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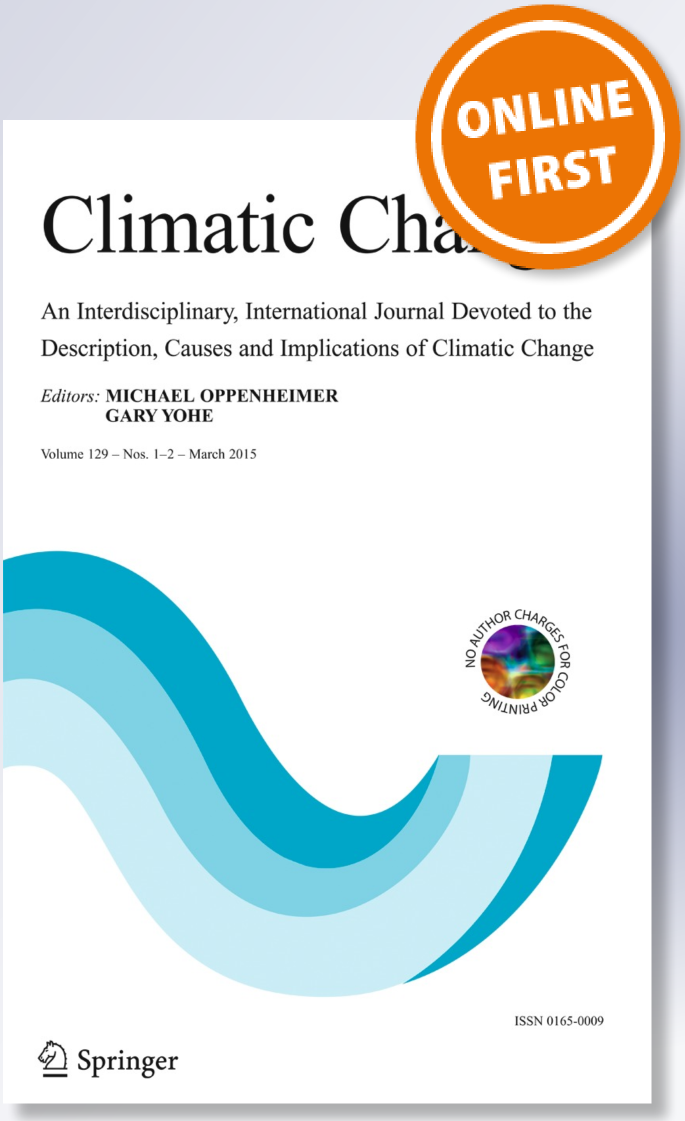
Climatic Change
An Interdisciplinary, International Journal Devoted to the Description, Causes and Implications of Climatic Change

ISSN 0165-0009

Climatic Change
DOI 10.1007/s10584-015-1377-3



Climatic Cha

An Interdisciplinary, International Journal Devoted to the Description, Causes and Implications of Climatic Change

Editors: MICHAEL OPPENHEIMER
GARY YOHE

Volume 129 – Nos. 1–2 – March 2015

ISSN 0165-0009

Springer

Springer

Climatic Change
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Received: 11 April 2014 / Accepted: 1 March 2015

Abstract Coastal erosion, salt-water intrusion, and flooding due to sea-level rise threaten to degrade critical coastal strand and wetland habitats. Because habitat loss is a measure of the risk of extinction, managers are keen for guidance to reduce risk posed by sea-level rise. Building upon standard inundation mapping techniques and suitability mapping, we develop a ranking system that models sea-level rise vulnerability as a function of six input parameters defined by wetland experts: type of inundation, time of inundation, soil type, habitat value, infrastructure, and coastal erosion. We apply this model under the mid-century and end-of-century RCP8.5 sea-level projection (0.30 m by 2057, and 0.74 m by 2100) according to the Intergovernmental Panel on Climate Change Fifth Assessment Report. To demonstrate this method, the model is applied to three coastal wetlands on the Hawaiian islands of Maui and O'ahu. Each ranked input parameter is mapped upon a 2 m horizontal resolution raster and final vulnerability is obtained by calculating the weighted geometric mean of the input vulnerability scores. Areas that ranked with the 'highest' vulnerability should be the focus of future management efforts. The tools developed in this study can be a guide to prioritize conservation actions at flooded areas and initiate decisions to adaptively manage sea-level rise impacts.

1 Introduction

Globally, coastal strand and wetland habitats have high conservation value due to the role they play in the preservation of endangered and endemic organisms. Wetlands provide a variety of functions that reduce storm damage and stabilize shorelines (Gedan et al. 2011), trap land-based sediments, retain nutrients, and alleviate flooding (Bruland 2008). In the Pacific region alone, over 2500 islands and atolls harbor a diverse range of freshwater, coastal, and marine wetlands (Ellison 2009). It is noted that the disappearance of small wetlands will cause a dire

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Electronic supplementary material The online version of this article (doi:10.1007/s10584-015-1377-3) contains supplementary material, which is available to authorized users.

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reduction in the ecological connection among remaining species populations (Semlitsch and Bodie 1998).

Sea-level rise (SLR) is a growing problem on low lying coastal plains and threatens costal strand and wetland habitats with increased erosion (Romine et al. 2013), frequency of extreme high water events (Tebaldi et al. 2012), pond water levels, and salinity (Kuan et al. 2012). The Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (AR5) predicts under a worst case scenario (RCP8.5), global sea-level will increase 0.30±0.08 m by 2057 (mid-century) and 0.74±0.23 m by 2100 relative to 1986–2005 (Church et al. 2013). Equatorial Pacific regions may experience sea-level values between 10 and 20% above the global mean (IPCC 2013). Islands within the tropics are especially vulnerable because species have narrow tolerances for changes in climate (Mora et al. 2013), and microtidal (<2 m tidal range) environments do not allow for large concentrations of marine suspended sediment to aid in vertical accretion in response to SLR (Kirwan et al. 2010).

To date, much of insular SLR vulnerability research has focused on summarizing potential impacts on a global scale (e.g., Wetzel et al. 2012; Bellard et al. 2013). For example, Bellard et al. (2013) found that approximately 6% of the 4447 islands investigated worldwide would be entirely submerged under 1 m of SLR. Global assessments are beneficial for demonstrating the general consequences of SLR, however through the use of low resolution elevation data sets, the final vulnerability maps are produced with large errors (Cooper et al. 2013b). Furthermore most management occurs at regional and local scales, thus fine-scale vulnerability assessments are more relevant for direct decision making (Halpern et al. 2007).

Prior regional scale assessments define SLR vulnerability based upon uncertainty in high-resolution elevation along with tidal datums (Gesch 2013) and SLR estimates (Cooper and Chen 2013). However within highly managed areas such as Hawaiian wetlands, landscape vulnerability should relate to the site-specific goals of decision makers. Previous studies have gained stakeholder's support by ranking the vulnerability of marine ecosystems to anthropogenic threats based on survey results of experts (e.g., Halpern et al. 2007; Fuentes and Cinner 2010; Selkoe et al. 2008). These studies employ an expert elicitation process that involves synthesizing expert's opinion of relative impacts and assessing the uncertainty of those views (Halpern et al. 2007; Fuentes and Cinner 2010). Expert knowledge is used both in instances when data is scarce, and to supplement empirical data (Hameed et al. 2013). To aid in the prioritization of conservation actions those ecosystem threats that are identified as having the greatest impact generally are dealt with first (Fuentes and Cinner 2010).

Most threat ranking studies to date include SLR in an array of multiple stressors, and do not provide maps that spatially represent areas assumed to experience the greatest impacts. Previous studies have generated suitability maps that use ranked variables to identify potential wetland mitigation and restoration sites (Van Lonkhuyzen et al. 2004; White and Fennessy 2005). Integrating expert elicitation with suitability mapping in a GIS would allow decision makers with site-specific goals to evaluate landscape vulnerability due to future SLR.

We present an important case study that depicts a localized approach to prioritizing management strategies in response to SLR based upon a number of predetermined factors such as the time and nature of flooding, environmental features that influence flood severity, and the loss that would result from flooded high value habitats and infrastructure. The vulnerability ranking process may be easily refined and replicated to other small, microtidal islands to accommodate different planning needs, data availability, and sources of expert knowledge.

2 Methods

The modeling approach used in this study to assess SLR vulnerability is outlined in Fig. 1. Study sites were identified based upon the biological integrity of managed resources within an area, the existence of experienced and knowledgeable management staff, and the availability of mappable layers such as high resolution topographic data (Fig. 1a). We defined vulnerability to future SLR of 0.30 m by 2057 and 0.74 m by 2100 (Church et al. 2013) from a management perspective by mapping those parameters that best characterize how SLR will impact decision makers' ability to accomplish mandated goals and objectives (Fig. 1b). A literature search and elicited expert knowledge were used to rank vulnerability parameters for each study site from very high (5) to very low (1) (Fig. 1c). We used ArcGIS 10.2 to apply the ranked vulnerability scores and a weighted geometric mean to map the cumulative vulnerability (Fig. 1d–f). Areas with the highest vulnerability were identified and should be used to guide adaptive management planning (Fig. 1g–h). Wetland experts may modify the model and refine the definition of vulnerability as new information becomes available (Fig. 1i).

2.1 Study area

In conjunction with the Hawai‘i Wetland Joint Venture, a group that represents state, federal, and local wetland managers, three low lying coastal environments were identified: James Campbell National Wildlife Refuge (north O‘ahu), Kanaha Pond State Wildlife Sanctuary (north Maui), and Keālia Pond National Wildlife Refuge (south Maui). The refuges are composed of three dominant environments including coastal wetlands, a coastal strand, and

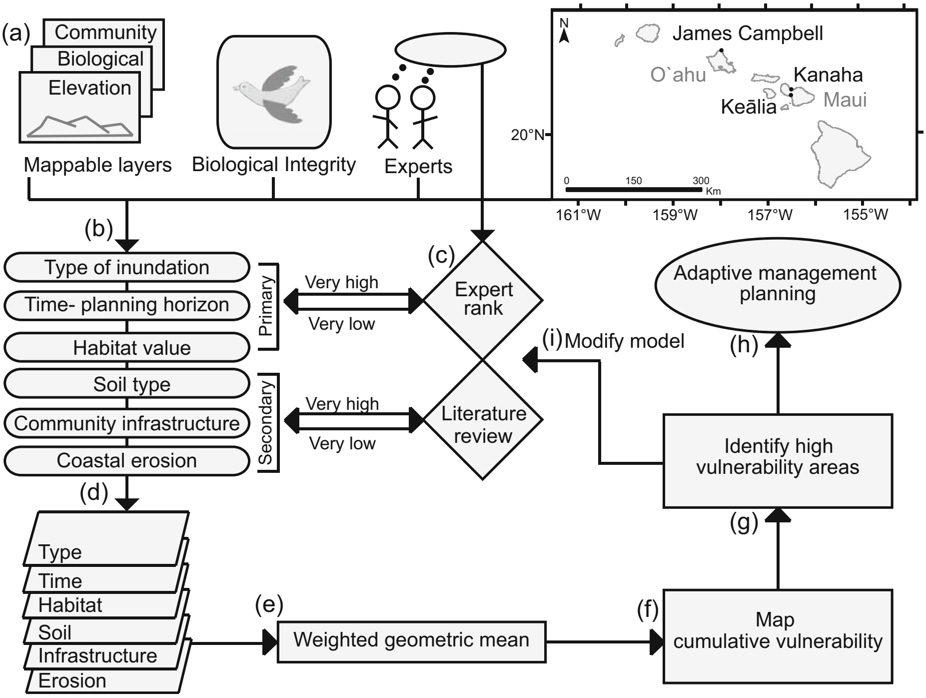

Fig. 1 Depiction of the vulnerability ranking process

upland habitats. The wetlands consist of freshwater impoundments and natural ponds that are fed by groundwater, and rainfall. The wetlands are largely buffered from marine flooding and sediment inputs by a narrow coastal strand and 2–4 m sand dunes (U.S. Fish and Wildlife Service 2011a). The refuges provide habitats for the recovery of endangered waterbirds, the endangered Hawaiian monk seal (*Monachus schauinslandi*), the threatened Hawaiian green sea turtle (*Chelonia mydas*), seabirds, and migratory shorebirds. Identified at each study site were one to two senior wetland experts (four experts total) who from training, research, and personal experience (5–20+ years) possess the greatest capacity to assess how SLR will impact future management strategies.

2.2 Defining sea-level rise vulnerability

Here, we define SLR vulnerability as having primary and secondary parameters. Through discussions with local wetland managers it was expressed that they are primarily concerned with prioritizing management at flooded areas upon the value of each site. The primary parameters are defined by 1) type of inundation, 2) time of inundation, and 3) habitat value. The secondary parameters refine the definition of threatened resources based upon the availability of ancillary data. Secondary parameters are defined by 1) soil type, 2) community infrastructure, and 3) coastal erosion hazard zones.

2.3 Ranking vulnerability parameters

Expert judgment was used to rank the primary vulnerability parameters defined in Section 2.2. Face to face surveys were conducted and experts were asked a series of questions where they ranked the vulnerability of their refuge to SLR from very low (1) to very high (5) (Table 1). After each survey question, respondents were asked to use the same ranking scale to indicate how confident they were about the depth of knowledge used to determine vulnerability (Halpern et al. 2007; Selkoe et al. 2008; Fuentes and Cinner 2010). Assessing survey confidence gives greater importance to values with higher certainty (Halpern et al. 2007), and allows for gaps in knowledge to be identified (e.g., low confidence responses). Equation 1 defines vulnerability input rank as the summed product of vulnerability score and confidence, divided by the sum of the confidence.

$$\text{Vulnerability Input Rank} = \frac{\sum \text{Vulnerability Score} \times \text{Confidence}}{\sum \text{Confidence}} \tag{1}$$

Secondary input parameters were ranked by the authors, and relied on data collected from a literature review. Secondary ranks were based upon the presence and intensity of the identified parameters. Secondary input parameters and their ranks are explained in more detail in the following section.

2.4 Mapping vulnerability

GIS layers for each input parameter were compiled, and 2 m horizontal resolution rasters were produced such that each cell represented a corresponding vulnerability rank (Table 1). SLR inundation maps were generated by interpolating LiDAR ground returns as a 2 m horizontal resolution LiDAR DEM. The Federal Emergency Management Agency (FEMA) contracted Airborne 1 in 2006 to collect LiDAR data for Keālia, while LiDAR at James Campbell and

Table 1 Sea-level rise vulnerability ranked from very low (1) to very high (5) for each of the six input parameters

Parameter	Weight	James Campbell	Keālia	Kanaha
Type of inundation	2			
Groundwater		5	4	4
Marine		4	5	4
Not inundated		1	1	1
Time of inundation	2			
2057 (0.30 m)		2	4	3
2100 (0.74 m)		3	4	5
Not inundated		1	1	1
Habitat value	2			
Coastal strand		4	3	2
Upland shrub/forest		2	2	3
Wetlands		5	5	5
Soil type	1			
Hydric		3	3	None[a]
Non-hydric		1	1	
Community infrastructure	1			
3 types		5	5	5
2 types		4	4	4
1 type		3	3	3
None		1	1	1
Coastal erosion	1			
Erosion hazard		5	5	5
Hardened shoreline		3	None[a]	3
None		1	1	1

[a] Hydric soils or hardened shorelines were not found at these study areas

Kanaha were collected in 2007 by the Joint Airborne LiDAR Bathymetry Technical Center of Expertise (JABLTCX) for the U.S. Army Corps of Engineers (USACE). The LiDAR data has a vertical uncertainty of 0.18–0.20 m (1 σ) and was collected in geographic coordinates and ellipsoid heights relative to the North American Datum of 1983 (NAD83). Ellipsoid elevations were transformed to orthometric elevations using the Geoid03 model, before they were finally adjusted to the Local Tidal Datum of MSL.

To assess the percent probability that a location is inundated we accounted for the uncertainty of the LiDAR data and the SLR projections. The IPCC's fifth assessment report (AR5) Representative Concentration Pathways 8.5 (RCP8.5) scenario provides yearly global mean sea-level and associated uncertainty (1 σ) values up until the year 2100. We use values (Church et al. 2013) of projected sea-level by mid-century (0.30±0.08 m by 2057) and end-of-century (0.74±0.23m by 2100). A cumulative percent probability approach, similar to NOAA (2010), and also used by Mitsova et al. (2012) was carried out to calculate a standard-score (SS_{xy}) or the number of standard deviations a value differs from the mean.

$$SS_{xy} = \frac{\mu_s - \mu_z}{\sqrt{\sigma_s^2 + \sigma_z^2}} \tag{2}$$

The standard-score was calculated through a cell by cell approach where the difference between the projected sea-level value above MHHW (μ_s) and the DEM elevation (μ_z) was divided by the joint-uncertainty of SLR projections (σ_s) and LiDAR data (σ_z). The standard score was converted to a percent probability via a look-up table. Similar to Cooper and Chen (2013) the probability rasters were reclassified by assigning the range of probability values 0–0.49 equal to 0 (not inundated), 0.50–0.79 equal to 50 (low probability), and 0.80–1 equal to 80 (high probability).

One of the key issues for managing wetlands is identifying which areas may be impacted by marine (salty) inundation or groundwater (potentially fresh or brackish) inundation as waterfowl and vegetation are sensitive to both increased pond water levels and salinity (U.S. Fish and Wildlife Service 2011a, b). To determine areas of marine inundation, DEM grid cells hydrologically connected to the ocean were identified using the 8-sided connectivity approach. This approach identifies those flooded cells that are either directly adjacent to the ocean or connected to the ocean via adjacent grid cells in the cardinal and diagonal directions (Cooper et al. 2013b; Poulter and Haplin 2008). Inundated areas disconnected from the ocean were assumed to be flooded by rising groundwater levels (Rotzoll and Fletcher 2012). Wetland experts ranked the vulnerability of their study area to both types of inundation by considering natural and constructed features that may impede future surface inundation, as well as their dependency upon groundwater sources to maintain pond water levels.

The ability of highly managed ecosystems to successfully adapt to SLR lies in the capacity of coastal decision makers to develop and apply adaptive management plans. The time of inundation parameter ranked wetland managers' ability to implement strategies to manage 0.30 m of SLR by 2057, and 0.74 m of SLR by 2100. The IPCC's RCP8.5 scenario was used to correlate mean sea-level heights with time (Church et al. 2013). Due to time and human resource restraints most refuges do not plan beyond 10–15 years into the future (e.g., USFWS 2011a, b). Thus as a decision maker's ability to adaptively respond to the threats of SLR diminishes, managed resources become increasingly vulnerable to SLR.

To assess the ecological value of coastal sites that may potentially be flooded by SLR, experts were asked to rank the emphasis that is placed upon the management of a list of predetermined species within mapped coastal strand, wetland, and upland habitats. Managed areas that have a high habitat value were ranked highly vulnerable to SLR because these areas will result in the greatest loss in endangered and native organisms if impacted by SLR. For example, globally, coastal strand habitats are managed to support important nesting sites for sea turtles (Fuentes and Cinner 2010), resting areas for monk seals (Baker et al. 2006), and winter staging sites for migrant shorebirds (Galbraith et al. 2002). Wetland areas delineated by the National Wetlands Inventory (http://www.fws.gov/wetlands/Data/Mapper.html) are managed primarily to provide habitat for Hawai'i's four endemic and endangered waterbirds. Upland habitats are defined as the non-wetland or coastal strand area.

The presence of hydric soils is one of the primary indicators used to identify the occurrence of historical wetlands that no longer exist due to changes in hydrology and vegetation, as well as potential areas to support the establishment of future wetland ecosystems (Richardson and Gatti 1999; Van Lonkhuyzen et al. 2004; White and Fennessy 2005). Poorly drained and moderately to strongly saline hydric soil types were identified in each study area using soil maps derived from the Natural Resource Conservation Service (NRCS) web soil survey (http://websoilsurvey.nrcs.usda.gov/app/). Hydric soils included Kealia silt loam, Kaloko clay, Keaau clay, and Pearl Harbor clay. Due to the low draining potential of hydric soils, we assumed that areas with hydric soils are very highly vulnerable to prolonged flooding, whereas non-hydric soil areas have a very low vulnerability.

Coastal and wetland managers have a commitment to mitigate flood impacts upon both refuge and surrounding community infrastructure (U.S. Fish and Wildlife Service 2011a, b). To assess the proximity of flooded areas to infrastructure, we mapped a 50 m buffer around three infrastructure types including roads, 2010 U.S. census designated urban areas (http://planning.hawaii.gov/gis/download-gis-data/), and rural areas (http://www.csc.noaa.gov/digitalcoast/data/ccapregional). Flooded areas that intersect buffered infrastructure were ranked very highly vulnerable because refuge flooding may impact the nearby community.

We modeled the effects of accelerated SLR on beaches with a hybrid model that extrapolated the long-term trend from historical shoreline data collected by the University of Hawaii Coastal Geology Group (Fletcher et al. 2013), and added the change in shoreline positions due to accelerated SLR by employing the Bruun Rule (Bruun 1962) using the difference between projected and current SLR. The Bruun Rule has been highly criticized (Thieler et al. 2000; Cooper and Pilkey 2004) due to its limiting assumptions (e.g., closed sediment system, offshore-only transport), yet field and laboratory studies have argued that the Bruun Rule does provide first approximations to SLR-induced shoreline response in limited settings (Hands 1979; Mimura and Nobuoka 1995; Zhang et al. 2004). With no other viable alternative, we use the hybrid model to extend the Bruun Rule to account for the local sediment budget which often largely influence shoreline change in Hawaiian settings (Norcross et al. 2002). Erosion hazard zones mapped in this study encompass the area occupied between the current shoreline and the future shoreline position predicted under a 0.74 m rise in sea-level at the year 2100. Hazard zones projected from sandy shorelines are ranked very highly vulnerable to SLR, while those projected from hardened shorelines are ranked moderately vulnerable.

2.5 Identifying high vulnerability areas

Once each of the individual vulnerability parameters were ranked (Table 1) and mapped (Fig. 2a–f), cumulative vulnerability was determined. The final spatial variation of vulnerability for each study area was found by combining the individual vulnerability parameter rasters using a weighted geometric mean (Eq. 3).

$$\text{Final Vulnerbility} = \left(\text{type}^2 \times \text{time}^2 \times \text{habitat}^2 \times \text{soil} \times \text{infrastructure} \times \text{erosion}\right)^{\frac{1}{9}} \quad (3)$$

This approach is mathematically similar to the wetland suitability modeling methodology used by Van Lonkhuyzen et al. (2004). Primary input parameters (type, time, and habitat) were ranked higher than secondary parameters (soil, infrastructure, and erosion) because the primary input parameters most directly reflect manager's goals and objectives.

3 Results and discussion

For several reasons future sea level position is uncertain. This study uses the global mean values of RCP8.5 to map mid-century (0.3 by 2057) and end of the century (0.74 by 2100) impacts of SLR. We chose RCP8.5, the IPCC AR5 worst case scenario, because recent studies have argued that sea-level could very likely exceed 1 m by the end of the century (e.g., Horton et al. 2014; Kopp et al. 2014). Regional assessments (e.g., Slangen et al. 2012; Spada et al. 2013; IPCC 2013) of SLR provide estimates for the departure of the central Pacific (Hawai‘i's location) from the global average. For example, by the year 2100 the tropical Pacific is predicted to reach a sea-level value between 10 and 20% above the global mean (IPCC 2013). In our study we apply global SLR rates to Hawai‘i because regional models fail to

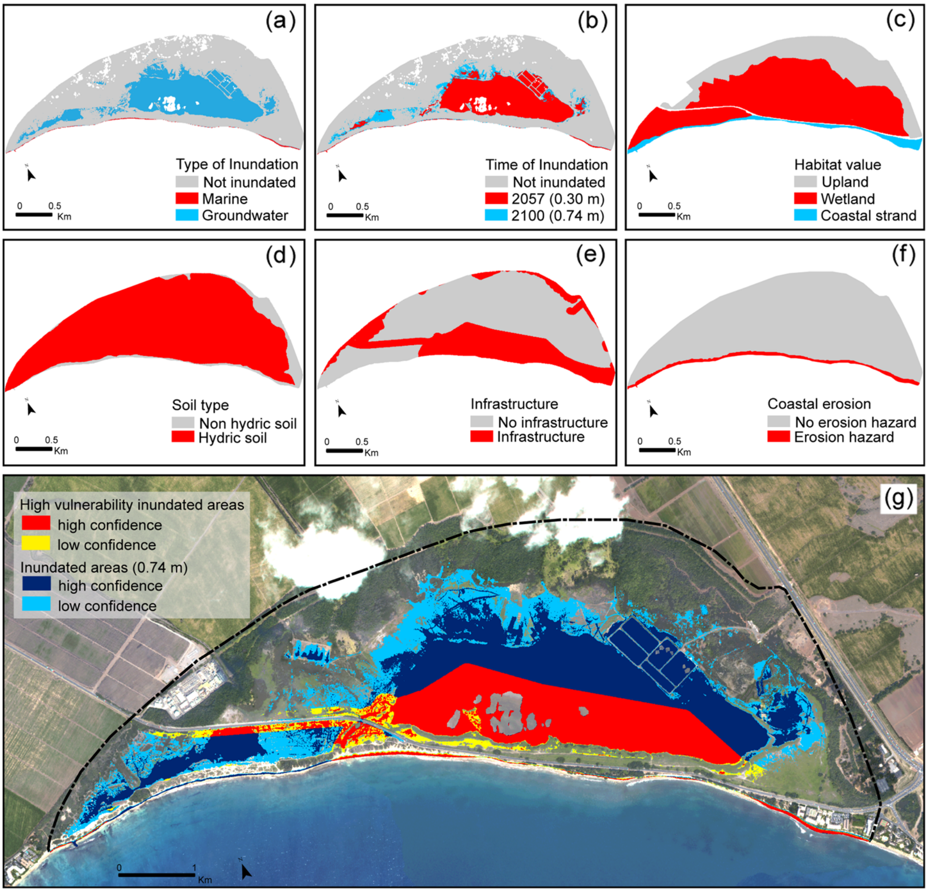

Fig. 2 Example vulnerability maps for Keālia National Wildlife Refuge. Vulnerability is defined and high confidence areas (80 % probability of flooding) are mapped for six input parameters; type of inundation (**a**), time of inundation (**b**), habitat value (**c**), soil type (**d**), infrastructure (**e**), and coastal erosion (**f**). Input parameter vulnerability maps are combined (**g**) and areas of the highest vulnerability (*red* and *yellow*) are identified as a subset of the total area inundated at 0.74 m by 2100 (*blue*). High vulnerability areas are mapped at high (80% probability of flooding) and low confidence (50%)

capture observed local weather patterns, local subsidence, produce inconsistencies among projections (Tebaldi et al. 2012), and map SLR for only one point in time (rather than providing a SLR curve).

With the exception of James Campbell wetland, experts expressed a moderate to very low ability to manage mid and end of the century SLR impacts according to the RCP8.5. The greatest gap in knowledge arose when defining long-term plans from the perspective of climate science models and wetland experts. Wetland experts typically do not plan beyond 15 years into the future due to limited staff coupled with a high number of daily responsibilities, and uncertainty in future funding. In addition much of the uncertainty in SLR projections is irreducible and stakeholders are challenged with making decisions given greater long-term uncertainty. On the other hand, current vertical uncertainty associated with LiDAR and datum errors makes it difficult to generate accurate inundation maps using short-term SLR planning targets (Cooper et al. 2013b). We suggest a compromise such as the rolling short-term (approx.

20 year) planning horizon of Donner and Webber (2014) which allows for continuous and gradual revision of policies and measures in response to the observations of impacts, new scientific findings, and improvements in SLR projections and mapping techniques. Integrating the rolling planning horizon into the management of 'high vulnerability' areas may increase decision maker's confidence in responding to a changing climate.

Poor draining, high salinity, hydric soils occupy relatively large areas at James Campbell and Keālia. The hydric soil layer at these two study areas includes both existing wetlands and surrounding upland areas which may in the future be prone to long periods of standing water due to poorly drained soils. Furthermore increases in pond water height and salinity directly impact managers' ability to provide suitable habitat for endangered waterbirds. Assuming a 0.74 m rise in sea-level we found that groundwater inundation represents over 90% of the total inundation at all study areas (Table 2). Post end of century SLR may result in increased frequency and intensity of marine flooding as sea-level breaches sand dunes, narrow outlet ditches, dikes and other natural coastal buffers. At James Campbell, wetland managers believed it would be more difficult to pump wetlands to alleviate increased groundwater inputs, while at Keālia, salty hydric soils are currently impacting waterbirds and vegetation.

Mapped wetland habitat types were found to be the most important habitats due to the role they play in preservation of endangered waterbirds, and were ranked highly vulnerable to SLR. At James Campbell and Keālia, coastal strand habitats ranked second based upon the priority each refuge gives to the management of native coastal plants, the monk seal, and sea turtles. At Kanaha, upland habitats are valued as potential sites to relocate wetland habitats.

Table 2 Area impacted by type and time of inundation under the RCP8.5 scenario

Type of inundation				
	High confidence (80%)		Low confidence (50%)	
Study Area	% Area	% Total inundation	% Area	% Total inundation
James Campbell				
Groundwater	1.4	93.2	8.2	98.2
Marine	0.1	6.8	0.2	1.8
Kanaha				
Groundwater	25	99.1	4.4	97.1
Marine	0.2	0.9	0.1	2.9
Keālia				
Groundwater	27.6	98	11.7	98.3
Marine	0.6	2	0.2	1.7
James Campbell				
2057 (0.30 m)	0.1	5.6	0.7	7.9
2100 (0.74 m)	1.5	94.7	8.3	92.7
Kanaha				
2057	24.9	98.7	0.2	3.9
2100	0.3	1.3	4.5	96.1
Keālia				
2057	21.9	77.7	3.4	22.4
2100	6.3	22.3	11.8	77.6

On the basis of infrastructure alone, the areas of the highest vulnerability are located near refuge infrastructure or along the refuge boundaries that are bordered by community infrastructure. As the number of community infrastructure types increases there is a greater risk that flooding within the refuge will impact bordering roads, urban, and rural communities. This is especially true at Kanaha, which is located in downtown Kahului, Maui and is completely surrounded by development. Accounting for land and building values in Kahului, Cooper et al. (2013a) found that a 0.75 m rise in sea-level would result in a loss of $18.7 million dollars. At Keālia the majority of infrastructure is located on the narrow coastal strip and is bordered on both sides by inundation.

All three study areas are currently experiencing chronic coastal erosion (Fletcher et al. 2013). Two out of the three study areas have roads, houses and other developed structures that will prevent the natural landward migration of beaches as sea-level rises and potentially limit the availability of coastal habitats. Coastal erosion and sea-level extreme events may be exacerbated by increased storminess and associated storm surges, however limited geographical coverage of current studies and the uncertainties related to future storminess, prevents a local assessment of impacts (IPCC 2013).

Composite vulnerability scores were compiled and the areas with the highest vulnerability rank were identified. At all three study areas, the dominant factor in determining vulnerability is whether or not an area is inundated, which is an artifact of the weighting scheme that was applied. Wetland managers, however, may find it useful to prioritize management efforts at flooded areas and thus the other input parameters were applied. Keālia most successfully exemplifies the applicability of this methodology. Figure 2g illustrates both the areas predicted to be inundated by 2100 (blue) as well as a subset of the inundated areas that ranked a higher vulnerability (yellow and red). Referring to our input vulnerability maps (Fig. 2a–f), the areas of highest vulnerability are defined as inundated hydric soil wetlands and the eroded coastal strand that fall within 50 m of infrastructure. At Keālia, infrastructure serves as the distinguishing feature in determining high vulnerability, as the majority of the flooded area encompasses wetlands habitat and hydric soils. At Kanaha high vulnerability areas are defined as inundated wetlands, uplands and coastal stand habitats that fall within the erosion hazard zone (Online Resource 1). High vulnerability areas at James Campbell are defined as inundated coastal strand environments within the erosion hazard zone, and inundated wetlands with hydric soil (Online Resource 2).

4 Conclusions

Under changing climate conditions it will be increasingly difficult to achieve all conservation objectives for habitats, species and protected areas (Hossell et al. 2003). This study is unique in that it couples expert knowledge and empirical data to define and map input parameters that systematically rank SLR vulnerability. The ranking process is translated into a series of maps that identify 'high vulnerability' areas where adaptive management efforts are needed most. The entirety of this process should encourage discussion of how managing 'high priority' or' high vulnerability' areas will impact current management objectives and goals. For example coastal decision makers should identify low lying areas and discuss how management of these areas may be impacted by marine and groundwater sources of flooding. Creating an inventory of infrastructure, valued habitats, and cultural assets that fall within the predicted areas of flooding may assist in prioritizing which flooded habitats to manage first.

The expert knowledge elicitation process greatly benefits from face-to-face surveys that allow input parameters to be adequately defined or updated so that they are truly beneficial in determining rank. This process encourages decision makers to feel more confident in focusing resources to manage 'high vulnerability' areas. Our study employed a small sample size due to limited management staff at each study site. Rather than consulting a larger group of experts who may have a general idea of how each coastal ecosystem functions, wetland managers found it more beneficial to sample a smaller number of experts who have in-depth knowledge of site specific characteristics, historical factors, and management goals of each coastal environment.

The strength of this approach is that the rankings as well as the input parameters can be tailored to reflect the goals and objectives of various management groups and regions. As regional projections of SLR, storminess, and storm surge improve they can be incorporated into the vulnerability model to refine the designation of 'high vulnerability' areas. Currently there is much uncertainty surrounding future sea-level position. Thus, the real world adaptive challenge is to make decisions given great long-term uncertainty. Flexible, adaptive management strategies that identify gaps in knowledge, develop alternative hypothesis, and gradually revise policies in response to new scientific findings and observations are necessary to produce management plans that openly address an uncertain future.

Acknowledgments This project was supported by the U.S. Department of Interior Pacific Islands Climate Change Cooperative grant # 6661281, the School of Ocean Earth Science and Technology, and the Native Hawaiian Student Engineering Mentorship Program.

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