

# **The role of mangrove fringe areas in providing feeding habitat for the New Zealand eagle ray (*Myliobatis tenuicaudatus*)**



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## Executive summary

This report details the results of an investigation into the use of mangrove fringe habitat by the eagle ray *Myliobatis tenuicaudatus*. The study was conducted in Matua estuary, Tauranga Harbour, and examined differences in the use of natural mangrove fringe vs. trimmed (mangrove removal and maintenance) mangrove fringe areas by rays. The use of these areas as feeding habitat was determined by quantification of feeding excavations.

Mangrove fringe was found to be an important feeding habitat for this species with high numbers of pits recorded adjacent to mangrove forest throughout the study. There were consistently fewer feeding pits present in the trimmed fringe habitat compared with natural fringe suggesting that modified mangrove habitats may be of lower quality feeding habitat for this species. Mean numbers of pits per plot over the study period were 4.03 ( $\pm 1.64 SE$ ) in the trimmed fringe, and 13.83 ( $\pm 4.16 SE$ ) in the natural fringe. The observed differences in habitat use may be due to differences in infaunal prey densities or factors such as protection from predators. The small mangrove trees present in the natural fringe habitat may provide protection or the perception of protection from predators.

We conclude that the fringes of mangrove forests are an important feeding habitat for eagle rays with our results indicating that modified (trimmed) mangrove fringe may be of lower ecological value than unmodified fringe. We recommended that these findings be taken into account when the removal and maintenance of mangrove areas is considered in the future.

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## Introduction

Mangroves in many tropical areas are declining rapidly as a result of a number of anthropogenic activities, including, but not limited to, coastal development, agriculture and aquaculture (Laegdsgaard & Johnson, 2001; Walters et al., 2008). The ecological and economic benefits of tropical mangroves (ecosystem services) are numerous and well documented, and include carbon sequestration, the provision of fisheries-important nursery habitats and protection of coastlines from erosion (Ellison, 2008; Polidoro et al., 2010; Walters et al., 2008). Less well documented is the importance of mangrove habitat for larger predatory species, such as elasmobranchs (sharks and rays), which inhabit estuaries for part or all of their lives. Given the clear ecological value of mangrove habitat, the observed global decline of this habitat is of particular concern, especially in tropical regions (Dulvy et al., 2014; Rivera-Monroy et al., 2017). In New Zealand, there is some knowledge of the ecological value of mangroves for small species (reviewed by Dencer-Brown et al., 2018) whereas there is a distinct lack of information regarding the role of temperate mangrove habitat to larger species. This should be of concern given the widespread clearance of mangroves by resource management agencies throughout the North Island.

Mangrove forests have the potential to be important for elasmobranchs, both as feeding and protective habitat. The morphology of mangroves, with their close growing structure of often buttressed stems (in tropical regions), aerial roots and pneumatophores, may prevent large predators from accessing these areas, therefore providing protection for small species of fish and juveniles of larger species (Davy et al., 2015). Mangroves often support nurseries of teleost fishes, and therefore have the additional potential to be a productive feeding habitat for small piscivorous elasmobranchs (Kimirei et al., 2013; Robertson & Duke, 1987; Simpfendorfer & Milward, 1993). The sediments below and immediately adjacent to

mangroves typically contains a high mud content and has the potential to provide highly productive habitat for benthic and infaunal invertebrate species (Alfaro, 2006). In New Zealand, mangroves may constitute an important trophic resource for benthic feeding fish species such as flounder, parore, grey mullet and benthic feeding elasmobranch species such as eagle rays. However, our understanding of their importance is limited as there have been few investigations into the use of temperate mangrove areas in New Zealand by fish species (Dencer-Brown et al., 2018).

In addition to the value of mangroves as feeding habitat, batoid elasmobranchs may use mangroves as nursery habitat. Davy et al. (2015), using passive acoustic telemetry found that juvenile mangrove whiprays (*Himantura granulata*) moved into mangrove areas at high tide. It was suggested that this behaviour may have been driven by the threat from large sharks present in the area (Davy et al. (2015). White and Potter (2004) used gill nets in unvegetated areas immediately adjacent to dense and sparse stands of the temperate mangrove *Avicennia marina* in Shark Bay, Western Australia, to quantify numbers of elasmobranchs entering mangrove adjacent areas on the incoming tide. They found that the mean number of elasmobranch species caught and the mean catch rate at these sites was higher than at an unvegetated site within 200 m. They also found that the majority of individuals caught were juvenile, with umbilical scars, suggesting the area may be used as a nursery area (White & Potter, 2004).

While the loss of mangrove habitat has been relatively rapid in the tropics, in New Zealand, the endemic mangrove species, *Avicennia marina* var. *australasica* (hereafter called *Avicennia marina*), has been steadily expanding its coverage in estuaries within its biogeographic range (De Lange & De Lange, 1994; Lovelock et al., 2010; Lundquist et al., 2012; Morrissey et al., 2010; Park, 2004; Swales et al., 2008). This mangrove expansion has been correlated with an increase in sedimentation associated with changes in land-use from



native forest to agriculture, forestry and coastal/catchment urbanisation (De Lange & De Lange, 1994; Lovelock et al., 2007). To a lesser extent, the expansion in mangroves is also linked to nutrient enrichment (Lovelock et al., 2007). Public perception of mangroves is often negative, due to a perceived or real loss of water views, recreational and boating access, or habitat/biodiversity (De Luca, 2015; Green et al., 2003; Harty, 2009; Lundquist et al., 2014; Morrissey et al., 2007). Regional Councils are under increasing pressure to engage in, or support, various forms of mangrove management to prevent further spread; or to restore open sandflat areas that are perceived to be ecologically and recreationally more valuable (De Luca, 2015; Green et al., 2003; Harty, 2009; Lundquist et al., 2012; Lundquist et al., 2014; Morrissey et al., 2007). Consequently, there has been widespread removal and management (both legal and illegal) of mangroves in recent decades (Lundquist et al., 2012; Lundquist et al., 2014). The ecological consequences of mangrove management remain largely unknown, with particular paucity of information relating to the effects of mangrove removal on higher trophic levels (De Luca, 2015; Dencer-Brown et al., 2018).

Removal methods can be either manual (chainsaws) or mechanical (tractors with low pressure tyres) with waste material either removed or left *in situ* to degrade (with or without burning) (Lundquist et al., 2014). Management of mangroves in New Zealand is often carried out by Estuary Care groups made up of local volunteers, and consists of working parties removing seedlings and pneumatophores with hand-tools (Lundquist et al., 2014). The removal and management of mangroves generally results in a significant disruption to the sediment. Negative impacts on benthic infauna have been recorded immediately after management, followed by possible increases in abundance and diversity depending on sediment properties (Alfaro, 2006, 2010). The recovery of managed areas to the desired sandflat conditions post-removal has been varied. Observations of some sites more than 5 years following a removal event recorded much sandier conditions, however most sites

showed minimal recovery, retaining high proportions of muddy sediments (Lundquist et al., 2014). Where cut mangrove material has been left in place, or where the remnants of extensive, formerly well-established root and pneumatophore systems remain, there has been a persistence of anoxic, muddy sediments and low infaunal diversity and biomass (Lundquist et al., 2012). As the ecological value of mangroves in New Zealand is poorly understood (Alfaro, 2010; Dencer-Brown et al., 2018), the consequences of mangrove removal remains largely unknown.

One elasmobranch species known to use estuarine and potentially mangrove habitats in New Zealand is the eagle ray (*Myliobatis tenuicaudatus*). This species of eagle ray is not currently threatened and is considered ‘Least Concern’ by the IUCN red-list analysis (Kyne, 2016). However, habitat degradation, including mangrove removal, is considered one of the major risk factors for coastal elasmobranchs, particularly batoid (ray) species (Dulvy et al., 2014). Therefore, with the removal and management of mangroves in New Zealand a common occurrence, it is important to assess the use of mangrove habitats by this and other elasmobranch species.

*Myliobatis tenuicaudatus* is a small to medium sized myliobatid ray that exhibits a highly specialised method of extracting benthic and infaunal prey by using jets of water to excavate a pit (Gregory & Ballance, 1979). In intertidal areas these pits survive over a number of tides and can be used to quantify the feeding effort of this species (Hines et al., 1997). Feeding excavations of *M. tenuicaudatus* have been observed in the pneumatophore habitat immediately fringing *Avicennia marina* areas in several New Zealand estuaries (H. Jones, pers. comm), suggesting that this habitat is a profitable feeding area for this species. It has been shown that the fringing pneumatophore habitat of *Avicennia marina* habitat contains a higher diversity and abundance of organisms than sediments within the stands, although

lower than in sandflat areas (Alfaro, 2006), further suggesting that the fringe area is an important habitat.

Mangrove fringe is classified as the 30 m wide strip immediately adjacent to an area of dense *A. marina* trees. In an untrimmed, natural site, this area is often characterised by mangrove pneumatophores, mangrove seedlings, small plants (<1 m in diameter) and mud crab holes at varying densities. A trimmed edge may have as little as 50 cm of pneumatophore spread beyond the dense mangrove forest and seedlings are removed on a regular basis. In Tauranga Harbour, in northern New Zealand, eleven areas of mangrove were consented for management or removal in 2008 and 2009. Approximately 32 kilometres of mangrove edge are now trimmed in an attempt to prevent mangrove expansion (calculated using consent maps, courtesy of Bay of Plenty Regional Council and Google Earth). It was postulated that due to current management (manual removal of seedlings and pneumatophores) likely resulting in sediment disruption, and natural fringe habitat providing an aspect of protection from predators, feeding intensity by eagle rays would be lower in the trimmed fringe habitat compared to the natural (unmodified) fringe habitat

As part of a larger investigation into *M. tenuicaudatus* feeding habitats in Tauranga Harbour, the current study aimed to a) determine if rays are using mangrove fringe habitat for feeding and b) assess whether trimming of mangrove fringe alters the way in which rays use this habitat.

## Methods

### Study Area

The survey was conducted in the Matua sub-estuary, Southern Tauranga Harbour, New Zealand ( $37^{\circ}\text{E}$ ,  $175^{\circ}\text{S}$ , Figure 1). In this sub-estuary, an area of consented management trimming is located adjacent to an area of natural fringe, separated by a persistent channel ( $>50$  cm deep at lowest astronomical tide) (Figure 2). This channel provides equal access to both habitats.



*Figure 1. Satellite image showing Tauranga Harbour (North Island, New Zealand) indicating the location of the Matua sub-estuary study site. Image: Google, Digital Globe, CNES/Airbus*

## Survey methodology

Two fringe habitats were surveyed between January and August 2017 to compare *M. tenuicaudatus* feeding effort at natural and trimmed fringe mangrove fringe (Figure 2). In both managed (trimmed, Site B) and unmanaged (natural, Site A) fringe habitats, counts were performed at approximately two-week intervals between January and August 2017, exact observation dates and times were dependant on tides and weather conditions. In addition, on three occasions, pits were counted at an adjacent outer sandflat site (Site C) and on one occasion at an adjacent inner sandflat site (Site D) in order to determine whether pit densities in mangrove fringe were comparable to those in sandflat habitat, and whether the position within the embayment had any effect on pit densities.

A circular plot counting method designed by Hines et al. (1997) was used to quantify density of ray feeding pits (Figure 3). This methodology was also used for a parallel study of *M. tenuicaudatus* feeding in the wider Tauranga Harbour (Cadwallader, In Prep). In each habitat, feeding pits were counted in ten circular plots, each with a radius of 15 metres and an area of 707 m<sup>2</sup>. Plots were placed at varying distances along a 300 metre transect of the fringe zone using a random number generator to determine distance from the beginning of the transect. In the sandflat areas, plots were placed using a random number generator to determine compass direction and distance from a central point.

## Statistical analysis

To allow ease of comparison with the only previous work of this type, density was described as the number of pits per 707 m<sup>2</sup> plot. A Scheirer-Ray-Hare test (a non-parametric ranked variant of the two-way ANOVA (Dytham, 2011)) was used to compare pit density between habitats. A *post-hoc* pairwise Mann-Whitney U tested where differences between trimmed and untrimmed sites lay. All statistics were performed in SPSS (IBM, version 24). In

addition, densities of feeding pits in the two mangrove fringe areas were compared with adjacent outer sandflat habitats (Sites C and D) using the same methodology.



Figure 2. Matua sub-estuary, Tauranga Harbour. Shaded overlay shows the area consented for Mangrove (*Avicenna marinara*) removal and maintenance. Lines show survey transects, A) Untrimmed mangrove fringe, B) mangrove fringe within trimmed area. C) Adjacent outer sandflat area, D) Adjacent inner sandflat area. Consent area information courtesy of Bay of Plenty Regional Council. Image: Google.



Figure 3. New Zealand eagle ray (*Myliobatis tenuicaudatus*) feeding pits in Tauranga Harbour, New Zealand. A) a fresh pit in sandy sediment B) a slightly degraded pit in muddy sediment in a mangrove fringe habitat.



## Results

*Myliobatis tenuicaudatus* feeding pits were counted at both trimmed and untrimmed sites on 15 occasions between January and August 2017 (Table 1). The maximum observed density of pits was 82 pits per 707 m<sup>2</sup> plot, observed in the untrimmed site on 28<sup>th</sup> June. Minimum observed density was 0 pits per 707 m<sup>2</sup> plot. Pits were recognisable for a maximum of 6 tides at both sites.

Table 1. Dates of and mean pits per plot counted on each sampling event. \*Indicates an extra observation 24 hours following trimming event by hovercraft.. All observations were >4 days apart, pit-longevity at this site was observed to be maximum 6 tides and so all observational events were deemed independent for analysis

EVENT	DATE	TRIMMED	NATURAL	SANDFLAT (OUTER)	SANDFLAT (INNER)
1	14 <sup>th</sup> January	18	43.2	51.8	51
2	25 <sup>th</sup> January	0	0.2		
3	7 <sup>th</sup> February	0.9	6.4		
4	27 <sup>th</sup> February	0.1	11.9		
5	21 <sup>st</sup> March	0.3	4.4	34	
6	2 <sup>nd</sup> April	1.2	24.6		
7	20 <sup>th</sup> April	0	0.2		
8	10 <sup>th</sup> May	11.6	26		
9	28 <sup>th</sup> May	13.7	18.3		
10	7 <sup>th</sup> June*	0.7	8.2		
11	11 <sup>th</sup> June	0.4	3.7		
12	28 <sup>th</sup> June	12.5	52.1		
13	20 <sup>th</sup> July	0	2.5		
14	5 <sup>th</sup> August	0	1.2		
15	19 <sup>th</sup> August	1.1	4.5	14.6	
Mean		4.03	13.83	33.47	51

Combining all plots on observational events, a higher total number of pits were observed in the natural fringe zone with 2074 pits counted in the natural zone and 605 pits in the trimmed zone. Density was significantly higher in the natural fringe zone than the trimmed fringe zone (13.82 pits /plot versus 4.03 /plot; Scheirer-Ray-Hare:  $p < 0.001$ ; d.f. = 1, Figure 4). Pit density varied between sampling events ( $p < 0.001$ ; d.f. = 14) suggesting a possible seasonal

pattern. The time x treatment interaction term was not significant ( $p = 0.196$ ); indicating a consistent relationship in the difference between trimmed and untrimmed sites over the observational period. Post-hoc Mann-Whitney-U analysis showed significantly higher pit density at natural fringe zone than trimmed fringe zone on 11 out of 15 observations (Table 2, Figure 5).

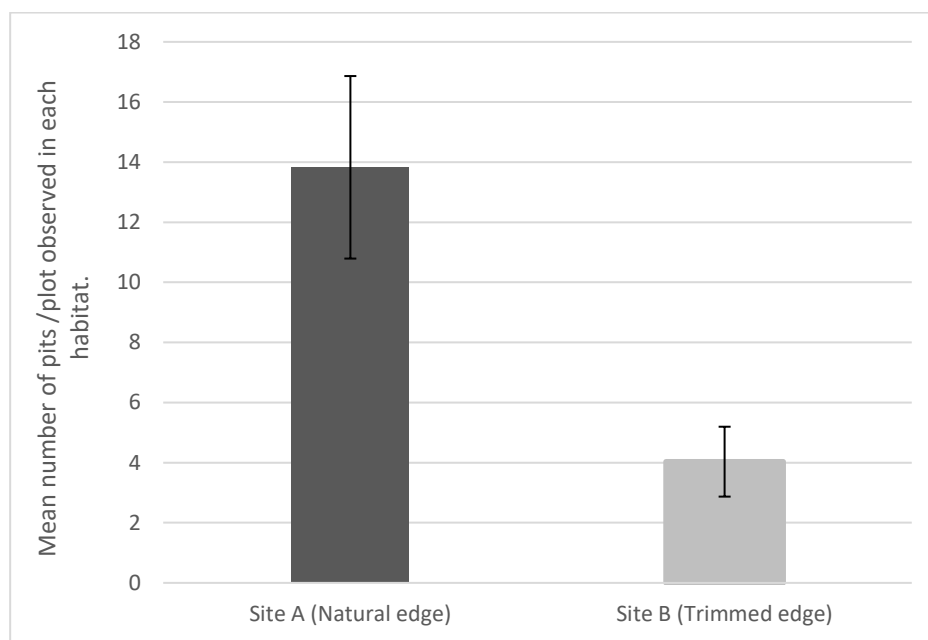


Figure 4. Mean number of pits /plot overall in natural edge and trimmed edge mangrove fringe habitats in the Matua sub-estuary, Tauranga harbour, New Zealand during 15 observational events between January and August 2017. Error bars  $\pm$  95% CI

When compared to feeding pit densities in an adjacent sandflat area (Site C , Figure 6), the Summer observation (January 14<sup>th</sup>) revealed significantly lower densities in the trimmed fringe site ( $U = 13$ ,  $p = <0.005$ ), whereas the natural fringe site showed no significant difference in pit density to the adjacent sandflat ( $U = 46.5$ ,  $p = 0.791$ , Table 3). In the Autumn and Winter observations (March 21<sup>st</sup> and August 15<sup>th</sup> respectively) density of pits at both natural and trimmed mangrove fringe zones were significantly lower than within the adjacent sandflat area (Natural:  $U = 4.0$ ,  $p < 0.001$ , Trimmed:  $U = 0$ ,  $p < 0$ , Table 3). On one occasion, during the summer observation (January 14<sup>th</sup>) an additional sandflat (Site D) was



surveyed in order to determine whether the effect of location within the estuary was a factor. No significant difference in pit density was found between the two sandflat areas C and D ( $U = 45$ ,  $p = 0.71$ ), nor between the inner sandflat (Site D) and the natural mangrove fringe zone ( $U = 40$ ,  $p = 0.45$ ). However the density of pits in the trimmed fringe zone was significantly lower than at the inner sandflat area ( $U = 7.5$ ,  $p = 0.001$ , Table 3).

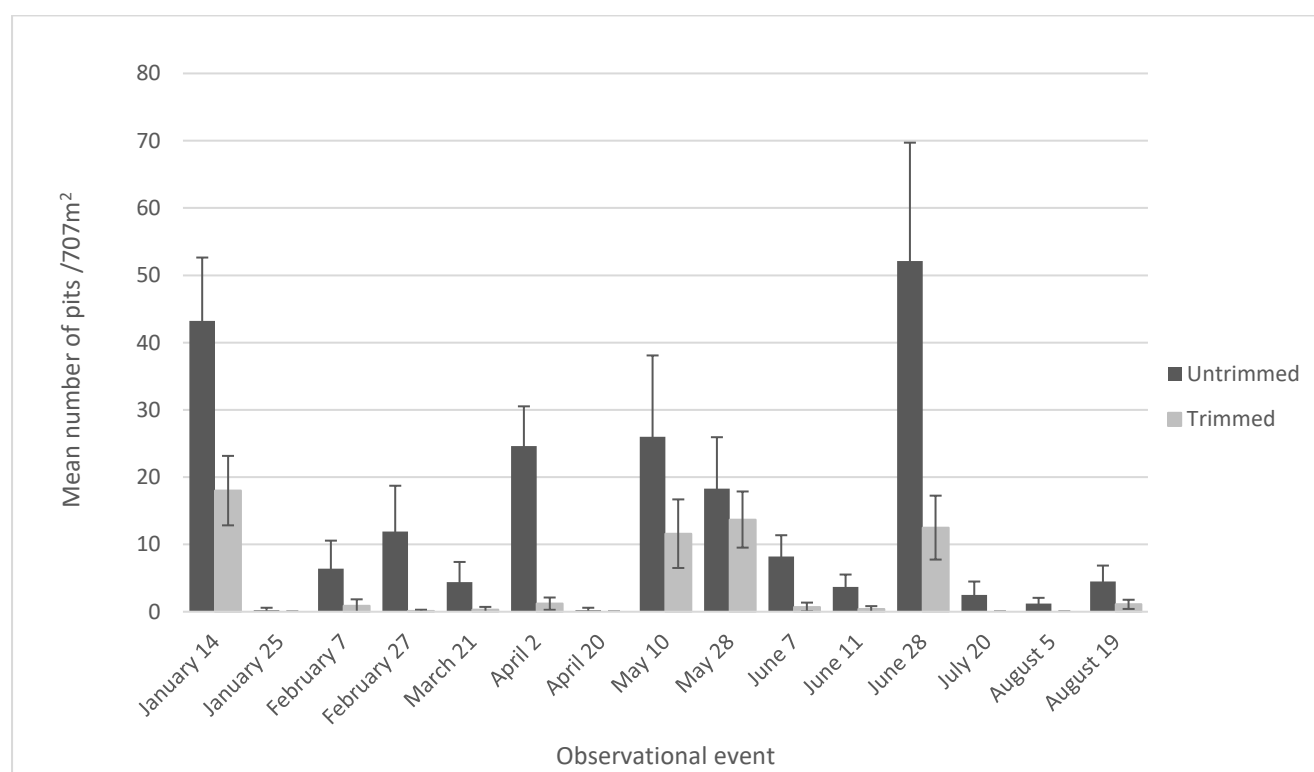


Figure 5. Mean number of pits per 707m<sup>2</sup> plot at trimmed and untrimmed mangrove fringe habitats in the Matua sub-estuary, Tauranga harbour, New Zealand during 15 observational events between January and August 2017. Error bars +/- 95% CI

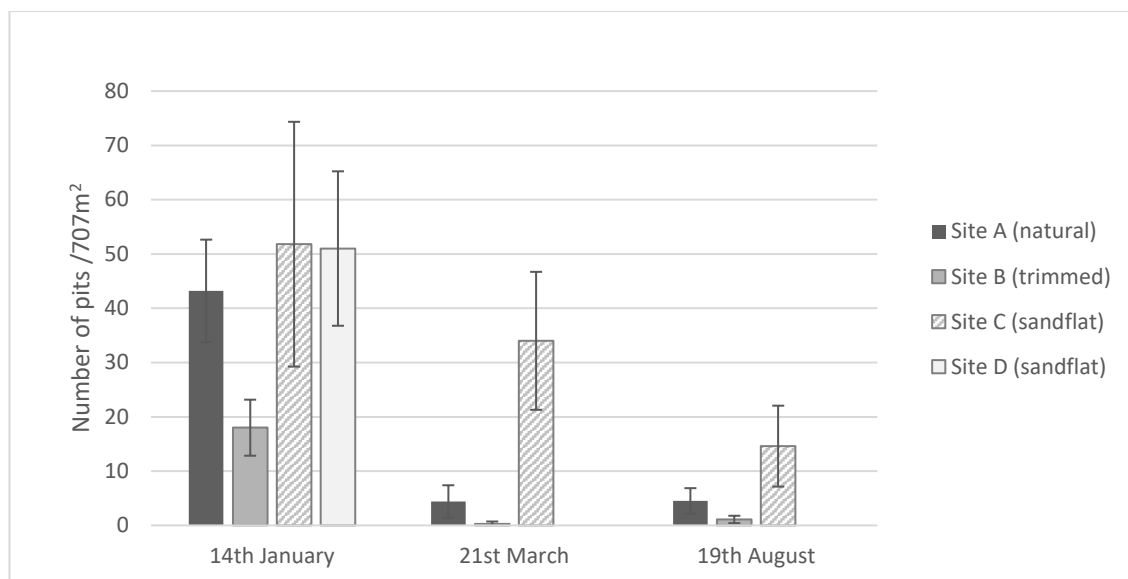


Figure 6. Showing mean number of *M. tenuicaudatus* feeding pits in mangrove (*Avicennia marina*) fringe habitats that have been left natural, been trimmed and two adjacent sandflat habitats. Error bars +/- 95% CI

Table 2. Post hoc Mann-Whitney U test analyses for density of ray feeding pits at a trimmed versus an untrimmed mangrove fringe site in Matua sub-estuary, Tauranga Harbour, New Zealand. \* indicates a significant difference between the sites.

EVENT	U	P
1	3.5	<0.001*
2	45	0.317
3	27.5	0.063
4	0.5	<0.001*
5	14.5	0.004*
6	0	<0.001*
7	45	0.317
8	22.5	0.037*
9	34.5	0.241
10	1.5	<0.001*
11	19.5	0.014*
12	15	0.008*
13	25	0.013*
14	20	0.005*
15	23.5	0.041*

Table 3. Matrix of Mann-Whitney  $U$  comparison of density of *M. tenuicaudatus* feeding pits between Untrimmed (A), Trimmed (B), Sandflat (C) and Inner Sandflat (D) locations in Matua Sub-Estuary, Tauranga harbour, New Zealand on three occasions during 2017. \*denotes a significant result. Note: Inner Sandflat was sampled on one occasion only.

	January				March			August		
	A	B	C	D	A	B	C	A	B	C
A		$U = 3.5$ $p < 0.001^*$	$U = 46.5$ $p = 0.79$	$U = 40.0$ $p = 0.45$		$U = 14.5$ $p = 0.004^*$	$U = 4.0$ $p < 0.001^*$		$U = 23.5$ $p = 0.041^*$	$U = 12.5$ $p = 0.005^*$
B			$U = 13$ $p = 0.005^*$	$U = 7.5$ $p = 0.001^*$			$U = 0$ $p < 0.001^*$			$U = 0.5$ $p < 0.001^*$
C				$U = 45$ $p = 0.71$						

## Discussion

The presence of feeding pits in the vicinity of mangroves on multiple occasions over a relatively long period, shows that *M. tenuicaudatus* are using mangrove fringe areas for feeding. The consistent reduced occurrence of feeding pits in the trimmed fringe zone indicates that this habitat may be of lesser value vs. natural fringe zone. Observation of *M. tenuicaudatus* during foraging periods, where animals swim, then stop, turn, and dig, has led to the conclusion that prey is located using electroreception, much like other ray species feeding behaviour on benthic prey (Blonder & Alevizon, 1988; Tillett et al., 2008), leading to the assumption that feeding pits are likely only dug when appropriate food items are present. It may therefore be the case that trimmed fringe is a lesser quality feeding habitat, with lower infaunal densities or altered species composition relative to a natural fringe. Lower infaunal densities or altered species composition may mean that it is less optimal energetically for animals to forage in these areas, leading to reduced use. Potential infaunal differences may be due to the repeated disturbance of the sediment during trimming events (Rossi et al., 2007). However, infaunal characteristics were not examined as part of the study.

Alternatively, if the infaunal density or species composition in both fringe habitats is similar then the observed difference in pit counts may be that the small (<1m diameter) bushes, seedlings and pneumatophores that characterise the natural fringe habitat may be providing

some protection from potential predators. Choice of foraging patch is thought to be not only a factor of food density but also predation risk in that patch (Ahrens et al., 2012). Orca (*Orcinus orca*) are a major predator of small elasmobranch species in New Zealand, particularly rays, (Visser, 1999). The presence of orca in the Tauranga Harbour often results in large aggregations of *M. tenuicaudatus* seeking refuge in shallow protected areas. It may be that the perception by the rays that the natural fringe area provides protection is driving a consistently higher use of this habitat.

The possibility that the lower densities in the trimmed fringe zone may be due to the position of the zone within the embayment, with the natural fringe zone being closer to the embayment mouth, seems to be countered by the summer (January 14th) sampling occasion including an inner sandflat area. There was no significant difference between densities of pits between the inner and outer sandflat areas, and the natural fringe zone, but all densities were significantly higher than that of the trimmed zone.

High variation in pit densities between the observational events is not unexpected. Seasonal differences in *M. tenuicaudatus* feeding pit density was initially suggested by Hines et al. (1997) over a short time period and has been confirmed in the Tauranga Harbour over a two year period (Cadwallader, In Prep). Additionally, feeding pit presence in intertidal areas in another New Zealand estuary (Whangateau Estuary) depended on time of day and meteorological conditions: fewer pits were present after high rainfall, and more pits were present during early morning low tides than during afternoon low tides (Le Port, 2003).

Similar variation may be occurring in Matua estuary. Furthermore, myliobatid rays can be highly mobile and some species cover large areas on a daily basis. For example, the cownose ray (*Rhinoptera bonasus*) is known to travel up to 50 km per day (Ajemian & Powers, 2014) and have daily core areas of use of up to 9.7 km<sup>2</sup> (Collins et al., 2007). The distance travelled by 10 *M. tenuicaudatus* individuals tracked using buoy-based GPS telemetry in a study by

Riding et al. (2009) can be calculated at an average of 1.34 km per hour, or 32 km per day. Therefore, low pit abundance in Matua during some periods of observation is likely due to the animals feeding elsewhere. Variation in pit density between observational events may also be due to presence or absence of large predators such as the Orca. Pods of Orca in the Tauranga Harbour are relatively common, but occur irregularly. Their presence may drive large numbers of rays into or out of certain areas within the harbour depending on their location. Despite this variation, the differences between feeding effort at trimmed and natural fringe zones remains consistent over all events with the interaction term effect not showing significance.

The significantly lower pit density in both fringe habitats compared with the adjacent sandflat area in the autumn and winter sampling events may be attributed to a higher abundance of preferred infaunal species in the sandflat habitat. Alfaro (2006) identified that pneumatophore zones in Matapouri Estuary, New Zealand, had an intermediate diversity and abundance of infaunal species, with sandflats higher, and inner mangrove habitat lower.

Barnes (2017) in Moreton Bay, Queensland, Australia, found a similar pattern in study sites with pneumatophore zones that bordered sandflats, but also found that when pneumatophore zones bordered seagrass (*Zostera* spp.) areas the biodiversity in the fringe habitat was comparatively high, similar to that in the seagrass area. The lack of difference between densities of feeding pits between the natural fringe zone and sandflat areas during the summer observation event may be due to a higher number rays generally over this time period resulting in increased competition for resources and effort was spread over a larger area.

From the results of this investigation, it appears that in terms of ecological value to *M. tenuicaudatus* measured by pit density, some sandflats are more valuable than natural mangrove fringe, which are in turn more valuable than trimmed mangrove fringe. Therefore,

if removal events result in increased suitable sandflat then this will in turn increase the availability of valuable feeding habitat for this species. However, this recovery has not been the case in all monitored mangrove removal locations and is reliant on many factors including but not limited to the hydrodynamics of the site and the removal methodology (Bulmer et al., 2017; Lundquist et al., 2017; Lundquist et al., 2012). In addition, the suitability of sandflat areas for *M. tenuicaudatus* is likely to be not purely dependant on abundance of food items. A parallel study surveying feeding habitats of *M. tenuicaudatus* has found that the density of pits and thus the feeding effort is highly dependent on location (Cadwallader, In Prep), and this is likely to be the case for the use of mangrove fringe. However, feeding pits observed in the mangrove fringe for this study as well as others outside of the study area, indicate that mangrove fringe is an important habitat for this species.

This study has confirmed the use of mangrove fringe as feeding habitat for *M. tenuicaudatus*. In addition, it is clear from the difference in pit densities between natural and trimmed mangrove fringe zones that the current management strategy of removal of seedling and pneumatophore to mitigate mangrove spread, may be result in a lower quality foraging habitat for *M. tenuicaudatus*. When extrapolated to the whole of the Tauranga harbour, the approximately 32 km of currently trimmed mangrove fringe results in over 95 hectares of degraded habitat assuming a fringe zone diameter of 30 m. Although *M. tenuicaudatus* is currently considered 'Least concern' by the IUCN red list (Kyne, 2016), the potential risk to coastal elasmobranch species is high (Dulvy et al., 2014). Therefore, this continued degradation of large areas of habitat for 'aesthetic consideration' is troubling and the use of an area by elasmobranch species should be included in considerations for future mangrove removal consents.

In conclusion, this study has confirmed the use of mangrove fringe as feeding habitat for *M. tenuicaudatus*. It is clear from the difference in pit densities between natural and trimmed mangrove fringe zones the current management strategy of seedling and pneumatophore removal to mitigate mangrove spread may result in a lower quality foraging habitat for *M. tenuicaudatus*. However, it is also clear that sand flat, at least in this location, may be a more valuable habitat for eagle ray feeding. Hence, the outcome of a recovery process, post mangrove removal, is critical to establishing whether more or less foraging habitat is available to stingrays. This study therefore indicates that the ecological and hydrodynamic processes that ensue following a mangrove removal campaign (Bulmer et al., 2017), must be investigated thoroughly before further mangrove management action is undertaken.

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