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 (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.

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 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.