional são dependentes da comunicação acústica, como a seleção sexual e a defesa de territórios (Kleist et al., 2016; Ortega, 2012). Mesmo assim, são poucos os estudos que abordam os impactos do ruído nesse nível biológico.

Sabe-se que a presença do ruído antrópico pode filtrar grupos específicos de aves de acordo com suas características vocais (Francis et al., 2011b), provocar alterações na composição de espécies (Saha & Padhy, 2011), e reduzir a riqueza e a diversidade taxonômica (Arevalo & Newhard, 2011; Summers et al., 2011). Sabe-se que as características vocais das espécies podem estar relacionadas às suas histórias evolutivas, pois as vocalizações das aves podem conter elementos consistentes com sua filogenia (Price & Lanyon, 2002). Mas pouco se sabe sobre os efeitos do ruído na diversidade filogenética. De forma geral percebe-se que o impacto do ruído na diversidade pode ser examinado por métricas que vão além da redução da riqueza de espécies. A perda da diversidade é o efeito mais alarmante causado pelo ruído antrópico, no sentido de que ajustes individuais (por exemplo, mudanças no comportamento do canto) não são suficientes para prevalecer sobre os impactos do ruído em algumas espécies (Slabbekoorn & Ripmeester, 2008).

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CAPÍTULO I

The allometrics of noise and the sensitivity of Amazonian birds

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The allometrics of noise and the sensitivity of Amazonian birds

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The allometrics of noise and the sensitivity of Amazonian birds

Abstract

Anthropogenic noise can severely affect acoustic communication efficiency, impairing the essential biological activities of birds. Some species can be most prone to be affected by noise, and characteristics such as body size and the need for long-distance communication may be determinants of sensitivity. We evaluated the species potentially affected by noise according to the allometric relationships between body mass, frequency use and territory size, using species recorded in Serra dos Carajás (Pará State) as a community model, and considering four noise source thresholds. The allometric models for the acoustic traits were built using 289 species, including a variety of acoustic codes, ranging from 164.3Hz to 12.5kHz. The allometric models for the territory size were made using 213 species, ranging from 0.25ha to 80ha. Body mass varied from 2.4f to 4.8kg. Body mass was significantly correlated with all acoustic traits and territory size. Our results show that the potentially most affected species are large, use lower frequencies, and require long-distance communication, either for reasons of territorial size or spatial flock dynamics. These species, already threatened by anthropogenic actions such as hunting and deforestation, may also be threatened by noise.

Keywords: threatened species, game birds, body mass, long-distance communication

Introduction

Acoustic communication can be defined as how one mind affects another (Weaver, 1949) and involves at least two individuals, a sender and a receiver. Acoustic signals can provide the sender's location and allow individuals to recognize and interact with each other over great distances (Aubin et al., 2004; Naguib & Haven-Wiley, 2001). This advantage could be essential in forest habitats where visual cues can become unpractical with increasing distances (Aubin et al., 2004; Chappuis, 1971). Anthropogenic noise, however, can severely affect the receiver's capability to detect and discriminate the sender`s signal (Nemeth & Brumm, 2010), and such effect can reduce communication efficiency and affect important biological activities (Leonard & Horn, 2012; Quinn et al., 2006).

Acoustic communication plays an essential role in many aspects of birds' ecology (Brumm & Ritschard, 2011; de Araújo et al., 2011; Janik & Slater, 2000). Individual survival is often related to finding a mating partner and deterring rivals, activities often dependent on efficient acoustic communication (Brumm, 2013; Kumar, 2003; Manica et al., 2014). Acoustic signals can also be used for territory maintenance, ensuring exclusive access to specific sites and resources (Tomaz & Alves, 2009). However, noise can reduce the maximum distance at which the signal can be detected, making the active range of the signal decrease (Nemeth & Brumm, 2010; Ryan & Brenowitz, 1985). Additionally, this problem can be compounded by the size of the territories. While small birds may vary in territory size, large birds require more resources and larger territories, and therefore, are more dependent on long-distance communication (Francis et al., 2011). Therefore, territory size is a trait of interest for comprehension of the impact of noise.

Species that use communication to complex social interactions, such as parrots, might also require long-distance communication (Moura, 2007). Parrots are gregarious species and, while foraging, flocks are constantly splitting and regrouping, forming groups rarely larger than 20 individuals, which at the end of the day assemble into a large roosting flock (Carraca et al., 2007; de Araújo et al., 2011). Given such flock dynamics a regular emission of long-range signals is essential to maintain group cohesion (de Araújo et al., 2017). For example, the Yellow-faced Parrot (*Alipiopsitta xanthops*) emits synchronized alarm calls, a behaviour that increases the signal intensity and allows communication over longer distances of up to 800m (de Araújo et al., 2011). The entire spatial dynamics of parrots, and their capability to split and regroup flocks, seems to depend on efficient long-range acoustic communication (de Araújo et al., 2011; Martins & de Araújo, 2020), which probably makes parrots vulnerable to noise.

Even though some species can change their calls to reduce the harmful effects of noise (Brumm, 2013), morphology can be limiting for significant frequency shifts. Each species seems to have an optimal acoustic frequency relative to its body mass (Fletcher, 2004; Torres et al., 2017; Torres et al., 2020), and the larger the species, the lower the frequencies used. As anthropogenic noise usually occupies the lower portions of the spectra, larger species should have a greater probability of experiencing acoustic masking by anthropogenic noise (Francis et al., 2011). However, although most noise sources emit acoustic energy at lower frequencies, anthropogenic noise is not restricted to masking low frequencies. Industrial noise sources such as gas wells with compressors, for example, can occupy the spectrum up to 5kHz (Ortega & Francis, 2012), presumably affecting even more species. Therefore, it may be practical to predict the species prone to be affected by noise using the allometric relationship between body mass and frequencies, based on the frequency bands of the noise source.

The spread in human activities worldwide has caused a rise in especially low-frequencies noise (Slabbekoorn & Ripmeester, 2008). This disruption in information transmission may be a new selective force shaping bird ecology (Francis et al., 2011). This impact could be especially challenging for large species that emit low-frequency signals and communicate over long distances, whether a territorial species defending a large territory or gregarious species with complex spatial flock dynamics. We evaluated the species potentially affected by noise according to the allometric relationships between body mass, frequency use and territory size, using species recorded in Serra dos Carajás (Pará State) as a community model under different noise thresholds. We expect to find that species most prone to be affected by anthropogenic noise are those with large bodies, which use low frequencies and long-distance communication, either because of their large territories or flock behaviour.

Materials and methods

Data collection

Bird vocalizations

We used published data of 189 Amazon bird species from Serra dos Carajás – Pará state (Torres et al., 2020), and 1600 sound recordings belonging to another 100 Amazon bird species available at the Jacques Vielliard Neotropical Sound Library (https://www2.ib.unicamp.br/fnjv/), Wikiaves (wikiaves.com.br/) and Xenocanto (xeno-canto.org/). To reduce geographic song variation, which could hinder the analysis, we only used recordings made within the Xingu endemism area, where Carajás is located. We used acoustic traits that tend to reflect better the morphological (e.g., body mass) limits of signal emission (Torres et al. 2020): the dominant frequency, which represents the frequency with the highest energy, and the minimum and maximum fundamental frequencies, which represent the minimum and maximum values found within the fundamental harmonic. All audio files were edited by applying a bandstop filter (with an upper limit of 125Hz) and amplified to a peak threshold of 0dB (Torres et al., 2017). Both audio editing and song analysis were performed in Raven Pro software version 1.6.1 (Center for Conservation

Bioacoustics, 2019), using default visualization settings (brightness and contrast at 50%), an FFT window of length 512 (Hann) with 50% overlap between successive windows.

Territory size and body mass

Because there was little available data for the sampled location, we used territory size data for species taken from the Amazon and Atlantic Forest, and Tropical Rainforest from Panamá, Peru and Colombia. The data we used is available in: Chiarani & Fontana (2015); Cintra & Cancelli (2008); Damasceno (2011); Del-Rio (2014); Duca et al. (2006); Fedy & Stutchbury (2004); Freitas & Rodrigues (2012); Gussoni (2014); Johnson et al. (2011); Kattan & Beltran (2002); Lima & Roper (2009); Lopes & Marini (2006); Reinert et al. (2007); Ribon & Marini (2016); Robinson et al. (2000); Stouffer & Vega Rivera (2007); Terborgh et al. (1990); Vieira & Klemann Jr (2013); see details in Supplementary Material I. Instead of analyzing territory area directly, we calculated the range of territorial defence, which is the radius of a circle of area equals the area of the territory. The average body mass of each species was obtained from Dunning (2008), Terborgh et al. (1990) and (Buzzetti et al., 2013).

Allometric models

We examined the allometric relationships between the acoustic traits and territory size of species with their body mass, using type II ranged major axis regression (RMA) after a $\log_{10}$ transformation of the variables. The RMA technique provides the least biased estimate when error variances are unknown (Kiltie, 2000) and allows evaluating how body mass affects the vocal frequency and territory size. We plotted the impact thresholds of the following sources of anthropogenic noise on the frequency allometric models: compressors used in gas wells, which emit noise up to 5kHz (Francis et al., 2011; Ortega & Francis, 2012), noise commonly linked to urban centres, which reaches up to 2kHz (Slabbekoorn & Peet, 2003), traffic noise that generally reaches up to 1kHz (Nemeth & Brumm, 2010), and mining noise, which can be limited to 500Hz (Duarte et al., 2015). We considered the lower portion of the spectra to be entirely occupied. Using our models, we count the species in which the acoustic traits are within the threshold of each noise source. Statistical analyses were made using lmodel2 package (Legendre, 2014) implemented in R software version 4.0.3 (R Core Team, 2020)

Results

The allometric models of acoustic traits included 289 species (Supplementary Material II), with the FFMIN ranging from 164.3 to 8760.3 Hz, FFMAX from 502.5 to 12.5kHz, and FDOM from 328.1 to 9.4kHz. We obtained data on territory size for 213 forest birds, ranging from 0.25ha (28.2m radius) to 80ha (504.8m radius). We successfully gathered body mass data for all these species, ranging from 2.4g to 4.8kg. Body mass was significantly correlated with all acoustic traits and territory size (Table 1).

[Table 1 near here]

Noise sources should affect species according to the frequency profile of noise itself and the frequency used by each species (Table. 2). While strictly low anthropogenic noise may affect fewer species, such as that produced by mining activity, broadband noise produced by compressors can affect 76% to 96% of the species present. Smaller birds use a wide range of frequencies, while most large-bodied birds use lower frequencies (Fig. 1). Smaller birds also have a varied territory size, while species with a body mass greater than 350g will only have large territories, with a range above 23.8ha (~275m radius). Parrots and terrestrial species comprise most of the large species (above 350g) that use lower frequencies and need to defend territories using long-range communication. Half of these species, which

FFMIN is below 500Hz, have their signals at least partially masked by all referred noise sources.

[Table 2 near here]

[Figure 1 near here]

Our models indicate that large Psittaciformes such as macaws and parrots are prone to be affected by noise, as are large Tinamiformes (*Tinamus* and *Crypturelus)*, Gruiformes (*Psophia*) and Galliformes (e.g., *Penelope, Crax, Ortalis motmot, Aburria cujubi*). Due to their size, these species use mostly low frequencies to communicate, overlapping the frequencies of more noise sources. In addition, from the species shown in figure 1, five are assessed as vulnerable: *Anodorhynchus hyacinthinus, Penelope pileata, Crax fasciolata, Aburria cujubi* (IUCN, 2021), and *Psophia interjecta* (Dornas et al., 2017).

Discussion

The noise impact over forest birds was evaluated based on the allometric relationships between acoustic traits and territory size with body mass. Our results indicate that large species are the most affected due to their use of lower frequencies and requirements for long-distance communication in larger territories or spatial flocks with complex dynamics.

Body mass appears to be a predictor of noise sensitivity. First, as most large birds use lower frequencies (Torres et al., 2020), and anthropogenic noise is thought to be concentrated at lower frequencies, large birds should be more prone to suffer the effects of low-frequency noise pollution. Second, large species have larger territories and depend on long-distance communication to defend their territories. Studies have used the term ‘territory’ in various classifications, including the area where mating, nesting, and feeding takes place, the surroundings of a nest, the winter territory, or a roosting territory (Nice 1941).

Nevertheless, all these definitions of territory require defence by at least an individual, which is birds is typically made using acoustic communication. This is especially true in forest environments, where birds are mainly detected acoustically (Vielliard & Silva, 1990).

By definition, a territory is established when the benefits provided are larger than the costs for its maintenance (Brown, 1964). The advantages of defending a territory include food access, an increased chance of finding a mating partner and obtaining a protected nesting site. Territory defence costs include the risk of being injured and the time and energy spent defending it (Tomaz & Alves, 2009). As noise reduces the active communication space (Lohr et al., 2003; Parris et al., 2013), birds would defend a smaller portion of their territory within noisy environments. In such a case, birds might exhibit three responses: 1) changes in song parameters (e.g., increasing call frequency), which might help reduce the adverse effects of noise; 2) increase the energy expenditure to defend their territory, or 3) defend a smaller territory. These alternatives are not mutually exclusive, even though all will likely result in fitness costs.

Psittaciformes are also known to depend on long-distance communication, even though they do not defend territories. Parrots forage in small groups that search the environment for sparse resources (de Araújo et al., 2011; Paranhos et al., 2007). Some species roost within a large roosting flock at night, often including hundreds but sometimes thousands of individuals (Carraca et al., 2007; Moura et al., 2012). Such dynamics are usually thought to improve feeding efficiency during the day while supplying protection during the night (de Araújo et al., 2011; Paranhos et al., 2009) and can depend on long-range communication. Parrots' communication has been recorded at distances of 800m, and flying Macaws responded to calls uttered at over 1km (de Araújo, 2011). Allometric studies indicate selective pressures for optimizing communication range in parrots but not terrestrial birds (Torres et al., 2017). Due to the logarithmic nature of sound propagation, even lesser amounts of noise can drastically reduce the active communication space of a species (Lohr et al., 2003; Parris et al., 2013). Parrots are known to use the limits of their communication range and could suffer the effects of noise at lower intensities.

Despite the shocking number of species affected by high-frequency noise (~5kHz), sound absorption increases with frequency (Chappuis, 1971; Morton 1975), so high-frequency noise should not propagate over long distances (Tyagi et al., 2006). Therefore, high-frequency noise effects are localized and restricted to the areas close to the acoustic source. In this sense, we consider frequencies below 500Hz to exhibit the most significant impact because anthropogenic noise sources usually comprise these frequencies and because low-frequency propagates more efficiently due to lower absorption rates (Chappuis, 1971; Morton 1975).

Our data shows that terrestrial species are most likely to be affected by noise, with four of these known to be threatened by anthropogenic action (Dornas et al., 2017; IUCN, 2021). All terrestrial species tagged in the graphs are territorial, and most defend their territory year-round (Tobias et al., 2016). Territorial species, such as tinamids, are often hunted using their songs as bait. Most species are attracted and killed, especially during the breeding season, when species vocalize for hours to defend their territories (Sick, 2001). Tinamiformes are among the most important groups of Brazilian game birds (Ferreira, 2014). Many species of Galliformes are threatened due to habitat loss and illegal hunting. Galliformes are highly territorial species, and when acoustic communication does not deter the rival, physical disputes can occur (Sick, 2001). The group has one of the first species in Brazil to go extinct, the *Mitu mitu* (IUCN, 2021), and the Cracidae family is considered the most endangered bird group in the Americas (Eisermann et al., 2006). In addition, as mining noise is practically intermittent, with no big breaks during the day, makes this noise source especially challenging for these species, as temporal adjustments (i.e. singing only in silent periods) will not be enough to compensate for noise impact.

It is becoming clear that the worldwide spread of anthropogenic noise has countless adverse effects on wildlife (Slabbekoorn & Ripmeester, 2008). And now, we have discovered that species already threatened by anthropogenic actions, such as hunting and deforestation, may also be threatened by noise. Further, these effects may have significant consequences for ecosystem functioning. The effectiveness of conservation programs will only be achieved if the problem is faced in a transdisciplinary approach (Ferreira, 2014). As noise can decrease the value of once-ideal habitats (Ware et al., 2015), this approach becomes essential for managing protected areas. Although protection from deforestation and hunting practices solves much of the anthropogenic impact over a given forest fragment, species diversity and ecological functions may continue to be under the deleterious effects of a noisy matrix. The reduction of anthropogenic noise levels can certainly benefit protected areas but will almost certainly need financial compensation, as mitigating measures are expensive or counteract economic values.

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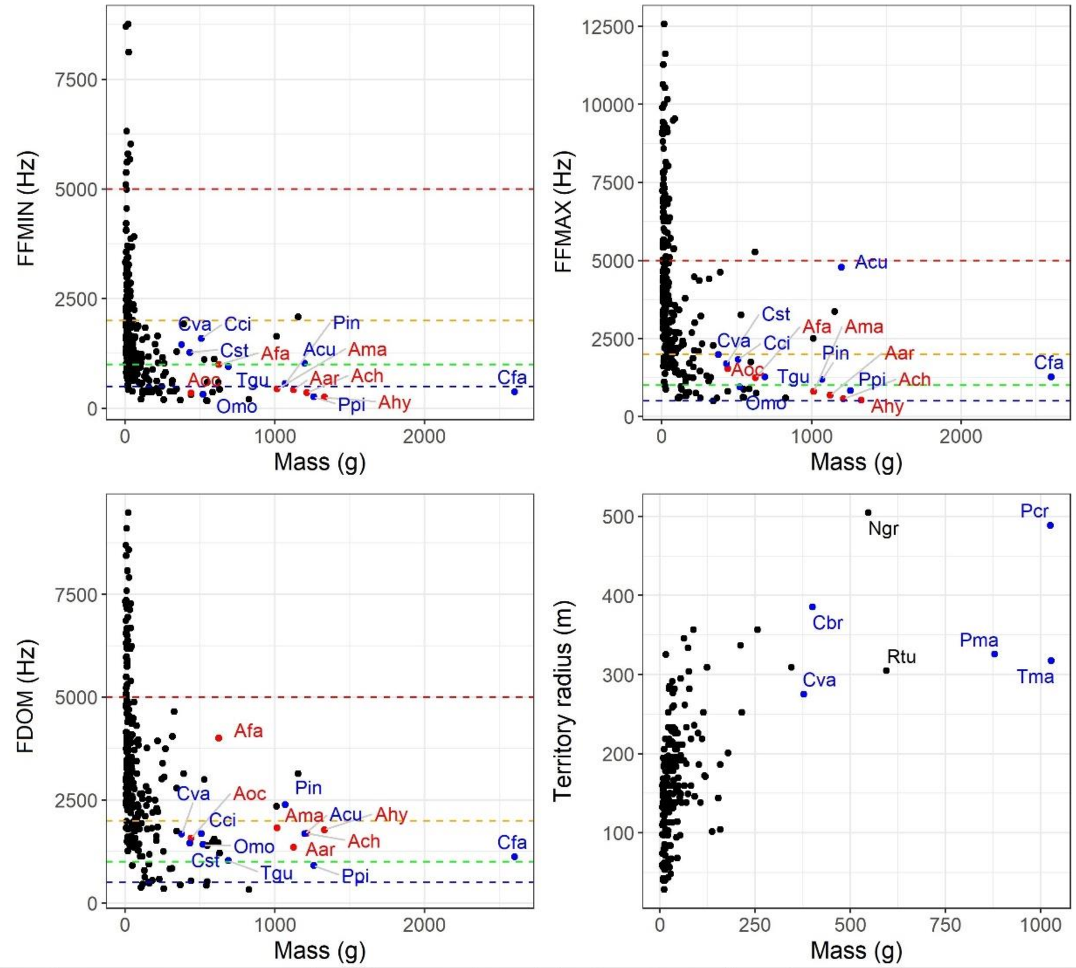
Table 1. The relationship between Log10-transformed parameters and body mass of bird data. For the song parameters (FFMIN, FFMAX and FDOM), species recorded in Carajás were used, and for the territory size, species from tropical forests.

	Intercept	**Slope**	**CI**	**p-perm**	**r2**
FFMIN	9.588	-2.552	-2.96 | -2.23	0.01	0.41
FFMAX	11.031	-2.701	-3.13 | -2.37	0.01	0.41
FDOM	12.955	-3.300	-3.91 | -2.85	0.01	0.35
Territory size	-4.828	2.887	2.39 | 3.59	0.01	0.31

FFMIN, minimum fundamental frequency; FFMAX, maximum fundamental frequency; FDOM, dominant frequency; CI, 95% confidence intervals; p-perm, permutational probability.

Table 2. The number of species that can have each respective acoustic trait masked according to the noise source. FFMIN, minimum fundamental frequency; FFMAX, maximum fundamental frequency; FDOM, dominant frequency.

	Mining	**Traffic**	**Urban centres**	**Gas well compressors**
FFMIN	38 species	92 species	203 species	279 species
FFMAX	-	28 species	79 species	220 species
FDOM	8 species	20 species	80 species	247 species



Aar	*Ara ararauna*	Aoc	*Amazona ochrocephala*	Cva	*Crypturellus variegatus*	Pma	*Penelope marail*
Ach	*Ara chloropterus*	Cbr	*Crypturellus brevirostris*	Ngr	*Nyctibius grandis*	Ppi	*Penelope pileata*
Acu	*Aburria cujubi*	Cci	*Crypturellus cinereus*	Omo	*Ortalis motmot*	Rtu	*Ramphastos tucanus*
Afa	*Amazona farinosa*	Cfa	*Crax fasciolata*	Pcr	*Psophia crepitans*	Tgu	*Tinamus guttatus*
Ahy	*Anodorhynchus hyacinthinus*	Cst	*Crypturellus strigulosus*	Pin	*Psophia interjecta*	Tma	*Tinamus major*
Ama	*Ara macao*						

Fig 1. Allometric models of the acoustic traits and territory size with body mass. FFMIN, minimum fundamental frequency; FFMAX, maximum fundamental frequency; FDOM, dominant frequency. Blue dashed line: mining noise, threshold at 500Hz; Green dashed line: traffic noise, threshold at 1kHz; Yellow dashed line: urban noise, threshold at 2kHz; Red dashed line: gas well compressors, threshold at 5kHz. Species above 350g are tagged; red represents parrots; blue represents terrestrial species, and black other species. Parrots were not considered in the territory size allometric model, as they are not typically territorial species.

Supplementary Material I

Table 1. Data gathered on territory size and their respective references, countries of origin and main habitat type.

Species	Habitat type	Country	Territory size (ha)	Territory radius (m)	Reference
Emberizoides ypiranganus	Atlantic Rainforest	Brazil	1.1	59.2	Chiarani & Fontana, 2015
Hylophylax poecilinotus	Amazon Rainforest	Brazil	8.2	161.6	Cintra & Cancelli, 2008
Merulaxis stresemanni	Atlantic Rainforest	Brazil	3.6	107.1	Damasceno, 2011
Formicivora paludicola	Atlantic Rainforest	Brazil	0.6	41.9	Del-Rio, 2014
Myiothlypis flaveola	Atlantic Rainforest	Brazil	2.0	79.8	Duca & Marini, 2005
Myrmeciza longipes	Tropical Rainforest	Panama	0.7	48.2	Fedy & Stutchbury, 2004
Embernagra longicauda	Atlantic Rainforest	Brazil	2.9	96.6	Freitas & Rodrigues, 2012
Phylloscartes kronei	Atlantic Rainforest	Brazil	0.6	42.6	Gussoni, 2014
Tinamus major	Amazon Rainforest	Brazil	31.7	317.7	Johnson et al., 2011
Psophia crepitans	Amazon Rainforest	Brazil	75.0	488.7	Johnson et al., 2011
Penelope marail	Amazon Rainforest	Brazil	33.3	325.7	Johnson et al., 2011
Ramphastos tucanus	Amazon Rainforest	Brazil	29.2	304.9	Johnson et al., 2011
Crypturellus brevirostris	Amazon Rainforest	Brazil	46.7	385.7	Johnson et al., 2011
Crypturellus variegatus	Amazon Rainforest	Brazil	23.8	275.3	Johnson et al., 2011
Ramphastos vitellinus	Amazon Rainforest	Brazil	30.0	309.1	Johnson et al., 2011
Campephilus rubricollis	Amazon Rainforest	Brazil	35.6	336.7	Johnson et al., 2011
Patagioenas plumbea	Amazon Rainforest	Brazil	12.7	201.1	Johnson et al., 2011
Momotus momota	Amazon Rainforest	Brazil	10.9	186.3	Johnson et al., 2011
Grallaria varia	Amazon Rainforest	Brazil	9.2	171.2	Johnson et al., 2011
Hylexetastes perrotii	Amazon Rainforest	Brazil	20.0	252.4	Johnson et al., 2011
Trogon melanurus	Amazon Rainforest	Brazil	20.0	252.4	Johnson et al., 2011
Piaya melanogaster	Amazon Rainforest	Brazil	10.9	186.3	Johnson et al., 2011
Monasa atra	Amazon Rainforest	Brazil	6.7	146.1	Johnson et al., 2011
Trogon viridis	Amazon Rainforest	Brazil	17.5	236.1	Johnson et al., 2011
Dendrocolaptes picumnus	Amazon Rainforest	Brazil	40.0	356.9	Johnson et al., 2011
Frederickena viridis	Amazon Rainforest	Brazil	29.0	303.9	Johnson et al., 2011
Dendrexetastes rufigula	Amazon Rainforest	Brazil	11.0	187.2	Johnson et al., 2011
Dendrocolaptes certhia	Amazon Rainforest	Brazil	17.1	233.4	Johnson et al., 2011
Celeus undatus	Amazon Rainforest	Brazil	21.5	261.7	Johnson et al., 2011
Jacamerops aureus	Amazon Rainforest	Brazil	37.5	345.6	Johnson et al., 2011
Formicarius analis	Amazon Rainforest	Brazil	11.5	191.4	Johnson et al., 2011
Piculus flavigula	Amazon Rainforest	Brazil	11.5	191.4	Johnson et al., 2011
Turdus albicollis	Amazon Rainforest	Brazil	27.3	294.9	Johnson et al., 2011

Trogon rufus	Amazon Rainforest	Brazil	12.3	197.9	Johnson et al., 2011
Capito niger	Amazon Rainforest	Brazil	12.0	195.5	Johnson et al., 2011
Myrmothera campanisona	Amazon Rainforest	Brazil	8.0	159.6	Johnson et al., 2011
Formicarius colma	Amazon Rainforest	Brazil	7.3	152.5	Johnson et al., 2011
Malacoptila fusca	Amazon Rainforest	Brazil	4.1	114.3	Johnson et al., 2011
Saltator grossus	Amazon Rainforest	Brazil	13.3	205.8	Johnson et al., 2011
Trogon violaceus	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Automolus rubiginosus	Amazon Rainforest	Brazil	13.3	205.8	Johnson et al., 2011
Attila spadiceus	Amazon Rainforest	Brazil	16.0	225.7	Johnson et al., 2011
Dendrocincla fuliginosa	Amazon Rainforest	Brazil	16.8	231.3	Johnson et al., 2011
Cymbilaimus lineatus	Amazon Rainforest	Brazil	11.0	187.2	Johnson et al., 2011
Bucco tamatia	Amazon Rainforest	Brazil	8.7	166.5	Johnson et al., 2011
Veniliornis cassini	Amazon Rainforest	Brazil	6.0	138.2	Johnson et al., 2011
Caryothraustes canadensis	Amazon Rainforest	Brazil	11.4	190.5	Johnson et al., 2011
Lamprospiza melanoleuca	Amazon Rainforest	Brazil	17.1	233.4	Johnson et al., 2011
Campylorhamphus multostriatus	Amazon Rainforest	Brazil	24.0	276.5	Johnson et al., 2011
Automolus infuscatus	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Xiphorhynchus pardalotus	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Cyanoloxia rothschildii	Amazon Rainforest	Brazil	21.0	258.6	Johnson et al., 2011
Sirystes sibilator	Amazon Rainforest	Brazil	26.7	291.6	Johnson et al., 2011
Rhytipterna simplex	Amazon Rainforest	Brazil	14.5	214.9	Johnson et al., 2011
Schiffornis turdina	Amazon Rainforest	Brazil	16.4	228.5	Johnson et al., 2011
Galbula dea	Amazon Rainforest	Brazil	16.8	231.3	Johnson et al., 2011
Philydor erythrocercum	Amazon Rainforest	Brazil	11.4	190.5	Johnson et al., 2011
Percnostola rufifrons	Amazon Rainforest	Brazil	5.6	133.5	Johnson et al., 2011
Myrmeciza ferruginea	Amazon Rainforest	Brazil	7.1	150.4	Johnson et al., 2011
Vireolanius leucotis	Amazon Rainforest	Brazil	11.8	193.9	Johnson et al., 2011
Sclerurus mexicanus	Amazon Rainforest	Brazil	25.5	285.0	Johnson et al., 2011
Deconychura longicauda	Amazon Rainforest	Brazil	15.0	218.6	Johnson et al., 2011
Galbula albirostris	Amazon Rainforest	Brazil	6.3	141.6	Johnson et al., 2011
Sclerurus rufigularis	Amazon Rainforest	Brazil	17.1	233.4	Johnson et al., 2011
Conopias parvus	Amazon Rainforest	Brazil	12.2	197.1	Johnson et al., 2011
Cyphorhinus arada	Amazon Rainforest	Brazil	20.0	252.4	Johnson et al., 2011
Lepidocolaptes albolineatus	Amazon Rainforest	Brazil	12.0	195.5	Johnson et al., 2011
Thamnophilus murinus	Amazon Rainforest	Brazil	5.6	133.5	Johnson et al., 2011
Sittasomus griseicapillus	Amazon Rainforest	Brazil	9.0	169.3	Johnson et al., 2011
Tachyphonus cristatus	Amazon Rainforest	Brazil	8.2	161.6	Johnson et al., 2011
Certhiasomus stictolaemus	Amazon Rainforest	Brazil	12.0	195.5	Johnson et al., 2011
Pachyramphus marginatus	Amazon Rainforest	Brazil	10.0	178.5	Johnson et al., 2011
Willisornis poecilinotus	Amazon Rainforest	Brazil	5.6	133.5	Johnson et al., 2011
Thamnomanes ardesiacus	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Pheugopedius coraya	Amazon Rainforest	Brazil	3.1	99.4	Johnson et al., 2011

Tolmomyias assimilis	Amazon Rainforest	Brazil	9.1	170.2	Johnson et al., 2011
Piprites chloris	Amazon Rainforest	Brazil	10.8	185.5	Johnson et al., 2011
Thamnomanes caesius	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Glyphorhynchus spirurus	Amazon Rainforest	Brazil	5.2	128.7	Johnson et al., 2011
Onychorhynchus coronatus	Amazon Rainforest	Brazil	33.2	325.2	Johnson et al., 2011
Euphonia cayennensis	Amazon Rainforest	Brazil	10.0	178.5	Johnson et al., 2011
Corythopis torquatus	Amazon Rainforest	Brazil	5.7	134.7	Johnson et al., 2011
Cercomacra cinerascens	Amazon Rainforest	Brazil	3.4	104.1	Johnson et al., 2011
Myiopagis gaimardii	Amazon Rainforest	Brazil	5.6	133.5	Johnson et al., 2011
Platyrinchus platyrhynchos	Amazon Rainforest	Brazil	5.0	126.2	Johnson et al., 2011
Hypocnemis cantator	Amazon Rainforest	Brazil	4.0	112.9	Johnson et al., 2011
Hylophilus muscicapinus	Amazon Rainforest	Brazil	11.2	188.9	Johnson et al., 2011
Hylophilus ochraceiceps	Amazon Rainforest	Brazil	8.6	165.5	Johnson et al., 2011
Tolmomyias poliocephalus	Amazon Rainforest	Brazil	6.0	138.2	Johnson et al., 2011
Hemitriccus josephinae	Amazon Rainforest	Brazil	4.6	121.0	Johnson et al., 2011
Xenops minutus	Amazon Rainforest	Brazil	13.3	205.8	Johnson et al., 2011
Conopophaga aurita	Amazon Rainforest	Brazil	6.3	141.6	Johnson et al., 2011
Myiopagis caniceps	Amazon Rainforest	Brazil	6.0	138.2	Johnson et al., 2011
Microbates collaris	Amazon Rainforest	Brazil	4.0	112.9	Johnson et al., 2011
Herpsilochmus dorsimaculatus	Amazon Rainforest	Brazil	8.0	159.6	Johnson et al., 2011
Ramphocaenus melanurus	Amazon Rainforest	Brazil	10.7	184.6	Johnson et al., 2011
Myrmotherula longipennis	Amazon Rainforest	Brazil	11.9	194.7	Johnson et al., 2011
Platyrinchus coronatus	Amazon Rainforest	Brazil	3.8	110.0	Johnson et al., 2011
Picumnus exilis	Amazon Rainforest	Brazil	4.7	122.3	Johnson et al., 2011
Epinecrophylla gutturalis	Amazon Rainforest	Brazil	10.6	183.7	Johnson et al., 2011
Hemitriccus zosterops	Amazon Rainforest	Brazil	3.9	111.4	Johnson et al., 2011
Myrmotherula menetriesii	Amazon Rainforest	Brazil	11.9	194.7	Johnson et al., 2011
Phylloscartes virescens	Amazon Rainforest	Brazil	7.5	154.5	Johnson et al., 2011
Myrmotherula axillaris	Amazon Rainforest	Brazil	8.9	168.4	Johnson et al., 2011
Coereba flaveola	Amazon Rainforest	Brazil	8.3	162.6	Johnson et al., 2011
Zimmerius acer	Amazon Rainforest	Brazil	8.2	161.6	Johnson et al., 2011
Terenotriccus erythrurus	Amazon Rainforest	Brazil	4.7	122.3	Johnson et al., 2011
Ornithion inerme	Amazon Rainforest	Brazil	6.7	146.1	Johnson et al., 2011
Tyrannulus elatus	Amazon Rainforest	Brazil	5.7	134.7	Johnson et al., 2011
Lophotriccus vitiosus	Amazon Rainforest	Brazil	3.8	110.0	Johnson et al., 2011
Terenura spodioptila	Amazon Rainforest	Brazil	8.0	159.6	Johnson et al., 2011
Myrmotherula brachyura	Amazon Rainforest	Brazil	5.5	132.3	Johnson et al., 2011
Grallaria squamigera	Tropical Forest and alder plantations	Colombia	3.3	101.7	Kattan & Beltran, 2002
Grallaria nuchalis	Tropical Forest and alder plantations	Colombia	9.3	172.1	Kattan & Beltran, 2002
Grallaria milleri	Tropical Forest and alder plantations	Colombia	3.0	96.9	Kattan & Beltran, 2002

Grallaria rufocinerea	Tropical Forest and alder plantations	Colombia	1.5	68.0	Kattan & Beltran, 2002
Conopophaga melanops	Atlantic Rainforest	Brazil	2.9	96.8	Lima & Roper, 2009
Suiriri islerorum	Cerrado	Brazil	14.0	211.2	Lopes & Marini, 2006
Suiriri suiriri affinis	Cerrado	Brazil	11.2	188.9	Lopes & Marini, 2006
Stymphalornis acutirostris	Cerrado	Brazil	0.3	28.2	Reinert et al., 2007
Baryphthengus ruficapillus	Atlantic Rainforest	Brazil	3.4	104.1	Ribon & Marini, 2016
Mackenziaena severa	Atlantic Rainforest	Brazil	6.6	145.0	Ribon & Marini, 2016
Attila rufus	Atlantic Rainforest	Brazil	7.5	154.5	Ribon & Marini, 2016
Campylorhamphus falcularius	Atlantic Rainforest	Brazil	2.8	94.4	Ribon & Marini, 2016
Automolus leucophthalmus	Atlantic Rainforest	Brazil	5.4	131.1	Ribon & Marini, 2016
Habia rubica	Atlantic Rainforest	Brazil	4.0	112.9	Ribon & Marini, 2016
Pyriglena leucoptera	Atlantic Rainforest	Brazil	1.4	66.8	Ribon & Marini, 2016
Lepidocolaptes squamatus	Atlantic Rainforest	Brazil	2.2	83.7	Ribon & Marini, 2016
Lochmias nematura	Atlantic Rainforest	Brazil	2.2	83.7	Ribon & Marini, 2016
Xiphorhynchus fuscus	Atlantic Rainforest	Brazil	3.3	102.5	Ribon & Marini, 2016
Conopophaga lineata	Atlantic Rainforest	Brazil	0.6	43.7	Ribon & Marini, 2016
Thamnophilus caerulescens	Atlantic Rainforest	Brazil	1.3	64.3	Ribon & Marini, 2016
Corythopis delalandi	Atlantic Rainforest	Brazil	1.0	56.4	Ribon & Marini, 2016
Tolmomyias sulphurescens	Atlantic Rainforest	Brazil	1.6	71.4	Ribon & Marini, 2016
Synallaxis ruficapila	Atlantic Rainforest	Brazil	1.6	71.4	Ribon & Marini, 2016
Synallaxis cinerascens	Atlantic Rainforest	Brazil	2.1	81.8	Ribon & Marini, 2016
Dysithamnus mentalis	Atlantic Rainforest	Brazil	0.9	53.5	Ribon & Marini, 2016
Myiopagis viridicata	Atlantic Rainforest	Brazil	0.5	39.9	Ribon & Marini, 2016
Lathrotriccus euleri	Atlantic Rainforest	Brazil	0.9	53.5	Ribon & Marini, 2016
Leptopogon amaurocephalus	Atlantic Rainforest	Brazil	2.3	85.6	Ribon & Marini, 2016
Drymophila ferruginea	Atlantic Rainforest	Brazil	1.2	61.8	Ribon & Marini, 2016
Xenops rutilans	Atlantic Rainforest	Brazil	6.0	138.2	Ribon & Marini, 2016
Platyrinchus mystaceus	Atlantic Rainforest	Brazil	2.0	79.8	Ribon & Marini, 2016
Hemitriccus nidipendulus	Atlantic Rainforest	Brazil	0.5	39.9	Ribon & Marini, 2016
Todirostrum plumbeiceps	Atlantic Rainforest	Brazil	1.7	73.6	Ribon & Marini, 2016
Myiornis auricularis	Atlantic Rainforest	Brazil	1.1	59.2	Ribon & Marini, 2016
Anabazenops fuscus	Atlantic Rainforest	Brazil	3.3	102.5	Ribon & Marini, 2016
Todirostrum poliocephalum	Atlantic Rainforest	Brazil	1.5	69.1	Ribon & Marini, 2016
Formicivora serrana	Atlantic Rainforest	Brazil	1.0	56.4	Ribon & Marini, 2016
Campephilus melanoleucos	Tropical Rainforest	Panama	40.0	356.9	Robinson et al., 2000
Baryphthengus martii	Tropical Rainforest	Panama	6.5	143.9	Robinson et al., 2000
Otus guatemalae	Tropical Rainforest	Panama	15.0	218.6	Robinson et al., 2000
Querula purpurata	Tropical Rainforest	Panama	16.0	225.7	Robinson et al., 2000
Celeus loricatus	Tropical Rainforest	Panama	25.0	282.2	Robinson et al., 2000
Melanerpes pucherani	Tropical Rainforest	Panama	8.0	159.6	Robinson et al., 2000
Xiphorhynchus lachrymosus	Tropical Rainforest	Panama	6.0	138.2	Robinson et al., 2000
Automolus ochrolaemus	Tropical Rainforest	Panama	11.0	187.2	Robinson et al., 2000

Habia fuscicauda	Tropical Rainforest	Panama	25.0	282.2	Robinson et al., 2000
Sclerurus guatemalensis	Tropical Rainforest	Panama	7.0	149.3	Robinson et al., 2000
Cyphorhinus phaeocephalus	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Vireolanius pulchellus	Tropical Rainforest	Panama	5.0	126.2	Robinson et al., 2000
Gymnopithys leucaspis	Tropical Rainforest	Panama	25.0	282.2	Robinson et al., 2000
Thamnophilus atrinucha	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Hylophylax naevioides	Tropical Rainforest	Panama	4.0	112.9	Robinson et al., 2000
Cercomacra tyrannina	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Euphonia fulvicrissa	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Myrmotherula fulviventris	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Microrhopias quixensis	Tropical Rainforest	Panama	1.0	56.4	Robinson et al., 2000
Myiornis atricapillus	Tropical Rainforest	Panama	3.0	97.7	Robinson et al., 2000
Myrmornis torquata	Amazon Rainforest	Brazil	16.0	225.9	Stouffer, 2007
Hylopezus macularius	Amazon Rainforest	Brazil	11.7	193.0	Stouffer, 2007
Sclerurus caudacutus	Amazon Rainforest	Brazil	21.3	260.5	Stouffer, 2007
Nyctibius grandis	Amazon Rainforest	Peru	80.0	504.8	Terborgh et al., 1990
Pteroglossus beauharnaesii	Amazon Rainforest	Peru	20.0	252.4	Terborgh et al., 1990
Chamaeza nobilis	Amazon Rainforest	Peru	30.0	309.1	Terborgh et al., 1990
Monasa morphoeus	Amazon Rainforest	Peru	6.0	138.2	Terborgh et al., 1990
Liosceles thoracicus	Amazon Rainforest	Peru	15.0	218.6	Terborgh et al., 1990
Monasa nigrifrons	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Celeus grammicus	Amazon Rainforest	Peru	35.0	333.9	Terborgh et al., 1990
Electron platyrhynchum	Amazon Rainforest	Peru	7.0	149.3	Terborgh et al., 1990
Turdus hauxwelli	Amazon Rainforest	Peru	6.0	138.2	Terborgh et al., 1990
Trogon collaris	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Melanerpes cruentatus	Amazon Rainforest	Peru	14.0	211.2	Terborgh et al., 1990
Trogon curucui	Amazon Rainforest	Peru	7.0	149.3	Terborgh et al., 1990
Glaucidium minutissimum	Amazon Rainforest	Peru	14.0	211.2	Terborgh et al., 1990
Nystalus striolatus	Amazon Rainforest	Peru	9.0	169.3	Terborgh et al., 1990
Phlegopsis nigromaculata	Amazon Rainforest	Peru	14.0	211.2	Terborgh et al., 1990
Myrmeciza goeldii	Amazon Rainforest	Peru	16.0	225.7	Terborgh et al., 1990
Myrmeciza hyperythra	Amazon Rainforest	Peru	6.0	138.2	Terborgh et al., 1990
Attila bolivianus	Amazon Rainforest	Peru	12.0	195.5	Terborgh et al., 1990
Pachyramphus minor	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Eubucco richardsoni	Amazon Rainforest	Peru	10.0	178.5	Terborgh et al., 1990
Campylorhynchus turdinus	Amazon Rainforest	Peru	25.0	282.2	Terborgh et al., 1990
Xiphorhynchus ocellatus	Amazon Rainforest	Peru	14.0	211.2	Terborgh et al., 1990
Xiphorhvnchus spixii	Amazon Rainforest	Peru	14.0	211.2	Terborgh et al., 1990
Philydor ruficaudatum	Amazon Rainforest	Peru	12.0	195.5	Terborgh et al., 1990
Icterus cayanensis	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Hyloctistes subulatus	Amazon Rainforest	Peru	12.0	195.5	Terborgh et al., 1990
Percnostola lophotes	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Thamnophilus aethiops	Amazon Rainforest	Peru	10.0	178.5	Terborgh et al., 1990

Galbula cyanescens	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Pygiptila stellaris	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Legatus leucophaius	Amazon Rainforest	Peru	7.0	149.3	Terborgh et al., 1990
Thamnophilus schistaceus	Amazon Rainforest	Peru	6.0	138.2	Terborgh et al., 1990
Myrmoborus myotherinus	Amazon Rainforest	Peru	4.0	112.9	Terborgh et al., 1990
Ramphotrigon ruficauda	Amazon Rainforest	Peru	8.0	159.6	Terborgh et al., 1990
Dichrozona cincta	Amazon Rainforest	Peru	7.0	149.3	Terborgh et al., 1990
Myrmeciza hemimelaena	Amazon Rainforest	Peru	4.0	112.9	Terborgh et al., 1990
Euphonia rufiventris	Amazon Rainforest	Peru	5.0	126.2	Terborgh et al., 1990
Hylophylax naevius	Amazon Rainforest	Peru	15.0	218.6	Terborgh et al., 1990
Euphonia chrysopasta	Amazon Rainforest	Peru	5.0	126.2	Terborgh et al., 1990
Myrmotherula hauxwelli	Amazon Rainforest	Peru	4.0	112.9	Terborgh et al., 1990
Myrmotherula iheringi	Amazon Rainforest	Peru	4.0	112.9	Terborgh et al., 1990
Terenura humeralis	Amazon Rainforest	Peru	6.0	138.2	Terborgh et al., 1990
Todirostrum chrysocrotaphum	Amazon Rainforest	Peru	7.0	149.3	Terborgh et al., 1990
Myiornis ecaucatus	Amazon Rainforest	Peru	3.0	97.7	Terborgh et al., 1990
Scytalopus iraiensis	Atlantic Rainforest	Brazil	0.5	39.9	Vieira & Klemann Jr, 2013

Supplementary Material II

Table 1. Species and their respective mean acoustic traits. FFMIN, minimum fundamental frequency; FFMAX, maximum fundamental frequency; FDOM, dominant frequency.

Species	**FFMIN (Hz)**	**FFMAX (Hz)**	**FDOM (Hz)**
Tinamus tao	1031.6	1573.7	1288.8
Tinamus guttatus	946	1268	1031
Crypturellus cinereus	1592.1	1830.2	1683.9
Crypturellus soui	1648.6	2125.6	1845.5
Crypturellus strigulosus	1271.3	1697.6	1453.3
Crypturellus variegatus	1453.3	1984	1671.3
Crypturellus tataupa	1036.1	3004.1	2214.7
Penelope pileata	259	840.7	913
Aburria cujubi	1021	4787	1687
Ortalis motmot	317	951	1421
Crax fasciolata	375	1270	1125
Odontophorus gujanensis	560.3	1135.7	840.5
Patagioenas speciosa	190.2	597.4	354.7
Patagioenas cayennensis	350	750	580.5
Patagioenas subvinacea	630.7	1205.4	1182
Geotrygon montana	393.8	600	445.3
Leptotila verreauxi	361.4	660.4	484.2
Leptotila rufaxilla	396.6	660.8	515.6
Dromococcyx pavoninus	1955.3	2674.9	2312.5
Piaya cayana	931.3	3432	2648
Nyctibius grandis	164.3	606.5	516.8
Nyctibius aethereus	312.5	809.5	538.9
Nyctibius griseus	474	1102	947
Nyctiphrynus ocellatus	821.2	2036.1	1623.5
Antrostomus rufus	1028	2046.5	1439
Antrostomus sericocaudatus	1240.7	2153.7	1962
Lurocalis semitorquatus	1276	1912.7	1503.3
Nyctidromus nigrescens	775.2	2101.7	1601.2
Nyctidromus albicollis	627.1	2741.9	2331.6
Hydropsalis torquata	3913.6	5704.6	4875
Chaetura spinicaudus	3215	7862	5770
Chaetura brachyura	2599	7329	6933
Glaucis hirsutus	3560	9246	8437
Phaethornis ruber	3328	9121.3	7317.9
Phaethornis superciliosus	5373.5	9429.8	7229.8
Campylopterus largipennis	2309	9894	5490
Chionomesa fimbriata	1860	7229	6546
Hylocharis sapphirina	8704.3	9896	8687

Chlorestes cyanus	5100	9056.7	6858.7
Psophia interjecta	566.1	1191.2	2389.5
Vanellus chilensis	382	1253	4651
Elanoides forficatus	1921	4621	3143
Spizaetus tyrannus	1640.5	2507	2346.5
Urubitinga urubitinga	2083	3369	3140
Rupornis magnirostris	820.3	2328.7	3746
Buteo nitidus	1113	3254	3000
Megascops choliba	406.5	870	632.7
Megascops usta	734	964.5	796.8
Lophostrix cristata	172.6	624.4	430.7
Pulsatrix perspicillata	204	593.4	328.1
Strix huhula	183	600	445
Glaucidium hardyi	1020.9	1644	1351.5
Trogon melanurus	573.8	1349	1130.7
Trogon viridis	592.7	1681.7	1273.9
Trogon ramonianus	775.6	1632.5	1431
Trogon rufus	843.8	1625.5	1462.5
Momotus momota	207	589.5	380.8
Chloroceryle inda	1464	5102	4435
Galbula cyanicollis	891.6	4072.3	2849
Galbula dea	1140.6	2292	3083.1
Jacamerops aureus	2239.1	3098.1	2767.5
Monasa morphoeus	1031.6	2694.2	2202.6
Nonnula ruficapilla	1292.2	3046.1	2606
Malacoptila rufa	3684.7	4911.6	4459.8
Notharchus tectus	2048	5131	4546
Notharchus hyperrhynchus	1184.9	2201.4	1796.6
Bucco capensis	869.7	1721.8	1483.9
Nystalus torridus	1368.5	2164.8	1908.2
Ramphastos tucanus	1121.9	1753.3	1553.8
Ramphastos vitellinus	1287.2	2276.9	1746
Selenidera gouldii	370.8	785.8	552.8
Pteroglossus inscriptus	672.5	1336	1792.5
Pteroglossus aracari	1029	3215	3057
Pteroglossus bitorquatus	859	1421.5	2368
Picumnus aurifrons	6312.7	7834.7	7577.7
Melanerpes cruentatus	921.5	2455.4	2812.1
Veniliornis affinis	1467.3	1667.3	3140
Campephilus rubricollis	622.7	1593.7	2300.3
Campephilus melanoleucos	742.5	1457.5	1256.5
Dryocopus lineatus	883	1707	2203
Celeus torquatus	1837.8	2276.4	2109.1
Celeus undatus	699	1974.3	1745.5
Celeus flavus	1463	2685	2497

Piculus leucolaemus	1621.3	2388.5	2062.3
Piculus flavigula	1355	1692	1525
Piculus laemostictus	1877.3	2486.9	2335.9
Herpetotheres cachinnans	369	886.5	1485.5
Micrastur ruficollis	425.1	1255.7	1815.7
Micrastur mintoni	448	1242.8	1628.7
Micrastur mirandollei	594.8	877.3	1396.8
Micrastur semitorquatus	427.5	756	1218
Ibycter americanus	593.4	5269.7	1468.6
Milvago chimachima	624.5	4410	4047.5
Falco rufigularis	1236	2189	1937
Brotogeris chrysoptera	2368.5	6374.4	4484.2
Pyrilia vulturina	815.3	2429.3	2835.5
Pionus menstruus	1156.4	4363	3390.2
Amazona ochrocephala	346	1537.3	1577.6
Amazona farinosa	993	1242	4005
Pionites leucogaster	1177	3786	2718
Deroptyus accipitrinus	500	2094.3	3016.4
Pyrrhura anerythra	1005.5	1761	3444.5
Pyrrhura amazonum	1140.7	2358.5	3709.7
Anodorhynchus hyacinthinus	250.2	530.7	1778.9
Aratinga jandaya	1000.7	1420	3835.8
Ara ararauna	424.5	689	1359
Ara severus	444	502.5	2788.6
Ara macao	440.6	803.8	1828
Ara chloropterus	355.9	572.8	1696.5
Psittacara leucophthalmus	361.8	762.5	2518.5
Myrmornis torquata	1616.2	4286.5	3227.5
Pygiptila stellaris	2100.7	2909.7	2624.3
Epinecrophylla leucophthalma	4553.1	7445.3	6283.9
Epinecrophylla ornata	3072.2	7425.4	6664.2
Myrmotherula brachyura	1390	3811.4	2649.3
Myrmotherula axillaris	1228.1	3678.1	2472.3
Myrmotherula longipennis	2153.3	4939.7	3462.9
Myrmotherula menetriesii	1658	4673	3019.3
Formicivora grisea	721.3	1613.9	1992
Isleria hauxwelli	2463.8	5660.8	4101.1
Thamnomanes caesius	1102.2	5427	2865.3
Dichrozona cincta	2827.5	3773.5	3363
Herpsilochmus rufimarginatus	883.1	2532.2	2345.2
Thamnophilus schistaceus	498.7	987.4	1606.6
Thamnophilus stictocephalus	421.6	1379.5	1507.1
Thamnophilus aethiops	428.8	868.7	1436.6
Thamnophilus amazonicus	499.5	1691	1265
Cymbilaimus lineatus	1124	1588.3	1380.4

Taraba major	717.3	1784.3	1356
Hylophylax naevius	2027.7	6019.3	4098.2
Hylophylax punctulatus	1633.5	4760.1	4045.9
Sclateria naevia	1826.5	4541.5	3552.5
Myrmelastes rufifacies	2412.3	4468.2	4853.2
Myrmoborus myotherinus	1987.1	4047.1	3683.6
Myrmoborus leucophrys	1204.9	3266.6	2939.1
Pyriglena leuconota	1460.5	2772.3	2512
Cercomacra cinerascens	524.6	1117.1	2262.8
Cercomacroides nigrescens	1053.6	2594.9	2326.7
Hypocnemis striata	1134.5	3401.6	2549.6
Willisornis vidua	2225.4	4646	3475.6
Phlegopsis nigromaculata	1896.7	3876.6	3539.4
Conopophaga melanogaster	809.1	2209.9	4355.9
Conopophaga aurita	1457	4176.9	3472
Grallaria varia	329.9	639.3	462.7
Myrmothera berlepschi	383.6	1897.4	917.3
Formicarius colma	2001.7	3744.5	2598.7
Formicarius analis	1586.8	2275	1894
Sclerurus macconnelli	3266.1	5577.6	4010.7
Sclerurus rufigularis	2707.7	5534	4108.3
Sclerurus caudacutus	2856.5	5564	3375
Certhiasomus stictolaemus	1492.3	4172.7	3395.7
Sittasomus griseicapillus	1365	3396	3031
Deconychura longicauda	1806.5	3038.5	2273
Dendrocincla fuliginosa	838	1986	3375
Glyphorynchus spirurus	1925.7	4405.9	4807.4
Dendrexetastes rufigula	1252.5	3065.9	2371.5
Dendrocolaptes retentus	1048.7	3565.7	2174.9
Dendrocolaptes picumnus	918.3	2699.5	2142.8
Hylexetastes uniformis	1148.5	2682.3	2660.8
Xiphocolaptes carajaensis	1559.4	3423.9	2531.3
Xiphorhynchus spixii	1328.7	5594.4	3489.9
Xiphorhynchus guttatoides	1098.6	2255.1	1850.5
Dendroplex picus	835.5	3587	2192.8
Campylorhamphus multostriatus	1306	2847.1	2435.6
Lepidocolaptes layardi	2271.9	4605.5	3433
Xenops minutus	2525.7	6697.3	5987
Xenops rutilans	2787	6856	6187
Philydor erythrocercum	3053.5	8141.5	7905
Anabacerthia ruficaudata	1074	2943	1808
Automolus rufipileatus	717.1	1561.1	2335.7
Automolus paraensis	866	2512.4	3102.4
Synallaxis scutata	1875.3	4441.9	3464.4
Synallaxis rutilans	652.8	1691.2	2766.1

Synallaxis cherriei	944	1913.4	2973.8
Tyranneutes stolzmanni	2251.9	4145.6	2802.9
Chiroxiphia pareola	1362.6	4051.1	2489.1
Lepidothrix iris	1304.7	3303.1	2430.4
Pipra fasciicauda	1380	3003.8	3210.5
Machaeropterus pyrocephalus	3173.3	3920.3	3374.7
Ceratopipra rubrocapilla	3135.4	6247.8	4084.5
Querula purpurata	543.4	1466.1	1133.6
Lipaugus vociferans	782.8	5382.5	2388.2
Procnias albus	1285	1871.2	1519.3
Xipholena lamellipennis	786.4	1797.8	1574.2
Schiffornis turdina	2837	4058.3	3011.5
Laniocera hypopyrra	2002	3020	2554
Iodopleura isabellae	3441.7	4842	4515
Tityra semifasciata	797.4	5367.3	3140.3
Pachyramphus viridis	1543.8	3491.2	3778.6
Pachyramphus castaneus	2338	3873.2	3850.2
Pachyramphus marginatus	2252.3	3041.7	2655.7
Pachyramphus minor	3662	5886	3890
Oxyruncus cristatus	1844.9	5484.9	6234.3
Onychorhynchus coronatus	701.1	3083.3	2845.7
Terenotriccus erythrurus	4054.2	6972.3	5077.7
Piprites chloris	1299.7	1865.3	1602.1
Platyrinchus saturatus	1792	3817	3679
Platyrinchus platyrhynchos	2675.7	3646.1	3140.3
Mionectes oleagineus	912	3075	2885
Mionectes macconnelli	1045.4	3081.7	5835.6
Leptopogon amaurocephalus	1180	1771	1499.5
Corythopis torquatus	1622	4342	3724.5
Rhynchocyclus olivaceus	1414.8	2817.5	2718.8
Tolmomyias sulphurescens	2580.3	6954.5	5894.3
Tolmomyias assimilis	2504	6450.8	4398.1
Tolmomyias poliocephalus	1831.7	3963.3	3808.1
Tolmomyias flaviventris	5607.6	7018.3	5739
Todirostrum chrysocrotaphum	4057.6	6367.3	5962.4
Poecilotriccus capitalis	1018.4	2808.2	4816.7
Poecilotriccus sylvia	815.9	2557.9	1827.8
Myiornis ecaudatus	4216.9	5529.6	4754.1
Hemitriccus minor	1391.1	2994	5220.3
Hemitriccus griseipectus	1612.8	3087.1	4143.7
Hemitriccus margaritaceiventer	1558.6	4325.7	2964
Hemitriccus minimus	1988.5	2993.5	2390.6
Zimmerius acer	2193	4040.5	3037.3
Ornithion inerme	3065.3	5421.1	5016.6
Camptostoma obsoletum	1960.5	5023	3749.5

Myiopagis gaimardii	2978.6	7620.3	4734.3
Myiopagis caniceps	1715.4	6557.2	4593.4
Tyrannulus elatus	1925.7	2841.7	2631
Phaeomyias murina	898.3	4366.4	3992.8
Phyllomyias fasciatus	1245	3411	2437
Attila spadiceus	1549.2	2669.9	2338.8
Legatus leucophaius	2841	5934.6	3953.4
Myiarchus tuberculifer	1634.6	3737.8	4088.3
Myiarchus ferox	2095.3	3422.7	3128.7
Rhytipterna simplex	1205.5	2754.6	2359
Pitangus sulphuratus	848.6	2110.2	2925.1
Myiodynastes maculatus	978	4169	3000
Megarynchus pitangua	882.5	1576.4	3912.8
Myiozetetes cayanensis	2295.3	3481.7	3126.7
Myiozetetes luteiventris	921.1	2192.5	4054.5
Tyrannus melancholicus	3867	6813	5390
Sublegatus obscurior	2028	4761	3229
Colonia colonus	1435.6	4699.8	3250
Cnemotriccus fuscatus	1712.3	4317	3389.7
Lathrotriccus euleri	2239.1	4490.9	3891.3
Contopus nigrescens	2050.2	4623.6	4038.4
Cyclarhis gujanensis	1512.2	3580.7	2498.2
Vireolanius leucotis	1689.4	2758.9	2311.3
Hylophilus semicinereus	2512.4	3653.4	3295.4
Tunchiornis ochraceiceps	2823.6	3185.2	2951.1
Pachysylvia hypoxantha	2398.3	4677.5	3784.7
Vireo chivi	2242.5	4727.3	3492.9
Cyanocorax cyanopogon	1619	2323.3	3763.7
Progne chalybea	1219	8031	3187
Microcerculus marginatus	1638	3905.8	2355.1
Campylorhynchus turdinus	614.7	3558.5	1593.7
Pheugopedius coraya	817.4	2471.4	1796.4
Ramphocaenus melanurus	2134.6	3221.4	2979.9
Turdus leucomelas	1629	3324.5	2067
Turdus fumigatus	1887.5	3244	2086
Euphonia chlorotica	3287	4024	3875
Euphonia chrysopasta	2176	4339	3962
Euphonia rufiventris	2565	5666	4134
Arremon taciturnus	8115.1	11623.7	8579.4
Zonotrichia capensis	2087.4	7113.4	4078
Psarocolius decumanus	430	4474	3937
Psarocolius viridis	383.3	1312.7	829.3
Cacicus cela	1002	9545	3962
Icterus cayanensis	1030	6249	3314
Setophaga pitiayumi	3363.4	8806.9	6310.3

Myiothlypis flaveola	2303.3	6766.9	4906.8
Myiothlypis mesoleuca	1822	4379.4	3249.5
Basileuterus culicivorus	3252.9	9067.2	5287.4
Lamprospiza melanoleuca	2664.3	4667.9	4012.3
Granatellus pelzelni	2823.1	4480.9	3675.3
Cyanoloxia rothschildii	2622.7	4423	3290
Parkerthraustes humeralis	6025	9121.5	7265
Hemithraupis guira	3411.2	8578.9	6498.1
Cyanerpes caeruleus	1252.8	7338.1	7149
Dacnis lineata	4987	11272	9093
Saltator maximus	1641.8	6964	4081.1
Saltator coerulescens	1089.1	3617.5	1968.8
Saltator grossus	1492.5	3066.8	2447.5
Coereba flaveola	2473.8	10641.9	7354
Loriotus luctuosus	5678.5	8018	7123
Loriotus cristatus	5801	7310	6373
Lanio versicolor	2998.1	9383	5310.6
Ramphocelus carbo	1841.5	3791	4694
Conirostrum speciosum	3102.9	7794.5	6591.7
Cissopis leverianus	1813.3	9483	4312
Ixothraupis punctata	2714.3	12574.7	8073.8
Thraupis episcopus	2549.8	9238.3	6172.2
Thraupis palmarum	1497.8	10167.9	6679.4
Stilpnia nigrocincta	3704.5	10004.5	8062
Tangara gyrola	1917.5	3109	7191.5
Tangara mexicana	8760.3	10526.9	9479.7

CAPÍTULO II

Noise as an acoustic filter can drive bird species composition in the Amazon Rainforest

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Noise as an acoustic filter can drive bird species composition in the Amazon Rainforest

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Noise pollution can impair acoustic communication by overlapping signals in a process called acoustic masking. This is especially harmful to animals that rely on acoustic communication to perform their social interactions, thus interfering with their reproductive success and survival. While noise is becoming ubiquitous, it may be changing the species composition of natural communities, filtering out those species that use the same frequencies. We evaluated whether mining noise influences bird species composition according to their song acoustic traits. We surveyed five areas of the Carajás National Forest, PA state, Brazil, three under the influence of mining noise (15 sites) and two without mining noise (10 sites). We monitored the bird composition and the anthropogenic noise in each site from 2015 to 2019 and recorded/gathered the species' songs. We analyzed four acoustic traits: the minimum and maximum fundamental frequencies, the dominant frequency, and duration. We attained a mean species value for each acoustic trait, and we used the species composition of each site to calculate the mean acoustic trait values for each of the 25 sites. We fitted type II ranged major axis regression to evaluate the relationship between the mean acoustic traits and the background noise profile. We found that mining noise might have changed the species composition according to their song traits. The areas permeated by mining noise are composed of species that mostly use high-frequency songs with longer durations. Noise can decrease the habitat value without physical changes. The defaunation of species with low-pitched vocalizations could further impact the food-web structural complexity, community dynamics stability and ecosystem functioning. For this reason, it is essential to assess the impact of anthropogenic noise for the effective management of preserved areas and ecosystem conservation.

Keywords: acoustic integrity, acoustic traits, Carajás National Forest, habitat quality, mining noise, noise pollution.

INTRODUCTION

Sound pollution changes the natural background noise profile and, as a result, it can reduce the acoustic communication efficiency of many species (Lohr et al. 2003, Slabbekoorn and Ripmeester 2008, Francis et al. 2009). Acoustic signals have been selected based on their effectiveness in properly transmitting information under natural background noise (Slabbekoorn and Smith 2002, de Araújo et al. 2020a). Effective use of acoustic signals may be critical to the survival and reproductive success of many animals (Brumm and Slabbekoorn 2005). Beyond territory defense (Aubin et al. 2004) and mate attraction (Ballentine et al. 2004), vocalizations can also be used to avoid predation (Krams et al. 2006), request food (Marques et al. 2010), and mediate complex social interactions (Freeberg 2006, Balsby and Scari 2008, de Araújo et al. 2011, Bradbury and Balsby 2016). The detection and discrimination of signals by a receiver in the presence of masking noise can be decisive for the occurrence of the species (Lohr et al. 2003).

The susceptibility of birds to the impact of sound pollution depends on the frequency, loudness, consistency, and duration of the noise produced and the species' ability to adjust its singing behaviour to a background noise profile (Ortega 2012). Sound pollution can occupy specific portions of the acoustic space, reducing its availability and impairing the communication of many species (Francis 2015). In the presence of noise, some birds are known to shift their signals' temporal and spectral features (Brumm et al. 2004, Slabbekoorn et al. 2012, Gil et al. 2015, Bermúdez-Cuamatzin et al. 2020) to guarantee efficient acoustic communication. Despite the evidence of species thriving in noisy environments (Slabbekoorn and Peet 2003, Hu and Cardoso 2009, Slabbekoorn 2013), many are unable to cope with noise, experiencing physical and physiological constraints, and avoiding and evading noisy sites (Bayne et al. 2008, Ortega 2012, Brumm et al. 2021).

Noise can promote local bird diversity and composition changes (Francis et al. 2011a, McClure et al. 2013). For example, species that use low-frequencies songs are the most prone to be under the effects of noise pollution generated by mining operations, as machines are known to produce low-frequency noise (Duarte et al. 2015). By affecting a specific group of species, sound pollution can be seen as an environmental filter, which will affect the species that sing in low frequencies but not high-pitched songs (Francis et al. 2011a). If higher-pitched songs are less prone to the effects of mining acoustic pollution (Rheindt 2003, McClure et al. 2013), species capable of singing at higher frequency should be favored, while those species that sing at lower frequencies are expected to be filtered out of the community. Therefore, acoustic pollution should lead to changes in species diversity and composition. As song parameters such as the dominant frequency have a deep relation to body mass (Torres et al. 2017, 2020) and are driven mainly by shared ancestry (Mikula et al. 2020), sound pollution can functionally constrain specific portions of the phylogeny by occupying specific frequencies bands (Rheindt 2003, Arevalo and Newhard 2011, Francis et al. 2011a). Signal duration can also play an essential role in the effectiveness of acoustic communication under noisy scenarios since longer songs can increase signal's detectability (Brumm et al. 2004, Wiley 2006, Francis et al. 2011b). We expect those bird species that sing at higher pitches and for more extended time to be favored in noisy environments.

Despite evidence of the negative impact of anthropogenic noise on wildlife, mainly from open cast mining (Ghose 2004, Harding and Boothroyd 2004, Duarte et al. 2015), acoustic noise emissions are still poorly regulated worldwide. The purpose of this study is to evaluate whether acoustic degradation acts as an acoustic filter by promoting changes in the local bird composition according to song features. We expect mining noise to act as an environmental filter, selecting species that use higher frequencies and longer songs.

MATERIAL AND METHODS

Study area

We conducted our study within the Carajás National Forest (FLONA Carajás), located on the southeastern limit of the continuous forest of the Amazon rainforest in Pará, Brazil. FLONA Carajás occupies an area of 392,725 hectares of a heterogeneous scenario with great biological diversity that has been used for ore extraction for more than three decades (Martins et al. 2012). The mining activities are concentrated where *canga* vegetation predominates, which is a ferruginous substrate with a savanna-like vegetation, while our samplings were concentrated in areas of Ombrophilous Forest. We surveyed five areas managed by the Bioindicators Monitoring Project (Vale SA): two areas without mining noise (unmined: S11A and N8) and three areas under the influence of mining noise (mined: Mg, N4 and N5). In addition to being far from urban areas, there are severe access restrictions managed by the mining company (VALE SA) and by the official government agency responsible for protected areas in Brazil (ICMBio). It makes it impossible for the sampled areas to be influenced by other human activities, such as hunting, capture of wild animals, and unauthorized logging. Thus, the only anthropogenic impact in these areas is the mining operation and the noise that such activity produces.

All our sampling areas are close to an open environment, be it a mining pit or open savanna-like vegetation (canga), as shown in Figure 1. This way, we avoided a strong influence of confounding variables associated with the edge effect, influencing bird diversity only in the mined areas. Also, as none of the sampled areas is fragmented or isolated, there is a forest continuity throughout FLONA Carajás. It precludes the effects of possible confounding variables associated with fragmentation, reinforcing that the anthropogenic impact on bird diversity is just mining noise. It is important to point out that our sampled areas are not the only ones being actively mined in Carajás. The distances between the sampled transects and the nearest mining pit vary greatly between areas. N5 sites have the shortest distance to the mined area, ranging from 225m to 575m, with an intense mining operation. On the other hand, Mg and N4 reach up to 638m and 840m, respectively. N8 sites are 7.6km to 9.5km from the nearest mining pit, while S11A sites area 28km from the nearest mining pit. It is expected that the noise suffers greater attenuation and sound absorption at greater distances. Each area has a 2km trail with five sites (each 250m long and 500m apart) perpendicular to the trail but parallel to each other. Each site has five points, totaling 25 sites and 125 points. Birds were sampled at each site, and mining noise was sampled at each point, as described below.

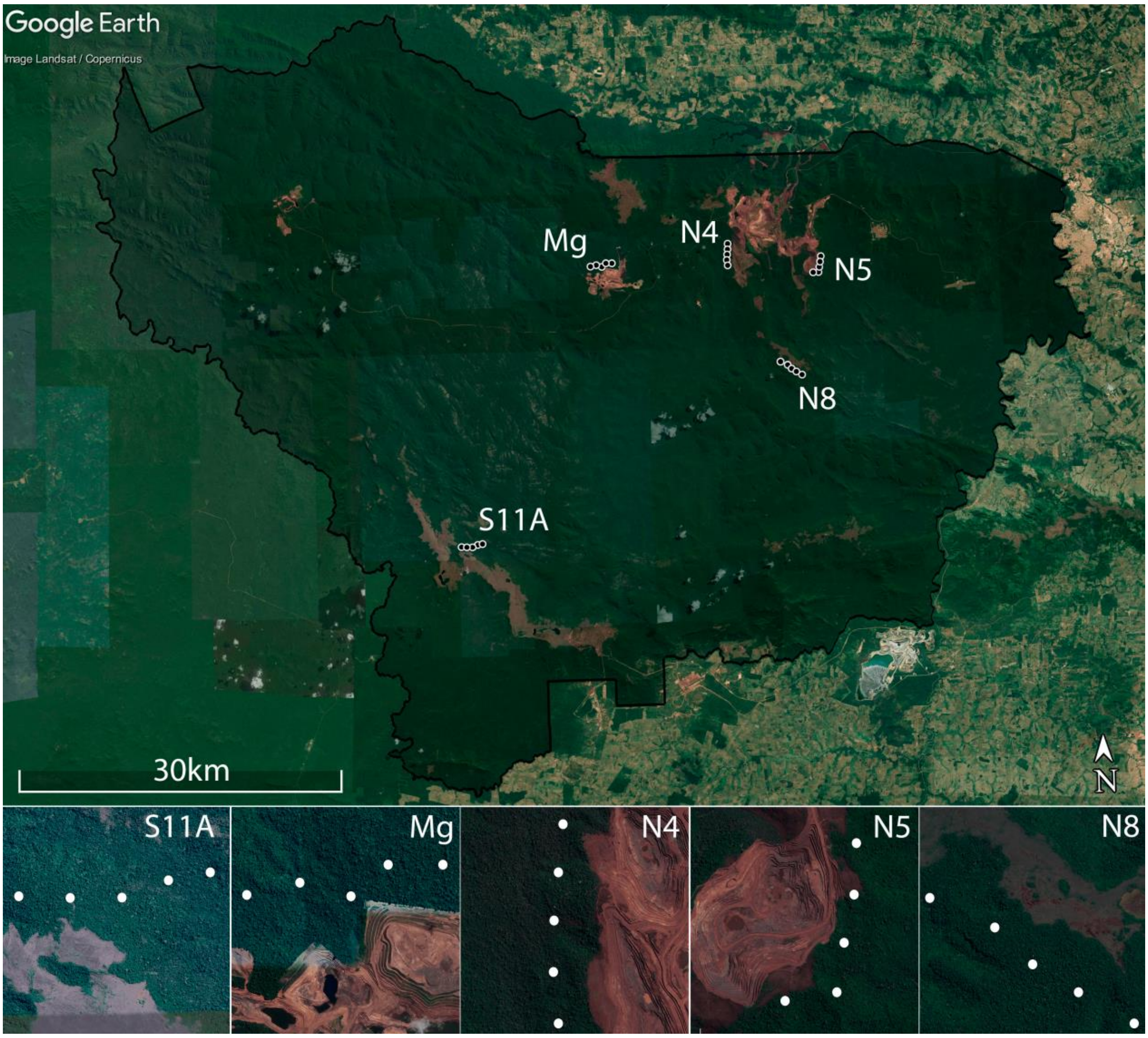

Figure 1. Above: The FLONA Carajás is outlined in black, and the sampled areas are S11A, Mg, N4, N5 and N8. Bellow: the sampled areas and their respective sites (white dots), showing the beginning of each parallel 250m trail, 500m apart. The S11A and N8 sites are close to savanna-like areas, with *canga* vegetation; Mg sites are close to manganese ore mines; N4 and N5 are close to iron ore mines.

Bird sampling

Species survey. To verify if our acoustic sampling accurately represents the local community, we sampled the species using MacKinnon lists (M-Lists; MacKinnon and Phillips 1993, Herzog et al. 2002) carried out at each site once a year for five years (2015 to 2019), totaling 125 working days. The sampling occurred in the mornings (5:30 am to 7:30 am) during the rainy season (March to May). We used the M-Lists to estimate species richness using the iNEXT package (Hsieh et al. 2016a), implemented in R software (R 2020). The iNEXT package uses interpolation and extrapolation to determine the accumulation of diversity (Hsieh et al. 2016b).

Species recordings. We recorded most species opportunistically in our sites and, for some listed species, we carried out active searches using the playback of the species` song. The sampling took place over 280 days, usually in the morning (5:20 am to 11:00 am), between 2013 and 2020 (Mar. 2013, Feb., Aug., Sept. 2014, Mar., Apr., May 2015-2021, Sept., Oct. 2019-2020). Details of the recording equipment are described in Supplementary Material I. Even though our recording effort was extensive, we could not make high fidelity recordings of all sampled species, so we gathered additional recordings from Xeno-canto (https://www.xeno-canto.org/) and Wikiaves (http://www.wikiaves.com.br/). To avoid population vocal variations, we only included recordings of the Xingu endemic area, a continuous biogeographic unit located between large rivers (Cracraft 1985, da Silva and

Garda 2010), where the FLONA Carajás is located. To allow proper comparisons (Zoolinger et al. 2012), all files were edited with a high-pass filter at 125 Hz and normalized to 0dB preceding the analysis. The standardization and acoustic parameters measurements were made in the COOL EDIT PRO software (Syntrillium Software Corporation 2002) through direct measurements over spectrograms and power spectra, built with 512 FFT size, Hamming window.

Acoustic traits. We measured four acoustic traits that could be assessed analogously among the species (Fig. 2), which best represent the morphological limits to frequencies emission (Torres et al. 2017, 2020). Within the frequency spectra, we measured the dominant frequency (FDOM; i.e. the frequency with the highest energy), the minimum (FFMIN) and the maximum (FFMAX) fundamental frequencies (i.e. the minimum and maximum values of the fundamental harmonic). We also measured the duration (DUR) of the song, defined as the structure that repeats itself that could consist of a single or multiple notes, as redundancy (or repetition) which might be relevant for communication amid sound pollution. We attained a mean species value for each acoustic trait, and we used the species composition sampled by the M-Lists of each site to calculate the mean acoustic trait values (MATV) for each of the 25 sites. To avoid pseudoreplication, we measured a single song of each sound recording. The mean trait value is a flexible and straightforward approach to assess a community's acoustic composition, which allows evaluating whether noise changes the composition of a community based on song characteristics.

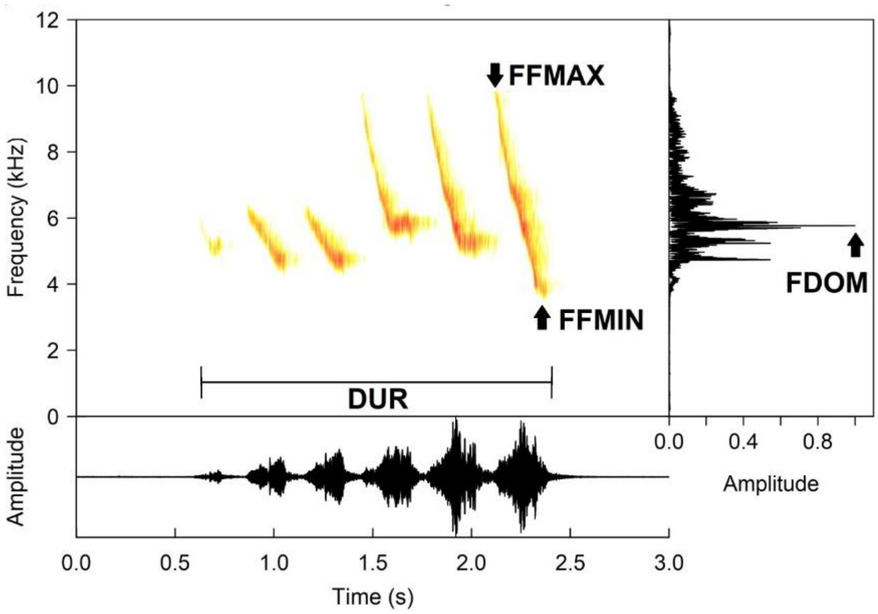

Figure 2. Sonogram of the song of the Golden-crowned Warbler Basileuterus culicivorus, illustrating the vocal parameters measured. FFMAX, maximum fundamental frequency; FFMIN, minimum fundamental frequency; FDOM, dominant frequency, DUR, song duration.

Acoustic integrity

We describe acoustic integrity in two distinct ways. First, to quantify the acoustic energy generated by mining noise, we used a Brüel & Kjaer BK2270 decibel meter to measure equivalent continuous A-weighted sound pressure level (LAeq) for one minute. Second, we used the concept of acoustic entropy (H index) to measure the distribution of such energy across the spectra (Sueur et al. 2008). The H index varies from 0 to 1, approaching 0 when the energy is concentrated in a narrow frequency band and 1 when the signal is evenly distributed in the frequency spectra. Thus, areas with mining noise - i.e. areas with a significant amount of energy concentrated in a narrow bandwidth in the lower portion of the spectra – will have high sound pressure values (LAeq) and low acoustic entropy values (H index).

To obtain the H index, we recorded the soundscape using a Tascam DR680 recorder coupled to a Behringer ECM8000 flat response omnidirectional microphone positioned at 2.4m above ground. We recorded the soundscape for three minutes, using a non-compressed wave file format with a 48kHz and 16bit resolution. The recordings were made between 9 am and 3 pm, after the peak of bird activity (de Araújo et al. 2020b), to avoid interferences from biophony and geophony. We calibrated the recordings using a signal of 1000Hz with 94dB, with an IMPAC type I calibrator. These files were edited to eliminate artefacts such as shocks on the microphone or conspicuous events of geophony or biophony. We used one minute of each edited recording to calculate the acoustic entropy (H index; Suer et al. 2008), using the package soundecology version 1.3.2 (Villanueva-Rivera and Pijanowski 2016) implemented in software R (R 2020). The LAeq and noise recordings were made at the 125 sampling points and, after calculating the H index, we used the median value of each site as a reference value for the sites. In addition, we described the recorded background noise trends by one-third octave bands to display the sound pressure levels present in each band and the specific bands in which the mining noise is concentrated.

Statistical analysis

We fitted type II ranged major axis regression (RMA) to evaluate the relationship between the acoustic traits (here, represented by the MATV) and the background noise profile, both quantitatively (LAeq) and qualitatively (H). RMA appropriately estimates the true functional relationships between variables in different units and are not controlled by the researcher (Legendre and Legendre 2000). The RMA can handle errors in both the predictor and response variables, using vertical and horizontal distances of the data points from the resulting line (Harper 2016). The RMA regressions were performed using the lmodel2 package (Legendre 2014), implemented in R software version 4.0.2 (R 2020). We considered a significant relationship when the regressions had a $p < 0.05$ and the confidence intervals of the slope did not include zero to avoid misinterpretation of the directionality of the relationship.

RESULTS

We recorded 287 species in 836 M-lists, representing sampling completeness of about 91% of the 316 estimated species (Supplemental material II). We successfully sound recorded 195 species on our sites and 16 species on the surrounding forest sites of Carajás. Also, we gathered recordings of another 54 species for the Xingu endemic region, totaling 265 species and 1988 analyzed songs. We did not manage to record or find sound recordings for 22 species. The recordings show an amazing acoustic diversity, in which dominant frequency ranged from 374 to 9689Hz, and the minimum and maximum fundamental frequencies ranged from 236 to 9688Hz and from 502 to 12852Hz, respectively. The song duration can be as short as 0.03s and as long as 32s (Data available in Supplementary Material III).

Our data show that sound pollution occupies the low portion of the spectrum, concentrating energy in the 25 Hz to 500 Hz one-octave bands (Fig 3). Our results indicate that bird composition is influenced by noise pollution (Table 1; Fig 5); as the mean value of the acoustic traits (dominant frequency, the minimum and maximum fundamental frequencies) were higher in areas of high background noise levels (LAeq), and low values of acoustic entropy (H index). That is, areas permeated by mining noise are composed of species that mostly use high-frequency songs. The song's duration was related only to the H index, indicating a weaker relationship with noise. Our models suggest that a 7dB increase on background levels can be related to an increase of 300Hz in dominant frequencies (Fig 4).

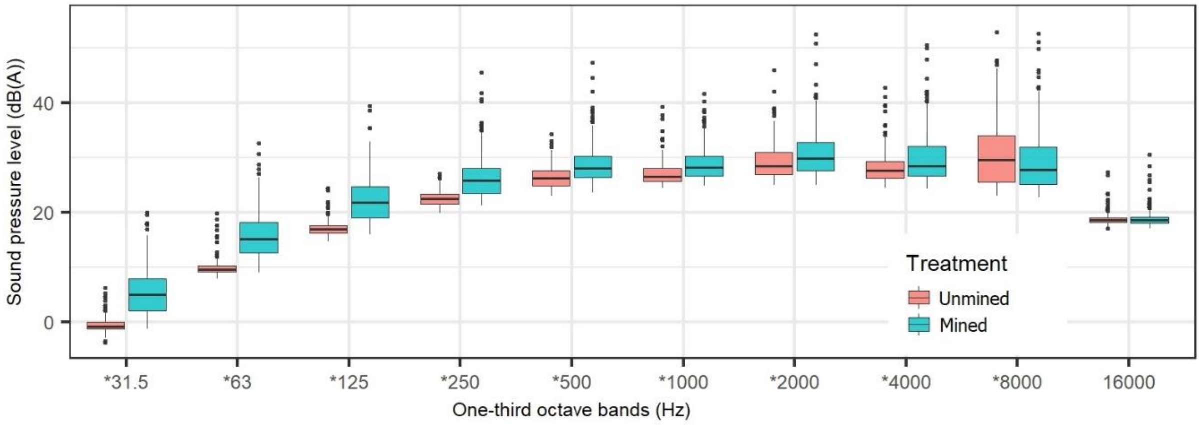

Figure 3. Sound pressure level per one-third octave bands present in mined areas (light pink), and unmined areas (blue), exhibiting a great amplitude (energy) concentrated in bands from 31.5 to 500Hz only in mined areas, while the other higher frequency bands have a more similar amplitude.

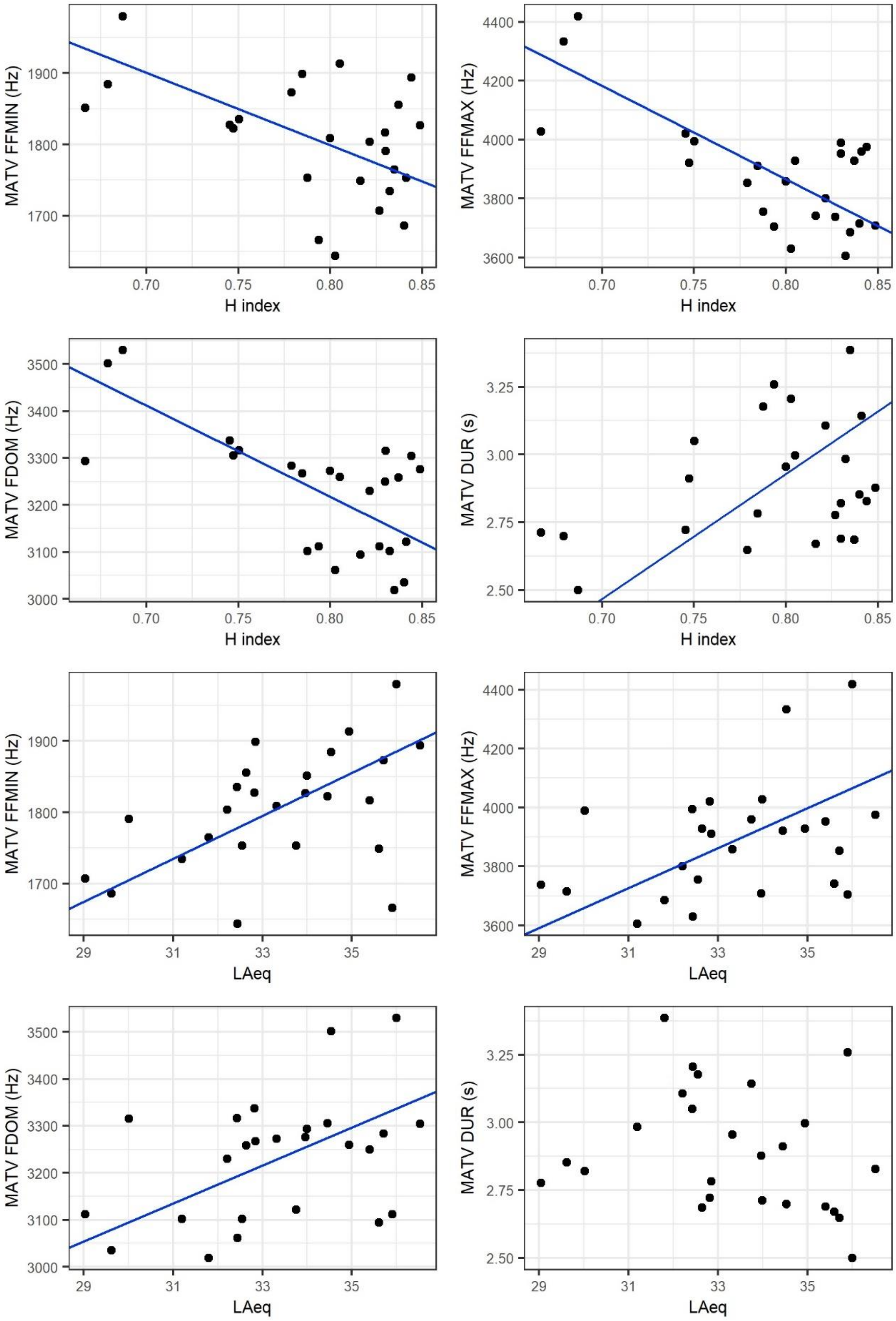

Figure 4. RMA regression models between the MATV of the acoustic traits and the acoustic entropy (H index), and the background noise level (LAeq).

Table 1. The relationship between the acoustical traits and mining noise. The acoustic traits were represented by the community's mean acoustic trait value (MATV) and the mining noise by the H index and LAeq. FFMIN, minimum fundamental frequency; FFMAX, maximum fundamental frequency; FDOM, dominant frequency; DUR, duration of the song.

	MATV	**Intercept**	**Slope**	**CI**	**p-perm**
H index	**FFMIN**	**2612.8**	**-1017.6**	**-2064.9 | -229.6**	**0.01**
	FFMAX	**6403.5**	**-3173.4**	**-4857.0 | -1865.7**	**0.01**
	FDOM	**4772.7**	**-1944.4**	**-3062.0 | -1017.8**	**0.01**
	FDUR	**-0.8**	**4.6**	**1.1 | 30.4**	**0.04**
LAeq	**FFMIN**	**801.5**	**30.1**	**9.4 | 60.9**	**0.01**
	FFMAX	**1625.0**	**67.8**	**6.6 | 175.3**	**0.03**
	FDOM	**1881.6**	**40.4**	**5.5 | 87.5**	**0.04**
	FDUR	8.9	-0.2	0.3 | -0.03	0.14

CI, 95% confidence intervals; p-perm, the permutational probability; Significant relationships are marked in bold.

DISCUSSION

We examined the influence of mining noise on the acoustic composition of the bird diversity. The mining activities influence the surrounding soundscapes, increasing the overall sound pressure levels of background noise and concentrating the acoustic energy in the lower portions of the spectra, triggering a major change in the availability of acoustic space. Our results suggest that mining noise filters certain species, driving changes in the composition of bird communities. The areas with low acoustic integrity retain species that mostly use high-frequency songs.

LAeq measurements and the H index seem to efficiently represent the acoustic integrity and predict the mean frequencies value used by the bird species. Anthropogenic noise can vary in frequency, loudness, consistency, and duration (Ortega, 2012), and how the acoustic energy is distributed along the spectrum is as important as the energy (i.e. the amplitude). Background noise measurements have been successfully used to evaluate acoustic changes in natural environments generated by anthropogenic noise and its impacts in bird communities (Nemeth and Brumm 2010, Barber et al. 2011, McClure et al. 2013, Perillo et al. 2017, Injaian et al. 2018). Even though the H index was not primarily created to evaluate anthropogenic noise, the typical low-frequency noise of the mining activity can reduce the acoustic entropy (Sueur et al. 2008; Golden 2020) so that the H index can be used as a proxy of acoustic degradation, describing how the acoustic energy of anthropogenic noise is distributed across the spectrum.

Our data suggest that species composition changes due to the acoustic degradation caused by mining noise. Human activities such as industry, construction and transportation mostly emit acoustic energy at low frequencies (Herrera-Montes and Aide 2011). By occupying a specific portion of the acoustic space, anthropogenic noise should affect species that use the same frequency band (Francis and Barber 2013). For example, it has been reported that near gas compressors and roads, species with lower frequencies vocalizations exhibit stronger negative responses (Rheindt 2003, Francis et al. 2011a). Noise can interfere with important biological phenomena such as mate attraction and territory defense (Brumm 2004). Frequent noise exposure can also change vigilance rates (Quinn et al. 2006), foraging behavior (Siemers and Schaub 2011), spatial distribution (Arevalo and Newhard 2011), increase stress while decreasing immune response (Kight and Swaddle 2011, Blickley et al. 2012), and affect survival and reproductive success (Habib et al. 2006, Leonard and Horn 2012).

From a functional perspective, the features of a bird’s song are linked to the species’ ability to persist in the face of environmental changes and disturbances – in other words, they represent functional response traits (Díaz et al. 2013) should be affected by noise. A decline of species with low-frequency vocalizations may have a ripple effect on ecosystem functioning. For example, there is a relationship between the frequency used and body mass, in which larger birds use lower frequencies (Fletcher 2004, Torres et al. 2017, 2020), making those species more susceptible to anthropogenic noise. The loss of larger frugivorous birds could impact the dispersal of large-seeded plants, and, as larger birds usually also have wider territories, the dispersal of seeds of any size over long distances could also be affected (Markl et al. 2012, Galetti et al. 2013, Donoso et al. 2017). In addition, noise can also influence predator-prey interactions. Species rely on alarm calls or the sound of a predator approaching, especially during feeding time, which can go unnoticed by the presence of noise (Slabbekoorn and Ripmeester 2008). Some species can respond to noisy environments by increasing vigilance rate at the expense of feeding time (Quinn et al. 2006). That is, noise can either increase the predation risk, or it can indirectly lead to a fitness cost for those trying to compensate for it. Although the delivery of ecosystem services by birds is not fully understood, key functions such as predator-prey interactions and seed dispersal seem important to maintain the biological diversity in forest ecosystems in the tropics (Gaston et al. 2018).

Biodiversity loss is a growing concern and, in addition to structural changes in natural habitats, the increasing noise brought by the constant expansion of human activities have been neglected to a certain extent. The anthropogenic noise seems to be ‘silently’ modifying natural communities, decreasing the habitat value without physical changes (Ware et al. 2015). Due to the biological importance of acoustic communication for birds, it is essential to evaluate the impact of unnatural noise for the effective management of protected areas and bird conservation. The defaunation of species with low-pitched vocalizations could further impact the food-web structural complexity, community dynamics stability and ecosystem functioning.

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Supplementary Material I

Equipment used to record the species in the Carajás National Forest.

Sony PCM-D50 recorder with a parabolic dish of 50 cm & 19cm deep and a Shure Beta 58 microphone;

Sound Devices 722 recorder with a Telinga parabolic dish of 57cm & 12cm deep and a Sennheiser ME62 microphone;

Sound Devices MixPre3 recorder with a Telinga parabolic dish of 57cm & 12cm deep and a Sennheiser ME62 microphone;

Sound Devices 722 recorder with a Sennheiser ME67 microphone;

Marantz 660 recorder with a Sennheiser ME67 microphone;

Marantz 661 recorder with a Sennheiser ME67 microphone;

Sony PCM-M10 recorder with a Sennheiser ME67 microphone;

Sony PCM-M10 recorder with a Rode NTG2 microphone;

Zoom H4n recorder with a Sennheiser ME67 microphone;

Tascam DR-680 recorder with a Behringer ECM 8000 microphone.

All recordings had a sampling rate of at least 48 kHz and 16 bits resolution.

Supplementary Material II

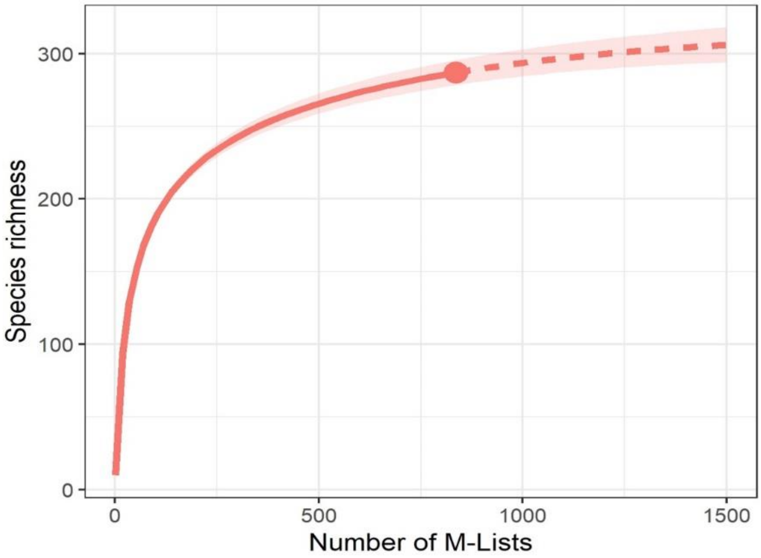

Figure 1. Species accumulation curve of the avian assemblage of FLONA Carajás, Brazil. Solid line: interpolated method; dashed line: extrapolated method.

Supplementary Material III

Table 01. Species used in the analysis and their respective acoustic traits.

Species	FFMIN (Hz)	FFMAX (Hz)	FDOM (Hz)	DUR (s)
Aburria cujubi	1021	4787	1687	0.124
Amazilia fimbriata	1860	7229	6546	2.009
Amazona farinosa	993	1242	4005	0.271
Amazona ochrocephala	414	1003	1195	0.916
Anabacerthia ruficaudata	1074	2943	1808	2.616
Anodorhynchus hyacinthinus	250	531	1779	0.348
Ara ararauna	425	689	1359	0.575
Ara chloropterus	370	580	1664	0.699
Ara macao	727	987	1781	0.452
Ara severus	412	503	2671	0.472
Aratinga jandaya	1113	1518	3997	0.242
Arremon taciturnus	8142	11600	8546	2.138
Attila spadiceus	1650	2644	2296	3.285
Automolus paraensis	880	2533	3113	1.229
Automolus rufipileatus	743	1620	2078	0.856
Basileuterus culicivorus	3268	9030	5277	1.956
Brotogeris chrysoptera	2650	4797	4383	0.156
Bucco capensis	888	1747	1518	8.778
Buteo nitidus	1113	3254	3000	0.698
Cacicus cela	1002	9545	3962	6.407
Campephilus rubricollis	635	1546	1996	0.49
Camptostoma obsoletum	1961	5023	3750	1.444
Campylorhamphus multostriatus	1306	2847	2436	2.067
Campylorhynchus turdinus	665	3492	1425	3.22
Celeus flavus	1463	2685	2497	2.219
Celeus torquatus	1781	2231	2062	1.708
Celeus undatus	699	1974	1746	0.741
Ceratopipra rubrocapilla	3275	6278	4012	1.83

Cercomacra cinerascens	548	1098	2363	3.911
Cercomacra nigrescens	1050	2595	2305	1.569
Certhiasomus stictolaemus	1492	4173	3396	1.442
Chaetura brachyura	2599	7329	6933	0.099
Chaetura spinicaudus	3215	7862	5770	0.797
Chiroxiphia pareola	1887	4087	2543	0.268
Chloroceryle inda	1464	5102	4435	1.51
Cissopis leverianus	1813	9483	4312	5.02
Cnemotriccus fuscatus	1712	4317	3390	0.385
Coereba flaveola	2700	10282	7389	1.773
Colonia colonus	1882	4941	3750	0.373
Conirostrum speciosum	4988	8049	5900	0.556
Conopophaga aurita	1676	4317	3550	2.209
Conopophaga melanogaster	841	2210	4393	0.649
Contopus nigrescens	2050	4624	4038	0.348
Crax fasciolata	375	1270	1125	0.621
Crypturellus cinereus	1626	1848	1683	1.448
Crypturellus soui	1594	2045	1759	1.502
Crypturellus strigulosus	1286	1665	1423	5.448
Crypturellus tataupa	1245	3106	2015	1.653
Crypturellus variegatus	1453	1984	1671	5.789
Cyanerpes caeruleus	1253	7338	7149	0.076
Cyanocorax cyanopogon	1619	2323	3764	0.3
Cyanoloxia rothschildii	2645	4509	3293	2.465
Cyclarhis gujanensis	1561	3523	2492	0.984
Cymbilaimus lineatus	1146	1585	1372	3.197
Dacnis lineata	4987	11272	9093	0.413
Dendrexetastes rufigula	1473	3048	2265	4.384
Dendrocincla fuliginosa	838	1986	3375	3.618
Dendrocolaptes picumnus	959	2877	2175	3.311
Dendrocolaptes retentus	1180	3738	2109	2.601
Dendroplex picus	787	3424	2024	4.13
Deroptyus accipitrinus	809	3552	3143	0.36

Dichrozona cincta	2900	3776	3359	15.045
Dryocopus lineatus	883	1707	2203	0.771
Elanoides forficatus	1921	4621	3143	2.316
Epinecrophylla leucophthalma	4600	7516	6335	2.749
Epinecrophylla ornata	3089	6902	6514	1.245
Euphonia chlorotica	3287	4024	3875	0.93
Euphonia chrysopasta	2176	4339	3962	0.556
Euphonia rufiventris	2565	5666	4134	0.56
Falco rufigularis	1236	2189	1937	1.51
Formicarius analis	1594	2274	1886	1.752
Formicarius colma	2041	3861	2570	2.624
Formicivora grisea	937	1983	2109	9.76
Galbula cyanicollis	916	4082	2854	5.715
Galbula dea	1205	2355	2767	2.175
Geotrygon montana	394	600	445	0.789
Glaucidium hardyi	1211	1609	1359	2.024
Glaucis hirsutus	3560	9246	8437	1.666
Glyphorynchus spirurus	2601	5373	6373	0.343
Grallaria varia	354	637	469	2.077
Granatellus pelzelni	2826	4400	3657	2.416
Hemithraupis guira	3833	8718	6497	2.042
Hemitriccus griseipectus	1818	3184	4700	0.112
Hemitriccus margaritaceiventer	1639	4606	2928	0.502
Hemitriccus minimus	2133	2528	2367	0.769
Hemitriccus minor	1474	3010	5293	0.771
Herpetotheres cachinnans	369	887	1486	29.201
Herpsilochmus rufimarginatus	883	2532	2345	1.626
Hydropsalis albicollis	875	2724	2078	0.551
Hydropsalis nigrescens	775	2102	1601	0.291
Hylexetastes brigidai	1136	2532	2668	3.024
Hylocharis cyanus	5100	9057	6859	1.95
Hylocharis sapphirina	8704	9896	8687	1.127

Hylopezus berlepschi	533	1538	984	1.751
Hylophilus hypoxanthus	2398	4678	3785	0.849
Hylophilus ochraceiceps	2830	3183	2938	0.818
Hylophilus semicinereus	2512	3653	3295	3.494
Hylophylax naevius	2032	6044	4078	3.786
Hylophylax punctulatus	2184	4852	4207	0.471
Hypocnemis striata	1162	3364	2537	2.795
Ibycter americanus	704	7054	1406	3.899
Icterus cayanensis	1030	6249	3314	3.528
Iodopleura isabellae	3442	4842	4515	0.327
Isleria hauxwelli	2523	5719	4104	14.51
Jacamerops aureus	2232	3062	2762	2.714
Lamprospiza melanoleuca	2664	4668	4012	0.432
Lanio cristatus	5801	7310	6373	0.197
Lanio luctuosus	5679	8018	7123	1.181
Lanio versicolor	3203	10714	5465	1.644
Laniocera hypopyrra	2002	3020	2554	3.069
Lathrotriccus euleri	2380	4589	3907	0.539
Lepidocolaptes layardi	2272	4606	3433	1.837
Lepidothrix iris	1441	3424	2428	0.276
Leptopogon amaurocephalus	1180	1771	1500	2.451
Leptotila rufaxilla	457	650	504	0.77
Lipaugus vociferans	868	5453	2502	2.195
Lurocalis semitorquatus	1276	1913	1503	0.102
Machaeropterus pyrocephalus	3173	3920	3375	0.142
Malacoptila rufa	3675	4834	4406	3.156
Megarynchus pitangua	1059	1881	4263	0.8
Megascops choliba	407	870	633	1.33
Megascops usta	734	965	797	14.796
Melanerpes cruentatus	1022	2141	2937	0.72
Micrastur mintoni	490	1304	1336	10.254
Micrastur mirandollei	595	877	1397	4.838
Micrastur ruficollis	495	1213	1718	1.012

Micrastur semitorquatus	428	756	1218	0.437
Microcerculus marginatus	1682	3950	2337	31.916
Milvago chimachima	625	4410	4048	0.692
Mionectes macconnelli	1423	3517	4640	1.89
Mionectes oleagineus	912	3075	2885	2.855
Momotus momota	236	585	375	0.383
Monasa morphoeus	1032	2694	2203	2.586
Myiarchus ferox	2095	3423	3129	0.251
Myiarchus tuberculifer	1702	4044	4820	0.634
Myiopagis caniceps	2060	6515	4218	3.12
Myiopagis gaimardii	3077	5863	5531	0.331
Myiornis ecaudatus	4239	5590	4750	0.17
Myiothlypis flaveola	2363	6776	4981	1.838
Myiothlypis mesoleuca	1858	4349	3316	3.512
Myiozetetes cayanensis	2295	3482	3127	1.196
Myrmelastes rufifacies	2412	4468	4853	2.005
Myrmoborus leucophrys	1248	3291	2959	4.097
Myrmoborus myotherinus	2014	4072	3709	2.665
Myrmornis torquata	1616	4286	3227	5.659
Myrmotherula axillaris	1228	3678	2472	2.719
Myrmotherula brachyura	1486	3748	2619	2.803
Myrmotherula longipennis	2224	4952	3454	2.897
Myrmotherula menetriesii	1671	4679	2967	6.573
Nonnula ruficapilla	1451	3035	2601	11.558
Notharchus hyperrhynchus	1317	2074	1713	5.953
Notharchus tectus	2048	5131	4546	7.773
Nyctibius aethereus	313	810	539	0.699
Nyctibius griseus	474	1102	947	3.04
Nyctiphrynus ocellatus	853	1985	1592	0.519
Nystalus torridus	1499	2126	1865	3.24
Odontophorus gujanensis	582	1139	853	16.335
Onychorhynchus coronatus	748	3078	2852	0.188
Ornithion inerme	3267	5466	5097	1.072

Oxyruncus cristatus	2293	5237	5109	3.181
Pachyramphus castaneus	2362	3865	3867	2.095
Pachyramphus marginatus	2252	3042	2656	1.488
Pachyramphus minor	3662	5886	3890	0.215
Pachyramphus viridis	1544	3491	3779	2.421
Patagioenas cayennensis	350	750	581	6.913
Patagioenas speciosa	238	541	388	3.506
Patagioenas subvinacea	667	1202	1265	1.177
Penelope pileata	297	810	905	12.897
Phaeomyias murina	1020	4727	4048	1.136
Phaethornis ruber	3506	9180	7225	2.641
Phaethornis superciliosus	5374	9430	7230	0.111
Pheugopedius coraya	838	2405	1770	1.549
Philydor erythrocercum	3054	8142	7905	0.103
Phlegopsis nigromaculata	1965	3950	3593	1.828
Phyllomyias fasciatus	1245	3411	2437	4.451
Piaya cayana	931	3432	2648	0.901
Piculus flavigula	1355	1692	1525	1.552
Piculus laemostictus	1877	2487	2336	1.708
Piculus leucolaemus	2005	2399	2297	1.887
Picumnus aurifrons	6313	7835	7578	0.581
Pionites leucogaster	1177	3786	2718	1.884
Pionus menstruus	1156	4363	3390	0.237
Pipra fasciicauda	1380	3004	3211	0.379
Piprites chloris	1331	1910	1604	2.573
Pitangus sulphuratus	942	2242	3122	0.785
Platyrinchus platyrhynchos	3050	3720	3093	1.955
Platyrinchus saturatus	1792	3817	3679	0.581
Poecilotriccus capitalis	1106	2938	4763	1.604
Poecilotriccus sylvia	990	2743	1593	1.064
Progne chalybea	1219	8031	3187	0.204
Psarocolius decumanus	430	4474	3937	4.601
Psarocolius viridis	383	1313	829	1.957

Psittacara leucophthalmus	362	763	2519	0.477
Psophia interjecta	566	1191	2390	4.589
Pteroglossus aracari	1029	3215	3057	0.454
Pteroglossus bitorquatus	859	1422	2368	0.387
Pteroglossus inscriptus	673	1336	1793	0.119
Pygiptila stellaris	2327	2991	2655	1.031
Pyriglena leuconota	1519	2756	2526	2.149
Pyrilia vulturina	815	2429	2836	0.056
Pyrrhura amazonum	1263	2419	3771	0.165
Pyrrhura lepida	1006	1761	3445	0.102
Querula purpurata	577	1489	1135	2.182
Ramphastos tucanus	1220	1787	1520	1.304
Ramphastos vitellinus	1422	2169	1766	4.056
Ramphocaenus melanurus	2150	3251	2988	2.006
Rhytipterna simplex	1293	2556	2216	3.692
Rupornis magnirostris	820	2329	3746	1.878
Saltator coerulescens	1208	3536	1898	3.111
Saltator grossus	1547	3095	2421	1.243
Saltator maximus	1560	6830	4244	1.509
Schiffornis turdina	2837	4058	3012	3.39
Sclateria naevia	1827	4542	3553	4.906
Sclerurus macconnelli	3266	5578	4011	2.742
Sclerurus rufigularis	2708	5534	4108	2.294
Selenidera gouldii	371	786	553	7.728
Setophaga pitiayumi	3395	8657	6539	2.214
Sittasomus griseicapillus	1365	3396	3031	2.25
Spizaetus tyrannus	1641	2507	2347	1.523
Synallaxis cherriei	941	1913	2975	0.446
Synallaxis rutilans	681	1722	2768	0.393
Synallaxis scutata	1875	4442	3464	0.962
Tangara episcopus	2678	8999	6352	2.365
Tangara gyrola	1918	3109	7192	0.352
Tangara mexicana	9688	12852	9689	12.583

Tangara nigrocincta	3705	10005	8062	1.462
Tangara palmarum	1670	10357	6890	5.876
Tangara punctata	2714	12575	8074	1.149
Taraba major	717	1784	1356	9.755
Terenotriccus erythrurus	4054	6972	5078	0.502
Thamnomanes caesius	1147	5263	2845	11.219
Thamnophilus aethiops	454	890	1443	3.082
Thamnophilus amazonicus	500	1691	1265	2.309
Thamnophilus schistaceus	510	1004	1603	2.874
Thamnophilus stictocephalus	435	1429	1525	2.973
Tinamus guttatus	946	1268	1031	7.084
Tinamus tao	1251	1523	1265	0.503
Todirostrum chrysocrotaphum	4747	6368	5781	0.036
Tolmomyias assimilis	2479	6186	4183	1.279
Tolmomyias flaviventris	5581	7048	5652	0.256
Tolmomyias poliocephalus	1897	4086	3909	0.294
Tolmomyias sulphurescens	3033	7036	5086	0.43
Trogon melanurus	712	1408	1153	9.143
Trogon ramonianus	851	1655	1428	8.691
Trogon rufus	906	1572	1430	3.02
Trogon viridis	659	1624	1253	5.669
Turdus fumigatus	1888	3244	2086	0.541
Turdus leucomelas	1629	3325	2067	2.724
Tyranneutes stolzmanni	2484	3788	2559	0.349
Tyrannulus elatus	1926	2842	2631	0.493
Tyrannus melancholicus	3867	6813	5390	0.355
Vanellus chilensis	382	1253	4651	7.961
Veniliornis affinis	1467	1667	3140	2.79
Vireo chivi	2394	4992	3470	0.297
Vireolanius leucotis	1689	2762	2310	0.349
Willisornis vidua	2226	4589	3471	4.92
Xenops minutus	2531	6672	5941	1.157
Xenops rutilans	2787	6856	6187	0.802

Xiphocolaptes carajaensis	1317	3356	2438	5.411
Xipholena lamellipennis	786	1798	1574	0.173
Xiphorhynchus guttatoides	1194	2325	1882	2.898
Xiphorhynchus spixii	1373	5593	3504	2.298
Zimmerius acer	2499	4217	3156	0.328
Zonotrichia capensis	2261	8307	4406	1.968

CAPÍTULO III

Diversity in a noisy world: amazon birds respond negatively to acoustic degradation

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Diversity in a noisy world: amazon birds respond negatively to acoustic degradation

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Acoustic communication is essential in the social interactions of many animals. However, the constant human expansion imposes unprecedented acoustic conditions to their communication. While noise is spreading worldwide, its impact on distinct diversity components has not yet been fully addressed, and conflicting factors such as vegetation structure often inhibit strong conclusions. We evaluated the impact of mining noise on the taxonomic and phylogenetic diversity of birds of the Carajás National Forest, Brazil, while considering differences in vegetation structure. Five areas were surveyed: three areas under the influence of mining noise and two without mining noise, each with five sites. We monitored the bird diversity and the mining noise in each site from 2015 to 2019. We found that anthropogenic noise had a negative effect on the taxonomic diversity and phylogenetic diversity but a positive effect on the average phylogenetic diversity per species. The decrease in phylogenetic diversity with an increase in the average phylogenetic diversity per species could result from two possible scenarios: a greater loss of phylogenetically similar species and/or changes in species composition, with the colonization or increase of human-tolerant and opportunistic species. As expected, vegetation structure influenced taxonomic and phylogenetic diversity, especially species richness. Our findings show that mining noise can act as a selective force influencing birds' diversity and vegetation structure. Noise is an invisible source of habitat degradation and may affect ecosystem functioning. Given that noise pollution is becoming ubiquitous, it is crucial to assess the effects of anthropogenic noise for the correct management of protected areas.

Keywords: mining noise, noise pollution, acoustic integrity, habitat quality, bird diversity, acoustic communication.

INTRODUCTION

Human activities pose new challenges to animal communication by acoustically affecting the environment (Brumm, 2013). The constant expansion of human activities changes the natural background noise imposing unprecedented acoustic conditions in which animals must communicate (Brumm & Slabbekoorn, 2005). If the acoustic characteristics of anthropogenic noise overlap with those of an animal's acoustic signal, it will affect the receiver's ability to detect or discriminate the signal within the soundscape (Lohr et al., 2003). Noise pollution is an invisible source of habitat degradation. Noise can decrease the habitat value without structural changes by occupying important portions of the acoustic space and reducing the availability of specific frequency bands (Ware et al., 2015). It is the least evaluated man-made threat in natural communities, as its impact is neither visible nor has sentimental appeal, such as deforestation and forest fires.

For birds, most biological interactions are strongly dependent on acoustic communication (de Araújo et al., 2011; Manica et al., 2016), making the acoustic space a contested resource among species (de Araújo et al., 2020; Torres et al., 2020). However, noise can be an unbeatable competitor, and negative impacts at distinct biological scales have been reported (Brumm, 2013). At the individual level, noise can cause changes in song component redundancy, frequency, signal amplitude and temporal shifts, as well as physical and physiological changes (Ortega, 2012; Ware et al., 2015). At the population level, anthropogenic noise affects the active communication space in the interactions between individuals (Lohr et al., 2003). For example, reducing signal range can interfere with parental care (Leonard & Horn, 2012), predator avoidance (Quinn et al., 2006), sexual selection and territorial defence (Brumm, 2004; Planqué & Slabbekoorn, 2008), thus affecting the fitness and survival of individuals. At the community level, anthropogenic noise can reduce species richness and abundance (Arevalo & Newhard, 2011; Saha & Padhy, 2011), filtering specific groups of birds according to their vocal characteristics (Francis et al., 2011). Filtering processes can be linked to their evolutionary histories, as bird vocalizations appear to contain conservative elements consistent with their phylogeny (Price & Lanyon, 2002).

The loss of species from once-ideal habitats may have implications for the ecosystem functioning. It is the most alarming damage caused by noise, in the sense that individual adjustments (e.g. changes in singing behaviour) were not enough to prevail over the impacts of noise. However, disentangling the impacts of anthropogenic noise from other factors such as habitat heterogeneity or vegetation structure remains a challenge, which often inhibits the ability to draw strong conclusions. The distribution and diversity of birds can be influenced by habitat heterogeneity, as complex heterogenic vegetation structures support richer biological communities (Stein et al., 2014; Willrich et al., 2019). A broader approach that includes additional determinants of bird diversity is necessary to understand the ecological impacts of noise.

We evaluated the impact of mining noise on the taxonomic and phylogenetic diversity of birds at the Carajás National Forest (Pará state - Brazil) whilst accounting for the effects of vegetation structure. We expect both diversity components to be negatively influenced by anthropogenic noise. Phylogenetic diversity should be more sensitive to changes in the profile of background noise, as noise may filter out specific phylogenetic branches which correlate with particular vocal characteristics. We also expect vegetation structure to influence the diversity of birds so that more complex environments should retain a great number of species. However, this influence should have a minor role compared to the acoustic impact of mining noise, as our sampling areas have similar vegetation structures.

MATERIALS AND METHODS

Study area

The study was carried out in the Carajás National Forest, Pará State, Brazil (FLONA Carajás). FLONA Carajás occupies an area of 392,725 hectares (Jackson & Castilho, 2012) in a sustainable-use region that has been used for ore extraction for over three decades. Nevertheless, FLONA Carajás is one of the few forest remnants still virtually intact on the eastern limits of the Amazon rainforest. The mining activity is concentrated in patches of *canga* vegetation, a kind of ferruginous outcrop savanna, but sampling was concentrated in the nearby Ombrophilous Forest. The Ombrophilous Forest is fully connected throughout the entire FLONA Carajás area (dark grey within the limits of the FLONA Carajás Fig 1), except for the *canga* vegetation patches and mining pits (light grey within the limits of the FLONA Carajás).

We surveyed five areas managed by the Bioindicators Monitoring Project (Vale SA): three areas under the influence of mining noise (Mg, N4 and N5) and two without the acoustic effects of mining (S11A and N8). All areas are connected, with no isolated forest fragments, ranging from 500 to 700m in altitude. Each area has five sites with five points, totalling 25 sites and 125 monitoring points (Fig. 1c). Birds were sampled at each site, while mining noise was sampled at each point. The sampling took place between February and April of 2015 to 2019, one day per year at each site. All our sampling areas are close to an open environment, reducing the possible influence of edge effects affecting the diversity only in the mined areas. The S11A and N8 sites are close to savanna-like areas, with *canga* vegetation, while N4 and N5 are close to iron ore mines and Mg sites are close to manganese ore mines. N5 sites have the shortest distance to the mined area, ranging from 225m to 575m, with an intense mining operation. On the other hand, Mg and N4 reach up to 638m and 840m, respectively. It is expected that the noise suffers greater attenuation and sound absorption at greater distances. Also, Mg has the least intense ore extraction, reaching its capacity and heading towards a complete shutdown. There are severe access restrictions managed by the mining company (VALE SA), and our sampling sites are far from urban areas. It makes it very unlikely for the sampled areas to be influenced by other human activities, such as hunting and capturing wild animals or unauthorized logging. In addition, there is no inappropriate disposal of pollutants near sampling areas.

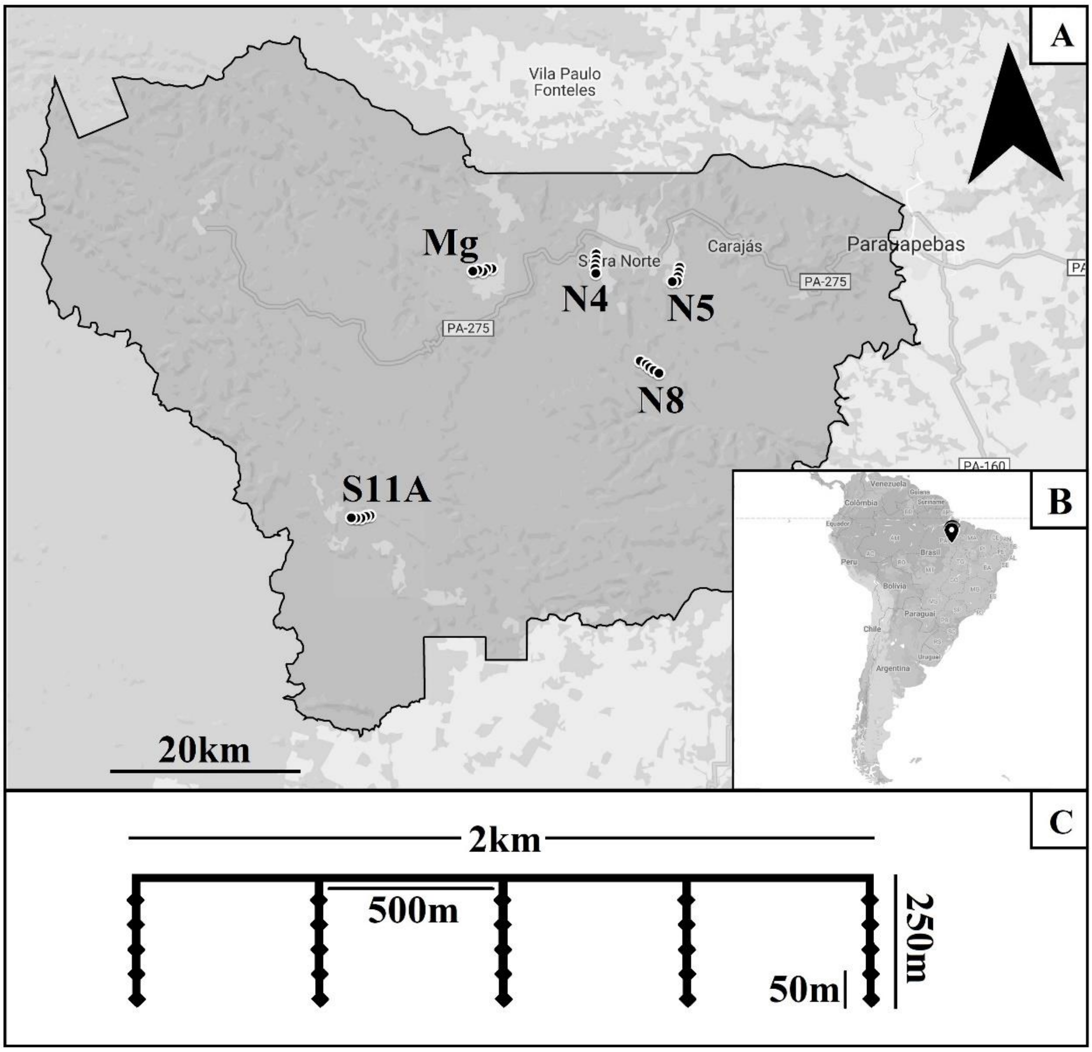

Fig 1. A. Limits of the FLONA Carajás and the sampled areas. B. Location of the Carajás Nacional Forest, Brazil. C. Sampling design adopted in each area: a 2 km trail with five perpendicular sites, each with five points.

Bird survey

Species sampling

Species were sampled using MacKinnon lists (M-lists; MacKinnon and Phillips 1993, Herzog et al. 2002), which is a rapid avifaunal assessment (auditory and/or visual) grouped into lists of 10 species. We made the M-lists on each of the 25 sites during the morning hours (5:30 am to 7:30 am). We used the accumulated M-lists over the 5-year study as a sample unit to represent the species composition of each site.

Taxonomic Diversity

We evaluated the taxonomic diversity using two of Hill's numbers (Hill, 1973): the Species Richness, which gives equal weight to dominant and rare species, and the Inverse Simpson's index, which gives greater weight to dominant species (Gotelli & Colwell, 2011). The M-lists were used as a sample unit to calculate taxonomic diversity indices using interpolation and extrapolation methods from the species occurrence matrix, allowing direct comparison of communities without loss of information (Chao & Jost, 2012). The extrapolation was established so that each site had 41 M-lists, taken as the maximum number of lists made in a single site. The taxonomic diversities were calculated using the iNEXT package (Hsieh et al., 2016), implemented in R software (R, 2020).

Phylogenetic diversity

We calculated the Phylogenetic Diversity (PD) and the Average Phylogenetic Diversity index (avPD). The first is the total length of the branches that comprises a tree subset, reflecting the number of phylogenetic features present in a community (Faith, 1992). The second is the total length of a subset divided by the number of species, reflecting the average amount of phylogenetic history per species (Knapp et al., 2012). PD and avPD can be considered complementary metrics for practical decision making, as the species richness influences each metric differently (Faith, 2018; Tucker et al., 2017). We obtained 10,000 phylogenetic trees from a global avian phylogeny (Jetz et al., 2012) containing all recorded species. We created a consensus tree with the phytools package (Revell, 2012) and calculated the PD and avPD for each site using the species composition recorded on the M-lists with the picante package (Kembel et al., 2010) implemented in the R software (R, 2020).

Acoustic integrity

We described acoustic integrity in two distinct ways. First, we quantified the mining noise in terms of the energy present by measuring the equivalent continuous A-weighted sound pressure level (LAeq) for one minute, with a Brüel & Kjaer BK2270 decibel meter. Second, we used the concept of acoustic entropy (H index) to evaluate how mining noise affects energy distribution (entropy) across the spectrum (Sueur et al., 2008).

We calibrated the recording system using a signal of 1000Hz with 94dB, with an IMPAC type I calibrator, and then recorded the soundscape for 3 minutes with a Tascam DR680 recorder coupled to a flat omnidirectional microphone (Behringer ECM8000) positioned at 2.4m above ground. The recordings were made after the peak of bird activity (8 am to 3 pm) at each point, aiming to evaluate the changes in background noise caused by mining activities (both in terms of energy and entropy) while avoiding interferences from biophony and geophony. Each recording was built using a resolution of 24 bit and 48kHz. We edited the recordings to eliminate artefacts such as wind distortions, unwanted physical shock noise or biophony before the analysis and used the first edited minute to calculate the acoustic entropy.

The H index varies from 0 to 1, approaching 0 when the energy is concentrated around a narrow frequency band and 1 when the signal is evenly distributed across the spectra (Sueur et al., 2008). Because biophony/geophony interferences were avoided during recording sessions, and mining noise is concentrated in a narrow bandwidth at the lower portion of the spectra (Duarte et al., 2015), higher values of energy (i.e. LAeq) and lower values of acoustic entropy indicates acoustic degradation. The acoustic entropy was calculated using the soundecology package (Villanueva-Rivera & Pijanowski, 2016) implemented in R software (R, 2020). As the noise produced by mining can vary over time, the LAeq measurements and noise recordings were made at the five points in each site, and we used the median value of each index as a reference value.

Vegetation structure

We visually estimated the vegetation structure within a 10m fixed radius by adapting a Kaufmann et al. (1999) procedure. We assigned values from 0 to 4 to determine the proportion of large trees (DBH > 15CM), small trees (DBH < 15cm), palm trees, lianas, soil cover and exposed rocks. A value of zero was assigned when the variable was absent, 1 when it was scarce (<10%), 2 when it was medium (10-40%), 3 when it was dense (40-75%) and 4 when it was very dense (> 75%). We counted the presence of standing dead trees, fallen trunks, glades, and ant/termite mounds. We measured canopy height with a Bushnell rangefinder. We obtained canopy openness by converting photographs (taken at the height of 1 m facing upwards) into binary colors (black and white) to evaluate the amount of area of black pixels (i.e. coverage), using the ImageJ program (Schneider et al., 2012). We repeated these assessments at each point to account for variation in habitat structure and used the mean values of each variable as a reference value. We used a Principal Components Analysis (PCA) to reduce the number of explanatory variables used in the statistical procedure, with the prcomp function of the stats package in R (R, 2020). Variable centering and scaling were performed prior to computing the correlation-based PCA to mitigate the influence of variable’s scales a nature.

Statistical analysis

We used the lme4 package (version 1.1-25) to fit linear mixed models using maximum likelihood (Bates et al., 2015) with a Gaussian distribution. We used four diversity indexes as response variables, species richness, inverse Simpson's diversity, Phylogenetic Diversity, and Average Phylogenetic Diversity. The predictor variables included the acoustic integrity, represented by the treatment category (mined vs non-mined), H index and LAeq measurements, and the vegetation structure, represented by the first three principal components (PC1, PC2 and PC3). The areas were used as a random effect to control their influence on the sampled sites, as the sites of each area are prone to have correlated variables. We verified model assumptions on the global model (all explanatory variables) using diagnostic plots of residual distribution and conditional modes of the random effects. The collinearity between variables was evaluated using the variance inflation factor with the car package (J. Fox & Weisberg, 2019), and variables with VIF < 4 will be dropped. Outliers were checked by Cook's distance with the predictmeans package (Luo et al., 2018).

We used the Akaike information criterion adjusted for small sample size (AICc) to compare all possible combinations of the global model, and the models with ΔAICc < 2 were defined as the most plausible set with substantial support (Burnham et al., 2010; Burnham & Anderson, 2002). The model selection was made using the dredge function of MuMin package (Barton, 2019), keeping only the models that had ΔAIC < 2 using the subset function, which renormalizes the weights based on the chosen cut number. As we have only five areas - the minimum number of levels indicated for running a mixed model - models can present a singular fit; that is, the estimated among-areas variance may be zero (Bolker, 2015). If the variance is zero, dropping the random effect variable would not influence the overall results (Pasch et al., 2013), whereas if a random effect has any influence on the response and predictor variables, it must be accounted for (A. Zuur et al., 2009). Therefore, we chose to keep the random effect variable and show a singular fit among our best-ranked models. This procedure ensured that the number of estimated parameters for the random effect matrix was the same since this is a key factor when calculating AICc.

Because more than one model had ΔAIC < 2, we used conditional model averaging to produce model-averaged parameters estimates. This procedure reveals the magnitude and direction of the relationship between the response and predictor variables (Burnham & Anderson, 2002), based on averages over the models where the parameters appear (Barton, 2019). We considered a reliable relationship when the confidence intervals did not include zero to avoid misinterpretation of its directionality. Also, we calculated the conditional coefficient of determination of each model presented, considering both fixed and random factors, with the r.squaredGLMM function of the MuMin package.

RESULTS

We documented 8360 birds (836 M-lists) from a total of 287 species (Supplemental Material I). We estimated the presence of 316 species in the study area so that even though we sampled 287 species (91% of the estimated), additional sampling is required to detect rare species (Supplemental Material II). Inverse Simpson's diversity improves sampling completeness to 99% of estimated bird diversity so that additional sampling would not imply a substantial change in Simpson's diversity estimates (Supplemental Material II).

The LAeq measurements varied across the areas, yet the lowest values were only found where the acoustic effects of mining were absent (Fig 2). On the other hand, the acoustic entropy (H index) showed more consistent results, exhibiting high sensitivity to mining noise. High H values were found only in the control areas with little variation, whereas low values were found in areas close to mining operations. A water stream close to some S11A sites did not affect the H index values but strongly affected the LAeq measurements. Since the noise generated by running water is broad-banded (Goutte et al. 2013), it occupies a wide frequency bandwidth like a white noise, making the acoustic entropy a better approach to evaluate the impact of noise.

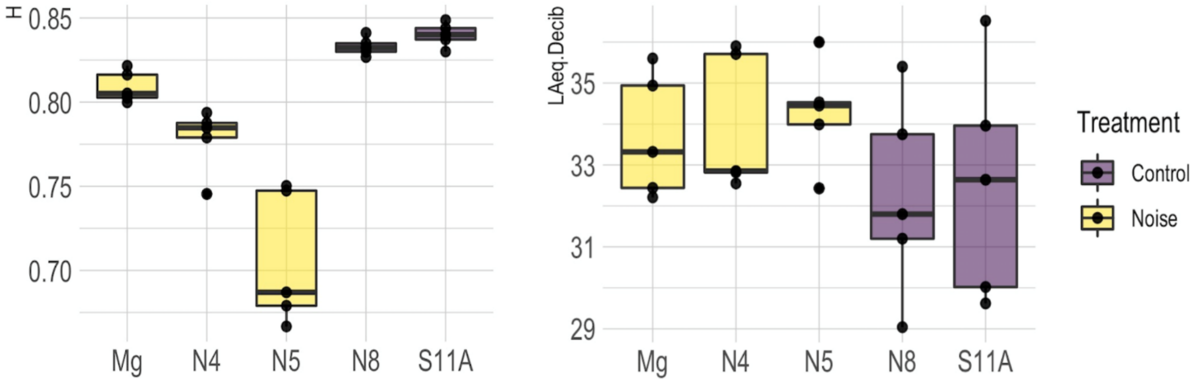

Fig 2. Boxplot of the H index (left) and the LAeq measurements (right) obtained for each sampled area.

The first three principal components can explain most vegetation structure variation (60.4% of the variance). PC1 was positively associated with fallen trunks and the presence of glades, and negative PC1 values seem to be associated with control areas. While PC2 was negatively associated with canopy height and ant/termite mounds, PC3 was positively associated with exposed rocks and negatively associated with soil cover (Supplemental Material III; Fig. 3).

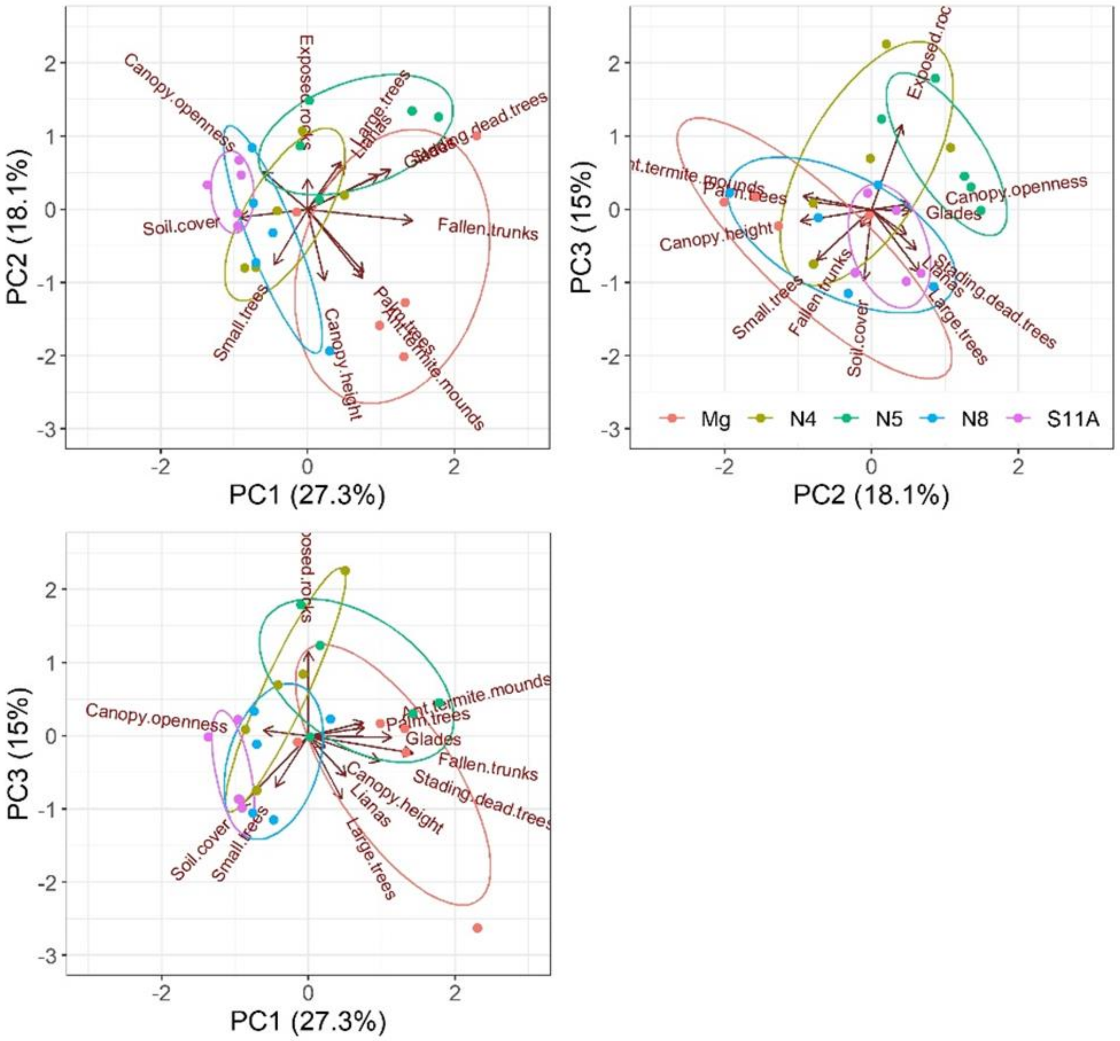

Fig 3. Principal Components Analysis among the evaluated environmental variables measured at the Carajás National Florest.

Candidate models that included PC2 and H index as predictors of taxonomic diversity had the highest support according to the AICc model selection (Table 1). On the other hand, the H index and treatment (mined) were present in all averaged phylogenetic diversity models (avPD), while the LAeq and PC2 were present in the phylogenetic diversity models (PD). The portion of habitat structure related to canopy height and the presence of ant/termite mounds was substantially present in the taxonomic diversity models. In contrast, the phylogenetic diversity (both indices) seems to suffer a stronger influence from acoustic degradation brought by mining operations.

The results of the conditional model averaging are presented in Table 2. The acoustic entropy and treatment positively affected the inverse Simpson's diversity, indicating that the diversity of frequent species is greater in habitats with high acoustic entropy. Background noise levels seem to directly affect the Phylogenetic Diversity (PD), as for each dB of background noise gained, PD is reduced by 49 units. However, the acoustic entropy is negatively correlated to the Average Phylogenetic Diversity. Results may be reflecting a loss of phylogenetic lineages in noisy environments, such as expected in communities subjected to environmental filters. Another possibility is a greater loss of recent lineages, i.e. shorter branches of the phylogenetic tree. For example, a sampling site in area S11A (control area) has 126 species with a PD of 3245.36, while a sampling site in area N5 (mined area) has 96 species with a PD of 2887.85. The avPD for the first is 25.75 (avPD = 3245.36/126), while for the second is 30.08 (avPD = 2887.85/96). While reducing the number of species, increasing the average phylogenetic diversity could mean that the remaining species have a higher phylogenetic "value", that is, species with very distinct evolutionary histories. In this case, the avPD metric is implemented as a complementary metric for PD. Since PD is affected by noise, represented by the background noise level, it supports the above interpretation of the avPD results. PC2 negatively affected taxonomic diversity (both indexes) and PD. Thus, areas with taller forest canopies and a greater presence of ant/termite mounds should have higher taxonomic diversity.

Table 1. Best ranked models (AICc < 2) showing the influence of noise and vegetation structure on the taxonomic and phylogenetic diversity of the bird community of the FLONA Carajás, Pará, Brazil. R2c: conditional coefficient of determination, including both fixed and random effects.

					Random effects			
Response variable	**Model**	**AICc**	**ΔAICc**	**wAICc**	**Variance**	**Residual**	**Singular fit?**	**R2c**
Species Richness	~ PC2	176.4	0	0.303	15.83	36.01	No	0.50
	~ PC2 + H	176.7	0.37	0.252	7.89	34.94	No	0.57
	~ PC2 + H + PC3	177.6	1.22	0.165	7.1	31.41	No	0.59
	~ PC2 + H + Treatment	177.9	1.53	0.141	0.82	36.21	No	0.57
	~ PC2 + LAeq	177.9	1.55	0.139	13.92	34.12	No	0.53
Inverse Simpson`s diversity	~ PC2 + H + Treatment	177	0	0.364	1.63	34.16	No	0.56
	~ PC2	177.4	0.43	0.294	12.15	39.27	No	0.43
	~ PC2 + H	178.4	1.46	0.175	13.15	35.23	No	0.54
	~ H + Treatment	178.5	1.57	0.166	0.43	43.25	No	0.44
Phylogenetic diversity	~ LAeq + PC2	344.5	0	0.649	2520	31208	No	0.39
	~ LAeq	345.7	1.23	0.351	7884	34143	No	0.32

Average Phylogenetic diversity	~ H + Treatment + PC1 + LAeq	87.4	0	0.317	0	0.85	Yes	0.67
	~ H + Treatment + PC1	87.5	0.06	0.308	0	0.99	Yes	0.61
	~ H + Treatment + PC1 + PC2	88.3	0.94	0.198	0	0.88	Yes	0.66
	~ H + Treatment	88.6	1.17	0.177	0	1.20	Yes	0.53
Full models	Species Richness ~ H + LAeq + Treatment + PC1 + PC2 + PC3				0	29.67	Yes	0.65
	Simpson`s diversity ~ H + LAeq + Treatment + PC1 + PC2 + PC3				0	33.54	Yes	0.57
	Phylogenetic diversity ~ H + LAeq + Treatment + PC1 + PC2 + PC3				1332	28109	No	0.45
	Av. Phylogenetic diversity ~ H + LAeq + Treatment + PC1 + PC2 + PC3				0	0.76	Yes	0.71

Table 2. Parameter estimates resulting from model averaging for each response variable. SE, standard error; CI, confidence interval. Confidence intervals that do not include zero are marked in bold. (-) The variable was not present in the model.

Predictor variable	**Species Richness**			**Inverse Simpson's diversity**			**Phylogenetic diversity (PD)**			**Average Phylogenetic diversity (avPD)**		
	Coef.	**SE**	**CI**	**Coef.**	**SE**	**CI**	**Coef.**	**SE**	**CI**	**Coef.**	**SE**	**CI**
H	76.21	38.49	-3.1 | 155.5	102.9	48.44	**4.5 | 201.4**	-	-	-	-30.58	5.67	**-42.4 | -18.8**
LAeq	-0.81	0.63	-2.1 | 0.5	-	-	-	-48.97	18.91	**-88.32 |-9.63**	-0.21	0.10	-0.4 | 0.01
Treatment (mined)	6.73	3.64	-0.9 | 14.4	10.62	3.85	**2.6 | 18.6**	-	-	-	-2.89	0.71	**-4.4 | -1.4**
PC1	-	-	-	-	-	-	-	-	-	0.34	0.14	**0.04 | 0.6**
PC2	-2.88	0.99	**-4.9 | -0.8**	-2.46	0.99	**-4.5 | -0.4**	-60.06	26.16	**-114.63 | -5.49**	0.26	0.15	-0.04 | 0.6
PC3	1.74	1.04	-0.4 | 3.9	-	-	-	-	-	-	-	-	-

DISCUSSION

Our study revealed that the acoustic integrity of the Carajás National Forest influences the taxonomic and phylogenetic diversity of birds. We found that acoustic degradation negatively affected taxonomic diversity when measured using Inverse Simpson's index and is associated with lower values of entropy (H). We also found that the acoustic degradation decreased phylogenetic diversity since sites, with an inverse relationship between LAeq and phylogenetic diversity. However, the acoustic degradation increased average phylogenetic diversity. Although this may seem counterintuitive at first, these opposite effects between phylogenetic diversity and average phylogenetic diversity suggest that mining noise acts as an environmental filter for species with similar evolutionary histories. Moreover, mining noise did not affect species richness. Lastly, vegetation structure, as expected, also influenced bird diversity.

We found that acoustic integrity positively influenced inverse Simpson's diversity. Species that were already frequent can increase their frequency in sites with high acoustic integrity (i.e., no mining noise disturbance). Current evidence indicates that anthropogenic noise can be an invisible source of habitat degradation (Ware et al., 2015), decreasing species richness and abundance. Roadway traffic noise caused a decline in bird abundance of around 25% and avoidance by some species (McClure et al., 2013). A similar result was found by Arevalo and Newhard (2011). Bird assemblages close to stone crushers had lower species diversity and population density (Saha & Padhy, 2011). Our results show that N5 has the highest acoustic degradation, and about 84 species present in unmined areas were not found in N5. Anthropogenic noise can impair important biological interactions, such as territorial defence (Kleist et al., 2016) and pairing success (Habib et al., 2007). For a species to persist close to a mine site, it must tolerate an acoustically degraded environment. However, even though some species tolerate acoustically degraded environments, it may result in fitness costs (Strasser & Heath, 2013). For instance, migratory species during migratory stopover showed a lower body condition and stopover efficiency in areas under roadway traffic noise (Ware et al., 2015).

According to our results, there is a lower phylogenetic diversity in areas with low acoustic integrity. This result suggests that the acoustic degradation may be influencing the structure of bird assemblages in FLONA Carajás. The acoustic degradation might be affecting specific bird groups, most likely those species with lower-frequency vocalizations that overlaps with mining noise. Francis et al. (2011) showed that noise could indeed filter bird communities based on their vocal features, which in turn are linked to their evolutionary history (Price & Lanyon, 2002). We have acoustic data for 264 of the 287 species in this study (in prep. by the authors). Of these, 11 species that use low frequencies (up to 570 Hz) were not found in N5, such *as Crax fasciolata, Thamnophilus amazonicus, Thamnophilus schistaceus, Thamnophilus aethiops*, *Nyctibius aethereus*, *Nyctibius griseus*, *Patagioenas speciosa, Geotrygon montana*, and a few others. There is a negative relationship between noise sensitivity and vocal frequency (Francis, 2015), and the stressful conditions generated by noise can make the most sensitive species evade and avoid noisy environments.

Species that remained in noisy areas have on average, a higher phylogenetic value. The loss of related species and/or recent species - which will have lower phylogenetic value in a community - may explain why the average phylogenetic diversity per species increased in noisy areas while the phylogenetic diversity decreased. Closely related species tend to have similar ecological attributes (Cavender-Bares et al., 2009), including song features (Price & Lanyon, 2002). For example, from the 11 species using low frequencies not found in N5, there are three main groups including seven of these, three from the genus *Thamnophilus*, two from the family Columbidae, and two from the genus *Nyctibius*. Another possibility, which can be happening simultaneously or not, is that the composition of the remaining species of our mined sites may have changed, increasing the average phylogenetic diversity per species. Saha and Padhy (2011) observed that bird communities near a stone mining operation not only had fewer species, but species composition also differed according to the severity of the impact between the sites. Opportunistic species can benefit from habitat degradation (Méndez et al., 2020) caused by mining, and species that can tolerate some level of human disturbance can be found in acoustically degraded environments (Cooke et al., 2020). Higher frequency signals are still effective in noisy environments, thus allowing such species to be present (Slabbekoorn & Peet, 2003). There were 33 species registered in N5 that were not found in non-mined sites, and most of them can easily be found on urban parks and even perched on street power lines, such as *Rupornis magnirostris, Pitangus sulphuratus, Megarynchus pitangua, Tachornis squamata, Zonotrichia capensis, Camptostoma obsoletum,* and many others.

Vegetation structure affected taxonomic and phylogenetic diversity, especially PC2 which is negatively associated with canopy height and ant/termite mounds. This result is not surprising since the influence of vegetation structure on niche availability imposes limits on the number of species supported by the habitat (Stein et al., 2014; Willrich et al., 2019). Moreover, our mined sites varied greatly with PC2 (Fig. 4), and unmined sites had overlapping values for PC2, reinforcing the influence of vegetation structure on bird communities. Therefore, it is important to evaluate other factors that affect the distribution and abundance of birds in conjunction with anthropogenic noise.

Other common impacts associated with industrial activities, such as mining, could influence diversity. We addressed that edge effects, hunting, the capture of wild animals, deforestation, and inadequate disposal of pollutants, would not influence our results. Although there is no visible accumulation of dust on the plant surfaces of the sampled areas, we do not assume the absence of impacts related to air and water quality or its influence over birds.

Our findings provide evidence that acoustic degradation can influence birds' taxonomic and phylogenetic diversity, along with vegetation structure. Likely, other specific impairments are also occurring, such as decreases in species abundance and fitness of individuals that remain in noisy areas (Ware et al., 2015), and further studies are necessary to identify these impacts. Nevertheless, our results can support management decisions regarding industrial developments. The successful insertion of species into the acoustic space is the result of a long evolutionary process (Naguib & Haven-Wiley, 2001; Torres et al., 2017, 2020), and this invisible source of habitat degradation may have implications on ecosystem functioning. Acoustic degradation will become more prevalent as it spreads worldwide, so it is crucial to assess how biological diversity responds to noise to implement adequate management practices. As sound decays as it moves away from its source, it may be appropriate to establish a minimum distance between protected areas and new industrial developments prior to their construction (Cooke et al., 2020; Strasser & Heath, 2013). For established industrial and urban activities, especially those with high background noise levels and low acoustic entropy, regulations and/or economic incentives will be needed. For example, the schedule of mining activities can be determined to avoid periods of the day with increased bird activity or even be shut down during the reproductive season. Also, more research is needed to determine the potential use of natural barriers to reduce noise propagation.

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Supplementary material I

List of species recorded in Carajás National Forest. OBS: 1. Species recorded in control areas (which are not under the influence of mining noise) that were not found in the noisiest mining area (N5); 2. Species that use frequencies below 600Hz recorded in control areas that were not recorded in the noisiest mining area (N5); 3. Species recorded in the noisiest area and that were not found in the control areas.

Species	Mg	N4	N5	N8	S11A	Mined	Non-mined	OBS
Aburria cujubi	0	1	1	1	1	2	2	
Amazilia fimbriata	0	0	0	0	1	0	1	1
Amazona farinosa	0	1	0	0	0	1	0	
Amazona ochrocephala	1	1	1	1	0	3	1	
Anabacerthia ruficaudata	1	0	0	0	0	1	0	
Anodorhynchus hyacinthinus	1	1	0	0	0	2	0	
Ara ararauna	0	0	1	0	0	1	0	3
Ara chloropterus	1	1	1	1	1	3	2	
Ara macao	1	1	1	1	1	3	2	
Ara severus	0	0	1	1	0	1	1	
Aratinga jandaya	1	1	1	1	1	3	2	
Arremon taciturnus	1	1	1	1	1	3	2	
Attila spadiceus	1	1	1	1	1	3	2	
Automolus paraensis	1	1	0	1	1	2	2	1
Automolus rufipileatus	1	1	1	1	1	3	2	
Basileuterus culicivorus	0	0	1	1	1	1	2	
Brotogeris chrysoptera	1	1	1	1	1	3	2	
Bucco capensis	1	1	1	1	1	3	2	
Buteo nitidus	1	0	0	0	0	1	0	
Cacicus cela	1	1	0	1	1	2	2	1
Cacicus haemorrhous	1	1	1	1	1	3	2	
Campephilus rubricollis	1	1	1	1	1	3	2	
Camptostoma obsoletum	1	1	1	0	0	3	0	3
Campylorhamphus multostriatus	1	1	0	1	1	2	2	1
Campylorhynchus turdinus	0	1	1	0	1	2	1	
Cathartes melambrotus	0	0	0	1	0	0	1	1
Celeus flavus	1	0	0	0	0	1	0	
Celeus torquatus	1	0	0	1	1	1	2	1
Celeus undatus	1	1	0	0	1	2	1	1
Ceratopipra rubrocapilla	1	1	1	1	1	3	2	

Cercomacra cinerascens	1	1	1	1	1	3	2	
Cercomacra nigrescens	1	1	1	1	1	3	2	
Certhiasomus stictolaemus	1	0	0	0	0	1	0	
Chaetura brachyura	0	0	0	1	1	0	2	1
Chaetura spinicaudus	0	0	1	1	0	1	1	
Chiroxiphia pareola	0	1	1	1	1	2	2	
Chloroceryle inda	0	0	0	0	1	0	1	1
Cissopis leverianus	1	1	1	1	1	3	2	
Cnemotriccus fuscatus	0	0	1	1	0	1	1	
Coereba flaveola	1	1	1	1	1	3	2	
Colonia colonus	1	0	0	0	0	1	0	
Conirostrum speciosum	0	1	0	0	0	1	0	
Conopophaga aurita	1	1	0	1	1	2	2	1
Conopophaga melanogaster	1	1	1	1	1	3	2	
Contopus nigrescens	1	1	1	1	1	3	2	
Coragyps atratus	0	0	1	1	0	1	1	
Cotinga cayana	0	0	0	1	1	0	2	1
Cotinga cotinga	0	0	1	0	0	1	0	3
Crax fasciolata	1	1	0	1	1	2	2	1, 2
Crypturellus cinereus	1	1	1	1	1	3	2	
Crypturellus soui	1	1	1	1	1	3	2	
Crypturellus strigulosus	1	1	1	1	1	3	2	
Crypturellus tataupa	0	0	1	0	0	1	0	3
Crypturellus variegatus	1	1	1	1	1	3	2	
Cyanerpes caeruleus	1	1	0	0	0	2	0	
Cyanerpes cyaneus	1	1	0	0	0	2	0	
Cyanocorax cyanopogon	0	0	1	0	0	1	0	3
Cyanoloxia rothschildii	1	1	1	1	1	3	2	
Cyclarhis gujanensis	0	1	1	1	1	2	2	
Cymbilaimus lineatus	1	1	1	1	1	3	2	
Dacnis cayana	1	1	1	1	1	3	2	
Dacnis lineata	1	1	0	0	1	2	1	1
Dendrexetastes rufigula	1	1	0	1	1	2	2	1
Dendrocincla fuliginosa	1	1	1	1	1	3	2	
Dendrocolaptes picumnus	1	1	0	1	1	2	2	1
Dendrocolaptes retentus	1	1	1	1	1	3	2	
Dendroplex picus	0	1	0	0	0	1	0	
Deroptyus accipitrinus	1	0	0	0	0	1	0	
Dichrozona cincta	0	0	0	1	0	0	1	1
Dryocopus lineatus	1	1	1	1	0	3	1	
Elanoides forficatus	0	0	0	1	0	0	1	1
Epinecrophylla leucophthalma	1	1	1	1	1	3	2	
Epinecrophylla ornata	1	1	1	1	1	3	2	
Euphonia chlorotica	0	1	1	0	1	2	1	

Euphonia chrysopasta	1	0	0	0	1	1	1	1
Euphonia rufiventris	1	1	0	0	1	2	1	1
Euphonia violacea	0	1	1	0	0	2	0	3
Falco rufigularis	1	1	1	1	1	3	2	
Formicarius analis	1	1	1	1	1	3	2	
Formicarius colma	1	1	0	1	1	2	2	1
Formicivora grisea	0	0	1	0	0	1	0	3
Galbula cyanicollis	1	1	1	1	1	3	2	
Galbula dea	1	0	0	1	0	1	1	1
Geotrygon montana	1	0	0	1	1	1	2	1, 2
Glaucidium hardyi	1	1	0	0	1	2	1	1
Glaucis hirsutus	1	1	1	1	1	3	2	
Glyphorynchus spirurus	1	1	1	1	1	3	2	
Grallaria varia	1	1	1	1	1	3	2	
Granatellus pelzelni	1	1	1	1	1	3	2	
Heliothryx auritus	0	1	1	1	1	2	2	
Hemithraupis guira	1	1	1	1	1	3	2	
Hemitriccus griseipectus	0	0	0	1	1	0	2	1
Hemitriccus margaritaceiventer	0	0	1	0	0	1	0	3
Hemitriccus minimus	0	0	0	0	1	0	1	1
Hemitriccus minor	1	1	1	1	1	3	2	
Herpetotheres cachinnans	0	1	1	0	0	2	0	3
Herpsilochmus rufimarginatus	1	1	1	1	1	3	2	
Hydropsalis albicollis	0	0	0	1	0	0	1	1
Hydropsalis nigrescens	0	0	1	1	0	1	1	
Hylexetastes brigidai	1	1	1	1	1	3	2	
Hylocharis cyanus	1	0	0	0	0	1	0	
Hylocharis sapphirina	1	1	0	0	0	2	0	
Hylopezus berlepschi	0	1	0	0	0	1	0	
Hylophilus hypoxanthus	1	1	1	1	1	3	2	
Hylophilus ochraceiceps	1	0	0	1	1	1	2	1
Hylophilus semicinereus	1	1	1	1	1	3	2	
Hylophylax naevius	1	1	0	1	1	2	2	1
Hylophylax punctulatus	0	0	0	0	1	0	1	1
Hypocnemis striata	1	1	1	1	1	3	2	
Ibycter americanus	1	1	1	1	1	3	2	
Icterus cayanensis	0	1	0	0	1	1	1	1
Iodopleura isabellae	1	1	0	1	1	2	2	1
Isleria hauxwelli	1	1	1	1	1	3	2	
Jacamerops aureus	1	1	1	1	1	3	2	
Lamprospiza melanoleuca	1	1	1	1	1	3	2	
Lanio cristatus	1	1	1	1	1	3	2	
Lanio luctuosus	1	1	0	1	1	2	2	1

Lanio versicolor	1	1	0	1	1	2	2	1
Laniocera hypopyrra	1	0	0	1	1	1	2	1
Lathrotriccus euleri	1	1	1	1	1	3	2	
Lepidocolaptes layardi	1	1	1	1	1	3	2	
Lepidothrix iris	1	1	0	1	1	2	2	1
Leptopogon amaurocephalus	1	0	1	1	1	2	2	
Leptotila rufaxilla	1	1	1	1	1	3	2	
Leptotila verreauxi	1	0	1	0	0	2	0	3
Lipaugus vociferans	1	1	1	1	1	3	2	
Lurocalis semitorquatus	1	1	1	1	1	3	2	
Machaeropterus pyrocephalus	0	0	0	0	1	0	1	1
Malacoptila rufa	1	1	1	1	1	3	2	
Megarynchus pitangua	0	1	1	0	0	2	0	3
Megascops choliba	0	0	1	0	0	1	0	3
Megascops usta	1	1	0	1	1	2	2	1
Melanerpes cruentatus	1	1	0	0	0	2	0	
Micrastur mintoni	1	1	0	1	1	2	2	1, 2
Micrastur mirandollei	1	1	1	1	1	3	2	
Micrastur ruficollis	1	1	1	1	1	3	2	
Micrastur semitorquatus	0	1	0	0	0	1	0	
Microcerculus marginatus	1	1	1	1	1	3	2	
Milvago chimachima	0	0	1	0	0	1	0	3
Mionectes macconnelli	0	0	0	0	1	0	1	1
Mionectes oleagineus	1	1	1	1	1	3	2	
Momotus momota	1	1	1	1	1	3	2	
Monasa morphoeus	1	1	1	1	1	3	2	
Myiarchus ferox	1	0	0	0	0	1	0	
Myiarchus tuberculifer	1	1	1	1	1	3	2	
Myiobius barbatus	1	0	0	1	0	1	1	1
Myiopagis caniceps	1	1	1	1	1	3	2	
Myiopagis gaimardii	1	1	1	1	1	3	2	
Myiornis ecaudatus	1	1	1	1	1	3	2	
Myiothlypis flaveola	0	0	1	0	1	1	1	
Myiothlypis mesoleuca	1	0	0	0	1	1	1	1
Myiozetetes cayanensis	0	0	1	0	0	1	0	3
Myrmelastes rufifacies	0	0	0	0	1	0	1	1
Myrmoborus leucophrys	1	0	1	0	1	2	1	
Myrmoborus myotherinus	1	1	1	1	1	3	2	
Myrmornis torquata	0	1	0	1	0	1	1	1
Myrmotherula axillaris	0	1	1	1	1	2	2	
Myrmotherula brachyura	1	1	1	1	1	3	2	
Myrmotherula longipennis	1	1	1	1	1	3	2	
Myrmotherula menetriesii	1	1	1	1	1	3	2	
Nonnula ruficapilla	1	1	1	1	1	3	2	

Notharchus hyperrhynchus	1	1	0	1	0	2	1	1
Notharchus tectus	1	1	1	1	1	3	2	
Nyctibius aethereus	0	0	0	1	0	0	1	1, 2
Nyctibius griseus	0	0	0	0	1	0	1	1, 2
Nyctiphrynus ocellatus	0	1	1	1	1	2	2	
Nystalus torridus	1	1	1	0	0	3	0	3
Odontophorus gujanensis	1	1	1	1	0	3	1	
Onychorhynchus coronatus	0	1	0	0	1	1	1	1
Ornithion inerme	1	1	1	1	1	3	2	
Oxyruncus cristatus	0	1	1	1	1	2	2	
Pachyramphus castaneus	1	1	1	0	1	3	1	
Pachyramphus marginatus	0	0	0	1	0	0	1	1
Pachyramphus minor	0	1	0	1	0	1	1	1
Pachyramphus polychopterus	0	0	1	0	0	1	0	3
Pachyramphus viridis	0	0	1	0	0	1	0	3
Patagioenas cayennensis	0	1	0	0	0	1	0	
Patagioenas plumbea	0	1	0	0	1	1	1	1
Patagioenas speciosa	1	0	0	1	1	1	2	1, 2
Patagioenas subvinacea	1	1	1	1	1	3	2	
Penelope pileata	1	1	1	1	1	3	2	
Phaeomyias murina	0	0	1	0	0	1	0	3
Phaethornis ruber	1	1	1	1	1	3	2	
Phaethornis superciliosus	1	1	1	1	1	3	2	
Pheugopedius coraya	1	1	1	1	1	3	2	
Philydor erythrocercum	1	1	1	0	0	3	0	3
Phlegopsis nigromaculata	1	1	1	1	1	3	2	
Phyllomyias fasciatus	0	0	1	0	0	1	0	3
Piaya cayana	1	1	1	1	1	3	2	
Piculus flavigula	1	1	1	1	1	3	2	
Piculus laemostictus	1	1	1	1	1	3	2	
Piculus leucolaemus	1	1	1	1	1	3	2	
Picumnus aurifrons	1	1	1	1	1	3	2	
Pionites leucogaster	1	0	0	0	0	1	0	
Pionus menstruus	1	1	1	1	1	3	2	
Pipra fasciicauda	1	0	0	0	1	1	1	1
Piprites chloris	1	1	1	1	1	3	2	
Pitangus sulphuratus	0	0	1	0	0	1	0	3
Platyrinchus platyrhynchos	0	0	0	0	1	0	1	1
Platyrinchus saturatus	1	0	0	0	0	1	0	
Poecilotriccus capitalis	1	1	1	0	1	3	1	
Poecilotriccus sylvia	1	0	0	0	0	1	0	
Polioptila paraensis	0	0	0	1	0	0	1	1
Progne chalybea	0	1	1	1	1	2	2	
Psarocolius decumanus	0	1	1	0	0	2	0	3

Psarocolius viridis	1	1	0	0	0	2	0	
Psittacara leucophthalmus	1	1	1	1	1	3	2	
Psophia interjecta	1	1	1	1	0	3	1	
Pteroglossus aracari	1	1	1	0	1	3	1	
Pteroglossus bitorquatus	1	0	0	1	0	1	1	1
Pteroglossus inscriptus	0	1	0	1	0	1	1	1
Pygiptila stellaris	0	0	1	0	0	1	0	3
Pyriglena leuconota	1	1	1	1	1	3	2	
Pyrilia vulturina	1	0	0	0	0	1	0	
Pyrrhura amazonum	1	1	1	1	1	3	2	
Pyrrhura lepida	1	0	0	1	0	1	1	1
Querula purpurata	1	1	0	1	1	2	2	1, 2
Ramphastos tucanus	1	1	1	1	1	3	2	
Ramphastos vitellinus	1	1	0	1	1	2	2	1
Ramphocaenus melanurus	1	1	1	1	1	3	2	
Rhytipterna simplex	1	1	0	1	1	2	2	1
Rupornis magnirostris	1	0	1	0	0	2	0	3
Saltator coerulescens	0	0	1	0	0	1	0	3
Saltator grossus	1	1	1	1	1	3	2	
Saltator maximus	0	1	1	1	1	2	2	
Sarcoramphus papa	0	0	0	0	1	0	1	1
Schiffornis turdina	1	1	1	1	1	3	2	
Sclateria naevia	0	0	0	0	1	0	1	1
Sclerurus caudacutus	0	0	0	0	1	0	1	1
Sclerurus macconnelli	1	1	1	0	1	3	1	
Sclerurus rufigularis	1	1	0	1	1	2	2	1
Selenidera gouldii	1	1	0	1	1	2	2	1, 2
Setophaga pitiayumi	1	1	1	1	0	3	1	
Sittasomus griseicapillus	1	1	1	1	1	3	2	
Spizaetus tyrannus	0	1	0	0	0	1	0	
Synallaxis cherriei	1	1	0	0	1	2	1	1
Synallaxis rutilans	1	1	0	1	1	2	2	1
Synallaxis scutata	0	1	1	0	0	2	0	3
Tachornis squamata	0	0	1	0	0	1	0	3
Tangara cyanicollis	1	0	0	0	1	1	1	1
Tangara episcopus	0	0	1	1	0	1	1	
Tangara gyrola	1	1	0	1	1	2	2	1
Tangara mexicana	1	1	1	0	0	3	0	3
Tangara nigrocincta	1	0	0	1	1	1	2	1
Tangara palmarum	1	1	1	0	1	3	1	
Tangara punctata	1	1	1	1	1	3	2	
Taraba major	0	1	0	0	1	1	1	1
Terenotriccus erythrurus	1	0	1	1	1	2	2	
Thalurania furcata	1	1	1	1	1	3	2	

Thamnomanes caesius	1	1	1	1	1	3	2	
Thamnophilus aethiops	1	1	0	1	1	2	2	1, 2
Thamnophilus amazonicus	0	0	0	1	0	0	1	1, 2
Thamnophilus schistaceus	1	1	0	1	1	2	2	1, 2
Thamnophilus stictocephalus	1	1	1	1	1	3	2	
Tinamus guttatus	1	0	0	0	1	1	1	1
Tinamus tao	1	1	1	0	1	3	1	
Tityra cayana	0	1	0	0	1	1	1	1
Tityra semifasciata	1	1	1	0	1	3	1	
Todirostrum chrysocrotaphum	1	1	1	1	1	3	2	
Tolmomyias assimilis	1	1	1	1	1	3	2	
Tolmomyias flaviventris	0	0	1	0	0	1	0	3
Tolmomyias poliocephalus	1	1	1	0	0	3	0	3
Tolmomyias sulphurescens	1	1	1	1	1	3	2	
Trogon melanurus	1	1	0	1	1	2	2	1
Trogon ramonianus	1	1	0	1	1	2	2	1
Trogon rufus	1	0	0	0	1	1	1	1
Trogon viridis	1	1	1	1	1	3	2	
Turdus albicollis	1	1	0	1	1	2	2	1
Turdus fumigatus	1	0	0	0	0	1	0	
Turdus leucomelas	1	1	1	1	1	3	2	
Tyranneutes stolzmanni	1	0	0	1	1	1	2	1
Tyrannulus elatus	0	0	0	1	1	0	2	1
Tyrannus melancholicus	0	0	1	1	0	1	1	
Vanellus chilensis	0	0	1	0	0	1	0	3
Veniliornis affinis	1	1	1	1	1	3	2	
Veniliornis passerinus	0	0	1	0	0	1	0	3
Vireo chivi	0	0	1	0	1	1	1	
Vireolanius leucotis	1	1	1	1	1	3	2	
Willisornis vidua	1	1	1	1	1	3	2	
Xenops minutus	1	1	1	1	1	3	2	
Xenops rutilans	1	1	1	0	1	3	1	
Xiphocolaptes carajaensis	1	1	0	1	1	2	2	1
Xipholena lamellipennis	1	0	0	1	1	1	2	1
Xiphorhynchus guttatoides	1	1	1	1	1	3	2	
Xiphorhynchus spixii	1	1	1	1	1	3	2	
Zimmerius acer	1	1	1	1	1	3	2	
Zonotrichia capensis	0	0	1	0	0	1	0	3

Supplementary material II

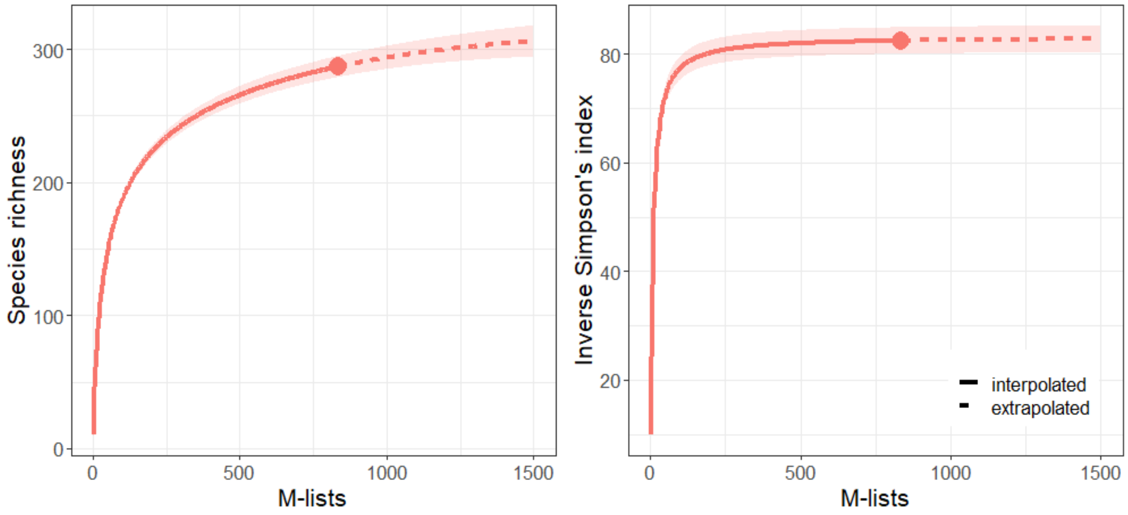

Fig 1. Species estimation curve of species richness (left) and inverse Simpson’s index (right) through interpolation and extrapolation methods.

Supplementary material III

Table 1. Principal Component Analysis loadings of the vegetation structure of the FLONA Carajás sites.

Variable	PC1	PC2	PC3
Exposed rocks	0.000	0.186	**0.571**
Soil cover	**-0.343**	-0.049	**-0.484**
Large trees	0.167	**0.294**	-0.427
Small trees	-0.167	-0.340	-0.346
Lianas	0.188	0.274	-0.272
Standing dead trees	0.356	0.214	-0.172
Fallen trunks	**0.526**	-0.069	-0.118
Ant/termite mounds	0.277	**-0.422**	0.092
Canopy height	0.085	-0.438	-0.080
Palm trees	0.281	-0.387	0.057
Canopy openness	-0.225	0.236	0.038
Glades	0.417	0.244	-0.009

CONSIDERAÇÕES FINAIS

Está cada vez mais claro que o ruído antrópico tem inúmeros efeitos negativos sobre os humanos e animais, especialmente aqueles que dependem da comunicação acústica para desempenhar papéis biológicos essenciais (Levak et al., 2008; Slabbekoorn & Ripmeester, 2008). A perda de biodiversidade é uma preocupação crescente, mas que por vezes é ignorada pelos órgãos governamentais. Se impactos antrópicos mais "comuns", como desmatamentos e queimadas, são notoriamente negligenciados, o que dirá o impacto causado pelo ruído. O ruído antrópico parece estar "silenciosamente" modificando as comunidades naturais, diminuindo o valor do habitat sem mudanças estruturais no ambiente (Ware et al., 2015).

Nossos resultados fornecem algumas evidências de que a degradação acústica tem influência na diversidade de aves, podendo afetar espécies já vulneráveis pelas ações antrópicas e modificar a composição da comunidade, a diversidade taxonômica e filogenética. Ainda assim, é provável que outros impactos específicos estejam ocorrendo, como diminuição na abundância das espécies, aumento do estresse e redução da resposta imunológica, afetando a aptidão dos indivíduos presentes em áreas ruidosas (Brumm, 2013; Brumm & Todt, 2002; Slabbekoorn & Ripmeester, 2008). A inserção das espécies no espaço acústico é fruto de um longo processo evolutivo no qual o papel da competição acústica gerou uma dispersão no uso das frequências para evitar o mascaramento acústico (Torres et al., 2020). O ruído antrópico surge como um competidor imbatível, interrompendo as interações entre as espécies, e potencialmente influenciando muitos organismos direta e indiretamente (Francis et al., 2009). É possível que consequências em cascata para as comunidades naturais estejam ocorrendo, o que pode ter implicações adicionais na estabilidade da dinâmica da comunidade e funcionamento do ecossistema, e mais estudos são necessários para identificar esses impactos.

Nossos resultados podem ser usados para apoiar as decisões de gestão sobre desenvolvimentos industriais. A poluição sonora está se tornando prevalente no mundo e o conhecimento de como as comunidades respondem a esse impacto, pode ser crucial proteger a biodiversidade. A efetividade dos programas de conservação só será alcançada se o problema for enfrentado de forma transdisciplinar (Ferreira, 2014), principalmente em áreas protegidas. Por exemplo, um fragmento florestal com um manejo correto e controle de práticas de caça e desmatamento, pode não estar efetivamente protegido se a matriz deste fragmento for muito ruidosa. Para mitigação do ruído, pode ser apropriado estabelecer uma distância mínima entre áreas protegidas e novos empreendimentos industriais antes de sua construção. Esforços para reduzir os níveis de ruído de áreas urbanas e industriais já estabelecidas, certamente dependerão de incentivos econômicos, pois as medidas de mitigação podem ser dispendiosas.

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