North Hilo (Hilo Palikū) Coastal Fishery Assemblages: Global Climate Change Impacts Modeled Along a Highly Constrained Hydrological and Precipitation Gradient

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By
James Jacob Akau

Thesis Committee:

Patrick Hart, Chairperson
Richard MacKenzie
Christian Giardina
Celia Bardwell-Jones

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Abstract

For centuries coastal fish communities have sustained human populations throughout the Hawaiian archipelago. Fish populations not only provide sustenance, but have also been and continue to be a source of inspiration for Hawaiian culture. The advent of global climate change and corollary increases in sea level temperature has compromised coral reef health and disrupted the ecological stability of nearshore systems. Global climate change and associated elevated atmospheric carbon dioxide has affected watershed function, especially rainfall magnitude and frequency. Since 1958 average rates of precipitation have decreased from 5 to 40 %. Coastal and estuarine environments are especially sensitive to runoff and stream discharge, which affect food availability for fishes, juvenile recruitment, benthic substrate, salinity levels, and fish community composition.   To examine the effects of global climate change on coastal ecosystems, I've compared salinity levels, benthic substrate, fish community assemblages, species relative abundances, and food web structure across a precipitation gradient from Pepeʻekeo (6000mm rain/yr) to Laupahoehoe (300mm rain/yr) along the Hilo Paliku (North Hilo), Hawaiʻi coastline.

To investigate the effects of variable fresh water inputs to near shore fishery assemblages, I compared recorded fish biodiversity, species relative abundances, and benthic substrate at Pepeʻekeo and Laupāhoehoe using fish visual surveys on SCUBA and free diving.   While free diving, I also used a Hobo data conductivity logger to measure salinity levels spatially and over time Pepe ʻekeo, Kolekole, and Pepeʻekeo. Corals, which provide habitat and food for fishes, are sensitive to extreme fluctuations in salinity. Most corals have a salinity tolerance range between 28.7. Coral percent cover and fish biodiversity was significantly greater at Laupāhoehoe when compared to Pepeʻekeo (P < 0.001). This demonstrates that more coral provide a more variable habitat for greater fish diversity, and that changes in rainfall and coastal salinity levels are important to maintain this diversity.

With an understanding of how decreased stream flow will impact near-shore fish communities, we can begin to develop adaptive management strategies to maintain stream flow during periods of drought or return water to streams from diversions. This information can help to develop and support effective near-shore fisheries management at the community and legislation level.

Keywords: precipitation gradient, fishery assemblage, food web dynamics

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Introduction

Global climate change and associated elevated atmospheric carbon dioxide has affected watershed function, especially rainfall magnitude and frequency, trans-evaporation, vegetative water use, humidity, and soil moisture (Strauch et al. 2015). Carbon dioxide levels on Mauna Loa, Hawaiʻi (19.5N, 155.6W, 3400m) had risen from less than 320 ppm in 1958 to 400 ppm in 2013 (Zhao and Zeng 2014). Average rates of precipitation have decreased from 5 to 40 % over this same time frame. The decrease in precipitation rates varies across each island, with an increasing contrast between windward and leeward sides (Strauch et al. 2016).

The combined effects of elevated carbon dioxide (Zhao and Zeng 2014), warmer temperatures, and diminishing rainfall patterns have changed delivery times and reduced freshwater discharge into nearshore environments (Strauch et al. 2016). Warmer atmosphere tend to hold more moisture, which strengthen storm events that feed coastal systems with runoff and stream discharge (Strauch et al. 2016). Contrary to typical climate change model predictions, some areas of Hawaiʻi have shown weakening singular storm events. In both instances, regardless of strengthening or weakening storm events, there has been a consistent observation of reduced total rainfall across Hawaiʻi (Bassiouni and Oki 2012, Chen and Chu 2014).

Coastal and estuarine environments are especially influenced by runoff and stream discharge (Strydom et al., 2003, Hoover and Mackenzie 2007, Atwood et al. 2012). In some cases, terrestrial subsidies have increased secondary production of nearshore ecosystems (Pace et al. 2007). However, other studies suggest that terrestrial subsidies are a small part of fishes that feed in the water column (Atwood et al. 2012). Although nutrients from runoff may or may not significantly affect secondary production (Pace et al. 2007), suspended sediments limit light availability and eventually settle on coral to restrict growth (Harvell et al. 1999, Ogston and Field 2010).

The loss of light limit coral photosynthesis, which when combined with shifts in temperature, salinity (Coles and Riegl 2013), and sedimentation can result in mortality (Smith et al. 2001, Anderson et al. 2002, Guan 2015). The loss of coral due to algal competition, physical as well as chemical stressors can result in a loss of habitat complexity and fish biodiversity (Friedlander et al. 2006). Excess nutrient and sedimentation correspond to algal blooms that produce harmful toxins linked to high incidences of invertebrate and fish mortality (Smith et al. 2001, Anderson et al. 2002). Environmental changes from eutrophication may result in a phase shift from coral to algal dominated substrate (Barnosky et al. 2012). The effects of which is a shift in fish community assemblages that along the nearshore (Friedlander et al. 2006).

Across the windward side of Hawaiʻi Island, overall losses to the water budget, increased erosion, amplified organic matter, nutrient and sediment loads have stressed coral and associated fish communities (Atwood et al. 2012, Wiegner et al. 2009, Strauch et al. 2016.). Both ends of the Hilo Palikū precipitation gradient represent the extreme climatic shifts that are expected to impact nearshore ecosystems. Loss of water to the overall water budgets affect salinity, temperature (Coles and Riegl 2013), nutrient and sediment loads into the nearshore (Ringuet & Mackenzie 2005, Wiegner et al. 2009). Changes in fresh water inputs will affect salinity and

temperature levels, both of which are important factors in larval fish community assemblages (Strydom et al., 2003, Ramos 2006).

Importance of Fisheries

Coastal reef ecosystems support highly productive and diversified communities of corals, invertebrates, and fishes (Hoover 1999) rivaling the biodiversity found in rain forests (Bowen et al. 2013, Kane et al. 2014). Although corals occupy less than 0.1 % of the global sea floor (Bowen et al. 2013), gross primary productivity can be 50 to 100 times greater (Dawes 1998). The heightened productivity of reef ecosystems and associated fish communities encompass nearly a third of known marine species (Bowen et al. 2013). In the Hawaiian archipelago, rates of fish endemism to 30 m in depth are 12.8 % and 21% in the Main Hawaiian Islands (MHI) and North Western Hawaiian Islands (NWHI) respectively (Kane et al. 2014). This rate is attributed to the high level of geographic isolation (Eldredge and Evenhuis 2003). In the mesophotic zone, between 30 to 150 m, of the NWHI, rates of endemism swell to 46%, twice that of any other tropical region (Kane et al. 2014).

The 23 Hawaiian hermatypic (reef-building) corals contribute to habitat complexity to affect fish biomass, abundances, species richness, and diversity (Friedlander et al. 2006, Wild et al. 2011). Reefs as autogenic engineers provide structural relief from predation and a refuge from the elements (Crooks 2012). For example, at Maniniholo, Kauaʻi *manini* (*Acanthurus triostegus*) migrate from their deeper burrows into shallow protected bays during the winter season to escape the unrelenting surf (Andrade 2008). In addition to shelter, over a hundred fish species feed on live coral (Jayewardene et al. 2009) as well as invertebrates that co-habit reef complexes (Friedlander & Parrish 1998).

Corallivores such as butterfly fish (*Chaetodon spp.*), barred filefish (*Cantherhines dumerilii*), and parrot fish function as ecosystem engineers who digest corals to produce sand (Jaywardene et al. 2009). The calcareous sand's biogenic permeability acts as a bio catalytic sieve, which recycles organic materials and hosts heterotrophic microbes (Wild et al. 2011). Although widespread and in some instances concentrated, the effects of corallivory on coral cover may be in some reef ecosystems nominal (Jaywardene et al. 2009).

Coastal fish communities have and continue to provide sustenance throughout the Pacific as an integral component of diet and economic livelihood (Dollar 1982, Friedlander and Parrish 1998). Throughout the Hawaiian archipelago, fish populations, supplemented with domestic pigs, chickens, and dogs, served as a primary source of protein prior to the introduction of large ungulates (i.e. cattle, goats, sheep, and deer) (Titcomb 1982, Andrade 2008, Jokiel et al. 2011). As a result, established cultural norms and designated harvest periods evolved to not only ensure fish population stability, but facilitated conditions to promote target species densities for optimal productivity (Kamakau1964, Kahaʻulelio 2006).

Hawaiian culture employs pedagogic methodologies involving oral traditions of allegorical song, chant, and dance supported by governmental decree to manage human roles, define stewardship responsibility, and harvest etiquette (Jokiel et al. 2011). Regulations set into law defines a *Kapu* (prohibition) system involving seasonal closures of fisheries, size limitations, gear constraints, no-take zones, and regulated entry (Jokiel et al. 2011). With the abolishment of

the *Kapu* system, contemporary policies permit lengthier harvest periods often with no bag limits; scant gear restrictions, and uninhibited access. Consequently, the abandonment of customary tenets and surging commercial activities have contributed to the demise of Hawaiʻi's fisheries (Jackson et al. 2001).

Hawaiʻi's economic shift over the past century from an artisanal based economy to industrial capitalism has had detrimental impacts on the local fisheries. As a mercantile enterprise, Hawaiʻi's reef ecosystems directly generate 360 million dollars a year (eg. fisheries, invertebrate harvest, aquarium trade, etc.) and indirectly generate an additional 800 million dollars via tourism (Tissot and Hallacher 2003, State of Hawaiʻi and NOAA, 2010). Declines are by and large due to the rise of densely populated areas, urban development, pollution, and commercial fishing pressure (Beets 2010, Birkeland and Dierking 2007, Friedlander et al. 2007, Arita et al. 2013). The commercialization of fisheries and increased catch per unit effort has caused a number of fish populations to crash (Friedlander et al. 2001).

Threats to coastal reefs

Fish introductions

To support Hawaiʻi's lucrative fishing market, the State of Hawaiʻi introduced fish species as a means to bolster dwindling stocks. Many of these species, introduced accidentally and intentionally, aggressively compete with native fish for food resources, habitat, or as predators (Birkeland and Dierking 2007, Keller et al. 2008). The loss of habitat especially impact endemic fishes with limited local distribution and constrained geographic ranges (Kane et al. 2014). For example, the State of Hawaiʻi introduced the peacock grouper (*Cephalopholis argus*; common name: roi) to Hawaiʻi from Moʻorea, French Polynesia in 1956 to improve fish catch and supplement local stocks (Dierking 2007, Dierking and Campora 2009). Roi, a once viable dish, is now linked to numerous cases of ciguatera poisoning and has since experienced dramatic declines in human consumption (Birkeland and Dierking 2007). In the absence of both natural and human predation, unchecked roi populations disproportionately prey upon native fish species (Birkeland and Dierking 2007, Courtenay et al. 2009). The introductions of species such as roi represent just one of many anthropogenic threats to Hawaii's coastal reef systems.

Deforestation

In Hawaiʻi, the conversion of forested watersheds into agricultural lands, industrial, and urbanized spaces (Ringuet & Mackenzie 2005) has diminished watershed resilience, water conservation, and soil retention (Yiqing Li and Mathews 2010). Subjected to decades of ungulate grazing, a third of the Island of Kahoʻolawe is now barren and has lost one to three meters of topsoil, much of it washed or blown into adjacent coastal waters (Warren and Aschmann 1993). The loss of vegetative cover reduces fog drip and rainfall, which are important sources of water for hydrogeological cycles (Giambelluca et al. 2008). Denuded landscapes also magnify terrestrial subsidies (i.e. runoff, pollution, organic matter, nitrogen, phosphorus, etc.) in river plumes (Vitousek et al. 1997). The increased sediments from denuded areas can be transported in into coastal waters and smother corals (Smith et al. 2001).

Global Climate Change

Since the industrial revolution and consequent unabated consumption of fossil fuels, the advent of global climate change and corollary increases in tropical cyclone frequency, duration, and strength have disrupted marine ecosystem processes and cycles (Jokiel and Brown 2004, Fletcher 2010, Keener et al. 2012, Strauch et al. 2015). Although storm intensity has increased, over the past century, Hawaiʻi Island has experienced a decline in annual precipitation with a 20-70% fall in stream base flow (Giambelluca et al. 1991, Chu 1995, Oki 2004, Chu and Chen 2005, Bassiouni and Oki 2012). Such decreases occur in concert and are in part attributed to diminishing prevailing northeasterly trade winds, primarily responsible for orographic precipitation (Collins et al. 2010, Garza et al. 2012, and Tokinaga et al. 2012). These decreases limit aquifer recharge, nearshore ground water discharge and fresh water dynamics (Chu et al. 2010).

Stream inputs are an important source of nutrients and organic matter that support high levels of nearshore fish productivity. Various models predict that Hawaii will be warmer and drier in the future, with fewer but more intense storm events. This will result in decreased stream flow and inputs to nearshore water that will reduce nutrient and organic matter loads. In Kaneohe Bay, storm delivered nutrients are rapidly utilized by phytoplankton and zooplankton (Ringuet and MacKenzie 2005). These plankton organisms, as well as terrestrial organic matter inputs, are then fed upon by many culturally and ecologically important species of fish (Atwood et al. 2012, Michaud and Wiegner 2011). A decrease in rainfall correspond to diminishing stream flow (Strauch et al 2015) and reduced organic matter and nutrient loads into Hawaiian nearshore waters (MacKenzie et al. 2016).

On Hawaiʻi Island's windward coast, intensified rainfall events and extended drought periods are projected to escalate episodic flash flooding (Chu et al. 2010, Elison Timm et al. 2011)), significantly increasing sediment, nutrient, and bacterial loads to nearshore waters (Strauch et al 2013, Strauch et al. 2014, MacKenzie et al 2016). This could have severe consequences on the nearshore environments as Hawaiʻi's coastal environments are particularly sensitive to rain and can rapidly respond to storms that amplify delivery of terrestrial inputs (Ringuet & Mackenzie 2005, Wiegner et al. 2009). Excess terrestrial subsidies entering the nearshore environment can suffocate corals, fuel eutrophication, and alter salinity and temperature (Harvell et al. 1999, Ogston and Field 2010).

The loss of forest increases water flow, erosion, and particulate suspended materials into stream systems that feed nearshore environments. Streams from oceanic island watershed systems rapidly respond to storms (Ringuet & Mackenzie 2005), and account for the large part of coastal fresh water inputs (Hoover and Mackenzie 2007). Along Oahu's windward coast, Hoover and Mackenzie (2007) found that storms account for the majority of river introductions into the nearshore. Increases in stream effluence and surface runoff increase nutrients and detritus levels that provide food for macrozoobenthos and invertebrates (Gocke et al. 2013). Native and endemic fish such as the ʻāholehole (*Khulia spp.*) benefit directly as the terrestrial introductions support their dietary needs (Tester and Trefz, 1956, Randall 1996). Erosion and storm runoff pollute and toxify coastal waters, bury corals, and intensify algal bloom frequency, duration, and proliferation (Harvell et al. 1999, Ogston and Field 2010). Invasive species and denuded

landscapes compound global climate change affects to create increasingly inhospitable coastal ecosystems.

Within hours after a storm, rivers reach maximum flow supplying coastal waters with allochthonous matter. In a matter of a few hours, water can move from the headwaters down to the sea (Ringuet & Mackenzie 2005, Wiegner et al. 2009). The river discharge and terrestrial runoff correspond to increased phytoplankton productivity and biomass (Harvell et al. 1999). Phytoplankton organisms use CO_2 and nutrients to produce particulate organic and inorganic matter. Phytoplankton productivity and primary production rates are determined by temperature as well as light and nutrient availability (Doney 2010). Although natural nutrient inputs support phytoplankton productivity, nutrient loads have increased 50-200 % from their preindustrial levels and have resulted in catastrophic eutrophication events (Harvell et al. 1999, D'elia et al. 2003, Fisher et al. 2006, Ogston and Field 2010). From the 1950s to 2001, French Siene estuary conditions experienced an unprecedented increase in nitrogen levels as a response to human population growth (Billen et al. 2001). In this case, typically nutrient limited algae receive excess nutrient loads in pollutants and runoff from denuded landscapes converted into agricultural, residential, and urban spaces.

In abundance, algal respiration and mortality create hypoxic conditions, and elevate carbon concentrations to change ocean chemistry that alter trophic structure and food web dynamics (D'elia et al. 2003, Fisher et al. 2006, Doney et al. 2009). In high concentrations, algae produce harmful toxin levels that correspond to high incidences of invertebrate, fish, and coral mortality (Smith et al. 2001, Anderson et al. 2002). These levels of fish mortality contribute to systemic changes with an increasing anoxic, sediment saturated euphotic zone. Algal dominance suffocates both coral and stifle fish population stability to shift ecosystem food web dynamics, community structures, and native species succession (Smith et al. 2001).

The intensity and frequency of storms affect coastal water characteristics and nearshore primary biomass (Wiegner et al. 2009). In Kaneʻohe Bay, Hawaiʻi, Ringuet and Mackenzie (2005) collected oceanic surface samples five days prior to and during the course of storm events to measure chlorophyll *a* (an indicator for phytoplankton productivity), nitrogen, and phosphorus levels. Incidences of storm events were associated with increased phytoplankton biomass. These storm events compromise 80% of the annual nutrient and sediment influxes and were responsible for increases in algal biomass during storms in May, October 2002, and February 2003 (Ringuet and Mackenzie 2005). Typically large inputs of dissolved and particulate terrestrial based carbon, from runoff and fluvial introductions, tend to dominate the carbon budget of aquatic systems (Pace et al. 2007). Intensified storm events due to climate change increase erosion and stream effluence which add to the overall carbon budget (Sharma and Uehara 1968, Elison Timm et al. 2011).

Increased global sea surface temperatures (SSTs) due to climate change are expected to affect nearshore reef ecosystems. Global SSTs are projected to rise anywhere from 1.3-2.8ºC by the end of the 21st century (Munday et al. 2009), and in the Hawaiian Archipelago have already risen 0.8ºC since 1956 (Friedlander et al. 2008). Current and projected temperature changes in water have caused corals to bleach, a process in which they exude their zooxanthella symbionts (Tolleter et al. 2013).   As oceanic conditions destabilize, the effects of temperature and escalating

atmospheric CO_2 levels pose a severe threat to coral populations (Doney et al. 2009, Tolleter et al. 2013).

In Hawaii, elevated sea temperatures diminish upward transport of nutrient enriched subsurface waters, which reduce phytoplankton productivity, respiration, and dissolved oxygen levels (Mallin and Pearl 1993. Shifting temperatures may reduce water nutrient circulation (Doney 2010), that fuel nearshore fishery net productivity (Pace et al. 2007, Atwood et al. 2012). Typically large inputs of dissolved and particulate terrestrial based carbon, from runoff and fluvial introductions, tend to dominate the carbon budget of aquatic systems (Pace et al. 2007). Intensified storm events increase erosion and stream effluence adding to the carbon budget (Sharma and Uehara 1968; Elison Timm et al. 2011).

Ocean acidification, another climate change stressor, is expected to adversely affect coral health. Atmospheric CO_2 levels are increasing at a rate of at least 100 times faster than has occurred naturally for the past 650,000 years (Veron et al. 2009). Since the industrial revolution, CO_2 concentrations have risen to 400 ppm with an average annual 2.5 ppm rise; this denotes a 40 % increase which exceeds levels from over the past million years (Ateweberhan et al. 2013). Current atmospheric CO_2 trends suggest an annual 3-4 ppm rise over the next century to place global mean temperatures 5 C° above pre-industrial levels (Meinshausen et al. 2009). The amplified amount of CO_2 dissolved in the ocean has already lowered seawater pH and reduced the availability of carbonate ions, primarily aragonite and calcite, used by calcifying organisms (including reef-building corals) to form their shells and skeletons (Doney et al. 2009). As a result, significant declines have been observed in coral populations across the Pacific.

Coral mortality, due to bleaching and acidification has led to a shift towards algae-dominated seafloors. Coral bleaching events have a particular importance in Hawai'i hosting a high percentage of endemic coral species (25-40%) (Doney et al. 2012). Shifts in global and local weather patterns will continue to affect ocean acidity (Doney et al. 2009), freshwater inputs, and sea temperature (Chu et al. 2010). To model the impact of global climate change, namely freshwater inputs from shifting rainfall regimes, to nearshore water quality, coastal habitat, and fishery assemblage, we will examine a highly constrained hydrological model system along the Hilo Palikū coast of Hawai'i Island.

Nearshore Salinity

Monitoring nearshore salinity levels can be used to measure the effects of fresh water (stream effluence, storm runoff, and ground water discharge) introduced into estuary and coastal environments. The changes in rainfall correspond to increases in nearshore salinity with dramatic declines during intense storm events. Typical oceanic salinity levels are approximately 35 ppt (UNESCO 1981, Comeau 2015); decreases may be due to fresh water inputs. Global climate change trends show a continuous and gradual reduction of annual precipitation and baseline stream flow (Chu et al. 2010, Bassiouni and Oki 2012). The Hilo Palikū precipitation gradient with its wet and dry ends will be used to examine how diminishing freshwater inputs affect nearshore salinity, and in turn benthic habitat, fishery assemblages, and food web dynamics.

Coastal environments are sensitive to environmental factors and experience rapid shifts in salinity, temperature, oxygen, and turbidity which are largely influenced by tidal fluctuations that mix marine and freshwater (Strydom et al., 2003). The oceanic and fresh water interface creates conditions that support highly diverse, dynamic, and prolific fisheries (Ramos et al. 2006). The spatial and temporal salinity shifts influence the presence, density, and growth of juvenile fish (Strydom et al., 2003, Ramos 2006). As a result, overall fish community composition is in part determined by salinity and temperature conditions (Ramos et al. 2006).

Changes in salinity levels can have serious effects on coral settlement and growth to degrade fish habitat (Coles and Riegl 2013). Most corals have a salinity tolerance limit range between 28.7 and 40.4 ppt. Substantial deviations from this range stress corals, and if prolonged result in mortality (Guan 2015). Salinity and temperature are largely responsible, in addition to geographic isolation and algal competition, for what coral species populate Hawaii's nearshore. In the Indo-Pacific region only 10 % of possible coral species occur, in large part due to extremes in salinity and temperature (Coles and Riegel 2013).   Although some corals are resilient to acute salinity deviations, like the *Siderastrea radians* in South Florida, most coral species are sensitive and become susceptible to bleaching, disease, and competition from algal growth. Under normal salinity conditions corals better withstand local disturbances. For example corals more effectively remove harmful sediments from run off under normal salinity conditions (Lirman and Manzello 2008).

Thesis

The objective of this research is to examine the potential response of coastal fish communities to shifts in fresh water inputs that result from decreased rainfall. The forecasted diminished annual rainfall translates to reduced terrestrial inputs (i.e. fresh water discharge, organic matter, and nutrient loads) into near-shore ecosystems; thereby decreasing terrestrial subsides that are an important food source for extant biological communities. Consequently, I hypothesize that potential shifts in rainfall patterns due to climate change will alter near shore fishery assemblage and food web dynamics. The precipitation gradient along the Hilo Palikū (North Hilo) coast from Pepeʻekeo (6000 mm mean annual rainfall) to Laupāhoehoe (3000 mm mean annual rainfall), Hawaiʻi will serve as a proxy for forecasted global climate change. This model system will be used as a space for time substitution because as rainfall decreases along this gradient, stream flow and stream flashiness also decrease. To investigate the effects of variable fresh water inputs to near shore fishery assemblages, I examined how fish biodiversity, species relative abundances, and food web structure shift along this gradient. To investigate the response of food web dynamics, I examined a body condition index, gut content, and isotopes from samples collected at various sites along the Hilo Palikū coast.

The overall hypothesis is that differences in freshwater inputs namely surface runoff, stream discharge, and precipitation along the coast of Hilo Palikū (North Hilo), Hawaiʻi from Laupāhoehoe (3000 mm mean annual rainfall) to Pepeʻekeo/Kolekole (6000 mm mean annual rainfall) will alter coastal water salinity levels, nearshore benthic composition, near shore fish community assemblages (composition, relative abundance, and species richness) and food web dynamics.

Predictions

Coastal Water Salinity Levels

P_1: The coastal water salinity levels at different depth ranges (surface, 0.1 – 3 m, 3 – 6 m, and 6 – 9 m) along Laupāhoehoe point will be greater than that of the estuary and nearshore environments of Pepeʻekeo. I predict that Laupāhoehoe salinity levels will be greater because it receives half the rainfall annually compared to Pepeʻekeo. Fresh water, which is less dense when compared to salt water, will remain closer to the surface and have a greater diminishing impact on salinity at Pepeʻekeo.

P_2: The coastal water salinity levels for the summer months at Laupāhoehoe and Kolekole will be greater than the nearshore salinity levels for the winter months at Laupāhoehoe and Pepeʻekeo. The reason for the discrepancy being the increased amount of rainfall, therefore lower salinity levels, for the winter months.

Nearshore Benthic Composition

P_3: The benthic composition (biotic and abiotic features) along Laupāhoehoe point will differ when compared to the estuary and coastal environments of Pepeʻekeo. In particular, I predict that coral species richness and percent cover will be greater at Laupāhoehoe because of reduced fresh water inputs. Fresh water inputs introduce sedimentation that can bury and suffocate coral as well as limit light availability.

Coral Reef Fish Community Assemblages

P_4: The nearshore coral reef fish diversity along Laupāhoehoe point will be greater than that of the estuary and nearshore environments of Pepeʻekeo. Fish diversity will differ between sites due to differences in benthic habitat that is influenced by changes in fresh water inputs. Habitat complexity variables such as reef biodiversity, geography, and porosity influence resident fish community assemblages (Friedlander & Parrish 1998, Friedlander et al. 2008).

Nearshore Food Web Dynamics

P_5: The Body Condition Index (BCI) of ʻāholehole, a common nearshore fish, at Laupāhoehoe point and Kaʻawaliʻi, the dry end of the precipitation gradient, will be smaller when compared to ʻāholehole sampled at Kolekole and Pepeʻekeo. BCI will be different with the wetter end of the gradient, Kolekole and Pepeʻekeo having more food available thus larger fish.

P_6: The mass and volume of ʻāholehole gut contents sampled throughout coastal environments at Laupāhoehoe point and Kaʻawaliʻi, the dry end of the precipitation gradient, will be different from the ʻāholehole sampled at Kolekole and Pepeʻekeo. The mass and volume of gut contents will be different between sites because of variations in food availability. Sites that receive more fresh water will have more detritus and nutrients.

P₇: The Carbon and Nitrogen stable isotope signatures of ʻāholehole at Laupāhoehoe, Kaʻawaliʻi, Kolekole, and Pepeʻekeo will be different at each site. The differences will be attributed to the variation in freshwater inputs. Fresh water introductions affect food availability, thus species presence and relative abundances.

Methodology

Site Description

Climate change models, which typically function on a global scale, cannot accurately detail potential regional and local effects on hydrological, and nearshore processes, as well as reciprocated interactions. These models fail to capture systems with complex topographical features such as tropical islands (Timm et al. 2011) and many of these models are highly variable and uncertain. The precipitation gradient across Hilo Palikū, Hawaiʻi provides a unique hydrological model space-for-time system to examine how changes in rainfall anticipated to occur with climate change affects salinity, benthic composition, as well as coastal fish community assemblages and food web dynamics.

Located along the windward slopes of Mauna Kea (summit 4205 m), a young massive conical shield volcano, the weather conditions of Hilo Palikū are largely influenced by Northeasterly trade winds and associated orographic rainfall (Erasmus 1986). The trade winds produced by the northeast Pacific Ocean circulation pattern deliver moisture-laden air that is driven up the slopes of Mauna Kea. At about 1500 m, within the temperature inversion layer, the atmospheric water precipitates as fog and rainfall (Erasmus 1986). Much of the atmospheric moisture is captured by watersheds at the Southern end of the gradient, and as a result receives on average twice as much rainfall compared to its Northern boundary. The rapid loss of moisture creates a sheer precipitation gradient across a small area (Strauch et al. 2014).

The watersheds exist in a highly-constrained 3000 mm mean annual precipitation gradient with comparable geologic, landscape, and biological features, (Strauch et al. 2015). Watersheds here are similarly shaped, relatively small and narrow, and contain single- or low-order rivers. The underlying geology throughout the model system consists of Hāmākua Volcanic bedrock beneath Laupāhoehoe Volcanics of similar age ranges (13,000–64,000 yr Mauna Kea flows; Vitousek 1995).

The district features denuded sub-montane, montane and subalpine forests (artifacts of agriculture, the sandal wood trade and ranching), highly invaded wet forest (both floral and faunal invasive species), and agricultural and rural development from the forest boundary to the coast. The forests that typically occur at elevations above 600 m are dominated by ohiʻa lehua (*Metrosideros polymorpha*) trees with a stand basal area greater than 85 % (Strauch et al. 2014). The coastline marked by sheer sea cliffs offer few accessible bays, and for most of the year is subject to hazardous oceanic and weather conditions. The harsh terrain along most of this coastline coupled with harsh weather limits fishing pressure, and as a result provides relatively healthy fisheries.

Laupāhoehoe and Pepeʻekeo are situated at opposite ends of the rainfall gradient and represent ideal study sites as catchments of river effluents (Figure 1). Laupāhoehoe, located at the northern thus drier end of the gradient receives an annual average of rainfall 3000 mm. At sea level, the point at Laupāhoehoe projects out from the base of a deep gulch. Steep sea cliffs surround the point in either the direction. The study site includes two ephemeral rivers, Laupāhoehoe to the North and Kīlau to the South of Laupāhoehoe point. Located South of Kīlau stream are a series of springs that flow out from the cliff into the sea. Pepeʻekeo, at the southern and thus wet end of the gradient receives an annual average rainfall of 6000 mm. Alia river, a perennial stream, enters Pepeʻekeo bay providing fresh water and terrestrial subsidies. Although sea cliffs surround the bay, when compared to Laupāhoehoe they are significantly shorter.

At Laupāhoehoe and Pepeʻekeo the benthic substrate is characterized by large boulders covered with coral, and deeper sand channels that extend out to sea. The reef structures at Laupāhoehoe include spur and grove formations as well as jagged lava rock formations, which form barriers for tide pools that serve as nursery habitat for many species of native fish. At Pepeʻekeo the fringing reef is made up of coral flats that extend from the shore line. Large boulders, sometimes covered in coral, are interspersed between the coral flats. Alia stream delivers nutrients and serves as shelter for juvenile and estuarine fish. Kolekole is approximately 6 miles north of Pepeʻekeo. Unlike the other three sites, the Kolekole study area did not occur in a bay. The site instead was located at a cliff with a spring flowing into the ocean. Kolekole River, just a hundred yards south of the sampled area, flows perennially into the sea. The benthic substrate again is characterized by large boulders covered with some coral.

Within this model system, decreasing rainfall, stream flow, and freshwater inputs along the gradient (Bassiouni and Oki 2012, Strauch et al 2015) suggests diminishing terrestrial subsidies important for nearshore fish diet (Mallin et al. 1993, Atwood et al. 2012).   From 1975 to 2006, Hawaiʻi experienced a 27.5 % annual decline in coastal precipitation (Chu et al. 2010) corresponding to a 23 % reduction in median base flow from 1943 to 2008 compared to 1913 to 1943 (Bassiouni and Oki 2012). The increasingly drier conditions (Chu and Chen 2005) have reduced aquifer recharge and stream discharge (Strauch et al. 2015).   20th century decadal trends indicate shifts in nearshore fish assemblages, trophic structure, and decreases in fish biomass. The following are methodologies I used to quantify and characterize nearshore salinity, benthic habitat, and fish community assemblage and food web dynamics.
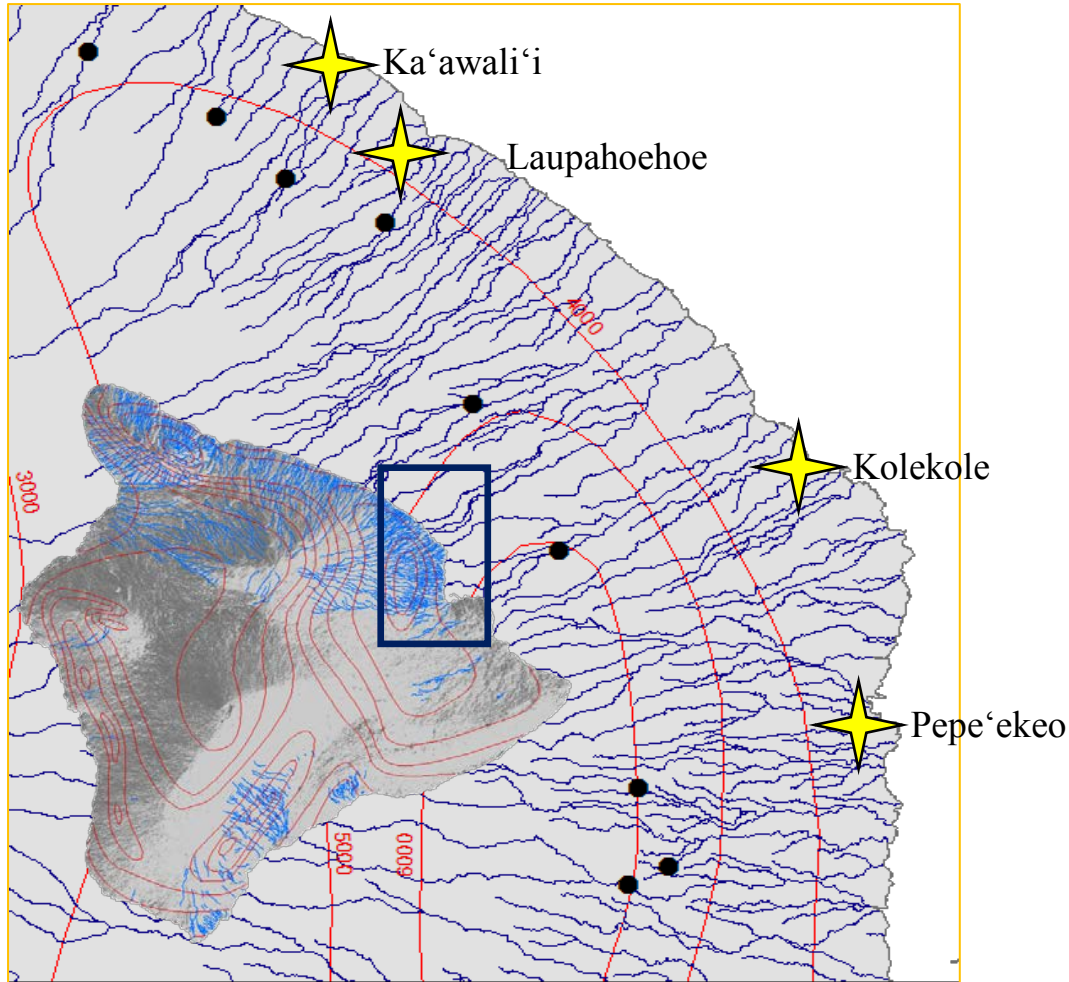
Figure 1. Map of study sites Kaʻawaliʻi, Laupāhoehoe, Kolekole, and Pepeʻekeo along the Hilo Palikū coastline, Hawaiʻi Island. The blue lines that run perpendicular to the shore are rivers while red lines running parallel represent precipitation isohyets. The black dots are stream gauge stations.

Data Collection

Nearshore Salinity

Estuarine and nearshore salinity levels, measured in parts per thousand (ppt), at Laupāhoehoe, Pepeʻekeo, Kaʻawaliʻi, and Kolekole were recorded with HOBO Conductivity loggers (brand name ONSET, www.onsetcomp.com). The logger measured Electrical Conductivity (EC) of the water as micro Siemens per centimeter (µS/cm). The data loggers were deployed avoiding periods following heavy rainfall events. The following formula (Bolland et al. 2002) was used to convert Electrical Conductivity into salinity as parts per thousand.

$$\text{Salinity (µS/cm)} = [\ 1\ EC\ (\text{µS/cm})\ /\ 25\ ^{\circ}C\ ]\ *\ temp\ (^{\circ}C)\ (\text{Eq. 1})$$

Salinity (ppt) = [Salinity (µS/cm) * 640 ppm/1000 EC (µS/cm)] * 1 ppm/1000 ppt (Eq. 2)

The summer and winter seasonal data sets at each site were used to investigate temporal shifts in salinity. The data loggers recorded Electrical Conductivity at 15 minute intervals. The loggers were housed within a PVC cylinder and secured to a cement hollow block that sat anchored at 3 m. At Laupāhoehoe the logger was deployed from July 7th, 2014 to September 8th, 2014. Here the logger was located to the south of Laupāhoehoe stream. At Kolekole the logger was deployed from June 13th,2014 to August 5th, 2014. Here the logger was located 6 m from shore where the spring flows from the cliff into the water. To compare data sets, we used the Kolekole data collected from July 7th to August 5th, 2014.

The same process to measure salinity for the summer months was utilized for the winter season with the exception of Pepeʻekeo as a substitute for Kolekole as monitored study site. The Kolekole site did not share enough physical similarities, lacking a river and bay, to be further considered for a detailed documentation of salinity measures as well as benthic substrate characterization and fish community assemblage. At Pepeʻekeo the logger was deployed from January 15th, 2015 and retrieved February 27th, 2015. Here the logger was located at the river mouth. At Laupāhoehoe the logger was deployed from January 14th, 2015 to May 6th, 2015. Here the logger was situated in the same place that it was deployed for the summer months. To compare data sets, we used the Laupāhoehoe data collected from January 15th, 2015 to February 27th, 2015.

At Laupāhoehoe and Pepeʻekeo salinity data helped to measure the diffuse effects of fresh water as depth increases and increased distance from the fresh water source (stream or spring).   The salinity points were distributed north and south from fresh water source (stream or spring). The depth ranges to measure salinity and were 0 - 3 m, 3 – 6 m, and 6 – 9 m. ‘ Fresh and salt water become stratified due to densities differences; these ranges were selected to measure the effective range of fresh water introductions. The data logger recorded Electrical Conductivity at one minute intervals. Electrical conductivity was measured for at least 15 minutes between point intercept transects, spaced at least 6 m apart, with the probe mounted on the dive float.   At the beginning of each transect, the probe was anchored to the sea floor and left to record for at least 15 minutes.

Twelve random points were recorded at each study location to measure the spatial variability of salinity. These transects were dispersed amongst the 12 point intercept transects. These transects did not involve benthic characterization. To measure for spatial variability along each transect, electrical conductivity was recorded at 5 m intervals along the 25 m transect for at least five minutes.

At Laupāhoehoe, from September 19th to January 14th, salinity was measured every 5 m along three 25-m long three transects within the 0.1 - 3 m depth range, at four transects within 3 – 6 m, and at five transects within 6 – 9 m. On January 22nd, 2015, twelve different random transects were conducted where salinity was recorded every 5 m for 25 m. At Pepeʻekeo, from October 22nd to January 15th, 2015 salinity was measured every 5 m at four transects within the 0.1 – 3 m depth range, at four transects within 3 – 6 m, and at four transects within 6 - 9m (at two

transects salinity points were measured every 5m for 25m). On January 15nd, 2015, twelve different random transects were conducted where salinity was recorded every 5 m for 25 m. At Kaʻawaliʻi, from September 18th to September 22nd, 2015 salinity was measured every 5 m at one transect within the 3 – 6 m depth range, and at one transect within 6 – 9 m. The salinity measures were conducted over a period of a few months due to weather and availability of other divers.

To normalize the effects of difference in stream flow over time at each location, a Q_{50} was calculated at each location and used to compute stream flow. The tide during the time of the sampling regime was also measured to normalize it's the effects on salinity levels. A Q_{50} represents the median flow, or the flow that is exceeded 50% of the time. The following is a formula used to calculate each Q_{50}.

$$Q_{50} = 4.49 * [\text{ Drainage area (sq mi) } \wedge 0.0808] * [\text{ Mean channel elevation (ft) } \wedge 0.641]$$
$$* [\text{ Precipitation (in) } \wedge 0.985] \text{ (eq. 3)}$$

The Q_{50} for Honoliʻi stream was used to calculate stream flow daily averages at Laupāhoehoe and Pepeʻekeo. The Q_{50} for Honoliʻi was used because stream flow daily averages were well documented via stream gauges. The following formula was used to calculate stream flow daily averages for Laupāhoehoe and Pepeʻekeo.

$$\text{Stream flow (ft}^\wedge 3/\text{s)} = [\text{ Laupāhoehoe or Pepeʻekeo } Q_{50} / \text{ Honoliʻi } Q_{50}] * \text{Honoliʻi stream}$$
$$\text{flow daily average (eq. 4)}$$

Benthic Substrate Type Composition

Benthic substrate was measured using the point intercept transect method. The 25 m point intercept transects were oriented parallel with the shoreline and spaced at least 3 m apart. Paired divers worked in unison to deploy a 25 m transect tape along the benthos. The bottom left corner of a 1m x1m double strung quadrat was positioned at 5 m intervals along the 25 m transect tape. With the quadrat resting on the sea floor, a diver recorded the substrate type as sand, rock, turf algae, crustose coralline algae, and coral for a total of 5 quadrat readings for each 25 m transect. Each quadrat was photographed to accurately identify coral species and percent cover. From September 19th, 2014 to January 15th, 2016, over the time span a total of 12 point intercept transects were conducted for each study site, Laupāhoehoe and Pepeʻekeo, at three depth ranges (0.1 – 3 m, 3 – 6 m, and 6 – 9 m). At Laupāhoehoe, from September 19th, 2014 to January 14th, point intercept transects were conducted with a distribution of three transects within the 0.1 - 3 m depth ranges, at four transects within 3 – 6 m, and at five transects within 6 – 9 m. At Pepeʻekeo, from October 22nd, 2014 to January 15th, 2015, point intercept transects were conducted with a distribution of four transects within the 0.1 – 3 m depth range, at four transects within 3 – 6 m, and at four transects within 6 – 9 m.

Fish Community Composition

Divers conducted belt transects (25 m x 5 m x 5 m) established at Laupāhoehoe and Pepeʻekeo to visually identify fish biodiversity, relative abundance and size. In total, 24 belt

transects were completed. The 12 transects at each study site were oriented in the northern and southern direction of the primary fresh water input (stream or spring). The transects were spaced at least 6 m apart ran parallel to the coast, and fell within three depth ranges (0.1 – 3 m, 3 – 6 m, and 6 – 9 m). The divers in unison deployed the 25 m transect tape. Following transect deployment; each diver surveyed within a 2.5 m x 2.5 m x 2.5 m field of view on either side of the transect tape.   The divers noted fish species abundance and size (categorized into three size classes; 0 – 10 cm, 10 – 20 cm, 20 – 30 cm, and > 30 cm). Each transect was completed within 15 minutes (Kilgo 20120, Friedlander and Parrish 1998).

Dives were conducted at Laupāhoehoe and Pepeʻekeo between January 14th 2015 and March 23rd 2016. At Laupāhoehoe dives were conducted on the following dates; two transects were completed at depths 8 and 4 m on January 14th, 2014, three transects were completed at depths 7, 6.5, and 9 m on February 20th 2016, and eight transects were completed at 8.5, 6, 9.5, 5.5, 5, 8, and 9.5 m on March 23rd, 24th and 27th 2016. At Pepeʻekeo two transects each were conducted on June 12th, 18th, 19th, 23rd, and 27th.

For each study site, species richness, diversity (via a Shannon Index, eq. 5), and evenness (eq. 6) were calculated for each group of fish species (Magurran 2004). To measure diversity, I used the Shannon index where H is the Shannon diversity index, and p_i is the ratio of the ith fish species to the total number of fish observed at study site (eq. 5, Magurran 2004). To measure evenness, I used Pielouʻs index, where H equals the Shannon index, and S is the total number of fish observed at each study site (eq. 6, Iwashita et al. 2012).

$H = - (S / i = 1) \, p_i \ln p_i$ (eq. 5, Magurran 2004)

$J = H / \ln S$ (eq. 6, Iwashita et al. 2012)

Gut Content Analysis and Body Complexity Index

During the summer and fall of 2014, ʻāholehole (Khulia spp.) were caught with a reel and rod as well as six foot three prong spears. The tackle used on the fishing line included three-foot leader lines with split shot leads (0.44 g) and size ten hooks. Three-inch glow sticks were attached to the floater to attract fish while fishing at night. The ʻāholeahole (*Kuhlia spp.*) nocturnally migrate into the water column to feed on freely suspended zooplankton, polychaete worms, insects, and algae (Randall 1996).   Bait included store bought white shrimp and locally caught ʻopae (*Atyoida bisculcata*). Ten ʻāholehole were harvested at Laupāhoehoe, three at Kaʻawaliʻi, six at Pepeʻekeo, and four at Kolekole. Due to problems with access, weather, and funding, these sites were not sampled for all measurements; in these cases, the aforementioned subset of site data was used for seasonal salinity, gut content, BCI, and Nitrogen and Carbon stable isotope comparisons.

Fish caught were measured from mandible to caudal peduncle (standard length; cm) as well as to the tip of the tail (total length; cm). Subsequently, the somatic and gonadal weights were recorded (g). The gut contents after removal were preserved in ethyl alcohol for laboratory analysis where contents were weighed (g) wet as well as dehydrated. The Body Complexity Index (BCI) was calculated using the following equation.

$$\text{Body Complexity Index} = [\text{ Fish total length/weight}^3]\ (\text{eq. 4})$$

The extracted fish gut was preserved in ethyl alcohol for lab wet/dehydrate weights and volume displacement measurements (Figure 2). In addition, both the Scanning Electron Microscope (SEM) and naked eye identified fish gut contents to characterize fish diet: crustaceans, algae, insects, etc. (Figure 3; Carlini et al. 2015). To aid in identification, unknown dried gut contents were gold coated on an aluminum stub for further inspection with the Scanning Electron Microscope (Carlini et a. 2015). Finally, a section of muscle tissue at least 3 - 5 g located under the right side of secondary dorsal was collected for isotopic analysis with the mass spectrometer (Figure 4).

Figure 2. Volume displacement content (crustacean; megalopa) from Kaʻawaliʻi.
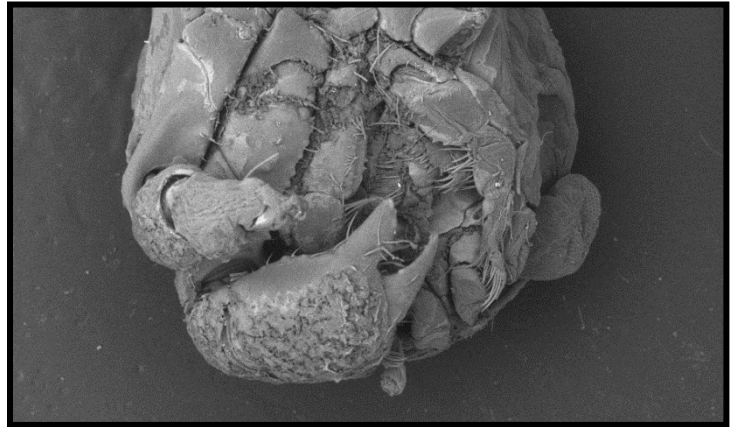
Figure 3. A scanning electron microscope image of a crustacean (Megalopa larvae) from the gut of a *ʻāholehole* caught at Kaʻawaʻaliʻi.

Figure 4. Muscle tissue sample for isotope of gut analysis.

Isotope Analysis

Sample/Standard Preparation

Three to five grams of muscle tissue were removed from directly beneath the posterior end of the dorsal fin on the left side of the fish. This muscle tissue was dried at 70° C for 48 hours. The tissue was then pulverized into a fine powder. Approximately 0.5 – 1 mg of pulverized dried fish tissue was weighed into a tared 5 x 9 cm tin and extracted for lipids to minimize lipid effects on C and N isotope signatures. For lipid extractions, I mixed a solution (2:1) of 133 ml chloroform and 66 ml methanol. Three ml of the chloroform-methanol solvent were added to only 0.10 g +/- 0.05 of dried fish muscle tissue and soaked for more than 12 h. The remaining fish muscle tissue was analyzed without undergoing lipid extraction. I centrifuged the mixture for 20 min at 4200 RPM, after which I removed the supernatant. I then added 1 ml of solvent to the precipitate and centrifuged for 10 min at 4200 RPM.

I washed the supernatant solvent-lipid solution with 1.50 ml of 0.88% KCl solution (0.3168 g in 36 ml water), let separate, and removed the lipid-solvent layer to a reweighed aluminum dish. I washed the vial with 1 ml 0.88% KCl and added the mixture to the aluminum dish. Dishes were left to dry in a fume hood at room temperature overnight (>12h), then weighed to within 0.0001g.

Instrument setting

To set the electron flow meter, I adjusted the Helium (He) gas flow to 80 ml / min and turned Carbon Dioxide (CO_2) reference gas on. Helium gas is used to transport the volatilized samples because of its inert quality. I opened the He vacuum to $\sim 1.1 \times 10^{-6}$ psi to set the conflow interface, when closed the vacuum pressure is $\sim 8.0 \times 10^{-8}$ psi.

The Elemental Analyzer settings are adjusted to combust samples and standards. I set the oven set at 50° C, the left furnace at 1000° C, and the right furnace at 800° C. The left furnace houses the chromium oxide column, the oxidizing agent, while the right furnace houses the copper column, the reducing agent. I checked the water trap for moisture because water affects N signatures. Finally, I set the computer controls to REMOTE.

Figure 5. The University of Hawaiʻi at Hilo Elemental analyzer used to volatilize fish muscle tissue samples for isotope analysis.

Statistical Analysis

      A series of statistical analyses tested for significant differences in salinity between study sites, depth ranges, and seasons (Table 2). An Anderson-Darling assessment was used to test for normality of each data set. If normal, variances were compared to test for equality. If not normal, the data sets were compared with a non-parametric analysis. Two sample T-test or Mann-Whitney tests compared salinity between depth ranges at both sites, between sites, and seasonally. A One way ANOVA or Kruskal-Wallis test compared salinity of thee three depth ranges at each Laupāhoehoe and Pepeʻekeo. A regression analysis measured correlation between stream flow (Q_{50}) and salinity, as well as tide and salinity over time.

      To compare percent cover of coral, CCA, and turf algae between Laupāhoehoe and Pepeʻekeo, the data was first arc-sin transformed. An Anderson-Darling assessment was used to test for normality of each data set. The data sets were not normal, therefore a Mann-Whitney analysis tested for significant difference of coral, CCA, and turf algae percent cover between Laupāhoehoe and Pepeʻekeo.

      A set of statistical analyses compared fish communities at Laupāhoehoe and Pepeʻekeo. The analyses compared fish abundance totals, by site, size, depth, and fish group both between and within study sites. An Anderson-Darling assessment was used to test for normality of each data set. If normal, variances were compared to test for equality and a Paired T-test compared fish abundance between sites for damsels, surgeon fish, wrasses, hawk fish, and butterfly fish. If data distribution was not normal, a Mann-Whitney analysis compared difference between sites for total fish abundances, total fish at each size bin (0 – 10, 10 – 20, and 20 – 30 cm), total fish within depth ranges (3 – 6 and 6 – 9 m), and between targeted and non-targeted fish groups.

A Kruskal-Wallis analysis was used to test total differences between sizes at each respective location. Finally, a two sample T-test compared stable ^{15}N and ^{13}C isotopes from ʻāholehole muscle tissue between study sites as well as between lipid and non-lipid extraction at each site. There were separate comparisons for muscle tissue that did and did not undergo lipid extraction.

Results

Coastal Salinity

There were significant salinity differences within and between study sites Laupāhoehoe, Pepeʻekeo, and Kolekole. At Laupāhoehoe, there was a significant difference between summer and winter months (Table 1, summer $\mu = 26.710$, $2\sigma = 0.781$, winter $\mu = 23.150.516$, $2\sigma = 0.261$; Table 2, $P < 0.0001$). At Pepeʻekeo, there was a significant difference at depth (Table 1, $\mu = 26.578$, $2\sigma = 3.011$; Table 2, $P = 0.039$), specifically between $0.1 - 3$ (Table 1, $\mu = 30.516$, $2\sigma = 0.259$) and $6 - 9$ m (Table 1, $\mu = 25.961$, $2\sigma = 0.168$; Table 2, $P = 0.0067$). A Mann-Whitney analyses of salinity between Laupāhoehoe and Pepeʻekeo identified a significant difference at depth (Table 1, Laupāhoehoe $\mu = 26.090$, $2\sigma = 0.161$, Pepeʻekeo $\mu = 26.578$, $2\sigma = 3.011$; Table 2, $P = 0.0008$), specifically at $0.1 - 3$ m (Table 1, Laupāhoehoe $\mu = 26.181$, $2\sigma = 0.157$, Pepeʻekeo $\mu = 30.516$, $2\sigma = 0.259$; Table 2, Figure 6, $P = 0.0085$), and for the winter months (Table 1, Laupāhoehoe $\mu = 23.150$, $2\sigma = 0.261$, Pepeʻekeo $\mu = 24.742$, $2\sigma = 0.266$; Table 2, $P < 0.001$). There were significant differences between Laupāhoehoe and Pepeʻekeo during the winter months (Tables 1, Laupāhoehoe $\mu = 23.150$, $2\sigma = 0.261$, Pepeʻekeo $\mu = 24.742$, $2\sigma = 0.266$; Table 2, $P < 0.001$) and between Laupāhoehoe and Kolekole for the summer months (Table 1, Laupāhoehoe $\mu = 26.710$, $2\sigma = 0.781$, Kolekole $\mu = 30.867$, $2\sigma = 0.482$; Table 2, $P < 0.001$).

To account for time between sampling, I measured how changes in stream flow (Q_{50}) and tide over time affect salinity. At Laupāhoehoe, stream flow was positively correlated with salinity at depth (Figure 7, $F = 4.073$, $P < 0.001$) and negatively correlated with surface salinity (Figure 8, $F = 0.0744$, $P = 0.264$). Tidal height was negatively correlated with salinity (Figure 9, $F = 6.746$, $P < 0.001$). At Pepeʻekeo, stream flow was positively correlated with both salinity at depth (Figure 10, $F = 0.763$, $P < 0.001$) and at the surface (Figure 11, $F = 0.526$, $P = 0.367$). Salinity was positively correlated with tidal height (Figure 12, $F = 3.462$, $P < 0.001$). Although, there were correlations between salinity and streamflow as well as tide, the variation in salinity did not have a strong correlation with the independent variable (stream flow and tidal height; $F > 0.05$).

Table 1. At Laupāhoehoe, Pepeʻekeo, Kolekole, mean salinity (ppt) and standard deviation (μ +/- 2 σ) at the surface, depth ranges (0.1 - 3 m, 3 - 6 m, 6 - 9 m, and 0 - 9m), and seasons.

Location	Variable	Description	STDEV	Mean
Laupāhoehoe	Salinity (ppt)	Surface	2.542	12.892
Laupāhoehoe	Salinity (ppt)	0.1-3m	0.157	26.181
Laupāhoehoe	Salinity (ppt)	3-6m	0.123	26.047
Laupāhoehoe	Salinity (ppt)	6-9m	0.168	25.961
Laupāhoehoe	Salinity (ppt)	Summer	0.781	26.710
Laupāhoehoe	Salinity (ppt)	Winter	0.261	23.150
Laupāhoehoe	Salinity (ppt)	0.1–9 m	0.161	26.090
Pepe'ekeo	Salinity (ppt)	Surface	2.961	14.189
Pepe'ekeo	Salinity (ppt)	0.1-3m	0.259	30.516
Pepe'ekeo	Salinity (ppt)	3-6m	3.762	26.726
Pepe'ekeo	Salinity (ppt)	6-9m	2.667	26.342
Kolekole	Salinity (ppt)	Summer	0.482	30.867
Pepe'ekeo	Salinity (ppt)	Winter	0.266	24.742
Pepe'ekeo	Salinity (ppt)	0.1-9m	3.011	26.578
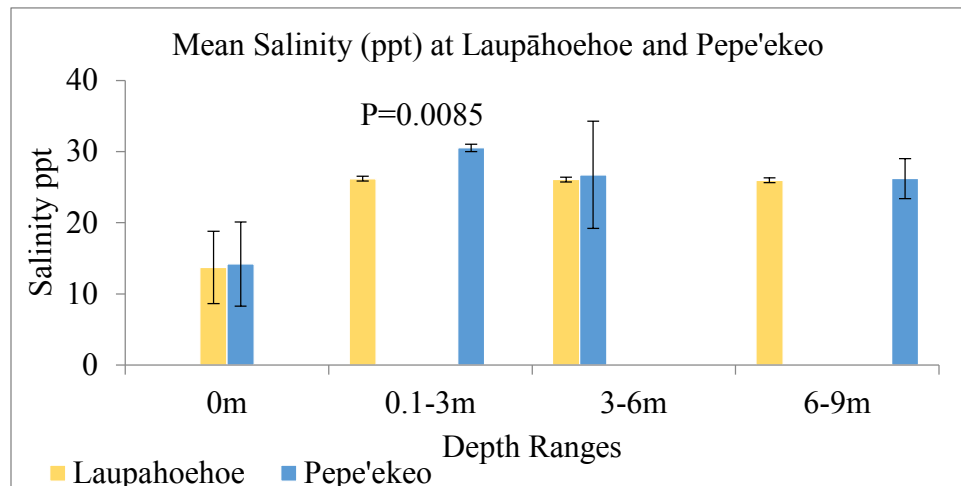
Figure 6. Laupāhoehoe and Pepeʻekeo mean salinity (ppt) data with standard deviation bars (μ + / - 2 σ); there were significant differences at 0.1 - 3 m (P = 0.0085).

Table 2. A list of statistical analyses conducted to measure significant differences in salinity (ppt) levels at Laupāhoehoe, Pepeʻekeo, Kolekole.

Statistical Analysis	Location(s)	Comparison	Statistical Variables	H₀
Mann-Whitney	Lau vs Pee	Salinity at depth	P=0.0008, W=155	FALSE
Mann-Whitney	Lau vs Pee	Depth 0.1-3m	P=0.0085, W=36	FALSE
One way Anova	Pepe'ekeo	Salinity at depth ranges	P= 0.039, F=3.88	FALSE
Mann-Whitney	Pepe'ekeo	Depth 0.1-3 vs 6-9m	P= 0.0067, W=64	FALSE
Mann-Whitney	Laupāhoehoe	Summer vs Winter	P<0.001, W=5.7x10^7	FALSE
Mann-Whitney	Lau vs Kole	Summer	P<0.001, W=2.9x10^6	FALSE
Mann-Whitney	Lau vs Pee	Winter	P<0.001, W=3.1x10^8	FALSE
Mann-Whitney	Pee vs Kole	Summer vs Winter	P<0.001, W=2.5x10^7	FALSE
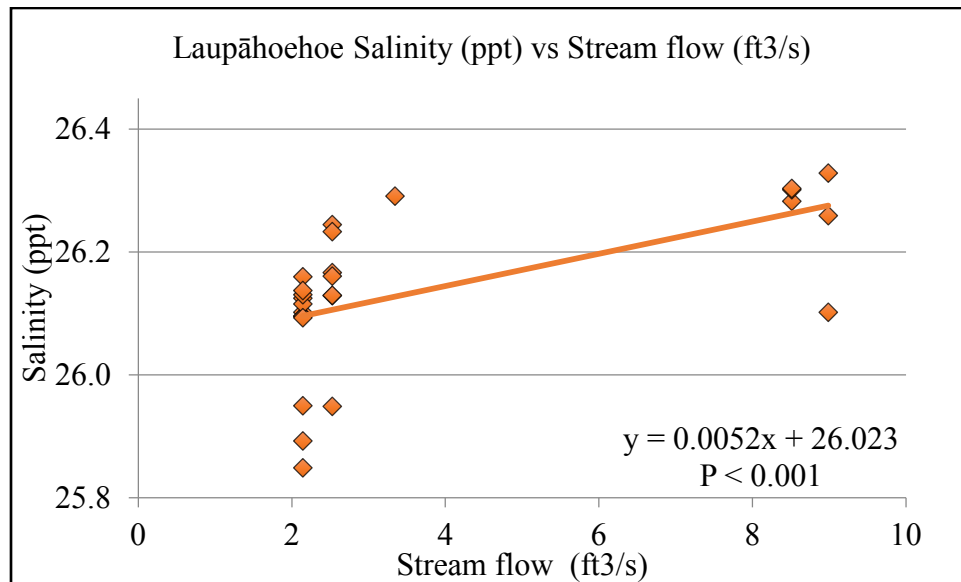
Figure 7. The effect of stream flow (ft^3/s) on adjacent nearshore salinity (ppt) levels from 0.1 – 9m at Laupāhoehoe (P < 0.001, Q_{50} =10.693).
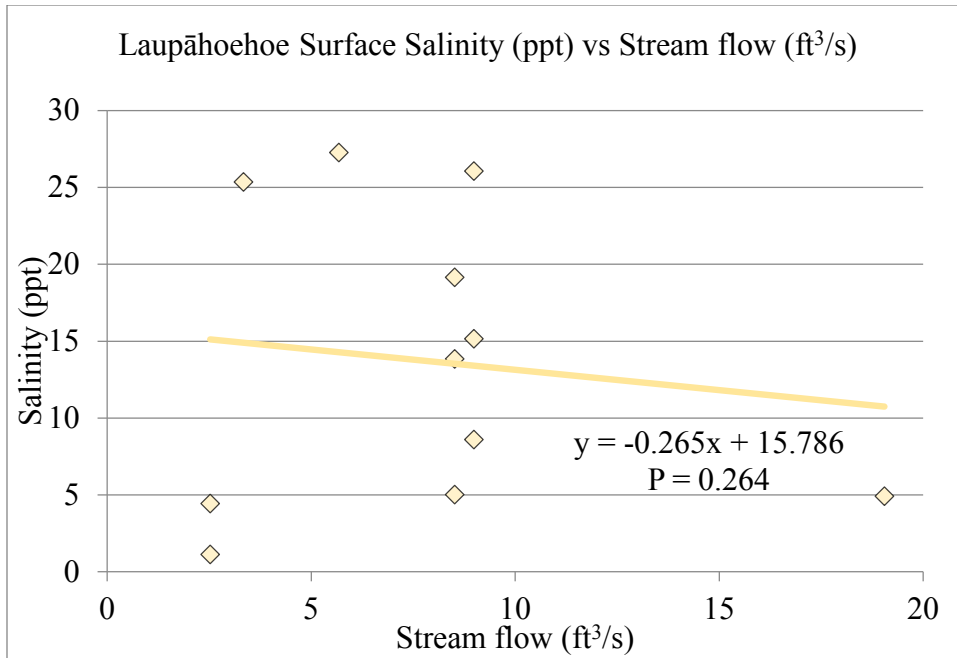
Figure 8. The effect of stream flow (ft^3/sec) on adjacent nearshore surface salinity (ppt) levels
($P = 0.264$, $Q_{50} = 10.693$).
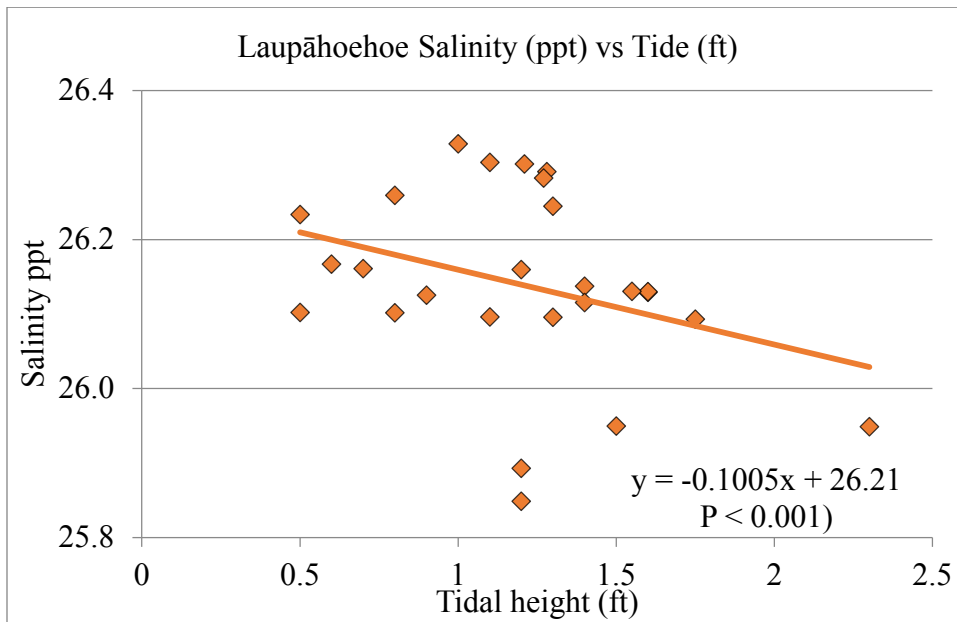
Figure 9. The relationship between tidal height (ft) and nearshore salinity (ppt) levels at
Laupāhoehoe ($P < 0.001$).
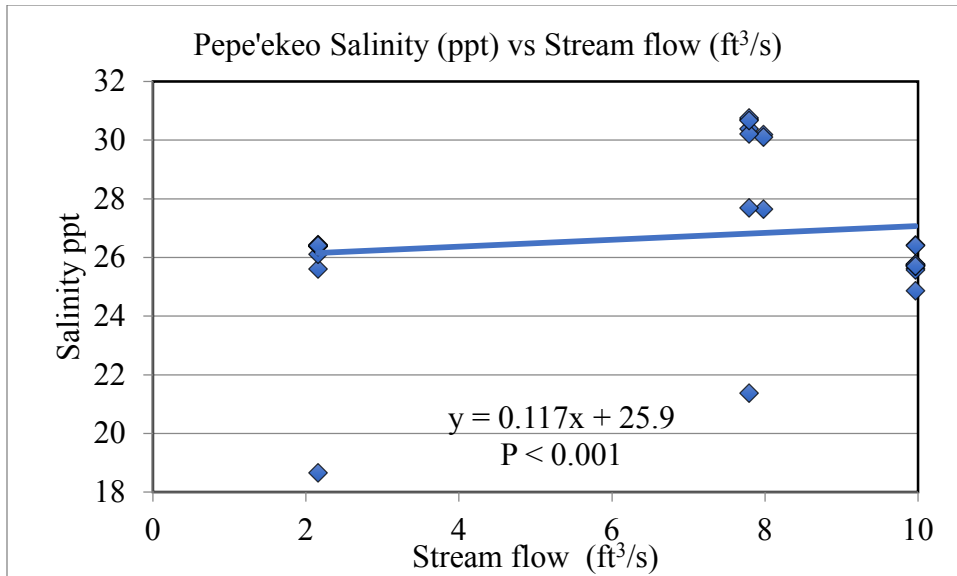
Figure 10. The relationship between stream flow (ft^3/sec) and adjacent nearshore salinity (ppt) levels at Pepeʻekeo (P < 0.001, Q_{50} = 9.360).
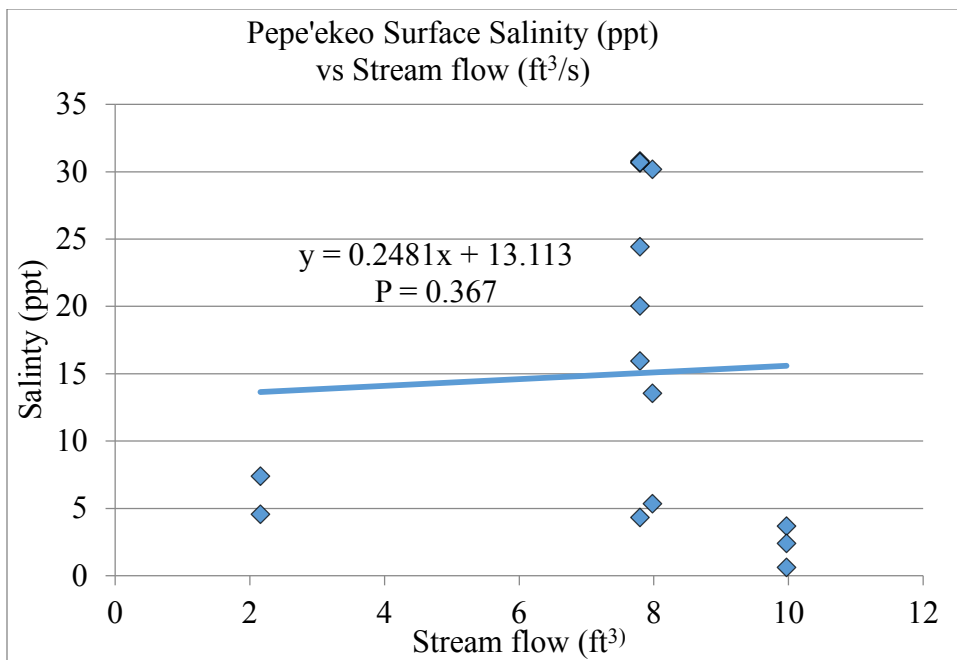
Figure 11. Pepeʻekeo river stream flow (ft^3/sec) compared to adjacent nearshore surface salinity (ppt) levels at Pepeʻekeo (P = 0.367, Q_{50} = 9.360).
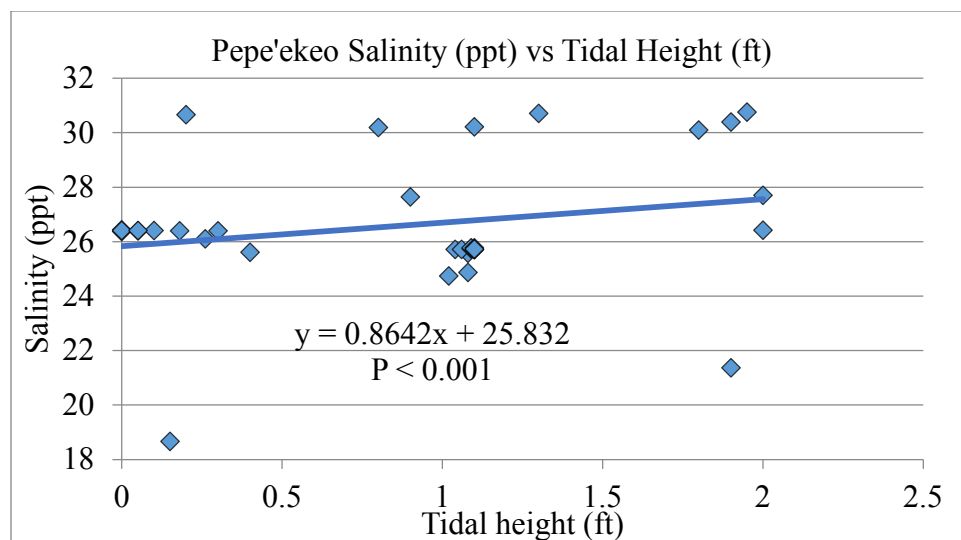
Figure 12. The relationship between tidal height (ft) and nearshore salinity (ppt) levels at Pepeʻekeo (P < 0.001, $Q_{50} = 9.360$).

Nearshore Benthic Composition

The benthic substrate types at Laupāhoehoe and Pepeʻekeo were characterized as coral (by species), turf algae, and Crustose Coralline Algae (CCA). At Laupāhoehoe, the benthos was covered by 51.8 % turf algae and 13.1 % CCA (Figure 13). At Pepeʻekeo, the benthos was covered by 68.4% turf algae and 0.7 % CCA (Figure 14). Between sites, turf algae (P = 0.0148) and CCA (P = 0.0004) were significantly different (Table 5, Figure 15). The abiotic features make up small percentage of the substrate at both sites (Table 3, Figure 13 and 14). At Laupāhoehoe, rock made up only 0.3 % with sand at 3.6 % of the sea floor (Figure 13). At Pepeʻekeo, the sea floor was 8.0 % rock and 3.6 % sand (Figure 14).

Coral percent cover was significantly different between sites (Table 5, Figure 15, P < 0.001). Laupāhoehoe had a greater percent coral cover at 31.5 % (Figure 13) compared to 2.7% at Pepeʻekeo (Figure 14). At Laupāhoehoe, coral mean percent cover at 0.1 – 3 m was 17.213 %, 37.063 % at 3- 6 m, and 39.323 % at 6- 9 m (Table 4). At Pepeʻekeo, coral mean percent cover at 0.1 – 3 m was 5.150 %, 0.421 % at 3- 6 m, and 39.323 at 6- 9 m (Table 4).

The coral species observed at Laupāhoehoe include *Porites meandrin Zoanthid species*, *Montipora capitata*, *Porites lutea*, *Porites lobata*, *Montipora flabellata*, *Pavona varians*, and *Montipora patula* (Table 6, Figure 16). The coral species that had the highest percent cover were *Monitopora flabellata* at 13.9 %, *Porites meandrina* at 7.7 %, and *Porites lobata* at 4.1 %. The coral species that had the lowest cover was *Porites lutea* at 0.159 % (Table 6, Figure 16). The coral species observed at Pepeʻekeo included *Porites meandrina*, *Montipora capitata*, *Porites lutea*, *Porites lobata*, and *Montipora flabellata* (Table 6, Figure 17). The coral species that had the highest percent cover were *Porites meandrina* at 1.445 % and *Montipora capitata* at 0.407 % (Table 17, Figure 17)

Table 3. Mean percent cover of substrate type (coral, biotic, and abiotic) at Laupāhoehoe (orange color scheme) and Pepeʻekeo (blue color scheme).

Location	Variable	Depth Range	Mean
Laupāhoehoe	Coral % cover	0.1-9m	31.548
Laupāhoehoe	Biotic % cover	0.1-9m	64.880
Laupāhoehoe	Abiotic % cover	0.1-9m	3.919
Pepe'ekeo	Coral % cover	0.1-9m	2.722
Pepe'ekeo	Biotic % cover	0.1-9m	69.030
Pepe'ekeo	Abiotic % cover	0.1-9m	28.853

Laupāhoehoe Benthic Substrate % Cover

- Coral % cover
- Turf % cover
- CCA % cover
- Rock % cover
- Sand % cover

0.3
3.6
13.1
31.5
51.8
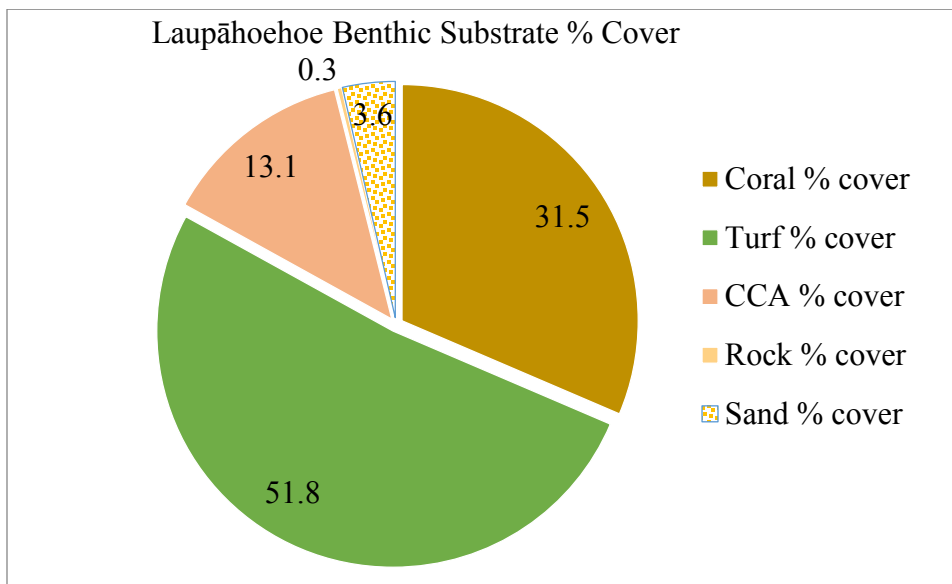
Figure 13. Benthic substrate type at Laupāhoehoe; coral (31. 5 %; brown), turf algae (51.8 %; green), Crustose Coralline Algae (CCA; 13.1; pink), rock (0.3 %; light orange), and sand (3.6 %; patterned orange) percent cover.
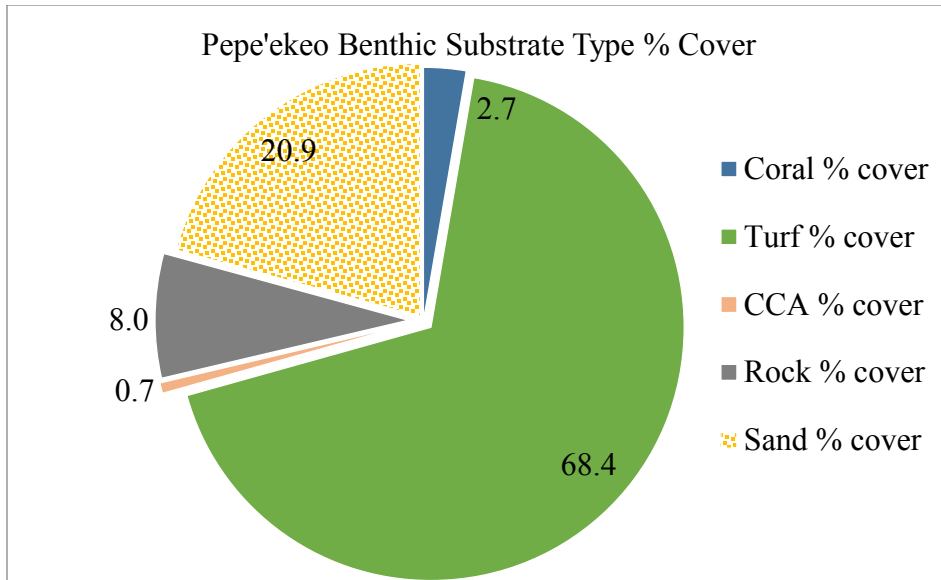
Figure 14. Benthic substrate type at Pepeʻekeo; coral (2.7 %; blue), turf algae (68.4 %; green), Crustose Coralline Algae (CCA; 0.7; pink), rock (8.0 %; gray), and sand (3.6 %; patterned orange) percent cover.

Table 4. Average percent cover of biotic and abiotic substrate type at Laupāhoehoe and Pepeʻekeo.

Location	Substrate	% Cover 0-3m	% Cover 3-6m	% Cover 6-9m	% Total Cover
Laupāhoehoe	Coral	17.213	37.063	39.323	31.548
Laupāhoehoe	Rock	0.933	0.000	0.000	0.311
Laupāhoehoe	Sand	0.517	0.118	10.190	3.608
Laupāhoehoe	Turf	52.562	52.353	50.420	51.778
Laupāhoehoe	CCA	28.775	10.466	0.067	13.103
Pepeʻekeo	Coral	5.150	0.421	2.733	2.722
Pepeʻekeo	Rock	5.275	15.111	3.500	7.962
Pepeʻekeo	Sand	9.910	18.389	34.375	20.891
Pepeʻekeo	Turf	78.995	66.895	59.169	68.353
Pepeʻekeo	CCA	0.670	1.000	0.363	0.678
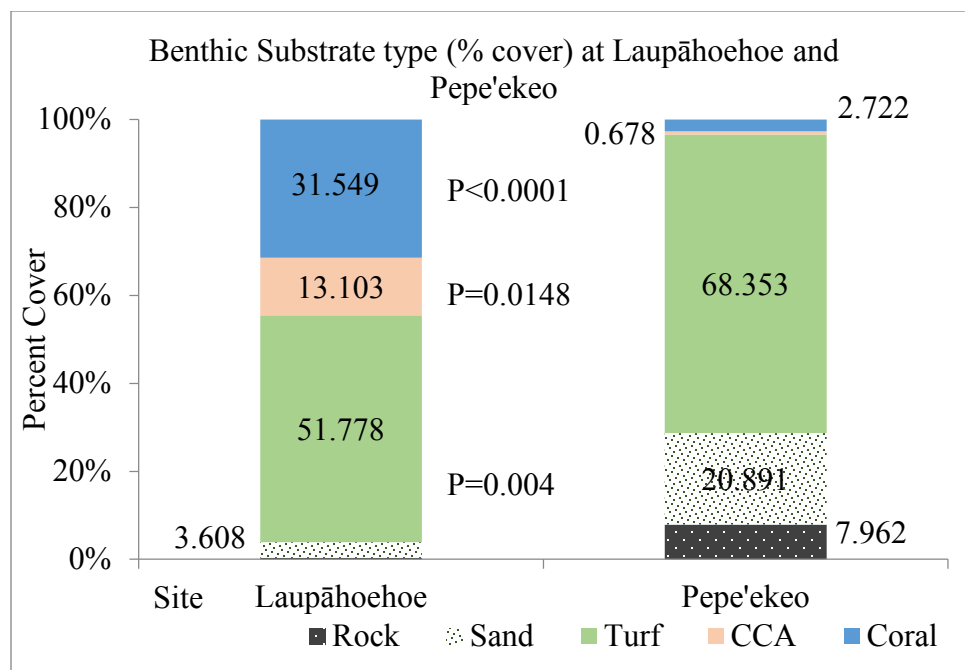
Figure 15. Percent cover of benthic substrate recorded at 0.1 – 9 m for both Laupāhoehoe and Pepeʻekeo. There were significant differences in coral (P < 0.001, W = 681755.5), CCA (P = 0.0148, W = 3213.5), and turf algae (P = 0.0004, W = 2316.5).

Table 5. Statistical analyses to measure for significant differences in total percent cover of coral, Crustose Coralline Algae (CCA), and turf algae between Laupāhoehoe and Pepeʻekeo.

Statistical Analysis	Locations	Variable	Statistical Variables	H$_o$
Mann-Whitney	Lau vs Pee	% Coral Cover	P<0.001, W=681755.5	FALSE
Mann-Whitney	Lau vs Pee	% CCA Cover	P= 0.0148 ,W=3213.5	FALSE
Mann-Whitney	Lau vs Pee	% Turf Cover	P=0.0004,W = 2316.5	FALSE

Table 6. Coral by species % cover for each depth range (0.1 – 3 m, 3 – 6 m, and 6 – 9 m) and mean % cover for 0.1 – 9 m at Laupāhoehoe and Pepeʻekeo.

Location	Coral spp.	% Cover 0.1-3m	% Cover 3-6m	% Cover 6-9m	Mean
Laupāhoehoe	*P. mea*	9.130	8.889	5.044	7.688
Laupāhoehoe	*Z. spp.*	4.187	0.000	0.579	1.589
Laupāhoehoe	*M. cap*	2.317	1.835	7.588	3.913
Laupāhoehoe	*P. lut*	0.000	0.000	0.476	0.159
Laupāhoehoe	*P. lob*	1.283	3.379	7.575	4.079
Laupāhoehoe	*M. fla*	0.417	22.518	18.690	13.875
Laupāhoehoe	*P. var*	0.000	0.359	0.262	0.207
Laupāhoehoe	*M. pat*	0.000	0.118	0.000	0.039
Pepeʻekeo	*P. mea*	0.475	0.105	0.156	0.246
Pepeʻekeo	*M. cap*	0.750	0.158	0.313	0.407
Pepeʻekeo	*P. lut*	0.050	0.079	1.000	0.376
Pepeʻekeo	*P. lob*	0.635	0.079	0.031	0.248
Pepeʻekeo	*M. fla*	3.240	0.000	1.094	1.445
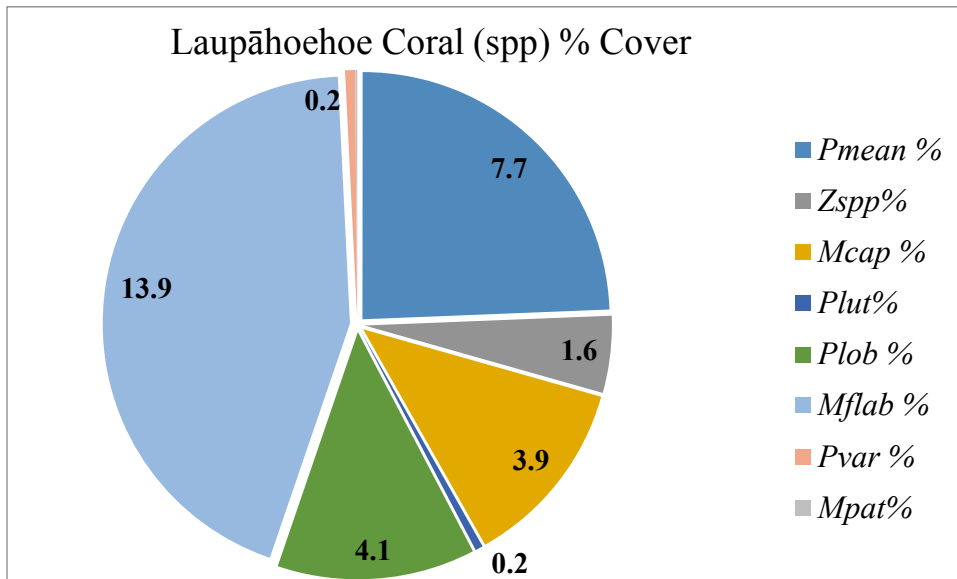
Figure 16. Coral percent cover by species at Laupāhoehoe. The percentages for each species are based on the total averages for all depth ranges (0.1 – 9 m). Coral species observed include *Porites meandrina* (blue), *Zoanthid species* (dark gray), *Montipora capitata* (orange), *Porites lutea* (dark blue), *Porites lobata* (green), *Montipora flabellata* (light blue), *Pavona varians* (pink), and *Montipora patula* (gray).
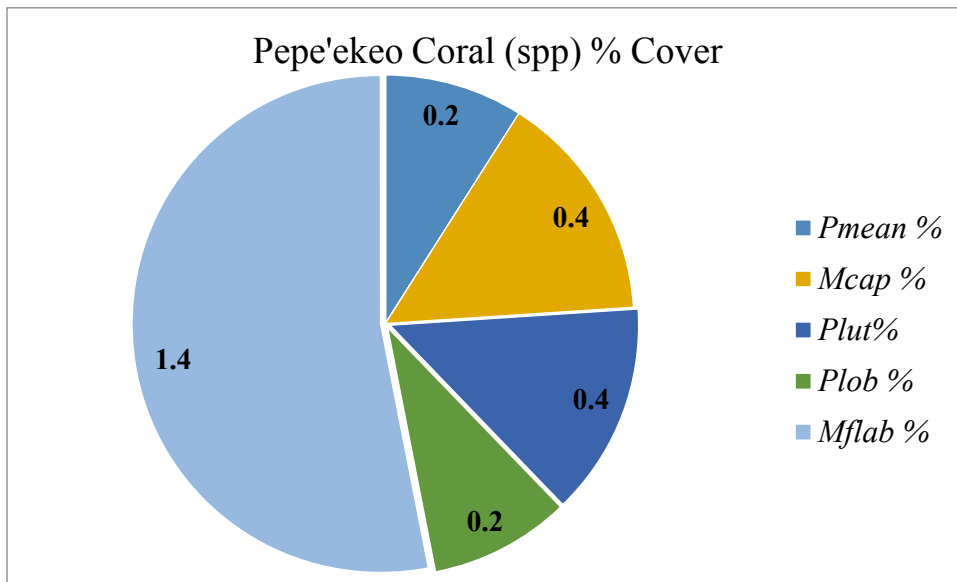
Figure 17. Coral percent cover by species at Pepeʻekeo. The percentages for each species are based on the total averages for all depth ranges (0.1 – 9 m). Coral species observed include *Porites meandrina* (blue), *Montipora capitata* (orange), *Porites lutea* (dark blue), *Porites lobata* (green)ˏ and *Montipora flabellata* (light blue).

Coral Reef Fish Community Assemblages

Species richness, diversity (via a Shannon Index, eq. 5), and evenness (eq. 6) were calculated for each group of fish species at Laupāhoehoe and Pepeʻekeo (Magurran 2004). Fish diversity, which is a measure of species richness and its proportional representation, was calculated using theShannon index. H is the Shannon diversity index, and p_i is the ratio of the *i*th fish species to the total number of fish observed at study site (p_i ; eq. 5, Iwashita et al. 2012).

H = - (S / i = 1) p_i ln p_i (eq. 5, Magurran 2004)

The number of species (species richness) observed at both Laupāhoehoe and Pepeʻekeo were 65 and 60 respectively. The total number of fish observed at Laupāhoehoe and Pepeʻekeo were 1617 and 1251 respectively. Fish diversity at Laupāhoehoe was 1.864 compared to 1.547 at Pepeʻekeo. To measure evenness, I used Pielouʻs index, where H equals the Shannon index, and S is the total number of fish observed at each study site (eq. 6, Iwashita et al. 2012). Evenness at Laupāhoehoe was 0.250 compared to 0.217 at Pepeʻekeo.

The fish groups observed at Laupāhoehoe included butterfly fish (Chaetodontidae), porcupine (Diodontidae)/box (Ostraciidae)/puffer fish (Tetraodontidae), surgeon fish (Acanthuridae), damsel fish (Pomacentridae), goat fish (Mullidae), trigger (Balistidae)/file fish (Monacanthidae), snappers (Lutjanidae), wrasses (Labridae), hawk fish (Cirrhitidae), squirrel fish (Holocentridae)/big eyes (Priacanthidae), blennies (Blenniidae), jacks (Carangidae), parrot fish (Scaridae), flag tail fish (Kuhliidae) and other fish species (Figure 18; Randal 1996). The

fish groups observed at Pepeʻekeo included butterfly fish, porcupine/box/puffer fish, surgeon fish, damsel fish, goat fish, trigger/file fish, snappers, wrasses, hawk fish, squirrel fish/big eyes, parrot fish, and other fish species (Figure 18).

Fish groups were ranked from most to least abundant and identified as target and non-targeted species. The most abundant fish groups at both sites were Damselfishes (Laupāhoehoe 539, Pepeʻekeo 496), Surgeon fish (Laupāhoehoe 415, Pepeʻekeo 244), Butterfly fish (Laupāhoehoe 222, Pepeʻekeo 39), Wrasses (Laupāhoehoe 243, Pepeʻekeo 351), and Hawk fish (Laupāhoehoe 539, Pepeʻekeo 496; Figure 18). At both sites, there were significant differences in abundances for wrasses (P = 0.049), surgeon fish (P = 0.0006), butterfly fish (P = 0.014), and hawk fish (P < 0.001; Figure 18). There was a significant difference between targeted (snappers, goatfish, other, squirrelfish/bigeyes, parrot fish, jacks, and flag tail) and non-targeted fish (damsels, surgeons, butterfly fish, wrasses, hawk fish, trigger/file fish, porcupine/box/puffer fish, and blennies; P<0.0001, Figure 18).

The fishes were catalogued by the depth range (3 – 6 m, 6 – 9 m) at which they were identified. At Laupāhoehoe, 583 fish were observed between 3 – 6 m (μ = 4.7, σ = 8), and 1033 fish were observed between 6 – 9 m (μ = 6.1, σ = 10; Figure 19). At Pepeʻekeo, 432 fish were observed between 3 – 6 m (μ = 4.5, σ = 6), and 819 fish were observed between 6 – 9 m (μ = 5.4, σ = 7; Figure 19). There were no significant differences between sites for fish abundances by depth range.

The observed fishes were further categorized by size classes 0 – 10 cm, 10 – 20 cm, and 20 – 30 cm. At Laupāhoehoe, 930 fish were observed in the 0 – 10 cm size class (μ = 92.6, σ = 153), 516 fish in the 10 – 20 cm size class (μ = 43.5, σ = 69), and 90 fish in the 20 – 30 cm size class (μ = 7.6, σ = 8; Figure 20). There were significant differences amongst size classes (P < 0.001, Table 7) between 0 – 10 and 10 – 20 cm (P < 0.001), between 0 – 10 and 20 – 30 cm (P <0.001), and between 10 - 20 and 20 - 30 cm (P = 0.0009; Figure 20). For surgeon fishes, there were significant differences amongst size classes (P = 0.012), between 0 – 10 and 20 – 30 cm (P = 0.0146), and between 10 -20 and 20 -30 cm (P = 0.0395; Figure 21). For wrasses, there were significant differences amongst size classes (P = 0.003, Figure 23). For hawk fishes, there were significant difference amongst size classes 0 – 10 and 10 – 20 cm (P = 0.004, Figure 24)

At Pepeʻekeo, 929 fish were observed in the 0 – 10 cm size class (μ = 82.4, σ = 144), 253 fish in the 10 – 20 cm size class (μ = 28.1, σ = 41), and 64 fish in the 20 – 30 cm size class (μ = 7.1, σ = 10; Figure 20). There were significant differences amongst size classes (P < 0.001, Table 7), between 0 – 10 and 10 – 20 cm (P = 0.0002), and between 0 – 10 and 20 – 30 cm (P = 0.0001; Figure 20). For damsel fishes, there were significant differences between size classes 0 – 10 and 10 – 20 cm (P = 0.005, Figure 21). For surgeon fishes, there were significant differences between size classes 10 -20 and 20 -30 cm (P=0.0496, Figure 22). For wrasses, there were significant differences between size classes (P < 0.001), between 0 -10 and 10 - 20 cm (P = 0.003), and between 0 – 10 and 20 – 30 cm (P = 0.002; Figure 23). For hawk fishes, there was a significant difference between size classes 0 – 10 and 10 – 20 cm (P = 0.049, Figure 24).

When comparing both sites, there was a significant difference between total fish for size class 10-20 cm (P = 0.038; Figure 20). For wrasses, there was a significant difference for size

class 0 – 10 cm (P = 0.002), and 10 – 20 cm (P = 0.017; Figure 23).   There were no other significant differences between sites for fish abundances by size class.

The data and samples gathered at Kolekole were four ʻāholehole and salinity from June 13ᵗʰ to August 5ᵗʰ, 2014. Three ʻāholehole was harvested from Kaʻawaliʻi; no salinity, benthic, and fish community data was recorded at this site. Due to problems with access, weather, and funding, these sites were not sampled for all measurements; in these cases, the subset Kaʻawaliʻi data was used for seasonal salinity, gut content, BCI, and nitrogen and carbon stable isotope comparisons.

Fish BCI and carbon (C13) and nitrogen (N15) stable isotopes were compare to better understand food web dynamic differences between Laupāhoehoe and Pepeʻekeo. Statistical comparisons for carbon (C13) and nitrogen (N15) stable isotope and BCI between sites were not significant (Table 9).
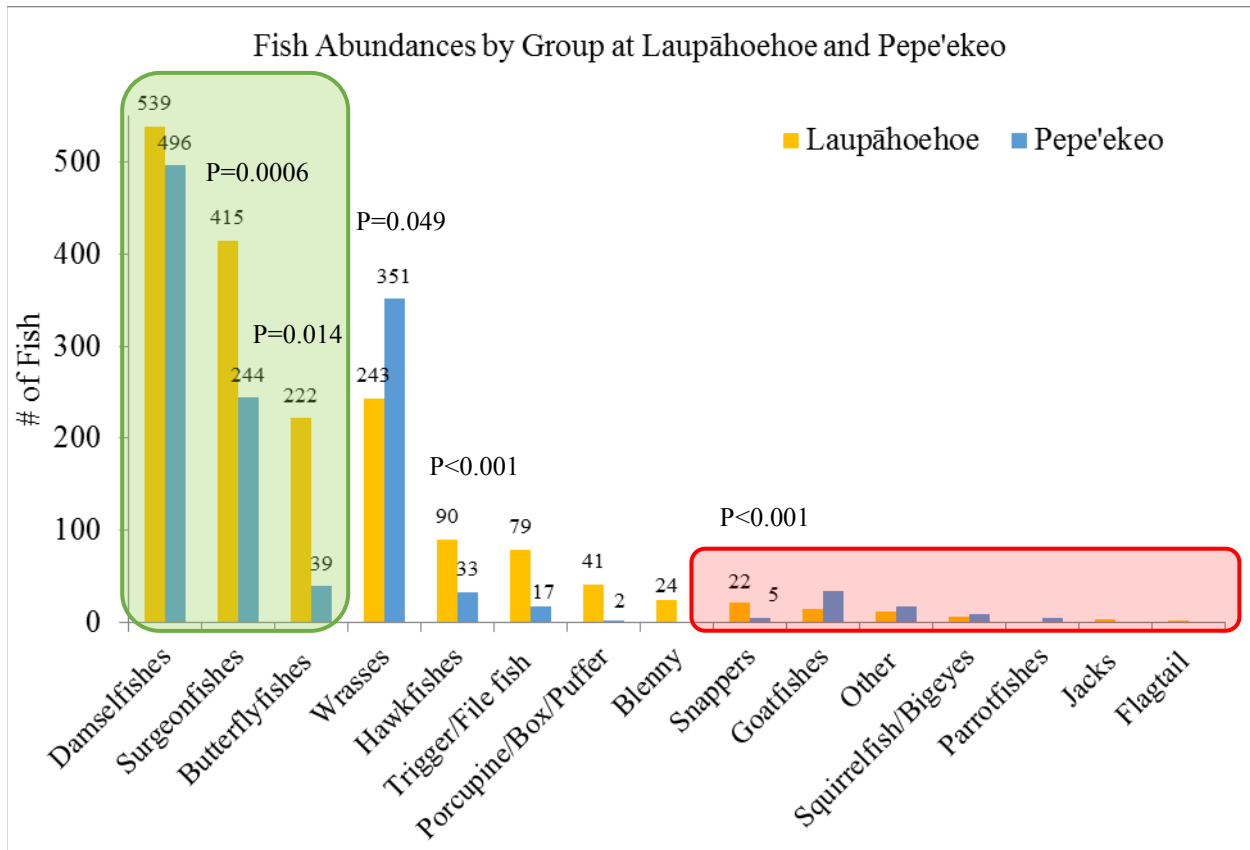
Figure 18. Fish abundances observed at Laupāhoehoe and Pepeʻekeo. The transparent red region identifies low abundance species that are subject to fishing pressure. The green area identifies high abundance herbivorous species. There were significant differences in wrasses (P = 0.049), hawk fish (P < 0.001), surgeon fish (P = 0.0006), butterfly fish (P = 0.014), and between targeted and non-targeted species (P < 0.0001).

Figure 19. Number of fish observed by depth range; 3 – 6 m (Laupāhoehoe μ = 4.7, σ = 8, Pepeʻekeo μ = 4.5, σ = 6), and 6 – 9 m (Laupāhoehoe μ = 6.1, σ = 10, Pepeʻekeo μ = 5.4, σ = 7).

Figure 20. Number of fish observed by size class; 0 – 10 cm (Laupāhoehoe μ = 92.6, σ = 153, Pepeʻekeo μ = 82.4, σ = 144), 10 – 20 cm (Laupāhoehoe μ = 43.5, σ = 69, Pepeʻekeo μ = 28.1, σ = 41), and 20 – 30 cm(Laupāhoehoe μ = 7.6, σ = 8, Pepeʻekeo μ = 7.1,, σ = 10). There were significant difference for size class 10-20 cm (P = 0.038).

Figure 21. Number of Damsel fish by size class; 0 – 10 cm (Laupāhoehoe μ = 10.6, σ = 15, Pepeʻekeo μ = 9.6, σ = 8) 10 – 20 cm (Laupāhoehoe μ = 2.9, σ = 2) 20 – 30 cm (Pepeʻekeo μ = 6.3, σ = 7).

Figure 22. Number of Surgeon fish by size class; 0 – 10 cm (Laupāhoehoe μ = 1, σ = 3.15, Pepeʻekeo μ = 0.79, σ = 3.1), 10 – 20 cm (Laupāhoehoe μ = 2, σ = 7.2, Pepeʻekeo μ = 1, σ = 3.9), 20 – 30 cm (Laupāhoehoe μ = 0, σ = 0.9, Pepeʻekeoμ = 0.07, σ = 0.62).

Figure 23. Number of Wrasses by size class; 0 – 10 cm (Laupāhoehoe μ = 4.8, σ = 4, Pepeʻekeo μ = 5.9, σ = 8), 10 – 20 cm (Laupāhoehoe μ = 2.6, σ = 2, Pepeʻekeo μ = 2.5, σ = 3), 20 – 30 cm (Laupāhoehoe μ = 1, σ = 0, Pepeʻekeo μ = 1.2, σ = 0.4). There were significant difference for size class 0 – 10 cm (P = 0.002), and 10 – 20 cm (P = 0.017).
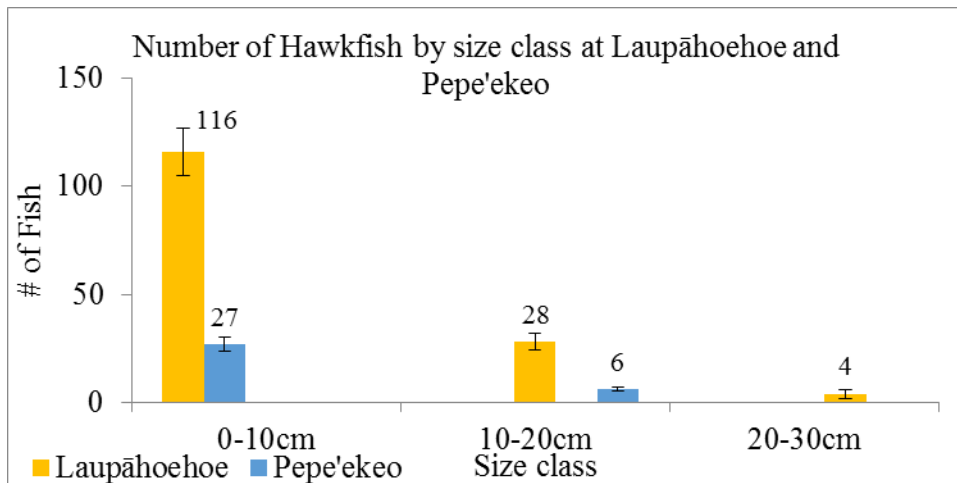
Figure 24. Number of Hawk fish by size class; 0 – 10 cm (Laupāhoehoe μ = 7.7, σ = 6, Pepeʻekeo μ = 2.4, σ = 1.6), 10 – 20 cm (Laupāhoehoe μ = 3, σ = 2, Pepeʻekeo μ = 1.2, σ = 0.4), 20 – 30 cm (μ = 1.3, σ = 0.6).

Table 7. Statistical analyses conducted to measure significant difference in fish assemblage parameters at Laupāhoehoe and Pepeʻekeo.

Statistical Analysis	Location(s)	Comparison	Statistical Variables	H₀
Mann-Whitney	Lau vs Pee	Total fish @ 10-20cm	P=0.038, W=12937.5	FALSE
Mann-Whitney	Lau vs Pee	Wrasse Total	P=0.0039, W=633.5	FALSE
Two sample T-Test	Lau vs Pee	Wrasse 0-10cm	P=0.002, T=3.58	FALSE
Two sample T-Test	Lau vs Pee	Wrasse 10-20cm	P=0.017, T=2.81	FALSE
Kruskal-Wallis	Laupāhoehoe	Total between sizes	P<0.001, H=41.30	FALSE
Mann-Whitney	Laupāhoehoe	Total fish 0-10 vs 10-20cm	P<0.001, W=23526.0	FALSE
Mann-Whitney	Laupāhoehoe	Total fish 0-10 vs 20-30cm	P<0.001, W=16153.5	FALSE
Mann-Whitney	Laupāhoehoe	Total fish 10-20 vs 20-30cm	P=0.0009, W=11019.0	FALSE
Kruskal-Wallis	Laupāhoehoe	Surgeon size classes	P=0.012, H=8.92	FALSE
Mann-Whitney	Laupāhoehoe	Surgeon 0-10 vs 20-30cm	P=0.0146, W=366.0	FALSE
Mann-Whitney	Laupāhoehoe	Surgeon 10-20 vs 20-30cm	P=0.005, W=527.0	FALSE
Kruskal-Wallis	Laupāhoehoe	Wrasse Size classes	P=0.003. H=11.64	FALSE
Two sample T-Test	Laupāhoehoe	Hawk 0-10 vs 10-20cm	P=0.004, T=3.35	FALSE
Kruskal-Wallis	Pepeʻekeo	Total between sizes	P<0.001, H=28.5	FALSE
Mann-Whitney	Pepeʻekeo	Total fish 0-10 vs 10-20cm	P=0.0002, W=19312.0	FALSE
Mann-Whitney	Pepeʻekeo	Total fish 0-10 vs 20-30cm	P=0.0001, W=14491.5	FALSE
Mann-Whitney	Pepeʻekeo	Damsel 0-10 vs 10-20cm	P=0.005, W=1634.0	FALSE
Kruskal-Wallis	Pepeʻekeo	Wrasse Size classes	P<0.001, H=16.86	FALSE
Mann-Whitney	Pepeʻekeo	Wrasse 0-10 vs 10-20cm	P=0.003, W=1972.5	FALSE
Mann-Whitney	Pepeʻekeo	Wrasse 0-10 vs 20-30cm	P=0.002, W=1598.5	FALSE
Two sample T-Test	Pepeʻekeo	Hawk 0-10 vs 10-20cm	P=0.049, T=2.19	FALSE

Table 8. Statistical comparisons of most abundant fish groups, and targeted vs non-targeted fish groups at Laupāhoehoe and Pepeʻekeo.

Statistical Analysis	Locations	Variable	Statistical Variables	Ho
Paired T-test	Lau vs Pee	Total # of Damsels	P=0.765, T=0.30	TRUE
Paired T-test	Lau vs Pee	Total # of Surgeon fish	P=0.0006, T=2.76	FALSE
Paired T-test	Lau vs Pee	Total # of Butterflyfish	P=0.014, T=2.48	FALSE
Paired T-test	Lau vs Pee	Total # of Wrasses	P=0.049, T =-1.98	FALSE
Paired T-test	Lau vs Pee	Total # of Hawk fish	P<0.0001, T=5.36	FALSE
Mann-Whitney	Lau vs Pee	Total # of targeted vs non	P<0.0001, W=122335.5	FALSE

Table 9. Averages of Body Complexity Index (BCI), stable isotopes Nitrogen (15N) and Carbon (13C), and lipid extraction tissues of ʻāholehole harvested from Laupāhoehoe, Kaʻawaliʻi, Pepeʻekeo, and Kolekole.

Location	Variable	Description	Average
Laupāhoehoe	Stable Isotope	15N	9.973
Laupāhoehoe	Stable Isotope	13C	-17.110
Laupāhoehoe	Stable Isotope	15N lipid	7.991
Laupāhoehoe	Stable Isotope	13C lipid	-18.217
Laupāhoehoe	ʻāholehole BCI	Total	0.003
Pepe'ekeo	Stable Isotope	15N	9.703
Pepe'ekeo	Stable Isotope	13C	-17.254
Pepe'ekeo	Stable Isotope	15N lipid	7.660
Pepe'ekeo	Stable Isotope	13C lipid	-17.976
Pepe'ekeo	āholehole BCI	Total	0.129

Discussion

The overall findings from this study support the hypothesis that global climate change may have an effect on nearshore marine fishery assemblages. At the Laupāhoehoe and Pepeʻekeo study locations located at extremes of the Hilo Palikū precipitation gradient, there were significant differences in community fish assemblages and notable variations in abundances for surgeon fish, damsel fish, and wrasses. The fish population data highlight that different fish communities exist across this gradient and thus may be affected by fresh water inputs. The difference in fish populations at the extremes of the gradient also coincide with differences in benthic habitat, as there were significant differences in percent cover for coral, CCA, and turf algae. The gut content data did not show differences in food web dynamics. however sample sizes were likely too small to detect a difference. Other studies suggest that terrestrial inputs may be a minor contributing factor for fish diet (Van den Meersche et al. 2009). The results for salinity levels were significantly different amongst both study sites, but did not follow the expected trend. Laupāhoehoe salinity levels were less than the levels at Pepeʻekeo. These trends may be explained by unidentified sources of fresh ground water discharge that may have skewed salinity readings, as well by the large time frame during which samples were gathered. The relationship amongst salinity and stream flow, and salinity and tide was used to normalize for other factors that may have had an impact. The findings from these comparisons show that relationships amongst the three variables were weak.

Fish Resources

Throughout the Main Hawaiian Islands targeted fish species include jacks, goat fishes, parrot fish, and soldier fish (Nado et al. 2015).   These fishes represented a small portion of species observed along the nearshore at both of the study sites. Studies that analyzed population abundances have shown that there are significant differences between targeted and non-targeted fish species (Friedlander et al. 2006). This is likely due to limited fishing regulations and

policing throughout the state, especially at rural sites like Laupāhoehoe and Pepeʻekeo. The visual transects performed exhibited a biomass three to four times lower compared to unfished populations in the Northwestern Hawaiian Islands (Nado et al. 2015). While the sample sizes in this study are too small, years of anecdotal observation at both these sites have shown considerable decreases in targeted fish species. Although access to these sites are challenging with rough terrain marked by a predominantly cliff coastline, as well as adverse weather conditions, the isolation of both sites make them more difficult to monitor and enforce rules. This makes these sites susceptible to exploitation by commercial fishers that have exhausted the resources of areas that are relatively easy to access.

The significant differences in juvenile wrasses and hawk fish between sites highlight differences in juvenile fish populations at the extreme ends of the precipitation gradient. This suggests that variability in salinity, temperature, and other parameters affected by freshwater inputs may affect juvenile fish population abundances. Other studies have also found that nearshore temperature and salinity levels, which are subject to fresh water introductions (Hoover and Mackenzie 2007), are a major determining factor of juvenile fish presence, abundance, and growth (Ramos et al. 2006).

There were significant differences between size classes 0-10 cm and 10-20 cm for total fish (Figure 20), and for wrasses (Figure 23). At both Laupāhoehoe and Pepeʻekeo fish populations dropped by almost half from size classes 0-10 cm to 10-20 cm. The differences demonstrate juvenile fish susceptibility to predation, competition, and the harsh oceanic conditions. There is evidence that demonstrate greater fish abundances with greater reef rugosity, which act as refuges from predation, as well as an inverse relationship between prey fish and resident piscivores (Hixon and Beets 1993). There is a similar pattern specifically for damsel fish and hawk fish, where populations decline when transitioning from a juvenile into an adult (Figure 21 and 24). This pattern is typical of most fish populations, as larval fish are susceptible to the many threats as listed above. This pattern was observed across all fish species. This again may be due to predatory pressure or limited habitat as demonstrated by the coral data (Table 5, Figure 15).

Diversity of fish species at Laupāhoehoe was greater than Pepeʻekeo. This difference in diversity may be attributed to abiotic factors such as salinity and nutrient loads, as well as reef structure, which can play a role in determining food availability and shelter type. In Hawaiʻi, there are 18 families of fish species that utilize sandy areas that surround reefs as feeding grounds. In Hanalei Bay fishes routinely move between sandy areas, reef edges, and surf zones, which help to demonstrate the complex interaction between fishes and their habitat (DeFelice and Parrish 2003). However, these differences may also be due to biotic features namely coral cover and diversity (Friedlander & Parrish 1998). There was a greater amount of coral cover and diversity, thus habitat and food availability, at Laupāhoehoe, which may be why there was higher fish species diversity at that site. This suggests that drier conditions will have an impact on not only fish community assemblages, but will also affect overall biodiversity of the nearshore habitat.

Salinity

The salinity results show decreasing trends toward the northern end of the gradient, contrary to predictions, but may be explained by the short time period for sampling, unaccounted plumes of ground water discharge, and/or variability in stream discharge over time. Due to a lengthy sampling period, the salinity data was normalized with stream flow and tidal height data. Although there were apparent correlations between salinity and streamflow as well as tide, the variations in salinity were not significant (stream flow and tidal height; F > 0.05, P > 0.05, Figure 7 - 12). However, the salinity levels at both study sites were significant at depth ranges 0.1 – 9 m (P = 0.0008, Table 2), specifically at 0.1 – 3 m (P = 0.0085, Table 2, Figure 6). These results do not support the hypothesis that Laupāhoehoe (μ = 26.181 ppt) salinity levels are higher compared to Pepeʻekeo (μ= 30.516; Table 2).

Corals

Corals are sensitive to extreme fluctuations in salinity, excess nutrients, and sedimentation (Harvell et al. 1999; Ogston and Field 2010). Most corals have a salinity tolerance range between 28.7 and 40.4 ppt. Deviations from this range stress corals, and if prolonged may result in mortality (Guan 2015). In this study, salinity levels increased with depth at both Laupāhoehoe and Pepeʻekeo (Table 1, Figure 6) indicating that there is relatively less impact from freshwater to marine life with increased depth. At Laupāhoehoe, the coral percent cover increases with depth, which corresponds to increases in salinity (Table 1, Figure 6, Table 6). However, at Pepeʻekeo, the coral percent cover does not demonstrate this pattern. This may be due to differences in sedimentation in the water column and nutrient loads. Anecdotally, Pepeʻekeo did have more sediments in the water column that increase light attenuation to inhibit photosynthetic production of corals. This may explain why salinity levels are not as strongly related to coral percent cover as demonstrated at Laupāhoehoe.

Excess nutrient loads fuel algal growth, which compete for substrate space across the ocean floor (Smith et al. 2001). Suspended sediments in the water column reduce light availability and eventually settle on corals to suffocate and impede photosynthetic rates that slow coral growth (Harvell et al. 1999; Ogston and Field 2010). Across the precipitation gradient, between Laupāhoehoe and Pepeʻekeo, there was a significant difference in coral cover (P < 0.001, Table 5, Figure 15). Although the differences in rainfall across the gradient were not represented in the salinity data (Table 2, Figure 6), coral cover at Laupāhoehoe was greater (31.5 %, Figure 13) than Pepeʻekeo (2.7, Figure 14). This reflects the hypothesized outcomes, and demonstrates trends in coral cover loss with increases in fresh water subsidies. Corals also directly compete with turf algae for space on the ocean floor (Ogston and Field 2010). This competition may be a major contributing factor for differences in substrate composition. As turf and macro algae compete with coral and CCA, this may be linked to the higher sediment and possibly nutrient inputs along the wetter end of the gradient. Coral species diversity was comparatively lower at Pepeʻekeo, which may be due to a more algal dominated habitat (Friedlander et al. 2006). The lack of coral cover at Pepeʻekeo (2.7 % Figure 14) may be explained by turf algae competition. Turf algae are the most dominant substrate type at both study sites; the benthic substrate at Laupāhoehoe is 51.8 % turf (Figure 13) and 68.4 % at Pepeʻekeo (Figure 14). There was a significant difference for turf algae percent cover between

Laupāhoehoe and Pepeʻekeo (Table 5, Figure 15). The difference in turf algae percent cover between sites is 16.6 %, which corresponds to a 28.8% difference in coral cover; this is nearly twice as much. Higher coral and CCA abundances may be attributed to differences in sediment and nutrient loads as well as reduced algal competition.

Herbivorous fish such as damselfish, surgeonfish, and butterflyfish consume algae and relieve competitive pressure on corals to influence benthic composition. At Laupāhoehoe, there was a significantly higher abundance of herbivorous fish (P < 0.001, Figure 18). Herbivory is a key process on coral reefs that can assist reef-building corals. In the Caribbean, 80 percent of herbivores are parrotfish. Herbivores remove the algae in constant competition with corals. When corals face any kind of disturbance (e.g. bleaching, disease, hurricanes) algae quickly colonize new availably space. Today we see increasingly more reports that indicate a phase shift from coral-dominated to algal-dominated reefs (Barnosky et al. 2012).

Targeted fish species, such as parrot and flag tail fish consume algae to decrease competition for coral populations. This study demonstrates the effects of fishing pressure on these fish species; at Laupāhoehoe there were no parrot fish and only two flag tail fish, at Pepeʻekeo there was four parrot fish and no flag tail fish (Table 10, Table 11, Figure 18). The loss of these species has influences upon the benthic habitat, unfortunately there have been no prior studies conducted to document changes to both the fish community and benthic substrate. This study can serve as a baseline study for future work to document shifts over time to both fish and coral communities across the Hilo Palikū gradient.

Crustose Coralline Algae (CCA) facilitates coral settlement, via chemical cues and physical recognition, as well as help in the post-settlement process. CCA increase coral survival, protecting corals from competition and sediments typically trapped in turf (Harrington et al. 2004). The greater percent cover of coral at Laupāhoehoe correspond comparatively greater percent cover of CCA. There percent cover of CCA is 13.1 % at Laupāhoehoe (Figure 13) compared to 0.7 % at Pepeʻekeo (Figure 14). Furthermore, the differences in CCA percent cover between sites are significant (P = 0.0148, Table 5, Figure 15). This demonstrates the function of CCA as an ally of corals, and help to explain the differences in substrate type between sites.

Future improvements to methodology

Sampling at additional sites along the gradient would be a major way to improve confidence in the results. In addition, future studies should involve the measure of shifting nutrient levels. Here, the impacts of terrestrial subsidies were captured primarily as changes in salinity. This was demonstrated with two approaches, with the measure of differences amongst sites along the gradient as well as variations ascribed to seasonal shifts within a site (Table 1, figure 6). As mentioned earlier, there were noted differences along the precipitation gradient but there may be larger differences between seasons, as illustrated in comparisons of seasonal differences between and at each site (Table 1).

To improve the methodology and better measure fresh water impacts, a future protocol would preserve aspects from my salinity measurement methodology but would take into account potential ground water plumes and suspended sediments. The framework that would persist is

the use of depth ranges and seasonal salinity measures. To amend the protocol, I would replace each transect perpendicular to the coastline. The main point of focus for the reorientation focuses on an accurate representation of fresh water influences at main water ways such as rivers, waterfalls, and ephemeral gulches. Another aspect of this protocol would be to identify subterranean ground water discharge sources. Finally, measuring salinity levels before and after storm events would be of great importance as this would better demonstrate the impact of fresh water on nutrient and sediment loads.

The other methodological amendment that would be incorporated to improve the study would be the measure of nutrient and sediment loads. Both of these factors are influenced by increases in the water budget and intensified storm events. The several ways that these would be measured include the documentation of changes in light attenuation and nutrient levels. To measure light availability, I would recommend use of a secchii disk. From a glance, Laupāhoehoe receives far less terrestrial subsidies (Figure 37 and 38) when compared to Pepe‘ekeo (Figure 39 and 40).

To more accurately measure fish population abundances, I would elect to perform timed fish surveys in conjunction with the fish visual transects. The time visual surveys would be done free diving at depths between 0 and 6 m with the help of host community organizations. The format of the timed survey should be a five minute period with the use of a GPS unit to mark the both the beginning and end of each transect. This would provide both the distance and location of each transect. Transects should be performed parallel to the coast and may, depending on the depth, be done parallel and simultaneous with other transects. The known distance and time of each transect will be calculated so that the methodology may be replicated.

To better study the differences in food web dynamics across the gradient, I would increase the sample size of fish from all four sites (Ka‘awali‘i, Laupāhoehoe, Kolekole, and Pepe‘ekeo). The results from the gut content analysis of the ‘āholehole did showed significant differences across the gradient, thus obtaining more samples for a robust data set may better demonstrate variations in diet. With an increased sample size, other parameters such as gamete weight, Body Complexity Index (BCI) as well as stable isotope comparisons may also be reported confidently. This may also highlight differences in reproductive cycles of ‘āholehole across the precipitation gradient, and may be indicative of variable reproductive cycles of other fish species across the gradient.

Conclusion

Climate change is likely to have an impact on nearshore fishery assemblages, but may not play as important of a role as fishing pressure. This supports the idea that management actions may have a greater and more direct impact on the health of nearshore fishery communities. This study, as being the first of its kind along the Hilo Palikū coast, may serve as a baseline for community organizations and the State Department of Land and Natural Resources to create a management plan for these fisheries. Although difficult and hazardous to access, these fisheries still show signs of significant fishing pressure that if unregulated, will result in the continued loss of fish stocks.

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Appendix

Table 10. Number of fish observed at Laupāhoehoe by species and fish group.

Location	Fish Group	Fish Species	# of individuals
Laupāhoehoe	Butterflyfishes	C. lunula	1
Laupāhoehoe	Butterflyfishes	*C. multicinctus*	14
Laupāhoehoe	Butterflyfishes	*C. ornatissimus*	27
Laupāhoehoe	Butterflyfishes	*C. quadrimaculatus*	16
Laupāhoehoe	Butterflyfishes	*F. longirostris*	18
Laupāhoehoe	Butterflyfishes	*F. flavissimus*	1
Laupāhoehoe	Butterflyfishes	*Z. Cornitus*	10
Laupāhoehoe	Butterflyfishes	*C. fremblii*	1
Laupāhoehoe	Butterflyfishes	*C. potteri*	2
Laupāhoehoe	Porcupine/Box/Puffer	*C. jactator*	3
Laupāhoehoe	Porcupine/Box/Puffer	*O. meleagris*	12
Laupāhoehoe	Surgeon fish	*A. dussumieri*	8
Laupāhoehoe	Surgeon fish	*A. nigrofuscus*	112
Laupāhoehoe	Surgeon fish	*A. leucopareius*	187
Laupāhoehoe	Surgeon fish	*A. nigricans*	20
Laupāhoehoe	Surgeon fish	*A. nigroris*	57
Laupāhoehoe	Surgeon fish	*A. olivaceus*	1
Laupāhoehoe	Surgeon fish	*A. triostegus*	12
Laupāhoehoe	Surgeon fish	*C. hawaiiensis*	10
Laupāhoehoe	Surgeon fish	*C. strigosus*	7
Laupāhoehoe	Surgeon fish	*N. lituratus*	1
Laupāhoehoe	Damselfishes	*A. sordidus*	9
Laupāhoehoe	Damselfishes	*A. abdominalis*	10
Laupāhoehoe	Damselfishes	*C. verator*	58
Laupāhoehoe	Damselfishes	*C. hanui*	19
Laupāhoehoe	Damselfishes	*C. agilis*	88
Laupāhoehoe	Damselfishes	*C. ovalis*	150
Laupāhoehoe	Damselfishes	*C. vanderbilti*	89
Laupāhoehoe	Damselfishes	*P. imparipennis*	22
Laupāhoehoe	Damselfishes	*P. johnstonianus*	29
Laupāhoehoe	Damselfishes	*S. marginatus*	65

Laupāhoehoe	Goatfishes	*M. vanicolensis*	3
Laupāhoehoe	Goatfishes	*P. insularis*	12
Laupāhoehoe	Goatfishes	*P. multifasciatus*	9
Laupāhoehoe	Trigger/File fish	*S. bursa*	16
Laupāhoehoe	Trigger/File fish	*M. niger*	56
Laupāhoehoe	Trigger/File fish	*M. vidua*	5
Laupāhoehoe	Trigger/File fish	*C. dumerilii*	2
Laupāhoehoe	Snappers	*A. furca*	3
Laupāhoehoe	Snappers	*L. kasmira*	37
Laupāhoehoe	Snappers	*L. fulva*	1
Laupāhoehoe	Wrasses	*M. grandoculis*	2
Laupāhoehoe	Wrasses	*B. albotaeniatus*	9
Laupāhoehoe	Wrasses	*C. gaimard*	15
Laupāhoehoe	Wrasses	*L. pthirophagus*	3
Laupāhoehoe	Wrasses	*T. ballieui*	16
Laupāhoehoe	Wrasses	*T. duperrey*	118
Laupāhoehoe	Wrasses	*H. ornatissimus*	45
Laupāhoehoe	Wrasses	*O. unifasiatus*	2
Laupāhoehoe	Wrasses	*O. octotaenia*	2
Laupāhoehoe	Wrasses	*G. varius*	10
Laupāhoehoe	Hawkfishes	*C. pinnulatus*	8
Laupāhoehoe	Hawkfishes	*C. fasciatus*	5
Laupāhoehoe	Hawkfishes	*P. arcatus*	126
Laupāhoehoe	Hawkfishes	*P. forsteri*	9
Laupāhoehoe	Blenny	*E. brevis*	6
Laupāhoehoe	Squirrelfish/Bigeyes	*M. kuntee*	2
Laupāhoehoe	Squirrelfish/Bigeyes	*M. berndti*	6
Laupāhoehoe	Squirrelfish/Bigeyes	*Nioniphon spp*	3
Laupāhoehoe	Jacks	*C. melampygus*	3
Laupāhoehoe	Flagtail	*K. xenura*	2
Laupāhoehoe	Other	*A. chinensis*	1
Laupāhoehoe	Other	*C. argus*	1
Laupāhoehoe	Other	*Kyphosus spp.*	18
Laupāhoehoe	Other	*S. cacopsis*	2

Table 11. Number of fish observed at Pepeʻekeo by species and fish group.

Location	Fish Group	Fish Species	# of individuals
Pepe'ekeo	Butterflyfishes	*C. lunula*	4
Pepe'ekeo	Butterflyfishes	*C. multicinctus*	7
Pepe'ekeo	Butterflyfishes	*C. ornatissimus*	4
Pepe'ekeo	Butterflyfishes	*C. quadrimaculatus*	14
Pepe'ekeo	Butterflyfishes	*F. longirostris*	10
Pepe'ekeo	Butterflyfishes	*C. auriga*	4
Pepe'ekeo	Porcupine/Box/Puffer	*D. hystrix*	1
Pepe'ekeo	Porcupine/Box/Puffer	*O. meleagris*	1
Pepe'ekeo	Surgeon fish	*A. blochii*	1
Pepe'ekeo	Surgeon fish	*A. nigrofuscus*	127
Pepe'ekeo	Surgeon fish	*A. leucopareius*	58
Pepe'ekeo	Surgeon fish	*A. nigroris*	3
Pepe'ekeo	Surgeon fish	*A. olivaceus*	2
Pepe'ekeo	Surgeon fish	*A. triostegus*	51
Pepe'ekeo	Surgeon fish	*C. strigosus*	2
Pepe'ekeo	Damselfishes	*A. sordidus*	2
Pepe'ekeo	Damselfishes	*A. abdominalis*	40
Pepe'ekeo	Damselfishes	*A. vaigiensis*	26
Pepe'ekeo	Damselfishes	*C. verator*	4
Pepe'ekeo	Damselfishes	*C. hanui*	62
Pepe'ekeo	Damselfishes	*C. agilis*	
Pepe'ekeo	Damselfishes	*C. ovalis*	58
Pepe'ekeo	Damselfishes	*C. vanderbilti*	178
Pepe'ekeo	Damselfishes	*P. imparipennis*	24
Pepe'ekeo	Damselfishes	*P. johnstonianus*	57
Pepe'ekeo	Damselfishes	*S. marginatus*	26
Pepe'ekeo	Goatfishes	*M. flavolineatus*	7
Pepe'ekeo	Goatfishes	*P. insularis*	4
Pepe'ekeo	Goatfishes	*P. multifasciatus*	11
Pepe'ekeo	Goatfishes	*P. porphyreus*	7
Pepe'ekeo	Goatfishes	*P. pleurostigma*	5
Pepe'ekeo	Trigger/File fish	*S. bursa*	3
Pepe'ekeo	Trigger/File fish	*R. rectangulus*	7

Pepe'ekeo	Trigger/File fish	*R. aculeatus*	1
Pepe'ekeo	Trigger/File fish	*C. dumerilii*	6
Pepe'ekeo	Snappers	*L. kasmira*	5
Pepe'ekeo	Wrasses	*B. albotaeniatus*	21
Pepe'ekeo	Wrasses	*C. gaimard*	20
Pepe'ekeo	Wrasses	*L. pthirophagus*	4
Pepe'ekeo	Wrasses	*N. taeniourus*	3
Pepe'ekeo	Wrasses	*T. ballieui*	14
Pepe'ekeo	Wrasses	*S. balteata*	23
Pepe'ekeo	Wrasses	*T. duperrey*	113
Pepe'ekeo	Wrasses	*T. trilobatum*	46
Pepe'ekeo	Wrasses	*H. ornatissimus*	92
Pepe'ekeo	Wrasses	*G. varius*	10
Pepe'ekeo	Wrasses	*M. geoffroy*	1
Pepe'ekeo	Wrasses	*C. venusta*	4
Pepe'ekeo	Hawkfishes	*C. pinnulatus*	3
Pepe'ekeo	Hawkfishes	*C. fasciatus*	3
Pepe'ekeo	Hawkfishes	*P. arcatus*	26
Pepe'ekeo	Hawkfishes	*P. forsteri*	1
Pepe'ekeo	Squirrelfish/Bigeyes	*M. kuntee*	3
Pepe'ekeo	Squirrelfish/Bigeyes	*P. meeki*	5
Pepe'ekeo	Squirrelfish/Bigeyes	*Nioniphon spp*	1
Pepe'ekeo	Parrotfishes	*C. carolinus*	4
Pepe'ekeo	Other	*P. kallopterus*	4
Pepe'ekeo	Other	*A. chinensis*	1
Pepe'ekeo	Other	*C. argus*	7
Pepe'ekeo	Other	*Synodus spp.*	4
Pepe'ekeo	Other	*S. cacopsis*	1