NOTE

Using unmanned aerial vehicles (UAVs) to investigate shark and ray densities in a shallow coral lagoon

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ABSTRACT: Unmanned aerial vehicles (UAVs) are being increasingly used in studies of marine fauna. Here, we tested the use of a UAV (DJI Phantom II®) to assess fine-scale variation in densities of 2 elasmobranchs (blacktip reef sharks *Carcharhinus melanopterus* and pink whiprays *Himantura fai*) on reef systems off Moorea (French Polynesia). We flew parallel transects designed to sample reef habitats (fringing, channel and sandflat habitats) across 2 survey blocks. Block 1 included a shark and ray provisioning site with potentially higher elasmobranch densities, whereas Block 2 most likely had lower densities with no provisioning activities. Across 10 survey days in July 2014, we flew 3 transects (400 m) within each survey block (n = 60 total transect passes). As expected, densities (animals ha^{-1}) were significantly higher in Block 1 than in Block 2, particularly where provisioning activities occur. Differences between habitats surveyed were also found. Our study provides the first direct estimates of shark and ray densities in coral-reef ecosystems and demonstrates that UAVs can produce important fishery-independent data for elasmobranchs, particularly in shallow-water habitats.

KEY WORDS: Elasmobranchs · Distribution · Abundance · Aerial surveys · Unmanned aerial drones

INTRODUCTION

Measuring spatiotemporal variation in densities of elasmobranchs is important for determining drivers of abundance, elucidating ecological roles and importance and developing effective management plans (e.g. Heithaus et al. 2010). Elasmobranch density estimates (or rates of detection) are generally derived from fishing methods (catch per unit effort), diver surveys and remote video cameras (e.g. baited remote underwater videos [BRUVs]; Goetze & Full-

wood 2013, Rizzari et al. 2014a,b). Although all of these methods can provide important information, each requires specific assumptions or may be inappropriately invasive in some situations (e.g. potential mortality of critically endangered species).

Aerial surveys have been used extensively to estimate marine animal distribution, absolute abundance and population trends over large spatial scales (e.g. Marsh & Sinclair 1989, Martin et al. 2016) of large subsurface elasmobranchs (e.g. Rowat et al. 2009, Robbins et al. 2014) and small elasmobranchs

in shallow-water habitats (Kessel et al. 2013). However, the use of aircraft has a number of logistical (prohibitive cost, risk for observers) and scientific (human observer error and misidentifications, low resolution of local data) drawbacks (Christie et al. 2016). For studies of elasmobranch densities using aircraft, observers are also limited by environmental factors that affect the ability to detect sharks (e.g. water depth and substrate composition; Robbins et al. 2014). In recent years, there has been a considerable rise in the use of unmanned aerial vehicles (UAVs or 'drones') to study wildlife distribution, abundance and behaviour in both terrestrial and marine ecosystems (e.g. Anderson & Gaston 2013, Hodgson et al. 2013, Christie et al. 2016). This tool is useful for quantifying animal densities, investigating the behaviour of individual and aggregating animals and evaluating the impacts of human activities on natural systems or protected species at a relatively low cost (Anderson & Gaston 2013, Hodgson et al. 2013). The use of UAVs equipped with high-resolution video cameras also enables continuous recording of data, and post-processing videos permits enhanced detection rates, more accurate species identifications and the potential to estimate true densities (Hodgson et al. 2016). Small UAVs operate over a small range (line-of-sight flights) and at a low altitude (Anderson & Gaston 2013). They are easy to manoeuvre and can be potentially used in coral reef studies to monitor species distributions (particularly in shallow-water habitats not accessible by boats or even divers) and coral reef habitats.

With growing concern about the large-scale losses of reef sharks worldwide, efforts have been made to convert extractive practices to non-lethal uses of sharks or to develop tourism around sharks to prevent incentives to initiate fisheries (Brena et al. 2015). While previous studies have shown that this activity can modify the relative abundance and movements of sharks, less attention has been paid to how these activities might influence spatiotemporal patterns of shark densities relative to provisioning activities. Such an understanding is important for elucidating the potential effects of elasmobranch feeding on coral reef habitats since the presence of these apex and mesopredators has the poten-

tial to influence reef communities through changes in predation rate and the spatial and temporal pattern of risk (or 'non-consumptive') effects induced by sharks (Heithaus et al. 2010).

In this study, we aimed to use a UAV (DJI Phantom II®) to investigate elasmobranch densities in the shallow lagoons off Moorea (French Polynesia). Our main objectives were to use a small UAV for estimating reef shark and ray densities in shallow coral reef habitats, and gain insights into the possible effects of provisioning on elasmobranch spatial distribution (including microhabitat preferences) and densities.

MATERIALS AND METHODS

Study area

This experiment was conducted at Moorea, a high island located in the Society Archipelago, French Polynesia (17° 30' S, 149° 51' W, Fig. 1). Shark and ray tourism, including provisioning (where animals are baited and fed for viewing purposes), has occurred
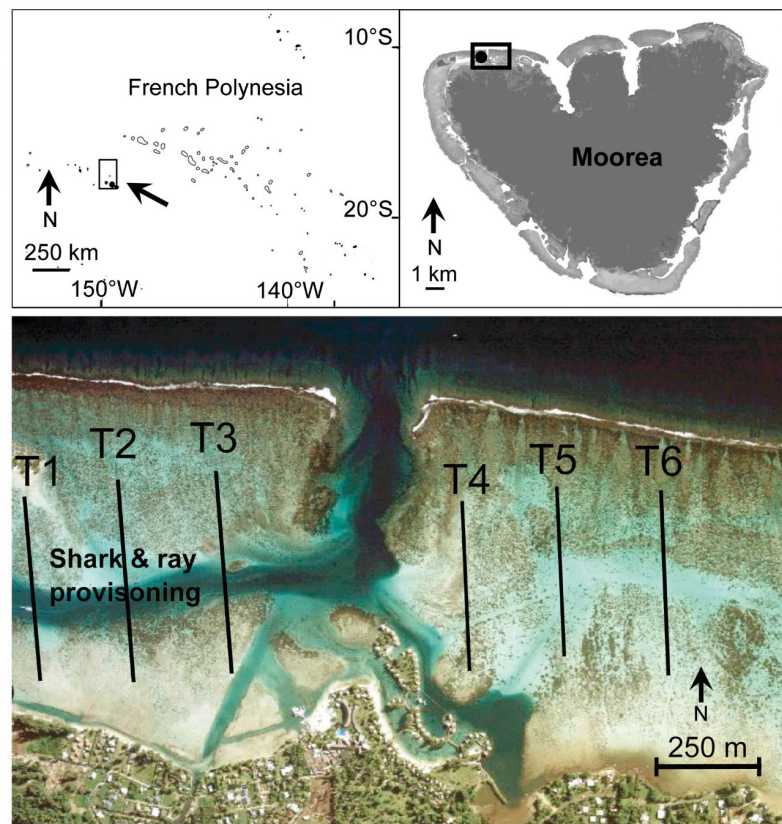

Fig. 1. Study area off Moorea (French Polynesia) and sampling design (Block 1: T1–T3; Block 2: T4–T6). The shark and ray provisioning site is located on transect 2 (T2), on the inner portion of the barrier reef and in the vicinity of the channel

for 2 decades inside the lagoon for pink whiprays *Himantura fai* and blacktip reef sharks *Carcharhinus melanopterus* (Gaspar et al. 2008). Our study occurred in the inner waters of the lagoon off the north coast (Fig. 1), including at the shark and ray provisioning site where tourists are present almost all day.

In order to estimate elasmobranch densities and the impact of provisioning on these densities, 2 survey blocks, each with 3 parallel transects oriented perpendicular to the coast, were defined with relatively similar habitat characteristics (fringing reef, deeper channel and internal portion of the barrier reef). The middle transect of Block 1 (T1–T3) contained the provisioning site (T2, Fig. 1), while Block 2 (T4–T6) was distant from any provisioning activities (Fig. 1). Depth in the study area was from 1.5 to 2 m on reefs (fringing and barrier) and varied from 2 to 7 m in the channel habitat. The tidal range in the study area is small at ca. 0.4 m. The depths of the channel in Block 1 are greater than those in Block 2, but good water visibility enabled detection of every targeted object, based on complete agreement of 2 independent observers of video data. No regulations or flight restrictions were in place in French Polynesia when our surveys were conducted, and this project was conducted under the supervision of USR 3278 CNRS-EPHE-UPVD, Centre de Recherche Insulaire et Observatoire de l'Environnement (CRIOBE).

Equipment and sampling

Drone flights were conducted using a DJI Phantom II® UAV quadcopter (30 × 30 × 18 cm; >1 kg mass) equipped with a GoPro Hero 3+ Silver edition video camera (with a 64 GB mini SD card) mounted underneath, as well as a set of six 5200 mAh lithium polymer batteries (Fig. 2). Each battery offers up to 25 min of flight time, but batteries were replaced between transects that lasted 4 to 7 min. The Phantom II includes a Naza-M V2 multi-axis flight controller, GPS and compass that allow for stable flight, maintenance of a consistent altitude, wind compensation, station holding and reliable user control. We flew transects designed to sample fringing, channel and barrier reef habitats in the lagoon of Moorea (Fig. 1). Transects were only flown in sea conditions <1 (Beaufort scale) to ensure optimal detection of targeted objects at all depths.

In order to define the adequate flying height and to be able to properly detect our targeted species, we experimentally flew transects over known targets at

Fig. 2. DJI Phantom II® unmanned aerial vehicle (UAV) quadcopter equipped with a video camera GoPro Hero 3+ Silver edition used for this study (photo credit: Valere Sabatier)

varying heights (10, 12, 15 and 20 m). The 12 m height was selected because it allowed a relatively large spatial coverage while ensuring that we would be able to identify all elasmobranch species in the study area including our 2 target species (*H. fai* and *C. melanopterus*). Field and in-facility measurements of the field of view were performed, highlighting that we surveyed a 35 m wide bandwidth during flights. Each transect was flown at a speed of 2 m s^{-1}, and the length of each transect was 400 m. UAV height was determined using a Nikon Aculon 6 × 20 rangefinder (±1 m resolution), with height readings taken at the beginning and end of each flight. Transect locations were recorded using a handheld Garmin 76Cx GPS. During each flight, the camera was set to record video at full high definition resolution (1920 × 1080) and 30 frames s^{-1} in the wide field of view setting. The camera was positioned to film straight down, and a polarizing filter was used to minimize glare. An H3-3D 3-axis gimbal (DJI) was used to stabilize the GoPro. Video coverage area was determined using premeasured landmarks and a network of buoys located across the study area. In order to ensure that each targeted object was observed, 2 independent observers extracted counts from each video collected. Counts from each observer were strictly identical, which confirms that all targeted objects were detected. However, in the absence of an availability bias experiment, we cannot reject the possibility of having missed targeted objects.

Density estimates

The strip transect survey methodology was applied to our dataset (Buckland et al. 2005), assuming that all targeted objects are detected. For each transect,

densities for *H. fai* and *C. melanopterus* were calculated using the following formula:

$$D_i = \frac{n_i}{w.L} \tag{1}$$

where n_i is the number of individuals detected, w is the strip width (measured in the laboratory using a measuring tape and verified *in situ* during each flight using the known size of our research vessel), and L is the length of the transect (in metres). Densities were expressed in number of animals ha^{-1}.

The Shapiro-Wilk test was used to assess the assumptions of parametric tests and fit parametric requirements. Differences in abundance and density of elasmobranchs between the two sampled blocks were investigated using pairwise *t*-tests. Spatial variations of densities between transects and microhabitat sampled were investigated using a 1-way ANOVA. However, due to a large number of zeros in our dataset, a zero-inflated Poisson model was used to compare densities between transects. Statistical analyses were performed using R v3.1.2.

RESULTS

We flew transects on 10 different days between 2 and 25 July 2014. Flights occurred between 08:00 and 10:00 h, and only when visibility conditions were optimal (absence of glare and wind). On each survey day, all 6 transects were flown once (Fig. 1). Our sampling design enabled easy detection and identification of elasmobranchs (Fig. 3). We recorded 3 species of elasmobranch including *Carcharhinus melanopterus* (n = 152 sightings), *Himantura fai* (n = 99) and ocellated eagle rays *Aetobatus ocellatus* (n = 2, off effort during transit time). Sea turtles of unidentified species (either hawksbill *Eretmochelys imbricata* or green sea turtles *Chelonia mydas*) were recorded on 3 occasions. Mean abundances of elasmobranchs were significantly higher in Block 1 (mean ± SD = 1.43 ± 4.36 animals) than in Block 2 (0.03 ± 0.21 animals, t_{153} = 4.22, p < 0.0001, Fig. 4A) as were densities (Block 1: 0.93 ± 2.82 animals ha^{-1}, Block 2: 0.02 ± 0.11 animals ha^{-1}, t_{153} = 4.24, p < 0.001; Fig. 4B). However, we noted significant spatial variation among transects (1-way ANOVA, $F_{5,85}$ = 20.82, p < 0.001, Fig. 5A). Indeed, sharks and rays were only detected on a single transect in Block 1 (T2) and a single transect in Block 2 (T6). The zero-inflated Poisson model confirmed a significantly higher abundance of sharks and rays at the provisioning site (z_9 = 4.491, p < 0.0001). Elasmobranch numbers were also signifi-
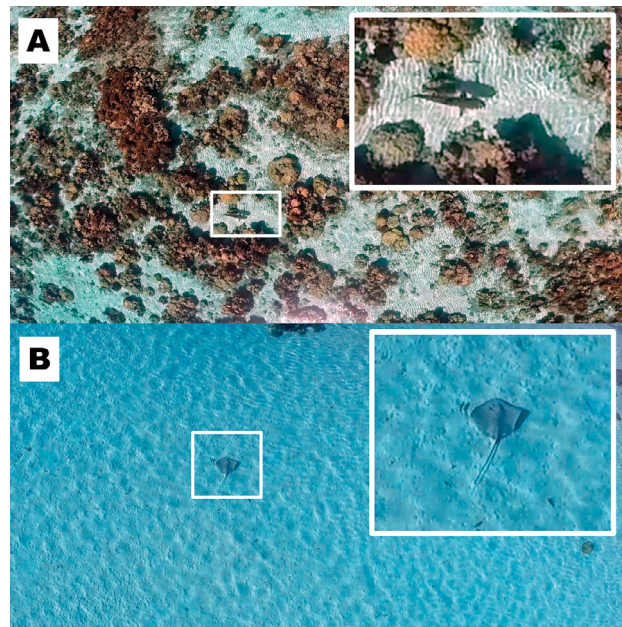

Fig. 3. Screen captures of (A) a blacktip reef shark *Carcharhinus melanopterus* on the barrier reef and (B) a pink whipray *Himantura fai* on the sandflat collected during unmanned aerial vehicle (UAV) surveys

cantly different across the 3 micro-habitats sampled, with higher densities in the inner portion of the barrier reef and in the channel (1-way ANOVA, $F_{2,90}$ = 6.27, p < 0.01, Fig. 5B).

DISCUSSION

In shallow coral reef habitats, fishery-independent density data are difficult to generate for fish, including elasmobranchs, but we have shown that UAVs are extremely valuable to produce such information, especially when working at a relatively small spatial scale (e.g. less than a few km^2). UAVs are potentially useful to investigate population trends and habitat use patterns, and to assess the effect of human activities (e.g. tourism) on abundance, particularly in coastal and shallow habitats where visibility enables animal detection from the surface to the bottom.

Provisioning activities seem to result in sharks and rays aggregating in a relatively confined area for prolonged periods. However, due to a lack of replication, it is not possible to definitively ascribe differences in density between the sites to provisioning. Nevertheless, aggregation in response to provisioning remains the most parsimonious explanation for spatial variation in abundance given variation among transects within blocks as well as differences between them. It is important to note that provisioning
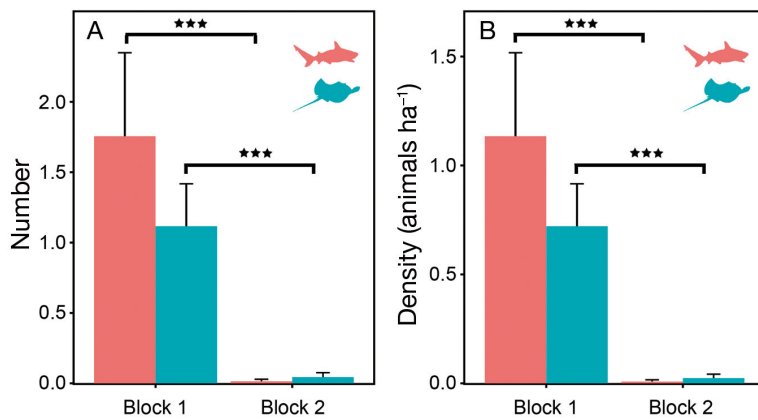
Fig. 4. Mean (±SD) (A) abundance and (B) density of blacktip reef sharks *Carcharhinus melanopterus* and pink whiprays *Himantura fai* in the 2 surveyed blocks. Lines above bars indicate significant differences (***p < 0.001, *t*-test) between blocks for each species
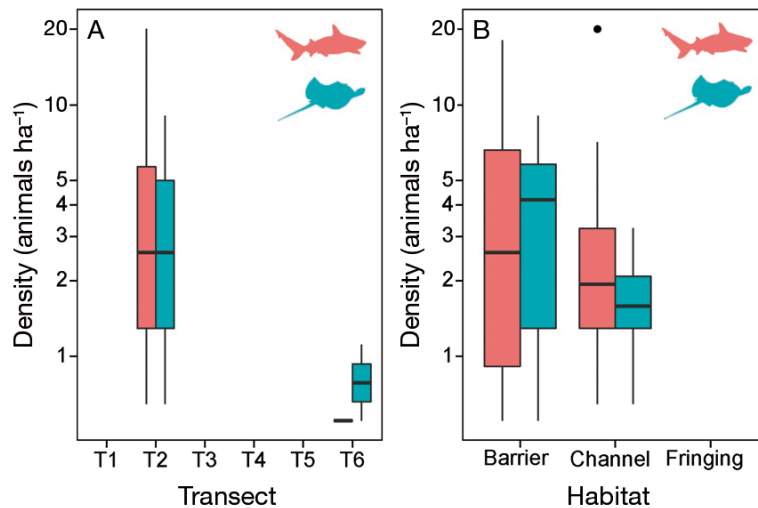

Fig. 5. Distribution (median, 50 and 75 percentiles and outliers) of the density of blacktip reef sharks *Carcharhinus melanopterus* and pink whiprays *Himantura fai* for (A) each transect and (B) each micro-habitat surveyed

Previous studies on the movements and site fidelity of pink whiprays off the north coast of Moorea, including at the provisioning site we sampled, have shown that tagged rays significantly increase their residency at the shark and ray feeding site, including beyond hours when provisioning occurs (Gaspar et al. 2008). Overall, our study suggests that provisioning activities can significantly affect the distribution and densities of reef sharks and rays in Moorea, as previously documented (Clua et al. 2010, Brena et al. 2015). However, expanding surveys to areas farther from the provisioning site and across all daylight hours will provide important insights into the spatial scale at which provisioning might affect shark and ray densities. Previous work using underwater photo-identification methods on blacktip reef sharks estimated that the mean number of individuals at the provisioning site was (mean ± SE) 8.97 ± 0.72 (Mourier et al. 2012), which is significantly lower than the mean number of individuals observed on transect T2 during our study (15.1 ± 2.77). We cannot reject the hypothesis of an increase in the abundance of blacktip reef sharks between the studies (February 2008 to June 2010 versus July 2014). However, it is likely that UAV surveys provide more accurate abundance estimates since individual sharks may respond differently to the presence of an observer in the water. Finally, our results suggest that densities varied significantly across microhabitats inside the lagoon, with higher densities on the inner slope of the barrier reef and in the adjacent channel. However, this difference was most likely driven by the location of provisioning activities that occur on the inner portion of the barrier reef in T2, close to the channel. While further studies are needed to fully elucidate microhabitat preferences of elasmobranchs in reef habitats, especially in the absence of anthropogenic factors, elasmobranchs were not detected on the fringing reef, suggesting that these habitats might be less suitable for reef elasmobranchs, particularly small carcharhinids (Rizzari et al. 2014a).

In conclusion, regardless of the driver(s) of variation in densities within and across blocks, this study has shown that UAVs can be used to quantify densities of actively swimming elasmobranch species

activities were occurring throughout the times of day during which we flew transects, so these patterns might be different at other times of the day when feeding activity decreases. In Block 2, densities were very low, supporting the hypothesis that reef elasmobranchs occur at low densities in the inner waters of the lagoon of Moorea, or that these densities are significantly impacted by provisioning activities, affecting the distribution of reef sharks and rays on peripheral reef systems off the north coast. However, unlike Block 1, Block 2 did not have a deep channel. The presence of a deeper channel could provide access for individuals coming from the outer slope, which could have an effect on observed densities in Block 1.

in coral reef habitats, at a relatively low cost (<US$ 2500) and over a short period of time. Our dataset also revealed the potential of UAV videos and data to assess the distribution of smaller taxa (reef fishes, echinoderms) that could be coupled with other *in situ* measurements and counts (snorkeling fish transect surveys). Despite the considerable potential of UAVs for wildlife monitoring and research, however, it is important to consider government approval and navigational stipulations that impose restrictions on the use of UAVs before undertaking research projects involving the use of UAVs (Vincent et al. 2015). Lastly, UAVs may have undesirable effects on studied species, and a careful consideration of risks of disturbance should be carried out to mitigate or alleviate them (Hodgson & Koh 2016).

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