USING BENTHIC FORAMINIFERAL ASSEMBLAGES TO ASSESS WATER QUALITY CHANGES IN MO'OREA, FRENCH POLYNESIA

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Abstract. As coral reefs face global threats of rising temperatures and ocean acidification, monitoring local threats like anthropogenic runoff becomes ever more important to assess baseline health and predict resilience to future global changes. Benthic foraminifera prove to be excellent proxy organisms for assessing favorability of water conditions for coral growth and sustained health. This study revisited sites on the island of Mo'orea from a foraminiferal survey conducted in 1992 with the objective of assessing changes in community composition and water quality index values along the length of Cook's Bay over time. Comparisons were made between 2018 assemblages of living foraminifera, 2018 assemblages of living and dead foraminifera, and 1992 assemblages of living and dead foraminifera. A gradient of increasing water quality index values from the inner bay to the back reef was observed in 1992 assemblages but not in 2018 assemblages. This suggests the nature of the pollution source has changed from a point source to a non-point source, but differences could also be due to variation in sampling locations and techniques. A principle component analysis revealed systematic differences in proportions of several opportunistic genera between 2018 assemblages of living foraminifera and 2018 assemblages of living and dead foraminifera. More information is needed to determine the cause of these differences. This study provides updated baseline data on foraminiferal assemblages in Cook's Bay and is intended to aid future water quality monitoring efforts with benthic foraminifera.

Key words: anthropogenic runoff; Cook's Bay; Pao Pao watershed; urbanization; nutrient enrichment; sedimentation; bioindicators, tropical foraminifera; FORAM Index.

INTRODUCTION

Coral reefs are in decline worldwide (Roberts et al. 2017). They face the global threats of rising temperatures and ocean acidification, regional scale impacts such as disease related mortality events, and more localized threats including tourism and anthropogenic runoff (Hallock et al. 2003). By understanding local disturbances within coral reef ecosystems, we can determine a baseline assessment of reef health to predict how reefs will be affected by future global changes (Sandin et al. 2008). As a result of increased terrestrial urbanization, agriculture, deforestation, and dredging, 25% of coral reefs are affected by terrestrial runoff, which can contain sediments, excess nutrients, and pollutants (Humanes et al. 2017). These factors have been shown to negatively impact reef health and coral growth (Roberts et al. 2017). According to Jones et al. (2015), coral can be affected by excess suspended sediment in the following ways. As broadcast spawners, coral gametes and larvae are disrupted by sediment particles in the water column. Physical contact with sediment irritates the polyps and can inhibit coral feeding and cleaning. Sediment particles can block sunlight from reaching the photosynthetic zooxanthella of adult corals and sinking sediment particles can generate anoxic conditions. Numerous studies have also confirmed that nutrient enrichment negatively affects all life stages of most coral polyps (D'Angelo and Wiedenmann 2014).

Because coral reefs face a wide variety of threats, monitoring and tracking changes in reef health is of utmost importance. However, this can be a challenging task. Nutrient values in water samples do not always accurately

represent nutrient availability in the system, because they do not account for the fact that the community can sequester nutrients (Risk 1999). Furthermore, the fact that coral larval settlement is complex and growth is slow means that the absence of coral from an area does not necessarily mean conditions are not suited for coral growth (Hallock et al. 2003). Due to these limitations in assessing conditions suitable for coral through the organism itself it can be more appropriate to use other taxa as a proxy for water conditions favorable for coral growth (Pisapia et al. 2017). Benthic foraminifera are an ideal proxy organism because many use a symbiotic relationship with algae similar to that which coral has with zooxanthellae (Uthicke et al. 2010). This means that the category of foraminifera, known as larger symbiont-bearing foraminifera, require similar water conditions to coral (Hallock 1988). Hallock (1988) asserts that by looking at foraminiferal community assemblages and comparing proportions of larger symbiont-bearing foraminifera to those of other functional groups, such as heterotrophic and opportunistic, it is possible to determine how favorable water conditions are for coral growth and recovery from disturbances. A numerical value comparing these proportions was created by Hallock et al. (2003) and titled the FORAM Index (abbreviated FI in this paper). The FI is widely accepted and used to compare the water quality of different locations (Schönfeld et al. 2012). Information provided by FI values is helpful for informing management as changes in water quality over time could promote regrowth of coral following destructive events (Pisapia et al. 2017).

Foraminifera are ideal proxy organisms for many reasons (Hallock et al. 2003). For instance, they are highly abundant in the sediment, making it easy to collect samples at nearly any location in a marine environment. High relative abundancy also allows for obtaining large sample sizes without damaging the ecosystem. Additionally, the fact that benthic foraminifera are found in the sediment makes them easy to collect without detriment to the surrounding reef community. Foraminifera are short-lived, meaning that they respond to conditions with less delay than coral. Looking at only living foraminifera, which can be identified with dye techniques (Green 2001), provides a snapshot of present community composition and therefore present water quality conditions. Furthermore, as foraminifera die or reproduce, they leave their tests behind in the sediment—effectively leaving a trace of their presence and preserving information about past water conditions in the layers of the sediment (Hallock et al 2003). Analyzing both living and dead foraminifera in a sample provides information about average community composition--and therefore average water quality--over several decades (Sen Gupta 1999, Fajemila et al. 2015, Ivo Duijnstee, UC Berkeley, pers. comm. 2018). For this reason, foraminifera can be used to determine reef health in the study of long-term changes.

Fajemila et al. (2015) conducted an extensive foraminiferal survey around the island of Mo'orea, French Polynesia documenting community assemblages in samples from 1992. In the Fajemila et al. (2015) study, living and dead foraminifera were used to calculate a FI value for each site visited. Results of this 1992 survey showed higher FI values in the inner parts of Cook's Bay compared to the back reef.

The present study aims to see how foraminiferal assemblages in Cook's Bay have changed over the past 26 years, during which development of the bay and the associated Pao Pao watershed has occurred rapidly (Duane 2006, Vincent Resh, UC Berkeley, pers. comm. 2018). In fact, Duane (2006) used GIS data to show that 80% of road length increase and structures built occurred in the Pao Pao watershed. Furthermore, Duane (2006) also found that road surface area is the land use category most correlated with increased sedimentation on Mo'orea. Using the FI, comparisons of assemblages provide information on water quality changes associated with increased urbanization in this part of Mo'orea. Changes in foraminiferal communities over time are assessed through two points of comparison. 1) Assemblages of both living and dead foraminifera from 2018 samples are compared with living and dead assemblage data from the 1992 survey to determine changes in average assemblages. 2) Living and dead assemblages (referred to as LD from here on) are compared with living only

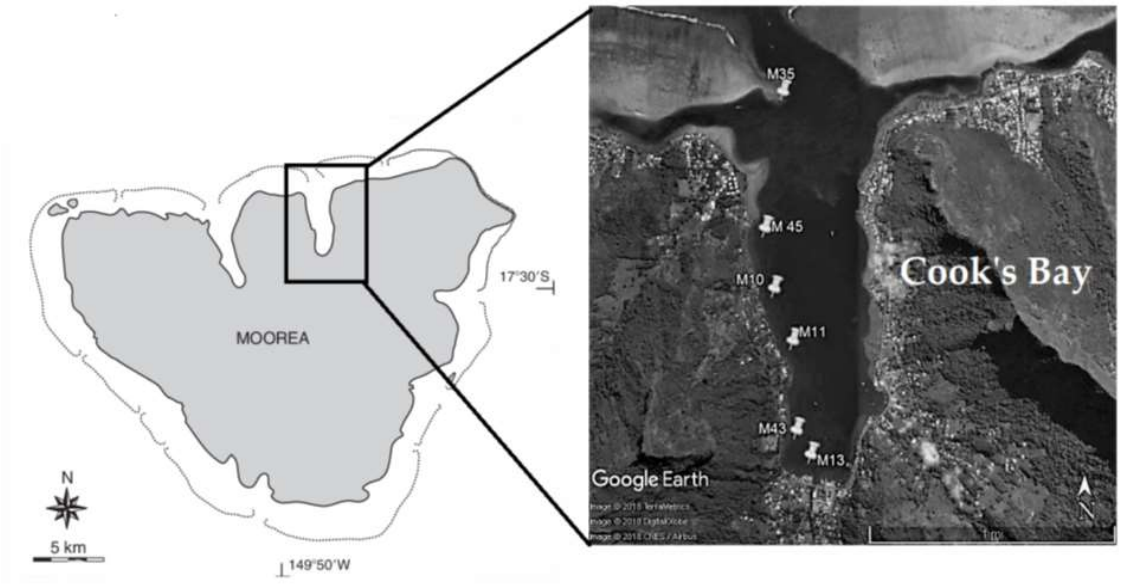

FIG. 1. Map of Moorea showing a close view of Cook's Bay with sampling sites marked (Carroll et al. 2016).

assemblages (referred to as LO from here on) to determine if present assemblages are representative of recent average assemblages. I predicted that 1992 LD, 2018 LO, and 2018 LD assemblages would differ and this would be reflected In differing FI values between the data sets. I also predicted that decreases in FI values will be most drastic at sites near the inner part of the bay that receives the most direct runoff from the adjacent valley.

METHODS

Sample collection

The island of Mo'orea is located in the Society Archipelago of French Polynesia, 25 km Northwest of Tahiti (Fajemila et al. 2015). This island is surrounded by a barrier reef that creates a lagoon encircling the island, and a fringing reef closer to the shore. The two major bays on the island, Cook's Bay and Opunohu Bay are located on the North of the island. For this study, sites in Cook's Bay from Fajemila et al. (2015) were revisited and sampled (Fig. 1). These sites are distributed linearly along the west side of the Bay, showing a 'transect' of positions from the mouth of the watershed at the inner-most part of the bay to the back-reef. These sites covered three different reef environments: inner bay, fringing reef, and back-reef.

A handheld GPS (Garmin Inc. 2013) was used to get as close as possible to the coordinates of each of the previously sampled sites. When coordinates used in Fajemila et al. (2015) did not match the present depth found at their location, replication of correct depth was was prioritized over location. Discrepancies in depth at 1992 coordinate sites, often inexact to several meters, could be due to less advanced GPS technology used in 1992 compared to that available for this study. Natural changes in bay depth and erosion rates may have also contributed to differences in depths at revisited sites (Goineau et al. 2015). At the locations where this occurred, M43, M10, M11 and M45, we found the coordinate location, then kayaked in a straight line toward the shore, until reaching a location where the depth matched the corresponding depth recorded in the Fajemila et al. (2015) survey (Table 1).

At each location, sediment was collected by snorkeling and free diving using a corer constructed of PVC pipe. For LO samples, the top 1 cm was collected using a corer of diameter 5.5 cm following the suggested protocol established in Schönfeld et al. (2012). For LD samples, the top two centimeters were

TABLE 1. Sample sites showing environment type, coordinates, distance from mouth of the watershed, and depth.

Site	Environment Type	Coordinates 1992	Coordinates 2018	Distance from Mouth of Watershed (m)	Depth (m)
M13	Inner Bay	17°30'21.29"S 149°49'19.95"W	17°30'22.00"S 149°49'19.80"W	80	1.5
M43	Inner Bay	17°30'15.68" S 149°49'23.4"W	17°30'15.61"S 149°49'23.73"W	304.5	5.4
M11	Fringing Reef	17°29'55.20"S 149°49'24.30"W	17°29'56.00"S 149°49'26.30"W	901.1	2.2
M10	Fringing Reef	17°29'43.20"S 149°49'28.80"W	17°29'44.60"S 149°49'31.30"W	1295.8	1.1
M45	Fringing Reef	17°29'29.40"S 149°49'31.00"W	17°29'29.30"S 149°49'34.50"W	1727.65	2
M35	Back-Reef	17°28'58.07"S 149°49'26.68"W	17°28'57.90"S 149°49'26.70"W	2651.6	2.6

collected using a corer of diameter 5.5 cm, consistent the methods of Fajemila et al. (2015). The sediment was then transferred to a Ziplock bag and transported to the UC Berkeley Gump Research Station for processing.

Laboratory protocol

In the laboratory, immediately after collection, a 2g/L solution of rose bengal dye and Ethanol was added to LO samples following the approach of Schönfeld et al. (2012). These samples were shaken gently and set aside to allow the dye to stain the living individuals. After one week they were processed with the same protocol as the LD samples, which were not dyed. LD samples were processed immediately after collection.

Each sample was washed with freshwater over a 2 mm sieve and then a 63 μm sieve. The <2mm, >63 μm size fraction was transferred to a round basin with a small amount of freshwater. This basin was moved in a circle for 10 seconds to distribute particle sizes uniformly in a circular pattern. The sediment was then split into sixteen equal sections using a separating tool of exactly the same length as the diameter of the basin to make radial divisions across the middle of the basin in the shape of equal sized pie slices. Sediment from one of the sixteen slices was removed via pipette and placed in two petri dishes with 95% ethanol to be examined under a dissecting microscope. The remainder of the sample was placed in a Falcon tube with 95% ethanol.

Under the dissecting microscope, foraminifera were picked using a paintbrush and secured to micropaleontological slides using water soluble glue. 160 foraminifera were picked from LD samples. This number was chosen based on recommended sample sizes established in Hallock et al. (2003). 100 foraminifera were picked from LO samples. This number was lower than the LD samples due to the time limitations of finding living specimens which are estimated to make up only 10% of foraminiferal tests in a sample (Hallock 2012). Numbers of foraminifera considered in each sample also decreased during species identification due to errors in identifying foraminifera (Ivo Duijnstee, UC Berkeley, pers. comm. 2018).

Next, foraminifera belonging to the genera that make up the larger symbiont-bearing and the opportunistic functional groups were identified and counted following updated classifications consistent with the Fajemila et al. (2015) study (Table 2) (Loelblich and Tappan 1994, Debenay 2012, Horton et al. 2018, Ivo Duijnstee, UC Berkeley, pers. com 2018, Jere Lipps, UC Berkeley, pers. comm. 2018, Martin Langer, University of Bonn, pers. comm. 2018).

TABLE 2. Classification of foraminifera genera into functional groups based on ecological preferences following Fajemila et al. (2015). Groupings of similar genera to facilitate identifications are linked by forward slashes.

Larger Symbiont-Bearing	*Amphistegina, Amphisorus, Assilina, Borelis, Coscinospira, Heterostegina, Monalysidium, Parasorities, Peneroplis, Sorites*
Oppor-tunistic	*Allasoida, Ammonia, Bolivina/Loxostomina Bulimina/Buliminella/Fursenkoina Elongobula, Elphidium, Euglandulina, Hopkinsina, Nonion/Nonionoides, Reussella, Sigmavirgulina/Trifarina*
Hetero-trophic	All other genera

Some genera of the opportunistic functional group were combined into genera groups based on morphological similarities that make it difficult to differentiate between them. Individuals that did not belong to genera groups classified as opportunistic or larger symbiont-bearing were then classified as heterotrophic and counted (Table 2).

Data analysis

Using counts of the genera groups in each sample, I determined the proportional abundances of each functional group. These values were used in the following equation (1) designed by Hallock et al. (2003) to obtain an FI value for each site.

$$FI = (10 \times P_s) + P_o + (2 \times P_h) \quad (1)$$

Where FI = FORAM Index, T= total number of foraminifera counted, Ps = Number of larger symbiont-bearing taxa/T, P_O = Number of the opportunistic taxa/T, P_h = Number of heterotrophic taxa/T.

Figures to visualize changes in functional group percent abundances and FI values between samples and data sets were created in Microsoft Excel. A Principle Component Analysis (PCA) was conducted to provide a more in-depth analysis of differences in genera group composition between 2018 LO and LD samples to compare the community composition of each sample. The PCA was created using the software program PAST (Hammer et al. 2001).

RESULTS

Picked specimens were successfully classified into functional groups based on genera group level identifications (Table 2). The percent abundances of the three functional groups in each sample (Fig 2.) were calculated using functional group counts (Table A1). The proportion of opportunistic taxa in each sample decreases gradually moving from sites near the mouth of the watershed towards sites at the back-reef. The proportion of heterotrophic taxa in each sample slightly increases from sites at the mouth of the watershed to sites at the back-reef. The proportion of larger symbiont-bearing taxa in samples is more variable along this distance and follows no apparent trend. The average percent abundance of heterotrophic taxa is 76% for LO assemblages and 85% for LD assemblages. The average percent abundance of opportunistic taxa is 18% for LO assemblages and 10% for LD assemblages. The average percent abundance of larger symbiont-bearing taxa is 6% for both LO assemblages and LD assemblages.

Percent abundance values of functional groups (Table A1) were used to calculate FI values for LO and LD assemblages at each site (Fig. 3). Each site's distance from the mouth of the watershed was mapped using the Ruler tool in Google Earth Pro (Fig. 3). In the 1992 LD assemblages, a gradient of increasing FI values from the mouth of the watershed to the back-reef is clear. Both the LO and LD assemblages from 2018 do not show this gradient and have FI values that are more homogenous across the bay. FI value means and standard deviations were LO 2018 = 2.30 ± were LO 2018 = 2.30 ±0.40, LD 2018 = 2.35 ± 0.28, LD 1992 = 2.21 ± 0.79.

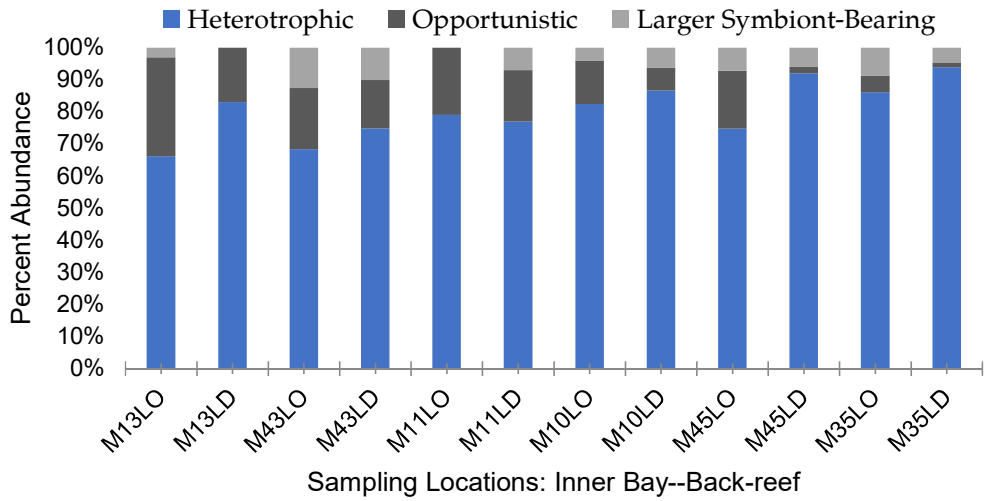

FIG. 2. Percent abundances of opportunistic, heterotrophic and larger symbiont-bearing foraminifera at each sampling location along the length of the bay.

A Principle Component Analysis (PCA) was conducted to analyze the genera composition of each sample. The proportion of each of the genera groups classified as larger symbiont-bearing and opportunistic and the total proportion of heterotrophic taxa were the explanatory variables. Genera groups that were not found in the assemblages (*Elongobula, Euglandulina, Reussella, Sigmavirgulina/Trifarina, Amphisorus, Borelis, Coscinospira, Bolivinella*) were not included in the PCA. PC1 accounts for 76.93% of the variance, and PC2 accounts for 10.29% of the variance, encompassing the majority of variance in the data set. Points from 2018 LO and LD assemblages are clustered spatially, and these groups are linearly separable (Fig. 4). The different genera groups (Table 2) are

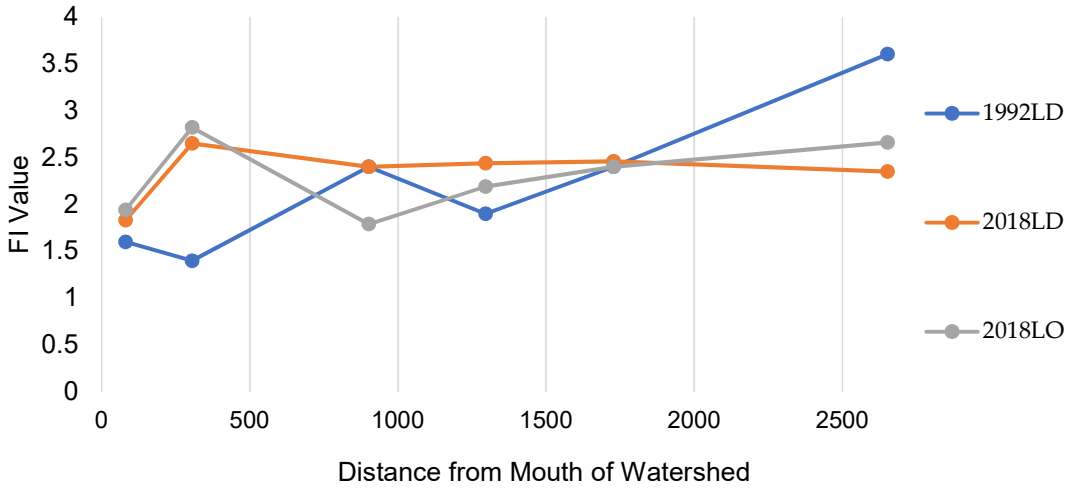

FIG. 3. FI values for 1992 LD, 2018 LO, and 2018 LD assemblages for each site distributed along the length of the bay with sites arranged in increasing distance from the mouth of Pao Pao watershed.

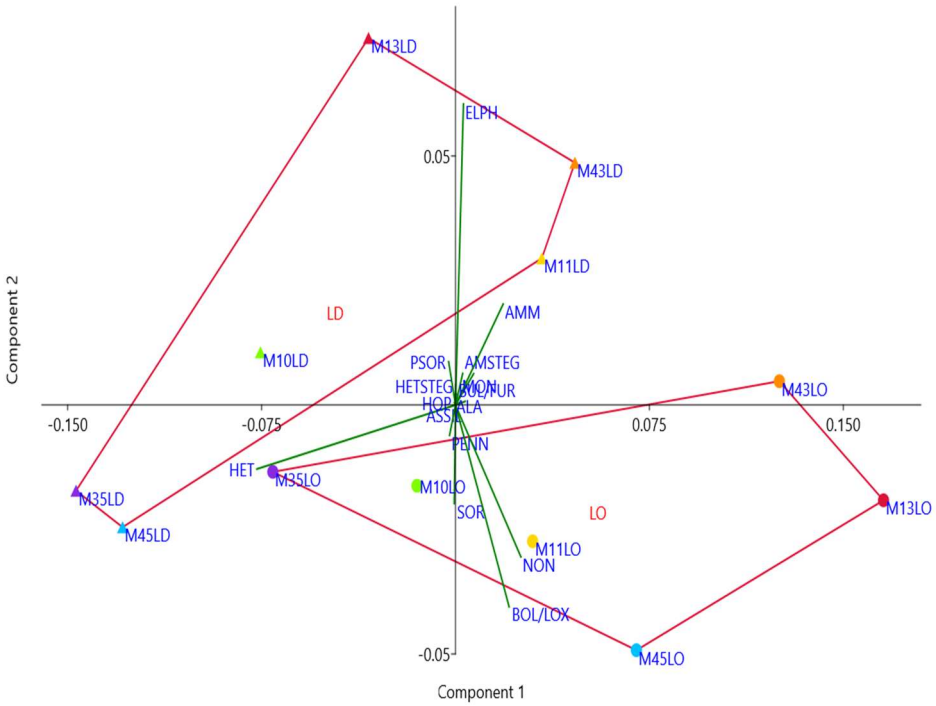

FIG. 4. Principle component analysis illustrating the generic group make up of each sample. LO and LD assemblages are clustered with convex hulls. The vectors show explanatory variables of generic groups. PC1 accounts for 76.93% of the variance and PC2 accounts for 10.29% of the variance. Abbreviations of genera groups are as follows: *Allasoida* (ALA), *Ammonia* (AMM), *Bolivina/Loxostomina* (BOL/LOX), *Bulimina/Buliminella/Fursenkoina* (BUL/FUR), *Elphidium* (ELPH), *Hopkinsina* (HOP) , *Nonion/Nonionoides* (NON). *Amphistegina* (AMSTEG), *Assilina* (ASSIL), *Heterostegina* (HETSTEG), *Monalysidium* (MON), *Parasorities* (PSOR), *Peneroplis* (PENN), *Sorites* (SOR), and all heterotrophic genera (HET).

shown as vectors. LO assemblages are distinctly characterized by higher proportions of *Bolivina/Loxostomina* and *Nonion/Nonionoides*, while LD assemblages are characterized by higher proportions of *Elphidium*. Other genera groupings were less influential to clustering of LD and LO samples (Fig. 4).

Fig 5. is a representation of the same data as in Fig 4. with lines connecting the LD sample to the LO sample for each site shown. The directionality of these lines indicates systematic differences between LO and LD assemblages, changing in genera group composition from the upper left corner to the lower right corner of the plot.

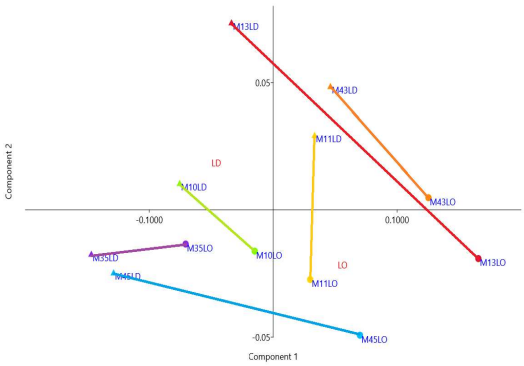

FIG. 5. PCA with lines connecting LO assemblage and LD assemblage for each site. Directionality of the lines suggests systematic differences between LO and LD assemblages.

DISCUSSION

Overall, differences were detected between LD 2018 and LD 1992 FI values. According to the FI guidelines, sites M10 and M43 moved from unsuitable status for coral growth (FI<2) to marginal status for coral growth (2<FI<4) (Hallock et al. 2003). However, these classifications are less meaningful outside of the Caribbean, where the FI was developed (Hallock 2012). Therefore, I considered raw FI values in comparison to each other rather than these classifications. The average FI value of the 1992 LD sample (2.21) is lower than the average FI value of the 2018 LD sample (2.35). This suggests change, but the most notable change is in the relationship of samples along the length of the bay. The gradient of increasing FI values from the mouth of the watershed toward the back reef in 1992 was not observed in 2018 data. This is reflected in the fact that the standard deviation of the 2018 LD FI values (0.28) is much lower than the standard deviation of the 1992 LD FI values (0.79).

Although the original gradient was not observed in the 2018 LD FI values (Fig. 3), slight gradients were observed in opportunistic and heterotrophic percent abundances (Fig. 2). This shows that the FI is not always sensitive enough to detect slight changes in proportions of the functional groups.

Differences between LD 1992 and LD 2018 samples could be related to sediment disturbance such as bioturbation caused by movement of benthic organisms (Murray 1973) or anthropogenic activity like docking boats or fishing. The FI is most appropriate for samples collected at depths between 3 and 15 meters, because shallower samples are likely to be disturbed by routine wave action (Hallock 2012). Sedimentation disturbance likely affected the data from site M43, which was collected from a steep slope where sliding of sediment could have easily occurred, mixing and exposing older sediment. This likely explains the site's unusually high proportion of larger symbiont-bearing foraminifera. Whatever the cause, this increased proportion of larger symbiont-bearing foraminifera undoubtedly contributed to the fact that 2018 LD FI has a higher average than the 1992 LD FI assemblages.

Another explanation of the gradient absence in 2018 could be related to sampling location discrepancies. At most sampling locations, (M43, M11, M10, M45) present depth did not match depth recorded in Fajemila et al. (2015). I chose to prioritize depth over location due to the possibility of GPS error in technology available in 1992. As a result, at each of the sites, my sampling location was closer to land than the point from the original survey. Consequently, my locations could be receiving higher, more consistent levels of anthropogenic runoff than locations farther from the shoreline. This consistent influx of pollutants would result in a decrease of the gradient, as samples would be more homogeneous in the levels of pollution to which they are exposed. Alternatively, the nature of the source of runoff could have changed from a point source of pollution, which is known to result in a distinct gradient in foraminifera (Alve 1995), to a non-point pollution source, which would result in consistent values throughout the bay (Carnahan 2009). This explanation is plausible based on land use changes on Mo'orea since 1992, specifically in the Cook's Bay region.

The first public wastewater treatment facility was created in 2004 (Mahe 2005). Literature suggests that prior to the construction of this facility, wastewater and sewage would run directly into the ocean, at least in the most developed areas of the island (Salvat 2002, Degorges et al. 2010, Fajemila et al. 2015), which would likely include Cook's Bay and Pao Pao watershed (Duane 2006). In the early 2000s, the island made a distinct effort to improve reef health with the implementation of the PGEM (The Plan de Gestion d'Espace Maritime) which created Marine Protected Areas (MPAs) around the island (Hunter 2017). Limiting pollution—including that in the form of anthropogenic runoff—was stated as one of the four main objectives of this plan (Walker and Endemano 2001). However, the MPAs only regulate fishing within their exclusionary zones, so they fail to directly address pollution (Walker and Endemano 2001). Anthropogenic runoff is an important issue to the local island community, as 35% of Mo'oreans interviewed for a study stated that pollution is the biggest threat toward their reef ecosystems (Hunter 2017). Information about runoff regulations on

the island is not readily available, so it is difficult to determine present sources of runoff that are affecting the bay. Based on GIS analysis Duane (2006) found that agriculture decreased in the Pao Pao watershed and moved spatially to increase in the Opunohou watershed between 1982 and 2001. This is a factor that could have also contributed to decreased runoff coming from the mouth of Pao Pao watershed decreasing the effect of this point source pollution in the bay. Ultimately, it is likely that sources of pollution have changed since 1992, which could be the cause of the changes observed in FI values.

Although the FI values of LO assemblages are more variable from site to site than LD assemblages, the mean FI value in the 2018 LO assemblage (2.30) is similar to the mean FI value in the 2018 LD assemblage (2.35). This similarity is interesting due to contention over whether it is more appropriate to use LO or LD assemblages for water quality analysis (Hallock 2012). Arguments for the use of LD samples draw on their ability to provide a more reliable representation of assemblages, because they are less susceptible to high variability over space and time—described as 'pulse patchiness' by Buzas et al. (2002). Alternatively, arguments for the use of LO samples maintain that they provide the most recent representation of water quality conditions (Schönfeld et al. 2012). Also, due to the taphonomical processes of transport or destruction of tests, LD assemblages are not always representative of actual past assemblages (Murray and Alve 1999). The fact that LO and LD samples are similar in average FI values suggests that both assemblages are appropriate for assessing water quality in this system. It also ensures that the results are cross verified: the values are 1) representative of recent water quality and 2) not affected by 'pulse patchiness'. This verifies the 2018 FI values generated in the present study and thus suggests that water quality conditions have not experienced recent drastic changes. However, no change in water quality does not mean no change in community composition—it merely indicates little to no change in functional group proportions.

As seen in Fig. 5, there were systematic differences in generic composition of LO and LD 2018 samples. This indicates recent change in community composition, but this is inconclusive due to several factors. LO samples consisted of the top 1 cm of sediment, whereas LD samples consisted of the top 2 cm of sediment. It is possible that based on depth preferences of different genera, this sampling discrepancy would result in different concentrations of specific genera between LD and LO samples, which would cause differences in counts. Vertical distribution of foraminifera is highly variable across space and time bioturbation both moves foraminifera as well as creates conditions that could allow individuals to live at abnormal depths (Murray 1973, Sen Gupta 1999). Generalizations about depth distribution for foraminifera species rarely extend reliably to genus level or across locations (Sen Gupta 1999), so it is difficult to speculate how this sampling discrepancy may have affected samples. However, Murray (1973) states that the increased validity of LO assemblages from samples thicker than 1 cm does not outweigh the considerable increase in difficulty finding specimens due to the diluted nature of living specimens in thicker samples. A follow up study comparing LO assemblages from the top 1 cm and the top 2 cm of sediment would provide insight on how these depth preferences occur in the sediment of Mo'orea.

Differences in LO and LD assemblages could also be related to changes in populations that occur on a short temporal scale. Populations of certain genera can dominate a community based on seasonal conditions (Murray 1973). This would show an increased proportion of these genera in LO assemblages compared to the time averaged LD assemblages, which scope beyond seasonality and are outside what these changes would affect (Goineau et al. 2015).

Opportunistic species are also more probable to dominate as they are more likely to bloom due to their ability to reproduce rapidly in response to nutrient enrichment (Hallock et al. 2003). Similarly to seasonality, this may also have caused more opportunistic genera to appear in LO samples. Given that LO assemblages had higher proportions of *Bolivina/Loxostomina* and *Nonion/Nonionoides*—all of which are opportunistic genera—this could explain the variance in genera composition between LO and LD assemblages. It is also likely that

interactions between populations of genera are occurring (Murray 1973), which would result in one opportunistic group diminishing while another proliferates, e.g. the higher proportions of *Elphidium* in LD assemblages than in LO assemblages. An opportunistic bloom could also explain why the average proportion of opportunistic genera is larger In the LO samples (18%) than the LD samples (10%) (Fig. 2). Whether this proliferation is due to routine seasonality or blooming of a population needs further research.

Differences in LO and LD assemblages could also be due to the rate at which tests are being deposited into the sediment (Goineau et al. 2015). When foraminifera die or reproduce, they leave their tests in the sediment (Sen Gupta 1999). Therefore, genera with higher reproductive rates and shorter life spans would appear to be more prevalent in LD assemblages than other genera. This could affect the proportions of functional groups in LD assemblages. Specifically, opportunistic foraminifera have been shown to exist in higher proportions in LD assemblages than LO assemblages due to their rapid reproduction rates (Goineau et al. 2015). Reproduction rates can be assessed by examining test size. If rapid reproduction were occurring, most tests found would be smaller than average (Murray 1973). Examining test size in specimens from this study and future studies could shed light on reproductive rates of foraminifera of Mo'orea.

Another factor to consider when comparing LO and LD assemblages is that genera with test morphologies that are more susceptible to dissolution or crushing will be less represented in LD assemblages than LO assemblages. Goineau et al. (2015) found that *Bolivina* were especially susceptible to crushing. This may explain why higher proportions of *Bolivina/Loxostomina* were observed in the LO assemblages compared to the LD assemblages.

In sum, we can only speculate about what has caused the differences in LO and LD assemblages in foraminifera. These findings suggest recent changes in community composition, but it is inconclusive as to whether these changes are due to anthropogenic runoff or natural factors. Further studies on the basic life histories and ecology of benthic foraminifera, especially those found in the sediment of Mo'orea, are necessary. This information would provide insight on factors influencing LO and LD community composition to make conclusions about the past more reliable.

The Fajemila et al. (2015) study aimed to provide baseline data for comparison of future samples. This follow up study found that despite a slight gradient in functional group proportions, water quality as indicated by FI values has changed and no longer reflects a gradient from the mouth of the watershed to the back reef within the bay. This suggests changes related to pollution sources, however, with present data we cannot reasonably conclude the causation behind any specific changes. The genera proportion and composition of the opportunistic functional group was observed to vary between LO and LD assemblages. This could be due to recent changes in water quality, or due to a number of other factors.

This study presents baseline data for continued monitoring of foraminiferal assemblages in Cook's Bay. Past studies have successfully used annual surveys of LO assemblages to track long-term environmental changes (Kelmo and Hallock 2013). I highly recommend this for future study in Cook's Bay. More routine sampling of LO assemblages, with standardized sampling locations and coring techniques would provide more reliable and higher resolution data to better monitor changes within the bay. As climate change and ocean acidification continues to increase, information about changes in local stressors will be of the utmost importance when it comes to preserving and predicting future states of the biodiverse reefs of Mo'orea.

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APPENDIX

TABLE A1. Counts of specimens in each functional group: heterotrophic (HET), opportunistic (OPP), and larger symbiont-bearing (LBF) shown for each sample. Calculated FI values based on these counts are also shown.

	2018 LO				2018 LD				1992 LD
Site	HET	OPP	LBF	FI	HET	OPP	LBF	FI	FI
M13	43	20	2	1.94	93	19	0	1.83	1.6
M43	54	15	10	2.82	104	21	14	2.65	1.4
M11	64	17	0	1.79	110	23	10	2.4	2.4
M10	61	10	3	2.19	123	10	9	2.44	1.9
M45	62	15	6	2.4	137	3	9	2.46	2.4
M35	68	4	7	2.66	122	2	6	2.35	3.6

TABLE A2. Proportions of each genera group in each sample are shown. Total specimen count for each sample is also indicated. Abbreviations are as follows: *Allasoida* (ALA), *Ammonia* (AMM), *Bolivina/Bolivinella/Loxostomina* (BOL/LOX), *Bulimina/Buliminella/Fursenkoina* (BUL/FUR), *Elphidium* (ELPH), *Hopkinsina* (HOP) , *Nonion/Nonionoides* (NON). *Amphistegina* (AMSTEG), *Assilina* (ASSIL), *Heterostegina* (HETSTEG), *Monalysidium* (MON), *Parasorities* (PSOR), *Peneroplis* (PENN), *Sorites* (SOR), *Elongobula (ELO), Euglandulina* (EUG), *Reussella* (REU), *Sigmavirgulina/Trifarina* (SIG/TRI), *Amphisorus* (AMSOR), *Borelis* (BOR), *Coscinospira* (COS), *Bolivinella* (BELLA), and all heterotrophic genera (HET).

	M13LO	M43LO	M11LO	M10LO	M45LO	M35LO	M13LD	M43LD	M11LD	M10LD	M45LD	M35LD
HET	0.662	0.684	0.790	0.824	0.747	0.861	0.830	0.748	0.769	0.866	0.919	0.938
AMSTEG	0	0.063	0	0.014	0.012	0.013	0	0.029	0.014	0	0	0.008
HETSTEG	0	0	0	0	0	0	0	0.007	0.007	0.007	0	0
ASSIL	0	0.013	0	0	0	0	0	0	0	0	0	0
PENN	0	0.025	0	0	0.024	0.038	0	0.007	0.028	0.021	0.027	0.008
AMSOR	0	0	0	0	0	0	0	0	0	0	0	0
BOR	0	0	0	0	0	0	0	0	0	0	0	0
COS	0	0	0	0	0	0	0	0	0	0	0	0
MON	0	0.025	0	0	0	0	0	0.022	0	0	0.007	0
PSOR	0	0	0	0.014	0	0.013	0	0.029	0.014	0.021	0	0.015
SOR	0.031	0	0	0.014	0.036	0.025	0	0.007	0.007	0.014	0.027	0.015
AMM	0.062	0.051	0.037	0.054	0.048	0	0.036	0.065	0.077	0.035	0	0
NON	0.108	0.051	0.049	0.027	0.072	0.013	0.018	0.007	0.028	0	0	0
ELPH	0.046	0.025	0.037	0.014	0	0.025	0.116	0.050	0.042	0.028	0.013	0.015
BOL/LOX	0.077	0.051	0.086	0.041	0.060	0.013	0	0.014	0.014	0	0.007	0
BUL/FUR	0.015	0.013	0	0	0	0	0	0.007	0	0	0	0
SIG/TRI	0	0	0	0	0	0	0	0	0	0	0	0
REU	0	0	0	0	0	0	0	0	0	0	0	0
HOP	0	0	0	0	0	0	0	0	0	0.007	0	0
ELO	0	0	0	0	0	0	0	0	0	0	0	0
BELLA	0	0	0	0	0	0	0	0	0	0	0	0
EUG	0	0	0	0	0	0	0	0	0	0	0	0
ALA	0	0	0	0	0	0	0	0.007	0	0	0	0
Total Specimens	65	79	81	74	83	79	112	139	143	142	149	130