Multiscale spatio-temporal patterns of boat noise on U.S. Virgin Island coral reefs

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ABSTRACT

Sound-sensitive organisms are abundant on coral reefs. Accordingly, experiments suggest that boat noise could elicit adverse effects on coral reef organisms. Yet, there are few data quantifying boat noise prevalence on coral reefs. We use long-term passive acoustic recordings at nine coral reefs and one sandy comparison site in a marine protected area to quantify spatio-temporal variation in boat noise and its effect on the soundscape. Boat noise was most common at reefs with high coral cover and fish density, and temporal patterns reflected patterns of human activity. Boat noise significantly increased low-frequency sound levels at the monitored sites. With boat noise present, the peak frequencies of the natural soundscape shifted from higher frequencies to the lower frequencies frequently used in fish communication. Taken together, the spectral overlap between boat noise and fish communication and the elevated boat detections on reefs with biological densities raises concern for coral reef organisms.

1. Introduction

Coral reefs host some of the highest diversity of life per unit area on Earth. About one-quarter to one-third of all marine species live in coral reefs (Knowlton et al., 2010; Plaisance et al., 2011; Reaka-Kudla, 1997). Reefs provide essential ecosystem services such as shoreline protection and carbon dioxide uptake (reviewed in Moberg and Folke, 1999). They also provide substantial economic value associated with tourism, fisheries, and the aquarium trade (Moberg and Folke, 1999; Spalding et al., 2017). Yet in recent decades, myriad stressors such as overfishing, ocean warming, disease, and acidification have driven coral reefs into global decline (Hughes et al., 2018; McClenachan et al., 2017; Tsounis and Edmunds, 2017). In addition to these classically studied stressors, there is increasing awareness that the resilience of reefs and basic ecological processes on coral reefs are threatened by anthropogenic noise pollution (e.g. McCormick et al., 2018; Simpson et al., 2016a, 2016b).

Acoustic signals are used by many coral reef organisms because of the efficient propagation of sound in water (Au and Hastings, 2008; Myrberg, 1981). In healthy coral reefs with diverse biological communities, these soniferous organisms create a biophony comprising of fish choruses and the sounds of invertebrates such as hermit crabs and snapping shrimp (Freeman et al., 2014; Kaplan et al., 2015). These animals use sound for ecologically vital behaviors such as larval orientation (Leis et al., 2003; Radford et al., 2011; Simpson et al., 2008), agonistic territoriality (e.g. Herberholz and Schmitz, 1998), and mate attraction (Myrberg et al., 1986).

With this growing awareness of reef acoustic ecology, there is a corresponding understanding that noise from vessels may impact key behaviors and have substantial physiological effects on coral reef organisms (Slabbekoorn et al., 2010). For example, acoustic masking occurs when the presence of one noise increases the detection threshold of another (Clark et al., 2009). In coral reefs, boat noise masks acoustic cues and disrupts orientation behavior in settlement-stage coral reef fish larvae (Holles et al., 2013; Simpson et al., 2016a). Furthermore, evidence suggests that noise from small motor vessels induces physiological stress responses in coral reef fishes. For instance, exposure to boat noise was associated with an increase in metabolic rate in Ambon damselfish (Simpson et al., 2016b) and an increase in heart rate of staghorn damselfish embryos (Jain-Schlaepfer et al., 2018).

Perhaps most alarmingly, acoustic disturbance from boats may impose direct consequences on individual fitness and induce mortality in certain coral reef organisms. For instance, in situ playback of boat noise has been shown to increase embryonic mortality in sea hares and

increase predation rates of settlement-stage Ambon damselfish (Nedelec et al., 2014; Simpson et al., 2016b). Recent evidence suggests that boat noise could also have more indirect and latent consequences on individual fitness. For example, boat noise has been shown to decrease offspring fitness by impairing parental care in a brooding reef fish, the spiny chromis (Nedelec et al., 2017). Exposure to outboard motor noise during critical learning periods has also been shown to impair predator-learning behavior and impact subsequent responses to predators in juvenile Ambon damselfish (Ferrari et al., 2018).

While several studies have suggested that boat noise may affect essential biological functions in coral reef organisms, little is known about the pervasiveness of this stressor. The actual levels, occurrence rates, and reef-based variation of boat noise in coral reefs are rarely reported. Several studies have suggested that passive acoustic recordings are a useful means to monitor and quantify boat activity, but this has been on a limited scale (just a few reefs and relatively short time-frames) (Kaplan and Mooney, 2015; Lammers et al., 2008). Furthermore, while noise predictions and propagation studies have long been conducted in open ocean environments (e.g., cetsound.org) (Wenz, 1962), the physically complex shallow water environments make noise propagation modeling and predicting noise levels on reefs more challenging, thereby limiting noise predictions that are comparable to open ocean environments. Thus, multiple in situ noise measurements become vital as we seek to understand acoustic patterns and pervasiveness of noise pollution on coral reefs.

Here, we utilized passive acoustic monitoring to quantify boat activity at nine coral reefs and one sandy shore comparison site off the island of St. John, U.S. Virgin Islands (USVI) over the course of one year. The coral reef sites are assigned the following numerical codes by which they will be referred hereafter, with site number increasing from west to east: Dittlif Point (1), Cocoloba Cay (2), Joel's Shoal (3), White Point (4), Europa Bay (5), Tektite (6), Yawzi Point (7), Booby Rock (8), and Ram Head (9) (Fig. 1; Table 1). Reef Bay is an off-reef sandy bottom comparison site and will be referred to as Site S. St. John is home to the USVI National Park, a popular tourist destination, and many visitors charter small boats to visit local coral reefs. We described the spatio-temporal patterns of boat noise at each site on diel, weekly, and seasonal scales. By comparing the relative prevalence of boat noise at each site to surveys of benthic cover and fish communities, we sought to better understand the potential drivers of boat noise. Furthermore, we evaluated the effects of boat noise on the local soundscape by calculating the sound pressure level (SPL) in two separate acoustic bands: one predominantly used by fish, the other dominated by snapping shrimp. We quantified the peak acoustic frequency defined by the frequency with the highest acoustic power to evaluate how the soundscape changed in the presence of boat noise. By providing occurrence patterns and spectral consequences of boat noise, this study provides valuable baseline data that can inform management, monitoring, and experimental methodology.

2. Methods

2.1. Acoustic data collection

We deployed passive acoustic recorders (SoundTrap ST300, Ocean Instruments NZ, Inc.) at nine shallow coral reef sites and one sandy-shore control site (all 7–12 m depth) along the southern shore of St. John, U.S. Virgin Islands from May 2016–July 2017 (48 kHz sampling frequency) (Fig. 1A). Acoustic recorders were attached ~0.75 m above the seafloor to a rebar stake using hose clamps and cable ties, with the omnidirectional hydrophone facing the water surface (Fig. 1B). Recording units were programmed on a 10% duty cycle and collected one-minute recordings every 10 min. At four times throughout the study

We conducted all analyses in Matlab 9.2 (MathWorks, Inc., Natick, MA). We calibrated recordings according to each hydrophone's sensitivity which was provided by the manufacturer. Spectrograms for each 1-minute recording were generated using a 16,384-point FFT in 0.25-second windows with 25% overlap between contiguous windows. The average power spectrum for each recording was estimated using Welch's method (Hanning window, non-overlapping 0.5-second averages) (Welch, 1967).

We visually inspected spectrograms and average power spectra to identify recordings containing boat noise. Visual identification was based on broadband high intensity sound levels (Kaplan and Mooney, 2015; Kaplan et al., 2016). Boat noise usually caused substantial changes to soundscapes and were easily detectable by visual inspection of spectrograms (Fig. 2). If visual identification was ambiguous, we aurally audited to confirm or reject the presence of boat noise. We binned each recording by hour, day, and month, and we calculated the net percentage of recordings containing boat noise for each site within these bins.

For each 1-minute audio file, we calculated the low-frequency root-mean-square sound pressure level (SPL_{rms}) (50–1500 Hz), the high-frequency SPL_{rms} (2 kHz–20 kHz), the overall SPL_{rms}, and the frequency with the highest acoustic power, herein called the peak frequency. The low-frequency band was selected due to its association with fish communication, and the high-frequency band was selected to assess sound generated from snapping shrimp (Kaplan and Mooney, 2015; Kaplan et al., 2015). The low-frequency band is also highly influenced by boat noise as well as geophonic sounds such as wind and waves. Thus, to determine natural diel patterns of SPL_{rms} at our sites, we considered only files without boat noise.

2.2. Benthic and fish surveys

We conducted benthic visual point surveys for soft corals, hard corals, algae, and substratum cover along six 10-m transects at each site from 7 June to 10 June 2016 and again from 17 July to 24 July 2017. Benthic cover at the point directly below each transect was recorded every 10 cm. Cover was classified as dead coral (bleached, newly dead, or dead), algae (crustose coralline algae, cyanobacteria, turf algae, or macroalgae), invertebrate (aggressive invertebrate or other invertebrate), live hard coral (identified to genus), live soft coral (identified to genus), or substratum (pavement, rubble, or sand). We calculated coral cover for each survey by dividing the total number of points identified as hard coral by the total number of points surveyed at each site ($n = 600$ per survey). The arithmetic mean of coral cover in the 2016 and 2017 surveys is herein called the average coral cover. The total number of hard coral genera present at each site is herein called the genus richness (R).

We conducted four 30-m SCUBA video transects at each site between 7 June and 10 June 2016 and again between 23 July and 26 July 2017 to assess reef fish abundance and diversity. Videos were recorded on an Olympus PEN Lite E-PL5 camera. Attached to the camera's underwater housing and in the field of view was a 1 m long section of PVC with a 25 cm cross bar located 50 cm from the camera lens and a 50 cm cross bar located 1 m from the camera lens. The cross bars were marked at 5 cm increments. This setup assisted with estimating the width of the swath to be examined and for the estimation of fish size. At the sandy site (S), fish were rarely seen and thus a camera was not employed. Instead, the occasional fish was counted, identified by the diver, and recorded underwater.

At each reef site, transects began in the vicinity of the acoustic recorder with transect bearings being haphazardly chosen. Bearings were restricted to those that largely covered reef structure to avoid surveys over large portions of sand. A diver swam along slowly holding the
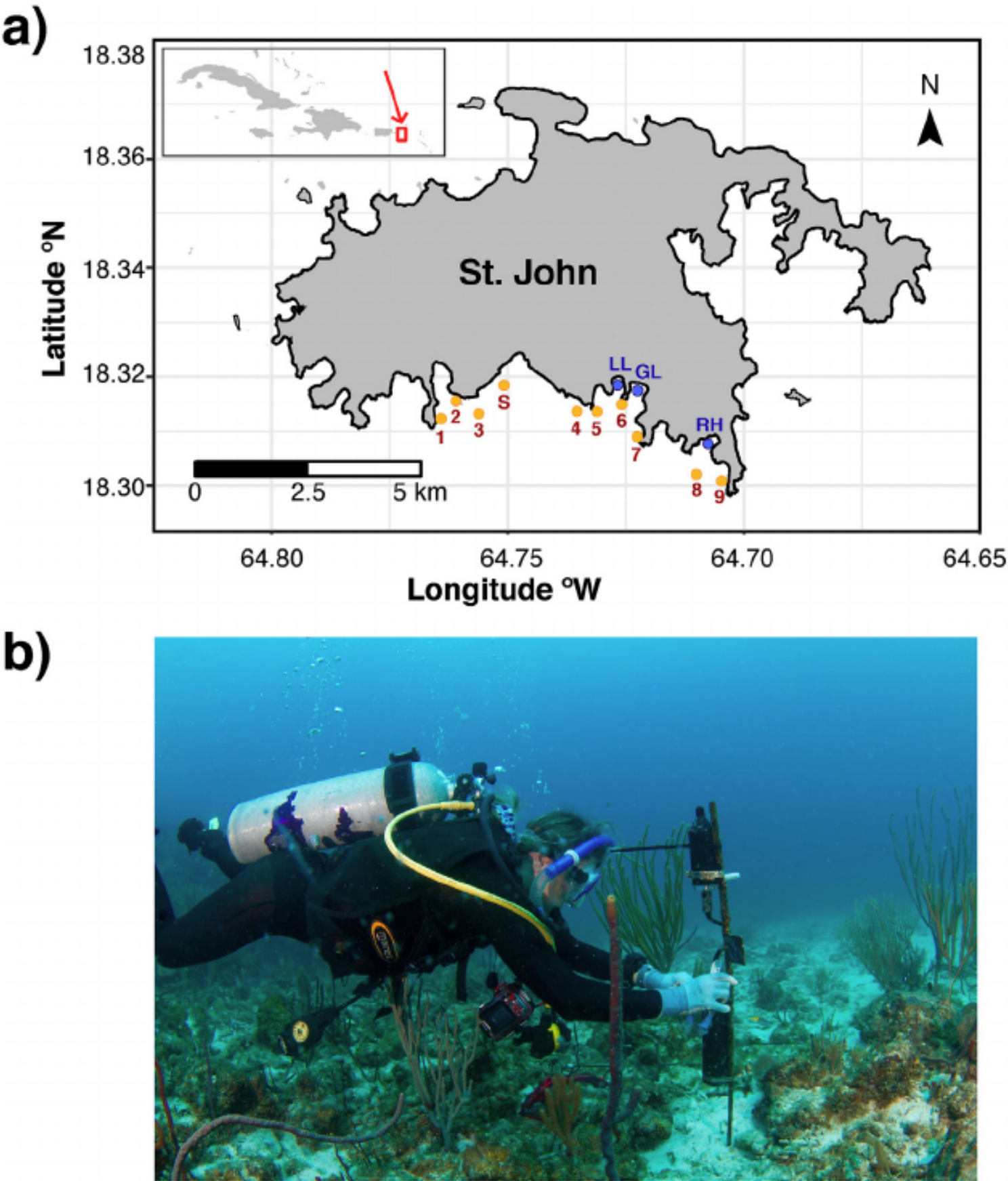
(June 2016, August 2016, October 2016 and March 2017), acoustic recorders were removed for 1–3 days in order to offload data and recharge batteries and were then redeployed.

camera system as close to but above the reef, remaining parallel to the reef structure while at the same time laying out the transect tape. This was repeated for a total of four transects at each location.

Fig. 1. Field sites at St. John and deployment of hydrophones. (a) Map displaying locations of ten acoustically monitored sites off the southern shore of St. John, U.S. Virgin Islands (orange points). The red box indicates the location of St. John within the Caribbean. Blue points indicate mooring sites at Little Lameshur Bay (LL), Greater Lameshur Bay (GL), and Ram Head (RH). (b) Example of passive acoustic recorder deployment. Recording units were attached to a rebar stake and deployed with the omnidirectional hydrophone facing the sea surface. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1

Summary table describing the type of ecosystem, type of boat mooring present, and percentage of days with boat detections at each site. Boat noise was frequently detected at the two sites with SCUBA moorings (6 and 8) and at sites with high coral cover and fish density (see Fig. 5). Generally, the sites without moorings (4, 5, and S) showed lower boat noise occurrences with the exception of Reef 3, where higher boat noise detections might be expected due to increased depth and distance offshore.

Site name	Reef number	Ecosystem	Moorings	Percentage of days with boat noise detections
Dittlif Point	1	Coral reef	Overnight	72.10%
Cocoloba Cay	2	Coral reef	Overnight	84.10%
Joel's Shoal	3	Coral reef	None	96.80%
White Point	4	Coral reef	None	81.20%
Europa Bay	5	Coral reef	None	29.80%
Tektite	6	Coral reef	SCUBA	96.50%
Yawzi Point	7	Coral reef	Overnight	93.40%
Booby Rock	8	Coral reef	SCUBA	91.20%
Ram Head	9	Coral reef	Overnight	97.30%
Reef Bay	S	Sandy shore	None	85.50%

We analyzed video transects on a high-resolution monitor to max-

6–10 cm, 11–20 cm, 21–30 cm, 31–40 cm, and > 40 cm. We calculated total abundance and species richness of reef fishes at each site. Species richness (R) was determined as the total number of reef fish species present at each reef.

At each site, we calculated Shannon-Weiner diversity indices (H) on a genus level for corals and species level for fishes using the following formula:

$$H = -\sum_{i=1}^{R} p_i * \ln(p_i)$$

where p is the proportion of total taxa that taxon i represents and R is the genus richness for corals or species richness for fishes. H is the diversity term with larger values representing more diverse communities. We calculated equitability of Shannon-Weiner indices to compare fish and coral diversities across sites using the following formula:

$$E_x = \frac{H_x}{\ln(R)}$$

where H_x is the Shannon-Wiener index at site x, R is the genus richness for corals or species richness for fishes at site x, and E_x is the equitability at site x (Pielou, 1966). Larger values of equitability represent more diverse communities.

imize capability and confidence of reef fish identification. We randomly selected three transects for analysis if all four transects contained suitable footage and no equipment malfunctions occurred. The three suitable transects were used otherwise. All fish observed in the video were enumerated and identified to lowest possible taxonomic unit. Fish sizes were estimated and grouped into the following size classes: 0–5 cm,

2.3. Statistical analyses

We conducted statistical analyses using the R programming language (RStudio Inc., Boston, MA). We modeled temporal variation in boat noise using binomial generalized additive mixed models (GAMMs)
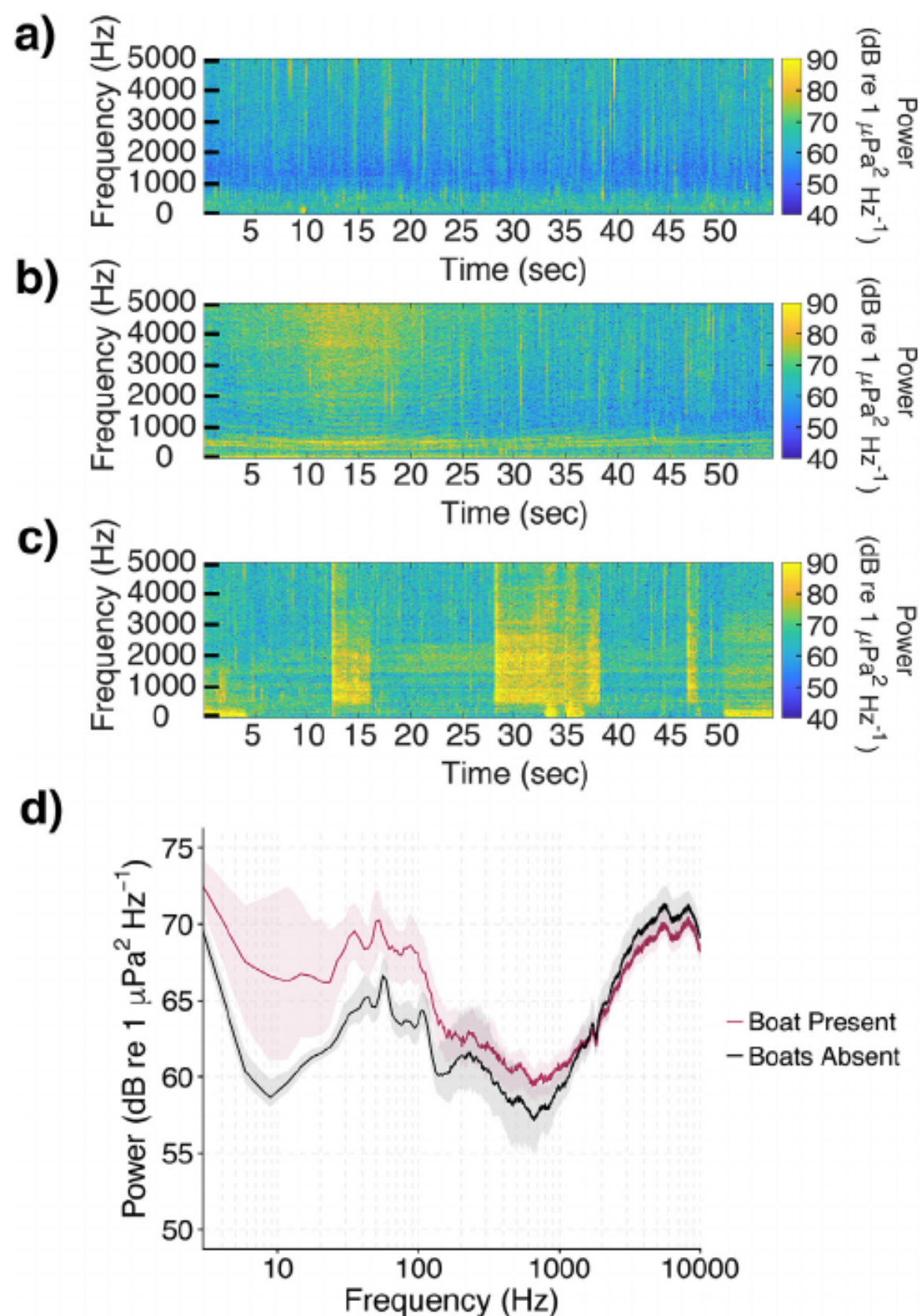
Fig. 2. Spectral comparisons between recordings with and without boat noise. Spectrograms without boat noise (a) and with boat noise (b and c). Sites with boat noise show broadband elevated acoustic power in low frequencies compared to spectrograms without boat noise. (d) Average power spectra show elevated acoustic power below 1000 Hz. Recordings with boat noise exhibit high variability in acoustic power at frequencies below 100 Hz. Lines represent median acoustic power, while shaded areas represent 25th–75th percentiles. Average power spectra were generated from recordings taken between 3/26/2017 and 4/10/2017 at Reef 6 ($n = 2073$). This time period and site were chosen by selecting the two first weeks of data from a randomly selected deployment at a randomly selected site. To generate spectrograms, two recordings with boat noise and one recording without boat noise were randomly selected from within this deployment.

in the R package 'mgcv' (Wood, 2018). We removed recordings collected while our team was conducting experiments in the field from analysis of diel patterns to account for the influence of daily experiments conducted by our group. Because the samples for daily analysis were different than the samples for yearly and weekly analysis, we fitted two GAMMS: one using weekday and month as fixed effects with random slopes for weekday and month within each site, and another using hour of day as a fixed effect and hour of day within each site as a

To determine the acoustic effects of boat noise, we calculated median values and quantiles for SPL_{rms}, low-frequency SPL_{rms} (50–1500 Hz), high-frequency SPL_{rms} (2–20 kHz), and peak frequency at each site. We binned these metrics by site and presence of boat noise. For each site, we compared the median values of these metrics in files with and without boat noise using a Mann-Whitney U test. A post-hoc Bonferroni correction was applied wherever multiple comparisons were made to correct for family-wise error rate.

3. Results

3.1. Boat noise acoustic characteristics

Recordings with boat noise exhibited greater sound levels in the low-frequency band (< 1500 Hz) compared to files without boat noise (Fig. 3b). This difference was as high as 40 dB re 1 µPa $^2 \cdot$Hz^{-1} (Fig. 2; Supplementary material 1). Peak frequencies in recordings with boat noise were significantly lower than recordings of the natural soundscapes, dropping from 3000–6000 Hz to 100–500 Hz (Fig. 3d).

Overall SPL_{rms} and low-frequency SPL_{rms} were significantly higher in recordings with boat noise at all sites (Fig. 3a and b). The contribution of motorboats also caused significant differences in high-frequency SPL_{rms} at six of the ten sites (Fig. 3c; Supplementary Material 2). At the other four sites, the median high-frequency SPL_{rms} was lower in files with boat noise (Fig. 3c; Supplementary material 2). Despite statistical significance at six sites, the actual differences in median high-frequency SPL_{rms} in the presence of boat noise were small (i.e. within 1 dB) and likely an artifact of high statistical power from large sample size.

3.2. Spatial and temporal patterns

Boat noise was a nearly daily occurrence at most sites. Boat noise was observed in over 90% of days at half the sites and in at least 70% of the days at another 4 sites (Table 1). Only Reef 5 had relatively few boat detection days at $< 30\%$. The overall percentage of files with boat noise at each site varied between 1 and 5% (Fig. 4). Boats were most commonly detected at Reef 3 and least commonly detected at Reef 5 ($X^2 = 9480$, $p < 0.0005$) (Fig. 4; Supplementary Material 3). The percentage of recordings containing boat noise at a given site was positively correlated with average coral cover and fish density. There was no significant correlation between the percentage of recordings with boat noise and coral or fish diversity (Fig. 5).

Smoothing estimators were statistically significant for fixed effects of hour, day of week, and month ($p < 0.05$) (Fig. 6; Table 2). Boat noise was lowest at night between 20:00 and 04:00 (Figs. 7 and 8). Occurrences increased starkly at dawn (ca 06:00) and steadily increased throughout daylight hours until late afternoon for most sites (except Reef 7), with peaks in boat activity typically occurring at 15:00 and 16:00 (Figs. 7 and 8; Table 3).

Clear crepuscular peaks in SPL_{rms} were noted in the high-frequency band (Fig. 8). Dawn and dusk peaks were also notable in the low-frequency acoustic band, but SPL_{rms} generally remained elevated compared to nighttime levels throughout daylight hours. The elevations in the low frequencies were shorter in duration, with sound levels usually decreasing shortly after the 18:00 peak. Contrarily, temporal patterns of

nested random effect (Chen, 2000; Hastie and Tibshirani, 1990). Recordings from May, June, and July contained files from 2016 and 2017. For these months, we considered recordings from 2016 and 2017 together because GAMMs with cubic cyclic splines account for the cyclical nature of monthly variation.

We compared spatial variation of boat noise occurrence across sites using a Pearson's chi-squared test for independence. We used ordinary least-squares linear regression modeling to identify relationships between the percentage of recordings containing boat noise at a site and the coral cover, coral diversity, fish density, and fish diversity at that site.

SPL_{rms} in higher frequencies exhibited more gradual transitions.

Boat noise occurrence also varied among days of the week, with peaks between Thursday and Sunday (Table 3; Supplementary Material 4). However, weekly patterns were weaker than diel and annual patterns (Fig. 6). On an annual scale, boat noise was least common during the summer months (June–September) (Fig. 6; Supplementary Material 5). Boat noise detections increased through autumn and early winter before peaking in late winter and early spring (ca. January–March), coinciding with the general patterns in the tourist season (Fig. 6; Supplementary material 5).
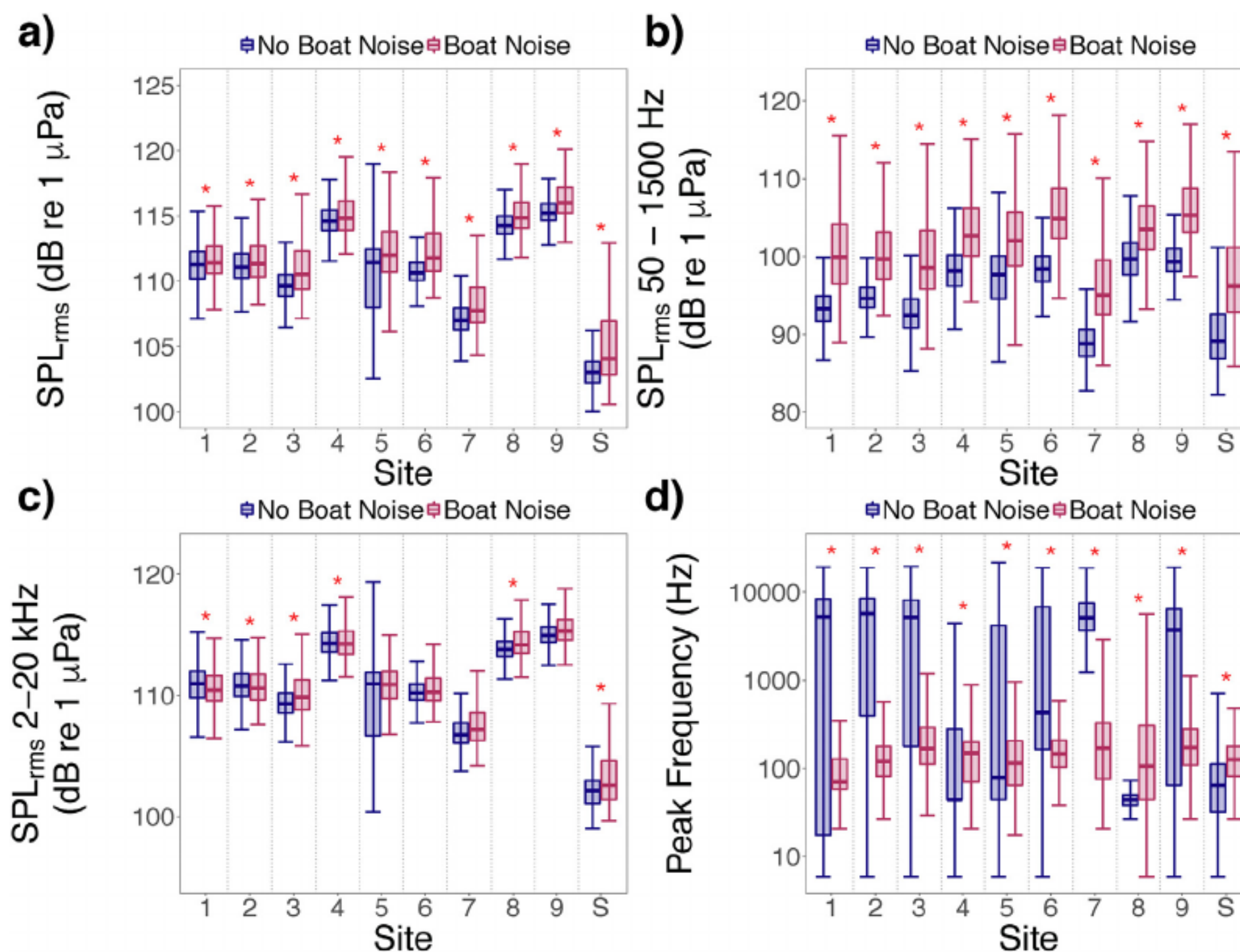
Fig. 3. Acoustic effects of boat noise. Boat noise was associated with significant increases in (a) overall SPL_{rms} and (b) low-frequency SPL_{rms} at all sites. (c) High-frequency SPL_{rms} was not greatly affected by boat noise despite several sites showing statistical significance. (d) Downward shifts in median peak frequency indicate an overall change in ambient soundscapes. Center lines indicate median values, boxes indicate 25th and 75th quantiles, and whiskers indicate maximal and minimal values excluding outliers. Asterisks denote statistical significance between natural noise and boat noise after the Bonferroni correction ($p < 0.0125$).
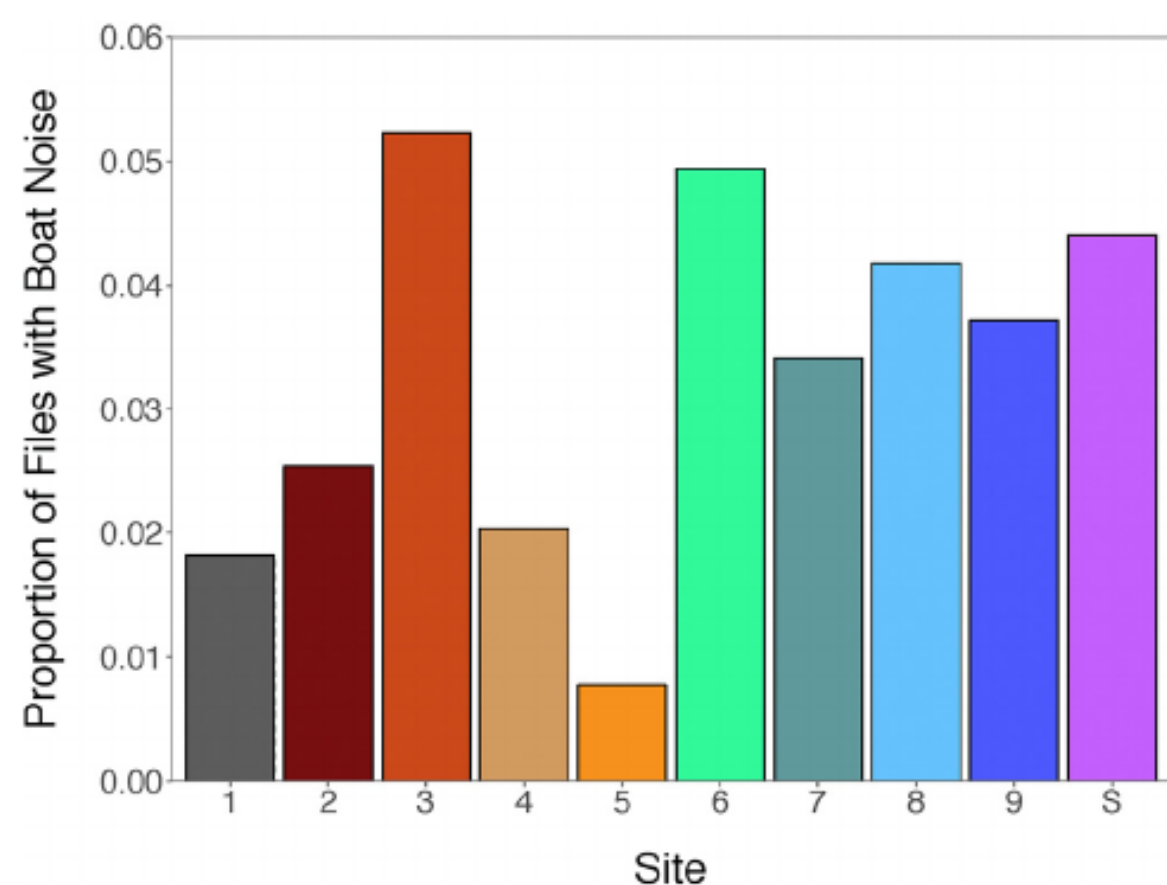

Fig. 4. Variation in boat noise occurrence between sites. Between 1 and 5% of

associated with increases in overall SPL_{rms} (Fig. 3a). The strongest increases arose in low frequencies, reflecting an overall change in the soundscape induced by boat noise (Fig. 3b). These lower frequencies are used by many fishes and some invertebrates for acoustic communication (Freeman et al., 2014; Kaplan et al., 2015; Myrberg, 1981; Tricas and Boyle, 2014). The percentage of files with boat noise also varied among reefs, with detections being most common at sites with high coral cover and fish density (Fig. 5). Taken together, the spatial, temporal, and acoustic patterns presented here suggest that boat noise can be a persistent acoustic stressor on coral reefs, especially on those with a high density of fishes and coral cover.

We do not have visual observation data that could indicate the likely sources of this noise. We suspect that this noise often comes from boats supporting recreational SCUBA diving and snorkeling activity at coral reefs. Two sites with high occurrence rates of boat noise (Reefs 6 and 8) have SCUBA moorings, supporting the suggestion that elevated boat noise is at least partly due to recreational diving. However, the exact sources of boat noise remain speculative until further study. We encourage future studies to use synchronous audio-visual recordings or

the total collected recordings at a given site contained boat noise. Detections were most common at Reef 3 and least common at Reef 5.

4. Discussion

Here, we found that boat noise was a near-daily occurrence on the coral reefs studied in the U.S. Virgin Islands National Park (Table 1). This was especially true at sites with the highest density of fishes and corals (e.g. Reefs 3 and 6), where boats were detected on 97–98% of recording days (Table 1). We observed clear temporal patterns with peaks in boat noise in the early afternoon, weekends, and in the late winter through early spring (Fig. 6). The presence of boat noise was

vessel transects to visually identify vessel types and activities associated with boat noise on coral reefs.

If SCUBA activity indeed is the primary driver of boat noise on the monitored coral reefs, then the detection of elevated boat noise at coral reefs with the highest density of biota suggests that recreational diving may impose an acoustic stressor on the ecosystems with the greatest number of organisms at risk. Further, if this pattern is indeed the result of recreational diving, monitoring such noise could be an indicator of coincident SCUBA-related stressors on coral reefs, including coral fragmentation (Hawkins and Roberts, 1993), sediment deposition (Zakai and Chadwick-Furman, 2002), and pathogen exposure (Lamb et al., 2014). Again, a future study utilizing synchronous or
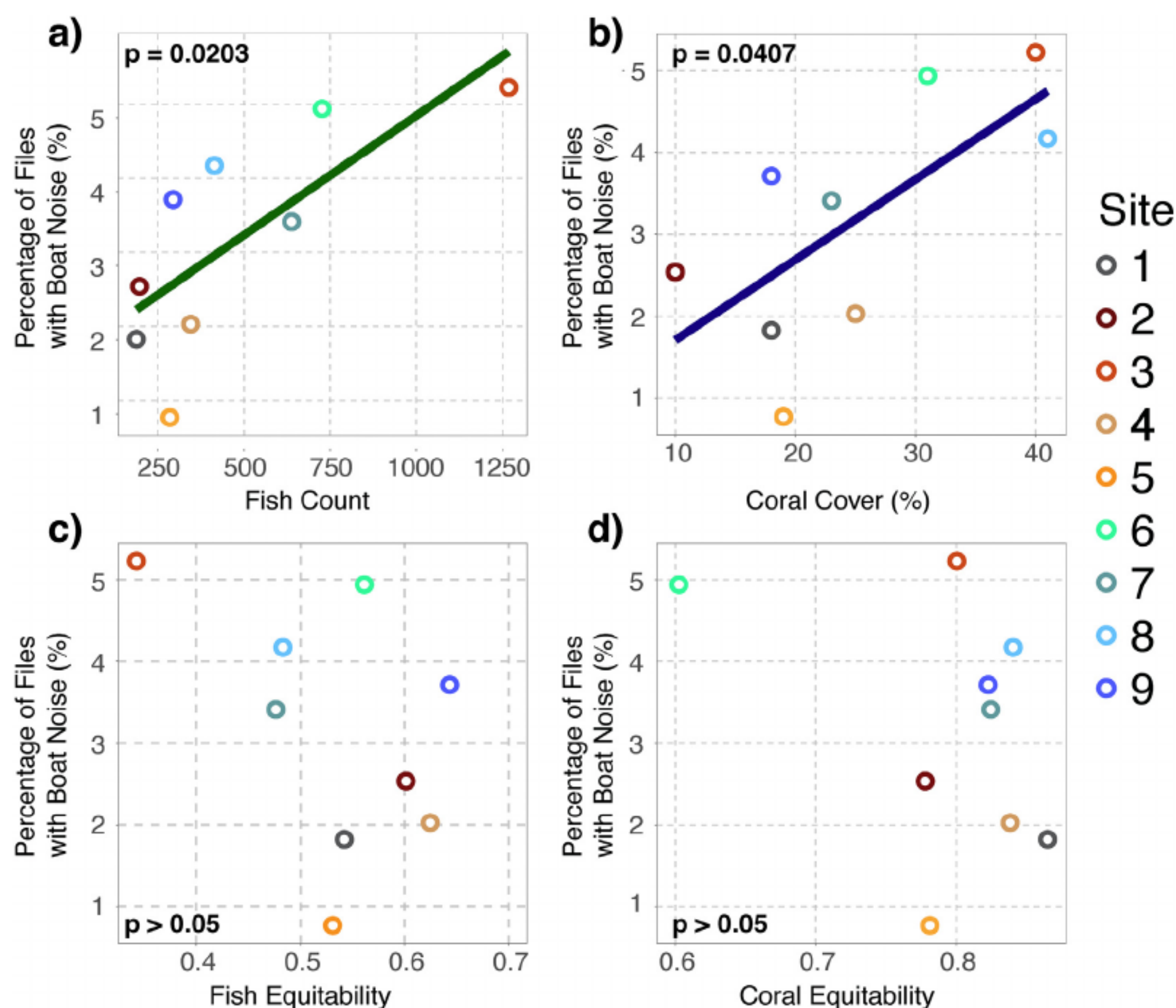
Fig. 5. Ordinary least-squares regressions of boat noise occurrence versus fish and coral metrics. There was a significant positive linear correlation between the percentage of files with boat noise and (a) fish density ($y = 1.62 + 0.00323 * x$, $R^2 = 0.498$, $F = 8.92$, $df = 7$, $p = 0.0203$) and (b) coral cover ($y = 0.724 + 0.0983 * x$, $R^2 = 0.397$, $F = 6.28$, $df = 7$, $p = 0.0407$). There was no significant correlation between the percentage of files with boat noise and (c) fish Shannon-Weiner diversity or (d) coral Shannon-Weiner diversity ($p > 0.05$). The sandy shore site (S) was not included in this analysis.

contemporaneous audio-visual observations is required to confirm that SCUBA activity is the primary source of boat noise before these applications can be appropriately developed.

Anchoring is not allowed in the USVI National Park. Thus it is unlikely that sites without short-term "day" moorings (8 of our 10 sites; though two of these sites, Reefs 7 and 9, are in close proximity to overnight moorings) are subject to high levels of diver-vessel associated boat noise. Therefore, several other contributing factors such as benthic

complexity, depth, and distance offshore may have affected the variation in boat noise rates between sites. For example, boat noise was more prevalent at the sandy site S, than at 6 of the 9 coral reefs. This is likely a function of improved propagation at sandy sites compared to complex benthic systems like coral reefs. As a sandy shore site, S also tends to have lower levels of biological sounds (unpublished data), and boats may be more easily detected in these quieter conditions.

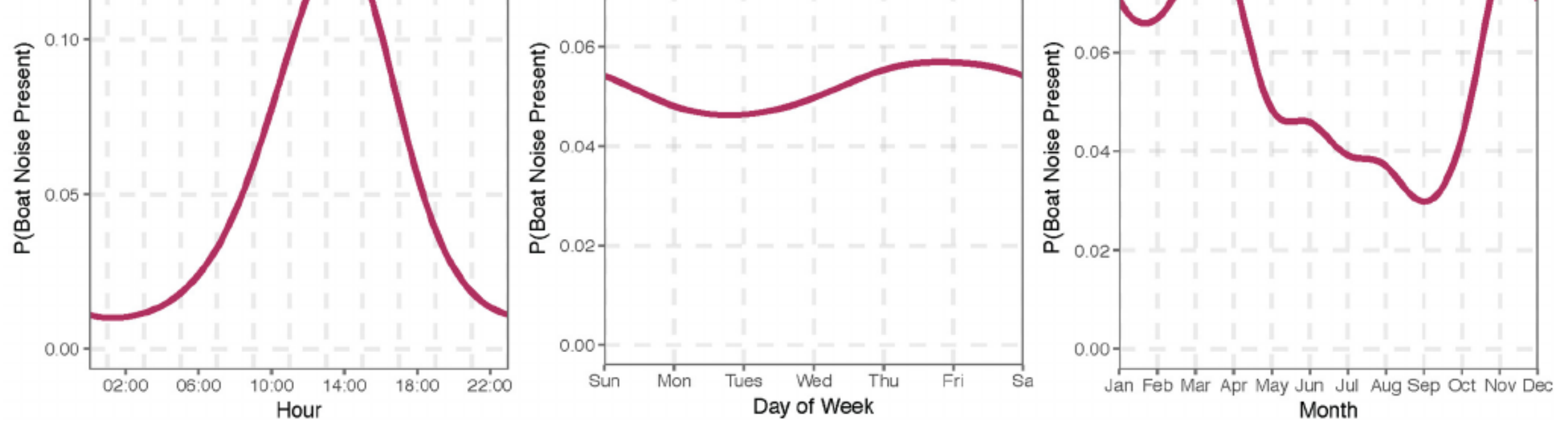Further, boats were most commonly detected at Reef 3. This site is

Fig. 6. Smoothing estimators of GAMM fixed effects after transformation to a binary scale. Knots were capped at 4, 7, and 12 for (a) daily, (b) weekly, and (c) yearly smooths, respectively.

Table 2

Summary of two fitted GAMMs. Estimated degrees of freedom, chi-squared, and p-values are shown for each fixed effect along with the deviance explained by each model.

Response	Fixed effects	edf	X^2	p-Value	Random effects	Deviance explained
Presence of boat noise	Day of week	3.674	230.1	$3.54 * 10^{-15}$	Day of week within site	4.11%
	Month	9.581	66,576	$< 2.2 * 10^{-16}$	Month within site	
Presence of boat noise	Hour	1.999	13,302	$< 2.2 * 10^{-16}$	Hour within site	9%

slightly further offshore and in deeper water than the other sites where boats traveling the coastline are more likely to pass. Additionally, Reefs 6 and 7 are located at the entrance to Greater Lameshur Bay that contains U.S. National Parks Service overnight moorings, which may have influenced boat noise occurrences (Fig. 1). Reefs 1, 2, and 9 are also located near inlets where boat moorings are present (Fig. 1). However, many, if not most, of the moored boats were sailboats and catamarans (pers. obs.), which are likely quieter boats than motor vessels. It therefore remains unlikely that boats at overnight moorings are the sole contributors to elevated boat noise at nearby sites.

Occurrences of boat noise were elevated during daylight hours, reflecting levels of human activity. We found that patterns in boat noise occurrences overlapped temporally with patterns in fish calling, although the exact patterns of the two sounds did not match precisely. We used natural low-frequency SPL_{rms} to approximate diel patterns in fish calls at our recorded sites. In this analysis, we calculated SPL_{rms} using only recordings without boat noise, thereby separating the influence of boat noise and fish chorusing on SPL_{rms}. Low-frequency SPL_{rms} were elevated during daylight hours compared to nighttime hours, and crepuscular peaks were present at most sites, consistent with previous studies (Kaplan et al., 2015) (Fig. 8). Thus, while peaks in boat noise were not exactly coincident with crepuscular peaks in ambient fish calls and natural sound levels, elevated presence of boats during daylight hours when fishes are commonly calling may still pose a potential risk for acoustic masking.

The presence of boat noise increased SPL_{rms} particularly in lower frequencies, reflecting the dominant frequencies of the noise from the vessel engines. Yet, the SPL_{rms} measured in this study may still underestimate the acoustic levels experienced by coral reef organisms because many fishes and invertebrates detect sound primarily in the form of particle motion (e.g. Montgomery et al., 2006; Radford et al., 2016). Acoustic particle motion of boat noise exceeds SPL_{rms} within 5 m from the noise source (Wahlberg et al., 2008). Thus, while SPL_{rms} measurements are sufficient for identification of boat noise, future studies that quantify the particle motion associated with boat noise in shallow-water environments may be more appropriate to fully understand the extent to which boat noise may affect coral reef organisms.

Coincident with occurrences of boat noise, peak frequencies switched from the higher, snapping shrimp-dominated bandwidth (ca. 3–6 kHz) to the fish communication bandwidth (ca 100–500), suggesting that boat noise changes the patterns of the overall soundscape. Reefs 4 and 8 were the two exceptions where naturally occurring soundscapes were dominated by low-frequency sounds. The anomaly at these two sites is likely the result of increased wave action, wind, and other geophonic sources that predominantly affect frequencies lower than 1 kHz (Wenz, 1962).

Intense acoustic energy at frequencies below 1 kHz raises concerns for reef fishes, whose hearing thresholds and sound production lie within this range (Maruska et al., 2007; Myrberg et al., 1986; Wright et al., 2005). The frequency overlap between boat noise and natural noise raises concerns for acoustic masking in coral reef fishes, who use sound for ecological functions including courtship (Myrberg et al., 1986), individual recognition (Myrberg and Riggio, 1985), and larval settlement (Radford et al., 2011). While acoustic masking has not been well-documented in adult coral reef fishes, it has been demonstrated in several non-reef fishes (e.g. Codarin et al., 2009; Vasconcelos et al., 2007). One concern is that acoustic signals in fishes are often not very high in amplitude (Amorim, 2006), making them vulnerable to masking. Formal calculations of masking are beyond the scope of this study. However, given the frequency range of reef fish communication and the temporal patterns of boat noise presented here, we believe that coral reef fishes are prone to masking, and we highly encourage future study to quantitatively assess the effects of boat noise on communication space in these systems.

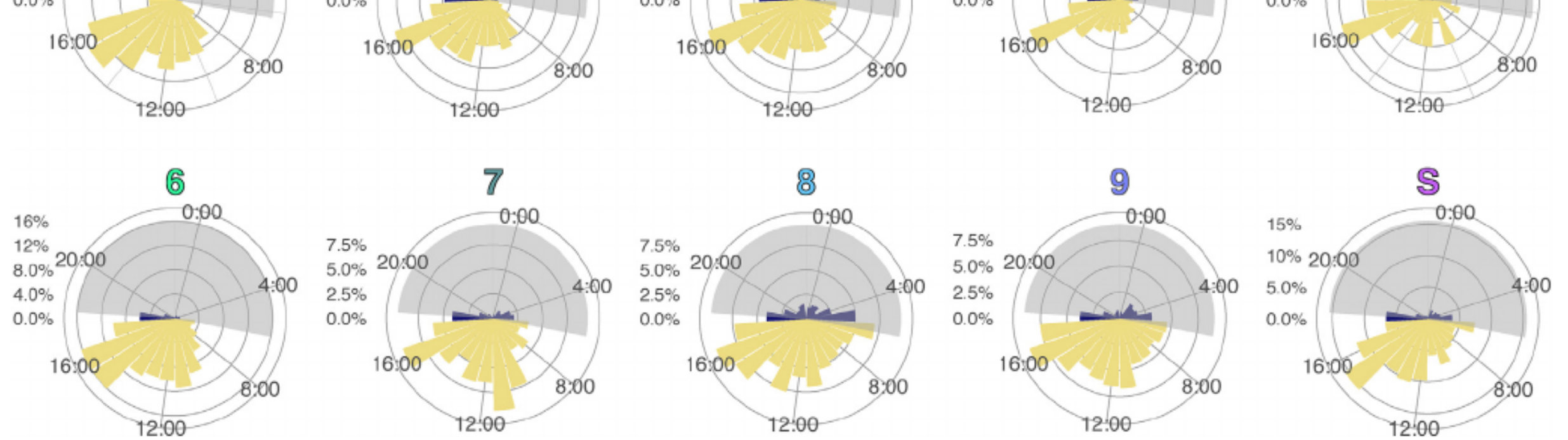The chronic and repeated exposures to boat noise may also have

Fig. 7. Diel patterns in boat noise occurences at each site. Most sites showed low boat occurrence rates at night with a sudden increase occurring in the early daylight hours. Peaks in boat noise are observed in the late afternoon (ca 15:00). The gold bars with a white background denote hours of daylight while the navy bars with a gray background denote hours of darkness based on winter (i.e. November–January) daylight hours. Scales vary between plots, and tick labels are noted to the left of the graph. Colors of the site numbers correspond to the coloring scheme seen in other figures. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)
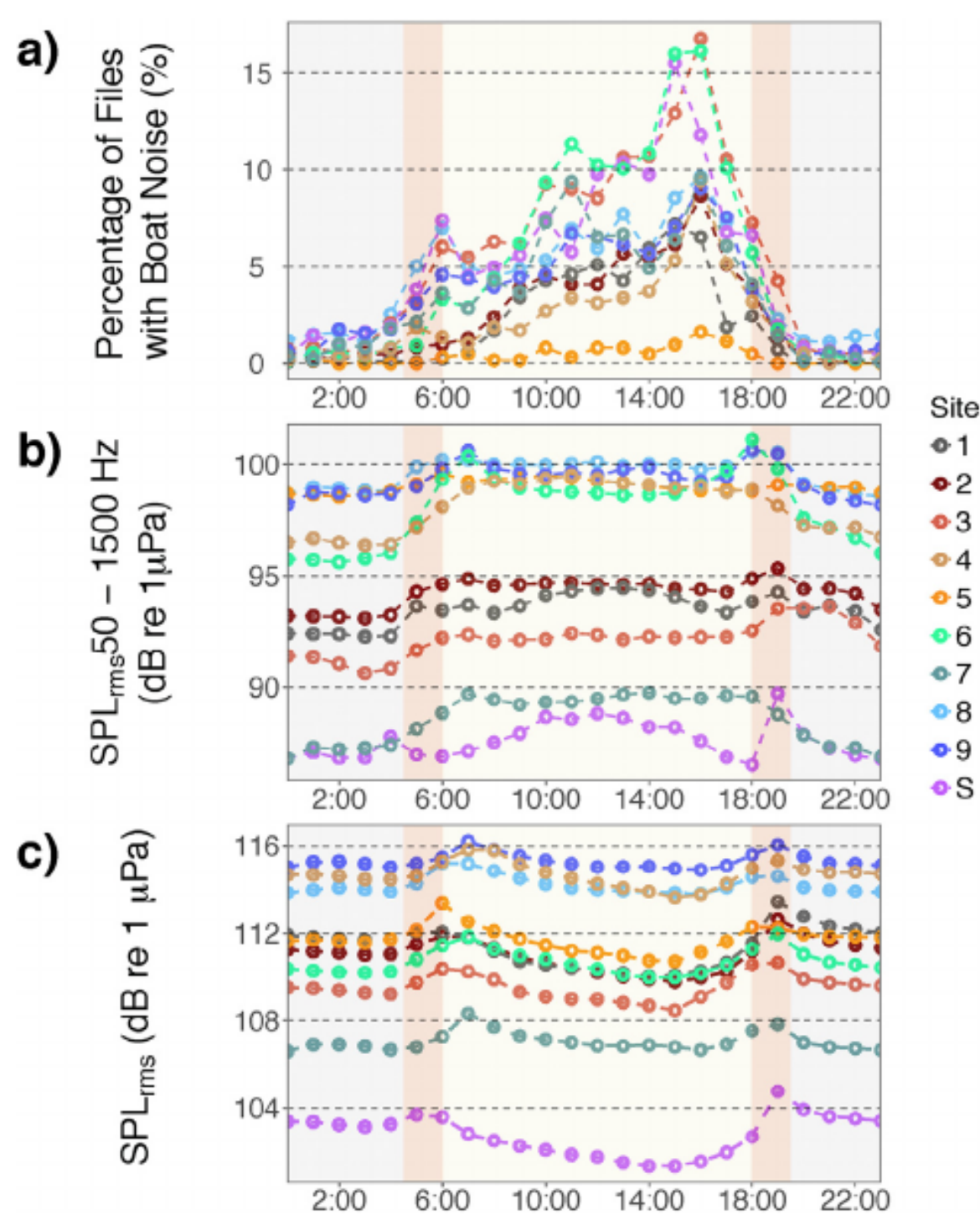
Fig. 8. Diel patterns in boat noise and ambient sound levels. (a) Boat noise was lowest during nighttime hours, sharply rose at dawn, and peaked in the late afternoon. (b) Median SPL_{rms} in the low-frequency fish communication band were elevated during daytime hours, concurrent with high boating activity. (c) Median overall SPL_{rms} exhibit crepuscular peaks. Only files without boat noise were selected for SPL_{rms} calculations in (b) and (c), thereby eliminating the effect of boat noise in calculated SPL_{rms}. Recordings were binned by hour and pooled across the entire deployment year to determine the percentage of files with boat noise and to calculate median SPL. Background colors correspond to the time of day, with light purple indicating night, red indicating dusk and dawn, and yellow indicating daylight. Daylight designations are based on winter daylight hours (November–January). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

unlikely given the temporal unpredictability and the acoustic variability between boat noise occurrences. Even within a national park, which can be some of our most protected marine areas, daily exposure to boat noise still poses a chronic stressor to resident coral reef fishes and invertebrates. The need to include anthropogenic noise in ecosystem management plans becomes increasingly clear given the frequent detections of boat noise presented here and the burgeoning body of scientific evidence elucidating the biological consequences of small vessel noise. Noise from small boats can have lethal ecological and physiological effects on coral reef organisms, with known consequences ranging from larval recruitment (Holles et al., 2013; Simpson et al., 2016a) and cognition (Ferrari et al., 2018) to behavior (Holmes et al., 2017; McCormick et al., 2018; Nedelec et al., 2017), and embryonic development (Jain-Schlaepfer et al., 2018; Nedelec et al., 2014). The data presented here suggest that there is potential for these effects to already be manifesting, even within national parks, and will continue to exacerbate with increasing demand for tourist activities on coral reefs (Gil et al., 2015; Piggott-McKellar and McNamara, 2017). The U.S. National Parks Service (NPS) addresses threats associated with anthropogenic noise through the Natural Sounds and Night Skies Division. However, most efforts outlined in this program specifically address terrestrial ecosystems and noise pollution from overflights. The data presented here suggest that natural sounds of aquatic national parks are also at risk to noise pollution and may face frequent noise events. Thus, we encourage the expansion of U.S. NPS acoustic monitoring and prevention initiatives to include marine and coastal national parks. Boat noise is easily detectable and potentially preventable, and management of this ubiquitous pollutant could abate a chronic stressor in an ecosystem that is already facing myriad other chemical and physical threats.

Our results show that acoustic monitoring data can quantify boat activity and noise exposure on relatively fine spatial (individual reefs) and temporal (hourly) scales. This tool is particularly useful in remote areas where visual inspection is infeasible or not supported and in marine protected areas where boat activity is potentially restricted. These methods could be used to inform ecosystem managers regarding not only noise exposure, but also potentially fishing and harvesting in closed areas, diving in restricted areas, and other usage violations in marine protected areas. Furthermore, the data presented here provide valuable baseline information elucidating the extent and patterns of anthropogenic noise in coral reefs. These baseline data can be used to assess current and future levels of noise pollution and can be applied to inform methodology of laboratory studies.

Table 3

Summary of temporal peaks in boat noise at each site. Daily peaks occurred in the late afternoon (i.e. 15:00–16:00) at all sites except for Reef 7. Weekly peaks varied but occurred on weekends (i.e. Friday–Sunday) at 7 out of 10 sites. Monthly peaks varied but most commonly occurred between January and March.

Reef	Hour with peak boat frequency	Weekday with highest boat frequency	Month with peak boat frequency
1	15:00	Tuesday	March
2	16:00	Friday	January
3	16:00	Thursday	January
4	16:00	Sunday	February
5	16:00	Friday	May
6	15:00	Friday	April
7	11:00	Thursday	March
8	16:00	Sunday	January
9	16:00	Sunday	March
S	15:00	Friday	March

implications on the behavior, physiology, and sensory neurobiology of coral reef organisms. Some evidence suggests that coral reef fishes may desensitize to boat noise during short-term continuous exposure (Holmes et al., 2017). However, long-term desensitization here is

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2018.09.009.

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