Tidepool Fish Assemblages at Wai`Opae Marine Life Conservation District, Hawai`i: Monitoring the effects of mangrove eradication on nearshore fish assemblages

A preliminary report prepared for Malama O Puna

by
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INTRODUCTION

Mangrove forests are found in intertidal zones along coastlines, lagoons, and estuaries of tropical and subtropical regions throughout the world. Within their native ranges, mangroves provide valuable ecosystem services, such as sediment retention (Victor et al. 2004) and provision of habitat for fish, shrimp, and crabs (Manson et al. 2005). However, the ecological value of mangroves may be limited when mangroves are introduced to areas outside their native range (Demopoulos et al. 2007). The red mangrove (*Rhizophora mangle*) was introduced to the island of Molokai, Hawaii, in 1902. Since then, red mangroves have quickly spread to neighboring islands, where established mangrove populations are having negative impacts on coastal habitat. Mudflats that were once valuable habitat for native water birds are now overgrown with mangrove forests. Colonization of coastal areas has also resulted in altered hydrological cycles and damaged Hawaiian archeological sites (Allen 1998). A preliminary study has also suggested that dense stands of invasive mangroves may influence community structure of nearshore fishes by providing habitat that is more beneficial for non-native fish (Vander Veur and Beets 2006).

The rapid spread of red mangroves throughout the five main Hawaiian islands (i.e., Kauai, Hawaii, Maui, Molokai, Oahu) has been attributed to: 1) the amount of coastal habitat available for invasion due to the absence of native mangroves and 2) the lack of organisms that feed on mangrove leaves and propagules (Allen and Krauss 2006, Chimner et al. 2006). Recent efforts to remove or control these invasive trees have been costly and time consuming. For example, 20 years of volunteer efforts and 2.5 million dollars were required to mechanically remove > 20 acres of red mangrove from the Nu`upia Ponds Wildlife Management Area on Oahu (Rauzon and Drigot 2002). Clearly, more cost effective and labor efficient techniques are needed.

Malama O Puna recently acquired funding to eradicate mangroves along the shoreline adjacent to the Wai`Opae Marine Life Conservation District using glyphosate and imazapyr. The US Fish and Wildlife Service requested that Malama O Puna monitor fish assemblages to determine if and how mangrove eradication would impact nearshore fish. As a result, researchers from the USDA Forest Service were contracted to monitor the fish community before and after the mangrove eradication. Results presented in this report are a preliminary assessment of the fish community structure before and after herbicides were applied to eradicate mangroves. Fish monitoring may continue in May of 2011, pending funding.

METHODS

Study Site

The Wai`Opae Marine Life Conservation District is located on the eastern coast of the island of Hawai`i. Designation as an MLCD was granted by the Hawaii Department of Land and Natural Resources in 2003 and prohibits all fishing and collecting of marine life within the area. Tide pools and coral gardens within the Wai`Opae MLCD are regarded as important nursery habitat for fishes and foster high diversity of corals and fish. These tide pools are also an important component of Hawaii's tourism industry as they attract 1000's of snorkelers each year. Surrounding the tide pools are dense stands of red mangrove trees intermixed with smaller stands of native hau (*Hibiscus tiliaceus*) and milo trees (*Thespesia populnea*).

Study Design

The impacts of mangrove eradication on fish communities were examined using a before-after-control-impact (BACI) design. Fish were sampled from six mangrove, three native vegetated, and six open tide pools in 2008 (before mangrove eradication) and again in 2009 and 2010 (after mangrove eradication). Mangrove pools were tide pools adjacent to mangrove stands, native vegetated pools were tide pools surrounded by milo and hau, and open pools were tide pools with >50% of pool edge consisting of un-colonized basalt. After the initial sampling period in 2008, the majority of mangroves (16 acres) were eradicated from June of 2008 until May of 2009 from a coastal area surrounding three of the mangrove tide pools (impacted); the remaining mangrove, native vegetated and open tide pools functioned as control pools. Immediately after the sampling period in 2009, remaining stands of mangroves (1 acre) were eradicated from the end of June 2009 until May of 2010. This resulted in two groups of impacted pools: 2009 mangrove (sampled immediately and one year after mangrove eradication) and 2010 mangrove (sampled immediately after mangrove eradication). Native vegetated and open pools functioned as control pools. From here on, these four groups of pools will be referred to as **2009 MN**, **2010 MN**, **native vegetated**, and **open**, respectively (Fig. 1).

Two independent techniques were used to monitor fish communities in the Wai`Opae tide pools: visual surveys and fyke nets. Visual surveys allowed us to determine if fish abundance and community composition in tide pools changed after mangroves were eradicated. Fyke nets allowed us to examine abundances and community composition of fish assemblages accessing the flooded vegetated surfaces and if mangrove eradication would impact them.

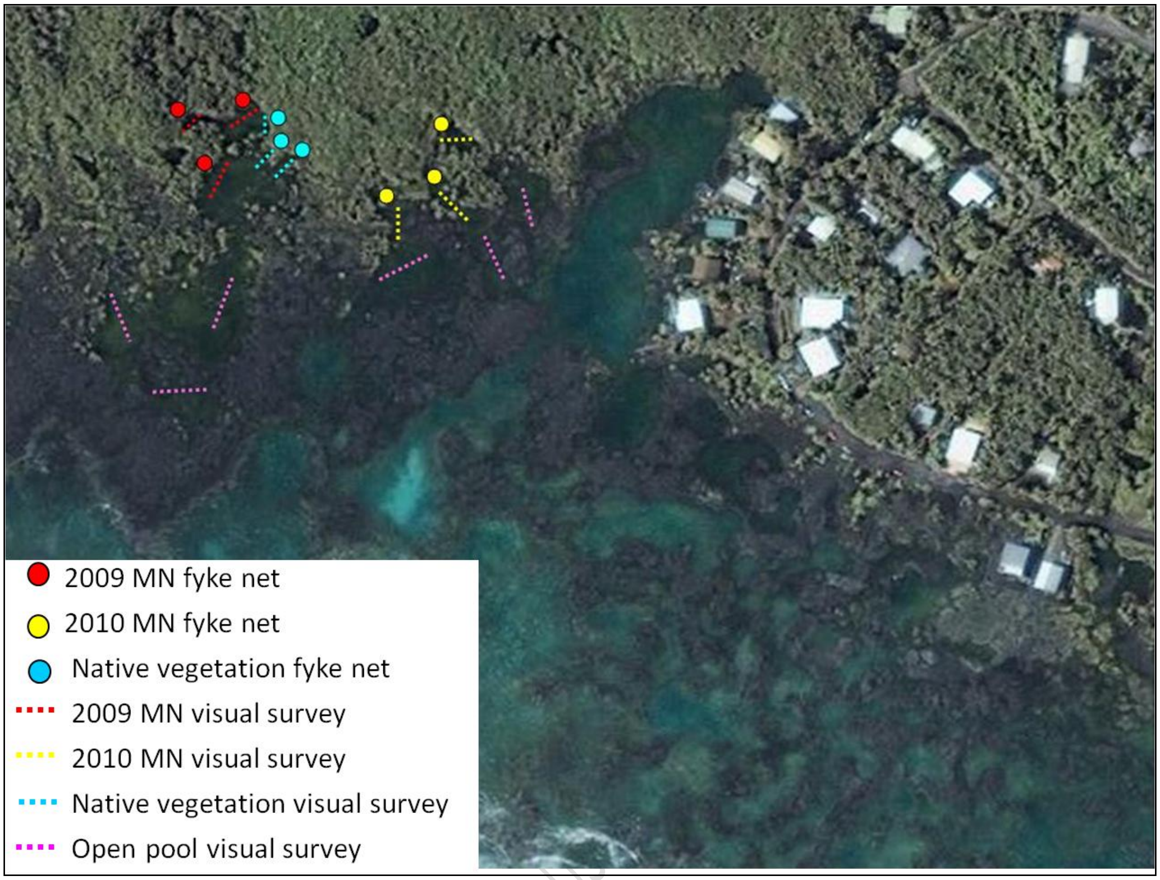

Figure 1. Sampling stations within the Wai`Opae MLCD. All sites were sampled each year (2008, 2009, 2010).

Visual Surveys

Fish communities were sampled using visual surveys conducted in all four treatments during high tides in May of 2008 (before mangrove eradication) and June of 2009 and 2010 (after mangrove eradication) using standard belt transect methodology (Brock 1954) in each of the four treatment pools. Transects were 25 m long x 2 m wide, with 3-m buffers on each end. In the event that a tidepool could not accommodate a transect survey of that size, the transect was proportionally downsized to fit the pool. In the vegetated treatments, visual surveys were conducted near the locations of the fyke nets. In the larger open tidepools, two 25 m long x 2m wide belt transect visual surveys were conducted simultaneously. The visual surveys recorded fish species, total length, and noted any juvenile coloration if present.

Fyke Nets

Fish communities were also sampled from the flooded surfaces adjacent to the 2009 MN, 2010 MN, and native vegetated pools using 3-mm mesh, mini-fyke nets. Fyke nets were deployed during the full moon phase in May of 2008 (before mangrove eradication) and in June of 2009 and 2010 (after mangrove eradication). Three fyke nets were deployed into the three replicate pools of each treatment on consecutive days. Fyke nets were placed directly in front of the vegetation at low tide with the openings facing the vegetated areas. The side wings of the nets were secured to the substrate using daisy chain knots, thereby minimizing the area obstructed from fish and invertebrates attempting to access the flooded mangrove prop roots or native vegetation. At or near slack high tide, the wings of the fyke net were released and lifted through the water so that the fish exiting the vegetation with the falling tide would be caught in the net. Nets were then sampled at the next low tide and fish were identified to species, weighed to the nearest mg, and total lengths measured to the nearest mm.

Physicochemical parameters

Temperature, specific conductivity (mS), and dissolved oxygen concentrations (DO mg/L) were measured in each of the three vegetated treatments (2009 MN, 2010 MN, and native vegetated pools) each year using YSI 600XLM data sondes. Two sondes were deployed into two of the three replicate pools for each treatment on consecutive days by attaching to the sides of the fyke nets at or near slack high tide.

Nutrients were measured from all four treatment pools (2009 MN, 2010 MN, native vegetated pools, open pools), but only during 2009, post mangrove eradication. Fifty mL of water was sampled from each pool at or near low tide to minimize mixing from adjacent pools. Water samples were then filtered through a GF/F filter into acid washed centrifuge tubes and returned to the laboratory on ice where they were stored frozen until they could be analyzed. Nutrients were measured on a Technicon Pulse 2 Autoanalyzer and included ammonium (NH_4^+) (USGS 1-2525-89), nitrate and nitrite (ΣNO_3^-) (USEPA 353.2), soluble reactive phosphorous (PO_4^{3+}) (USEPA 365.1), total nitrogen (TN) (Shimadzu TNM-1), and total phosphorous (TP).

Statistical Analyses

Water temperatures, dissolved oxygen concentrations, specific conductivity, and pH were compared from similar tidal cycles (full moon, spring tide) at similar times of the day (18:00). Comparisons were made near dusk to minimize the influence of the sun on water temperature and dissolved oxygen. Water temperatures, dissolved oxygen concentrations, and specific conductivity levels from 2008, 2009, and 2010 were averaged from 18:00 to 19:00. Average values were then compared among years (2008, 2009, 2010) as well as among treatments (2009 MN, 2010 MN, native vegetation) using a two- way analysis of variance (ANOVA). Nutrients were compared among treatments using a one way ANOVA.

Fish densities (no/m^2) from visual surveys were determined by dividing the total number of fish observed along a transect by the area of that transect. Fish densities units from fyke net data were catch per unit effort (CPUE). Native and exotic fish densities from visual surveys and fyke net samples were first individually compared among treatments and between years using a two-way ANOVA to determine if mangrove eradication had impacted fish assemblages in the Wai`Opae tide pools. A second set of two way ANOVAs were then conducted on the 2008, 2009, and 2010 visual survey and fyke net data sets to examine if community composition of fish assemblages (native vs. exotic fish) differed within the different tide pool treatments sampled. Factors included fish type (native vs. exotic), treatments, as well as the two way interactions.

Species richness was determined by summing up the total number of fish species observed or collected from each site for each year. Species richness for visual surveys and fyke nets were then compared between years as well as among sites using a two-way ANOVA.

Size classes of flagtails (*Kuhlia xenura*), the most abundant native species present in fyke net samples, were compared between years and among treatments using a two-way ANOVA. Size classes were pooled within each treatment in each year and average lengths were then compared.

Fish densities from both visual surveys and fyke net samples did not meet assumptions of normality and equal variance and were therefore (log +1) transformed prior to statistical analysis. Post-hoc comparisons were conducted using Fisher's Least Square Differences and all statistical analyses were performed in a generalized linear model (PROC GLM) from SAS 9.1 (2002. SAS Institute, Cary, South Carolina) at an α level of 0.05.

RESULTS AND DISCUSSION

Physicochemical parameters

Temperature, specific conductivity, dissolved oxygen (DO), and pH were all similar among the three different vegetated treatments (Figs. 2-5); fluctuations were attributed to tidal input and time of day. Temperature was generally higher and specific conductivity, dissolved oxygen, and pH lower in 2009 and 2010 than in 2008. Statistical comparison of values from dusk revealed that average values were not significantly different among sites (Fig. 6). However, there were significant differences when average specific conductivity ($p < 0.01$, $F = 8.8$, $df = 2$), temperature ($p < 0.001$, $F = 15.6$, $df = 2$), DO ($p < 0.01$, $F = 7.03$, $df = 2$), and pH values ($p < 0.01$, $F = 9.6$, $df = 2$) were compared among years. *Post hoc* comparisons revealed that differences in specific conductivity, temperature, DO, and pH were due to significantly fresher, warmer, less-oxygenated, and acidic waters in 2009 compared to 2008.

The lack of differences among the different tide pools treatments was likely due to mixing of waters between the different tide pools that occurred at high tides. The significantly fresher, warmer, more acidic waters in 2009 compared to 2008 were attributed to increased inputs of geothermally warmed groundwater inputs in 2009 that would have also been lower in dissolved oxygen. While this may have been due to reduced tidal inputs, tides were actually higher in 2009 (0.78 ± 0.01 m) than in 2008 (0.76 ± 0.0 m). This suggests that the eradication of mangroves around Wai`Opae tidepools contributed to increased inputs of groundwater as groundwater would no longer be lost by uptake and/or evapotranspiration by mangrove trees. The lower pH and DO values in 2009 may have also been influenced by the decomposition of the massive amounts of leaf litter that fell into the pools; higher temperatures may have been influenced by the loss of tree canopy surrounding the pools and increased inputs of solar radiance. This was evident by massive amounts of benthic algae growing on rocks in 2009 that were absent in 2008. In 2010, water was still fresher, warmer, and more acidic

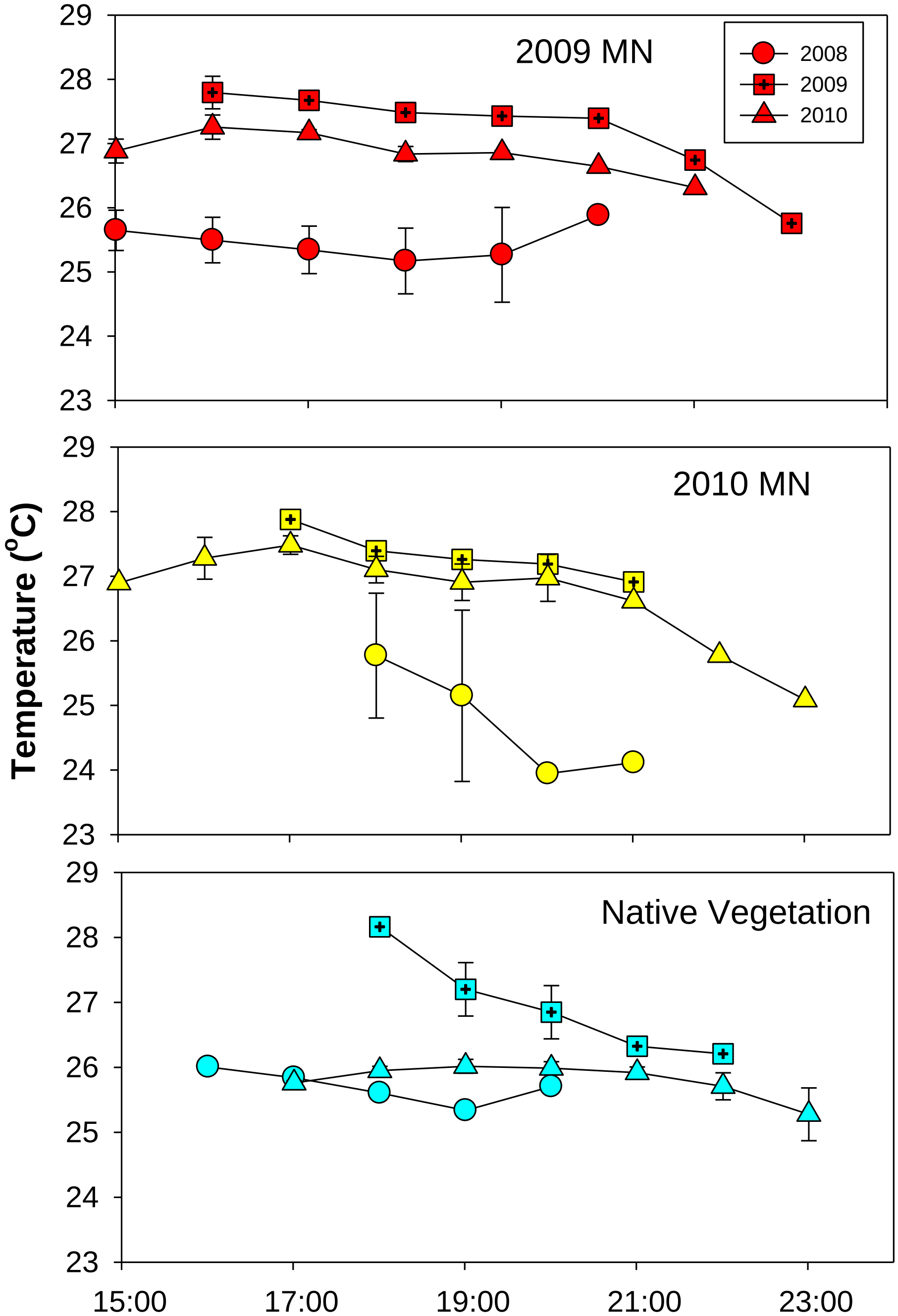

Figure 2. Average water temperature values (^{o}C) measured from the three vegetated treatments (2009 MN, 2010 MN, native vegetated pools) measured in 2008, 2009, and 2010.

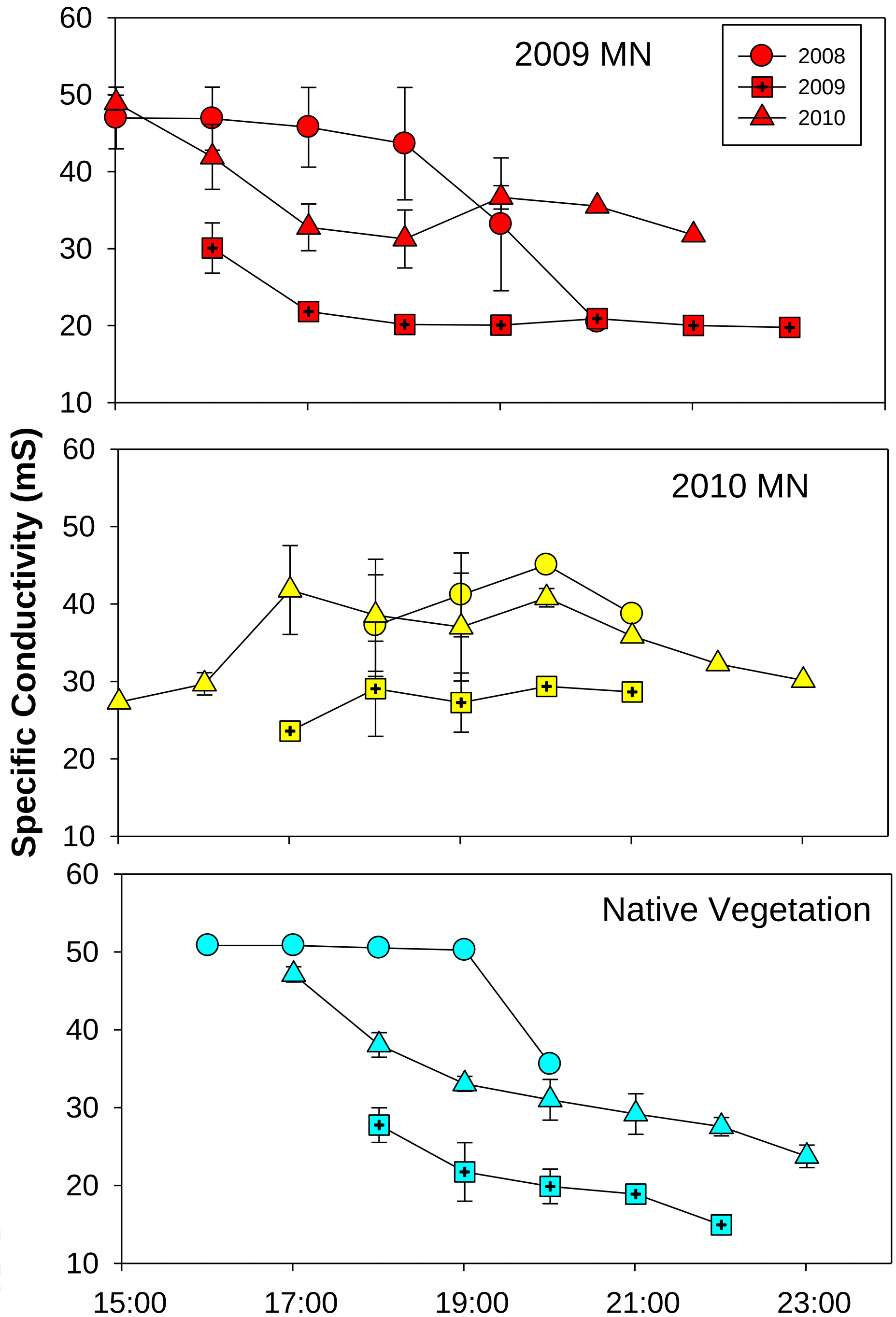

Figure 3. Average specific conductivity values (mS) measured from the three vegetated treatments (2009 MN, 2010 MN, native vegetated pools) measured in 2008, 2009, and 2010.

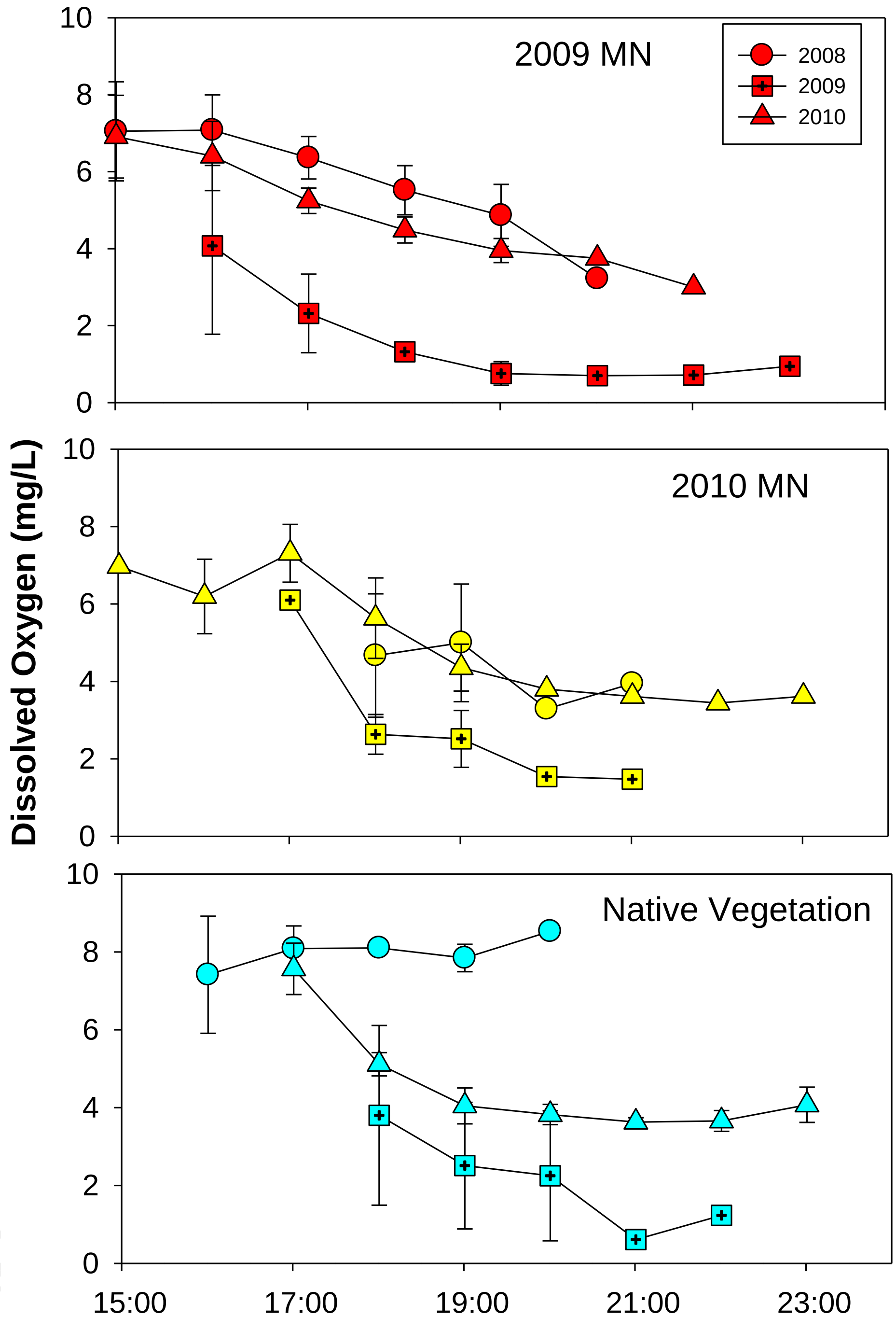

Figure 4. Average dissolved oxygen concentrations (mg/L) measured from the three vegetated treatments (2009 MN, 2010 MN, native vegetated pools) measured in 2008, 2009, and 2010.

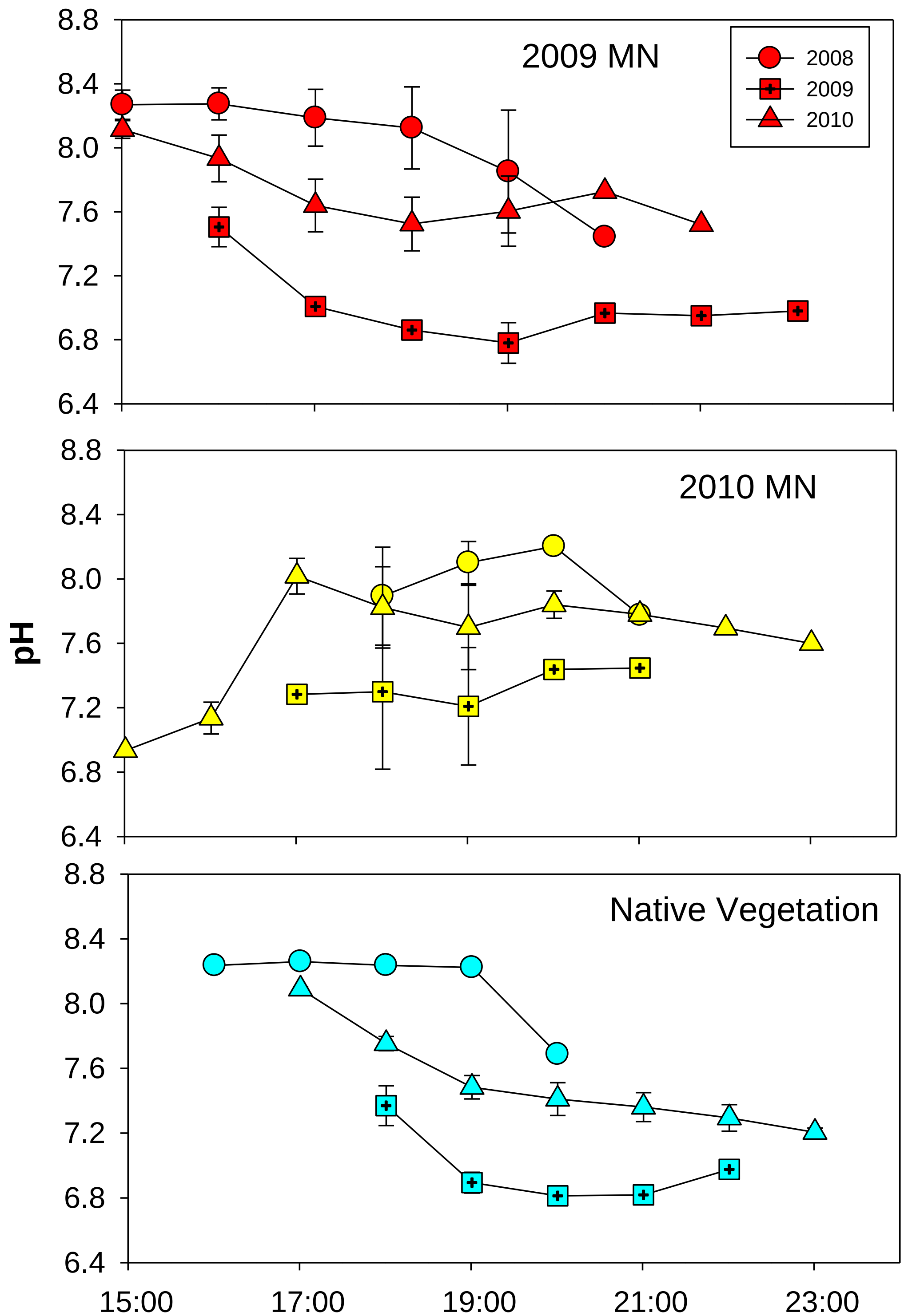

Figure 5. Average pH values measured from the three vegetated treatments (2009 MN, 2010 MN, native vegetated pools) measured in 2008, 2009, and 2010.

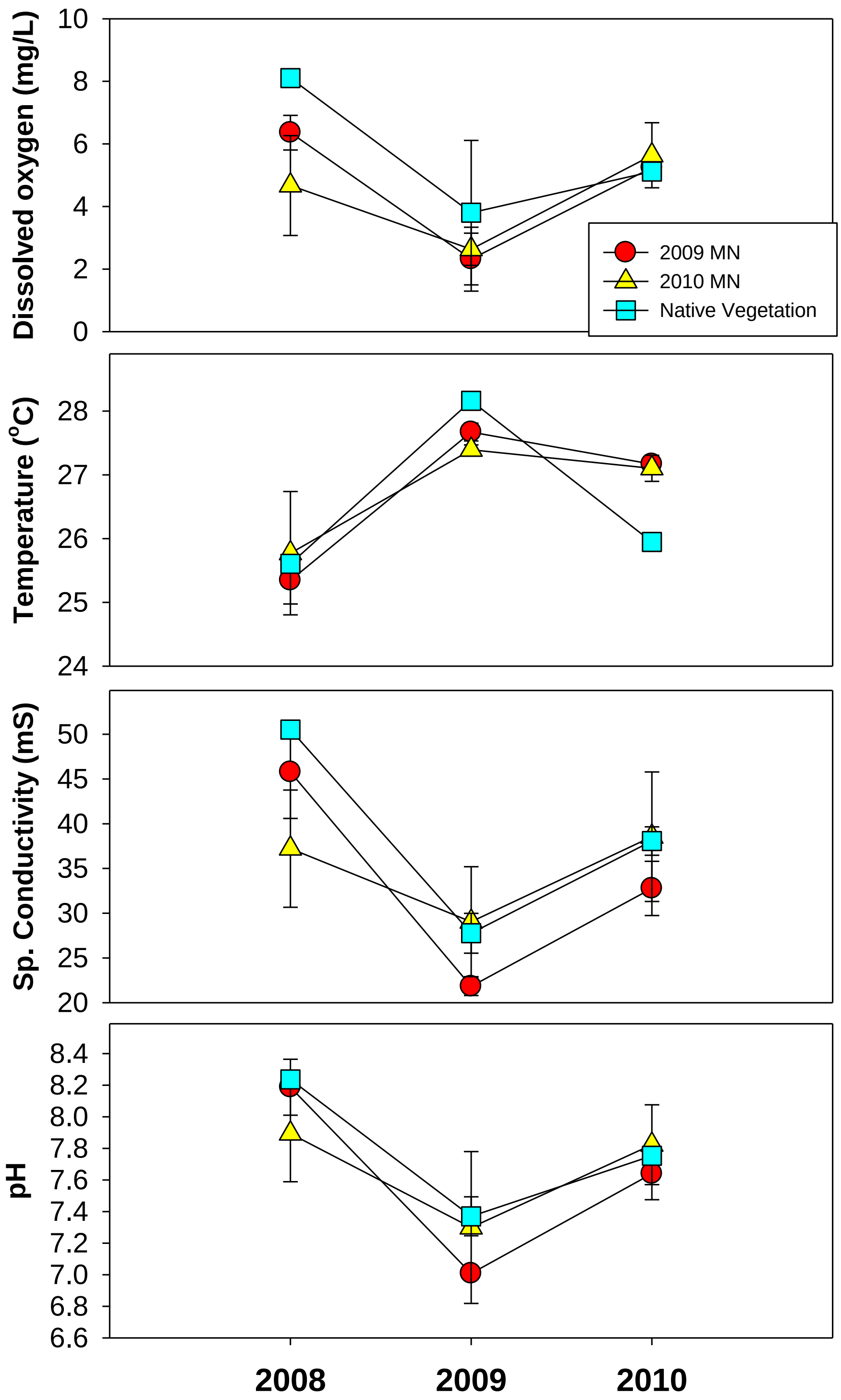

Figure 6. Average dissolved oxygen concentrations (mg/L), water temperature (°C), and specific conductivity values compared across treatments and time. Values were measured at 18:00 and during the same tidal cycle (spring tide, full moon) from each treatment each year in order to minimize influence from the angle of the sun, cloud cover, and tidal inputs.

than 2008, but these differences were not as great as those observed between 2009 and 2008. Thus, the increased groundwater inputs that resulted from mangrove eradication may have been offset by increased tidal inputs as the high tides during the 2010 sampling event (0.82 ± 0.01 m) were higher than 2009 or 2008. Alternatively, the El Niño event that happened in 2010 may have also lowered groundwater inputs that year.

Levels of NH_4^+, PO_4^-, and TP were all below the level of detection (1.0, 0.25, and 0.25 µM, respectively). Total dissolved nitrogen and ΣNO_3^- ranged from 9.3 - 26.0 µM and 6.0 - 28.4 µM, respectively (Fig. 7). The high amounts of ΣNO_3^- relative to TDN suggest that TDN was largely made up of ΣNO_3^-; dissolved organic forms of nitrogen were a minor component. Comparisons could not be made between years as data was not collected in 2008. However, comparisons among the four treatments revealed no significant difference in either TDN or ΣNO_3^- concentrations. Because water samples were collected at or near low tide, mixing between adjacent pools was assumed to be at a minimum. While decomposition of mangroves could have contributed to nutrients measured, the lack of statistical differences among sites suggests that nutrient

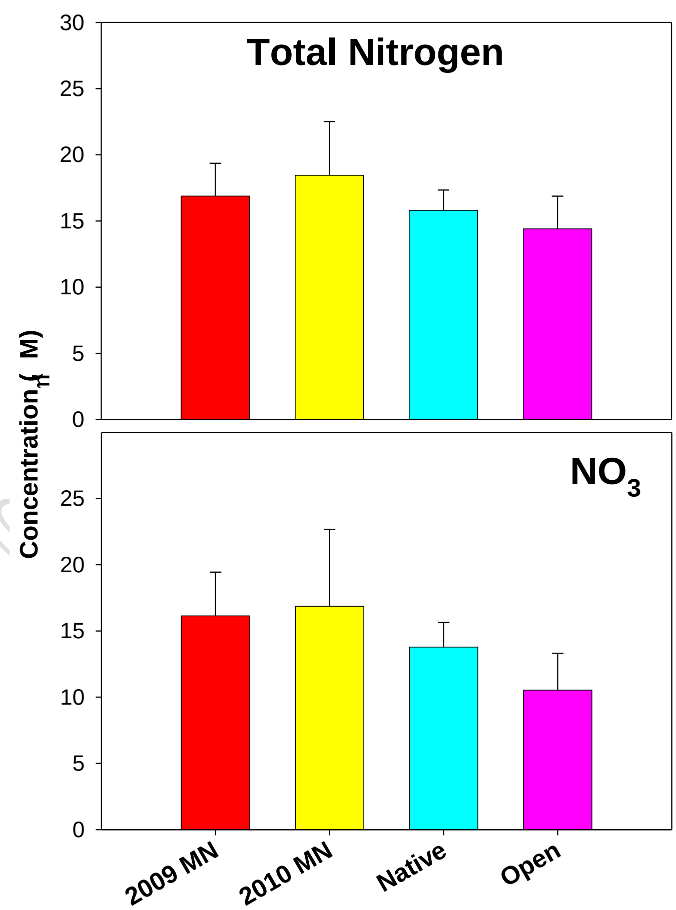

Figure 7. Comparison of total dissolved nitrogen and ΣNO_3^- concentrations among the four treatment pools. There were no statistically significant differences among treatments.

contributions from groundwater inputs elevated in ΣNO_3^- levels from human activities (e.g., cess pools, agricultural activity) were a larger contributor. High levels of bacteria in Wai`Opae tide pools have been attributed to inputs from nearby cesspools, which are also a major source of nitrogen entering this area.

Nekton community composition

Visual surveys. Densities of native fish from visual surveys averaged across all treatments were significantly lower in 2009 (1.6 ± 0.3 no/m^2) than in 2008 (3.3 ± 0.5 no/m^2) or 2010 (5.3 ± 0.7 no/m^2) ($p < 0.001$, $F = 16.03$, df =2). Densities of native fish averaged across years were not significantly different among 2009 MN, 2010 MN, native vegetated, or open pools (Fig. 8). Examination within each treatment between years revealed that densities of native fish initially decreased in 2009, despite the fact that mangrove eradication only occurred in the 2009 MN pools. However, native fish densities in all 2010 treatments were equal to or greater than the initial densities observed in 2008 prior to mangrove eradication (Fig. 8). Changes in fish densities were largely due to densities of the native flagtail, *Kuhlia xenura* (Tables 1, 2, 3). These results suggest that the large scale eradication effort that occurred during 2009 may have had a short-lived effect on the native fish communities, but fish densities in the Wai`Opae quickly recovered one year later in 2010. Alternatively, differences in fish densities may have also been due to inter-annual variation in fish populations.

Densities of exotic fish from visual surveys were also significantly different among years ($p < 0.05$, $F = 4.33$, df =2) and treatments ($p < 0.001$, $F = 7.27$, df =3). However, the significant interaction between site and years revealed that changes in densities were not consistent across sites or years (Fig. 8). Densities of exotic fish were almost always significantly higher in the vegetated pools (2009 MN, 2010 MN, native vegetated) than the open pools in each of the three years. However, densities fluctuated among the three vegetated pools. In 2008, densities were significantly higher in the 2009 MN and native vegetated pools than the 2010 MN pools. In 2009, exotic fish densities were significantly higher in the 2009 MN pools. In 2010, exotic fish densities were significantly higher in the 2010 MN pools. These patterns were largely due to densities of mollies (*Poecilia* sp.) (Tables 1, 2, 3). These results suggest that while exotic fish prefer vegetated habitat over open pools, they do not appear to prefer exotic over native vegetation.

In 2008, 2009, and 2010, native fish had significantly greater densities than exotic fish ($p < 0.001$, $F = 23.31$, df = 1; $p< 0.001$, $F = 14.24$, df=1; $p < 0.001$, $F = 57.44$, df = 1, respectively) (Fig. 8). In 2008, prior to mangrove eradication, comparisons within each treatment revealed that native and exotic fish densities were similar in the 2009 MN and native vegetated pools. However, densities of native fish were significantly higher than exotic fish densities ($p < 0.01$, $F = 5.31$, df = 3) in open (3.56 ± 0.91 and 0.01 ± 0.01 no/m^2, respectively) and 2010 MN pools (2.65 ± 0.38 and 0.64 ± 0.64 no/m^2, respectively). Comparison of total fish densities (sum of native and exotic fish) among treatments revealed that densities were higher in the 2009 MN and native vegetated pools than the 2010 MN and open pools, although this was only significant for the native

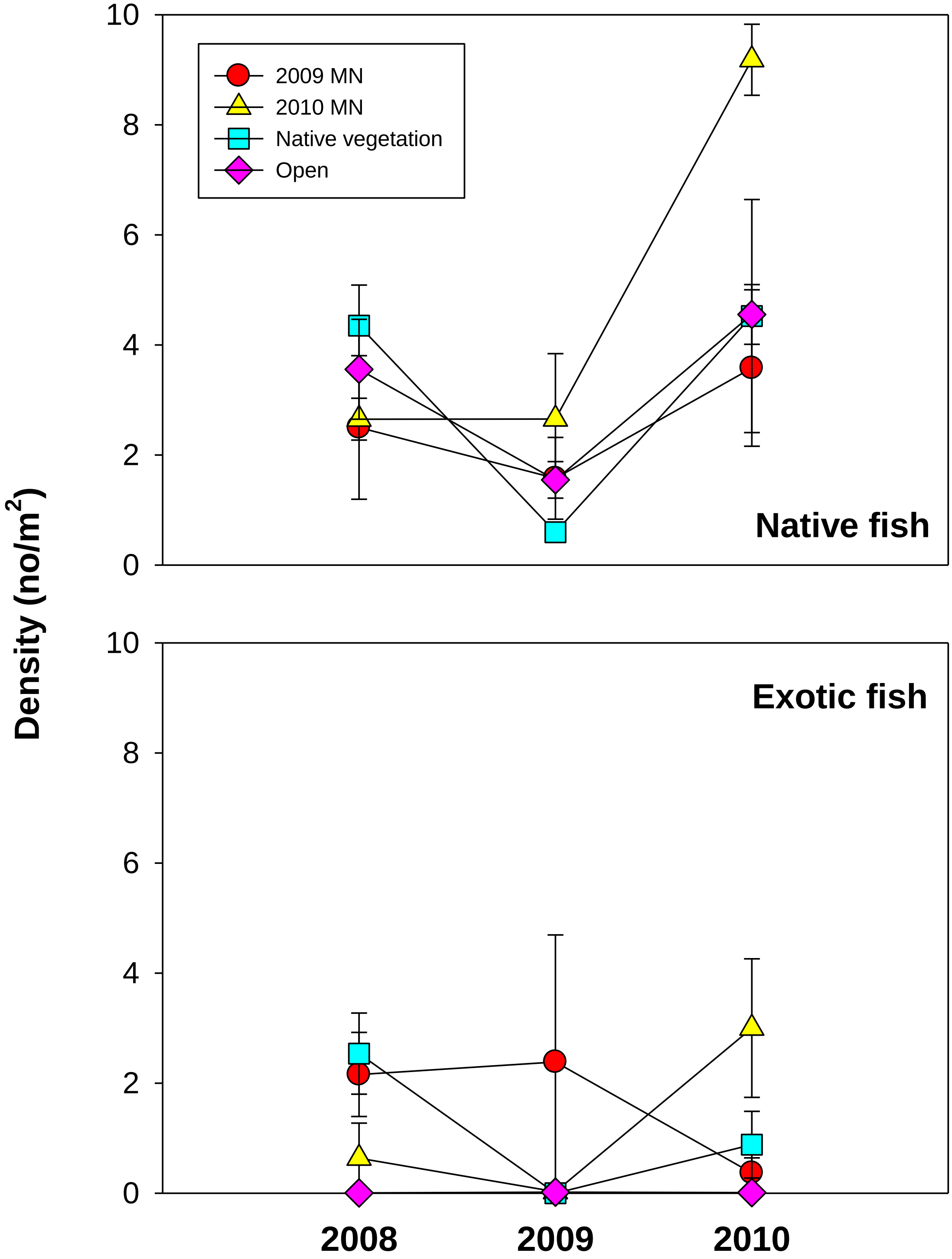

Figure 8. Average (± 1 SE) native and exotic fish densities (no/m^2) from visual surveys in Wai`Opae in 2008, 2009, and 2019.

vegetated pools ($p < 0.01$, $F = 5.13$, $df = 3$) (Table 1). In 2009, comparisons within each treatment revealed that total fish densities did not significantly differ among treatments (Table 2) nor were there any significant interactions, although native fish densities were generally higher than exotic fish densities in the 2010 MN, native vegetated, and open pools; densities of exotic fish were higher in the 2009 MN pools. In 2010, comparisons of total fish densities across treatments revealed that densities in the 2010 MN pools were significantly higher than any of the other three pool treatments ($p < 0.01$, $F = 7.28$, $df =3$) (Table 3). While there were no significant interactions, native fish densities were always greater in all four treatments than exotic fish densities.

In both years, native fish species in the Wai`Opae tide pools were largely comprised of flagtails (*Kuhlia xenura*). In 2008, flagtails represented 19-45% of all fish from visual surveys (Table 1). In the 2009 visual surveys, overall densities of flagtails decreased and only represented 0-18% (Table 2) of all fish, respectively. In 2010, densities of flagtails increased and represented 13-69% of all fish from visual surveys (Table 3). Exotic fish species densities were largely comprised of mollies (*Poecilia* sp.). In 2008, mollies represented 0-46% of all fish from visual surveys (Table 1). Mollies were observed in all treatments except open pools in the visual surveys. In 2009, visual surveys recorded mollies from only one of the 2009 MN pools; mollies were absent from all other pools. In 2009, mollies represented 0-58% of all fish from visual surveys (Table 2). In 2010, mollies were again present in all pools except open pools, but were present in lower numbers than the previous two years, representing 0-17% of all fish from visual surveys. In 2008, the less abundant peacock groupers (*Cephalopholis argus*) were only present in open pools and black tail snappers (*Lutjanus fulvus*) in 2010 MN pools. In 2009, black tail snappers and peacock groupers were only observed in the visual surveys conducted in the open, 2009, and 2010 MN pools. In 2010, black tail snappers were only observed in 2009 MN pools; peacock groupers were only present in 2009 MN, 2010 MN, and open pools.

Fyke nets. Densities of native fish accessing the flooded vegetated areas were significantly lower in 2009 (14.9 ± 4.6 CPUE) than in 2008 (41.6 ± 8.9 CPUE) or 2010 (95.4 ± 18.3 CPUE) ($p< 0.001$, $F = 10.2$, $df = 2$). While densities of native fish were actually higher in 2010 than in 2008, these differences were not significant. Densities of native fish averaged across years did not significantly differ among 2009 MN, 2010 MN, or native vegetated pools (Fig. 9). Comparison of densities among years within each treatment revealed that native fish densities were lower, although not significantly, in 2009 than 2008 or 2010 (Fig. 9), despite the fact that eradication efforts only occurred near the 2009 and 2010 MN pools. These patterns were largely due to decreased densities of flagtails in 2009 compared to 2008 and 2010 (Tables 4, 5, 6). Densities of exotic fish averaged across treatments were higher in 2010 and then decreased from 2009 to 2008, although this was not significant (Fig. 9). Densities of exotic fish averaged across years were significantly different among the three treatments ($p < 0.01$, $F = 9.27$, $df = 2$), with significantly higher densities in the 2009 MN and native vegetated pools compared to the 2010 MN pools (Fig. 9). Comparison of densities between years within each treatment revealed that densities of exotic fish from the 2009 MN were higher in 2009 than 2008 or 2010. Densities of exotic fish from 2010 MN pools showed the opposite pattern, while

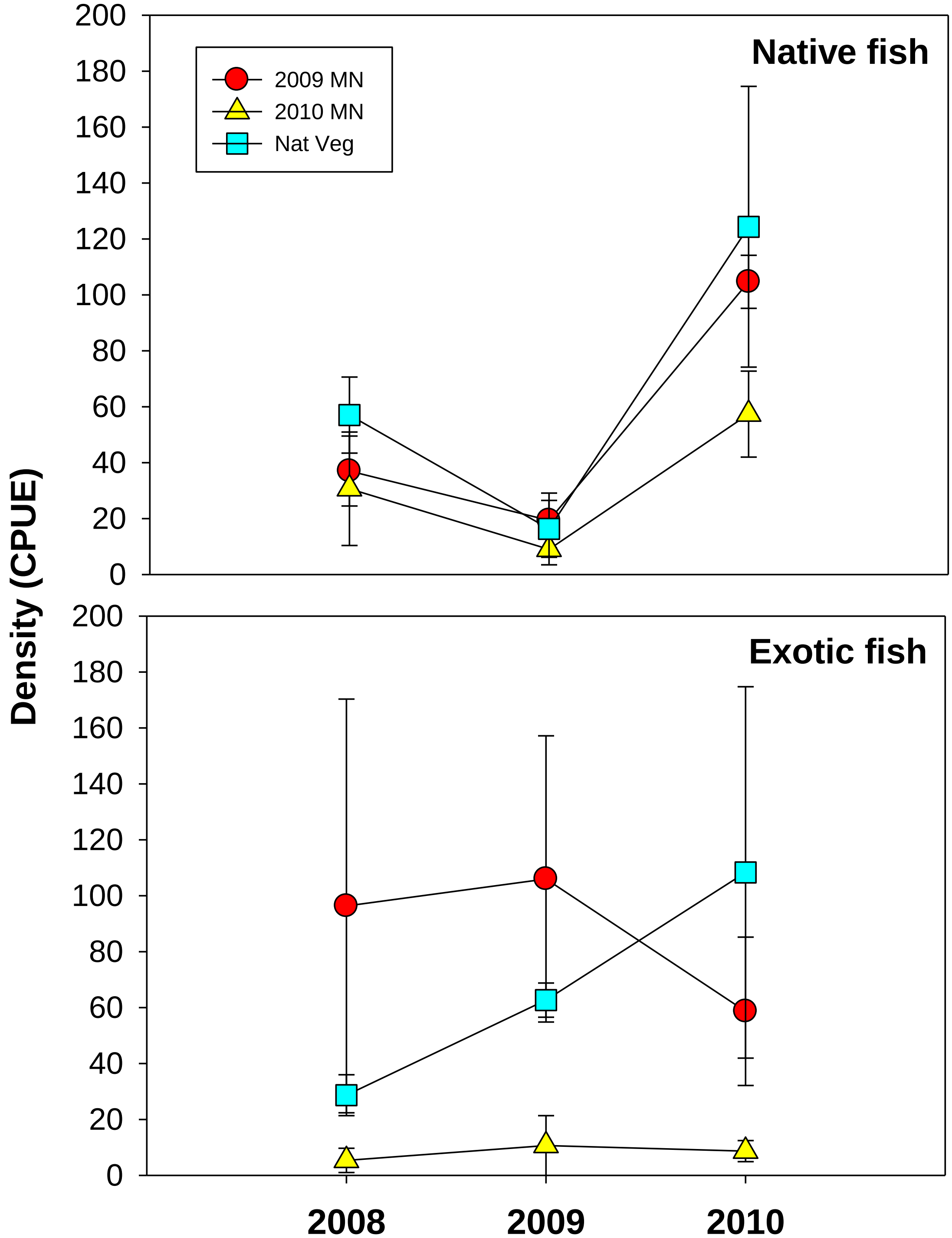

Figure 9. Average (± 1 SE) native and exotic fish densities (CPUE) from fyke net samples in Wai`Opae in 2008, 2009, and 2010.

densities of exotic fish increased each year for native vegetated pools. None of these patterns were significant.

Fish densities from 2008 and 2009 fyke net samples did not significantly differ between fish type (native vs. exotic) or among treatment pools. In 2008, densities of exotic fish were generally higher than native fish in the 2009 MN pools, while native fish densities were higher in the 2010 MN and native vegetated pools (Fig. 9, Table 4). In 2009, densities of exotic fish were generally higher than densities of native fish in all vegetated treatments (Fig. 9, Table 5). In 2010, significantly more native fish were collected in fyke nets than exotic fish ($p < 0.05$, $F = 6.3$, $df = 1$). While there was no significant interaction, native fish had higher densities in all three of the treatments compared to exotic fish (Fig. 9, Table 6).

In all three years, native fish species accessing the flooded vegetated areas in Wai`Opae were largely comprised of flagtails (*Kuhlia xenura*). In 2008, flagtails represented 24–74% of all fish from fyke net samples (Table 4). In the 2009 fyke nets, overall densities of flagtails decreased and they only represented 3-7% (Table 5) of all fish. In the 2010 fyke nets, densities of flagtails were 2x higher than those observed in 2008 and represented 47-72% of all fish collected in fyke net samples (Table 6). Exotic fish species were largely comprised of the abundant mollies (*Poecilia* sp.). In 2008, mollies were present in all treatments and represented 12–76% of all fish collected from fyke net samples (Table 4). In 2009, mollies were again present in all treatments and represented 54-85% of all fish collected from fyke net samples (Table 5). In 2010, mollies were again present in all treatments and represented 5-37% of all fish collected (Table 6). In 2008, peacock groupers were only present in the 2010 MN pools, black tail snappers in 2010 MN pools, and Marquesan mullet (*Valamugil engeli*) in the 2009 MN, 2010 MN, and native vegetated pools. In 2009 and 2010, none of these latter three fish were collected in fyke nets.

The native shrimp, *Palemon debilis*, was also a dominant portion of the nekton collected in the fyke net samples. Densities were higher in 2009 than in 2008 or 2010, however these differences were not significant (Fig. 10). Densities were also similar across sites.

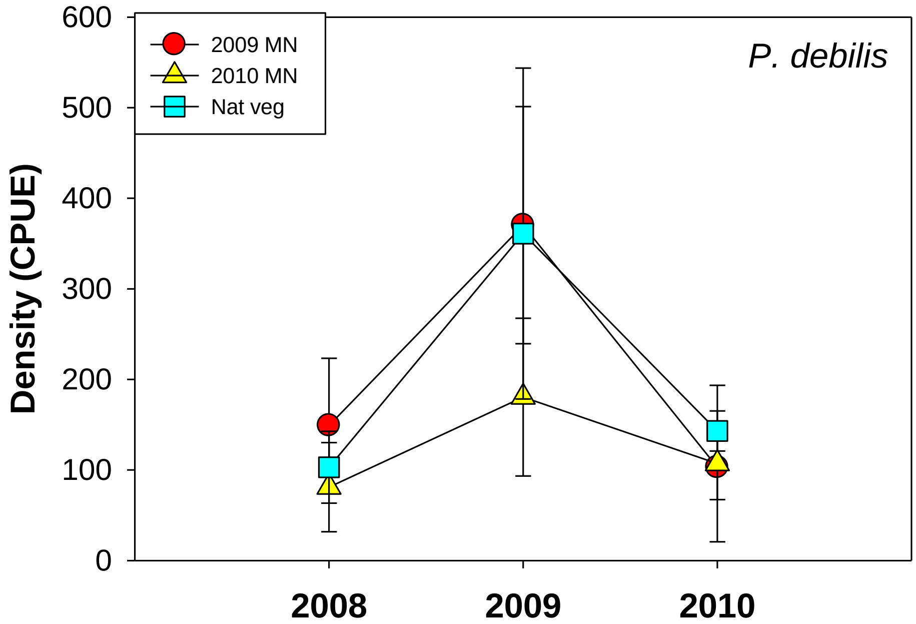

Figure 10. Average (± 1 SE) densities of the shrimp, Palaemon debilis (CPUE) from fyke net samples in Wai`Opae in 2008, 2009, and 2010.

Species Richness

Species richness from visual surveys was not significantly different between years, but was significantly different among treatments ($p < 0.001$, $F = 14.3$, $df = 3$). Species richness was significantly higher in the open pools compared to the 2009 MN, 2010 MN, and native vegetated pools. Species richness from fyke nets was significantly higher in 2008 and 2010 compared to 2009 ($p < 0.001$, $F = 13.85$, $df = 2$), but was not different among treatments. Results from visual surveys suggest that diversity of fish assemblages within tide pools is inversely correlated to vegetation around pools as more species were always observed in the open pools compared to the pools surrounded by vegetation. Results also suggest that one year post-eradication, diversity of fish assemblages within tide pools does not appear to be impacted by the removal of mangroves. Results from fyke net data suggest that the removal of mangroves may have had some effect on the fish community assemblage in 2009, but the fish community appears to have recovered by 2010, with species richness equal to or greater than values reported in 2008.

Table 1. Average (± 1 SE) fish densities (no/m^2) from 2008 visual surveys. Trophic: PL = Planktivore, O = Omnivore, H = Herbivore, P = Predator. Status: N = Hawaiian endemic, IP = Indo-Pacific, E = Exotic, CCG = Circumglobal. See Appendix 1 for species abbreviations.

Trophic	Status	Species	2009 MN	2010 MN	Native Vegetation	Open
PL/O	N	ABAB	0.17 ± 0.10	0.17 ± 0.06	0.05 ± 0.05	0.23 ± 0.07
O	IP	ABSO	0.13 ± 0.08	0.14 ± 0.05	--	0.12 ± 0.02
H	IP	ACTR	0.33 ± 0.20	0.63 ± 0.32	0.44 ± 0.25	0.55 ± 0.08
H	N	blk blenny	--	--	--	0.01 ± 0.01
P	E	CEAR	--	--	--	0.01 ± 0.01
O	IP	ARME	0.01 ± 0.01	--	--	--
O	IP	CHAU	0.01 ± 0.01	--	--	0.01 ± 0.01
O	IP	CHLU	0.04 ± 0.02	0.02 ± 0.02	0.03 ± 0.03	0.03 ± 0.01
H	IP	CHSO	--	0.09 ± 0.09	--	0.05 ± 0.03
P	IP	GOVA	--	0.13 ± 0.08	--	0.11 ± 0.02
P	N	*Kuhlia*	1.66 ± 1.24	0.62 ± 0.34	3.11 ± 0.76	1.01 ± 0.94
H	IP	*Kyphosus*	--	0.02 ± 0.02	--	--
P	E	LUFU	--	0.02 ± 0.02	--	--
H	CCG	*M. cephalus*	0.01 ± 0.01	0.43 ± 0.07	0.34 ± 0.21	0.22 ± 0.11
P	IP	PAMU	--	--	--	0.01 ± 0.00
H	IP	parrot jv	--	--	--	0.06 ± 0.05
O	E	*Poecilia* sp.	2.16 ± 0.77	0.62 ± 0.62	2.54 ± 0.74	--
H	IP	SCPS	--	0.10 ± 0.10	--	0.5 ± 0.14
H	IP	SCRU	--	--	--	0.01 ± 0.01
P	CCG	SPBA	0.01 ± 0.01	0.01 ± 0.01	--	--
P	N	STBA	--	--	--	0.01± 0.01
H	IP	STFA	--	--	--	0.01 ± 0.01
P	N	THDU	0.13 ± 0.08	0.30 ± 0.13	0.39 ± 0.19	0.61 ± 0.12
P	IP	THTR	--	0.01 ± 0.01	--	0.02 ± 0.01
Site Average Density (No/m^2)			**4.66 ± 1.97**	**6.89 ± 1.45**	**3.56 ± 0.91**	**3.29 ± 0.33**

Table 2. Average (± 1 SE) fish densities (no/m^2) from 2009 visual surveys. Trophic: PL = Planktivore, O = Omnivore, H = Herbivore, P = Predator. Status: N = Hawaiian endemic, IP = Indo-Pacific, E = Exotic, CCG = Circumglobal. See Appendix 1 for species abbreviations.

Trophic	Status	Species	2009 MN	2010 MN	Native Vegetation	Open
H	IP	A. nigricans	--	--	--	--
PL/O	N	ABAB	0.08 ± 0.06	0.39 ± 0.19	0.03 ± 0.02	0.09 ± 0.03
O	IP	ABSO	0.17 ± 0.06	0.21 ± 0.01	--	0.14 ± 0.02
PL/O	IP	ABVA	--	0.01 ± 0.01	--	--
H	IP	ACBL	0.11 ± 0.09	--	--	0.01 ± 0.01
H	IP	ACNI	--	0.02 ± 0.02	--	0.01 ± 0.01
H	IP	ACTR	0.31 ± 0.28	0.7 ± 0.18	0.08 ± 0.06	0.54 ± 0.14
P	CCG	SPBA	0.03 ± 0.01	--	--	--
H	N	blk blenny	--	0.01 ± 0.01	--	--
O	IP	CAAM	--	0.02 ± 0.02	--	--
P	E	CEAR	0.01 ± 0.01	0.01 ± 0.01	--	0.01 ± 0.01
O	IP	CHAU	0.01 ± 0.01	--	--	0.02 ± 0.01
O	IP	CHEP	0.01 ± 0.01	--	--	--
O	IP	CHLU	0.06 ± 0.03	0.05 ± 0.04	0.04 ± 0.04	0.01 ± 0.01
PL	N	E. purpurea	--	0.08 ± 0.08	--	--
P	IP	GOVA	--	0.11 ± 0.04	--	0.05 ± 0.02
P	N	*Kuhlia*	--	0.47 ± 0.47	--	0.25 ± 0.17
H	IP	*Kyphosus*	0.01 ± 0.01	0.03 ± 0.03	0.01 ± 0.01	0.02 ± 0.0
P	N	LAPH	--	--	--	--
P	E	LUFU	0.04 ± 0.03	0.02 ± 0.01	--	0.01 ± 0.0
H	CCG	*M. cephalus*	0.02 ± 0.02	0.05 ± 0.05	0.06 ± 0.06	--
P	IP	MUFL	0.16 ± 0.16	--	--	--
H	IP	*N. leuciscus*	--	0.03 ± 0.03	--	--
P	N	PAPO	--	--	--	0.01 ± 0.01
H	IP	parrot jv	0.01 ± 0.01	0.05 ± 0.05	--	0.05 ± 0.03
O	E	*Poecilia*	2.33 ± 2.33	--	--	--
H	IP	SCPS	0.03 ± 0.02	--	--	0.01 ± 0.01
H	IP	STFA	--	--	--	0.01 ± 0.01
P	N	THDU	0.57 ± 0.33	0.42 ± 0.16	0.36 ± 0.12	0.31 ± 0.02
P	IP	THTR	--	0.01 ± 0.01	--	0.01 ± 0.01
Site Average Density (No/m^2)			**3.96 ± 1.57**	**2.68 ± 1.2**	**0.60 ± 0.02**	**1.57 ± 0.33**

Table 3. Average (± 1 SE) fish densities (no/m^2) from 2010 visual surveys. Trophic: PL = Planktivore, O = Omnivore, H = Herbivore, P = Predator. Status: N = Hawaiian endemic, IP = Indo-Pacific, E = Exotic, CCG = Circumglobal. See Appendix 1 for species abbreviations.

Trophic	Status	Species	2009 MN	2010 MN	Native Vegetation	Open
PL/O	N	ABAB	--	0.17 ± 0.07	0.21 ± 0.12	0.32 ± 0.1
O	IP	ABSO	0.09 ± 0.06	0.11 ± 0.03	0.01 ± 0.01	0.28 ± 0.03
PL/O	IP	ABVA	--	--	--	0.05 ± 0.05
H	IP	ACBL	0.03 ± 0.03	0.01 ± 0.01	--	--
H	IP	ACDU	--	--	--	--
H	IP	ACNI	--	--	--	--
H	IP	ACTR	0.18 ± 0.15	0.5 ± 0.32	1.27 ± 0.52	0.91 ± 0.16
O	IP	ARHI	--	--	--	0.01 ± 0
P	IP	ASSE	--	0.03 ± 0.03	--	0.13 ± 0.03
O	IP	CAAM	--	--	--	--
O	N	CAJA	--	--	--	0.01 ± 0
PL/O	E	CEAR	0.02 ± 0.02	0.01 ± 0.01	--	0.01 ± 0
O	IP	CHAU	--	--	--	0.02 ± 0.01
PL/O	IP	CHCH	0.01 ± 0.01	--	--	--
O	IP	CHLU	0.04 ± 0.03	0.01 ± 0.01	0.1 ± 0.08	0.04 ± 0.01
H	IP	CHSO	--	--	--	0.28 ± 0.13
P	IP	*Gymnothorax* sp.	0.01 ± 0.01	0.01 ± 0.01	--	--
P	IP	*Gnatholepis* sp. cf.	0.04 ± 0.02	--	--	--
P	IP	GOVA	--	0.02 ± 0.02	--	0.18 ± 0.09
P	N	*Kuhlia*	2.73 ± 1.42	3.14 ± 1.74	7.08 ± 0.84	0.6 ± 0.58
P	N	LAPH	--	--	--	--
P	E	LUFU	0.01 ± 0.01	--	--	--
H	CCG	*M. cephalus*	0.08 ± 0.08	0.11 ± 0.09	0.01 ± 0.01	0.04 ± 0.02
P	IP	MUFL	0.01 ± 0.01	0.07 ± 0.07	--	--
P	IP	MUVA	--	--	--	--
H	IP	*N. leuciscus*	--	0.01 ± 0.01	--	0.26 ± 0.12
O	N	*E. strasburgi* cf.	0.02 ± 0.02	--	--	--
P	IP	PAIN	--	--	--	--
P	IP	PAMU	--	--	--	--
P	IP	PAPL	--	--	0.03 ± 0.03	--
H	IP	parrot juvi	0.04 ± 0.04	--	--	0.19 ± 0.09
PL/O	IP	PLIM	--	--	--	0.01 ± 0.01
O	E	*Poecilia*	0.34 ± 0.29	0.65 ± 0.65	2.04 ± 0.95	--
H	IP	SCPS	--	0.14 ± 0.14	--	0.22 ± 0.09
P	IP	snowflake eel	--	--	--	--
P	CCG	SPBA	--	0.01 ± 0.01	--	--
P	N	STBA	--	--	--	0.02 ± 0.01
H	IP	STFA	0.01 ± 0.01	--	--	0.07 ± 0.02
P	N	THBA	--	--	--	--
P	N	THDU	0.28 ± 0.28	0.17 ± 0.06	0.46 ± 0.18	0.87 ± 0.15
P	IP	THTR	0.01 ± 0.01	--	0.02 ± 0.02	0.02 ± 0.01
H	E	*V. engeli*	--	0.22 ± 0.18	0.96 ± 0.39	--
Site Average Density (No/m^2)			**3.95 ± 1.67**	**5.41 ± 2.2**	**12.18 ± 1.38**	**4.56 ± 0.54**

Table 4. Average (± 1 SE) fish densities (CPUE) from 2008 fyke nets.

Species	2009 MN	2010 MN	Native Vegetation
Bathygobius cocosensis	2.00 ± 1.00	1.00 ± 1.00	3.00 ± 1.00
Cephalopholus argus	--	0.67 ± 0.58	--
Eleotris sandwicensis	0.33 ± 0.58	2.67 ± 3.06	--
Gymnothorax eurostus	0.67 ± 0.58	--	--
Kuhlia xenura	31.67 ± 22.9	26.67 ± 37.86	52.00 ± 21.52
Lutjanus fulvus	--	0.33 ± 0.58	--
Mugil cephalus	2.33 ± 3.21	0.33 ± 0.58	1.67 ± 1.53
Poecilia sp.	95.00 ± 128.95	4.33 ± 7.51	26.67 ± 10.21
Sphraena barracuda	--	--	0.33 ± 0.58
Valamugil engeli	1.33 ± 2.31	--	2.00 ± 3.46
Site Average Total Density (CPUE)	**133.33 ± 146.62**	**36.0 ± 31.32**	**85.67 ± 17.9**

Table 5. Average (± 1 SE) fish densities (CPUE) from 2009 fyke nets.

Species	2009 MN	2010 MN	Native Vegetation
Abudefduf abdominalis	--	0.33 ± 0.33	--
Acanthurus triostegus	--	1.00 ± 1.00	--
Bathygobius cocosensis	2.33 ± 0.33	4.67 ± 3.71	8.33 ± 4.63
Eleotris sandwicensis	3.33 ± 2.85	0.33 ± 0.33	0.33 ± 0.33
Gymnothorax eurostus	0.33 ± 0.33	0.67 ± 0.33	--
Oxyurichthys lonchotus cf.	1.00 ± 1.00	0.33 ± 0.33	--
Kuhlia xenura	8.00 ± 8.00	0.67 ± 0.67	5.67 ± 5.67
Mugil cephalus	4.33 ± 1.86	1.00 ± 1.00	2.00 ± 1.53
Poecilia sp.	106.00 ± 51.16	10.67 ± 10.67	62.67 ± 6.12
Site Average Density (CPUE)	**125.33 ± 55.84**	**19.67 ± 8.37**	**79.0 ± 13.2**

Table 6. Average (± 1 SE) fish densities (CPUE) from 2010 fyke nets.

Taxa	2009 MN	2010 MN	Native Vegetation
Abudefduf abdominalis	--	0.33 ± 0.33	--
Abudefduf sordidus	0.33 ± 0.33	--	--
Acanthurus triostegus	0.33 ± 0.33	1.33 ± 0.88	1.33 ± 0.67
Bathygobius cocosensis	3.67 ± 0.88	4 ± 3.06	5.33 ± 2.19
Black/Gold Cardinalfish	--	0.33 ± 0.33	--
Eleotris sandwicensis	0.33 ± 0.33	0.67 ± 0.67	1 ± 0.58
Black/White Goby	0.67 ± 0.67	0.33 ± 0.33	--
Gymnothorax eurostus cf.	--	0.33 ± 0.33	--
Kuhlia xenura	99 ± 10.02	48.33 ± 11.61	108.67 ± 43.34
Mugil cephalus	--	1 ± 1	8 ± 7.02
Mulloidichthys flavolineatus	0.33 ± 0.33	--	--
Poecilia sp.	58 ± 26.76	3 ± 2.52	61.67 ± 29.63
Pristiapogon taeniopterus	--	0.33 ± 0.33	--
Sargocentron sp.	--	0.33 ± 0.33	--
Valamugil engeli	0.67 ± 0.67	5.67 ± 4.18	46.67 ± 37.95
Site Average Density (CPUE)	**163.33 ± 32.2**	**66 ± 14.53**	**232.67 ± 99.17**

Fish Size Class Distribution:

Size classes of flagtails collected in fyke nets were significantly different among the different vegetated treatments ($p < 0.001$, $F = 7.3$, $df = 2$). Larger flagtails were collected from native vegetated pools (61.5 ± 1.6 mm) compared to 2009 MN (41.9 ± 1.0 mm) and 2010 MN (48.0 ± 1.2 mm). Significantly larger flagtails were also collected in 2009 (80.7 ± 4.8 mm) than in 2008 (58.9 ± 2.3 mm) or 2010 (47.0 ± 0.7 mm) ($p < 0.001$, $F = 19.27$, $df = 2$). The significant interaction between year and treatment ($p < 0.001$, $F = 7.1$, df – 4) was largely due to significantly larger flagtails being collected in 2009 than 2008 and 2010 in the 2009 MN and native vegetated sites (Fig. 11). These results revealed that fewer young of the year were collected in 2009 compared to 2008 and 2010. Lower levels of dissolved oxygen and warmer water temperatures coupled with the observed benthic algal blooms on the bottoms of the tide pools in 2009 suggest that lower recruitment rates of flagtails in 2009 may have been due to poor habitat quality, which may have been due to increased inputs of groundwater or an effect of mangrove eradication. Regardless, these effects were not apparent in 2010, when young of the year densities were actually greater than densities reported in 2008, before mangrove eradication began. Alternatively, lower recruitment rates in 2009 may have been a natural, inter-annual variation.

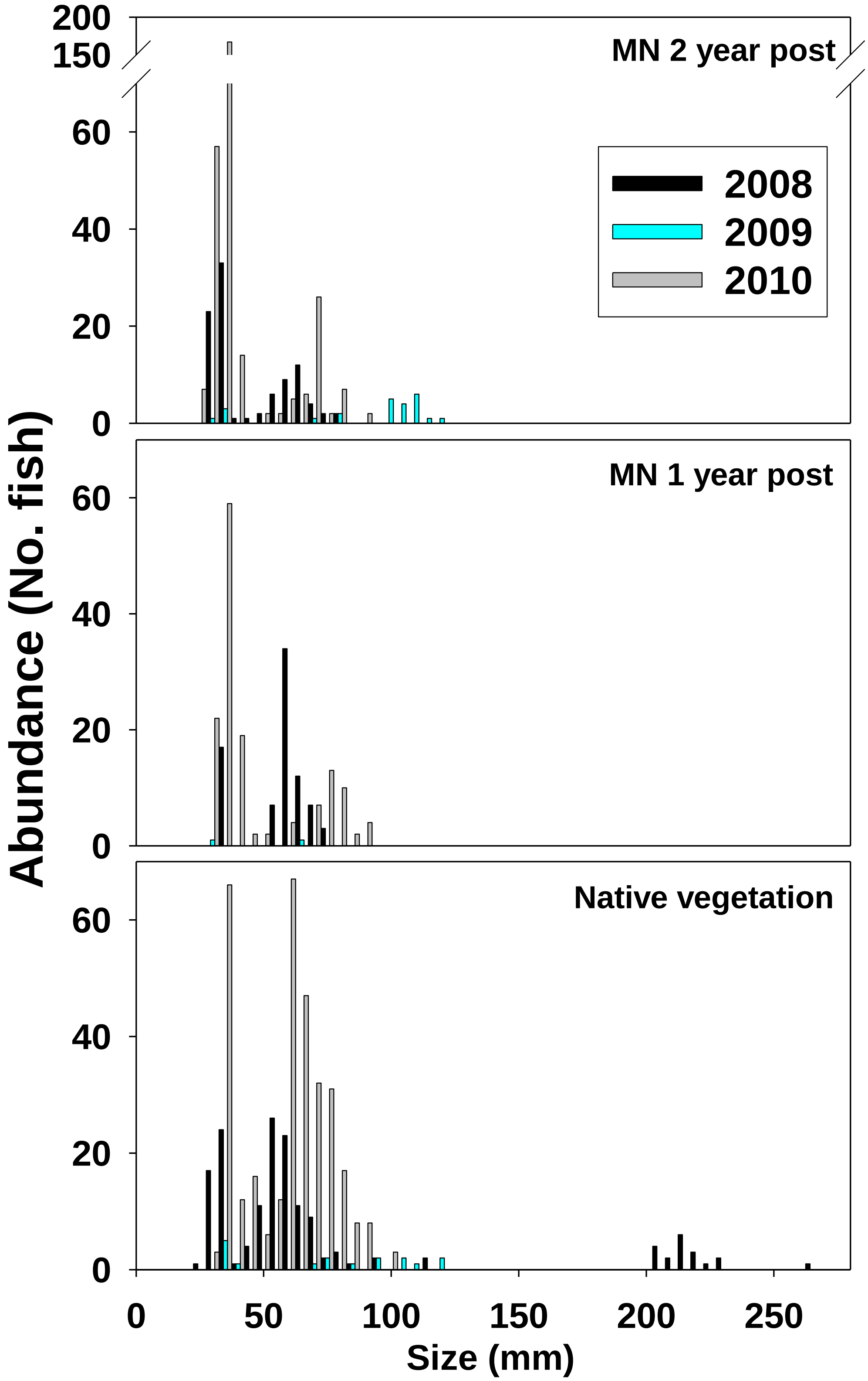

Figure 11. Flagtail size class distributions from 2008, 2009, and 2010 fyke net samples.

Summary

Results reported here are a summary of preliminary data collected immediately before and immediately and one year after mangroves were eradicated from tide pools at the Wai`Opae MLCD. Continued fish monitoring is needed to increase our understanding of how invasive mangroves are impacting nearshore fish communities as well as to verify that any negative impacts from mangrove eradication are short lived.

Densities of native fish from visual surveys and fyke nets appear to have recovered to pre-mangrove eradication levels one year after the majority of mangroves were eradictated from the Wai`Opae tide pools. It is not clear if the reduction in native fish densities was a result of mangrove eradication, increased groundwater inputs, or natural inter-annual variation in fish populations. Densities of exotic fish varied across sites and years with no consistent pattern. This may have been due to the fact that exotic fish were dominated by poeciliids, which can tolerate extreme fluctuations in dissolved oxygen, salinities, and temperatures (MacKenzie and Bruland, unpublished data). Thus, the lower oxygenated, warmer water in 2009, which was thought to have negatively affected native fish densities, did not appear to affect exotic fish densities.

The invasion of ecosystems by one exotic species can often lead the way to future invasions by other exotic species, a phenomenon ecologists have anthropomorphized with the apocalyptic term "invasional meltdown". Whether or not this is occurring in the Wai`Opae tide pools remains unclear. Well established populations of exotic fish have been reported from coastal wetlands throughout Hawaii that have also been invaded by exotic vegetation (e.g., *Paspalum vaginatum*, *Batis maritima*) (MacKenzie et al., unpub). High densities of exotic fish were present in Wai`Opae tide pools regardless of whether vegetation was exotic or native, which suggests that exotic fish do not prefer exotic over native vegetation. However, densities of exotic fish were significantly higher in vegetated pools than open pools. This suggests that if mangroves are colonizing bare, basalt areas at a greater rate than they are invading native vegetated areas, mangroves may be creating more suitable habitat for exotic fish. More studies are needed that determine how mangroves influence growth rates of exotic fish in order to further test this hypothesis.

Densities of native fish were generally higher than densities of exotic fish, with fyke net samples from the 2009 MN pool being the one exception. This suggests that the presence of vegetation surrounding tide pools may somehow benefit native fish, especially young of the year flagtails. Future studies are also needed to examine how native and exotic vegetation affects the growth rates of native fish to determine if coastal vegetation provides valuable habitat for native fish.

Results revealed that if mangrove eradication in Wai`Opae had any negative effects on the fish or shrimp community, those effects were short lived. One year after the majority of mangroves were eradicated, the fish community appeared to be similar to the fish community present immediately before mangrove eradication began. Additional monitoring is needed to verify this. Longer term monitoring is also needed to determine if or how standing woody debris that remains after the herbicide is applied or is dislodged might impact fish and coral assemblages. Finally, future studies are needed using stable isotopes, growth rates, as well as production of fish to determine the ecological value of native versus exotic vegetation in supporting native (and exotic) fish assemblages.

Appendix 1. Wai`Opae visual survey species key for 2008, 2009, and 2010.

Species Code	Latin name	Common name	Hawaiian/ local name
A. nigricans	*Acanthurus nigricans*	Goldrim Surgeonfish	
ABAB	*Abudefduf abdominalis*	Hawaiian Sergeant	Mamo
ABSO	*Abudefduf sordidus*	Blackspot Sergeant	Kupipi
ABVA	*Abudefduf vaigiensis*	Indo-Pacific Sergeant	
ACBL	*Acanthurus blochii*	Ringtail Surgeonfish	Pualu
ACDU	*Acanthurus dussumieri*	Eyestripe Surgeonfish	Palani
ACLE	*Acanthurus leucoparieus*	Whitebar Surgeonfish	Maikoiko
ACNI	*Acanthurus nigrofuscus*	Lavender Surgeonfish	Ma`i`i`i
ACTR	*Acanthurus triostegus*	Convict Tang	Manini
ANCU	*Anampses cuvier*	Pearl Wrasse	`Opule
ARHI	*Arothron hispidus*	Stripebelly Puffer	`O`opu hue
ARME	*Arothron meleagris*	Spotted Puffer	`O`opu hue
ASSE	*Asterropteryx semipunctatus*	Halfspotted Goby	
blk blenny	*Istiblennius zebra*	Zebra Rockskipper	Pao`o
CAAM	*Canthigaster amboinensis*	Ambon Toby	
CAJA	*Canthigaster jactator*	Hawaiian Whitespotted Toby	
CEAR	*Cephalopholis argus*	Peacock Grouper	Roi
CHAU	*Chaetodon auriga*	Threadfin Butterflyfish	Kikakapu
CHCH	*Chanos chanos*	Milkfish	Awa
CHEP	*Chaetodon ephippium*	Saddled Butterflyfish	
CHLU	*Chaetodon lunula*	Raccoon Butterflyfish	Kikakapu
CHSO	*Chlorurus sordidus*	Bullethead Parrotfish	Uhu
COGA	*Coris gaimard*	Yellowtail Coris	Hinalea `aki-lolo
DIHO	*Diodon holocanthus*	Spiny Balloonfish	Kokala
E. purpurea	*Encrasicholina purpurea*	Hawaiian Anchovy	Nehu
E. strasburgi cf.	*Entomacrodus strasburgi*	Strasburg's Blenny	Pao`o
GOVA	*Gomphosus varius*	Bird Wrasse	Hinalea i`iwi
Gymnothorax sp.	*Gymnothorax sp.* (*G. eurostus* cf.)	eel	Puhi
Gnatholepis sp. cf.	*Gnatholepis* sp. cf.	Goby	
Kuhlia	*Kuhlia xenura*	Hawaiian Flagtail	Aholehole
Kyphosus	*Kyphosus* sp.	Sea Chub	Nenue
LAPH	*Labroides phthirophagus*	Hawaiian Cleaner Wrasse	
LUFU	*Lutjanus fulvus*	Blacktail Snapper	Toau
M. cephalus	*Mugil cephalus*	Stripped Mullet	`Ama`ama
MUFL	*Mulloidichthys flavolineatus*	Yellowstripe Goatfish	Weke`a
MUVA	*Mulloidichthys vanicolensis*	Yellowfin Goatfish	Weke `ula
N. leuciscus	*Neomyxus leuciscus*	Sharpnose Mullet	Uouoa

PAIN	*Parupeneus insularis*	Island Goatfish	Munu
PAMU	*Parupeneus multifasciatus*	Manybar Goatfish	Moano
PAPL	*Parupeneus pleurostigma*	Sidespot Goatfish	Malu
PAPO	*Parupeneus porphyreus*	Whitesaddle Goatfish	Kumu
parrot juvis	Unidentifiable juvenile parrotfishes		
Poecilia	*Poecilia* sp. hybrid complex	Mexican Mollies	
PLIM	*Plectroglyphidodon imparipennis*	Bright-eye Damselfish	
SCPS	*Scarus psittacus*	Palenose Parrotfish	Uhu
SCRU	*Scarus rubroviolaceus*	Redlip Parrotfish	Palukaluka
Snowflake eel	*Echidna nebulosa*	Snowflake Moray	Puhi kapa
SPBA	*Sphraena barracuda*	Great Barracuda	Kaku
STBA	*Stethojulius balteata*	Belted Wrasse	`Omaka
STFA	*Stegastes fasciolatus*	Pacific Gregory	
Synodus	*Synodus* sp.	Lizardfish	`Ulae
THBA	*Thalassoma ballieui*	Old Woman Wrasse	Hinalea luahine
THDU	*Thalassoma duperrey*	Saddle Wrasse	Hinalea lauwili
THTR	*Thalassoma trilobatum*	Christmas Wrasse	`Awela
V. engeli	*Valamugil engeli*		Kanda

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