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9 Abstract

10 Declining natural resources have led to a cultural renaissance across the Pacific that seeks to
11 revive customary ridge-to-reef management approaches to protect freshwater and restore
12 abundant coral reef fisheries. We applied a linked land-sea modeling framework based on remote
13 sensing and empirical data, which couples groundwater nutrient export and coral reef models at
14 fine spatial resolution. This spatially-explicit (60 x 60 m) framework simultaneously tracks
15 changes in multiple benthic and fish indicators as a function of marine closures, land-use and
16 climate change scenarios. We applied this framework in Hā‘ena and Ka‘ūpūlehu, located at
17 opposite ends of the Hawaiian Archipelago to investigate the effects of coastal development and
18 marine closures on coral reefs in the face of climate change. Our results indicated that projected
19 coastal development and bleaching can result in a significant decrease in benthic habitat quality
20 and community-led marine closures can result in a significant increase in fish biomass. In
21 general, Ka‘ūpūlehu is more vulnerable to land-based nutrients and coral bleaching than Hā‘ena
22 due to high coral cover and limited dilution and mixing from low rainfall and wave power,
23 except for the shallow and wave sheltered back-reef areas of Hā‘ena, which support high coral
24 cover and act as nursery habitat for fishes. By coupling spatially explicit land-sea models with
25 scenario planning, we identified priority areas on land where upgrading cesspools can reduce
26 human impacts on coral reefs in the face of projected climate change impacts.

Keywords: coral reef, land-use, nutrients, groundwater, bleaching, coastal development, scenario planning, impact, ridge-to-reef, management, land-sea models

1. Introduction

Over the past fifty years, climate change has become a global threat to coral reefs through impacts from bleaching (Hoegh-Guldberg 1999), ocean acidification (Hoegh-Guldberg et al. 2007), and intensified storms (Webster et al. 2005). In addition, growing human population and resource use directly and indirectly impacts coral reefs at the local scale, through increased fishing pressure and land-use changes (Nyström et al. 2000). Fishing pressure alters the composition and trophic structure of marine ecosystems, and the resulting changes to functional groups can have strong effects on overall system resilience (Jackson et al. 2001, Green and Bellwood 2009). For example, removal of keystone herbivores can drive phase shifts to less desirable ecosystem states dominated by algae (Hoegh-Guldberg et al. 2007, Mumby 2009, Bahr et al. 2015). Land-use change has altered terrestrial fluxes of freshwater (Vörösmarty and Sahagian 2000), sediments (Syvitski et al. 2005), and nutrients (Downing et al. 1999, Elser et al. 2007) to coral reefs (Kroon et al. 2014). Excess nutrients have been shown to impact coral reefs by promoting benthic algae growth and smothering corals (Fabricius et al. 2005, Fabricius 2005, Smith et al. 2016), thereby contributing to lack of recovery from bleaching (Wooldridge 2009, Wooldridge and Done 2009). Although the extent to which nutrient levels interact with elevated sea surface temperature (SST) to affect the outcome of bleaching events remains poorly understood, it is increasingly recognized that water quality plays a complex role in the fate of nearshore coral reefs under climate change (Anthony 2006, Wenger et al. 2015, Morgan et al. 2016).

Managing land-based pollutants on coral reefs requires linking their sources to potential downstream impacts. However, tracing those linkages is challenging because of multi-scale processes that affect pollutant export, retention, and eventual impacts on coral reefs. For instance, coastal development can occur in multiple areas within a watershed and increase nutrient discharge into groundwater and ultimately the coastal zone (Neil et al. 2002, Terry and Lal 2008, Bartley et al. 2014). Once in the ocean, nutrients are diffused by nearshore processes and impact coral reefs by promoting benthic algae growth which can affect coral health through

59 direct and indirect competition for space, negatively affect coral & fish settlement, and rapidly
60 colonize dead corals (Vermeij et al. 2010, Dixon et al. 2014). These impacts on the benthic
61 community result in reduced habitat quality and direct and indirect negative effects on some reef
62 fish taxa (Littler et al. 2006, Smith et al. 2006). The location and extent of coral reef impacts
63 from nutrient discharge also depend on marine drivers such as waves and tides (Rude et al.
64 2016). These processes create real challenges for identifying area on land, where management
65 action can mitigate downstream impact on the most affected reef areas (Klein et al. 2012, Brown
66 et al. 2017a).

67
68 These trends have led to the decline of important resources upon which human wellbeing
69 depends (Moberg and Folke 1999, Worm et al. 2006). Awareness of natural resource decline has
70 contributed to a cultural renaissance around the Pacific, where local communities seek to revive
71 local and place-based management (Johannes 2002), such as customary ridge-to-reef
72 management approaches (Minerbi 1999, McGregor et al. 2003), traditional closures, and
73 sustainable practices to foster social and ecological resilience (Johannes 2002, Foale et al. 2011).
74 Marine closures have been shown to protect coral reefs from direct threats, such as fishing
75 pressure (Halpern 2003). However, they have also been shown to fail in the absence of social
76 buy-in (Adams et al. 2011) and where impacts from land-based source pollution are not
77 considered (Halpern et al. 2013). Therefore, management strategies that reduce land-based
78 pollution have been widely advocated to foster coral reef resilience in the face of climate change
79 (Hughes et al. 2007). However, determining where and how to manage the land to promote coral
80 reefs in the face of multiple human drivers can be challenging and differs among places (Bruno
81 2016, Cinner et al. 2016, Fredston-Hermann et al. 2016). Anthropogenic impacts depend on local
82 conditions, including land-use change, topography and bathymetry, geology, and habitat. As a
83 result, local management should account for these underlying differences to reduce human
84 impacts on coral reefs (Klein et al. 2014).

85
86 In the Hawaiian Archipelago, two communities (Hā'ena, Kaua'i and Ka'ūpūlehu, Hawai'i)
87 embody this cultural renaissance (Pascua et al. 2017). In the face of climate change impacts and
88 declining resources, the Hā'ena and Ka'ūpūlehu communities are seeking to foster social and
89 ecological resilience by reviving place-based management practices. Due to their location at the

opposite ends of the main Hawaiian Islands, Hā‘ena is geologically old and highly eroded because of its exposure to heavy rains and very large open oceanic swells, while Ka‘ūpūlehu is young, dry, and sheltered from waves (Delevaux et al. 2018c). Both local communities are interested in a better understanding of how land-based sources of pollutants from golf courses, lawns and cesspools affect the health of their marine ecosystems. Coral reef fish caught near shore with nets, pole and line, or spears, are a very important component of community food systems (Vaughan and Vitousek 2013, Kittinger et al. 2015) and also a play key ecological role in terms of herbivory. Herbivorous fish species feed off benthic algae, which can bloom and cover the reef with land-based nutrients. By eating the algae, these protected fish create space for new corals to settle and ensure the persistence or resilience of the coral reefs.

Both communities initiated local-level fisheries management rules, including marine closures of different sizes, rooted in traditional ecological knowledge to foster coral reef resilience and are in need of additional information to guide land-based management actions (TNC 2015, DAR 2016). Even with healthy herbivorous fish populations, land-based pollution from increased coastal development can adversely impact coral reefs, especially under increased ocean temperature as a result of climate change (Fabricius 2005, Littler et al. 2006, Hughes et al. 2007). Therefore, it is important to these communities, and to the health of all marine ecosystems in the face of climate change, to ensure that future planning tries to minimize local human impacts on coral reefs. To support the local management efforts of these two communities and inform similar efforts throughout the region, we applied a novel land-sea modeling framework developed to inform sustainable development in Hawai‘i at the sub-watershed scale by tracking the effects of local management on coral reefs under projected climate change impacts (Delevaux et al. 2018b).

This spatially-explicit framework simultaneously tracks changes in multiple benthic and fish indicators under land-use and climate change scenarios and was developed independently for Hā‘ena and Ka‘ūpūlehu *ahupua‘a* (ridge-to-reef systems) in order to capture the different natural disturbance regimes and reflect the local ridge-to-reef processes. To determine the impact of co-occurring human drivers on coral reefs and support local management, we coupled human driver scenarios (climate change, coastal development, and marine closures) with these linked land-sea modeling frameworks. Specifically, we asked: (1) How climate change and land-based pollution

121 impacts on coral reefs differ within and between sites? (2) Can the established fisheries
122 management areas reduce climate change impacts? (3) Where can land-based management
123 reduce climate change impacts? Our approach was conducted in partnership with communities to
124 identify priority areas on land to help reduce downstream impacts on coral reefs and inform
125 place-based management.

128 **2. Methods**

129 **2.1 Study sites**

130 We focused on two *ahupua'a* at the opposite ends of the main Hawaiian Islands. Hā'ena is
131 located on the windward side of Kaua'i Island and Ka'ūpūlehu is located on the leeward side of
132 Hawai'i Island (Fig 1) (further described in Table 1). Geologically older and exposed to the trade
133 winds, Hā'ena receives high rainfall, resulting in steeply eroded cliffs, with high fluvial and
134 groundwater inputs (Calhoun and Fletcher 1999). The fringing reefs are wider and shallower
135 compared to Ka'ūpūlehu, where fringing reefs form a narrow band on the slope of the shield
136 volcano (Fig 1B-C) (Fletcher et al. 2008). The backreef areas form lagoons that are protected
137 from wave power by well-developed reef crests and support a benthic community dominated by
138 corals and high juvenile fish abundance (Goodell et al. 2018). The benthic community on the
139 wave-exposed forereefs is dominated by crustose coralline algae (CCA) and supports high adult
140 fish biomass (Friedlander et al. 2003, Jokiel et al. 2004). Geologically younger and located in the
141 rain shadows of Mauna Loa and Mauna Kea mountains, Ka'ūpūlehu is very dry and minimally
142 eroded, resulting in poorly developed ephemeral stream channels and large submerged
143 groundwater discharge (SGD) (Knee et al. 2010, Izuka et al. 2016). The reef slopes are
144 dominated by corals and have high habitat complexity, which supports higher fish biomass,
145 compared to the low habitat complexity of the shallow reef flats, dominated by turf algae and
146 supporting lower fish biomass (Minton et al. 2015).

147
148 The Hā'ena community worked for over a decade to create local-level fisheries management
149 rules based on indigenous Hawaiian practices (*pono*), which were passed into law in 2015.
150 Among these rules, the backreef of Makua was designated a marine refuge (*Makua Pu'uhonua*),
151 where corals can grow and provide habitat for juvenile fishes safe from powerful waves (DAR

2016) (Fig 1B). The same year, Ka‘ūpūlehu initiated a law implementing a 10-year fishing rest period (‘Try Wait’), which extends out to 120 feet deep (or 36.6 m) along a large portion of the coastline (Fig 1C). Both communities initiated these marine closures to enhance nearshore fisheries, a large portion of which consists of fish species that feed on algae (herbivorous fishes). At Hā‘ena, fishing pressure was relatively lower than Ka‘ūpūlehu prior to the establishment of the marine closures (Delaney et al. 2017) (Table 1).

2.2 Modeling approach overview

To spatially prioritize land management practices based on corresponding coral reef impacts, we determined the locations of projected climate and land-use change impacts on coral reefs and traced those back to the areas driving these impacts on land. In order to do so, we coupled various human driver scenarios (climate change, coastal development, and marine closures) with an adapted linked land-sea modeling framework (Delevaux et al. 2018b) (Fig 2). First, we designed land-use change scenarios that represent extreme projections of coastal development, using local county land zoning and climate change scenarios based on regional coral growth and mortality models for Hawai‘i (Hoeke et al. 2011). Second, we applied the land-sea framework, which combines groundwater models (MODFLOW, MT3D-MS, MODPATH), coastal plume modeling, marine driver models, and predictive models of coral reef resilience indicators (Delevaux et al. 2018b). To measure proxies of ecological resilience, the coral reef models focused on four benthic groups known to change under land-based nutrient discharge, and four fish indicator groups which also represent important cultural resources (Vaughan and Vitousek 2013, Minton et al. 2015). Finally, we undertook a spatially-explicit scenario analysis to assess the impacts of the proposed scenarios on coral reefs and identify priority areas on land where best management practices (BMPs) may reduce local human impacts on coral reefs.

2.3 Scenarios

2.3.1 Coastal development

We considered two feasible coastal development scenarios (low and high) at each study site that represent alternative projected and extreme land-use change specifically for each study site that generally could apply in many other watersheds across tropical high islands by 2050 (Table 2). Under current land cover/use, we estimated a total of 136 houses (99 cesspools; 37 septic

systems) and 6 ha of lawn in the model domain at Hā'ena, (Whittier and El-Kadi 2014), compared to Ka'ūpūlehu, which has 193 houses with 45 ha of residential green spaces, two resorts disposing of their wastewater after secondary treatment through an injection well, and a golf course (190 ha) (State of Hawaii 2003). The coastal development scenarios consisted of increasing the number of houses and lawns or residential green spaces according to current land zoning, building regulations, and current practices for wastewater treatment systems (septic tanks and injection wells). At Hā'ena, we assumed that new added houses are connected to a septic tank coupled with soil treatment (referred to as 'septic systems' hereafter) in both coastal development scenarios due to a recent statewide cesspool ban (HAR Title 11, Chapter 62). We assumed that existing houses remained on cesspools. Both wastewater treatment systems are associated with each house and discharge their effluent into the groundwater beneath the onsite sewer disposal system location. The low coastal development scenario in Hā'ena consisted of adding 20 houses (septic tanks) and lawns (total 1.3 ha) in accordance with existing regulations of only building one house per Tax Map Key (TMK). The high development scenario consisted of adding 73 houses (septic tanks) and lawns (total 5 ha), with up to two houses per TMK (which is illegal but occurs). At Ka'ūpūlehu, each parcel of current land use was an average 0.4 ha, with three levels of development intensity (i.e., low, medium, and high), which corresponded to 65%, 35%, and 10% of green space per residential lot, respectively. We considered two future coastal development scenarios where wastewater was disposed of through a wastewater injection well under the low coastal development scenario and through septic tanks coupled with soil treatment under the high coastal development scenario. Both coastal development scenarios consisted of adding 83 houses and green spaces with a total area of ~13 ha based on existing lots with low intensity development.

We assigned N and P flux to each land cover/use derived from local data where possible, and from literature values when local data were not available (see Table 3 for nutrient loading rates). The nutrient flux rates from developed parcels were based on the assumptions that each land parcel had a residential unit with three bedrooms at an occupancy rate of 1.5 persons per bedroom generating 435 m³.yr⁻¹ of wastewater (USEPA 2002). At Hā'ena, the groundwater recharge over lawns was computed assuming no irrigation given high rainfall in the area, with a recharge rate of 50 m³.ha.d⁻¹, fertilization rate of 192 kg.ha⁻¹, leaching 4.5 kg.ha⁻¹ of N and 0.2

kg.ha⁻¹ of P combined with the background concentrations (Wang et al. 2014). At Ka‘ūpūlehu, the coastal nutrient flux from residential green spaces and the golf course were based on an assumed irrigation rate of 46.8 m³.ha.d⁻¹ (Engott 2011, CH2MHill 2013), fertilization rates at 236 kg.ha⁻¹ of N and 122 kg.ha⁻¹ of P, and a leaching rate of 5% for both nutrients (i.e., 12 kg.ha⁻¹ of N and 6.1 kg.ha⁻¹ of P) (Throssell et al. 2009). We scaled the nutrient flux of the added injection well effluent to the proposed development and assumed the same loading and discharge rates as the existing injection well (State of Hawaii 2003).

2.3.2 Climate change

Two climate change scenarios (low and high impact in terms of levels of coral bleaching) were considered to assess the potential impact of increases in sea surface temperature (SST) on coral reefs by 2050 (Table 2). The coral bleaching scenarios were derived from regional climate change models calibrated using observed coral growth and mortality rates linked to mass coral bleaching episodes for the Hawaiian archipelago (Hoeke et al. 2011). An average greenhouse gas emissions scenario (A1) was assumed for the years 2000–2099 A.D. (21st century), which corresponds to a future with very rapid economic growth, global population peaks in mid-century and declines thereafter, a rapid introduction of new and more efficient technologies, and an energy system with no heavy dependence on one particular source (IPCC 2007). Based on SST projections using the Coupled Model Intercomparison Project phase 3 (CMIP3) for SST projections with a threshold for heat stress increasing by 0.1° C every decade, Hoeke et al. (2011) estimated a coral cover decline of 25% to 75% for across the Hawaiian Archipelago by the end of the century, using a combination of adaptation scenarios, coral growth and mortality models. Based on CMIP5 for SST projections and downscaling to 4 km, van Hooidonk et al. (2016) estimated that Hawai‘i is likely to experience high thermal stress likely to induce severe coral bleaching on an annual basis beginning around 2040-2045. During the last 2015-2016 massive coral bleaching event in Hawai‘i, Kramer et al. (2016) reported that bleaching prevalence averaged 53.3%, and resulted in an average coral cover loss of 49.7% along the West Hawai‘i shoreline. Another study found that some South Kohala reefs experienced 55-99% coral mortality due to bleaching in 2015 (Maynard 2016). Hence, under our low bleaching scenario, the current coral cover for reef areas shallower than 5 m was reduced by 30%. In our high bleaching scenario, the current coral cover between 0-5 m and 5-10 m was reduced by 50% and

25% , respectively, because deeper waters are cooler and can reduce the impact of increase in SST (Bridge et al. 2013). Given the large uncertainties and debate surrounding coral adaptation to heat stress (Baker et al. 2008), these scenarios should not be considered quantitative forecasts of percent coral cover change for these specific locations, but instead, conservative and large-scale probability-based estimates of the relative impact of predicted increases in SST on coral reefs in the Hawaiian Archipelago over the next 30 years.

2.3.3 Marine closures

To model the effects of the recently enacted marine closures, we assumed full compliance and no fishing within closure boundaries (TNC 2015, DAR 2016). In terms of size, the marine reserve at Hā'ena (7.1 ha) is relatively small compared to the fishing rest area at Ka'ūpūlehu (355.4 ha) (Table 2). To reflect the absence of fishing pressure in those locations, we increased the fish biomass in the water scaled by current fishing pressure and observed catch-per-unit-effort (CPUE) at each site. We calculated and mapped the catch levels (g.m^{-2}) per grid cell for three shore-based fishing gear types (i.e., net, line, and spear fishing) by multiplying the CPUE estimates (kg.hr^{-1}) from locally available creel surveys (Vaughan and Vitousek 2013, Delaney et al. 2017) (Table 1), with a high resolution fishing effort map (100m^2) spanning the Main Hawaiian Islands, expressed in (hr.ha^{-1}) (see Stamoulis & Delevaux et al. [2018] for more details). In ArcGIS, catch biomass was added to each fish indicator based on species susceptibility to gear type and the proportion of biomass per fish indicator on a cell-by-cell basis. We assumed that gears differentially targeted fish functional groups, where net and spear fishing gears target high proportions of herbivores as well as piscivores, while line fishing only targets piscivores (Cinner et al. 2009).

2.4 Coral reef indicators

To measure ecological resilience, we considered the percent cover of four benthic groups and the biomass of four fish groups (kg.ha^{-1}) based on their ecological roles and cultural importance to local communities (Friedlander et al. 2002, Poepoe et al. 2005, DAR 2016) (see Appendix S1: Table S1 for more information). The benthic indicators (crustose coralline algae (CCA), scleractinian corals, turf algae and macroalgae) are known to respond to changes in nutrient runoff and influence fish functional group assemblages (Brown et al. 2017b), and therefore

support aspects of coral reef ecological resilience (Green and Bellwood 2009, Smith et al. 2016). Distributions of fish taxa identified as important for subsistence and cultural practices by the local communities (e.g., Acanthuridae, Scarinae, Carangidae) (Friedlander et al. 2002, Poepoe et al. 2005) were categorized and modeled by functional group: (1) browsers, (2) grazers, (3) scrapers, and (4) piscivores (see Appendix S1: Tables S1 & S2 for more details) (Green and Bellwood 2009, Jupiter and Egli 2011). The percent cover of the benthic indicators and biomass of the fish indicators (g.m^{-2}) was derived from reef survey data collected by the Fisheries Ecology Research Lab (FERL) at the University of Hawai'i and The Hawai'i Nature Conservancy (TNC) reef monitoring program. The field dataset from Hā'ena was collected over two sampling periods (July 2013 and August 2014) and comprised 126 survey locations randomly stratified by habitat (nearshore, back-reef, and fore-reef areas), allocated across Makua and Pu'ukahua reef complex (Fig 1F) (refer to Goodell et al. (2018) for more details). The field dataset from Ka'ūpūlehu was collected over two sampling periods (2012 and 2013) and comprised 243 survey locations randomly stratified across two factors: management status (inside and outside the Fisheries Replenishment Area) and reef types (Fig 1G) (refer to Minton et al. (2015) for more details). Note that the reef survey data at both locations were collected prior to the implementation of the marine closure and the 2015-2016 massive bleaching event.

2.5 Linked land-sea modeling framework

Ka'ūpūlehu is located in a very dry region lacking perennial streams and surface runoff (Street et al. 2008, Knee et al. 2010). Whereas, Hā'ena is located in a wet region, where surface water discharge largely exceeds SGD but SGD nutrient flux account for over 70% of the total coastal nutrient discharge (Knee et al. 2008). Thus, SGD was found to be the primary vector of nutrients to coastal waters in both study sites (Knee et al. 2008, Street et al. 2008, Knee et al. 2010). The land-sea framework is made of three key components (refer to Delevaux et al. [2018b] for more details): (1) groundwater models (MODFLOW, MT3D-MS, MODPATH) (Figs 2A & 2D), (2) land-sea link models (flow tubes and coastal plume models) (Fig 2D), and (3) coral reef predictive models (including the marine drivers) (Fig 2E-F).

We leveraged empirical and remote sensing data, publicly available at the time of the modeling in these regions to calibrate and validate these models at both sites (Fig 1).

2.5.1 Groundwater models

The groundwater models boundary conditions were defined to comprise the groundwater flow path from the zones of recharge to coastal discharge, the entire *ahupua'a* boundaries, and the coastal development area, using MODPATH (Pollock 1994, Pollock 2012) to set: (1) a representative groundwater recharge flux at the upslope boundary; (2) no-flow condition at the lateral boundaries; and (3) the elevation of the groundwater discharge at the coast and submarine boundaries according to the greater seawater density compared to freshwater. We then estimated the coastal groundwater discharge ($\text{m}^3\cdot\text{yr}^{-1}$) using the groundwater model MODFLOW (Harbaugh 2005) and the dissolved inorganic nitrogen and phosphorus (hereafter – N and P, respectively) coastal flux ($\text{kg}\cdot\text{yr}^{-1}$) using the Modular Three-Dimensional Multispecies Transport Model (MT3D-MS) (Zheng and Wang 1999) on a 10 m x 10 m and 50 m x 50 m resolution grid at Hā'ena and Ka'ūpūlehu, respectively (Fig 1D-E). We assigned representative nutrient concentrations to the groundwater recharge based on concentrations measured in the groundwater within our model domains (Table 3). Human derived nutrient flux ($\text{kg}\cdot\text{yr}^{-1}$) were modeled, based on the nutrient loadings of current and projected land cover/use under each scenario (Table 3). We assumed that N is conservative (i.e. there is no decay, transformation, or attenuation) during transport to the ocean discharge points (Engott 2011, Wang et al. 2014). Given P binds to most soils (i.e., sorption process), P concentrations for lawns, green spaces, and the golf course reflected the leachable fraction available to the groundwater (Potter et al. 2006, Soldat and Petrovic 2008) (Table 3). Since wastewater from injection wells and cesspool discharges beneath the soil zone, we assumed P was also conservative (i.e., no sorption) (Glenn et al. 2013). The modeled nutrient concentrations were evaluated against measured nutrient concentrations in the model domains, using linear regression (R^2 and p-value) (Delevaux et al. 2018b).

2.5.2 Land-sea link

To connect the groundwater and coral reef models, a land-sea link was created by sub-dividing the groundwater model domain into 'flow tubes' (~200 m width) ending at pour points along the

shoreline using MODPATH (Fig 1F-G) (Deleaux et al. 2018b). To represent the SGD and map the terrestrial driver grid data (60 m x 60 m) under each scenario, we computed the coastal groundwater discharge ($\text{m}^3.\text{yr}^{-1}$) and nutrient flux ($\text{kg}.\text{yr}^{-1}$) from the nutrient concentrations ($\text{mg}.\text{L}^{-1}$) at the shoreline for each flow tube, using ZONEBUDGET (Harbaugh 1990), and diffused those values from pour points into the coastal zone using GIS distance-based models (Table 4, Fig 1F-G). We assumed that the nutrient chemistry of the SGD was similar to that of the groundwater because the high dissolved oxygen content (dominantly $> 80\%$) in the aquifers around the main Hawaiian Islands results in dominant stable oxidized forms of dissolved N and P (Marion 1998, Fackrell et al. 2016). However it is important to note that biogeochemical conversion of N and P to other species (i.e., denitrification and anaerobic ammonium oxidation) may occur under reducing conditions, which would result in over- and under-estimation of the amount of N and P, respectively, particularly at Ka'ūpūlehu, where wastewater is disposed of through injection wells (Glenn et al. 2013, Fackrell et al. 2016). The modeled nutrient concentrations were evaluated against measured nutrient concentrations in the nearshore area, using linear regression (R^2 and p-value) (Deleaux et al. 2018b).

2.5.3. Coral reef models

The boundaries of the marine model domains comprised the lateral boundaries of the *ahupua'a* to capture the spatial extent of this management unit and the offshore boundary corresponded to the maximum surveyed depth (i.e., 15 m at Hā'ena and 22 m at Ka'ūpūlehu) (Fig 1F-G). The marine drivers (geography, habitat, and wave power) were grid data at 60 m x 60 m derived from bathymetry maps and wave models were coupled with GIS-based models (described in Table 4 & processing methods in Appendix S1: Table S3). The coral reef predictive models used Boosted Regression Trees (BRT) calibrated on local coral reef survey data and generated response curves representing the relationships of each individual environmental driver to each coral reef indicator (Appendix S1: Fig S1) in the R software (R Core Team 2014) using the *dismo* and *raster* packages (Miller 2012, Hijmans 2014, Hijmans et al. 2014). The calibrated models were then used to generate predictive maps of the benthic (% cover) and fish ($\text{kg}.\text{ha}^{-1}$) indicators (60 m x 60 m) in R software (R Core Team 2014) using the *dismo* and *raster* packages (Hijmans 2014, Hijmans et al. 2014). We used the calibration Percent Deviance Explained (PDE) to determine the model fit and cross-validation (CV) PDE to obtain a relative indication of how

accurately the model predicted the observed data, with better performing models providing more confidence in the model predictions, especially within the temporal and spatial coverage of the observed survey data (Costa and Kendall 2016). The predicted values of the benthic and fish grid maps were sampled at the location of each reef survey in ArcGIS and compared to the surveyed coral reef indicators with linear regression (R^2 and p-value), and visually using the same color scale (Delevaux et al. 2018b).

2.6 Scenario modeling

To test the effect of human drivers on coral reef communities, we held the natural drivers constant. We applied this linked land-sea modeling framework to evaluate relative changes in benthic and fish indicators under each scenario compared to present conditions (Fig 2G). Using the groundwater models calibrated on present conditions (see Delevaux et al. [2018] for more details), the nutrient flux discharge was calculated for each coastal development scenario based on the nutrient loading rates associated with each land cover/use type. We then compared the change in the fraction of human-derived N and P flux from the present coastal development to the modeled scenarios at each site. Using the coral reef models calibrated for each site, we predicted and mapped the distribution of the benthic and fish indicators under each scenario. For each benthic and fish indicator, we predicted the effect of the coastal development and climate change scenarios independently and in combination (Table 2). The predictions were then used to produce maps of benthic and fish indicator distributions under each scenario (Elith et al. 2006, Elith et al. 2008).

2.7 Scenario impact assessment

We applied the framework as a decision-support tool using scenario planning to identify coral reef areas vulnerable to nutrient and bleaching impacts, under the proposed coastal development and/or climate change scenarios with the marine closure (Fig 2H). To do so, we calculated differences between predictions of coral reef indicators under the land-use change, climate change, and combined scenarios, compared to present conditions with and without the marine closures (Delevaux et al. 2018a, Stamoulis et al. 2018). We computed the significance of the pairwise differences per grid cell for each coral reef indicator relative to the mean and variance of all differences across the coral reef model domain using the SigDiff function from the R

package SDMTools (Januchowski et al. 2010). The grid cells representing significant differences ($\alpha = 0.10$) were reclassified to indicate where predictions were significantly different than present conditions under each scenario. For the combined scenarios, we overlaid the coral reef areas of significant differences under land-use and climate change scenarios and delineated areas of overlap, where both drivers could potentially interact. These areas were combined into a single map to display the spatial pattern of potential impact per scenario. Finally, the areas of significant differences for each coral reef indicator were used to quantify the relative changes in benthic habitat and fish biomass within those areas. The potential significant fish biomass recovery is summarized for the areas within the marine closures only. For areas where coral bleaching and nutrient discharge overlapped, we summed the changes in abundance of each coral reef indicator. Although interactions between nutrient discharge and bleaching were not explicitly modeled, this approach allowed us to delineate coral reef areas where cumulative impacts and interactions could occur. Excess nutrient runoff is known to promote algae growth, which inhibits recovery from bleaching (i.e., synergistic effect) (Wooldridge 2009, Wooldridge and Done 2009). However, given the limited understanding of these complex processes, we assumed that these effects were additive. Therefore, we applied the precautionary principle and considered these coral reef areas as vulnerable to cumulative impacts (Cooney 2004).

2.8 Spatial prioritization for management

In order to locate the most effective areas to prioritize land-based management (i.e., fertilizers BMPs or wastewater management) to reduce local human impacts on coral reefs, we linked the coral reef areas vulnerable to nutrient runoff to land areas within each *ahupua'a* that contributed the major nutrient load portion to those areas (Fig 2I) (Delevaux et al. 2018a). Using the nutrient export maps from the groundwater models, we identified land areas contributing nutrient discharge to coral reef areas likely to change under land-use and/or climate change scenarios, compared to present conditions. To do so, we linked the coral reef areas showing significant difference to the flow tubes upstream, which contributed the majority (> 66%) of the total nutrient load to those areas in ArcGIS. For climate change scenarios, we identified the linked land areas that contributed the majority (> 66%) of nutrient export to downstream coral reefs under current land-use. For the land-use cover scenarios, we calculated the relative differences in nutrient export in the linked flow tubes, compared to present condition, and selected the flow

tubes delivering the majority of nutrients (>66%). For the combined scenarios, we overlaid the land areas identified as priority for nutrient limitation under future land-use and climate change scenarios. To visually represent these results, the linked land-sea areas were combined into a single map to display where land-based management could help reduce human impacts on coral reefs based on both the land-use and climate change scenarios.

3. Results

3.1 Linked land-sea modeling framework validation

The comparison of the measured and modeled nutrient concentrations in groundwater and nearshore area at Ka‘ūpūlehu, indicated that the N model performed better compared to the P model, while data was insufficient to allow for a similar comparison at Hā‘ena (Appendix S1: Fig S2). The coral reef models’ spatial predictions more strongly correlated to the observed data when the calibration PDE and CV PDE of the model were higher. At Hā‘ena, the calibration PDE and CV PDE of the reef calcifiers, grazer, and scraper models were higher (41-74% and 10-51%, respectively) than the benthic algae, browser, and piscivore models (34-50% and 10-27%, respectively) (Appendix S1: Table S4). Correspondingly, the R^2 was higher for CCA and coral predictions compared to the turf and macroalgae predictions, and the grazer and scraper predictions performed better than the browsers and piscivores (Appendix S1: Fig S3). At Ka‘ūpūlehu, the calibration PDE and CV PDE of the coral, turf, grazer, and scraper models were higher (33-60% and 10-26%, respectively) than the CCA, macroalgae, browser, and piscivore models (22-32% and 5-10%, respectively) (Appendix S1: Table S4). Correspondingly, the R^2 was higher for coral and turf algae predictions compared to CCA and macroalgae predictions, and the scraper and piscivore predictions performed better than the browser and grazer (Appendix S1: Fig S4). All the baseline indicator predictions were significantly similar to the observed data, except for the piscivores at Hā‘ena (Appendix S1: Figs S3 & S4). In addition, those trends were also consistent with their relative empirical abundance and biomass at the survey sites (Appendix S1: Fig S5).

3.2 Coral reef impact assessment and priority management areas at Hā‘ena

Under the low coastal development scenario, the groundwater models indicated an additional 298 kg.yr⁻¹ of N and 62 kg.yr⁻¹ of P discharging at the shoreline, resulting in an increase of marine dissolved N and P behind Makua reef (Fig 3A-C). Consequently, the coral reef models predicted a statistically significant increase in benthic algae cover over 32 ha (4.2% modeled area), with an average gain of 0.2% macroalgae and 0.5% turf algae cover (Table 5, Fig 4A). Under the high coastal development scenario, the groundwater models indicated an additional 1,400 kg.yr⁻¹ of N and 703 kg.yr⁻¹ of P discharging at the shoreline, resulting in an increase of marine dissolved N and P downstream from the added development behind Makua reef (Fig 3D-F). Consequently, the coral reef models predicted a statistically significant increase in benthic algae cover over 58 ha (7.6% modeled area), with an average gain of 0.2% macroalgae and 0.6% turf algae cover (Table 5, Fig 4B). When adding the effects of the marine closure under the current coastal development, the coral reef fish models predicted a total increase in biomass of 34 kg in the marine closure (Table 6). When combining the effects of the marine closure with the coastal development scenarios, the coral reef fish models predicted an increase of 62-76 kg in fish biomass, due to increase in grazer biomass and a loss in browse and scraper biomass (Table 6). Based on the significant change of coral reef areas and the location of land areas contributing the largest human-derived nutrients under the low and high coastal development scenarios, two and four areas were identified as priority areas, respectively, for wastewater management to indirectly improve the conditions of coral reefs downstream (see pink zones in Fig 4E-F). The main sources of human-derived nutrients were septic tanks of houses under the new proposed development.

Under the climate change scenarios (low and high) and present land-use scenario, the coral reef models predicted statistically significant coral cover loss over 98-101 ha (12.8-13.2% modeled area), with an average loss ranging from 2.5-3.8 % coral cover (Table 5). The potential for bleaching impacts is higher around the back-reef of Makua (Fig 4C). When combining the effects of the marine closure and the climate change scenarios, the coral reef fish models predicted a fish biomass loss of 1,217-2,394 kg, due to a loss in grazer biomass (Table 6, Fig 4G). Based on the overlap of coral reef areas vulnerable to bleaching impacts and high levels of nutrients, and the locations of the linked land areas contributing the greatest amount of human-derived nutrients under present land-use, two land areas were identified as priority areas for

cesspool upgrades to indirectly improve the conditions of coral reefs downstream (see purple zones in Fig 4G). Under current land-use, the main sources of human-derived nutrients were cesspools of existing houses.

Under the combined coastal development and coral bleaching scenarios, the coral reefs most vulnerable to both nutrients and bleaching spatially overlap and were primarily located around the back-reef of Makua (Fig 4D). The coral reef models predicted a statistically significant change in habitat over 114-129 ha (14.9-16.9% modeled area), due to an increase in macroalgae and turf algae cover and loss in coral cover (Table 5). When adding the effects of the marine closure to the combined coastal development and coral bleaching scenarios, the coral reef fish models predicted 1,222-2,375 kg fish biomass loss (Table 6). By combining the main present and future sources of human-derived nutrients (i.e., cesspools from existing houses and septic tanks from future houses), wastewater management priority areas that can indirectly improve the conditions of coral reefs downstream were identified (see pink and purple zones in Fig 4H).

3.3 Coral reef impact assessment and priority management areas at Ka‘ūpūlehu

Under the low coastal development scenario, the groundwater models indicated an additional 741 kg.yr⁻¹ of N and 1,050 kg.yr⁻¹ of P discharging at the shoreline, resulting in an increase of marine dissolved N and P downstream from the added injection well to the north coastal area of the *ahupua‘a* (Fig 3G-I). Consequently, the coral reef models predicted a statistically significant change in habitat over 18 ha (5.7% modeled area), due to turf algae cover increase and a decrease in CCA (Table 5, Fig 5A). Under the high coastal development scenario, the groundwater models indicated an additional 7,704 kg.yr⁻¹ of N and 258 kg.yr⁻¹ of P discharging at the shoreline, resulting in an increase of marine dissolved N and P downstream from the added development to the north of the *ahupua‘a* (Fig 3J-L). Consequently, the coral reef models predicted a statistically significant change in habitat over 27 ha (7.9% modeled area), due to an increase in turf algae cover and a decrease in CCA cover (Table 5, Fig 5B). When combining the effects of the marine closure under the current coastal development, the coral reef fish models predicted a total increase in biomass of 692 kg, averaging 0.2 g.m⁻² in the marine closure (Table 6). When combining the effects of the marine closure to the coastal development scenarios, the coral reef fish models predicted a smaller increase of 666-674 kg biomass, due to a decrease in

browser biomass (Table 6). Note that the piscivores group does not change across all the coastal development scenarios because it does not respond to nutrients (Appendix S1: Fig S1). Based on the coral reef areas showing significant change and the land areas contributing the largest human-derived nutrients under the high coastal development scenarios, one area was identified as a priority for wastewater management to indirectly improve the conditions of the coral reefs downstream (see pink zones in Fig 5E-F). The main sources of human-derived nutrients were fertilizers applied to landscaped green spaces and septic tanks or the wastewater injection well for houses under the new proposed development.

Under the climate change scenarios (low and high) and present land-use scenario, the coral reef models predicted a statistically significant coral cover loss over 36-37 ha (11.4-11.7% modeled area), with an average loss ranging from 3.9-5.1 % coral cover (Table 5). The potential for bleaching impacts were higher along the shallow reef flats near to the south (Fig 4C). When adding the effects of the marine closure to the climate change scenarios, the coral reef fish models predicted a fish biomass loss of 102-309 kg, due to a loss in grazer and scraper biomass (Table 6). Note that the piscivores group does not change across all the coastal development scenarios because it does not respond to coral cover (see Appendix S1: Fig S1). Based on the overlap of coral reef areas vulnerable to bleaching impacts and high levels of nutrients and the location of land areas contributing the largest amount of human-derived nutrients under present land-use, three areas were identified as priority areas for fertilizer BMPs and wastewater management to indirectly improve the conditions of coral reef downstream (see purple zones in Fig 5G). Under current land-use, the main sources of human-derived nutrients were the golf course, lawns of existing houses, and the wastewater injection well.

Under the combined coastal development and coral bleaching scenarios, the coral reefs most vulnerable to nutrients and bleaching do not spatially overlap, but both affected areas are located along the shallow reef flats, downstream from proposed development (Fig 5D). The coral reef models predicted a statistically significant change in habitat of 57-64 ha (17.0-18.9% modeled area), due to a decrease in coral and CCA cover combined with an increase of turf algae cover (Table 5). When adding the effects of the marine closure to the combined coastal development and coral bleaching scenarios, the coral reef fish models predicted a loss of 120-335 kg in fish

biomass (Table 6). By combining the primary present and future sources of human-derived nutrients (i.e., the wastewater injection well from existing resorts, fertilizers for golf course, and septic tanks from future houses), wastewater management priority areas that can indirectly improve the conditions of coral reefs downstream were identified (see pink and purple zones in Fig 5H).

4. Discussion

We coupled the linked land-sea modeling framework with scenario planning and used this tool in partnership with Hā'ena and Ka'ūpūlehu communities to inform local management actions and identify priority areas on land that can reduce human impacts on coral reefs. Both communities were interested in restoring a ridge-to-reef approach to address contemporary environmental issues, including coastal development and fishing pressure impacts on coral reefs combined with bleaching from climate change. Due to their location at the opposite ends of the main Hawaiian Islands, Hā'ena is geologically old and has been exposed to and eroded by heavy rains and very large open oceanic swells, while Ka'ūpūlehu is young, dry, and sheltered from waves (Delevaux et al. 2018b). Given these inherent differences, local management actions relevant to each place are likely to differ. We located coral reefs vulnerable to local and global human stressors and linked them to areas on land where limiting sources of human-derived nutrients could prevent increases in benthic algae and promote chances of coral recovery from bleaching.

4.1 Coral reef impacts from coastal development and climate change

Our scenario analysis indicated that coral reefs sheltered from wave power are more vulnerable to bleaching impacts and coastal development, like the shallow reef flats of Ka'ūpūlehu and the back-reef of Makua, due to limited mixing of land-based nutrients, shallow depth, and exposure to higher SST (Bridge et al. 2013). Although the total area affected was lower at Ka'ūpūlehu than Hā'ena partly due to the smaller modeled area (Table 1), coral cover loss was higher on average at Ka'ūpūlehu due to more favorable wave conditions for coral growth that have enabled currently high coral cover. Conversely, the reef slopes of Hā'ena appear less susceptible to coral bleaching effects due to lower coral cover and higher CCA cover resulting from high wave exposure (Delevaux et al. 2018b). Similarly, coral reefs exposed to high rainfall and wave power

seem more resilient to coastal development impacts due to increased dilution and mixing, like the reef areas located between Makua and Pu‘ukahua reefs at Hā‘ena (Fig 6). This is also due to land zoning, which determines the location and intensity of feasible development and differs among places, having indirect impacts on coral reefs downstream. Projected increases in coastal development led to an increase in benthic algae at both sites and a loss of reef calcifiers at Ka‘ūpūlehu (Table 5). Managing wastewater in priority land areas is an important management strategy to maintain good water quality essential for healthy coral reefs nearshore and mitigate the impacts of climate change on these reefs (McCook et al. 2001, Szmant 2002).

When combining the impacts of coastal development and climate change, the predicted impacts on coral reefs worsened at both sites with a higher loss of reef calcifiers and an increase in benthic algae. Coral reef areas vulnerable to both do not spatially overlap at Ka‘ūpūlehu, while the back-reef of Makua at Hā‘ena is vulnerable to both coastal development and climate change due to limited wave mixing and proximity to coastal development. The findings for Ka‘ūpūlehu are consistent with past research conducted at the regional scale, which showed that wave-sheltered shores of Main Hawaiian Islands are more vulnerable to land-based source pollution, while wave-exposed shores are more resilient due to higher flushing (Rodgers et al. 2012). However, our spatially explicit models also revealed that coral reefs in these areas are also vulnerable at the local scale. This has important implications for future development in these areas and implies that accounting for finer spatial patterns is necessary to reduce impacts on coral reefs. The fine spatial scale of our models allowed us to discover the spatial nuances within both sites, which revealed the need for different marine and terrestrial management actions.

This suggests that increases in coastal development under projected climate change can reduce coral recovery after bleaching events through increased competition for space with algae. This is particularly true of the back-reef of Makua given the overlapping impacts of coastal development and coral bleaching in a habitat protected from waves and thus suitable for coral growth (Fig 4D). Although coral reef recovery from bleaching was not modeled, our results showed that nutrient enriched waters can promote benthic algae and hinder reef calcifiers, as reported elsewhere (Littler et al. 2006, Smith et al. 2010, Vermeij et al. 2010). As shown by Smith et al. (2016) turf algae has an advantage over corals and can be a fast colonizer in favorable conditions

at both sites, regardless of wave exposure. Other research has shown similar trends for benthic indicators under increased nutrients, which can lead to increases in grazing fish species (Gurney et al. 2013), as found in Hā'ena. Conversely, most herbivore biomass decreased at Ka'ūpūlehu due to higher dependence on reef calcifiers (Delevaux et al. 2018b). Therefore, adopting land use practices that reduce exposure of coral reefs to land-based nutrients minimizes turf and macroalgae growth providing space for coral recruits. This promotes coral recovery post-bleaching events, especially in dry regions or shallow back-reef areas with limited water circulation (Fabricius 2011).

4.2 Place-based management actions for coral reefs under climate change

As shown in a meta-analysis by Cinner et al. (2016), our results indicate that marine closures can reduce human impacts on coral reef by increasing fish biomass within the marine closures, with a smaller change at Hā'ena due to the smaller reserve area (Fig 1B-C) (Halpern 2003, Delaney et al. 2017). However, the Makua refuge is strategically located in a prime nursery habitat (Goodell et al. 2018), which can supplement adjacent reef through spillover (Stamoulis and Friedlander 2013). Because larger fishes are more susceptible to fishing impacts, biomass of piscivores, followed by grazers and scrapers, experienced the largest increase at both sites (Jennings et al. 1998, Jennings et al. 1999). The protection of herbivores can potentially compensate for some of the lost competitive ability of calcifying organisms over benthic algae (Adam et al. 2015) under coral bleaching and coastal development (Littler et al. 2006, Smith et al. 2010). By consuming benthic algae, herbivores free space for CCA and coral larvae recruitment (Bellwood et al. 2004, Green and Bellwood 2009), which is important for recovery from bleaching impacts. These results show that removal of fishing can offset impacts from coastal development and coral bleaching on the targeted fish community. Although the impacts of climate change and/or coastal development are significantly reduced when coupled with the marine closure, our results indicate that land-based impacts or bleaching impacts can persist within the boundaries of the closure at Ka'ūpūlehu and outside the boundaries of the closure at Hā'ena. As shown by Halpern et al. (2013), these spatial trends imply that marine closures are not always capable of addressing human drivers that impact the benthic community of coral reefs, which reinforces the need for ridge-to-reef management.

We also showed that limiting human-derived sources of nutrients in areas linked to coral reefs limits coral exposure to nutrient discharge and may reduce vulnerability to coral bleaching (McLeod et al. 2019). For instance, the wastewater injection well resulted in less human-derived nutrient discharge and a smaller impact footprint compared to septic systems for the same amount of development. This demonstrates the importance of selecting appropriate wastewater treatment technology while accounting for the specifics of the place, such as geological history and rainfall (Selvarajah et al. 1994). Furthermore, recent research found that coral reefs managed with high local community engagement and dependence on marine resources have a better chance of withstanding bleaching impacts related to climate change (Cinner et al. 2016). Both communities strategically designed their marine closures given the wave disturbance regime and land-based influence at each place. Makua back reef was designated a marine refuge where corals are able to grow and provide fish nursery habitat, safe from the powerful winter swells (Goodell 2015). The closed area at Makua is rather small due to compromise with competitive stakeholders' interests. However, these results highlight that targeted terrestrial management can benefit both places, particularly where marine closures may fall short, such as beyond the boundaries of the marine closure at Hā'ena. In this way, these findings reinforce that place-based ridge-to-reef management can reduce human impacts on coral reefs in the face of climate change.

4.3 Modeling assumptions and caveats

Our models were calibrated on contemporary conditions, using the latest data publicly available for our study sites at the time of modeling, which were from 2012-2014 prior to a large bleaching event that severely affected Ka'ūpūlehu (Kramer et al. 2016, Maynard 2016). Based on a number of assumptions, they can be used to forecast biological indicator distributions at a different point in time (Franklin 2010). A key assumption with predicting coral reef futures in our modeling approach is that species distributions are in equilibrium with current conditions and the identified relationships will remain constant over time (DeAngelis and Waterhouse 1987, Franklin 2010), which may not always be true (Carpenter 2002). For example, emerging evidence shows that corals may acclimatize to predicted increases in SST associated with climate change (Baker et al. 2008), in which case, these results are conservative. Static modeling approaches also do not account for species dispersal, migration, and interactions within the seascape, which can

influence management scale and outcomes (Guisan 2005, Stamoulis and Delevaux 2015). Given imperfect knowledge of both the effects of human drivers and how coral reefs respond to these drivers (Coreau et al. 2009), scenario modeling requires simplifications and assumptions, which lead to uncertainty in model projections. By using present conditions as the baseline for examining projected coral reefs, the comparative benchmark does not represent pristine ecosystems (Knowlton and Jackson 2008). Nevertheless, while sources of uncertainty in scenario analysis are inevitable, present conditions still provide an opportunity to identify the trajectory of coral reefs under different human drivers and provide guidance for management (Alagona et al. 2012). We used scenario modeling to illustrate the range of possibilities for the future of coral reefs and identify local management actions that can reduce the risk of impacts on coral reefs and associated marine resources (Coreau et al. 2009).

Although we did not account for the potential impacts of climate related changes in rainfall patterns in this study, climate change is predicted to impact oceanic islands through change in rainfall patterns and intensity (Giambelluca et al. 2012, Elison Timm et al. 2015). Rainfall is a key driver of groundwater input and groundwater runoff, likely to influence salinity, levels of nearshore nutrient input, and the associated benthic/fish impacts presented here. While the islands are expected to experience a greater contrast between the wet and dry regions in the future, rainfall is generally predicted to decrease over the Hawaiian Islands based on statistical downscaled modeling of the CMIP5 global model projections (Elison Timm et al. 2015). At both sites, climate change-related reductions in precipitation and increased temperatures are projected to further reduce groundwater recharge, increase nutrient concentration, and foster benthic algae growth in the nearshore areas (Elison Timm et al. 2015, Bremer et al. 2018).

Due to limited or lack of data, some models in our framework could not be fully validated. For example at Hā'ena, the groundwater model was parameterized with limited groundwater samples and the coastal discharge models could not be validated due to lack of coastal water quality data (Delevaux et al. 2018b). At Ka'ūpūlehu, the coastal discharge models were partially ground-truthed based on limited coastal water quality data (Delevaux et al. 2018b). In addition, we used existing wave data to represent mixing effects on the groundwater discharge, given circulation data was not available for our study sites (Delevaux et al. 2018b). Furthermore, species

composition and relative abundance can affect the coral reef model performance and predictability of selected indicators (Pittman et al. 2009), as shown by the low performance of the macroalgae model at Ka‘ūpūlehu. It is also important to note that models with high-unexplained variance coupled to datasets with high error, like the downscaled wave model data in the shallow nearshore areas, can propagate across the models, which may influence the precision and accuracy of the derived ecological relationships and the reliability of spatial predictions. However, the ecological relationships derived from these models were consistent with other local and global studies (Dollar 1982, Friedlander et al. 2003, Littler et al. 2006, Stamoulis et al. 2018), providing confidence in the relative accuracy of our models (Delevaux et al. 2018b). Furthermore, the areas we identified as vulnerable coincided with local observations from community members, providing additional confidence to our findings. In light of these caveats, the priority areas identified should be seen as target zones for wastewater management and further investigation of land-sea impacts (Delevaux et al. 2018b).

We tested how sensitive our modeling framework was to the linkages between the groundwater and the coral reef models by running the framework with various values of nutrient loading rates for the key land-uses that change across scenarios. We observed that the magnitude of change and the sizes of the spatial footprints of coral reef impacts as a function of nutrient discharge varied depending on the nutrient loading rates used. Though the relative changes between scenarios were small, partly because the coral reef model predictions tend to underestimate means of empirically measured values. The directionality of change in the coral reef indicators and the locations of coral reef vulnerable to nutrient discharge were consistently linked to the land areas that contributed the largest change in nutrient export, where wave flushing was minimal, and waters are shallow. Therefore, the priority areas on land linked to vulnerable coral reef areas did not change. Environmental decision theory has also established that uncertainties in the input parameters can alter predictions but do not change the relative priority of management options (McCarthy et al. 2003, Bode et al. 2008, Saunders et al. 2017). Our approach may under-estimate the magnitude of change but is sensitive to local-scale changes to describe ecological relationships at a fine spatial scale (Franklin 2010, Januchowski et al. 2010). Therefore, we concluded that this framework can reliably identify priority areas to undertake further field investigations.

It is increasingly recognized that fishing pressure and water quality interact with elevated SST and have a profound influence on management outcomes of nearshore coral reefs under climate change (Pratchett et al. 2008, Smith et al. 2016). Thus, better understanding where the cumulative effect of these human drivers is less than (antagonism), more than (synergism), or additive can help place-based management (Hughes and Connell 1999, Bruno 2016, Cinner et al. 2016). This research identified where potential fish recovery within closures and coral reef areas may be subject to either bleaching, nutrient exposure, or both, but did not explicitly model their potential interactions and cumulative impacts due to the poor understanding of those processes (Anthony 2006, Wenger et al. 2015). However, it is likely that nutrient enrichment would directly exacerbate bleaching rates through increased disease (Bruno et al. 2007). In addition, nutrients can contribute to lack of recovery from bleaching through promotion of algae growth (Wooldridge 2009, Wooldridge and Done 2009). On the other hand, work has shown that protecting herbivores can reduce impacts from increased SST by mitigating bleaching impacts through healthy grazing fish population that reduce algae cover (Littler et al. 2006, Smith et al. 2010). Therefore, the cumulative impacts of fishing and nutrients under elevated SST are challenging to model explicitly.

5. Conclusion

These findings provide three important management implications for coral reefs in a changing climate. First, effective management requires careful consideration of the combined impact of multiple human and natural drivers and potential interactions between them (e.g., (Nyström et al. 2000, Hughes et al. 2007)). Our results reveal that coral reefs can be more resilient to land-based sources of pollution and climate change in certain areas, depending on their natural disturbance regimes and local habitat geography. Second, local place-based management that reduces fishing and coastal development impacts, can protect important ecological functions and improve habitat quality, which in turn can increase chances of withstanding impacts from climate change (Gurney et al. 2013, Maina et al. 2013, Delevaux et al. 2018b). Therefore, this research supports the paradigm that managing local-scale human drivers fosters coral reef health in the face of climate change (Hoegh-Guldberg et al. 2007, Pandolfi et al. 2011). Lastly, this framework can

inform place-based management at spatial scales relevant to small oceanic islands. These models helped us identify where and how human stressors operating at different scales can impact coral reefs, and identify where on land local management can target sources of human derived-nutrients to best benefit coral reefs. We illustrate the utility of local-scale models in supporting local and place-based management to reduce human impacts on coral reefs on small high oceanic islands exposed to gradients of wave and rainfall disturbance regimes.

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Data Availability

Habitat and wave predictor layers are available online at <https://data.nodc.noaa.gov/cgi-bin/iso?id=gov.noaa.nodc:0155189>. Baseline data is available at Figshare (<https://doi.org/10.6084/m9.figshare.5877396>). Scenarios and associated data are available at: Figshare (<https://doi.org/10.6084/m9.figshare.7768562>).

Tables:

Table 1. Study sites attributes of Hā'ena and Ka'ūpūlehu.

Attributes	Hā'ena	Ka'ūpūlehu
Island age (MYA)	5.1	0.8
<i>Ahupua'a</i> size (km ²)	7.3	104
Maximum elevation (m)	1,006 (Ali'inui Peak)	2,518 (Hualālai Mountain)
Annual rainfall (mm.year ⁻¹)	High (4,040) ¹	Low (260-1,350) ²
Perennial streams	2	0
Coastline length (km)	4	7.4
Groundwater recharge (mg.L ⁻¹) ³	0.11 to 4.97	0.04 to 0.69
Groundwater discharge (m ³ .yr ⁻¹ or m ³ .m ⁻¹ .yr ⁻¹) ³	57.1 million or 10,279	22.7 million or 3,085
Background N (mg.L ⁻¹) ³	0.5-0.85	0.25-2.70
Background P (mg.L ⁻¹) ³	0.09-0.20	0.10-0.20
N flux (kg.yr ⁻¹ or kg.yr ⁻¹ .m ⁻¹) ³	36,320 or 6.0	55,540 or 7.1
P flux (kg.yr ⁻¹ or kg.yr ⁻¹ .m ⁻¹) ³	13,050 or 2.2	6,760 or 0.8
Fraction human-derived N (%) ³	16.4	10.7

Fraction human-derived P (%) ³	31.7	34.9
Modeled reef area (km ²)	7.6	3.2
Forereef dominant benthic substrate	Crustose coralline algae (CCA)	Coral
Mean total resource fish biomass (g.m ⁻²) ⁴	7.35	4.53
Management regime (year established)	Community Based Subsistence Fisheries Management Area (2016)	Community-led 10-year fishing rest period (2016)
Marine closure area (ha)	7.1	355.4
Current coastal development	136 private residences	193 private residences 2 large luxury resorts 1 golf course
CPUE for line, net, and spear (kg.hr ⁻¹) ⁵	0.09, 0.43, and 0.56	0.23, 0.39, and 0.51
Key land owners	State of Hawai‘i A non-profit organization (National Tropical	Private land owner (Kamehameha Schools)

¹ Value derived from (Calhoun and Fletcher 1999)

² Values derived from (Izuka et al. 2016)

³ Modeled values derived from (Delevaux et al. 2018)

⁴ Empirical survey mean values derived from (Delevaux et al. 2018)

⁵ Catch Per Unit Effort (CPUE) derived from (Delaney et al. 2017)

Table 2. Scenarios at Hā‘ena and Ka‘ūpūlehu. Low and high coastal development in terms of added houses and associated wastewater systems and lawns, coral bleaching in terms of coral cover loss, and the marine closure areas at both sites. A description of the combined scenarios is also presented for each site.

Scenarios	Hā‘ena	Ka‘ūpūlehu
Low development	+20 houses (20 septic tanks) +1.3 ha lawns	+83 houses (1 injection well) +13 ha green space
High development	+73 houses (73 septic tanks)	+83 houses (83 septic tanks)

	+ 5 ha lawns	+13 ha green space
Low bleaching	-30% x current coral cover 0-5m	-30% x current coral cover 0-5m
High bleaching	-50% and -25% x current coral cover 0-5m and 5-10m	-50% and -25% x current coral cover 0-5m and 5-10m
Marine closure	7.1 ha	355.4 ha
Combined low scenarios	Low bleaching x development x closure	Low bleaching x development x closure
Combined high scenarios	High bleaching x development x closure	High bleaching x development x closure

Table 3. Annual natural and anthropogenic nutrient loading rates per proposed land cover/use type. Adapted from Delevaux et al. (2018).

Sources and zones	[N] (mg.L ⁻¹)	[P] (mg.L ⁻¹)	N flux (kg.yr ⁻¹)	P flux (kg.yr ⁻¹)
Natural (background)				
Hā'ena	0.50	0.20	7.51.ha ⁻¹	3.00.ha ⁻¹
Ka'ūpūlehu Upland	2.70	0.20	8.55.ha ⁻¹	0.63.ha ⁻¹
Ka'ūpūlehu Lowland	0.25	0.10	0.65.ha ⁻¹	0.26.ha ⁻¹
Keauhou Upland	1.20	0.15	3.11.ha ⁻¹	0.26.ha ⁻¹
Keauhou Lowland	0.25	0.1	0.72.ha ⁻¹	0.29.ha ⁻¹
Anthropogenic				
Cesspool ^{a,b}	87	19	38	8.3
Septic system ^{a,c}	34.2	1.2	14.9	5.2
Current injection well ^d	5.25	6.8	843	1300
New injection well ^e	5.25	6.8	460	596
Lawn ^f	0.20	0.01	4.5.ha ⁻¹	0.2.ha ⁻¹
Golf course & green spaces ^g	7.59	0.54	12.ha ⁻¹	6.1.ha ⁻¹

Groundwater zones were assigned N and P recharge concentrations with the corresponding flux (i.e., concentration x annual recharge).

^a Each land parcel assumed a residential unit with three bedrooms at an occupancy rate of 1.5 persons per bedroom generating 435 m³.yr⁻¹ of wastewater (Whittier and El-Kadi 2014)

^b The nutrient loading rates were based on sampling conducted on Maui (HDOH 2017)

^c The nutrient loading rates were based sampling conducted on Hawai'i Island (Tasato and Dugan 1980)

^d Wastewater discharge is 160,600 m³.yr⁻¹ according to the State of Hawaii Injection Permit database (State of Hawaii 2003)

^e Wastewater discharge is 87,600 m³.yr⁻¹ according to the State of Hawaii Injection Permit database (State of Hawaii 2003)

^f Assuming a recharge rate of 50 m³.ha.d⁻¹ and a fertilization rate of 192 kg.ha⁻¹ of N (Wang et al. 2014)

^g Golf course and green spaces irrigation rate were assumed at 50 m³.ha.d⁻¹ 22(CH2MHill 2013), fertilization rates at 236 kg.ha⁻¹ of N and 122 kg.ha⁻¹ of P, and a leaching rate of 5% for both nutrients (Throssell et al. 2009)

Table 4. Description of the key terrestrial and marine drivers of coral reefs. Refer to Appendix S1: Table S3 and Delevaux et al. (2018) for more details on processing methods.

Type	Indicator	Metrics	Description	Unit
Terrestrial drivers	Groundwater discharge	Freshwater ¹	Volume of freshwater discharged yearly	m ³ .yr ⁻¹
		N ¹	Mass of dissolved nitrogen discharged yearly.	kg.yr ⁻¹
		P ¹	Mass of dissolved phosphorus discharged yearly.	kg.yr ⁻¹
Marine drivers	Wave	Power ²	Mean wave power derived from a 10 year (2000-2009) SWAN hindcast wave model.	kW.m ⁻¹
	Geography	Depth ³	Mean seafloor depth	m
		Distance to shore ⁴	Euclidean distance to the shoreline	m

Habitat topography	BPI ³	Relative topographic position of a point based its elevation and the mean elevation within a neighborhood (m)	m
	Slope ³	Maximum rate of change in seafloor depth between each grid cell and its neighbors	Degree
Habitat complexity	Planar curvature ³	Seafloor curvature perpendicular to the direction of the maximum slope (mean). Value indicates whether flow will converge or diverge over a point.	Radians.m ⁻¹
	Profile curvature ³	Seafloor curvature in the direction of the maximum slope (mean). Value indicates whether flow will accelerate or decelerate over the curve.	Radians.m ⁻¹
	Rugosity ³	Measure of small-scale variations of amplitude in the height of a surface (mean). Value range from 1 (flat) to infinity.	Unitless
Habitat exposure	Aspect ³	Downslope direction of maximum rate of change in seafloor depth between each grid cell and its neighbors (standard deviation, sine and cosine circular mean)	Degree

¹ Groundwater models (Delevaux et al. 2018)

² SWAN hindcast wave model at 500m native resolution (Stopa et al. 2013)

³ Bathymetry synthesis at 5m native resolution (HMRG 2015)

⁴ Coastline (OP 2000)

⁵ Coral reef models (Delevaux et al. 2018)

Table 5. Benthic coral reef impact assessment at Hā‘ena and Ka‘ūpūlehu. Habitat area (ha), average % cover, and % of modeled area predicted to significantly differ relative to present conditions are reported for the benthic indicators per scenario. The total column shows the total area (ha) and % of modeled area showing significant change accounting for spatial overlap amongst indicators, at each site. (Refer to Appendix S1: Fig S6 for more details on relative change by benthic indicator).

Hā‘ena	Ka‘ūpūlehu
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Benthic indicators	Units	CCA	COR	MAC	TUR	Total	CCA	COR	MAC	TUR	Total
Low development	ha	0	0	20	17	32	7	0	0	14	18
	% cover	0.0	0.0	0.2	0.5	-	-0.1	0.0	0.0	2.7	-
	% area	0.0	0.0	2.6	2.3	4.2	2.0	0.0	0.0	4.5	5.7
High development	ha	0	0	32	31	58	12	0	0	15	25
	% cover	0.0	0.0	0.2	0.6	-	-0.8	0.0	0.0	2.6	-
	% area	0.0	0.0	4.2	4.1	7.6	3.9	0.0	0.0	4.7	7.9
Low bleaching	ha	0	98	0	0	98	0	36	0	0	36
	% cover	0.0	-2.5	0.0	0.0	-	0.0	-3.9	0.0	0.0	-
	% area	0.0	12.8	0.0	0.0	12.8	0.0	11.4	0.0	0.0	11.4
High bleaching	ha	0	101	0	0	98	0	37	0	0	37
	% cover	0.0	-3.8	0.0	0.0	-	0.0	-5.1	0.0	0.0	-
	% area	0.0	13.2	0.0	0.0	13.2	0.0	11.6	0.0	0.0	11.7
Low bleaching x development	ha	0	98	20	17	114	-7	36	0	14	57
	% cover	0.0	-2.5	+0.1	0.5	-	-0.1	-3.9	0.0	2.7	-
	% area	0.0	12.8	2.6	2.3	14.9	2.0	11.4	0.0	4.5	17.0
High bleaching x development	ha	0	101	32	31	129	-12	37	0	15	64
	% cover	0.0	-3.8	+0.1	0.6	-	-0.8	-5.1	0.0	2.6	-
	% area	0.0	13.2	4.2	4.1	16.9	3.9	11.6	0.0	4.7	18.9

1209

1210

1211 **Table 6. Fish coral reef impact assessment at Hā'ena and Ka'ūpūlehu.** The total change in fish biomass (kg), average biomass
1212 change (g/m²), and percent change (%) relative to present conditions are reported for the fish indicators per scenario, including the
1213 effects of marine closures for all the scenarios. The total column shows the net change in fish biomass (kg) and the % change relative
1214 the total fish biomass prior to marine closure. (Refer to Appendix S1: Figs S7 & S8 for more details on relative change by fish
1215 indicator).

Fish indicators	Units	Hā'ena					Ka'ūpūlehu				
		BROW	GRAZ	SCRIP	PISC	Total	BROW	GRAZ	SCRIP	PISC	Total
Closure	kg	<0.1	17	<0.1	17	34	3	125	42	521	692
	g/m ²	<0.1	0.2	<0.1	0.2	-	<0.1	0.1	<0.1	0.5	-
	%	0.0	0.1	0.0	19.8	0.2	0.3	0.4	0.3	98.4	1.5
Low development	kg	-4	50	-1	17	62	-14	125	42	521	674
	g/m ²	<-0.1	0.1	<-0.1	0.2	-	<-0.1	0.1	<0.1	0.5	-
	%	-0.6	0.3	-0.1	19.7	0.3	-1.1	0.4	0.3	98.4	1.5
High development	kg	-7	70	-3	15	76	-22	125	42	521	666
	g/m ²	<-0.1	0.2	<-0.1	0.1	-	<-0.1	0.1	<0.1	0.5	-
	%	-1.1	0.4	-0.2	17.8	0.4	-1.7	0.4	0.3	98.4	1.5
Low bleaching	kg	0	-1234	0	17	-1217	3	-355	-272	521	-102
	g/m ²	0.0	-0.7	0.0	0.2	-	<0.1	-1.5	-0.4	0.5	-
	%	0.0	-8.6	0.0	19.8	-7.4	0.3	-1.2	-2.2	98.4	-0.5
High bleaching	kg	0	-2411	0	17	-2394	3	-548	-286	521	-309
	g/m ²	0.0	-2.4	0.0	0.2	-	<0.1	-1.6	-0.5	0.5	-
	%	0.0	-18.4	0.0	19.8	-15.7	0.3	-1.8	-2.4	98.4	-0.7
Low bleaching x	kg	-4	-1226	-1	17	-1222	-14	-355	-272	521	-120

development											
	g/m ²	<-0.1	-0.7	<-0.1	0.2	-	<-0.1	-1.5	-0.4	0.5	-
	%	-0.6	-8.6	-0.1	19.7	-7.4	-1.1	-1.2	-2.2	98.4	-0.3
High bleaching x development 1											
	3	-7	-2380	-3	15	-1996	-22	-548	-286	521	-335
	g/m ²	<-0.1	-2.4	<-0.1	0.1	-	<-0.1	-1.6	-0.5	0.5	-
	%	-1.1	-18.1	-0.2	17.8	-15.5	-1.7	-1.8	-2.4	98.4	-0.8

Figures:

Figure 1. Study sites location. (A) Location of study sites on Kauaʻi and Hawaiʻi along the main Hawaiian Island chain, with island age and the direction of the prevailing north-east tradewinds and ocean swell indicated. Current land use/cover and marine closures are shown for (B) Hāʻena and (C) Kaʻūpūlehu. Groundwater modeling domain and groundwater sample locations for (D) Hāʻena and (E) Kaʻūpūlehu. Coral reef modeling domain and coral reef survey locations for (F) Hāʻena and (G) Kaʻūpūlehu. Adapted from Delevaux et al. (2018).

Figure 2. Modeling framework. Human driver scenarios, including (A) coastal development, (B) marine closure, and (C) coral bleaching were designed for each study site (The Integration and Application Network 2016). (D) Nitrogen and phosphorus fluxes were modeled using calibrated groundwater models under each coastal development scenario. (E) Marine drivers were derived from the SWAN wave model and LiDAR bathymetry data. (F) Coral reef models calibrated for each site were used to predict the change in coral reef indicators distribution under each scenario. (G) Scenario modeling resulted in maps of coral reef indicators distribution under each scenario. The scenario analysis comprised (H) an impact assessment to identify coral reef areas impacted under the scenario considered and (I) a spatial prioritization to identify priority management areas on land. Adapted from Delevaux et al. (2018).

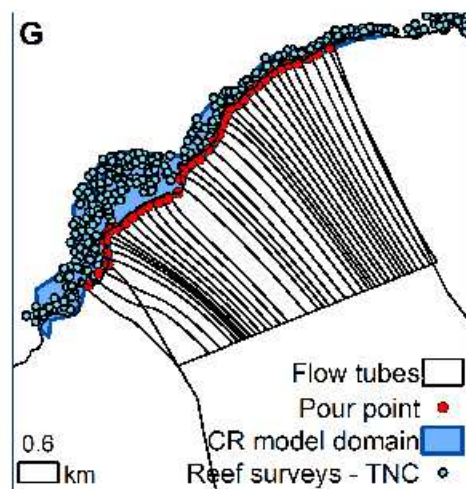
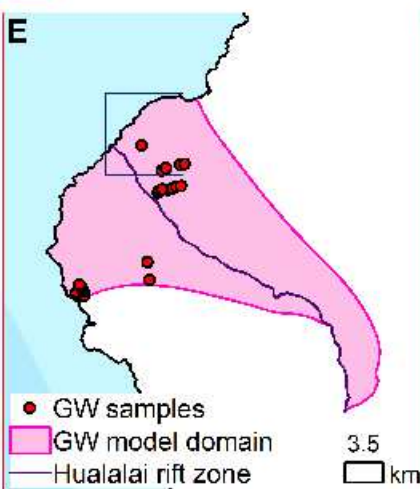
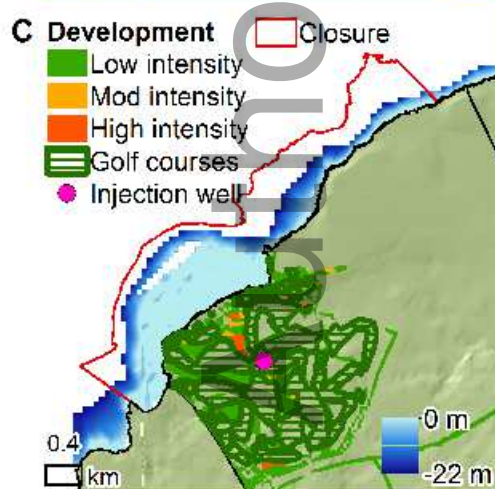
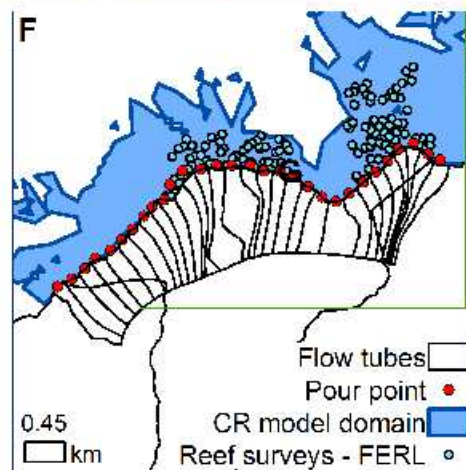
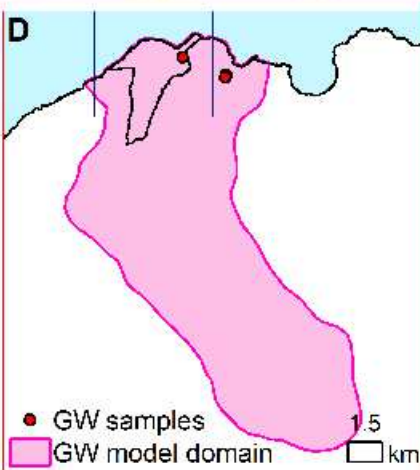
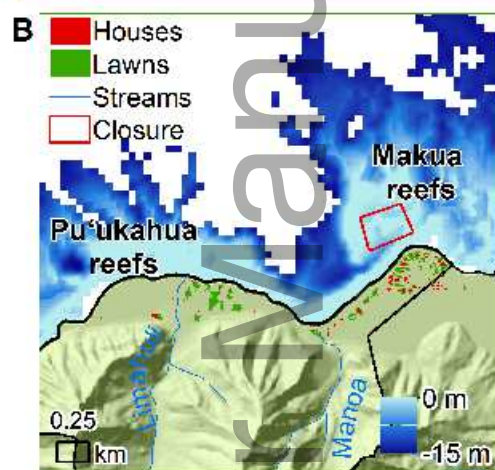
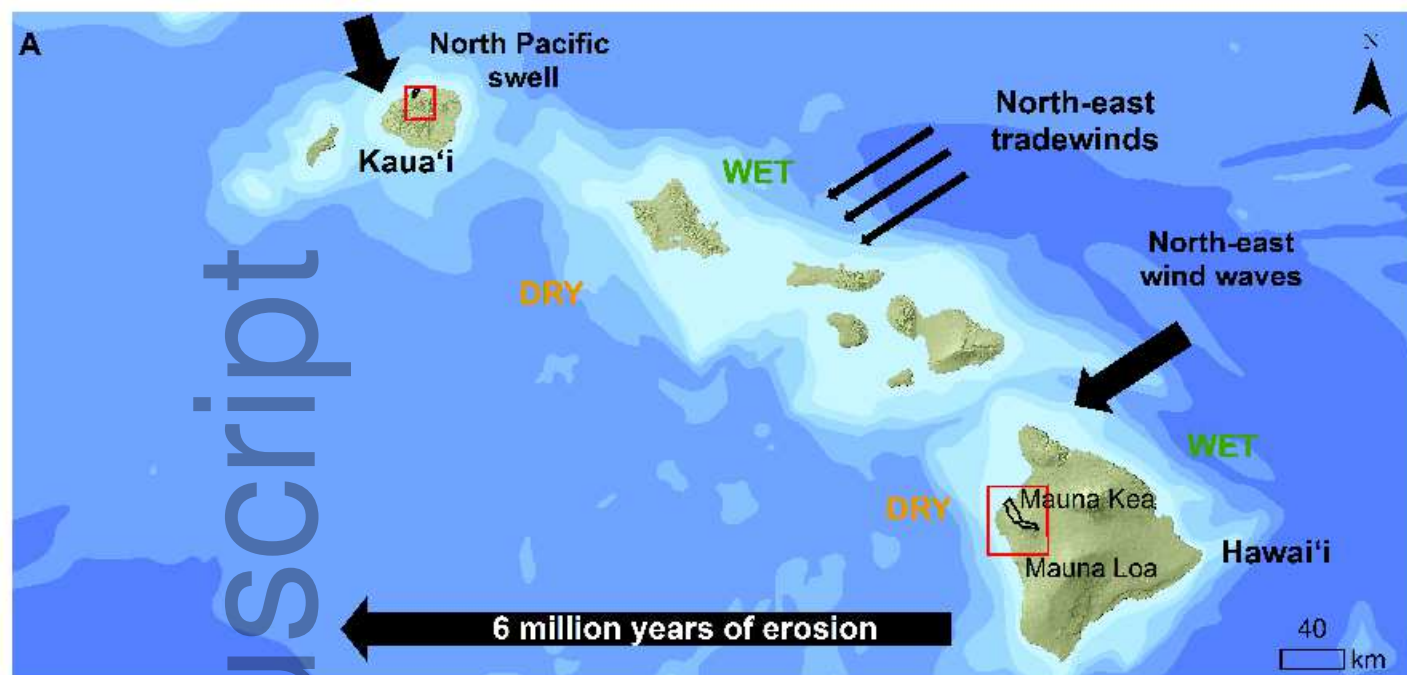
Figure 3. Coastal development scenarios and associated change in nutrient discharge at Hāʻena and Kaʻūpūlehu. (A) Low coastal development scenario and resulting relative increase in (B) modeled groundwater nitrogen (N) flux and coastal discharge and (C) modeled groundwater phosphorus (P) flux and coastal discharge, compared to present conditions at Hāʻena. (D) High coastal development scenario and associated relative increase in (E) modeled groundwater N flux and coastal discharge and (F) modeled groundwater P flux and coastal discharge, compared to present conditions at Hāʻena. (G) Low coastal development scenario and associated relative increase in (H) modeled groundwater N flux and coastal discharge and (I) modeled groundwater P flux and coastal discharge, compared to present conditions at Kaʻūpūlehu. (J) Low coastal development scenario and associated relative increase in (K) modeled N flux and coastal discharge and (L) modeled P flux and coastal discharge, compared to present conditions at Kaʻūpūlehu.

Figure 4. Effects of human stressors scenarios, priority areas for land-based management, and potential recovery within marine closure on coral reef at Hā‘ena. Coral reef areas vulnerable to: (A) Nutrient runoff under low coastal development scenarios, (B) nutrient runoff under high coastal development scenarios, (C) coral bleaching under the high climate change scenario, (D) coral bleaching and nutrient runoffs under the high climate change and coastal development scenario. Grey represents no change compared to the present conditions. The priority areas under the present costal development scenarios are shown in purple and the priority areas under future coastal development are in the pink zones. Land areas are linked to coral reef areas vulnerable to: (E) nutrient runoff impacts and potential fish recovery within the marine closure under low coastal development scenario; (F) nutrient runoff impacts and potential fish recovery within the marine closure under high coastal development scenario; (G) nutrient runoff and coral bleaching impacts and potential fish recovery within the marine closure under current land-use and high climate change scenario; (H) nutrient runoff and coral bleaching impacts and potential fish recovery within the marine closure under high climate change and coastal development scenario.

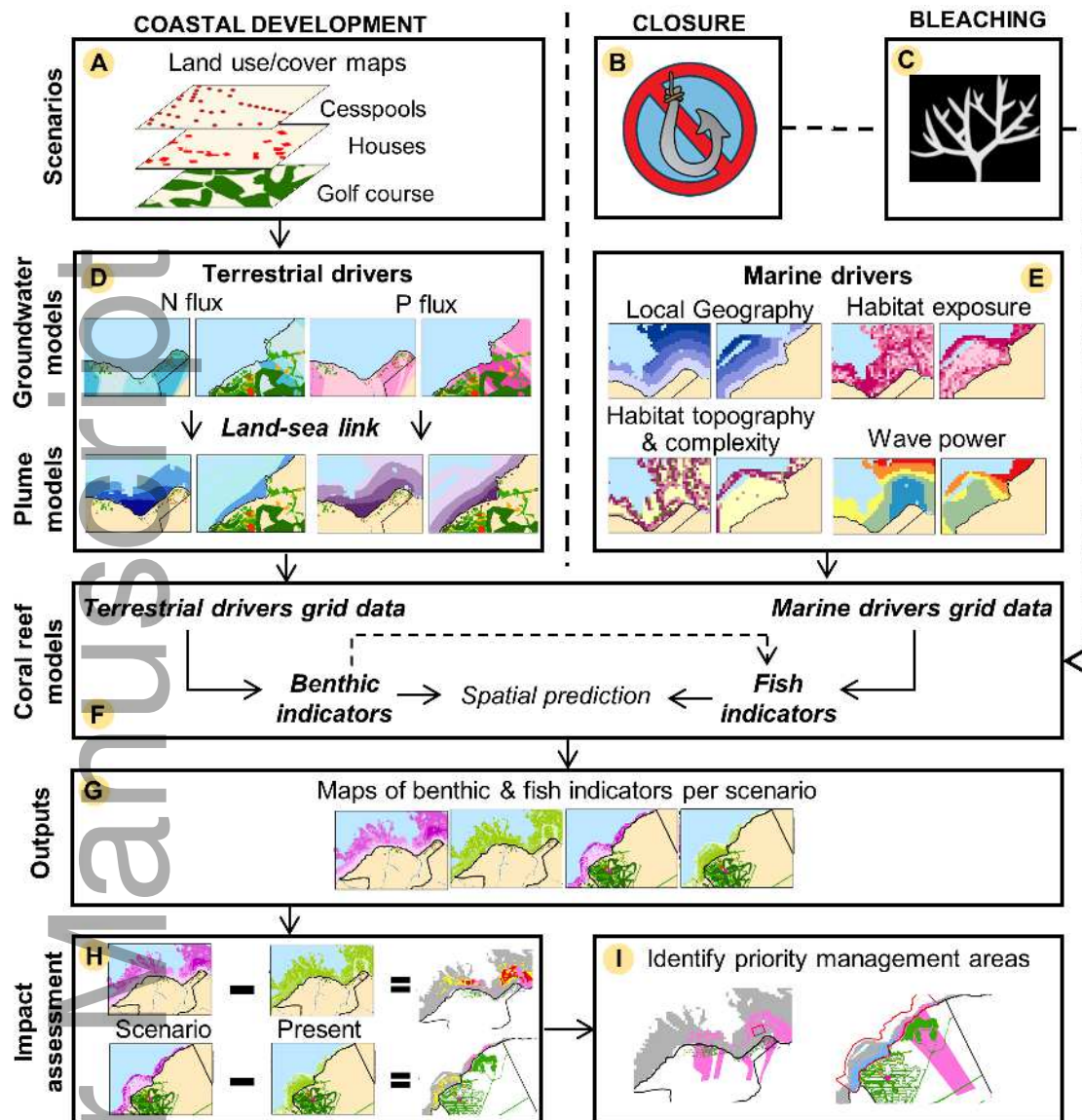
Figure 5. Effects of human stressors scenarios, priority areas for land-based management, and potential recovery within marine closure on coral reef at Ka‘ūpūlehu. Coral reef areas vulnerable to: (A) Nutrient runoff under low coastal development scenarios, (B) nutrient runoff under high coastal development scenarios, (C) coral bleaching under the high climate change scenario, (D) coral bleaching and nutrient runoffs under the high climate change and coastal development scenario. Grey represents no change compared to the present conditions. The priority areas under the present costal development scenarios are shown in purple and the priority areas under future coastal development are in the pink zones. Land areas are linked to coral reef areas vulnerable to: (E) nutrient runoff impacts and potential fish recovery within the marine closure under low coastal development scenario; (F) nutrient runoff impacts and potential fish recovery within the marine closure under high coastal development scenario; (G) nutrient runoff and coral bleaching impacts and potential fish recovery within the marine closure under current land-use and high climate change scenario; (H) nutrient runoff and coral bleaching impacts and

1278 potential fish recovery within the marine closure under high climate change and coastal
1279 development scenario.

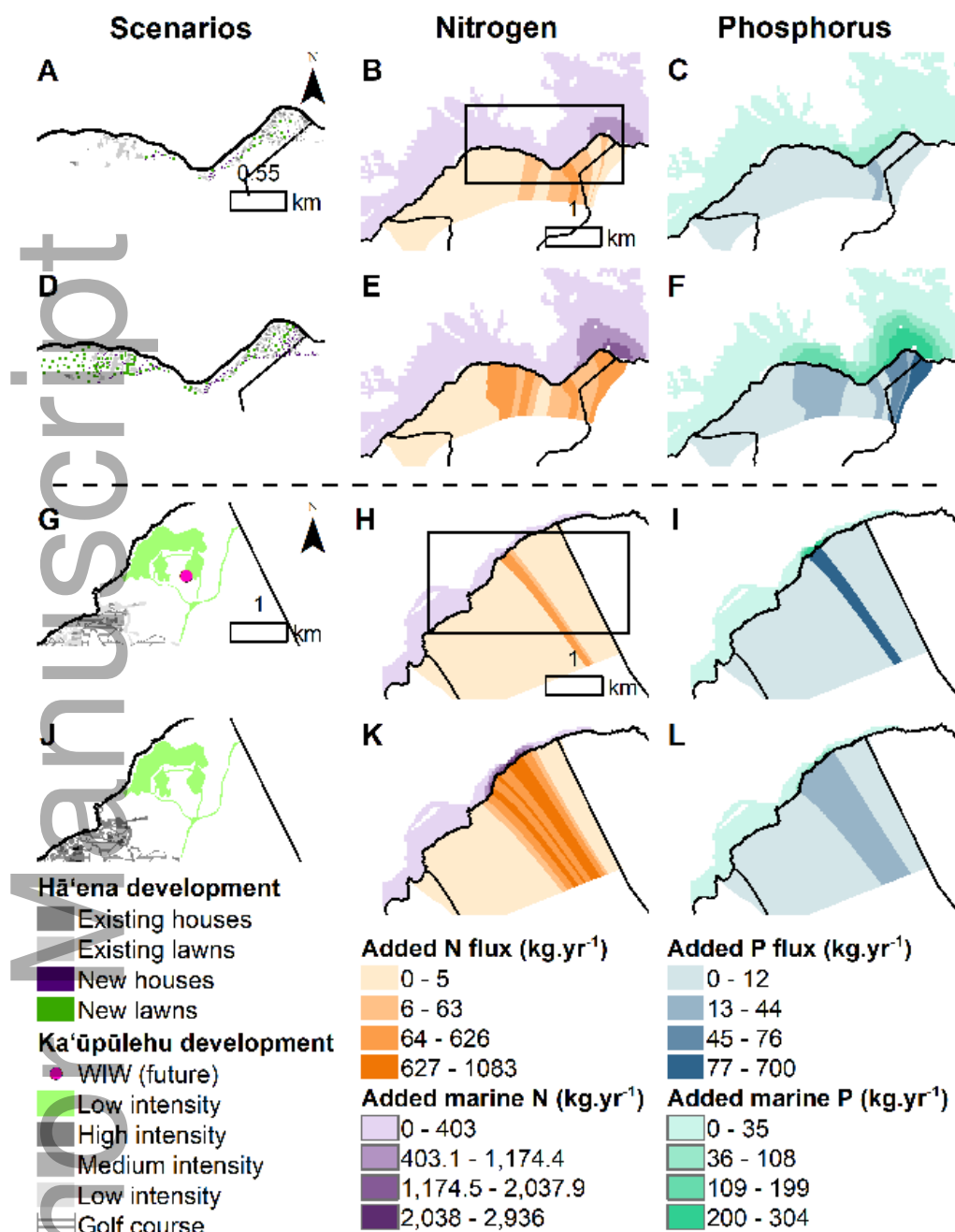
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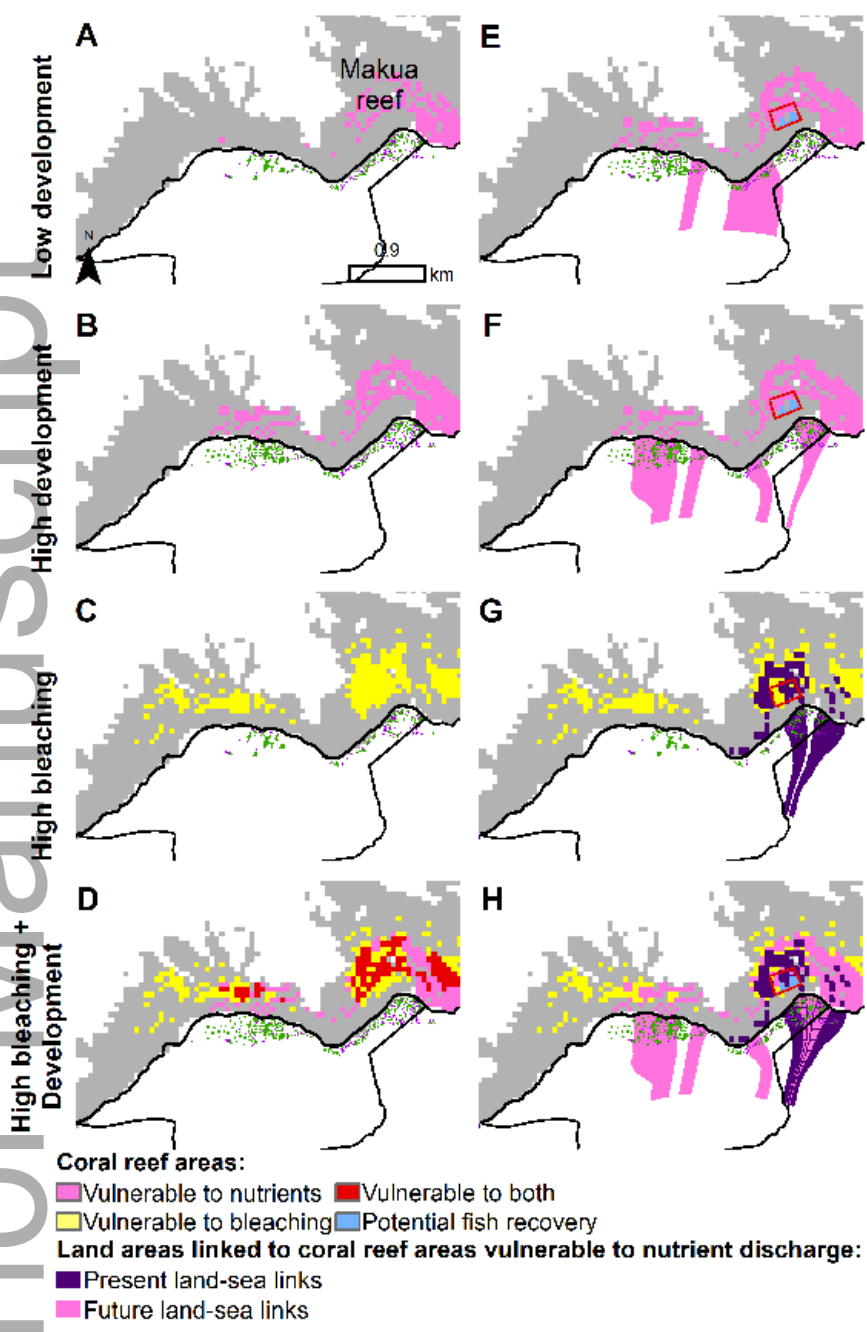
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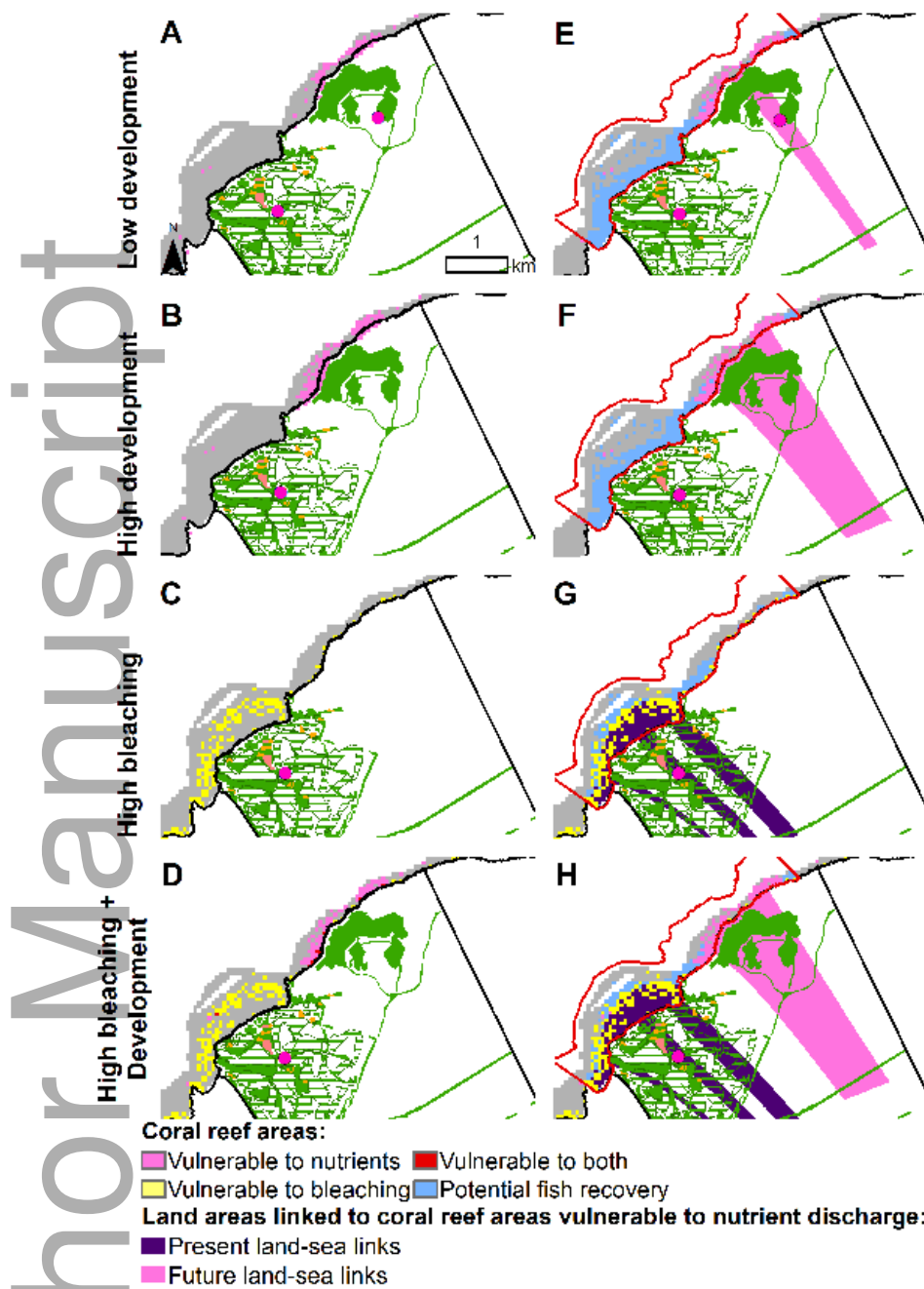
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