

Modeling Potential Shifts in Hawaiian Anchialine Pool Habitat and Introduced Fish Distribution due to Sea Level Rise

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Abstract Global mean sea levels may rise between 0.75 and 1.9 m by 2100 changing the distribution and community structure of coastal ecosystems due to flooding, erosion, and salt-water intrusion. Although habitats will be inundated, ecosystems have the potential to shift inland, and endemic species may persist if conditions are favorable. Predictions of ecosystem migration due to sea level rise need to account for current stressors, which may reduce the resilience of these ecosystems. This study predicts the potential consequences of sea level rise on the groundwater-fed anchialine pool ecosystem in Hawaii. Scenarios of marine and groundwater inundation were compared with current patterns of habitat, introduced fishes, and land use. Results show that current habitats containing endemic anchialine shrimp will be increasingly inundated by marine waters. New habitats will emerge in areas that are low lying and undeveloped. Because of subsurface hydrologic connectivity, endemic shrimp are likely to populate these new habitats by moving through the coastal aquifer. In some areas, rising sea levels will provide surface connectivity between pools currently containing introduced fishes (tilapia, poeciliids) and up to 46 % of new or existing pools that do not contain these fish. Results predicting future habitat distribution and condition due to sea level rise will support conservation planning. Additionally, the interdisciplinary approach may provide guidance for efforts in other coastal aquatic ecosystems.

Keywords Groundwater inundation · Brackish ecosystems · LiDAR · Nonnative fishes · Sea level rise · Atyid shrimp

Introduction

Global mean sea level may rise between 0.75 and 1.9 m by 2100 (Vermeer and Rahmstorf 2009; Parris et al. 2012). Coastal ecosystem distribution and community structure are expected to change significantly due to flooding, erosion, salt-water intrusion, or a combination of these phenomena (Nicholls and Cazenave 2010; IPCC 2013; Williams 2013). Coastal ecosystems may shift inland if open space is available and conditions are suitable (Woodroffe 1990; Kirwan and Magonigal 2013). Ideally, future potential habitat sites can be targeted for protection and incorporated into restoration efforts so that vulnerable ecosystems and the species associated with them will persist (Hannah et al. 2007; Stralberg et al. 2011).

Over the last century, multiple stressors including invasive species, elevated nutrient loads, and destructive land use practices have caused widespread coastal aquatic ecosystem degradation (Vitousek et al. 1997; Paine et al. 1998; Foley et al. 2005; Halpern et al. 2008). These current stressors may reduce the resilience of ecosystems to withstand climate change (Folke et al. 2004; IPCC 2013). Therefore, predictions of habitat and species shifts across the landscape due to a changing climate may be most useful if they incorporate stressors such as introduced species or human impacts (Brook et al. 2008; Lee et al. 2008; Pereira et al. 2010; Oliver and Morecroft 2014). In coastal areas, identifying where humans will restrict habitat transgression represents a simple and crucial step toward understanding whether these valuable ecosystems will expand or contract with sea level rise (Kirwan and Magonigal 2013). For example, spatial analysis of sea level rise impacts

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on tidal wetland habitat in San Francisco Bay revealed that relatively little area is naturally available to accommodate future marshes but that removal of levees and other barriers to tidal action would more than double potential new habitat (Stralberg et al. 2011). Landscape-scale analysis of coastal ecosystem vulnerability and shifts due to sea level change provides direction for restoration actions, cross-boundary management planning, decisions on future infrastructure development, and ecosystem protection. The unique anchialine ecosystem has had little attention in this regard.

Anchialine habitats are tidally influenced brackish water bodies without overland connection to the ocean (Holthuis 1973). Anchialine pools or caves are common in tropical coastal areas, such as the Indo-Pacific (Holthuis 1973; Maciolek 1983; Webb 2010), the Yucatan (Sanchez et al. 2002), and the Caribbean (Moritsch et al. 2014), where porous substrates, such as karst or lava, provide good hydrologic connectivity between the groundwater and ocean. At least 450 new species of anchialine organisms have been discovered and described in the past three decades highlighting the importance of this ecosystem to biodiversity (Iliffe and Kornicker 2009; Weese 2012). In Hawaii, anchialine pools support diverse endemic biota (Maciolek and Brock 1974), including seven species listed as candidate threatened or endangered species (USFWS 2015). Two of the most common species are the aytid shrimp *Halocaridina rubra* Holthuis 1963, and *Metabetaeus lohena* Banner and Banner 1960, which are located across the region in a wide range of salinities (2–25 parts per thousand (ppt); Maciolek and Brock 1974; Brock and Kam 1997; Marrack et al. 2015). *H. rubra* is the dominant grazer in the system eating benthic biofilms (Brock and Kam 1997). *Metabetaeus lohena* is a candidate endangered shrimp that preys on *H. rubra* and other invertebrates (Holthuis 1973). Similar to anchialine species elsewhere, adults and larvae of both *H. rubra* and *M. lohena* may disperse through subterranean groundwater (Kano and Kase 2004; Craft et al. 2008). Other endemic shrimp as well as gastropods are occasionally seen in pools, with some rare species found in highly restricted distributions (Maciolek and Brock 1974; Chai et al. 1989; Sakihara 2012).

Land use practices, introduced fishes, and sea level rise threaten anchialine habitats and associated endemic species. Development in coastal areas can negatively affect anchialine habitats due to infilling (Brock et al. 1987; Iliffe and Kornicker 2009) and groundwater pollution (Iliffe and Kornicker 2009). In Hawaii, nonnative fishes (*Poecilia reticulata*, *Gambusia affinis*, *Tilapia* sp.) prey on the dominant grazing shrimp *H. rubra* (Chai et al. 1989; Capps et al. 2009; Carey et al. 2011; Sakihara 2012) and are implicated as a factor leading to habitat degradation (Brock and Kam 1997). Preliminary analyses indicate that sea level rise could inundate some current pool habitats (Marrack and O'Grady 2014). Additionally, increased water levels may allow introduced

fishes to disperse into new areas, and current coastal development may restrict new habitat formation. To adequately predict the effect of sea level rise on anchialine habitats, current land use and introduced fish distribution should be examined along with future conditions.

Predictions of inundation due to sea level rise along any specific coastline carry uncertainty due to a number of global, regional, and local mechanisms (IPCC 2013). The rate and magnitude of ice sheet loss, particularly in Greenland and West Antarctica, are a major source of uncertainty for global sea level rise predictions (Vermeer and Rahmstorf 2009; NRC 2012; IPCC 2013). At the regional and local level, sea level rise will be affected by vertical land movement including subsidence or uplift. Maximum water levels will be a result of ocean dynamics including tides, storms, and hurricanes (Tebaldi et al. 2012; Merrifield et al. 2013; Ruggiero 2013), and basin-wide processes such as the Pacific Decadal Oscillation and ENSO (IPCC 2013; Scafetta 2013). Due to these complex interactions, scenarios of sea level rise are a useful method for illustrating future conditions, because they do not assume a particular timeframe and may incorporate various uncertainties (NRC 2012).

For this study, sea level rise was mapped in 0.5-m increments (0, 0.5, 1, and 1.5 m). Dates when these heights are projected to occur ultimately reflect future global carbon emissions (IPCC 2013). For example, based on the Vermeer and Rahmstorf (2009) global model, mean sea levels could rise between 0.3 and 0.53 m by 2050 and may reach 0.75 to 1.9 m by 2100. Regional models for the Central Pacific forecast an up to 1.5-m sea level rise by 2100 (Spada et al. 2013). On the island of Hawaii, tide gauges at Kawaihae and Hilo have recorded an average relative sea level rise of 3.5 mm/year (Vitousek et al. 2009). These data fit the measured global averages of sea level rise (Church and White 2011). However, satellite altimetry data indicate that the global acceleration of sea level rise due to thermal expansion and melting ice has not reached Hawaii and that the local long-term trend has been approximately 1.5 mm/year (Meyssignac and Cazenave 2012). Based on this information, the difference between local tide gauge measurements (3.5 mm/year) and the satellite altimetry measurements is most likely due to island subsidence rates. Subsidence for the island of Hawaii is estimated to be an average of 2.6 mm/year due to loading of the lithosphere by Kilauea volcano (Moore and Clague 1992; Zhong and Watts 2002). Although it is unclear if local subsidence rates and regional oceanographic processes will remain constant, sea level in Hawaii will continue to rise and rates are expected to increase by the middle of the twenty-first century (Marra et al. 2012).

Typically, spatial models of future coastal inundation compare sea level change scenarios with high-resolution topographic data that is collected using light detection and ranging (LiDAR) technology (Gesch 2009). Recent work has

highlighted the importance of incorporating groundwater levels into coastal inundation models (Bjerklie et al. 2012; Rotzoll and Fletcher 2013; Marrack 2014; Cooper et al. 2015). From an ecological perspective, groundwater can play an important role in the location and community composition of coastal ecosystems, especially those located slightly inland from shore (Doyle et al. 2010; Sakihara et al. 2015). Furthermore, in porous, unconfined coastal aquifers, the groundwater table is elevated above mean sea level sloping up and away from the shoreline and typically moves with the ocean surface (Oki 1999; Bjerklie et al. 2012). As a result, spatial models that do not include groundwater levels may underestimate coastal flooding (Bjerklie et al. 2012; Rotzoll and Fletcher 2013). On the west coast of Hawaii Island, detection of known anchialine pool locations was improved by 37 % when observed groundwater levels were included with maps of current marine flooding (Marrack 2014). Mapping of future sea level rise effects on groundwater-dependant coastal ecosystems should incorporate groundwater levels when possible.

Studies examining ecosystem response to sea level rise have highlighted coastal wetlands (Stralberg et al. 2011; Reynolds et al. 2012; Kirwan and Magonigal 2013; Kane et al. 2014), tidal forests (Doyle et al. 2010), and species like turtles or elephant seals that rely on beaches for reproduction (Fuentes et al. 2010; Funayama et al. 2013). This study is the first, to my knowledge, that predicts the effects of future sea level rise on anchialine pool habitats. Geospatial models included current habitat and species distributions, high-resolution elevation data (1 m), relative sea level rise scenarios, and local measurements of groundwater heights above mean sea level. The specific questions asked were as follows: (1) What is the current distribution of endemic anchialine species and introduced fishes within anchialine pools at the study sites? (2) What is the potential extent of pool inundation and the probable extent of future pool habitat within different land use types under various sea level rise scenarios? and (3) What is the potential extent to which elevated water levels will aid in the dispersal of introduced fishes from current locations to future pool habitats under various sea level rise scenarios? Results provide predictions of future habitat in both protected and developed coastal areas. Results also highlight areas where current introduced fish locations may pose a high risk to future habitat. Outcomes may be useful for conservation planning aimed at protecting this unique ecosystem under future conditions.

Methods

Study Site

The Hawaiian archipelago is the only place in the USA, other than the territory of Puerto Rico, where anchialine habitats are

found, and the study region represents one of the highest concentrations of pool habitats in the Hawaiian island chain (Brock and Kam 1997). Pools are considered surface expressions of groundwater, exhibit tidal fluctuations, and may range in salinity from 0.5 to 30 ppt (Maciolek and Brock 1974). Pool habitats may occur in bare lava substrate with no associated vegetation or on older lava flows surrounded by trees or wetland vegetation (Brock and Kam 1997). Over time, pools may become filled in with new lava flows or older sediments which would explain their sparse distribution or absence on older Hawaiian Islands (Maciolek and Brock 1974). Annual rainfall in the study area ranges from 25 to 50 cm per year (Giambelluca et al. 2013). In this study, six sites representing two National Parks, two state conservation areas, and two resort development lands were compared (Table 1 and Fig. 1).

Anchialine Pool Surveys

Data describing the habitat and faunal characteristics of anchialine pools were collected at each study site during daylight hours from July 12 to August 10, 2012 and July 2 to July 15, 2013. Methods were based on previous surveys conducted by the National Park Service (NPS) between 2007 and 2009 (Jones et al. 2011; NPS 2012) and Marrack et al. (2015). Data reported in this study includes pool location, surface area, distance from the shoreline, salinity, and the occurrence of aquatic animal taxa. For pool complexes that are connected at high tides, data was collected for the complex as a whole. Therefore, pool numbers reported for habitat surveys are lower than the overall number of pools used for the inundation models. In the case of the Makalawena site (MAKA), additional pools were identified on satellite imagery after the habitat surveys were completed. These pool locations were verified and recorded with a Global Positioning System (GPS) in March, 2014 and are used in the inundation models but are not included in habitat characteristic summaries.

Pool surface area was determined within ESRI's ArcGIS 10.0 using a polygon of the pool perimeter collected in the field with a Trimble GeoXH GPS (horizontal accuracy was less than 10 cm). Surface area for small pools or those with high canopy cover was calculated as pool length \times width measured with a transect tape at the high water line. Distance from coastal shoreline was calculated in ArcGIS as the Euclidian distance between the shoreline and the closest edge of each pool. Salinity was collected at the surface of each pool using a portable sonde (YSI 6500 or Hydrolab Quanta).

Endemic pool species as well as introduced fishes were recorded as present or absent based on visual surveys during daylight hours. *Poecilia reticulata*, *Poecilia mexicana*, and *G. affinis* were present in pools but were not always distinguishable in the field so were combined as poeciliids for statistical analysis. Tilapia species were also grouped together for this study. *Oreochromis mossambicus* probably makes up the majority of

Table 1 Study sites with total number of pools, number of pools with fish, and total length of coastline

Study site	Site name	Land use	Number of identified pools	Length of coastline (km)
Waikoloa resorts	WAIK	Resort	12	1
Maunalani resorts	MAUNA	Resort	8	4.5
Anaehoomalu to Keawa Iki	ABKW	Undeveloped (state conservation and private lands)	80	6
Makalawena	MAKA	Undeveloped (state conservation and private lands)	75	4.2
Puuhonua O Honaunau National Historic Park	PUHO	National park	14	3.9
Kaloko Honokona National Historical Park	KAHO	National park and state conservation lands	216	4.1

Current pools do not include fishponds with direct access to the ocean, man-made, or modified pools in which reef fish or koi are maintained

tilapia found on the island of Hawaii, but this is not certain since other tilapia species exist on Oahu and Kauai, and they are difficult to identify as juveniles (Mackenzie and Bruland 2012). Most pools were small and shallow enough that it was possible to visually examine the entire water body from the pool edges. Larger pools, where the center and bottom were not visible from the surface, were examined using mask and snorkel. Pools were examined for 15 to 40 min depending on size.

While some pools were visited multiple times during the study, some pools were only visited once due to restricted access. Previous work has indicated that although pools may exhibit increases in salinity with depth (Holthuis 1973; Havird et al. 2014), the surface of each pool remains relatively constant on annual, seasonal, and daily timescales (Bienfang et al. 2011; NPS 2012; Marrack et al. 2015). Additionally, repeated daytime visits to a subset of 58 pools show that introduced fishes and the endemic anchialine shrimp *H. rubra* and *M. lohena* occurred consistently within pools (Marrack et al. 2015). Therefore, although single site visits are not ideal for documenting variability within a habitat, it is assumed that observations made at each pool are characteristic of typical conditions.

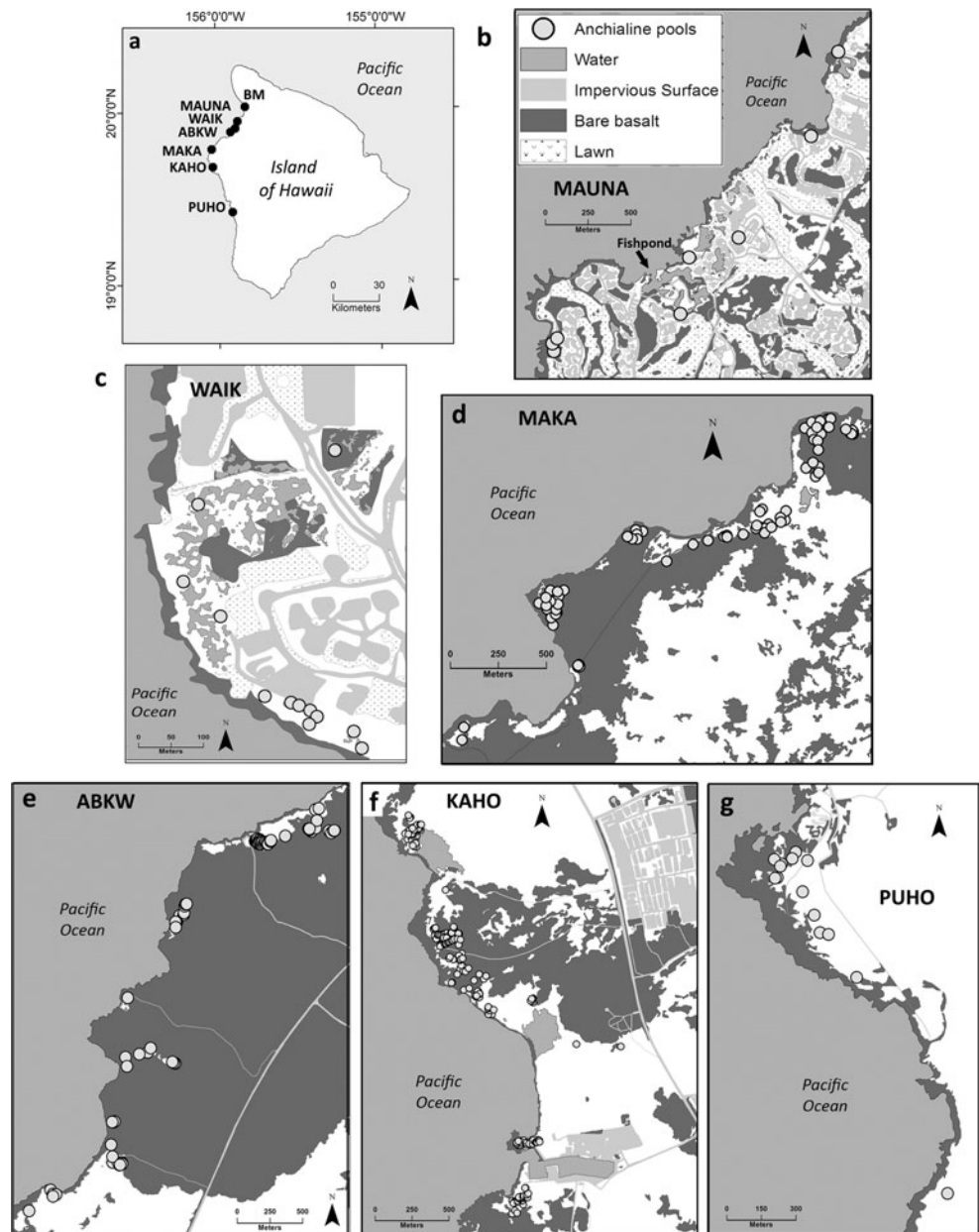
Two-factor contingency tables were used to determine whether fish occurrence was associated with the occurrence of *H. rubra* or *M. lohena*. Using data pooled from the six sites, frequency data were compiled for specific pool categories. For example, to examine the relationship between tilapia presence and *H. rubra* occurrence, counts were made of pools with tilapia and *H. rubra*; tilapia and no *H. rubra*; no tilapia and *H. rubra*; and no tilapia and no *H. rubra*. Frequencies were analyzed using a Pearson chi-square test with Yates correction (χ^2) and a significance level of $P \leq 0.05$ (Quinn and Keough 2009). The null hypothesis was that fish occurrence (tilapia or poeciliid) was independent of shrimp (*H. rubra* or *M. lohena*) occurrence.

To determine the current pattern of introduced fish distribution in pools, spatial autocorrelation was evaluated at each site using the Global Moran's I tool in ESRI's ArcGIS (10.1). The Global Moran's I tool evaluates whether features and associated attributes are clustered, dispersed, or randomly distributed across a landscape. The *P* value indicates statistical significance with the null hypothesis that features are randomly distributed across the study area. A significant *P* value and positive z-score indicate that the spatial distribution of high values and/or low values in the dataset is more spatially clustered than would be expected if underlying spatial processes were truly random. A positive Moran's I index value indicates tendency toward clustering whereas a negative Moran's I index value indicates tendency toward dispersion. The analysis typically is run with 30 or more features so Moran's index results for WAIK, MAUNA, and PUHO are included but not necessarily valid.

Modeling Methods

Sea level rise scenarios were combined with current pool locations, land use, and introduced fish distribution at the six sites to determine the combined effect of these factors on anchialine habitats under future sea level change. Under each scenario, the inundation of existing pools as well as the size, location, and number of new pools that will emerge on the landscape was mapped. To model future anchialine pool locations, gridded surfaces of sea level with and without groundwater heights were created using ESRI's ArcGIS 10.1. These water surfaces were then compared to 1-m-resolution topographic surfaces (Digital Elevation Models or DEMs) to determine where anchialine pools would be expected to occur. Areas where the water surface was higher than the topographic surface were classified as water bodies. Areas that became flooded in the inundation model but were not connected

Fig. 1 Location of sites on the island of Hawaii from north to south: Kawaihae tidal benchmark (BM), Maunalani Resort (MAUNA), Waikoloa Resort (WAIK), Anehoomalū Bay to Keawa Iki Bay (ABKW), Makalawena site (MAKA), Kaloko-Honokōhau National Historical Park (KAHO), and Puuhonua ō Hōnāunau National Historical Park (PUHO). Corresponding site maps (b–g) show locations of current anchialine pools and land cover/land use (NOAA 2005). Land cover/land use types include bare basalt, turf grass, and impenetrable surfaces such as roads and buildings. Unidentified land cover/land use (white) is a mix of undeveloped tree, shrub, and grass cover



overland to the ocean were considered potential new anchialine pools. Subsurface hydrologic connectivity between the coastal aquifer and the ocean was assumed to be high (Oki 1999; Bauer 2003).

To determine if sea level rise has the potential to aid in dispersal of introduced fishes from occupied to unoccupied habitats, I examined surface connectivity between pools under sea level rise scenarios at extreme tides. One assumption was that, unlike endemic anchialine shrimp, introduced fishes can only disperse over land through surface water connections between pools but may only need 3 mm of water depth to move (Alemadi and Jenkins 2008). The assumption that fish do not commonly disperse through subterranean groundwater in the Hawaiian anchialine ecosystem is supported by

personal observations of anchialine pools within the study area where unoccupied pools are found within 4 m of pools containing fish, and they have persisted this way for over 10 years. Another assumption was that the current suite of nonnative fishes are able to survive in a range of temperature and salinity environments, because they are generalists and found in a wide range of pool habitats (Maciolek and Brock 1974; Mackenzie and Bruland 2012).

Sea Level Scenarios

Sea level rise models included 0-, 0.5-, 1-, and 1.5-m relative sea level increases along with two tidal states. Tides were incorporated into sea level models because higher tides

increase coastal flooding and anchialine pools undergo tidal fluctuations. In Hawaii, tides are semi-diurnal with the highest recorded tide at 0.74 m above mean sea level (MSL) at the local tidal benchmark (Kawaihae–1617433B) in 1993 (NOAA 2012). Some anchialine pools become dry at low tides but are populated by endemic shrimp at higher tides; therefore, sea level scenarios used to determine new pool locations were created to reflect local mean higher high water (MHHW) or 0.374 m above MSL (1617433B; NOAA 2012). The value used for the extreme tide level in this study was 0.7 m above MSL, calculated as the mean of the six most extreme annual tides observed at the local tidal benchmark from 2001 to 2011 (NOAA 2012). This extreme tide value concurs with the time-averaged annual maxima water level empirically derived by Merrifield et al. (2013) for the same tide gauge ($66.4 \text{ cm} \pm 3.1$ (2 S.E.)).

Groundwater

The methods used to incorporate groundwater into sea level rise models were modified from previous studies (Rotzoll and Fletcher 2013; Marrack 2014). For this study, groundwater models were created at each site and sea level scenario (0, 0.5, 1, and 1.5 m) based on the unique shoreline predicted for that sea level. Theoretically, in a homogeneous unconfined coastal aquifer with constant recharge from an inland source, the groundwater level should be a function of the square root of distance from the coast (Glover 1959). On the west coast of Hawaii, the relationship between observed groundwater heights and distances from shore has been calculated as $h = 0.00989x^{1/2} + 0.16734$, where h is the average groundwater height in meters above MSL and x is the distance from shoreline in meters ($R = 0.93$; Marrack 2014, see Fig. 3a, b). For this study, the equation was incorporated into sea level rise models by creating a raster representing the Euclidean distance from shoreline (2-m resolution) using ArcGIS Spatial Analyst tools (ESRI 2011). A raster representing mean groundwater height above MSL was then calculated by applying the linear regression equation to each pixel.

A tidal efficiency component was then added to the groundwater raster to incorporate tidal effects on groundwater heights. Tidal effects decay with distance inland from shore and can be represented by the tidal efficiency ratio which is 1 at the shoreline decreasing to 0 inland where there is no tidal effect (Todd 1980). Tidal efficiency represents the ratio of mean daily groundwater tidal range at a site to mean daily ocean tidal range. For current sea level conditions within the study area, Marrack (2014) used observed ocean and groundwater tide heights to calculate $\log(y) = -0.00034x - 0.26646$, where y is the tidal efficiency and x is the distance from shoreline in meters ($R = -0.92$). For this study, raster surfaces representing tidal influence during MHHW and extreme tides

were combined with the groundwater surface over MSL raster to create the groundwater models.

To account for the potential decrease in future groundwater levels due to reduced rainfall (Giambelluca et al. 2013), another set of inundation maps were made without groundwater. Within this study, sea level rise scenarios that incorporate groundwater can be considered an upper bound to future flooding due to rising ocean levels whereas scenarios without groundwater are a lower bound.

Digital Elevation Models

Topographic data were derived from Federal Emergency Management Agency (FEMA) LiDAR data collected along the coast in 2006 for the Hawaiian Islands (Dewberry and Davis Corp 2007). LiDAR coverage includes the water line to the 15-m-elevation contour with an average point distance of 0.9 m for bare earth returns. Data were horizontally referenced to the North American Datum 1983 (NAD83) and were reported as horizontally accurate to 0.3 m with 68.2 % of laser returns. The North American Vertical Datum 1988 is specific to the Continental US and does not exist for Hawaii; therefore, the vertical datum was referenced to a local tidal datum with 0 m = mean sea level (MSL). For bare earth surfaces, the reported mean vertical error for the LiDAR points was 0.03 m and for all terrain types was 0.05 m (Dewberry and Davis Corp 2007).

Digital elevation models (DEMs) were created from the LiDAR bare earth returns using ESRI's ArcGIS. First, triangular irregular networks (TINs) were created from LiDAR point files. TINs were then converted to a raster format using linear natural neighbor interpolation. This interpolation technique constructs DEMs that retain original peak and valley elevation values which are useful for finding small isolated pits that exist in the lava surfaces (Maune 2007). Prior to subsequent analysis, DEMs were corrected by -0.25 m to account for vertical offsets in the LiDAR data detected during accuracy assessments that compared LiDAR DEM data to National Geodetic Survey benchmark orthometric heights (Marrack 2014).

Spatial Analysis

Within ArcGIS 10.1, each scenario was compared to the topographic surface (DEMs) of a site. Areas where the water surface was higher than the topographic surface were classified as water bodies and converted to polygons. Polygons within 1 m of each other were aggregated. Then, polygons were visually examined over Quickbird true color imagery (USDA 2008) to eliminate ocean surfaces and fishpond embayment features from the anchialine pool category. For each scenario, the inundation of current pools was calculated. A pool was considered inundated if it overlapped with an ocean or fishpond

surface. The number, surface area, and location of new pools created at each scenario were also calculated. New pools were defined as completely separated from other water bodies at MHHW, were at least 10 m from the shoreline, and did not overlap current pool surfaces. Also, inundation that occurred on unnatural surfaces such as paved areas or golf course was not considered a potential new pool habitat. All new pool calculations are relative to current land use conditions. Because endemic anchialine species disperse through groundwater, new pools are referred to as potential new habitat.

To examine potential dispersal of fish due to sea level rise, sea level scenarios were examined as time steps with current conditions as the first step, 0.5 m next, and so on. At each step, the extreme tide scenario was used to determine flooding extent. Flooding extent polygons that overlapped with pools containing introduced fish were assumed as potential dispersal routes to fishless pools. If pre-existing pools or newly created pools overlapped with the flood extent polygons, they were ranked as a potential fish pool, enumerated, and considered a source of fish for the next sea level step (Fig. 2). Ocean surfaces were not included in the flooding extent polygons because tilapia and poeciliids are not observed living in marine conditions in Hawaii and would therefore not be transported by marine flooding. Fishpond embayments at KAHO connect with narrow channels to the ocean but are lower in salinity than marine habitats and contain tilapia and poeciliids

(Mackenzie and Bruland 2012). These embayments were considered as potential sources of fish and were included in analysis. For simplicity, the fish dispersal analysis was only examined using sea level rise models without groundwater and therefore represents a conservative estimate of connectivity.

Results

Current Habitat Characteristics

Across all six sites, 405 pools were mapped with a GPS and 348 pools were surveyed for habitat characteristics. Salinities ranged from 1.3 to 25.6 ppt across all sites with the highest mean salinity of 14.5 ± 3.2 ppt at the KAHO site (Table 2). The lowest salinities were 1.3 ppt in pools from the ABKW site where the mean was 3.7 ± 1.1 ppt. Pool surface areas ranged from less than 1 to 7931 m² for one very large pool within the Waikoloa Pond Preserve. This large pool accounted for the high mean pool size for the WAIK site (1466 m²) compared to the mean pool size for other sites (14–124 m²). Pool distance to the ocean shoreline ranged from 10 to 600 m but averaged 40–135 m at all sites.

The most common endemic anchialine species observed during daylight surveys were *H. rubra* and the candidate endangered species *M. lohena*. At the ABKW site, *H. rubra* occurred in 76 % of 71 total pools whereas *M. lohena* occurred

Fig. 2 Potential dispersal of introduced fishes due to sea level rise and extreme tides. Time series shows **a** current pool and introduced fish distribution, **b** connectivity between fish occupied pool and unoccupied pools at 0.5-m sea level rise during an extreme tide, **c** potential distribution of pools and fish at 0.5 m sea level scenario at mean higher high tide after an extreme tide event

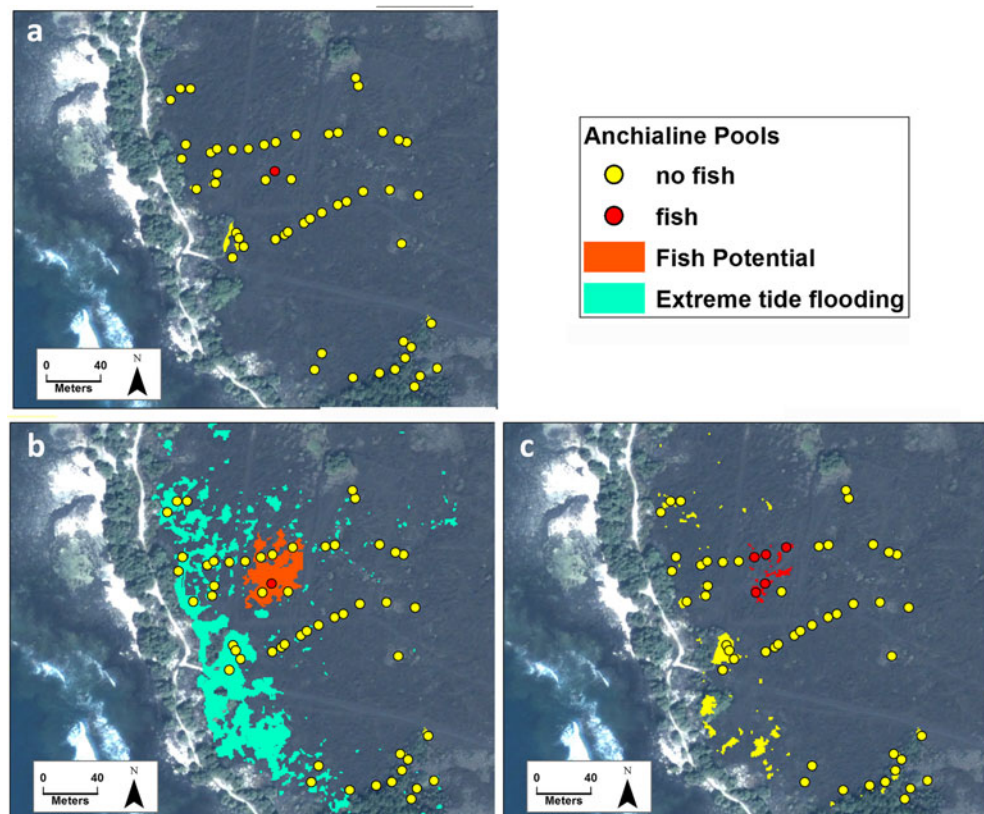


Table 2 Summary of habitat characteristic data collected at all sites

Study site	Salinity (ppt)	Surface area (m ²)	Distance to shoreline (m)	% with poeciliids	% with tilapia	% with <i>H. rubra</i>	% with <i>M. lohena</i>	# Surveyed pools
WAIK	3.5±4.5	1466±3197	43±23	0	18	36	9	11
MAUNA	4.1±2.7	91±96	94±109	57	0	29	14	7
ABKW	3.7±1.1	39±76	135±88	37	7	76	18	71
MAKA	6.1±2.9	124±223	96±52	43	7	50	14	44
PUHO	5.2±6.7	45±78	121±69	18	18	46	18	11
KAHO	14.5±3.2	14±48	113±69	19	0.5	60	12.5	184

Salinity, surface area, and distance are shown as mean ± standard deviation

in 18 % of total pools. At the MAKA site, *H. rubra* occurred in 50 % of 44 total pools and *M. lohena* occurred in 14 %. *H. rubra* occurred in 46 % of pools surveyed at PUHO ($n=11$) and 60 % at KAH0 ($n=184$), whereas *M. lohena* was present in 18 and 12.5 %, respectively. The lowest percent of pools with endemic shrimp was at MAUNA resort property where *H. rubra* was in 29 % of 11 pools. Other endemic anchialine species that were seen across the sites in less than 1 % of pools included gastropods (*Nerita picea*, *Neritina vespertina*, *Theodoxus cariosus*, and *Thiaridae* species), the endemic prawn *Macrobrachium grandimanus*, the glass shrimp *Palaemon debilis*, and the candidate endangered damselfly *Megalagrion xanthomelas*.

Introduced fish were found in pools throughout each site (Table 2). Poeciliids were more common than tilapia and occurred in 85 pools across all sites with up to 57 % of pools at any one site (MAUNA; Table 2). Tilapia were present in a total of 13 pools across all sites and were at every site but MAUNA. Endemic shrimp only co-occurred with tilapia in one pool and were in extremely low densities (3 per 10 m²) during the time of the survey. *H. rubra* co-occurred with poeciliids more frequently and were observed in 35 % of the pools with poeciliids. At the time of surveys, *M. lohena* only occurred in two pools with poeciliids.

Contingency table results show that *H. rubra* and *M. lohena* were significantly less likely to be observed when introduced fish were present. During daytime surveys across all sites, poeciliid presence was significantly associated with no *H. rubra* observed in pools ($\chi^2=33.1$, $P<0.00001$) as well as no *M. lohena* in pools ($\chi^2=5.5$, $P=0.02$). Similarly, tilapia presence was significantly associated with no *H. rubra* observed in pools ($\chi^2=13.6$, $P=0.0002$). The Pearson's chi-square test comparing tilapia with *M. lohena* frequencies was not significant ($\chi^2=1.2$, $P=0.28$); however, this result is likely an artifact of a zero count in one category; there were no pools with both tilapia and *M. lohena*.

Analysis of current patterns of fish distribution shows that pools with fish were clustered, not random or dispersed across the landscape. The Global Moran's index and z-score were positive at all sites and had statistical significance at KAH0,

PUHO, ABKW, and MAKA (Table 3). Therefore, at these sites, pools with fish were more likely to be near other pools with fish rather than pools without fish. In some cases, clusters of anchialine pools containing fish were located in close proximity to the edges of fishponds which also contain poeciliids and/or tilapia (Fig. 3). In other examples, pools with fish were relatively distant from either ocean or fishpond shorelines compared to clusters of pools without fish.

Modeling Results

Habitat Inundation

At all sites, rising sea levels will inundate current pools connecting them to the ocean. Projections of pool inundation vary by site and scenario (Figs. 4 and 5 and Table 4). Under the 0.5-m sea level rise scenarios, relatively minor percentages of pools become inundated at all sites (0–13 %) except at the Waikoloa resort site (WAIK; 92 %) and the KAH0 national park site when groundwater levels are included (32 %). At the WAIK site, all but one current pool are within 72 m of the current shoreline and a rise of only 0.5 m will cause these pools to connect overland to the ocean (Fig. 4). At the KAH0 site, incorporating current day groundwater levels in the scenario causes pools currently sitting at the edge of fishpond embayments to merge with fishponds. When groundwater is not part of the scenario, only 8 % of pools are inundated. Under the 1- and 1.5-m scenarios, increasing numbers of current pools will become inundated by marine waters at all sites. For the 1- and 1.5-m scenarios, groundwater causes a large difference in the number of pools inundated at the MAKA and ABKW sites. When groundwater is incorporated into the models, more pools merge with current fishpond embayments (Fig. 6b).

Formation of New Pool Habitat

Although current pools will become inundated as sea levels rise, new pools will emerge in the porous basalt substrate (Fig. 5c, d). Under all scenarios, geospatial models indicate

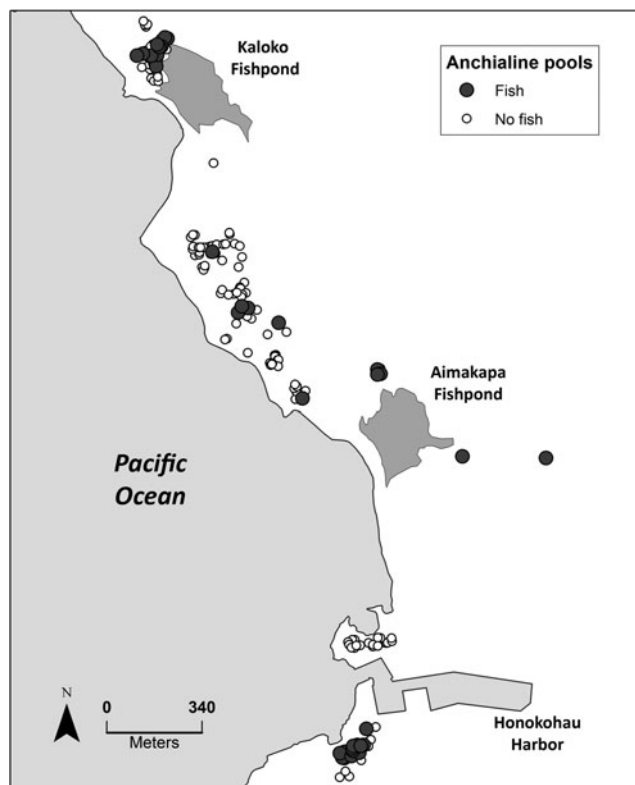
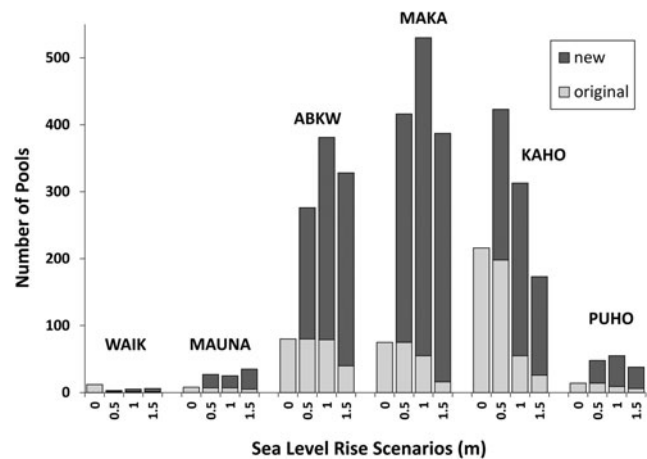
Table 3 Spatial patterns of fish distribution within anchialine pools at six sites as indicated by Global Moran's I index

Study site	Moran's index	z-score	P value	Variance	Distance (m)
WAIK ^a	0.06	1.17	0.24	0.019	227
MAUNA ^a	0.72	1.94	0.51	0.21	573
ABKW	1	17.52	<0.0001	0.004	378
MAKA	0.6	8.23	<0.0001	0.005	168
PUHO ^a	0.26	2.42	0.015	0.02	1000
KAHO	0.11	4.87	0.00001	0.0006	300

A positive index value, a positive z-score, and a statistically significant *P* value (<0.05) indicate a clustered distribution of features. Distance is the minimum value used in the calculation so that each pool has at least one neighbor

^a WAIK, MAUNA, and PUHO have less than the recommended 30 samples

that the greatest number of new pools will form within sections of ABKW and MAKA as well as in the KAHO national park (Table 5 and Fig. 4). For example, relative to current conditions at ABKW, up to 238 new pools will form at 0.5-m sea level rise, 336 at 1-m sea level, and 308 at 1.5 m. Once new pools are formed, they may be inundated by subsequent sea level rise, but additional pools will form inland in the rugose substrate (Fig. 5c, d).

**Fig. 3** Pattern of spatial distribution for anchialine pools with and without fish at the KAHO site**Fig. 4** Predicted changes in the number of anchialine pools at six sites under varying sea level rise scenarios. Bar plots include the inundation of existing pools (original) and creation of new pools (new). The current sea level scenario is 0 m. Results are from models without groundwater

Scenarios incorporating groundwater cause greater inland flooding than the same scenarios without groundwater. As a result, at all sea levels and sites, total pool surface area is higher for scenarios with groundwater compared to the same sea level scenario without groundwater (Table 5 and Fig. 6a). The only exceptions are at KAHO and MAKA where a 1.5-m sea level with groundwater creates 3617 and 67 m² less pool surface area compared to the same scenario without groundwater, respectively. This result is due to the fact that when groundwater is incorporated, some pools merge with fishpond embayments (Fig. 6b).

Increased pool surface area due to groundwater is not always reflected in pool counts. For example, at MAUNA, MAKA, and PUHO, total pool surface area increases when groundwater is incorporated into 1- and 1.5-m scenarios; however, total pool and new pool counts decrease (Table 5). In these cases, smaller pools in proximity to one another may merge into one pool when groundwater is included in the model.

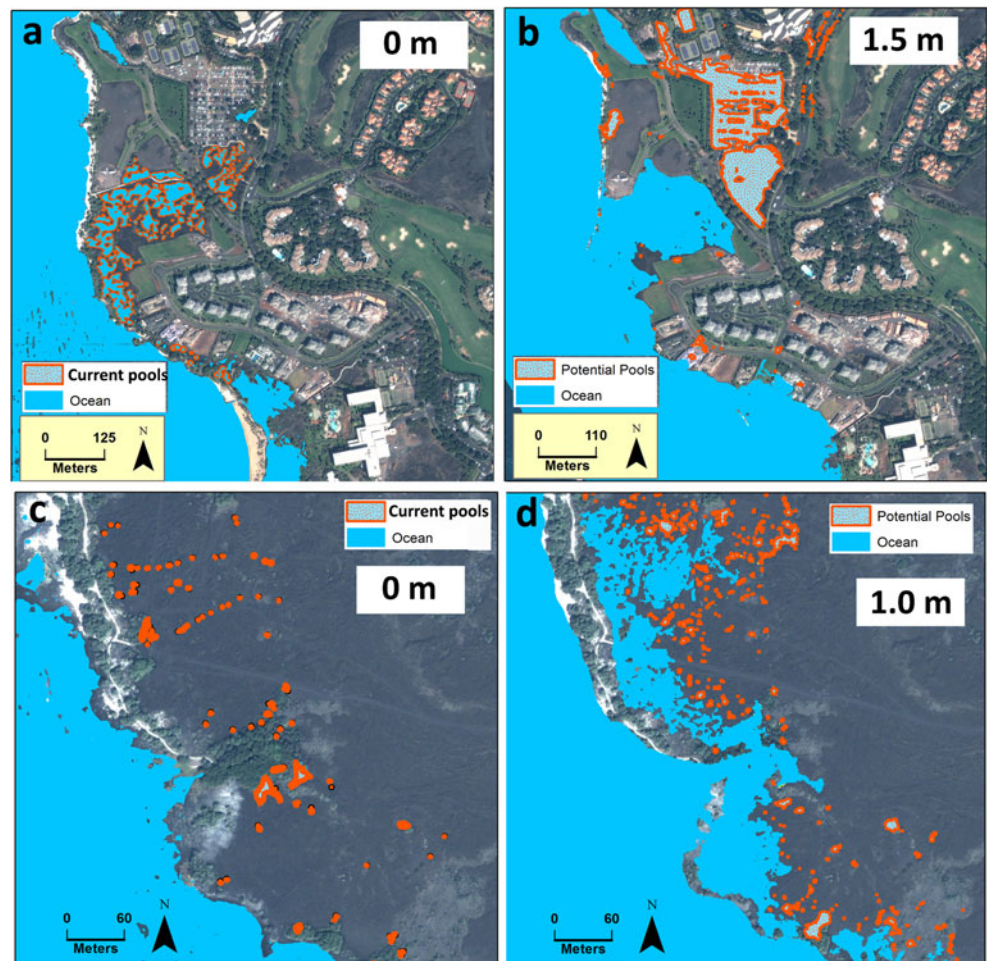
Land Use Effects

Land use does affect potential pool formation. Within WAIK and MAUNA resorts, increasing sea levels will cause inland flooding, but in the models, most of the flooding will occur on golf course turf or paved areas (Figs. 1 and 5a, b). Unless these areas were to be converted back to lava substrate, pools will not form in these locations. Some sections of the coastline are not developed but will not be conducive to pool formation because of high-elevation cliffs such as the southern section of PUHO.

Dispersal of Introduced Fishes

Rising sea levels will aid in the dispersal of introduced fishes from current occupied pools to unoccupied existing and future

Fig. 5 Inundation at two sites due to sea level rise scenarios prior to screening of potential pools by land use or proximity to ocean surface. **a, b** Waikoloa resort complex at 0- and 1.5-m sea level rise showing inundation over parking lots and grass turf; **c, d** portion of Kaloko-Honokōhau National Historic Park at 0- and 1-m sea level rise. The 0-m scenarios show current pool locations



pool habitats (Fig. 2). Introduced fishes currently inhabit at least 27 % of the existing anchialine pools ($n=405$) included in this study and are present at all sites (Table 5 and Fig. 7). Additionally, tilapia and poeciliids occur in fishpond embayments (Fig. 1: KAHO, MAUNA) and man-made resort water features (MAUNA, WAIK). With a 0.5-m sea level rise at an extreme tide, fish will have the potential to disperse from current habitats to new pools raising the total percent of pools potentially occupied with fish to 38.6 %. At the 0.5-m scenario, introduced fish have the potential to disperse into 27 to 70 % of all pools at different sites (Table 5). The percentage of potentially occupied pools decreases with further increases in sea level rise, because some of the source areas will become inundated by the ocean environment (Table 5). For example, at all sites but KAHO, potential occupancy of pools by introduced fishes drops to 0–17 % at the 1.5-m sea level rise scenario. These results are based on models that do not include groundwater, high wave events, or human transport of fish.

Introduced fishes currently occupy a range of pool sizes but poeciliids are commonly found in smaller pools than those occupied by tilapia (Fig. 8). Poeciliids are found in pools as with an average surface area of 77 ± 139 (standard deviation)

m^2 and in pools as small as 0.6 m^2 . Tilapia have been found in pools with an average surface area of $179 \pm 223 \text{ m}^2$ and in pools as small as 6.3 m^2 . Both taxa are commonly found in large ponds ($>400 \text{ m}^2$) or fishponds either alone or together. A comparison of pool sizes currently occupied by introduced fishes with pools that will be created under future sea level scenarios indicates that new pools will be of the range of sizes currently occupied by both poeciliids and tilapia. In addition, numerous pools will be created that will be smaller than currently observed with tilapia, but will be large enough to support poeciliids (Fig. 8).

Discussion

Efforts to conserve coastal biodiversity and ecosystems under changing climates have begun to include geospatial predictions of future habitat so that these land areas can be protected (Mawdsley et al. 2009). As sea levels rise, many low-elevation coastal ecosystems will become inundated more frequently through the tidal cycle until they are eventually underwater 100 % of the time. Ecosystems may gradually

Table 4 Percentage of anchialine pools inundated at various sea level rise scenarios within six sites

Site	Groundwater model	Percent of current pools inundated under sea level rise scenarios (m)		
		0.5	1	1.5
WAIK (12)				
	no GW	92	92	92
	GW	92	92	92
MAUNA (8)				
	no GW	13	13	38
	GW	13	13	38
ABKW (80)				
	no GW	0	1	50
	GW	0	36	60
MAKA (75)				
	no GW	0	27	79
	GW	0	62	98
PUHO (14)				
	no GW	0	36	57
	GW	0	36	57
KAHO (216)				
	no GW	8	75	88
	GW	32	82	89

The total number of current pools known at each site is included in parentheses. Water level models with groundwater (GW) and without groundwater (no GW) are included

migrate inland if conditions are appropriate and open space is available (Doyle et al. 2010; Stralberg et al. 2011). Results from this study, which examined 405 anchialine pools along

24 km of coastline, show that introduced fishes are prevalent in a variety of study sites and have a negative association with *H. rubra* and *Metabetaeus lohena*; however, these endemic shrimp species maintain a widespread distribution. Spatial analyses show that current pools with fish tend to be clustered together indicating that, at least in some areas, fish have dispersed from initial sites of introduction to nearby pools. Geospatial models show that current pool habitats will become increasingly inundated as sea levels rise and that new pools will form inland. Because there is high subsurface hydrologic connectivity within the coastal aquifer (Oki 1999), there is a high likelihood for these new habitats to be populated by anchialine shrimp and mollusk species which may disperse through subterranean groundwater (Kano and Kase 2004; Craft et al. 2008). Additionally, it is likely that some new habitats will be colonized by nonnative fishes as they disperse overland during high water events.

Sea Level Rise Models

This study is one of the few examples incorporating groundwater into predictions of ecosystem shifts under sea level rise (Bjerklie et al. 2012; Kane et al. 2014; Cooper et al. 2015). The methods applied may be useful for examining sea level rise effects on other ecosystems dependent on groundwater in unconfined coastal aquifers. Prior to model construction, the relationship between groundwater level and distance from shoreline should be calibrated to local conditions. In this study, the groundwater models were based on groundwater levels near Kāloko-Hōnokohau National Historical Park, which are assumed to be relatively representative of other areas examined in the present study (Marrack 2014).

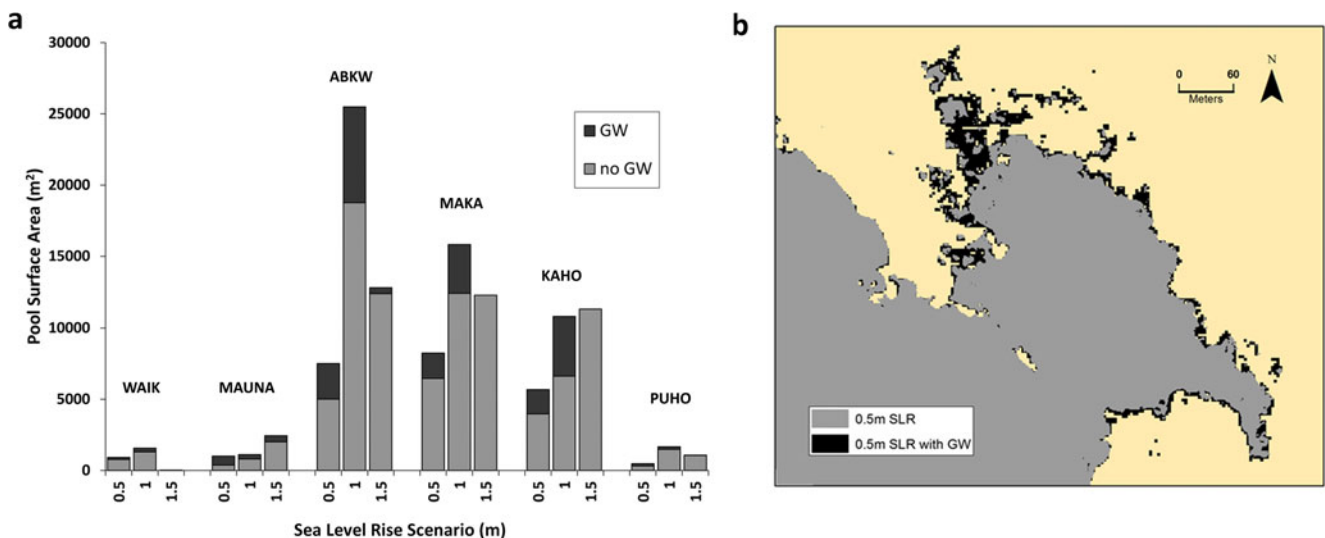


Fig. 6 Influence of groundwater on predicted changes to anchialine pool surface area and distribution. **a** Total anchialine pool surface area at six sites under varying sea level rise scenarios. Bars show results from models without groundwater and the additional pool surface area

created when groundwater is incorporated. **b** Incorporating groundwater in models increases flooded areas at the edges of a fishpond embayment within Kaloko Honokohau National Historical Park (0.5-m sea level rise at mean higher high water)

Table 5 The number and surface area of pools created under sea level rise scenarios at six sites on the western coast of the island of Hawaii

Site	Sea level rise (m)	Number of new pools		Total pool area (m ²)		Total number of pools		Average pool area (m ²) mean ± SD		Pools with fish access (% of total)
		no GW	GW	no GW	GW	no GW	GW	no GW	GW	
WAIK										
	0.5	2	5	784	919	3	6	392±390	184±350	33.3
	1	4	5	1312	1572	5	6	328±404	314±427	20.0
	1.5	5	1	13	28	6	2	3±1	28	16.7
MAUNA										
	0.5	20	15	385	1005	27	22	19±30	67±77	70.4
	1	18	9	811	1120	25	16	45±109	124±237	32.0
	1.5	30	16	2013	2440	35	21	67±239	153±365	11.4
ABKW										
	0.5	196	238	5015	7513	276	318	26±62	32±76	29.7
	1	302	336	18,762	25,492	381	387	62±236	76±409	11.5
	1.5	288	308	12,422	12,833	328	340	43±173	42±164	1.2
MAKA										
	0.5	341	362	6464	8248	342	437	19±51	23±57	39.5
	1	475	428	12,444	15,822	530	457	26±75	37±124	22.6
	1.5	371	374	12,310	12,243	387	376	33±150	33±169	0.5
KAHO										
	0.5	225	131	3977	5682	423	277	18±44	44±96	31.7
	1	258	122	6622	10,819	313	162	26±87	89±304	46.3
	1.5	147	66	11,332	7715	173	89	77±417	117±455	32.9
PUHO										
	0.5	34	37	322	484	48	51	10±30	13±41	27.1
	1	46	26	1486	1665	55	35	32±64	64±107	0
	1.5	32	23	1063	1084	38	29	33±74	47±69	0

Pools were enumerated for scenarios with (GW) and without (no GW) groundwater levels. All calculations of new pools are relative to current conditions

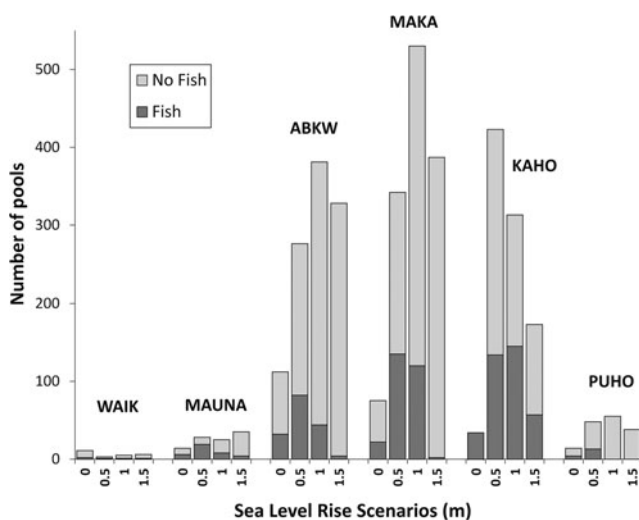


Fig. 7 Number of pools that have potential to be occupied with introduced fishes as a proportion of the total number of pools after extreme tides under rising sea level scenarios (0, 0.5, 1, and 1.5 m)

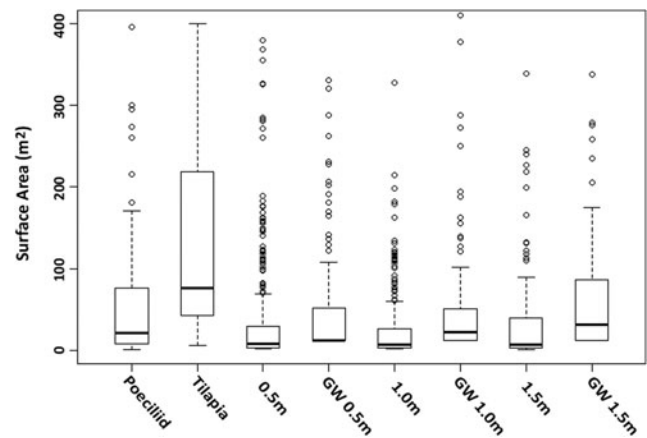


Fig. 8 Size distribution of pools currently containing poeciliids ($n=102$) and tilapia ($n=19$) compared to predicted pool sizes under sea level rise scenarios with and without groundwater (GW). Pools over 400 m² are not included in the plot

However, groundwater levels in the study area may become lower in the future due to increased water withdrawal (Oki 1999; Thornberry-Ehrlich 2011) and decreasing annual precipitation (Giambelluca et al. 2013). Therefore, the sea level models that incorporate current groundwater levels in this analysis may be considered an upper bound to future inland flooding. Although the scenarios without groundwater are considered a lower bound to flooding, it should be assumed that some groundwater is needed for brackish anchialine ecosystems to exist.

Due to uncertainties in future groundwater levels as well as limitations of the LiDAR data, calculations regarding pool formation, size, and placement should be considered estimates of future habitat extent. Accuracy assessments that examined the ability of LiDAR to detect current pool locations within the study area showed that LiDAR was not able to detect pools in caves or narrow fissures (Marrack 2014). Furthermore, LiDAR point data are sometimes spaced farther apart than pool areas, effectively missing small pools ($<5\text{ m}^2$) 59 % of the time (Marrack 2014). Although the models may underpredict the locations of small or cave-like pools, the general patterns of pool inundation and formation along with the locations of expected new pool complexes should accurately reflect future conditions under sea level rise.

Geospatial predictions of coastal change typically include coastal elevation data and local sea level rise scenarios (Gesch 2009) but may also incorporate wave dynamics and erosion models (e.g., Reynolds et al. 2012). Because anchialine pools are typically found in bedrock and are usually over 15 m inland from the shore, erosion and wave run-up are relatively unimportant for modeling sea level rise impacts to this ecosystem. Anchialine pools closest to the coast may be filled with sediment and ocean water during extreme wave events, but this is usually a temporary impact because groundwater rapidly flushes the system (unpublished personal observation).

Anchialine Pool Community Response to Sea Level Rise

For coastal ecosystems that are dependent on groundwater, changes to groundwater heights and salinities may drive shifts in community structure (Doyle et al. 2010). Surveys used in this study support previous findings that the dominant anchialine shrimp species *H. rubra* (Maciolek and Brock 1974; Havird et al. 2014) and *M. lohena* (Maciolek and Brock 1974) have wide salinity tolerances. Pool salinities vary by aquifer and may become less saline with increasing distances from the ocean, but the same dominant species are found throughout. These observations indicate that increasing pool salinity due to sea level rise will not necessarily lead to changes in pool community structure unless marine conditions dominate. However, it is unclear if *H. rubra* and *M. lohena* are able to complete their entire life cycle in the full range of salinities measured in pool habitats. It is also

unclear how sea level rise will affect the subterranean habitat except that some parts of the aquifer will become more saline. Furthermore, salinity tolerances are unknown for rarer endemic anchialine shrimp and mollusk species found within restricted ranges (Sakihara 2012). Work examining species-specific salinity tolerances is needed to understand how anchialine communities may shift with sea level rise. This is especially important because reduced rainfall and increased withdrawal could lead to lower groundwater levels and elevated salinities in the future.

Results show that sea level rise with extreme tides has the potential to aid in dispersal of introduced fishes from currently occupied pools to unoccupied habitats. Fish may not disperse immediately, but over time will spread into new habitats. The clustered pattern seen in current fish distribution indicates that some dispersal has occurred since fish were originally introduced. The mechanism for this dispersal is probably a combination of (1) dispersal during exceptionally high water levels when habitats containing introduced fish connect to previously unoccupied habitats such as fishponds and nearby anchialine pools, (2) dispersal due to humans, and (3) dispersal by the night heron (*Nycticorax nycticorax*) that has been seen flying between pools and dropping tilapia. The connectivity models used in this study do not account for human-aided or bird-aided dispersal, groundwater levels, or wave events that may increase fish dispersal rates.

In other freshwater systems, it has been well documented that tilapia (*O. mossambicus* and other species) and poeciliids (*Poecilia reticulata*, *G. affinis*) have a strong negative effect on native biodiversity due to predation and habitat modification (Global Invasive Species Database 2010). Because these fish are tolerant to a wide range of ecological conditions, have highly generalist dietary requirements, and have rapid reproductive rates, they have become successfully established in almost every region in which they have arrived (Canonica et al. 2005; Deacon et al. 2011). Out of numerous abiotic and biotic factors examined in anchialine pools on Hawaii Island, the occurrence of tilapia and poeciliids were the most important factors explaining the absence of *H. rubra* and *Metabetaeus lohena* shrimp during daylight surveys (Marrack et al. 2015). Predation pressure by poeciliids (*G. affinis*) causes the dominant endemic grazer (*H. rubra*) to hide during daylight hours (Chai et al. 1989; Capps et al. 2009; Carey et al. 2011; Sakihara 2012). Furthermore, Dalton et al. (2013) found that in pools with introduced poeciliids, epilithon (biofilm) biomass was higher compared to pools without fish. These results suggest that by causing *H. rubra* to hide during the day, fish may partially release primary producers from grazing. More work is needed to fully understand the effect of introduced fishes on anchialine ecosystems. Regardless, tilapia and poeciliids already inhabit a range of pool habitats in Hawaii, and it is likely that if they disperse into new habitats, they will continue to thrive. One exception

is that tilapia may be restricted from occupying the smallest pools ($<5 \text{ m}^2$). It would be best to target fish removals in locations where current distributions threaten to be the source of spreading to new pools as sea levels rise.

Land-Use and Coastal Topography Limit Future Habitat Formation

The lowest numbers of current and future pools relative to coastline length exist at the two resorts (WAIK and MAUNA) as well as at the PUHO national park. At the resorts, low lying land has been converted to paved areas or golf courses, and relatively little area exists for new pool formation. Coastal planning should consider utilizing geospatial models of sea level rise to determine where future pool habitats will emerge on the landscape so that these areas can be protected from development. This study does not examine the effects of land use on groundwater. However, nonpoint source pollution and groundwater withdrawals in populated areas are important considerations for the health of anchialine communities and should be examined.

Not all coastal areas are ideal for anchialine pool formation. The southern portion of the PUHO national park shoreline consists of 10–15-m-high cliffs. The only anchialine pool in this southern section of the park consists of an old well set back from the shoreline and dug down to groundwater level. Interestingly, the well contains endemic anchialine shrimp, which suggests that habitat for native biodiversity may be created in some areas by digging down to the groundwater.

Biodiversity Response to Climate Change

Identifying the response of biodiversity to climate change is a highly active area of research (Parmesan 2006). Various components of climate change are expected to affect biodiversity at every level, from individual organisms to biomes (Bellard et al. 2012). Due to climatic changes, species may no longer be adapted to the local environmental conditions and may only be able to persist through physiological responses (phenotypic plasticity or genetic) or range shifts (Parmesan 2006; Bellard et al. 2012). In addition, changes in climate may support the dispersal and establishment of nonnative species leading to novel communities (Lockwood et al. 2013). Observed changes in species distribution have been linked to local and regional climate change for numerous terrestrial (Colwell et al. 2008; Chen et al. 2011), freshwater (Hickling et al. 2006), and marine taxa (Sorte et al. 2010). Most of these studies focus on temperature and sometimes precipitation as the climatic drivers of species range shifts. Sea level rise, another component of climate change, has been recorded at tide gauges around the world with global mean sea levels increasing by approximately 0.19 m between 1901 and 2010 (IPCC 2013). However, evidence for associated changes in the distribution of species dependent on coastal ecosystems

(beach, dune, estuary, or anchialine pool) is scarce over this time frame. The lack of observed changes to biodiversity due to recent sea level rise may be because of the relatively small magnitude of change to date, difficulties associated with detecting change within dynamic coastal systems, anthropogenic impacts overriding climate change effects, and/or insufficient long-term monitoring in these systems. The rates and magnitude of sea level rise are expected to increase significantly in this century (Vermeer and Rahmstorf 2009); therefore, monitoring programs dedicated to detecting changes in species distributions, community assemblages, and ecosystem processes are recommended in coastal habitats.

Numerous modeling approaches have recently been developed to predict species range shifts due to climate change (Pereira et al. 2010; Bellard et al. 2012). Bioclimatic models, often referred to as species distribution models, are commonly used. These models typically relate current distributions of a species to multiple environmental variables in order to define an environmental or climatic niche for the organism (Elith and Leathwick 2009). Predicted changes to climate variables are then used to predict future niche space and potential species distributions. Bioclimatic models may also incorporate physiological tolerances, biotic interactions, and dispersal potential (Elith and Leathwick 2009; Kearney and Porter 2009; Soberón and Nakamura 2009). Although this study does not use the statistical methodology typically incorporated in bioclimatic models, the general approach is the same. Essentially, model the future location of a species' niche and assume the species may move into these areas. In this study, the basic niche or habitat for anchialine pool species is exposed groundwater in rocky coastal substrate. Because endemic *H. rubra* and *M. lohena* are currently found throughout the range of recorded pool temperatures and salinities (Marrack et al. 2015), these water properties were not incorporated into the models. Data on nonnative fish occurrence and land use were also examined. Based on our current knowledge of anchialine systems, these factors appear to be the key drivers of endemic shrimp occurrence. However, further research on the ecology and biology of *H. rubra* and *M. lohena* may reveal additional abiotic and biotic factors that should be added to predictive models. Furthermore, little is known about habitat requirements for less common anchialine species. Improving projections of the biological community response to climate change as a whole will require a better understanding of ecological and hydrologic processes in current anchialine habitats.

Conclusion

As sea levels rise and current pools become inundated, high subsurface hydrologic connectivity will cause future pools to emerge in the rugose lava terrain. Endemic anchialine shrimp and mollusks will be able to populate new pools, because

adults and larvae move in brackish coastal groundwater through subterranean cracks (Craft et al. 2008). Low-lying coastal areas that will become future anchialine habitat should be protected from development. Because sea level rise will provide a mechanism for introduced fishes to disperse from infected to uninfected habitats, fish removals will help protect future ecosystem integrity. Conservation efforts aimed at reducing introduced fish, minimizing groundwater contamination and withdrawal, and protecting low-lying coastal area from development will ultimately allow the unique Hawaiian anchialine pool ecosystem to persist in the face of climate change.

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