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# Retraction notice to 'Overexploitation of a seagrass-dominated fishery by fish fences in the Pacific Coral Triangle, Indonesia.' [*Marine & Freshwater Research* (2017) doi:10.1071/MF16236]

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Refers to: RETRACTED: Overexploitation of a seagrass-dominated fishery by fish fences in the Pacific Coral Triangle, Indonesia

*Marine and Freshwater Research*, published online 20 January 2017, doi.org/10.1071/MF16236. Ashley J. Endacott and Bruce Carlisle

After due consideration of various issues raised with respect to the Endacott and Carlisle paper, the co-authors unanimously agree to retract the above paper from *Marine and Freshwater Research*.

Reason: This paper is based on data collected as part of a long term project involving a wide team of researchers. It has come to light that there was inadequate consultation with the wider research team, and the appropriate permissions to publish these data had not been given. Therefore we retract our paper on data ownership grounds. We apologise for any inconvenience caused by our paper.

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***Perspective***

# Overexploitation of a seagrass-dominated fishery by fish fences in the Pacific Coral Triangle, Indonesia

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**Abstract.** Marine resources are vital to many coastal communities in South-east Asia, although their sustainable management is often neglected as they are perceived to be open access. In light of their small scale of operation, artisanal fisheries have been considered non-threatening to fish stocks and are frequently overlooked in comparison with industrial fisheries. In the present study, catch composition and fish fence attributes were measured from [illegible] fishing grounds on the island of Kaledupa, Indonesia, to investigate the effect of artisanal fishing on juveniles and ecologically important herbivores. The results do not support the expected differences between locations due to habitat type and quality, nor the expected influence of fence characteristics on the catch. However, there is a significant downward trend in catch size over a 10-year period. These results indicate that the characteristics of the fish community are being distorted by the fish fences. This suggests that the agreed fishery regulations are inadequate and this artisanal fishing technique is unsustainable.

**Additional keywords:** artisanal fishing, feeding guild, herbivory, juvenile, Malthusian overfishing, Marine Protected Areas.

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## Introduction

The Coral Triangle is an area of the west Pacific that spans the seas of Indonesia, Malaysia, Papua New Guinea, the Philippines, the Solomon Islands and Timor Leste. It is a global biodiversity hot spot with the highest diversity of coral reef fish and invertebrates on the planet (Briggs 2005). It accounts for only 1.1% of the world's area (Cabral *et al.* 2013), yet it contains 76% of reef-building corals (Veron 2000) and 37% of coral reef fish species (Allen 2008), along with the highest diversity of mangrove forests and seagrass beds in the tropics (Spalding *et al.* 2003, 2010).

In many coastal areas of South-east Asia, marine resources make up a large proportion of the protein requirements of the population (Exton 2005; Cullen 2007). This reliance is made evident through widespread overfishing and unsustainable resource usage (Newton *et al.* 2007). Overfishing, which is defined as taking too many fish too quickly (Duxbury and Duxbury 1996) and fishing above the sustainable limit (Wilkinson 2008), has resulted in over 87% of global marine fish stocks becoming exploited, overexploited or depleted (Pauly *et al.* 2002). Overfishing is a pressing issue in the Indo-Pacific (McClanahan *et al.* 2002; Smith *et al.* 2007). It is estimated that 95% of coastal fisheries in South-east Asia are at risk of being lost and, by 2050, all will be threatened (Burke *et al.* 2011). In Indonesia, 2.62 million people work in capture fisheries (Mallory 2015) and subsistence fishing provides ~70% of the protein requirements of the country (Resosudarmo 2005). Artisanal fisheries are low to very low investment and are important in subsistence and commercial activities (Berkes *et al.* 2001).

In light of their small scale of operation, artisanal fisheries have been considered non-threatening to fish stocks and are frequently overlooked in comparison with industrial fisheries (Hawkins and Roberts 2004). However, artisanal fisheries are often unselective in their catch, exploitative (Ashworth *et al.* 2004; Wilkinson 2004) and can trigger biodiversity loss (Wilkinson 2008). This occurs through the removal of individuals and trophic cascade effects (Bellwood *et al.* 2004; Exton and Smith 2011).

Fish fences, a type of fyke trap, are a widespread artisanal fishing method used across the world (FAO 2001), including Brazil (de Oliveira and Hanazaki 2011), the Caribbean (Dunn *et al.* 2010), Kenya (Mangi and Roberts 2006) and South-east Asia (Exton 2010). Fish fences are stationary structures consisting of wooden poles and nets that are sited upon seagrass beds near mangroves (Fig. 1). They are between 100 and 200 m in length, funnel shaped and with a landward-facing opening (Exton 2010). Fish fences are a low-effort, high-return form of fishing, being easy to erect (Ferry and Kohler 1987), inexpensive (Garrison *et al.* 1998) and able to be used on rough ground (Miller and Hunte 1987). They work by exploiting the natural movement of fish from the seagrass beds and mangrove forests to the reef during tidal recession, and coerce them into a collection point as the fish attempt to escape seaward.

Fish fences are highly unselective, being effective at catching various species (Stevenson and Stuart-Starkey 1980), including

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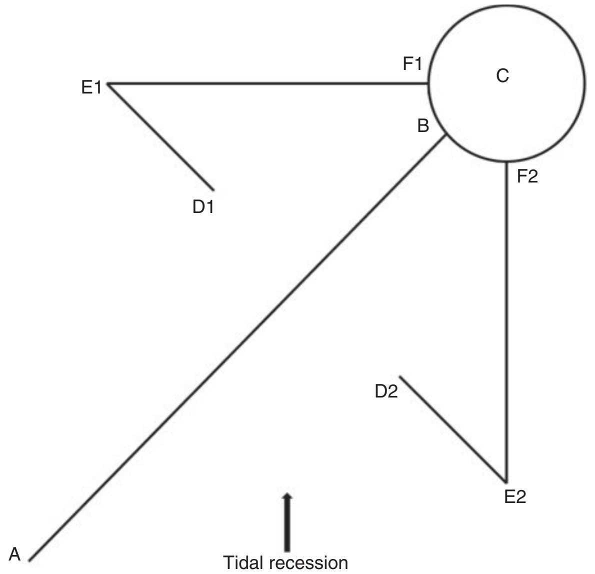


**Fig. 1.** Typical design of a fish fence. The dimensions vary, but the fish fences always follow this simple design. Spine length is calculated as the distance between A and B. Wing length is the combined length of D1→E1→F1 + D2→E2→F2. Aperture is the distance between D1 and D2. Mesh measurements were taken at collection point C only.

Acanthuridae, Balistidae, Carangidae, Clupeidae, Hemiramphidae, Lethrinidae, Mullidae, Nemipteridae, Scaridae and Siganidae (Exton 2010). This suggests that their effect on fish communities is likely to be significant (Robichaud *et al.* 19[illegible]; Exton 2010). Although certain species are [illegible] profitable at market, there is little concept of bycatch and juveniles are considered just as good a catch (Exton *et al.* 2015). Because they are primarily located on seagrass, fish fences have the potential to affect ecologically important herbivorous species. A survey conducted by Khalaf and Kochzius (2002) in the Jordanian Red Sea found that 79.9% of fish in seagrass beds were herbivores. Previous research has emphasised the [illegible] of herbivores in maintaining coral dominance over macroalgal benthic competitors (Mumby 200[illegible]). Hence, excessive fishing could have consequences for con[illegible] ecosystem health (Exton 2010). It is possible that fish fences can disproportionately affect juveniles. Seagrass beds are good nursery refuges for juveniles due to the complexity of their structure, the lower abundance of predators because of the distance from the reef, the decreased efficiency of predators due to the higher levels of water turbidity and the higher efficiency in intercepting planktonic larvae (Adams 1976; Nagelkerken *et al.* 2000*a*; Beck *et al.* 2003). Stanford (2008) found that the juvenile proportion of a fish fence catch can be as high as 100%.

When fishing pressure limits population renewal, fish assemblages are highly affected (Kronen *et al.* 2010) and overfished sites have assemblages with a recognisable reduction in the abundance and size of the fish species (McClanahan *et al.* 1999; Friedlander and DeMartini 2002; Sandin *et al.* 2008; Stuart-Smith *et al.* 2008). Despite this, there are few studies that have specifically looked at the association between fishing methods and the composition, diversity and selectivity of the catch (Dalzell *et al.* 1996; Pet-Soede *et al.* 2001; Campbell and Pardede 2006), with even fewer quantifying this relationship (McClanahan and Mangi 2004).

Most developing countries cannot afford effective monitoring, but this is essential to develop customised local fishery management strategies (Freire and Pauly 2005; Freire and Oliveira 2007; McClanahan and Castilla 2007). Better understanding of the effects of fish fences on a fish community is needed to allow strategies to be more precisely tailored to particular fisheries. The present study investigated the fish fence catch of two villages within the Wakatobi National Park (WNP). The catch composition of fish fences was determined in terms of species, feeding guild and maturation level; fishery health was assessed using biomass of fish catch; temporal changes in catch composition and fish fence attributes were examined and the effects of the physical attributes of fish fences on catch composition were also evaluated. The research allowed an assessment of the local fish community and improved our broader understanding of the ecological effects of fish fences.

It is hypothesised that: (1) the proportion of herbivore fish species caught within fish fences is higher than other feeding guilds; (2) fish fences situated on seagrass have a higher proportion of juvenile and herbivorous species than fish fences on other substrate; (3) over time, herbivorous and juvenile proportions have increased, biomass has fallen, mesh size has decreased and fish fence length has increased; and (4) there is an association between small mesh size and juvenile proportion within the catch.

## Materials and methods

### *Study site*

The WNP (Fig. 2) is one of fifteen Marine Protected Areas (MPAs) within the Pacific Coral Triangle. It was established in 1996 in response to the increasing use of destructive fishing practices in the area (Exton *et al.* 2015). Located south-east of Sulawesi (Clifton and Unsworth 2010), the WNP surrounds the four main islands of the Tukang Besi archipelago and forms the second largest MPA in Indonesia, covering 13 900 $km^2$, including 600 $km^2$ of coral reef (Exton *et al.* 2015).

The WNP supports rich and diverse marine communities (Turak 2003), with Halford (2003) identifying 590 species of fish. These levels of diversity are among the highest recorded in any coral reef ecosystem (Clifton and Unsworth 2010). Approximately 100 000 people live within the WNP, which makes it the most populated MPA in Indonesia (Pet-Soede and Erdmann 2003).

The present study took place on the fishing grounds of Lewuto and Peropa on the island of Kaledupa (Fig. 3). Kaledupa is economically reliant on fishing, accounting for 49.6% of the total number of fishers in the WNP (May and Coles 2004). Lewuto is on the north-east coast of Kaledupa, with the Bajau community of Sampela lying offshore. It is the main market town of Kaledupa. Peropa is situated on the south side of Kaledupa. The fishing grounds at Lewuto consist entirely of seagrass beds, whereas Peropa is a mixture of seagrass beds and reef flat. It is believed that at least 67% of fishers preferentially use seagrass (Unsworth *et al.* 2014). The fishing grounds of Lewuto and Peropa are easily accessed, and both villages have invested heavily in fish fences (May 2008).

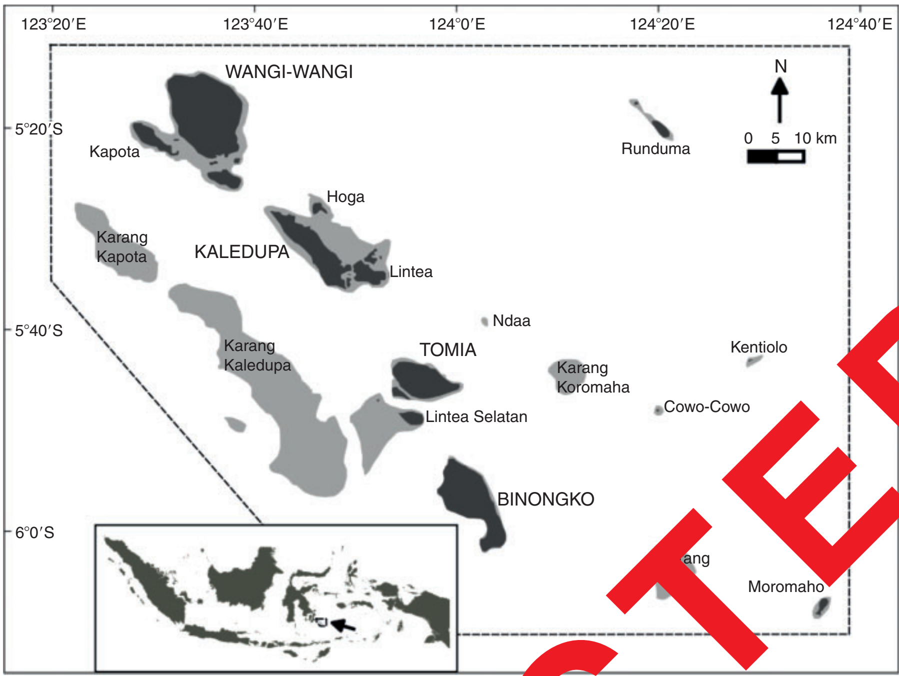


**Fig. 2.** Wakatobi Marine National Park, south-east Sulawesi. Map adapted from Clifton and Unsworth (2010).


**Fig. 3.** Location of the study sites on Kaledupa Island.

### Catch composition

Between 10 and 30 July 2014, 80 catches from the fish fences at Lewuto and Peropa were sampled using creel surveys. The monitoring of the catches took place daily, but at irregular times depending on the tides. When fishers returned to shore, all fish were measured and identified to species level.

The method for measuring fish lengths was adapted from Holden and Raitt (1974). Both total length (TL) and fork length (FL) were used. Fish lengths were measured to the nearest 0.5 cm. Fish were classified as juvenile if their length was less than one-third of that species' length at maturity ($L_m$; Unsworth *et al.* 2014). This was achieved using www.fishbase.org (accessed 14 October 2014).

The total catch was weighed to the nearest 0.1 kg. The biomass of each fish was calculated using the formula $aL^b$, where $L$ is the length of the fish and $a$ and $b$ are constants that are dependent on fish morphology and available from www.fishbase.org.

### Fish fence attributes

The catches came from 10 Lewuto fish fences and 10 Peropa fish fences. Several fish fence attributes were measured (Fig. 1). The length of the spine, wings and aperture of the fish fence were measured to the nearest 0.1 cm. Orientation to the shoreline and substrate were recorded. Mesh size was measured to the nearest 0.1 cm using callipers. This was measured in the collection point only because this is where fish are coerced and collected from. Twenty random mesh measurements were taken below the mean water level and averaged (Marine Management Organisation 2014).

### Historical data

Catch and fish fence data for the years 2004, 2007, 2009 and 2011 were provided courtesy of Operation Wallacea (D. Exton, pers. comm.). Analysis of temporal changes in catch per unit effort (CPUE) in the form of kilogram per fisher per day was not possible. Data were not to species level, and the surveyed villages were not specified. The attributes and abundance of fish fences around Kaledupa have also changed. This brings the reliability of using CPUE into doubt. Alternatively, we assess CPUE in the form of the number of individuals per landing per sampling year.

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### *Statistical analysis*

Statistical analyses were conducted using Minitab statistical software (ver. 17, Minitab, Inc., State College, PA, USA, see www.minitab.com) and IBM SPSS Statistics for Windows (ver. 22, IBM Corp., Armonk, NY, USA, see www-03.ibm.com/software/products/en/spss-statistics). Normality was tested using the Shapiro–Wilk method. All variables used in assessing the first three hypotheses deviated from a normal distribution and could not be normalised with transformations. Differences in the catch proportions of the feeding guilds, both number of individuals and biomass, were tested for using the Kruskal–Wallis test. Any significant results were further examined with SPSS's stepwise step-down multiple comparisons procedure to identify where significant differences occurred. Any differences in juvenile proportion and herbivorous proportion on seagrass and non-seagrass substrate were also tested for using the Mann–Whitney *U*-test. The Jonckheere–Terpstra test was used for temporal changes in the proportions of each feeding guild, the juvenile proportion of the catch and fish fence attributes, with stepwise step-down multiple comparisons of significant results. The juvenile proportions of biomass and number of individuals were normally distributed when grouped by village and substrate type, and Levene's test showed homogeneous variances ($F = 1.571$ ($P = 0.214$) and $F = 0.851$ ($P = 0.431$) respectively). So, a general linear model (GLM) was used to determine the relationship between fish fence attributes and juvenile catch.

## Results

### *Fish fence catch data*

In all, 262 species were identified from the fi[illegible]e catche[illegible] with carnivores, coralivores, detritivores, [illegible]bivor[illegible] invertivores, omnivores, piscivores and planktiv[illegible]ores[illegible] [illegible]ese, 85 species were identified as seagrass-[illegible]ociat[illegible]ecies (SAPs; *sensu* Unsworth *et al.* 2014). Mea[illegible]s.d.) fish a[illegible]dance was 63.9 ± 45.4 per catch. The mea[illegible]) number of species per catch was 18.4 ± 10.4. SAPs accoun[illegible] for 76.5% of the total fish catch, and for 74.8% [illegible]the catch fr[illegible]Lewuto and 78.2% from Peropa. The 10 m[illegible] abundant species are shown in Fig. 4.

### *Herbivore proporti[illegible]*

Herbivores c[illegible]ributed [illegible]45[illegible] to the total biomass and accounted [illegible] 1[illegible]3% of t[illegible] individuals caught. There were significa[illegible] differ[illegible] between guilds in terms of proportion of the biom[illegible] [illegible] = [illegible].388, $P = 0.000$). The herbivore proportion of biomass was significantly greater than that of coralivores, detritivores, piscivores and planktivores, but was significantly lower than the biomass of carnivores, invertivores and omnivores. There were also significant differences between guilds in terms of the proportion of individuals ($H_7 = 337.881$, $P = 0.000$). The herbivorous proportion of individuals was greater than coralivore, detritivore and piscivore individuals, but significantly lower than carnivores, invertivores and omnivores. Similar results were obtained when split by village, substrate and a combination of the two.

### *Seagrass catch composition*

Juveniles form 34.61% of the bio[illegible]s on se[illegible]rass, compared with 47.1% on non-seagr[illegible] (Table [illegible] Fo[illegible] per cent of the individuals on seagrass [illegible] juvenile, in c[illegible]ast with 49.57% on non-seagrass. Herbi[illegible]s ac[illegible]nt for 8.19% of the biomass on seagrass and 7.31% on [illegible]agrass [illegible]d 7.5% of the individuals on seagrass a[illegible]erbivor[illegible] co[illegible]ared with 10.94% on non-seagrass. Th[illegible] differences w[illegible] statistically significant, except for herbi[illegible]e b[illegible]ass.

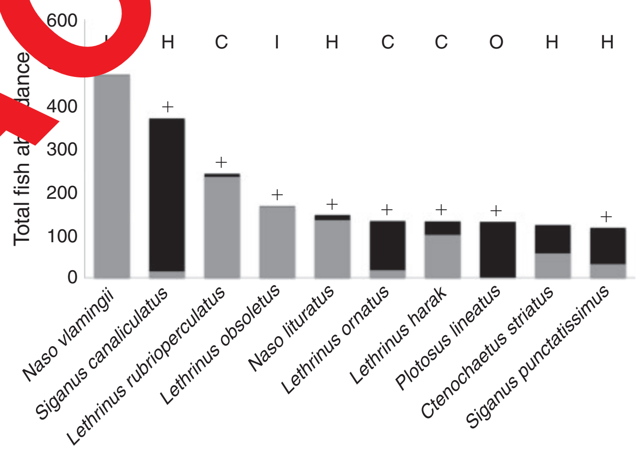


**Fig. 4.** Top 10 most abundant species recorded from catches at Kaledupa. Columns show the total number of fish caught (including juveniles), whereas the grey area within columns represents the juvenile proportion of the catch. +, seagrass-associated species; C, carnivore; H, herbivore; I, invertivore; O, omnivore.

**Table 1. Herbivore and juvenile proportion upon Kaledupa substrate**
Results of Mann–Whitney *U*-test significant at: *, $P < 0.1$, **, $P < 0.05$

| | Juvenile biomass (%) | Juvenile individuals (%) | Herbivore biomass (%) | Herbivore individuals (%) | Total biomass | Total individuals |
|---|---|---|---|---|---|---|
| Seagrass ($n = 53$) | | | | | | |
| Median | 34.61 | 40.0 | 8.19 | 7.5 | 5.36 | 44.0 |
| Non-seagrass ($n = 27$) | | | | | | |
| Median | 47.1 | 49.57 | 7.31 | 10.94 | 7.36 | 64.0 |
| Mann–Whitney *U* | 508.0 | 511.5 | 648.0 | 548.0 | 560.0 | 493.5 |
| Two-tailed *P*-value | 0.035** | 0.038** | 0.490 | 0.087* | 0.114 | 0.024** |

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### *Temporal changes*

The proportion of herbivore individuals increased from 0 to 8.9% between 2004 and 2014, but fluctuated in the intervening years (Table 2; Fig. 5). Over the same period, juveniles increased from 3.25 to 42.37%, but peaked at 46.36% in 2011. The proportion of herbivores and the proportion of juveniles was significantly higher in 2014 compared with 2004.

Total biomass fell from 11.9 kg per catch in 2004 to 6.24 kg in 2014, but was lowest in 2007 at 2.44 kg. Similarly, the number of individuals per catch declined from 444 in 2004 to 54 in 2014, but was lowest in 2011 at 32.5. Symbolic of increasing effort, fish fence length increased from 120 m in 2009 to 137.9 m in 2014. Mesh size increased from 1 cm in 2009 to 1.2 cm in 2014. No significant differences were observed for either variable.

### *Mesh size and juvenile proportion*

Two dummy variables were used to represent the village and substrate combination of a catch. Both of these were significantly correlated with juvenile biomass proportion (Lewuto: $r = -0.374$, $P = 0.000$; Peropa seagrass: $r = 0.210$, $P = 0.030$) and both were significantly correlated with the proportion of juvenile individuals (Lewuto: $r = -0.358$, $P = 0.001$; Peropa: $r = 0.229$, $P = 0.020$). Spine length was the only fence variable that had a significant correlation with the juvenile proportion of the biomass ($r = 0.233$, $P = 0.019$). Spine length was also the only fence variable significantly correlated with the juvenile proportion of individuals ($r = 0.246$, $P = 0.014$). Pearson's correlation coefficients showed no strong collinearity between predictor variables ($-0.6 < r < 0.6$). GLMs were constructed in

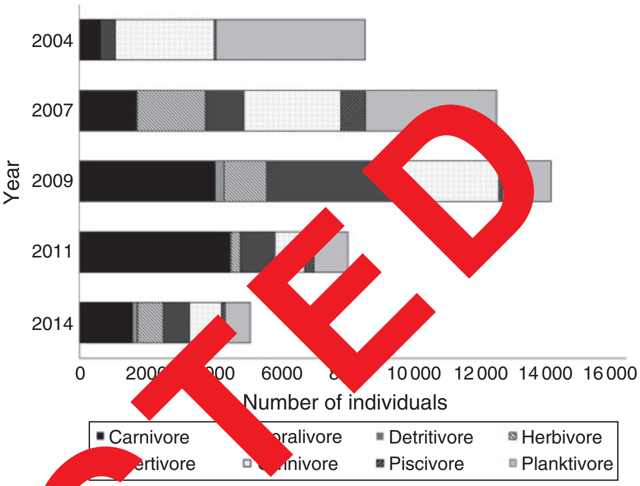


**Fig.** [illegible] Temporal [illegible]anges in the total number of individuals landed, split into d[illegible]ent feedi[illegible] guilds, for catches between 2004 and 2014.

**Table 2. Results of Jonckheere–Terpstra tests for temporal chang[illegible] in [illegible]res, juveniles, total individuals, total biomass, mesh and fish fenc[illegible]**

Median values that are significantly different between years ([illegible]ologically) are indicated by lower-case superscript letters

| | Test [illegible]t | Median values | | | | |
|---|---|---|---|---|---|---|
| | | [illegible]4 | 2007 | 2009 | 2011 | 2014 |
| Herbivore individuals | [illegible] = 40150.0<br>[illegible] = −3.169<br>[illegible] 0.002 | $0^{a}$ | 3 | $4.5^{a}$ | $1^{a}$ | $3^{a}$ |
| Herbivore biomass | $J$ [illegible]2.5<br>$z = 5.$[illegible]<br>$P = 0.000$ | $0^{a}$ | $0.08^{a}$ | No data | No data | $0.48^{a}$ |
| Within-catch herbivore propo[illegible] (%[illegible] | $J = 42875.0$<br>$z = -1.621$<br>$P = 0.105$ | 0 | 5.41 | 8.11 | 1.46 | 8.9 |
| Juvenile individua[illegible] | $J = 47103.0$<br>$z = 1.503$<br>$P = 0.133$ | 29.5 | 15 | 12.5 | 11.5 | 21 |
| Within-catch juven[illegible]oportion (%) | $J = 62018.5$<br>$z = 9.983$<br>$P = 0.000$ | $3.25^{a}$ | $17.03^{a}$ | 25 | 46.36 | 42.37 |
| Total individuals | $J = 35779.5$<br>$z = -5.555$<br>$P = 0.000$ | $444^{a}$ | $73^{a}$ | $49^{a}$ | $32.5^{a}$ | $54^{a}$ |
| Total biomass | $J = 4752.0$<br>$z = 2.548$<br>$P = 0.011$ | $11.9^{a}$ | $2.44^{a}$ | No data | No data | $6.24^{a}$ |
| Mesh size | $J = 25312.0$<br>$z = -1.152$<br>$P = 0.249$ | No data | No data | 1 | 1 | 1.2 |
| Fish fence length | $J = 669.5$<br>$z = -0.154$<br>$P = 0.878$ | No data | No data | 120 | 108 | 137.9 |

three blocks: first with just the village substrate dummy variables, second with spine length included and third with all fence variables included. For juvenile biomass proportion, the village substrate variables made a significant contribution to the model (adjusted $R^2 = 0.120$, $F = 6.380$, d.f. $1 = 2$, d.f. $2 = 77$, $P = 0.003$). Spine length and then other fence variables did not improve the model. Similarly, for the juvenile individuals proportion, the village substrate variables made a significant contribution to the model (adjusted $R^2 = 0.112$, $F = 5.986$, d.f. $1 = 2$, d.f. $2 = 77$, $P = 0.004$), and spine length and other fence variables did not improve the model.

## Discussion

The sampled fish fences remain effective at catching multiple species, and continue to catch juvenile fish at significant levels. Unsworth *et al.* (2014) documented 407 SAPs within the WNP, 85 of which were identified during the present study. Of particular significance is the abundance of *Naso vlamingii* (Acanthuridae; $n = 472$), *Siganus canaliculatus* (Siganidae; $n = 371$) and *Lethrinus rubrioperculatus* (Lethrinidae; $n = 242$) within the catches (Fig. 4), all of which are SAPs. *Lethrinus* spp. account for four of the ten most abundant species. Juvenile *N. vlamingii* and *S. canaliculatus* are known to be abundant within WNP seagrass (Unsworth 2010). However, a mean (±s.d.) 55.5 ± 41.7% of the 10 most abundant species are juveniles, with juvenile proportion of *N. vlamingii* and *L. rubrioperculatus* being 99.8 and 97.1% respectively. Acanthuridae and Siganidae are important to the reef system because of their functional role as herbivores, whereas Lethrinidae are carnivorous but are the most sought after food fish (Carpenter and Niem 2001).

It is known that populations that form dense spawning aggregations are particularly susceptible to exploitation by all forms of fishing (Myers and Ottensmeyer 2005). The abundance of *L. rubrioperculatus* could be explained by spawning activity. Research in the Ryukyu Islands, Japan, found this to occur between April and December (Ebisawa 1997). Similarly, the spawning season of *S. canaliculatus* in Palau has been found to occur between February and December (Hasse *et al.* 1977). These periods coincide with the data collection phase, possibly contributing to their over-representation. Moreover, species that form schooling aggregations are highly vulnerable to fishing because catch rates often remain high even though abundance is declining (Hilborn 2005). Schooling is another trait of *N. vlamingii* (Kuiter and Tonozuka 2001) and *S. canaliculatus* (Woodland 1990) that can explain their abundance within the catch. The results from the present study echo the findings of Unsworth *et al.* (2014): it is mainly SAPs that are removed, increasing the likelihood and scale of ecosystem and sustainability effects.

The findings of the study show that there are significantly more carnivores than herbivores within the catch, and that herbivore and juvenile catch proportion is significantly lower within seagrass than non-seagrass. The greater predator density has the ability to cause trophic cascades at the local level, characterised by the decline in prey species (Baskett *et al.* 2007). The lower abundance of herbivores could be due to the greater level of predation (Jennings and Polunin 1997). When carnivore levels are increased, herbivore levels decrease (Fig. 5). Alternatively, this trend could be attributed to the relocation of species of lower trophic levels to safer areas to avoid a higher risk of predation (Shepherd *et al.* 2010). Similarly, juveniles are known to use mangrove and reef habitats in close proximity to seagrass (Unsworth *et al.* 2008), providing an explanation for their lower abundance. However, the present study was constrained to a single season and only involved daytime sampling, therefore does not account for diel or seasonal changes in the use of seagrass.

Juvenile proportion has increased every year between 2004 and 2011 but fell in 2014, whereas herbivore proportion has fluctuated between 2004 and 2014. Total individuals have fallen markedly from 444 per catch in 2004 to 5[illegible] in 2014. These findings cannot be explained by fishing alone. Fish populations are also affected by factors such as habitat, location and availability of food items (Fenner 2014). Reefs with low abundances of fish are also associated with higher human populations, and more intensively populated areas (Fenner 2014). Larger and more p[illegible]ted areas have the ability to affect fish populations through increased sediment and nutrient run-off (Fenner 2014).

Such factors may explain the unexpected lower proportion of herbivores and juveniles on seagrass. This is supportive of previous research that identified higher herbivorous fish densities on the reef crest, where there is less sediment load (Purcell and Bellwood 2001; Wilson *et al.* 2003; Fox and Bellwood 2007). Research in the WNP has indicated that the Lewuto side of Kaledupa is subject to large sediment loads and reduced light availability (Crabbe and Smith 2002, 2003, 2005, 2006; Crabbe *et al.* 2004; Smith 2009). Our findings seem to support the claim that herbivory is suppressed by high densities of sediment (Bellwood and Fulton 2008). Although sedimentation is a natural occurrence, the levels here are largely affected and enhanced by anthropogenic activities (Smith 2009). Although it is known that seagrass acts as a nursery habitat for juvenile fish, it has been claimed that the loss of this habitat will not negatively affect the populations of juveniles that inhabit it. Simply, juveniles will move to another habitat (Mumby 2006). This is supported by Nagelkerken *et al.* (2000*b*), who identified that habitats close to the main reef of less than 3-m depth, including reef flats, can also provide nursery services to juvenile fish. These areas are less frequented by large predators due to the low water levels, while the living and dead corals provide sufficient refuge spaces to support large densities of juvenile fish (Nagelkerken *et al.* 2000*b*).

In response to the destructive power of fish fences in the WNP, a forum was held in 2009 to agree and implement a set of bylaws to manage the fishery. The key bylaws for fish fences include a minimum mesh size of 2.5 cm, a minimum 200-m distance between fences and the inclusion of a juvenile exit method (Exton 2010). Contrary to findings that show small mesh plays a role in catching juvenile fish, there was no statistically significant relationship between mesh size and juvenile proportion within the catch in the present study. This is despite all 20 sampled fish fences failing to comply with the minimum 2.5-cm mesh size, with the smallest measurement being 0.2 cm. Two fish fences in Lewuto and one in Peropa had nets with two layers of mesh. Doubling up further decreases the available space for fish to escape from the fish fences (Fig. 6).

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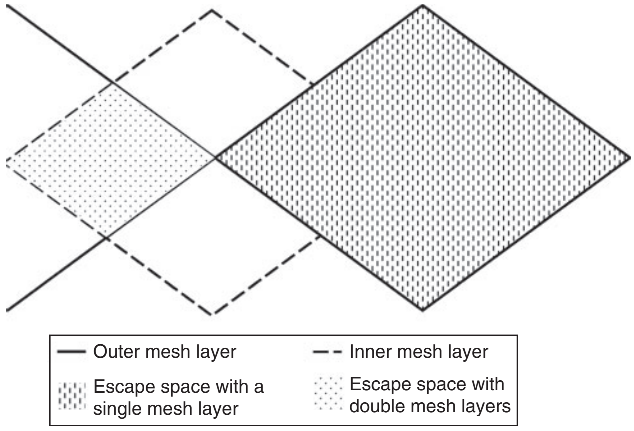


**Fig. 6.** Space for juveniles to escape from fish fences with single and double layers of mesh.

Although there was no noticeable difference in the mesh size of each layer, the mesh measurements in the situations where doubling occurs are overestimated. This is because the calliper prongs could only reach the outer layer, so were unable to account for the effect of the second layer.

The findings of the present study reveal that spine length is the fence characteristic most influential upon juvenile catch. This is possibly explained through the design of the fish fence where the spine protrudes into the areas normally inhabited by juvenile fish. Nevertheless, the mesh size on the spine is larger than that of the collection point. This provides suitable opportunities for juveniles to escape, so the effect of this feature cannot be explained based on these findings. For that reason, further research into the role this feature plays in determining juvenile catch needs to be conducted. The best predictors of juvenile proportion of biomass or individuals are the median values for each village substrate combination. That is, Lewuto seagrass, Peropa seagrass and Peropa non-seagrass. Fence characteristics have no effect and mesh size does not correlate with increased juvenile catch. However, only a narrow range of mesh sizes was present in the sampled fish fences in the present study, not allowing for differences to be investigated fully. It may be that Kaledupa is already overfished, meaning that herbivore and juvenile proportions are distorted.

Combining the escalation of fish fence numbers from 37 in 2002, to 100 in 2004, 119 in 2007 and over 200 in 2009 (May and Coles 2004; Exton 2011) and the reduction of mesh size by almost half between 2004 and 2007 (Exton 2010), it is easy to see why there is an increased herbivore and juvenile proportion. The clearest indication of overfishing at Kaledupa is seen in the CPUE for the years 2004–14 (Table 3). The total number of individuals caught increased between 2004 and 2009, but fell in 2011 and again in 2014. When paired against the number of catches sampled, it is possible to speculate that the outputs from the fishery are declining. Declining fish stocks and increasing effort are symbolic of collapsing fisheries (Froese *et al.* 2012; Watson *et al.* 2013; Miller and Russ 2014). For example, 8560 individuals were landed in 12 catches in 2004, whereas in 2014 it took 80 catches to reach just 5119 individuals. This is a time increase of 666.66%, yet the yield was only 59.8% of that in 2004. This itself helps explain the unexpected results: it points towards a skewed fish community with an irregular distribution leaving only a small segment to be sampled.

**Table 3. Individuals caught for the years 2004–14, an alternative analysis of catch per unit effort**

| Year | Individuals | Catches | Individuals per catch |
|---|---|---|---|
| 2004 | 8560 | 12 | 713.33 |
| 2007 | 12 501 | 75 | 166.68 |
| 2009 | 14 137 | 174 | 81.24 |
| 2011 | 8040 | 160 | 50.25 |
| 2014 | 5119 | 80 | 63.99 |

The increase in fish fences, reduced catch and the lack of alternative sources of income and protein suggest that Kaledupa is an example of Malthusian overfishing. Like many people in South-east Asia, the communities of Lewuto and Peropa are caught in a poverty trap, driven by a lack of alternative livelihoods and an inability to enter into more profitable income streams. Fishers are subsequently forced to remain in the fishery despite severe stock depletion and reduced catch (Dasgupta 199[illegible]; Cinner *et al.* 2009). It must be added that artisanal fisheries rarely operate on a purely subsistence basis (Altman *et al.* 2006); they interact considerably with the local cash economy, purposefully trying to catch more fish so as to meet their nutritional needs and to sell the surplus (Caughley *et al.* 1[illegible]).

However, the question remains as to what implications the results of the present study pose for the management of the WNP. It is clear that the current management does not work and change is needed. Successful management of a fishery involves the consideration of three interacting components: (1) the organisms (in the form of their taxonomy, ecology, population dynamics and life history, stocks, introductions and population reduction); (2) the habitat that supports the organisms (in the form of its limnology, water quality and quantity, structure and access); and (3) the people who interact with the habitat and extract the organisms (in terms of sociology, economics, politics, laws and regulations, planning, information and education; Nielsen 1999). Modern management incorporates techniques to benefit humans and to achieve sustainability usually through the regulation of human exploitation, habitat management and the manipulation of target species (Tyus 2012). Although this has occurred through the agreement of a set of bylaws in 2009, these bylaws have been largely ineffective.

Unlike other fishing methods that target certain species, it is unfeasible to consider a temporary exclusion on the use of fish fences. Rather, conventional policy options with understood rationale and outcomes are required (Bejarano *et al.* 2013). Endorsing periodic restrictions in designated areas around Kaledupa (Cinner *et al.* 2006), gear-based management measures (McClanahan and Mangi 2004) and enhancing the existing MPA status of the WNP are potential options.

The WNP was established as an MPA to close the habitat in order to improve yields and to enhance depleted or over-exploited fish stocks (Murray *et al.* 1999; Russ *et al.* 2004; McClanahan and Graham 2005). In theory, MPAs are the

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simplest method of managing overexploited fisheries (Roberts 1997). They work by enforcing top-down approaches to limit fishing pressure and the techniques used to extract fish (Exton *et al.* 2015). It has been suggested that the most successful MPAs incorporate no-take areas (NTAs), where resource extraction is firmly controlled (Mumby and Harborne 2010). When established successfully, NTAs can facilitate stock recovery and help protect biodiversity. NTAs also benefit neighbouring fishing grounds through 'spill-over' effects, which involve the movement of adults, larvae and eggs from recovered populations within NTAs to impoverished areas nearby (Gell and Roberts 2003). If fishing effort within these areas can be kept below the maximum sustainable yield, the benefits to fishers can be enhanced (Guidetti and Claudet 2010).

One major concern with fisheries management is that effort will be displaced elsewhere, so the reduction of pressure at one site will result in an increase at another (Halpern *et al.* 2004). It is often mistaken that artisanal fisheries can be managed in the same way as industrial and commercial fisheries (Castilla *et al.* 2007). The use of catch quotas is unsuitable in most scenarios due to the transience of landing sites, making policing such a scheme difficult (Exton 2010). This possibly explains why the WNP MPA has largely failed to address the issue of overfishing by fish fences as it focuses on commercial fishing rather than subsistence fishing. The solution is to reduce fishing capacity to an appropriate level by designating areas into limited, controlled or unfishable zones (Castilla and Defeo 2001). This has be[illegible] beyond the fishery. Epstein *et al.* (1999) found that 6 years [illegible] a small-scale no-use zone was implemented at Eilat's C[illegible]l Nature Reserve in the northern Red Sea, live coral cover w[illegible] threefold higher and significantly fewer co[illegible]ies we[illegible] partially dead.

The Kaledupan fishery has had no follow[illegible] fo[illegible]s s[illegible] the original one in 2009. In their absenc[illegible] it is n[illegible]common for fishers to selfishly pursue their o[illegible] personal int[illegible]s (Holling 2001), which often results in th[illegible] foraging model, where expenditure is minimised and r[illegible]source[illegible]traction is maximised (Aswani 1998). In situatio[illegible] where opport[illegible]ies are few, as seen in the WNP, this is lik[illegible] to result in the reduction of resources below a sustainable [illegible]d (M[illegible]anahan and Castilla 2007). The chance of reaching [illegible]eemen[illegible] among stakeholders and eliminating de[illegible]ctive g[illegible] is i[illegible]proved if resource users and managers [illegible] a[illegible]e. This, [illegible]wever, is wholly reliant upon forums [illegible] active [illegible]agement in order to achieve compliance.

## Conclusion

The use of fish fences around Kaledupa has been increasing over time. It has been assumed that the lack of selectivity is affecting ecologically important herbivores and juveniles. However, the results of the present study initially appear to contradict that claim after finding carnivores are the most abundant feeding guild. This may be because the fishery has overfished the other guilds, altering their relative abundance. The level of juvenile catch is still significantly high, but has not increased from the level in 2011. The bylaws agreed upon in 2009 are not adhered to and are not enforced. The increases in spine length, coupled with the small mesh size, despite not being significantly related to catch properties, are an indication of increased effort and have further reduced the selectivity of fish fences. This implies that the emphasis of management needs to be on gear restriction, with better enforcement and monitoring. The findings of the present study could act as a stimulus for improved conservation, monitoring and regulation of the WNP.

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