The economic value of coral reefs under future climate scenarios for the Main Hawaiian Islands

OR

Loss of recreational values of coral reefs for the Main Hawaiian Islands under future ecological scenarios

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Abstract

Coral reefs, characterized by their rich diversity, are productive ecosystems contributing to the provision of a wide range of ecosystem services, including recreation, coastal protection, and marine biodiversity. Climate change impacts, including ocean warming and acidification, pose a significant threat to coral reefs and the associated provisioning of ecosystem services. The variability of these impacts underlines the need to develop more spatially explicit tools in coastal ecosystem management that integrate and assess potential ecological and socio-economic outcomes. To address this gap, we employ a predictive ecological model and project changes in coral reef cover using downscaled predictions from socioeconomic pathway (SSP) climate scenarios. Using future scenarios, we estimate welfare impacts from recreational value of coral reefs across populations and landscapes. Our process considers both site-specific characteristics, income distributions and regional projected population growth to bridge the gap between ecological consequences and economic considerations. We highlight environmental justice concerns by identifying historically disadvantaged communities and the differing regional vulnerabilities. Our findings can inform policy decisions and resource allocation strategies promoting a more comprehensive and holistic approach to ecosystem management in the MHI.

JEL codes: Q26; Q51; Q54; Q57

Highlights

Graphical abstracts

Introduction

Although coral reefs cover only 0.2% of the seafloor, they play a vital role by supporting 25% of marine species (Souter et al., 2021). Coral reefs face increasing challenges in the face of anthropogenic climate change due to interconnected factors such as ocean acidification and rising sea temperatures (L. M. Brander et al., 2012; Hoegh-Guldberg et al., 2017; Pandolfi et al., 2011). Ocean acidification impedes coral growth and structural integrity by hindering their ability to build skeletons, while rising sea temperatures contribute to increased coral bleaching events. These climate-induced changes pose risk to the economic values communities derive from the health and quality of coral reefs (Narita et al., 2012; Armstrong et al., 2012; Moore & Fuller, 2022).

Coral reef habitats provide a diverse array of valuable ecosystems goods and services such as safety, coastal protection, food, wellbeing, and economic security of hundreds of millions of people (Cesar & Beukering, 2004; Colt & Knapp, 2016). They are an effective natural barrier from waves which allow human populations to settle on tropical coasts and reef islands (Ferrario et al., 2014; Pascal et al., 2016; Harris et al., 2018). The fisheries associated with coral reefs contribute substantially to the global economy, providing livelihoods for millions of people (Speers et al., 2016; Fernandes et al., 2017). The recreation and tourism sector greatly rely on the quality associated with the recreational opportunities coral reefs offer (H. S. J. Cesar, 2002; Cesar & Beukering, 2004; L. M. Brander et al., 2007; L. Brander & van Beukering, n.d.; Spalding et al., 2017; Fezzi et al., 2023). Spalding et al (2017) estimates 30% of the world's reefs are of value in the tourism sector. Despite their multifaceted contributions, recreational services of coral reefs are often undervalued in decision making due to their nonmarket nature, resulting in sub-optimal management of these fragile and important ecosystems (REF).

Given the significant impacts of climate change on coral reefs, understanding their vulnerability and assessing the economic consequences of their degradation is crucial. This enables environmental managers to develop strategies that better ensure their long-term resilience, safeguarding the well-being of both ecosystems and communities. However, climate change impacts on coral reefs will vary spatially, meaning different regions will experience varying degrees of degradation or resilience. For environmental managers, this calls for an integrated planning approach that captures both the physical and economic consequences of these spatial differences, contributing to tailor spatially targeted policies (Bateman et al., 2013; Ferrini et al., 2015).

This study develops an empirical approach that incorporates a complex long-term ecological modeling of changes in coral reefs with an economic valuation considering beach site heterogeneity and assessing the welfare tradeoffs under various climate scenarios. We do this by linking projected climate outcomes from an Atlantis ecosystem model with a valuation study providing site specific recreational values of coral reefs based ecological conditions and site amenities (Fezzi et al., 2023). We first expand the valuation to all recreational sites across the main Hawaiian Islands (MHI); Maui, Kauaʻi, Oʻahu, Lani, Molokaʻi and the Hawai'i Island and then impose the projected coral reef loss under climate change scenarios known as the shared socioeconomic pathways (SSPs) at site specific locations to measure the welfare impacts the population face under various climate futures (O'Neill et al., 2014).

Our findings provide spatially relevant information for decision-makers to maximize welfare by providing insights into targeted restoration and identify potential vulnerable communities under future ecological changes. It also provides much needed updated estimates of Hawaiiʻs economic value of coral reefs specifically to residents of the Hawaiian islands (Cesar & Beukering, 2004). Furthermore, our research aligns with current governmental managing agencies in the process to establishing new strategic management plans to address local and global impacts to there nearshore resources to meet the Sustainable Hawaii Initiative's goal to "effectively manage 30% of nearshore areas" in Hawaii by 2030 (NOAA 2022; DLNR 2022).

Data & Methods

Economic Data & Methods

A list of beach recreational sites were determined for the Hawaiian island of Maui, Oʻahu, Kauaʻi, Molakaʻi, Lani and the big island of Hawaiʻi. To identify all coastal sites of Kauaʻi, Oʻahu, Molakaʻi, Lani and the big island of Hawaiʻi, recreation sites were identified by merging existing spatial datasets (e.g., Hawaii Statewide GIS program, https://geoportal.hawaii.gov/) and Google Places API Places (e.g., https://developers.google.com/). Using the Nearby Search we identified bays, beaches, beach parks, harbors, public beaches, tourist attractions, hiking areas, state parks, scenic spots, boat ramps, and wildlife refuge. The Nearby Search was then manually cleaned and cross checked with guide books to verify the existence and location of sites across the island. Figure 1 shows beach sites identified across the islands and the population the sites serve. We also highlight areas under Marine Management Areas managed by the Division of Aquatic Resources (DAR). These areas are managed by DAR with a goal to manage, conserve and restore the state's unique aquatic resources and ecosystems for present and future generations.

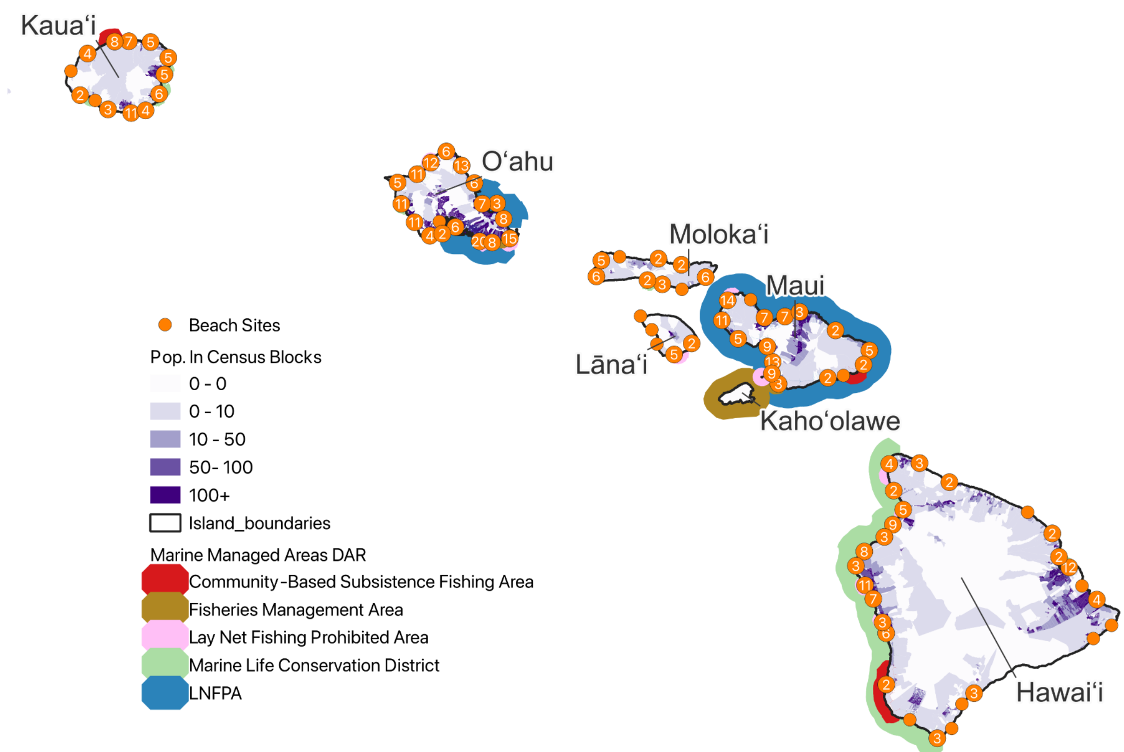

Figure 1. Beach Sites across the main Hawaiian Islands. (N = 444)

Note; Clustering of beach sites are represented by numbers in red dots to better represent beaches and reduce overlapping.

All beach site characteristics were also collected. This includes available recreational activities and amenities at each site such as surfing, swimming, snorkeling, a lifeguard, showers, parking, restrooms, sandy and rocky beaches as well as biophysical characteristics such as beach length, average coral cover and average fish biomass. In accordance with Fezzi et al. (2023), the mean coral cover was determined utilizing the methodology outlined in Asner et al., (2020) and representation of fish biomass relies on data sourced from the Marine Biogeographic Assessment of the Main Hawaiian Islands (Costa & Kendall, 2016). Table 1 summarizes the characteristics of the coastal sites.

We use the 2020 Census block level population estimates from each island collected from the Census Bureau (Figure 1). For each island, we created a population density map by mapping the islands into a 1km grid. For each 1km grid, we calculate the population density of each block area which is within each 1km grid area for the MHI. This grinding process allows for a richer calculation of the distance and travel time for each km^2 area to each site across the island. For each 1km, we calculate the distance and times to each site from each grid by using the Open Source Routing Machine (OSRM) (Giraud, 2022). OSRM's advanced routing algorithms ensure a cost-effective, accurate, and efficient solution for estimating travel times comparable to more costly options such as Google Distance Matrix API and ArcGIS Network Analysis (Fu et al., 2023). Income is matched at the census tract level to each subgroup population grid and calibrated to the medium income of the original population. We account for projected increases in human population growth over the 21st century by the rate of growth of the population over the period of one year over climate scenarios and at the geographical scale of the United States. [1] The US is expected to experience a medium level of population growth in both SSP1 ('Sustainability') and SSP2 ('Middle of the Road') which is driven by medium levels of fertility, mortality, and international migration. Population growth is low in SSP3 ('Regional Rivalry') due to low fertility and international migration along with high mortality.

Lastly, an important aspect of this work is to consider the welfare impacts from climate change and environmental justice. Recent government guidance in the United

[1] https://dataexplorer.wittgensteincentre.org/

States has outlined the importance of equity in Executive Order 14008 (Federal Register, 2021). The directive guides agencies to develop programs, policies, and activities that address the disproportionately high and adverse impacts on disadvantaged communities. These include human health risks, environmental degradation, climate-related challenges, and other cumulative burdens, along with the economic difficulties that accompany them. To better identify disadvantaged communities and refine policies, we use climate-simulated scenarios alongside the Climate and Economic Justice Screening Tool. This tool provides data on the various burdens faced by communities, using indicators derived from census tracts to inform more effective investments in nearshore environments and related areas.

Table 1
Descriptive statistics of coastal sites characteristics (N = 444)

Island	Maui (Original Study)	Oahu	Kauai	Hawaii	Lanai	Molokai
N(Grid * Sites)	191,948	257,174	95,542	1,089,184	4675	22533
Avg. distance (Mi) to Sites	34.9	28.1	32.5	63.6	14.5	20.6
Medium Income	86,872	87,722	82,818	65,401	59,053	47329
Pop. Total 18 Older	122K	810K	51K	159K	2.5K	5.5K
EJ Population						
Marine Managed Area ()						
Coral Reef Cover %	11.5	4.64	4.75	12.9	8.61	6.14
Avg. Resource Fish Biomass g/m2	16.4	5.62	18.7	15.8	4.38	12.2
Beach Length Meters	404	1801	2071	518	1073	781
Num Beaches	94	149	62	101	10	28
Num. Snorkel	60	64	26	46	2	8
Num. Surf	39	91	24	23	2	3

Our focus is to estimate the welfare losses from recreational values associated with changes in the ecological condition under different climate scenarios for the nearshore environment around the main Hawaiian Islands. We conduct a value function transfer using parameters derived from a multi-site travel cost (TC) model, which estimated the value of recreational ecosystem services for more than 170 outdoor sites located on the island of Maui, based on data from 300 residents who took nearly 3,000 trips a (Fezzi et al., 2023). The random utility model (RUM) forms the foundation of multi-site TC models, assuming that individuals maximize utility when choosing recreation sites based on factors such as travel cost, site amenities, and environmental attributes.. The multi-site TC model is workhorse for modeling recreational choices and is often applied to the valuation of coastal recreation

(Phaneuf & Smith, 2005; Leggett et al., 2018; Dundas & Von Haefen, 2020; Fezzi et al., 2023). Fezzi et al (2023) estimated parameters which enable the calculated welfare values associated with each sites' quality (Hanemann, 1999). The RUM assumes a standard approaches in economics such that the individual maximizes their utility function subject to their available income, m, and a vector of additional exogenous variables x_j which influence recreational demand (e.g time constraints, travel cost and recreation site characteristics). The result of the maximization of the Maui indirect utility function expressed maximum utility directly as a factor of exogenous factors, ecosystem services, and income.

$$V_{ij} = \alpha_j + \theta tc_{ij} + \beta' X_j, \tag{1}$$

where V_{ij} is the indirect utility function, tc_{ij} is the travel cost from the individual's home to the site which is equal to zero for staying at home, X_j is a vector of site characteristics (e.g. coral cover, fish biodiversity, presence of a amenities etc.) while each constant α_j identifies a different site-type (e.g., coastal locations, city parks, etc.), θ and β are the model parameters to be estimated. Following economic standard assumption of linear utility and additive income (Hanemann, 1999), it followed for each recreational choice occasion, the compensating variation in welfare for individual i associated with changes in sites' qualities is:

$$\theta^{-1} ln \left(\frac{\sum_{j=0}^{J} exp\left(V_{ij}^{*}\right)}{\sum_{j=0}^{J} exp\left(V_{ij}\right)} \right), \tag{2}$$

where V_{ij}^{*} is the attributes of the sites after the quality change and can be used to calculate the welfare changes for each individual in a sample. Alternatively, and for purposes of this study, it is aggregated at a regional level of the state of Hawaii by calculating the welfare changes by the number of people resident in each subgroups. Such welfare changes refer only to use values from recreation, and can be thought as the Willingness To Pay (WTP) of each individual i associated to the changes in quality from V_{ij} to V_{ij}^{*} for all possible sites J. Thus, we use the model as a welfare benefit function transfer which follows:

$$\underline{WTP_{ir}{}^{BT}} = f\left(\left[x_{ir}, x_{js}\right], \hat{\beta}_{js}\right) \tag{3}$$

where site $i \neq j$ and population, $s \neq r$ and the $\underline{WTP_{ir}}^{BT}$ is the adjusted observable difference between the predicted welfare estimate for original population in Maui, s, from the sites, j, with characteristics, x, to the new sites, i, and the subgroup, r. $\hat{\beta}$ is a conforming vector of estimated parameters from the original study. Each subgroup population is defined in km gridded squares for fine scale resolution. We then further stratify our results by income. For each income level we use the formula:

$$WTP_{ir} = \underline{WTP_{ir}}^{BT} * (\underline{y}_r / y_s)^{\eta} \tag{4}$$

where median income, $\underline{y}_r$, is matched to each census tract subgroup population, r, y_s is the original reported median household income from which WTP was calculated, n is the income elasticity. For simplification, we will be using η=1. This exercise uses economically sound transferable value functions and then spatially explicit based on-site characteristics (Bateman et al. 2011).

Ecological Data & Methods

Integrating high-resolution climate projections developed by regional oceanographic experts *(Friedrich et al., 2024),* we used the Atlantis ecosystem model framework (Fulton et al., 2004) previously tailored for the main Hawaiian Islands (MHI) *(Weijerman, 2020)* to simulate nearshore ecosystem outcomes to the end of the century. Atlantis is a spatially-explicit, three-dimensional, deterministic model that accounts for biophysical dynamics, predator-prey relationships, and species life histories. Modeled coral groups include branching and massive Porites, Pocillopora, Montipora, and Leptoseris.

Anthropogenic stressors like land-based pollution and fishing were simulated in the model, represented by sediment and nutrient pollution inputs *(Wedding et al., 2018)* and historical catch data[2], respectively. Climate forcing used high-resolution climate projections from the coupled Regional Ocean Modeling System (ROMS) and COBALT

2 https://www.pifsc.noaa.gov › wpacfin

models that incorporated global climate data from CMIP6. Three different Socioeconomic Pathways scenarios SSPs 1-3 representing low, moderate, and high emissions were used to simulate potential outcomes for the region.

Atlantis outputs represent coral biomass spatially by model cell and temporally by year. The modeled region covers approximately 8905 km^2 and reaches depths of up to 400 meters. The model's geometry captures spatial differences in marine habitats, biological communities, and management by dividing the area into user-defined polygons (referred to as "boxes"). These boxes are designed based on regions with similar biogeographic and oceanographic conditions and correspond to fishery reporting zones. A total of 79 dynamic boxes simulate biophysical processes, while 6 'boundary' boxes facilitate nutrient and plankton transfer to the dynamic boxes. There are also 8 island boxes, which have no internal processes. Total coral biomass and cover demonstrated substantial decline, with severity corresponding to climate intensity, respectively. Branching Porites showed the largest response, at almost 100% loss, while massive Porites retained the highest percentage of both biomass and cover, showing trends correlated with the intensity of climate change.

To integrate ecological conditions into our economic model, we calculate the total losses in coral biomass within each Atlantis grid cell. These losses are projected annually up to 2100, using the average coral biomass from 2015–2019 as the baseline. The base year is chosen to reflect the timing of the original survey. Each beach site is then matched to the nearest Atlantis grid cell, providing an identifier for estimating the projected changes in nearshore coral biomass over time.

Results

We convert Eq. (2) into an annual welfare measure by summing the values of a single choice occasion into a year and by the grid level population. Welfare measures are adjusted by medium income at the corresponding tract level. Figure 2-4 (a-b) maps the welfare under SSP1 & SSP3 in the near (2030), mid (2050) and late century (2100). Interpretation of these estimates should be thought of as how much the current resident would need to be compensated to return the resident to the original utility before an ecological change has occurred, i.e the original welfare of the 2020. Each estimate highlights the potential vulnerable locations under future ecological changes. Figure 2 represents SSP1 and SSP3 in the year 2030. Locations of note are the southwest portion of the island of Hawaiʻi which indicates a potential large reduction in live coral cover by the end of this decade and a large proportion of this ecological change falls within a fishery management area (Figure 1). The SSP1 does suggest only a 25% to 50% reduction whereas a SSP3 suggests a stark decline of more than 50% of the coral reef cover to the area.

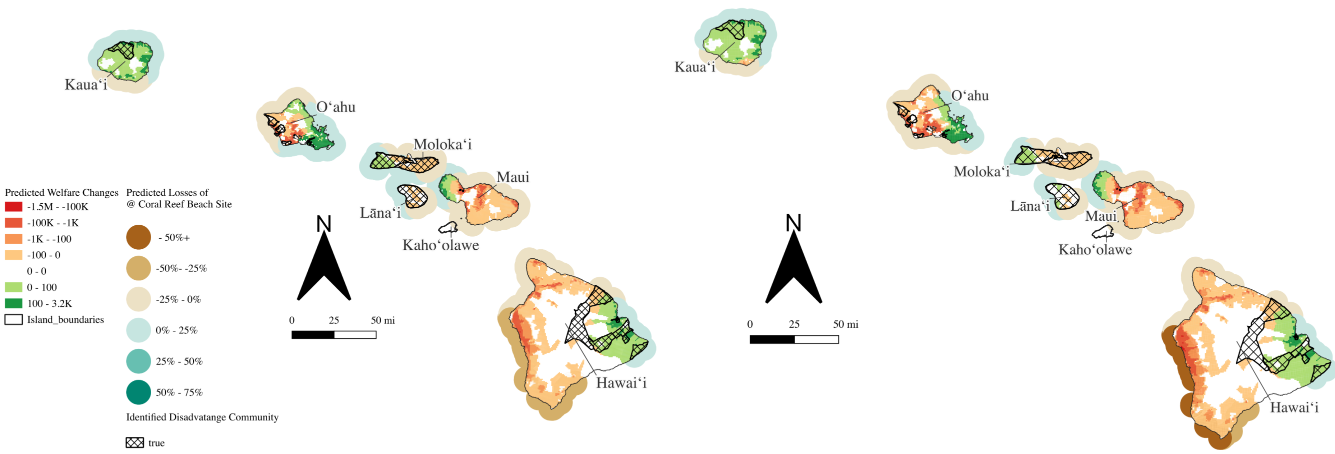

Figure 2. a) Atlantis using SSP1 (Left) b) SSP3 (Right) in 2030.

Figure 3 represents SSP1 and SSP3 in a mid-century snapshot at welfare changes (2050). Ecological changes are heterogenous across the islands predicted under the Atlantis model. One main difference between the scenarios is when the mid-century decline begins. The Atlantis model predicts larger immediate welfare gains in the near century under SSP1 but this is accompanied by mid-century decline that begins slightly sooner than SSP3. Both scenarios suggest the ecological changes to coral reefs are projected to see large declines by the mid-century with a near collapse by the late-century. This finding is consistent with the literature examining overall coral reef declines using climate scenarios around the Hawaiian islands (D. Lane et al., 2015). Welfare losses continue to be heterogeneous during the mid-century with both gains and losses reported across islands and locations.

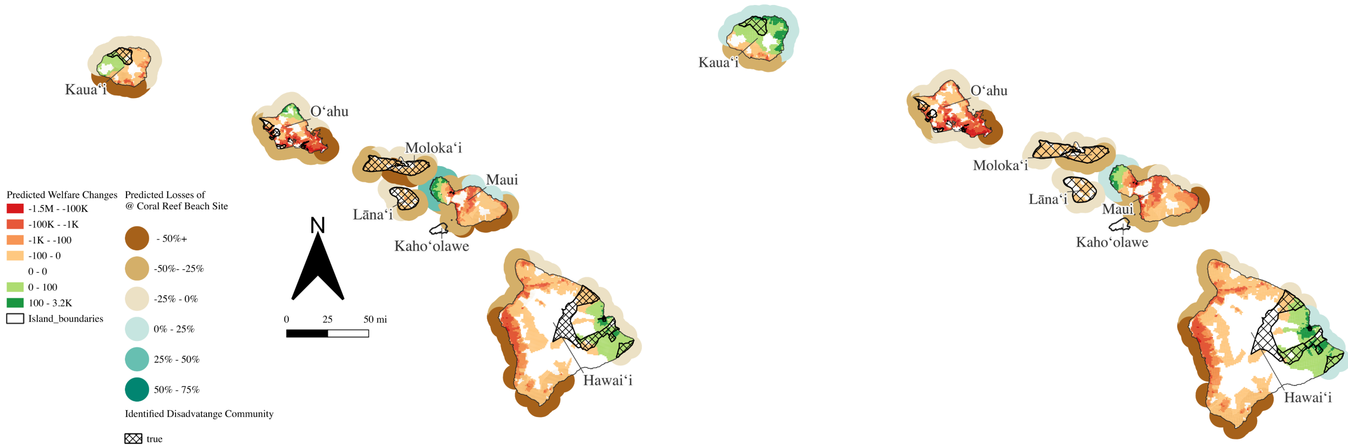

Figure 3. a) Atlantis using SSP1 (Left) b) SSP3 (Right) in 2050.

Figure 4 represents SSP1 and SSP3 in a late-century snapshot at welfare changes (2100). The ecological changes to coral reefs are projected to experience a near collapse at the late-century which is consistent with literature (D. Lane et al., 2015). All residents of 2020 would experience large welfare losses under future ecological quality decline scenarios to the nearshore coral reef biomass. Although both climate scenarios express large welfare losses, the analysis does highlight a few potential differences across the scenarios. While island residents experience reduction of welfare losses under climate change scenarios SSP1 & SSP3, the island of Molokaʻi experiences far less welfare losses under SSP1. This is relevant given the island is identified as an area of environmental justice concern.

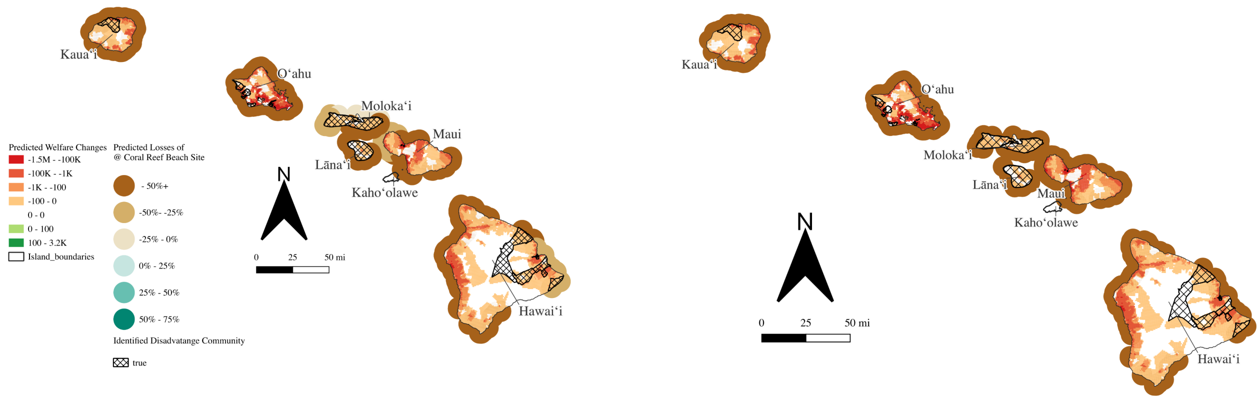

Figure 4. a) Atlantis using SSP1 (Left) b) SSP3 (Right) in 2100.

We further highlight potential vulnerable locations by identifying disadvantaged communities with environmental justice concerns (Federal Register, 2021). Furthermore, we provide the average welfare losses by island in Table 2. Examining the average welfare losses across identified disadvantaged and all other communities a few islands highlight potential concerns. The west side of Oahu both has high exposure to potential welfare losses from ecological changes and has on average higher welfare losses reported consistently across climate scenarios in disadvantaged communities than all other communities on the island (Table 2 A-B). Maui has two identified disadvantaged communities near Wailuku and Kahului which consistently have higher welfare losses than the other communities around the island. Lastly, the entire island of Lānaʻi and the near entire island of Molokaʻi are considered disadvantaged and thus are of particular concern when examining the welfare impacts across the islands.

A. Disadvantaged Communities Average Predicted Welfare Loss in 2020$

	SSP1			SSP3		
Island	**2030**	**2050**	**2100**	**2030**	**2050**	**2100**
Hawaii	48	22	-408	45	114	-645
Kauai	41	-4	-270	29	1101	-372
Lanai	-3	-14	-44	0	-12	-53
Maui	-24726	-67582	-236545	-44362	-130015	-315452
Molokai	-1	-3	-14	-2	-7	-27

Oahu	670	-42688	-824	-1518	-50764	-94536
Average	**-3995**	**-18378**	**-53291**	**.-7635**	**-30096**	**.-68514**

B. All Other Communities Average Predicted Welfare Loss

	SSP1			SSP3		
Island	**2030**	**2050**	**2100**	**2030**	**2050**	**2100**
Hawaii	-502	-1425	-1941	-847	-1301	-2038
Kauai	181	-175	-1544	115	377	-2065
Lani						
Maui	-255	-401	-11078	-1343	-4889	-15553
Molokai	-2	-1	-7	-2	-5	-24
Oahu	693	-30148	-60856	-741	-36970	-71232
Average	**23**	**-6430**	**-15085**	**-564**	**-85588**	**-181822**

Table 2 Snapshot of Predicted Average Annual Welfare Losses By Identified Disadvantaged Community per KM square grid. Reported average values per KM square in 2020$ by identified as historical and currently disadvantaged under the classification of the Justice40 Initiative (Federal Register, 2021). Highlighted values in A. note the on average higher impacts within islands respective disadvantaged communities than other communities.

We predicted a localized fit of the annual welfare loss across the islands using the long-term estimate of the social rate of time preferences over a 80-year horizon recommended through OMB Circular A-4 (Office of Management & Budget, 2023). Each scenario suggests a mid-century sharp decline in overall coral biomass in the nearshore environment accompanied by a large steep decline in welfare. However, the paths diverge in the late century, suggesting a mild recovery of welfare losses under the climate scenario SSP1 versus SSP3. The divergence stems from a few localized areas maintaining some of their coral reef cover in the SSP1 scenario whereas all localized areas experience a sharp 50% reduction in coral reef cover in the SSP 3 scenario. The overall analysis estimates welfare losses for residential recreational use between 2.6 to 3 billion in 2020 dollars for the year 2100.

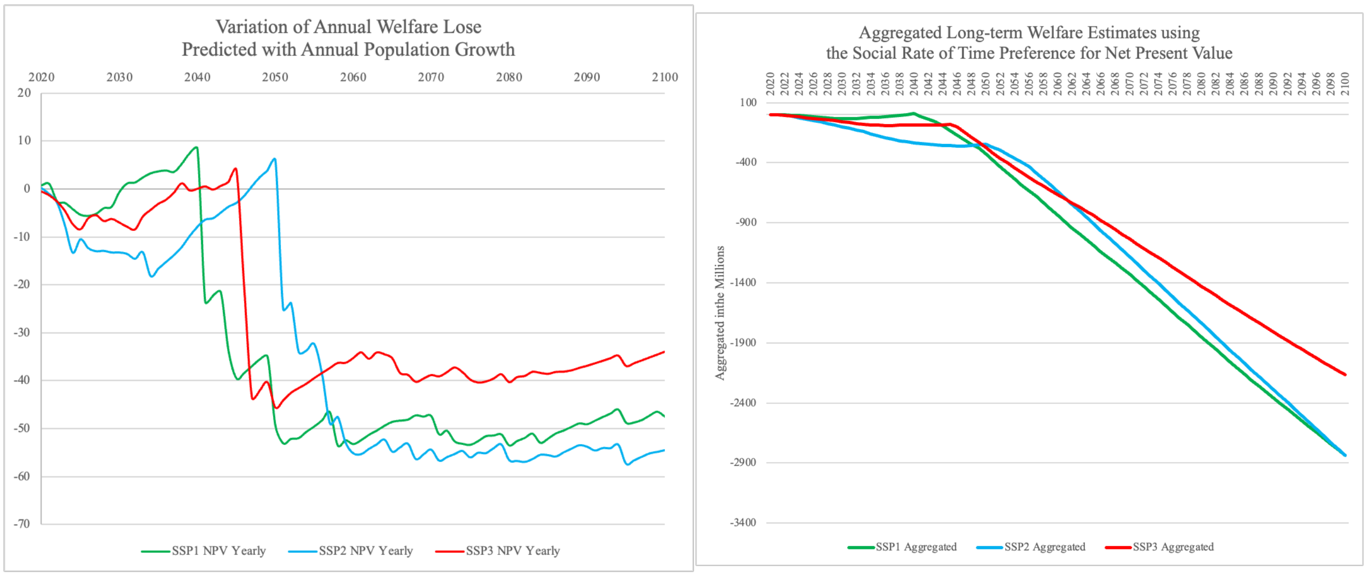

Figure 5 The Long Term Welfare Estimates using the Social Rate of Time Preference For Hawaii Residents and the recreational use of coral reef condition. Blue line represents the welfare estimates under the climate scenario SSP1 and the red line represents the welfare estimates under climate scenario SSP3.

Discussion

Coral reefs face significant threats from climate change, through drivers such as ocean acidification and bleaching from higher sea temperatures, resulting in reduced coral cover (Hoegh-Guldberg et al., 2007; Hoegh-Guldberg et al., 2017;). Reduced coral cover has economic consequences (Munday et al., 2008; Chen et al., 2015; Speers et al., 2016), including the loss of recreational values (Brander et al., 2007; Brander et al., 2012). These values are not directly reflected in market transactions, and are thus subject to the use of nonmarket valuation methods to assess their economic importance, otherwise, they risk being undervalued or ignored in policy formulation (Bateman et al., 2013). However, assessing future recreational value losses due to coral cover decline from climate change requires combining ecological models with nonmarket valuation. This integrated approach helps inform policy by providing insights into both the ecological consequences and the social welfare impacts of coral reef degradation.

In this study, we made projections of spatially explicit loss of coral reef cover under different SSPs using the Atlantis ecological model. We combined these projections with economic valuations of changes in recreational services using a value function transfer approach. Modeling these welfare far into the future have inherent uncertainties with large ecosystem models, such as Atlanatis (Bracis et al., 2020). There are disadvantages with requiring many parameters that can be unknown and must be estimated or calibrated to available data (Bracis et al., 2020). Hawaii Atlantis scaled uncertainties ….. Our results from ecological impacts under differing climate scenarios are similar to findings in the literature who have focused on the emission scenarios and ecological conditions of coral reefs in Hawaii (Lane et al., 2015).

Our main contribution to the literature arises from spatial heterogenous consideration of impacts to Hawaii residents. Our analysis estimated welfare losses for residential recreational use as a result of loss of coral reef is between 2.2 to 3 billion in 2020 dollars for the year 2100, which are far less than values projected in Lane et al., (2015). This is possible due to two main reasons 1) our estimates are restricted to Hawaii resident population preferences based on site specific amenity characteristics and ecological condition at each location previous welfare estimates based on climate projections were not spatially explicit and 2) were transferred by a weighting welfare estimate from South Florida given concerns of a downward bias from the only Hawaii study (Cesar & Beukering, 2004; D. R. Lane et al., 2013). The projected loss of coral reefs to the Hawaii islands by mid-century have exponentially larger impacts given the values used in this model are strictly based on revealed preferences of use values and do not include the non-use and tourism use values. Our estimates are intended to carefully examine residential recreational use values and highlight vulnerable communities by identifying welfare impacts based on disadvantaged communities that are relevant for policy concerns of environmental justice.

Furthermore, the ecological predictions in Atlantis model suggest some of the first initial declines are in marine protected areas located around the island of Hawaiʻi. The areas on the island of Hawaiʻi include management plans of Marine Life Conservation District (MLCD) and Community-Based Subsistence Fishing Area (CBSFA). The MLCDs are conservation areas that protect and preserve ecosystems in oceans and seas through planned management in order to prevent the over-exploitation of these marine resources. Our results support the continued management of these areas.

Notably, our welfare assumptions should be taken as extreme lower bounds only for the specified population it is derived from, i.e. residents of Hawaiʻi. The assumption underlying the compensating variation of the ecological change assumes the marginal changes are linear and that the estimated reduction into the future under each climate scenario are interpreted as marginal (Johnston & Wainger, 2015; Dugstad et al., 2021). However, the true functional form for extreme high biodiversity loss might be nonlinear and have considerable large "fat tails" (Weitzman, 2011). The critical probability distributions of coral reef collapse predicted here and elsewhere in the literature would fall into these highly uncertain damage functions. It is impossible to determine the true damage curve under these collapsed biodiversity scenarios and thus these estimates are an extreme lower bound of an extremely stark future. Nevertheless, the welfare estimates highlight important information for making environmental and economically sound decision-making.

Another limitation of this study is the reliance on benefit transfer. Specifically, we used a value function transfer approach, which means that we used information about how Maui residents value recreational services at coastal sites in Maui and applied that information to estimate values for other sites across the main Hawaiian Islands. Importantly, the value function transfer approach is generally preferred over other benefit transfer techniques, such as unit value transfer, as it allows for accounting for differences in site characteristics (Rosenberger & Loomis, 2017). However, our approach assumes that preferences for recreational services are constant across island populations, except that we adjusted for income differences. This assumption is, of course, subject to uncertainty, as preferences for recreational services may vary across different island populations. However, extensive research has been conducted on the validity and reliability of benefit transfer, including international benefit transfer (e.g., Ready & Navrud, 2006; Lindhjem & Navrud, 2008; Hynes et al., 2013; Dugstad & Navrud, 2023). Given the relatively close geographical and cultural proximity in our study, we believe that the likelihood of preferences being similar across the islands is higher compared to other benefit transfer applications, particularly those involving international transfers. Still, this consideration should be carefully examined in future research through examining transfer errors.

Another consideration for future research is the temporal stability of preferences. The survey on residents of Maui was conducted in 2017. Preferences for recreational services could have changed from then to now, when we apply the values elicited. This would have further implications on the values predicted for future coral loss under the different SSPs, as we assume preferences are constant over time. Rolfe & Dyack (2019) conducted a meta-analysis to test the temporal stability of recreation values, finding that TC values have increased up to 72 percent, and that the contingent valuation method is more reliable for benefit transfer over time than the TC method.

Conclusion

The results of this study are consistent with previous literature showing coral reefs are highly vulnerable to climate change and that adoption of policies focused on mitigating emissions suggest reduced impacts of the projected biological and economic impacts. The consensus suggests coral reefs appear to face a dire future under either climate scenario, however there are still clear benefits which are gained from mitigating emissions.

We also used welfare estimates that both consider substitutable activities and site specific attributes of all recreational locations across the Hawaii island. This includes both site specific characteristics and heterogenous ecological conditions at each site. We then mapped projected climate changes on ecological conditions to the nearshore environment and estimated the welfare changes to Hawaii residents. These welfare estimates include population growth under the various climate scenarios.

This paper combined an ecosystem model which considered all parts of the marine ecosystem - biophysical, economic and social- and an expanded spatial explicit recreational valuation model to the main Hawaiian islands. We showed how coral reef quality declined under differing climate scenarios will impact social welfare based on the heterogeneous nearshore coastal recreational areas. The most important aspect of our approach lies in its spatial nature. Our combined model is able to simulate a complex array of potential impacts of different geographical locations, thus providing decision makers relevant information for assessing the welfare impacts of a variety of coastal management strategies under these scenarios. It also illustrates the ethical implications of these management strategies.

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