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Smoke pollution disrupted biodiversity during the 2015 El Niño fires in Southeast Asia

Short title: Smoke pollution disrupted biodiversity during the 2015 El Niño

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Abstract

Forest and peatland fires during the 2015 El Niño drought were amongst the worst on record in Southeast Asia. They were a major contributor of carbon emissions across the region, with the associated smoke-induced haze causing an air pollution crisis that affected millions of people. We present evidence of air pollution impacts on biodiversity. Using daily acoustic recordings in central Singapore, we monitored the dawn chorus before, during and after the haze event. We demonstrate that levels of ecological community acoustic activity dropped dramatically during the haze, and that this decline was significantly associated with levels of air pollution considered ‘unhealthy’ to the human population. Acoustic disruption was apparent across four common indices of soundscape activity, with only a partial recovery to pre-haze levels observed 16 weeks after the smoke had dissipated. These impacts on ecological communities were likely to be even more severe closer to the fires, where air pollution levels were reported to be 15-fold greater than those recorded in Singapore. Our results indicate that large-scale air pollution crises may have hitherto underestimated and potentially far-reaching impacts on biodiversity, especially in parts of the world prone to extensive forest fires.

Introduction

Air pollution is increasing globally (World Health Organization 2016). While the human health, stress and mortality impacts associated with exposure to air pollutants, such as sulphur dioxide and particulate matter, are widely acknowledged as significant (World Health Organization 2016; Doherty *et al* 2009), the effects of air pollution on wildlife are much less understood. Previous studies indicate that animals respond negatively to pollution in similar ways to humans in terms of oxidative stress (Isaksson 2010), but it remains unclear whether these impacts translate to lowering of activity or extirpation for particular animal species. This is especially the case for adverse pollution episodes.

Between September and October 2015, much of maritime Southeast Asia was covered with a smoke-induced haze that originated from fires burning in Indonesia’s forests and peatlands.

Although uncontrolled fires occur annually in the region, those in 2015 were exacerbated by a prolonged drought caused by the El Niño–Southern Oscillation and Indian Ocean Dipole (Tacconi 2016, Huijnen *et al* 2016). An average of 11.3 Tg CO_2 was emitted each day during September and October 2015, a figure that exceeded the fossil fuel CO_2 emissions (8.9 Tg CO_2 per day) across the European Union (Huijnen *et al* 2016). The transboundary haze event was so serious that it forced airports and schools to shut in parts of the region, at an estimated economic cost of ~US$16.124 billion for Indonesia alone (World Bank 2016). Throughout the two months, the resulting air pollution regularly reached ‘unhealthy’ or ‘very unhealthy’ levels for humans, as measured by the Pollutant Standards Index (PSI) in Indonesia, Malaysia and Singapore (Tacconi 2016).

Harrison *et al* (2016) reviewed the potential impacts of the fire and resultant haze on Kalimantan’s biodiversity, speculating that the fauna of peat-swamp forests declined due to habitat loss, toxic particulate inhalation, drought and much reduced sunlight, and increased river acidity. To further examine impacts on biodiversity we used the 2015 haze event to examine ‘soundscape’ data from a wildlife-monitoring programme, which recorded acoustic activity each morning on an overpass in Singapore’s central forest block. Singapore is located approximately 300 km away from the locations of the fires, with the prevailing winds blowing the smoke over from Indonesia into the city-state. Acoustic recordings were made for two hours each morning after dawn for three weeks during the haze, and were compared to data available from eight months before the haze began, as well as three and 16 weeks after the smoke had dissipated. For each morning, we calculated acoustic activity using four indices that measure different aspects of the soundscape. The basic premise underpinning the application of acoustic indices is that ecological communities with more vocal species are characterised by greater acoustic diversity, and that biodiversity and ecosystem functioning are positively correlated with acoustic diversity (Gage *et al* 2001, Ji *et al* 2007, Pijanowski *et al* 2011). These validated metrics derived from acoustic monitoring can therefore be used to detect periods of environmental change (Sueur *et al* 2014).

Materials and Methods

Study system

Acoustic data were collected as part of a wildlife-monitoring programme on the “EcoLink” wildlife overpass (1.357°N, 103.784°E) in central Singapore. The overpass is 62 m in length and 50 m in width, and serves to re-connect two tropical lowland rainforest reserves, Bukit Timah Nature Reserve (163 ha) and Central Catchment Nature Reserve (c. 2000 ha); protected areas that have been separated by the Bukit Timah Expressway for 30 years. Construction and planting of native vegetation on the bridge were completed in December 2013. The likely fauna contributing to the soundscape include birds and Orthoptera. At least 55 bird species have been identified from the site via mist-netting (Yifei Chung, unpublished data), and *Nisitrus* sp. crickets and *Conocephalus* sp. katydids have been encountered (Ming Kai Tan, unpublished data). In addition, opportunistic camera trapping has detected macaques (*Macaca fascicularis*) and other small mammals using the overpass.

Monitoring of air pollution

We obtained air pollution data, measured by the Pollutant Standards Index (PSI), from the National Environment Agency (2015) of Singapore. Hourly PSI values are available for five regions on the island (north, south, east, west and central). We used PSI values from central Singapore to best reflect the location of the overpass. The PSI is derived from the average concentrations of six common air pollutants: (i) fine particulate matter ($PM_{2.5}$); (ii) particulate matter (PM_{10}); (iii) sulphur dioxide; (iv) carbon monoxide; (v) ozone; and, (vi) nitrogen dioxide. PSI values up to 50 are considered ‘good’ in relation to the human population; 51-100 ‘moderate’; 101-200 ‘unhealthy’; 201-300 ‘very unhealthy’; and, >300 as ‘hazardous’. During the peak of the haze event, PSI values reached 267, and residents were advised by the Singapore Government to avoid prolonged or strenuous outdoor physical exertion.

We compiled PSI values for our four acoustic monitoring sample periods in relation to the haze event: ‘before’; ‘during’; ‘3 weeks after’; and, ‘16 weeks after’ (Table 1). These periods were identified according to distinct shifts in PSI values and public health advisory announcements on air quality, as well as availability of quality recordings. As an equatorial island, Singapore is largely aseasonal, although it tends to experience two monsoons: December to March, and June to September. By sampling 16 weeks after the event, we were able to quantify acoustic activity mid-way through the northeast monsoon in a similar period sampled before the onset of haze. We used the 0800 hrs PSI value for each sampling day as this represented the midpoint of our two-hour morning recordings.

Table 1. Summary of the acoustic monitoring sample periods used to assess the impact of the 2015 haze event in central Singapore on ecological communities.

Sample periods	Dates	Number of days	Pollutant Standards Index (PSI) range
Before	11 Jan – 1 Feb 2015	22	31-68
During	24 Sep – 28 Oct 2015	21	97-267
3 week after	16 Nov – 14 Dec 2015	20	28-53
16 weeks after	29 Feb – 14 Mar 2016	15	37-55

Acoustic recording, processing and statistical analyses

We deployed a Song Meter SM2BAT+ unit (Wildlife Acoustics Inc., USA) at the centre of the overpass with a SMX-II weatherproof microphone attached to a 1.8-metre aluminium pole. The SMX-II is an omnidirectional and audio broadband microphone recording frequencies of up to 48,000 Hz, and with an estimated range of ca. 60 m radius. Our acoustic data therefore covered the overpass, as well as ca. 30 m of forest either side. The recordings comprised both ambient sounds from the ecological community, as well as non-biological phenomena, including those arising from human activities (Pijanowski *et al* 2011). The pole extended the microphone above vegetation to provide a clear recording space without obstruction from foliage or branches. The microphone was checked and tested regularly, with the foam windscreen changed after sustained periods of rain.

The Song Meter unit was configured to record in an uncompressed .wav format at a sampling rate of 24,000 Hz and 16 bits to target audible sound (i.e. up to ca. 11,025 Hz, the highest frequency of bird song in the region). A high pass-filter was applied at 180 Hz to attenuate excessive low-frequency sounds so that bird vocalisations could be recorded adequately. The device was placed in a locked box, from which the microphone extended, and secured to a tree trunk. Recordings were scheduled for between 0700 and 0900 hrs each day to capture the dawn chorus comprising bird vocalisations and insect calls. We checked all recordings within each file to assess levels of rain. Any recordings from mornings with heavy rain were subsequently excluded from the study, as bird and insect acoustic activity declines in rainy conditions (Robbins 1981) and the detection range of the microphone is reduced when wet. One morning was removed as a result. Each two-hour recording was split into 12 files of 10-minute duration using WavePad v6.37 (NCH Software), and acoustic indices calculated for each 10-minute segment. Sensitivity analyses using segments of various lengths (10, 30 and 60-minute) for all four soundscape indices revealed that 10-minute segments gave sufficient amount of variation across the duration of recordings for statistical comparisons to be made (Supporting Information Fig. S1).

Calculation of acoustic indices

We generated four acoustic indices for the Singapore soundscape recordings:

- The Bioacoustic (BA) index is designed to assess the relative abundance and composition of bird communities, and is a function of the acoustic level and the number of frequency bands used by animals (Boelman *et al* 2007). The index is derived from the area under each curve in all frequency bands, and so the area values are a function of both activity level and the number of frequency bands used by vocalising species, but in particular, avifauna. Here, calculations targeted the ecological frequency range of 2000-11,025 Hz (anthropogenic sound is typically <2000 Hz).

- Acoustic Complexity (AC) directly quantifies acoustic intricacy by calculating the variability of recording intensities, despite the presence of constant anthrophony (human-generated noise) in the background (Pieretti *et al* 2011, Farina and Pieretti 2014). The index correlates well with the number of bird calls in a community (Pieretti *et al* 2011), has been used to describe avian soundscapes (Farina *et al* 2011), to explore the association between these soundscapes and vegetation complexity (Farina and Pieretti 2014), and to investigate the influence of traffic noise on the singing of a bird community (Pieretti and Farina 2013).

- Acoustic Diversity (AD) is typically used as a direct proxy for species diversity, since it quantifies the number of unique sounds at unique frequencies, and these different sounds are largely emitted by different taxa (Villanueva-Rivera *et al* 2011). It is based on the Shannon index (Villanueva-Rivera *et al* 2011), and is derived by dividing the frequency spectrogram into bins and quantifying the proportion of sounds in each bin above a volume threshold (Sueur *et al* 2014, Villanueva-Rivera *et al* 2011). We applied this index to 1000 Hz bins.

- The Normalised Difference Soundscape Index (NDSI) reflects the relative contribution of ecological (biophonic) and anthrophonic sounds to the soundscape. The calculation involves segregating the spectral profile into two main frequency bands comprising anthrophonic (1000-2000 Hz) and ecological (2000-11,025 Hz) components (Ji *et al* 2007), before calculating the ratio of the two. The value therefore ranges from -1 to +1, with low and high values indicating the prevalence of anthrophony or biophony respectively.

Acoustic data were calculated in R version 3.2.4 using the ‘multiple_sounds’ function in the package ‘soundecology’ (Villanueva-Rivera and Pijanowski 2013), as well as the packages ‘seewave’, ‘ineq’, ‘tuneR’ and ‘vegan’ (using signal settings in S1 Table). Mean values of each index were calculated across the twelve 10-minute sound files recorded each morning, thereby producing a single average value per morning. The distribution of acoustic data across each sample period was checked for homoscedasticity (Levene's test), visualised using the R package 'ggplot2' to generate violin plots, and the residuals checked for normality. The variance in mean acoustic data for each index was unequal between sample periods. Kruskal-Wallis tests were thus used to determine if the indices varied significantly between sample periods, and pairwise Mann-Whitney tests were applied post-hoc to determine where differences occurred. A standard Bonferroni correction was applied to the Mann-Whitney tests to control for Type 1 error due to multiple testing. General linear models were then used to determine whether variation in each acoustic index across the entire study period could be predicted by PSI. Since samples collected after the haze were clearly not independent from those collected before or during the event, we report regressions restricted to the 'before' and 'during' data. However, our findings remain consistent even when all data are included in the models.

Results

Recordings were available for a total of 78 mornings between January 2015 and March 2016. Our analyses reveal that daily averages for all four acoustic indices corresponded clearly and significantly to the onset and, to some degree, the passing of the air pollution crisis (Fig. 1). The greatest acoustic activity was pre-haze. Subsequently, average levels of acoustic activity across all indices declined substantially (by up to 37.5%) during the haze event, a disruption that lasted for at least three weeks. A near complete recovery to pre-haze levels was evident after 16 weeks for the acoustic complexity and bioacoustic indices, two well-established indicators of bird vocalisation activity (Pijanowski *et al* 2011, Towsey *et al* 2014). However, acoustic diversity and normalized difference soundscape index, indices that reflect broader patterns of ecological activity relative to anthropogenic sounds (Gage *et al* 2001, Sueur *et al* 2008, Pijanowski *et al* 2011, Fuller *et al* 2015), remained low after 16 weeks. This suggests that some components of the ecological community continued to be absent or torpid for at least four months after the smoke dissipated. All acoustic indices correlated negatively and significantly with the average daily PSI readings for central

Singapore, and were particularly low when air pollution reached levels deemed unhealthy for humans (>100 PSI; Fig 1).

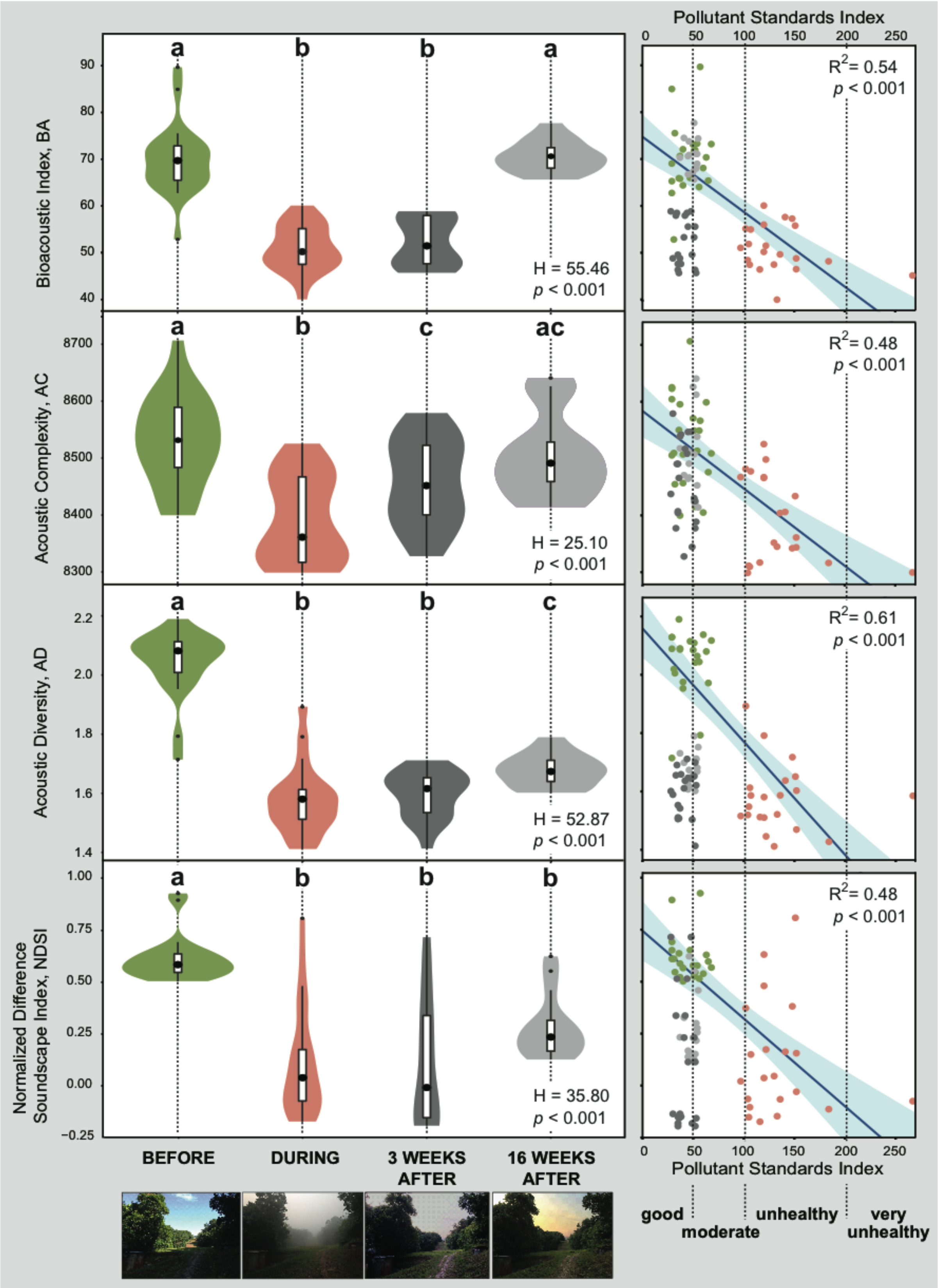


Figure 1. Daily variation in dawn chorus acoustic indices, and their association with air pollution levels, in Singapore before, during and after the 2015 El Niño forest and peatland fire smoke-induced haze event. Left panels: violin plots show variation (median, range, kernel

density; 25th–75th percentiles) in daily measures of dawn chorus acoustic activity ‘before’ (22 sample days), ‘during’ (21 sample days) and ‘after’ (20 and 15 sample days at 3 and 16 weeks respectively). Different letters denote significant differences between sample periods, based on Kruskal-Wallis and post-hoc Mann-Whitney tests. Right panels: regressions (+/- 95% CI, indicated by shaded area around line of best fit) of daily average acoustic indices against Pollutant Standards Index (PSI) values. Due to data non-independence issues associated with the two ‘after’ sample periods, the regressions are calculated from data ‘before’ and ‘during’ the haze event. Dashed lines show the public health advisory PSI thresholds.

Discussion

We provide a unique evaluation of the effects of the 2015 Southeast Asia forest and peatland fires on biodiversity, demonstrating that the transboundary haze crisis had a clear negative impact on the ecological community in Singapore. To the best of our knowledge, no other study has investigated the impacts of air pollution on biodiversity in this way. Previous studies have focused on the autecological responses of particular species to air pollution (e.g. great tits, *Parus major* (Gorissen *et al* 2005), yet acoustic indices give us a valuable and, importantly, rapid insight into the entire audible ecological community, which is not restricted to individual species.

Both the bioacoustic and acoustic complexity indices we used capture mainly bird vocalisations, with the former index being a reliable indicator of avian abundance (Boelman *et al* 2007) and the latter serving as a proxy for singing activity (Pieretti *et al* 2011). Our findings therefore imply that while bird abundance and associated activity were substantially impacted by the pollution event, both recovered within four months post-haze (Fig. 1). Conversely, the acoustic diversity index and normalised difference soundscape index showed little sign of recovery over this time period. In biodiversity assessments, acoustic diversity is renowned for being a reliable surrogate for overall (vocal) faunal richness, because it quantifies the number of unique sounds made at different acoustic frequencies, which are typically made by different species (Villanueva-Rivera *et al* 2011, Depraetere *et al* 2012). Low acoustic diversity values therefore indicate that sound is being recorded at fewer frequencies across all sound emitting taxa. The drop in NDSI, an index characteristically high and stable in undisturbed environments (Gage *et al* 2001), also reflects this trend.

Potential causes and mechanisms of acoustic activity change

There are several possible explanations for our findings, which may not be mutually exclusive. First, the drop in acoustic activity might signal a temporary cessation in ecological activity during the dawn chorus. For example, in temperate regions the singing behaviour of male great tits is known to be reduced and less complex at sites near to heavy metal air pollution sources (Gorissen *et al* 2005). In nearby Borneo, gibbons are reported to sing less during periods of poor air quality (Cheyne 2008). However, this seems unlikely to be the sole explanation of the soundscape patterns observed, given that acoustic recovery back to pre-haze levels was only demonstrated by two of the four indices. Second, smoke-induced air pollution may have caused direct mortality in key components of the ecological community. Although understudied, terrestrial vertebrates are predicted to suffer from the same direct effects of air pollution as humans, including respiratory diseases, hypoxia, irritation of the eyes and skin, increased stress and death (Isaksson 2010, Kimura *et al* 1988). During the 1997 El Niño, 90% of people in Indonesian villages exposed to similar levels of pollution as Singapore in 2015 had severe respiratory problems from inhaling toxic smoke (Kunii *et al* 2002). Third, mortality might have arisen indirectly via reduced fitness or foraging success, since airborne particles drastically increases atmospheric attenuation of light and sound (Hardy *et al* 2001, Pye 1971), and thus compromises the ability of animals to forage. This is particularly likely to be the case for flying and site-dependent species such as birds. Fourth, haze could reduce the abundance/size of prey species or affect plant phenology (see Harrison *et al* 2016), thereby driving bottom-up trophic cascades (Zvereva and Kozlov 2010). Fifth, there is scope for some seasonal variation in ecological activity to be occurring in the dataset although, given the highly aseasonal nature of Singapore and the scheduling of our sample periods, this is unlikely to have contributed to the observed trends. Finally, it is possible that our findings reflect a postponement of ecological activity to outside of the two-hour window we recorded at dawn. Delaying activity to sub-optimal times in this way would still be expected to have long-term repercussions for individual fitness and phenology (McNamara *et al* 2011). Nevertheless, this hypothesis is not well supported by anecdotal observations during the haze (i.e. forests were uncharacteristically quiet).

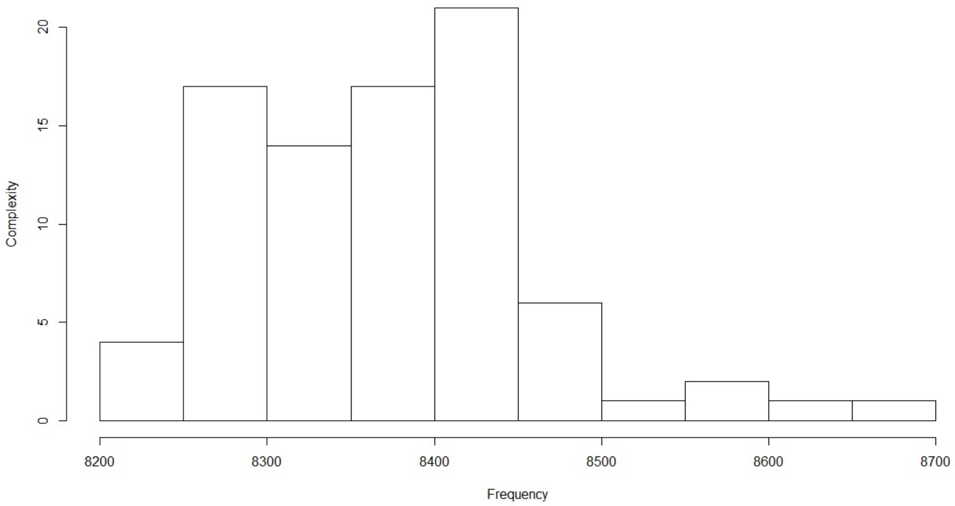
Conclusion

While we can only speculate about the precise mechanisms underpinning the reduction in acoustic activity during the 2015 haze crisis, the trends observed in Singapore could have far reaching implications for biodiversity if realised across pollution-stricken Asia. Ecological disruption across all taxa is likely to be greatest nearer to smoke sources. For example, PSI readings exceeded 2,000 in Indonesian Borneo during the 2015 fires (Chan 2015), which is 15-times greater than the levels recorded in Singapore. Without a major concerted international effort to tackle the causes of forest and peatland fires, annual air pollution events are expected to continue and worsen in Southeast Asia (Tacconi 2016, Huijnen *et al* 2016). Indeed, high levels of transboundary air pollution are not restricted to drought years and, in fact, occur almost annually (Gaveau *et al* 2014). The long-term prognosis for the region's imperilled biodiversity is therefore poor, particularly in burnt areas, which already support fewer taxa and depauperate wildlife populations compared to forests elsewhere (e.g. Cleary *et al* 2006). Further research into the effects of air pollution on tropical biodiversity is thus warranted.

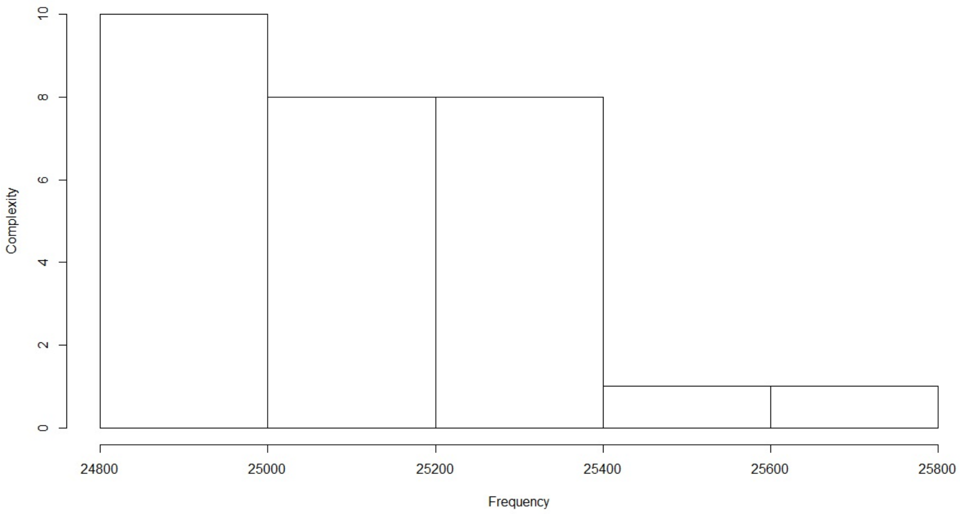
The funding and human capacity required to effectively document and monitor biodiversity in tropical countries, in addition to significant logistical challenges on-the-ground, are major barriers to large-scale implementation of traditional ecological field surveys (Harrison *et al* 2012). There is therefore a well justified need for rapid biological monitoring techniques, especially in situations in which health and safety is a concern. The capacity of acoustic techniques to yield information on species identity and turnover is currently still limited. However, as technological capabilities improve, it is highly probable that key vocalising taxa, such as those of conservation concern, will be identifiable. Such acoustic techniques therefore offer a practical and relatively cheap alternative to traditional wildlife monitoring techniques, could be safer to implement in fire/smoke affected areas, and provide much-needed data in near real-time. Moreover, acoustic indices are representative of entire ecological communities, rather than just a select few species that are indicators of environmental change. Just as with standard measures of species diversity, these proxies have the potential to be up-scaled over large landscape areas using spatial imagery (Boelman *et al* 2007, Pekin *et al* 2012), which is becoming increasingly available and affordable for tropical countries. Information on the spatial and temporal extent of acoustic disruption, and how well this corresponds to observed changes in ecological communities, will be fundamental to understanding the full environmental implications of future air pollution crises.

SUPPLEMENTARY INFORMATION

(a) Acoustic complexity – 10-minute samples

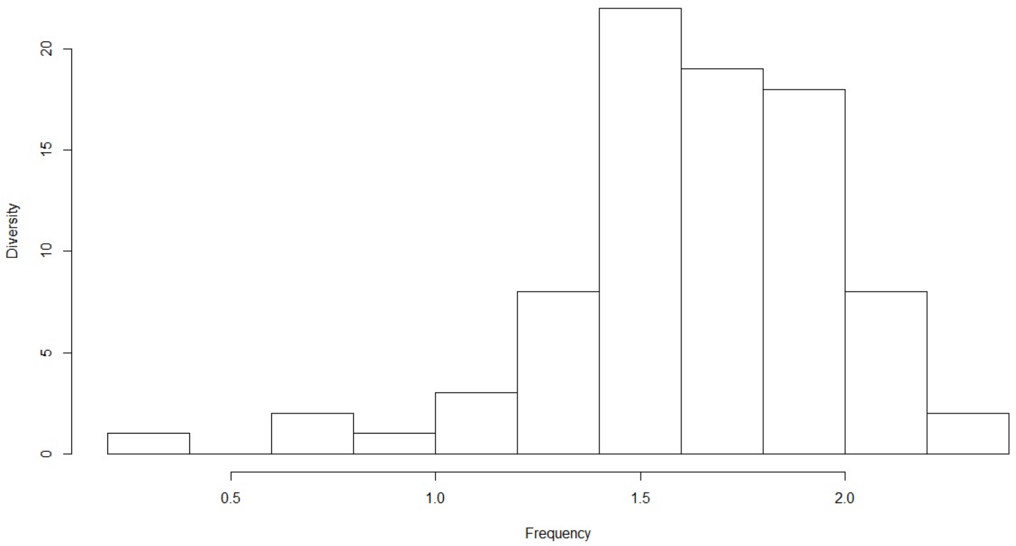


(b) Acoustic complexity – 30-minute samples

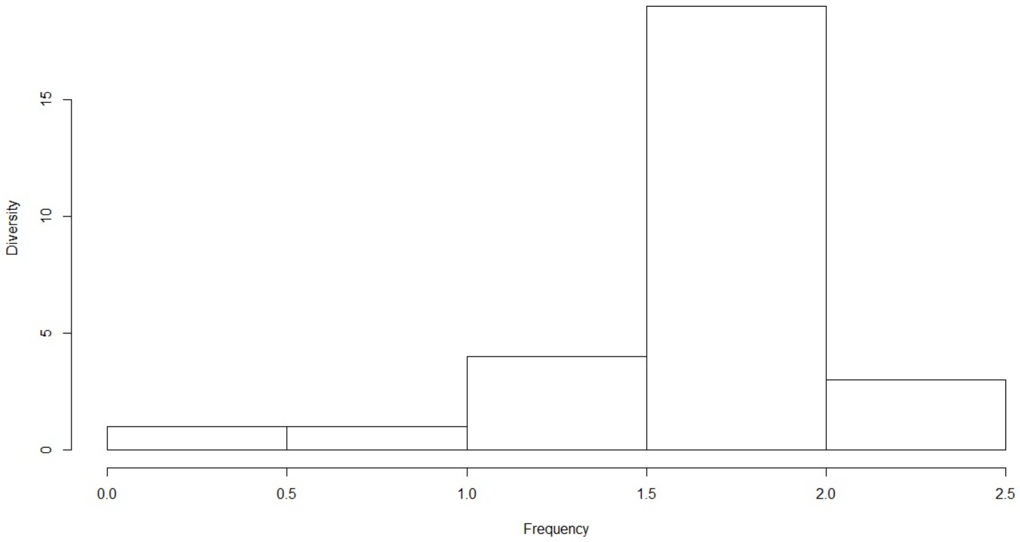


SI Figure 1a-b. Histograms showing variation in frequency of the acoustic complexity index calculated on 10- and 30-minute sound recording segments from Singapore.

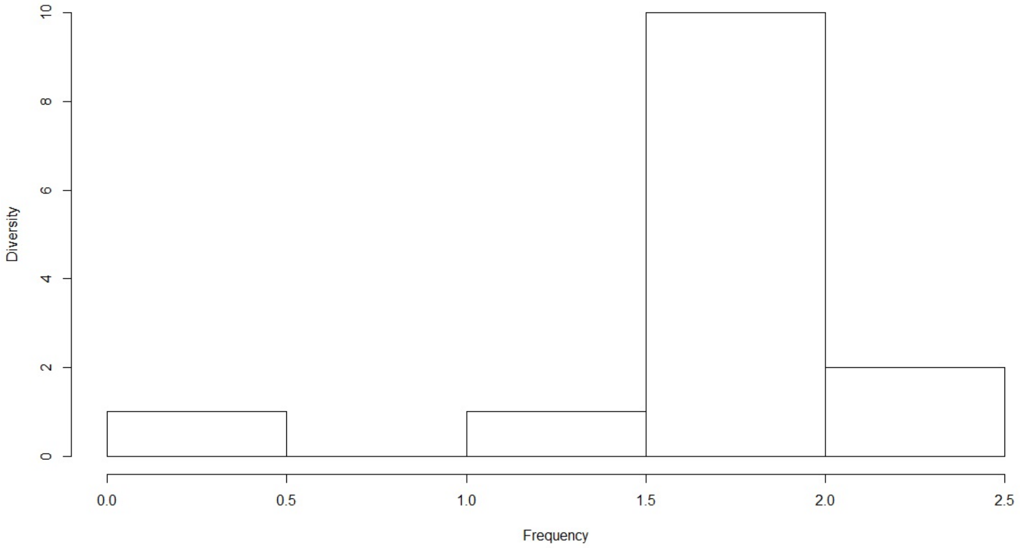
(c) Acoustic diversity – 10-minute samples



(d) Acoustic diversity – 30-minute samples

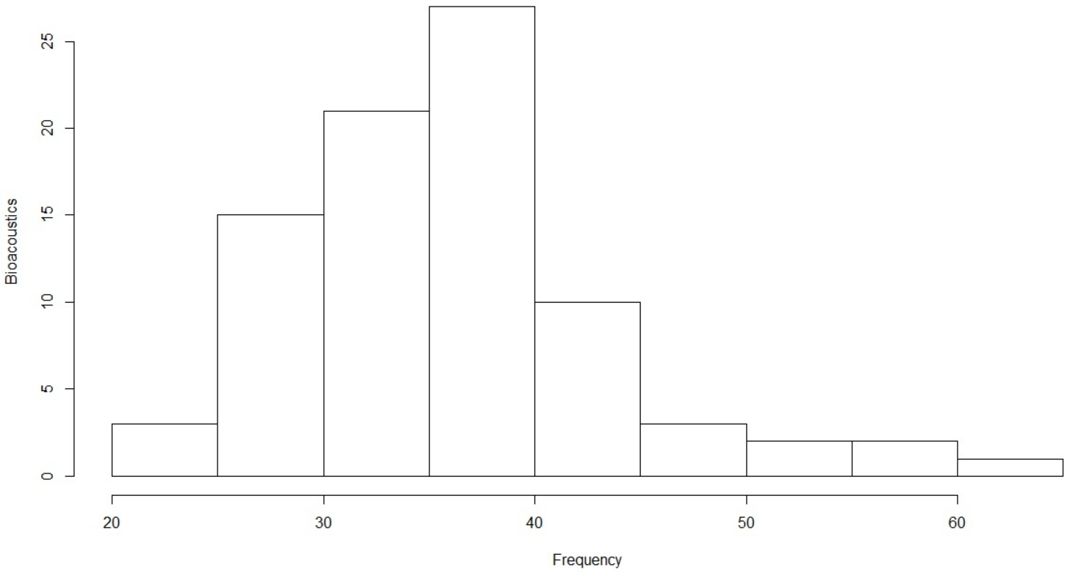


(e) Acoustic diversity – 60-minute samples

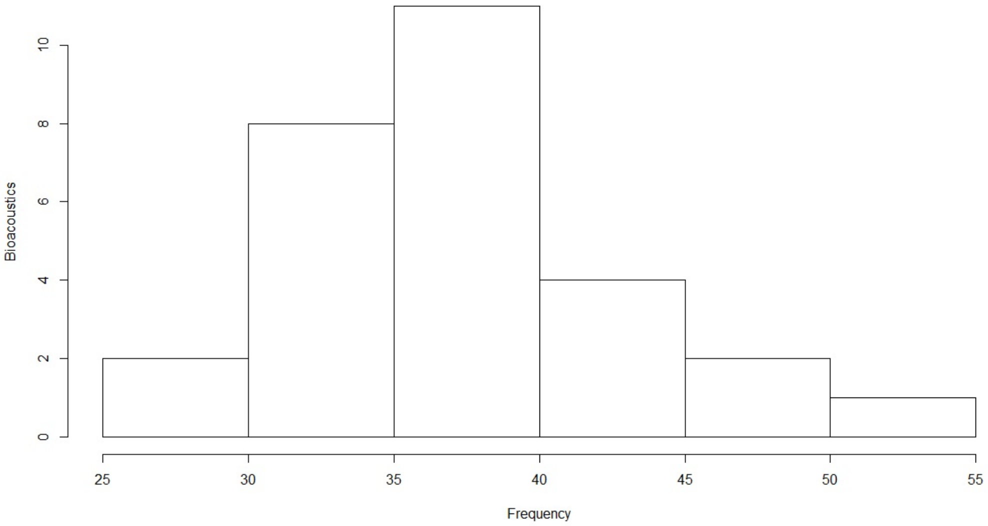


SI Figure 1c-e. Histograms showing variation in frequency of the acoustic diversity index calculated on 10-, 30- and 60-minute sound recording segments from Singapore.

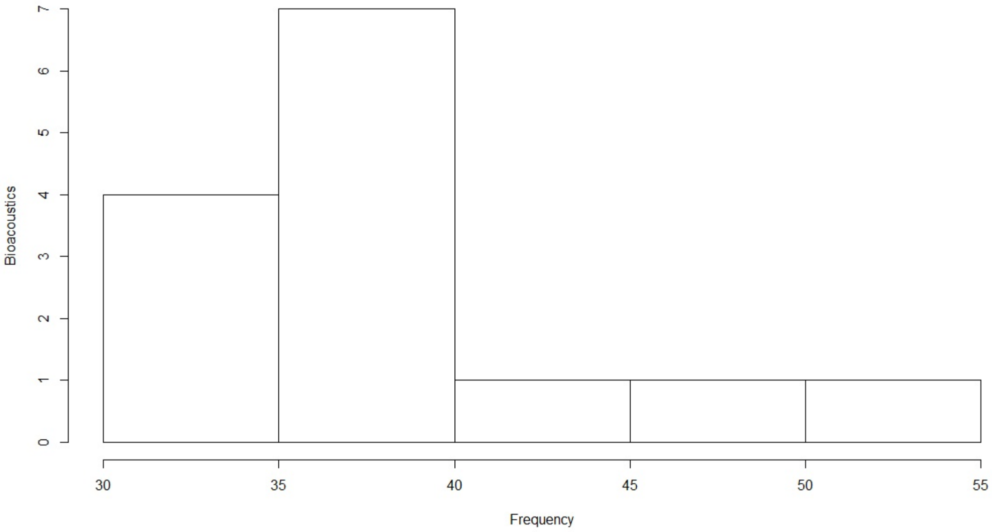
(f) Bioacoustic index– 10-minute samples



(g) Bioacoustic index – 30-minute samples

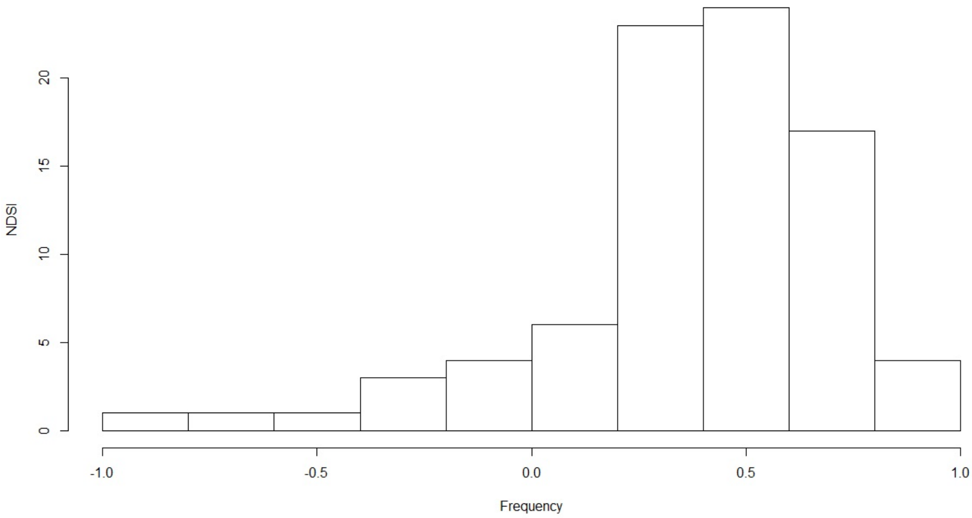


(h) Bioacoustic index – 60-minute samples

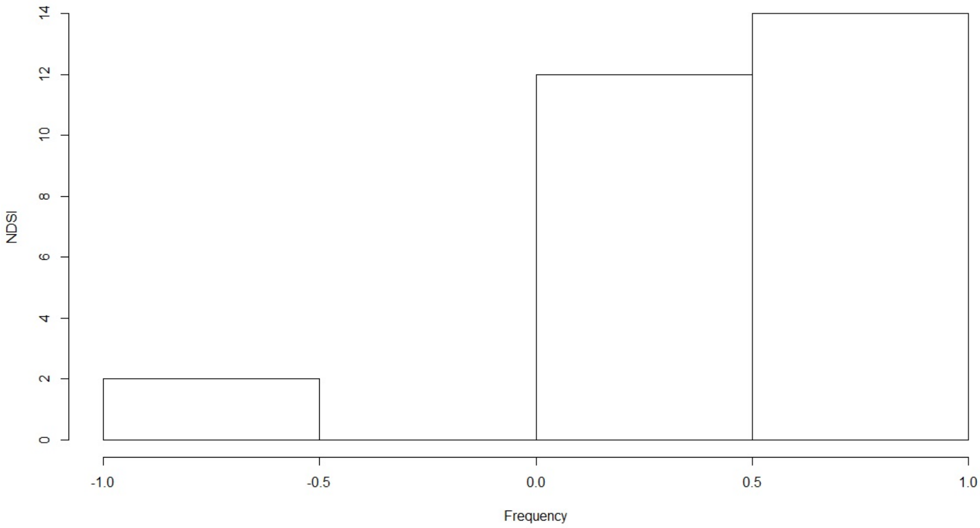


SI Figure 1f-h. Histograms showing variation in frequency of the bioacoustic index calculated on 10-, 30- and 60-munite sound recording segments from Singapore.

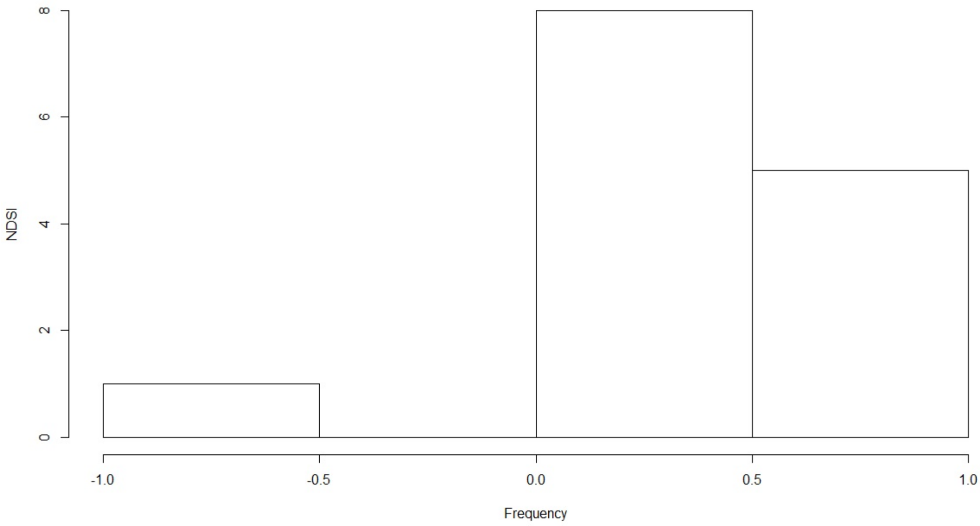
(i) NDSI – 10-minute samples



(j) NDSI – 30-minute samples



(k) NDSI – 60-minute samples



SI Figure 1i-k. Histograms showing variation in frequency of the Normalised Difference Soundscape index calculated on 10-, 30- and 60-munite sound recording segments from Singapore.

Table S1. Settings used for processing the acoustic indices in R. Optimal parameters for each index were determined after sensitivity analyses on a subset of sound recordings.

Index	J (cluster size) (s)	Min freq (Hz)	Max freq (Hz)	dB threshold (Hz)	Freq step (Hz)	Anthro min (Hz)	Anthro max (Hz)	Bio min (Hz)	Bio max (Hz)
Acoustic complexity (AC)	10	—	11025	—	—	—	—	—	—
Acoustic diversity (AD)	—	—	11025	-30	1000	—	—	—	—
Bio-acoustic (BA)	—	2000	11025	—	—	—	—	—	—
Normalised difference soundscape (NDSI)	—	—	—	—	—	1000	2000	2000	11025

J: cluster size in seconds (e.g. a cluster size of 10 s will partition analyses into 10-second clusters); **Min freq:** minimum frequency used when calculating the value, in Hz. Applied only to BA; **Max freq:** maximum frequency used when calculating the value, in Hz. Applied to AC, AD and BA; **dB threshold:** threshold to use in the calculation and a dB of -30 dBFS was chosen as it was the most optimal based on a visual comparison of boxplots and histograms. Applied only to AD; **Freq step:** size of frequency bands. Soundscape studies typically use 1000 Hz bands; **Anthro min and Anthro max:** minimum and maximum values of the range of anthrophony (human-generated sound); **Bio min and Bio max:** minimum and maximum values of the range of biophony (biologically-generated sounds).

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