THE ECOLOGICAL TRADEOFFS OF INVASIVE *RHIZOPHORA MANGLE* ON THE HAWAIIAN ISLANDS

by

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A professional paper submitted in partial fulfillment
of the requirements for the degree

of

Master of Science

in

Land Resources and Environmental Sciences

MONTANA STATE UNIVERSITY
Bozeman, Montana

May 2020

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ACKNOWLEDGEMENTS

Learning at Montana State University has been a worthy and rewarding experience. With gratitude, I would first like to acknowledge my advising committee Dr. Tracy Sterling, Dr. Robert Peterson, and Dr. Scott Powell. I would further like to thank Dr. Powell for his patience and kind words of encouragement as both my professional paper advisor and course instructor. The time and energy he puts into providing thoughtful and constructive feedback for every student is unparalleled; despite working remotely, he is by far the most attentive and helpful instructor I've ever had the pleasure of working with. Even amidst a global pandemic.

I would like to thank other inspiring professors like Dr. William Kleindl for his enthusiasm and willingness to talk to me for an hour on the phone while he hiked up a hill so I could pick his brain about novel ecosystems; Dr. Lisa Rew for my renewed love of plant ecology and for setting a high bar for what students can accomplish in one semester; Dr. John Long for just being cool because math is hard; and Dr. John Townsend-Mehler for reminding us that insects rule the world. I will miss this graduate program very much.

I am grateful for my parents and family who have always encouraged me to reach higher, even though they often had no idea what I was reaching for or why. This is for you, Papa.

Finally, I would like to thank Dr. Harshal Gupta who is a role model for all scientists. He is the best part of my life and together we are forever young.

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ABSTRACT

My research reevaluates historical nonnative mangrove eradication and habitat restoration goals on in Hawaii. As a part of this reevaluation, I establish a framework for weighing modern ecological services with negative ecological and socioeconomic costs. The purpose of my framework is to support land managers by providing updated assessments that accurately represent the contribution nonnative mangroves provide regarding Hawaii's current climate mitigation goals. This framework focuses on invasive *Rhizophora mangle* (red mangrove) stands as they are the most abundant type of mangrove on the islands. Recently published *Rhizophora mangle* carbon sequestration and soil accretion rates on the Hawaiian island of Molokai now allow us to estimate their ability to offset current greenhouse gas emissions and sea level rise rates. These offsets can then be weighed against the traditional view that all *Rhizophora mangle* must be removed. I estimate 4 km^2 of *R. mangle* currently sequesters 0.3% of Hawaii's current greenhouse gas reduction target and if removed would set this goal back by 8%. I also find that *Rhizophora mangle* vertically accretes sediment faster than current seal level rise and coastline erosion rates predicted for Hawaii, making this ecosystem service is a valuable asset in mitigating the effects of climate change. Historically, *Rhizophora mangle* removal is justified by a reduction in water quality and endemic bird habitat in both anthropogenic features like harbors, canals, and fishponds, and also sensitive coastline features like anchialine pools. By weighing the ecological impacts against services, I recommend strategic mangrove removal in sensitive areas and areas that hinder commerce. Mangrove removal efforts on the islands are currently active and on-going, financially supported by state and federal funding in conjunction with local conservation groups. I examine reported financial costs of previous mangrove removal projects and discuss the feasibility of island-wide eradication. Based on the cost of prior removal projects, I estimate the cost of full eradication above $41 million not including future monitoring and maintenance. With new climate mitigation goals being added to already burdened state and federal conservation and land management budgets, I promote a more pragmatic view of nonnative species by recommending strategic *Rhizophora mangle* management in lieu of full state-wide eradication.

INTRODUCTION

Mangrove forests grow in coastal intertidal zones between 40 degrees north and south latitudes worldwide (NOAA 2017). The term *mangrove* describes a diverse group of trees and shrubs based on physical structure, ecological function, and geographic location rather than a taxonomic grouping (Smithsonian 2018). Efforts to restore and conserve native mangrove systems across the globe are increasing with a renewed appreciation for the ecosystem services they provide (Smithsonian 2018). Mature mangrove forests provide a natural buffer for coastal communities vulnerable both to sea level rise and the more frequent and intense weather events caused by climate change (Rivera-Monroy et al. 2017.) A dense root system traps sediment and prevents erosion, filtering pollution from upland sources, and sequestering over twice the amount of carbon than tropical forests (Soper et al. 2019). Mangroves act as nurseries for soft-sediment invertebrates and larval fish, provide protection for nesting seabirds, and provide food and habitat for insects and reptiles (Rivera-Monroy et al. 2017).

Currently, the largest mangrove restoration efforts in the US are centered on *Rhizophora mangle* (red mangrove) in Florida, a broad-leaved evergreen tree native to coastlines around the Caribbean Sea (Kennedy et al. 2016; CSF 2019; FDEP 2019). However, and somewhat paradoxically, non-native mangroves like *Rhizophora mangle* are considered invasive in Hawaii and efforts to eradicate them involve widely respected agencies such as the National Oceanic and Atmospheric Administration (NOAA), the U.S. Fish and Wildlife Service (USFWS), the State of Hawaii, and local conservation groups like Malama O Puna (Allen 1998; Mitchell et al. 2005; FWS 2011; Agustin 2019).

Stands of *Rhizophora mangle* decrease water quality by both reducing flow with their large root structures and by dropping large amounts of organic detritus, resulting in nutrient loading and anoxia (Cox & Jokiel 1996). This issue is exacerbated by the location and mechanism of propagule disbursement—*Rhizophora mangle* is predominantly found in disturbed areas where large sediment deposits distribute propagules at the mouth of streams that flow into harbors, canals, and along the walls of pre-colonial fishponds (Figure 1).

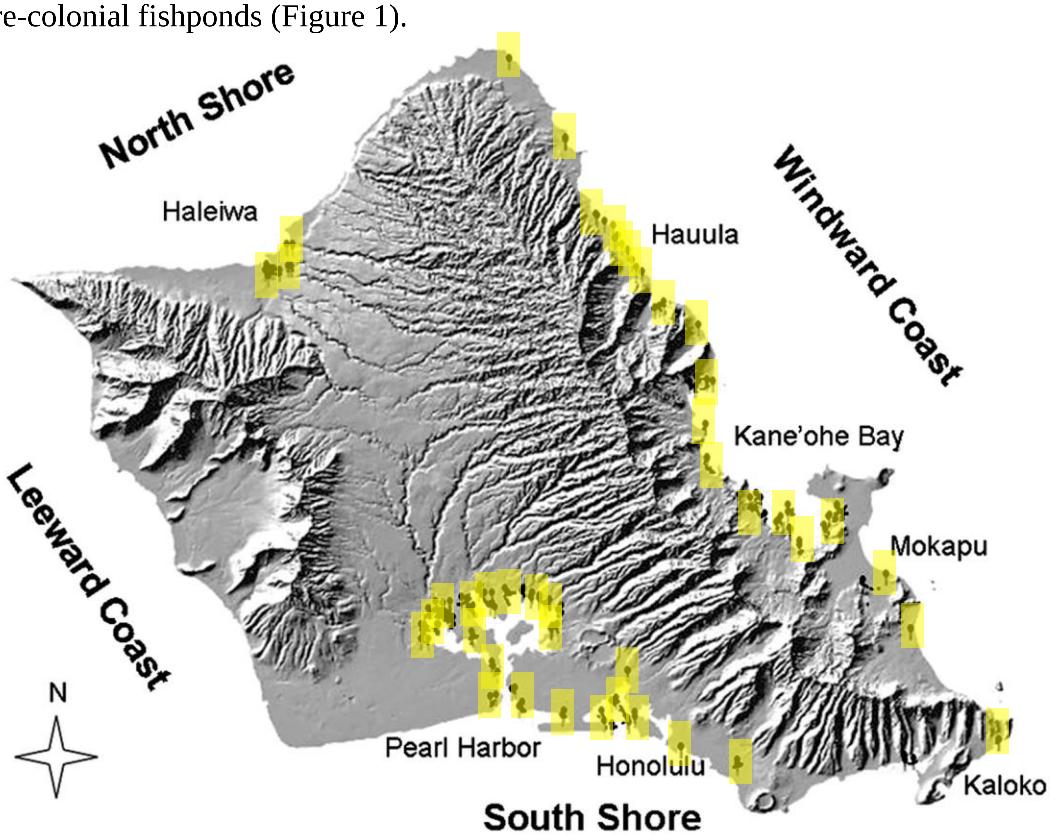

Figure 1. Location of *Rhizophora mangle* stands on the Hawaiian island of Oahu. *Rhizophora mangle* is dominantly found at the mouth of streams. Highlighted pins on map represent the presence of mangroves and not the size of stand. Map from Chimner et al. (2006)

Societal issues related to the establishment of *Rhizophora mangle* include both cultural and economic aspects. Dense stands of tall mangroves can outcompete native coastal vegetation—vegetation that is usually a small, shrub like ground cover easy to penetrate—thereby reducing coastal access for tourists and locals. Some stands block access to sacred burial grounds, ceremonial lands, and damage historical fishponds, many of which are protected by the U.S. National Register of Historic Places (Case 2017).

The negative impact *Rhizophora mangle* has on an ecosystem is usually site-specific and will outweigh their ecosystem services in sensitive conservation areas. The negative impact in sensitive conservation areas coupled with the poor water quality in commercial areas, like harbors and canals, prompts their removal elsewhere without reassessing potential benefits in other locations. For example, in Nu'upia Ponds Wildlife Management Area on Oahu, the reduction of water quality by *Rhizophora mangle* is thought to support the establishment of other non-native species, like *Batis maritima* (pickleweed) and varieties of tilapia fish originally introduced to control aquatic vegetation in the 1960s (Rauzon & Drigot 2002).

Rhizophora mangle is also thought to reduce nesting habitat in Nu'upia Ponds WMA for all four Hawaiian endemic shorebirds (Allen 1998; Rauzon & Drigot 2002). It has been generally stated that excess sediment from *Rhizophora mangle* increases turbidity in estuaries and is negatively correlated with coral growth (Allen 1998). However, more recent studies show *Rhizophora mangle* may protect coral reefs in other areas of the island by trapping large amounts of upland-derived sediments before they wash into coral habitat (D'Iorio 2003). Along the same lines, *Rhizophora mangle's* sedimentation rate has now been shown to outpace current sea level rise rates in Hawaii (Soper et al. 2019). Climate mitigation opportunities are lost when mangrove removal justifications are superimposed from one unique location to another without reevaluation.

In a pivotal paper about invasive mangroves in Hawaii, Allen (1998) first weighed the tradeoffs of *Rhizophora mangle* and concluded that despite the damage mangroves cause, the most reasonable strategy was to manage only the most sensitive sites. Allen reasoned that financial constraints alone would make island-wide eradication impossible. Although financial constraints are most certainly still true today, a modern understanding of *Rhizophora mangle's* ecosystem services may now even make full eradication *undesirable*.

Novel ecosystems where species have been introduced by humans are now ubiquitous (Hobbs et al. 2013). Ecologists are reexamining goals that aim to restore habitat to historical conditions because those conditions are rapidly changing (Miller & Bestelmeyer 2016; Davidson et al. 2018). With modern climate modelling, an updated consideration of the functional characteristics of vegetation like carbon storage of both native *and* non-native organisms—without prejudice of origin—is needed. New climate mitigation goals have been added to the already burdened state and federal conservation and land management budgets; therefore, my paper promotes a more pragmatic view of non-native species by making the argument for strategic *Rhizophora mangle* management in lieu of full state-wide eradication.

Research Goals and Objectives

The goal of this paper is to provide a modern framework for reexamining the ecological tradeoffs of *Rhizophora mangle* with an updated perspective on the ecological function and services mangroves provide. To achieve this, I will first address the history of *Rhizophora mangle* in Hawaii and why it is considered a threat to the native landscape.

This will allow the reader to appreciate the complexity of the issue before considering the potential mangroves have in meeting future coastal restoration goals. I will weigh the ecological tradeoffs of *Rhizophora mangle* as a potentially valuable member of a novel system *by synthesizing modern quantitative values of its ecosystem services relative to climate change mitigation goals*. To do this, the paper will address the following questions: 1) What ecosystem services do *Rhizophora mangle* stands provide in consideration of global climate change? 2) How significant are these services relative to current climate mitigation goals for the State of Hawaii? 3) What are the negative effects of *Rhizophora mangle* to local ecosystems? 4) What are the financial and ecological costs of removal and is full-eradication financially feasible?

To address the first two questions, I will use current carbon sequestration rates and area estimates of *Rhizophora mangle* in Hawaii to estimate the minimum amount of carbon stored per year. This number will be used to estimate the proportion of carbon stored by *Rhizophora mangle* relative to all carbon sinks reported in the “forest and land cover” category by the State of Hawaii. This will allow us to estimate what contribution *Rhizophora mangle* is currently making to support these state goals by sequestering carbon from the atmosphere. Using this information, I can also calculate the potential carbon stock loss full mangrove eradication may have on Hawaii’s state-wide Green House Gas (GHG) reduction goals.

Another climate mitigation goal in Hawaii is to reduce coastline erosion exacerbated by a sea level that is rising on average 1/8th of an inch per year worldwide (NOAA 2019). Sea level rise (SLR) as a result of rapid climate change has increased global flood risk for coastal communities by 300% since 1993 (NOAA 2019). I will compare current erosion and sea level rise (SLR) rates specifically for the Hawaiian Islands to the most recent soil accretion rates reported for *Rhizophora mangle* to determine the impact it has on maintaining coastal elevations.

To address the last two questions, I will outline the negative impact *Rhizophora mangle* has on the local environment using scientific literature and removal justifications published by the State of Hawaii. These impacts will be compared to the financial cost published for two distinct *Rhizophora mangle* removal projects using the same justifications. With these data, the cost of island-wide removal will then be estimated to discuss feasibility. Using these methods, I create a modern framework for land managers to assess the benefits and cost of continued *Rhizophora mangle* residency on the Hawaiian Islands.

BACKGROUND

Hawaii: Land Use and Vulnerability

The State of Hawaii is composed of eight main islands: Hawaii, Maui, Oahu, Kahoolawe, Lanai, Molokai, Kauai and Niiha, which have a combined area of 16,623 km^2 (HVCB.org). Hawaii is a tropical volcanic hot spot in the Pacific Ocean, more than 5,180 km from the nearest continent. Their unique geomorphic and geographic features have made them particularly vulnerable to human influences, which vary from island to island. Isolation combined with significant elevation gradients and habitat variation has led to high levels of endemism—more than 10,000 species are found nowhere else on earth (HBS 2014). The total combined area of the Hawaiian Islands is less than 0.2% of the land area of the United States but comprises more than 30%% of the nation's federally listed species (HSWAS 2010). In an area roughly the size of New Jersey, more than 4,000 nonnative species have been introduced through agriculture and urbanization. Of these, more than 900 species have naturalized, and more than 90 species are now on the federal list of noxious weeds (Smith et al. 1992).

Natural habitat for both terrestrial and aquatic species has been significantly reduced from agricultural practices, urban and residential development, military use, and tourism (Figure 2). As urbanization continues, an increase of wildfires on land overrun with nonnative grass further degrades the sensitive landscape (Smith et al. 1992).

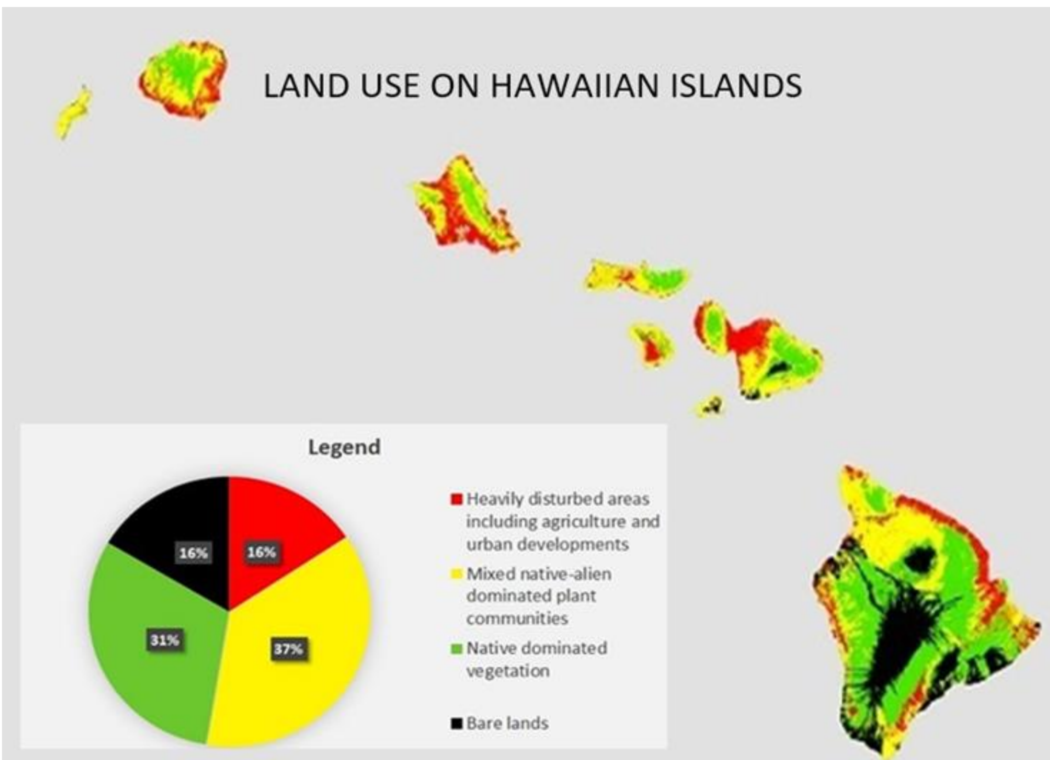

Figure 2. Coastlines are heavily disturbed (red) and novel plant communities dominate (yellow). Map credit: Pacific Islands Ecosystem Research Center (2014)

The decline in native species is mirrored by the loss of native habitat, with only 31% of the land surface covered with native-dominated vegetation (Figure 2; DLNR

2010). These conditions create serious challenges for conservation efforts; however, they may also provide a unique opportunity for new approaches to land management (Harrington & Ewel 1997; Allen 1998; DLNR 2010; Hobbs et al. 2013).

History and Controversy of *Rhizophora Mangle* in Hawaii

The Hawaiian Islands have conditions favorable to mangroves, but ocean currents and sheer distance have historically prevented mangrove propagules in places like the Caribbean and southeast Asia from reaching them (Chimner et al. 2006; Van der Stocken et al. 2019). Mangroves like *Rhizophora mangle* were introduced to Hawaii by plantation owners in the early 1900s to reduce the erosion caused by agriculture (Munro 1904; MacCaughey 1917). Their introduction is well documented by written records and aerial photography. This documentation fortunately provides ecologists ample data to examine how mangroves spread in Hawaii over the last century (Chimner et al. 2006; Soper et al. 2019). While a few different species of mangrove were introduced, only *Rhizophora mangle* has continued to expand (Figure 3, Chimner et al. 2006; Van der Stocken 2019).

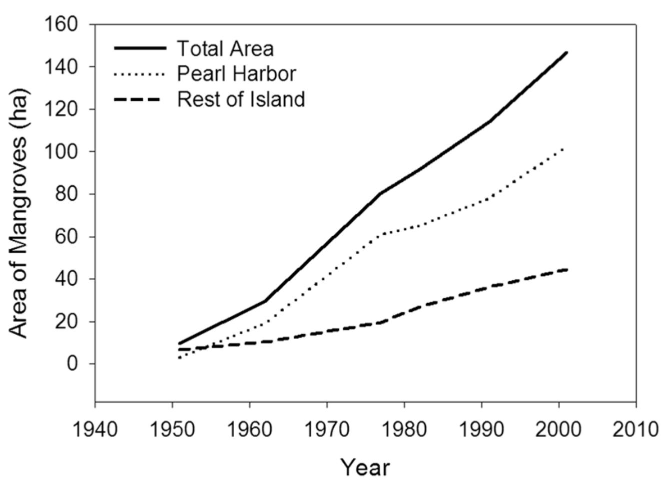

Figure 3. Spread of mangroves on the Hawaiian island of Oahu from 1951 to 2001. Over 70% of the mangroves on Oahu reside in Pearl Harbor near the city of Honolulu. Graph from Chimner et al. (2006)

The ecological tradeoffs of *Rhizophora mangle* have been debated since the 1970s when conservation laws promoted restoration of nationwide estuaries and stream systems, but until recently, most tradeoffs were subjective. A negative discourse about nonnative species gained momentum in the 1990s, using inflammatory language like ‘alien invaders’ and ‘war on invasives’ and while this language is largely outdated, the position that nonnatives threaten stability and biodiversity is one still shared by many modern ecologists (Simberloff 2015). Nonnative vegetation is thought to promote further colonization by other nonnatives in a positive feedback loop that upsets native diversity essential to healthy food webs (Harrington & Ewel 1997; Kuebbing, S., Nuñez, M. 2016). This perspective puts nonnative eradication at center stage in most modern restoration projects (Smith 1989; Eldredge & Miller 1996; Harrington & Ewel 1997).

In recent years, some public scrutiny of *Rhizophora mangle* removal has led to lawsuits by private citizens who question the widely accepted position that *Rhizophora mangle* is a noxious invasive species (Tummons 2011). However, a majority of conservation groups and volunteers are still working diligently to remove *Rhizophora mangle*—by hand, if needed—with support from local scientists like Rob Toonen, a researcher at the Hawaii Institute of Marine Biology (Funes 2018). In 2018, he was quoted as saying, “The belief that they are helping to trap sediments is simply that: a belief and not a fact.” Only a year later, research on the island of Molokai demonstrated that *Rhizophora mangle* indeed accretes sediment—up to twice the amount of its native pre-colonial mudflat counterpart—and its ability to store carbon is equally remarkable (Soper et al. 2019).

Carbon Sequestration

Greenhouse Gas Reduction Goals for the State of Hawaii

Greenhouse gases (GHG) released into the atmosphere absorb infrared radiation and trap heat. Carbon dioxide (CO_2) accounts for 81% of the excess GHG emitted through human activities (IPCC 2013). Human activities alter the global carbon cycle by adding more CO_2 to the atmosphere while simultaneously reducing the ability of natural sinks, like mangroves, to remove and store CO_2 (EPA). With GHG trapping excess heat thereby increasing global temperatures, the Hawaiian Islands will experience more severe weather events and coastal erosion from increased sea levels.

The State of Hawaii currently publicizes a GHG reduction goal with the wording "27% below 1990 levels by 2025" as a near term milestone for their ultimate goal of achieving net zero GHG emissions by 2050 (State of Hawaii 2016). The GHG emissions reported by the State of Hawaii in 1990 amounted to 23 million metric tons (MMT). By calculating 27% of 23 MMT, which is 6.21 MMT, and subtracting this amount from the 1990 level (23 MMT), we find that the state hopes to reduce GHG emissions to 16.97 MMT by 2025. The most current GHG emission data available to the public are for the year 2016 which report a GHG emission total of 19.58 MMT (State of Hawaii 2016). By subtracting the 2025 goal (16.97 MMT) from this current 2016 report (19.58 MMT), it appears the State of Hawaii aims to reduce GHG emissions by 2.61 MMT by 2025.

According to the same 2016 report, which breaks GHG emissions and offsets into ratios, mangrove forests fall under the category 'agriculture, forestry and land use' which offset GHG emissions by -6.51 MMT. In the next section, we will calculate how much of that offset is contributed by *Rhizophora mangle.* These calculations will also allow us to estimate the C storage impact island-wide eradication would have in relation to Hawaii's state-wide GHG reduction goals by 2025.

Carbon Sequestration by *Rhizophora Mangle* on Hawaiian Islands

High carbon sediment burial rates in water saturated conditions are major drivers of large carbon stocks in mangroves compared to other forested areas (Soper et al. 2019). Calculation of the average global carbon sediment burial rate for mangroves is 1.74 megagrams (Mg) organic carbon (C) per hectare (ha^{-1}) per year ($year^{-1}$) (Alongi 2012, 2014). This value ranges from 1.0 to 9.2 Mg C ha^{-1} $year^{-1}$ given differences in site conditions, root density, and age of stands (Alongi 2012). Mangroves store an average of 885 Mg C/ha globally (Kauffman & Bhomia, 2017), although an even higher value of 990 Mg C/ha was reported for oceanic mangroves in the Indo-Pacific region (Donato et al. 2011). For comparison, the U.S. Environmental Protection Agency (EPA) estimates an average of 210 Mg C/ha[1] stored in forests across the continental U.S. as of 2017 (See Appendix A for assumptions).

[1] These values include carbon in the five forest pools: aboveground biomass, belowground biomass, dead wood, litter, and soil organic and mineral carbon, and are based on state-level Forest Inventory and Analysis (FIA) data. Forest carbon stocks and carbon stock change are based on the stock difference methodology and algorithms described by Smith, Heath, and Nichols (2010).

On the Hawaiian island of Molokai, a recent study of three *Rhizophora mangle* stands between the ages of 69 and 75 years documented an annual sediment organic C burial rate of 4.5 ± 0.3 Mg C ha^{-1} $year^{-1}$ and total ecosystem C stock values ranging from 398 ± 5 to 501 ± 42 Mg C/ha, which is lower than global averages (Soper et al. 2019). The rate of carbon sequestration for *Rhizophora mangle* depends on its growth stage and age—in general, younger forest stands sequester carbon at a faster rate, but mature forest stands store more carbon overall (Toochi 2018). Older mangrove stands have greater amounts of biomass both above and below ground in root structures and have had the opportunity to sequester more carbon in soil sediment from organic detritus over longer periods of time. In other parts of the world, mangrove stands can live for hundreds to thousands of years (Ellison 2008). If managed instead of eradicated, carbon removed from the atmosphere and stored in mangrove ecosystems on Hawaii will continue increasing, further supporting Hawaii's GHG emission goals (Alongi 1998; Soper et al. 2019).

Impact of *Rhizophora Mangle* on Hawaiian Greenhouse Gas Reduction Goals

Rhizophora mangle has been documented on all the Hawaiian Islands but is most prolific on the islands of Molokai and Oahu. Molokai has approximately 150 ha of mangrove forest (D'Iorio 2003) and Oahu has 240 ha (Chimner et al. 2006). The area estimates of *Rhizophora mangle* currently available are outdated and the range of *Rhizophora mangle* has likely increased over the last 13 years (Figure 2; Chimner et al. 2006).

Mangroves on Molokai bury sediment organic C at a rate of 4.5 $\pm$ 0.3 Mg C ha^{-1} $year^{-1}$ (Soper et al. 2019). Multiplying the lower-end of this yearly organic C sediment burial rate with *Rhizophora mangle* area estimates (390 ha), mangroves on Molokai and Oahu sequester a lower-limit of 1,638 Mg C ha^{-1} each year. Over the next five years, this would total 8,190 MT or 0.3% of the 2.61 MMT GHG reduction goal for 2025 (1 Mg = 1 MT). For context, this value is equivalent to the yearly net GHG emissions from approximately 1,769 passenger cars in the US (See Appendix B for EPA calculations). The area estimates of *Rhizophora mangle* only represent two of the eight main Hawaiian Islands, where mangroves are also found albeit in smaller numbers. In addition, these lower-limit calculations do not address the amount of sequestered carbon *Rhizophora mangle* ecosystems would release if removed.

The upper limit of organic C ecosystem stocks of mangroves on Molokai total 543 Mg C ha^{-1} (Soper et al. 2019). With the assumption that mangroves on Molokai and Oahu are similar due to their similar ages, we can multiply this stock estimate to the area estimate (390 ha) to find that 211,770 Mg C would be released if removed. This is 8% of Hawaii's 2025 GHG reduction goal (2.61 MMT) by plants occupying only 0.18% of both islands combined (221,962 ha).

Determining the true impact removing Hawaii's mangroves would have on the State GHG emission goals is challenging. By removing mangroves, productive aboveground biomass can expose previously suboxic sediment C to microbial degradation and destabilization, resulting in sediment erosion (Lang'at et al. 2014). This sediment may be redeposited on reefs and other coastal systems ultimately returning sequestered C to the atmosphere as CO_2 or CH_4. Furthermore, sediment continues to show elevated decomposition of belowground biomass for at least 6 years after removal. Mangrove eradication projects on Oahu did not remove below ground root structures, suggesting that any CO_2 emissions associated with mangrove removal will carry into future emissions budgets (Rauzon and Drigot 2002; Sweetman et al. 2010).

Native Herbaceous Counterparts

Mangroves are vascular evergreen plants. In pre-colonial times, there were no vascular plants in coastal and intertidal zones where *Rhizophora mangle* now grows—the sheer biomass and longevity of *Rhizophora mangle* clearly indicates a higher carbon sequestration rate than any herbaceous native counterpart planted after their removal.

Nonnative mangroves are not replacing native tropical forests in Hawaii. While mangroves are often compared to tropical forests in terms of carbon sequestration, the tropical moist broadleaf forest ecoregions on the windward, lowland, and montane regions of Hawaii do not overlap with mangroves. Coastal lowland mesic forests are also above 100 m, while mangroves typically inhabit tidal zones protected from high-energy wave action (Comer et al. 2003).

If we compare the rate of carbon sequestration with that of a native counterpart, it would be with the various classifications of mudflats or sand flats found adjacent to mangroves in Hawaii's intertidal wetland systems. Some mudflats were created by 19th and 20th century cattle grazing and sugar crops—prompting the planting of mangroves to reduce erosion—while others occur naturally (Rauzon and Drigot 2002). Mangrove expansion into adjacent mudflats and sandflats can double or triple the amount of carbon stored in sediment (Demopoulos & Smith, 2010; Davidson et al. 2018; Soper et al. 2019). On Molokai, native mudflats adjacent to *Rhizophora mangle* demonstrated a carbon sediment burial rate of 0.35 to 0.51 Mg C ha^{-1} $year^{-1}$, eight-fold less than mangroves (Figure 4; Soper et al. 2019).

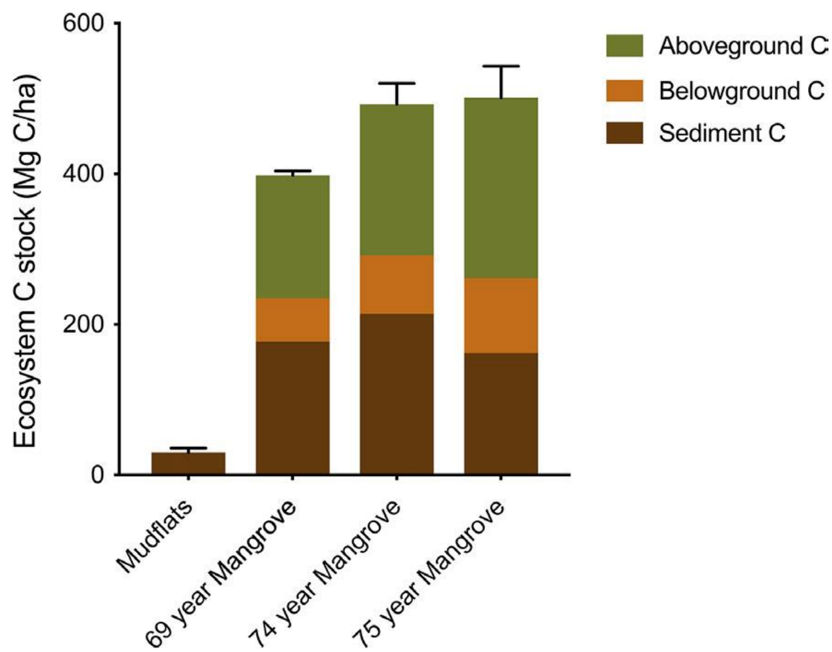

Figure 4. Carbon storage in major ecosystem components for native mudflats and nonnative *Rhizophora mangle* stands on Molokai study site (Soper et al. 2019).

By multiplying the average sediment organic C per year measured for native mudflats on Molokai (0.43 Mg C ha^{-1} $year^{-1}$) or nonnative *Rhizophora mangle* (4.5 Mg C ha^{-1} $year^{-1}$), by the number of hectares *Rhizophora mangle* occupies (150 ha) and the average number of years *Rhizophora mangle* has been on Molokai (73 years), we find that mudflats would have stored 93% less carbon had *Rhizophora mangle* never been introduced (Figure 4).

Countering Erosion and Sea Level Rise

Coastal Erosion Rates in Hawaii

The Hawaiian Islands were formed over a ‘hot spot’, a plume of magma beneath the Pacific plate. As the magma pushes through the crust, lava forms creating new crust. This crust builds up over time, creating the Hawaiian Islands. As the Pacific plate moves, these islands are being moved away from the hotspot, halting the volcanic activity that built the islands (Figure 5; Olsen 2004). Only the Big Island is still actively building in elevation, while the others are now beginning to erode (Figure 6).

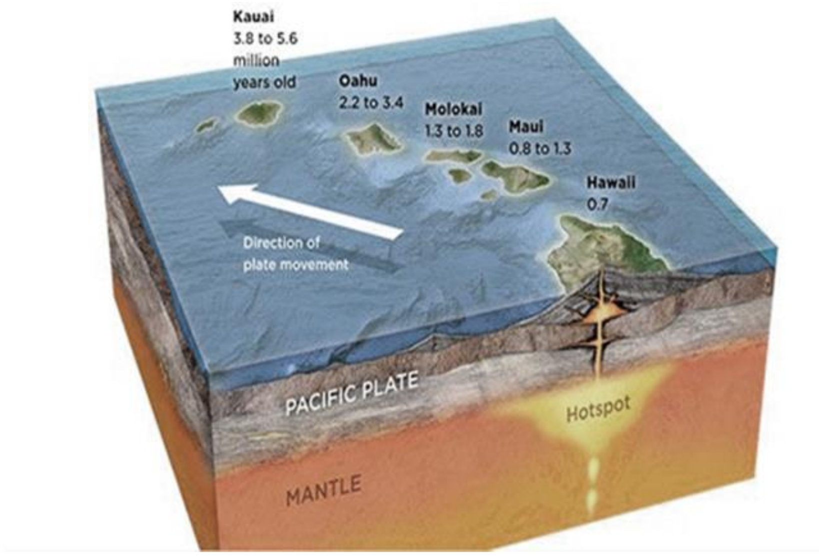

Figure 5. The Hawaiian Islands formed out of molten lava as the Pacific plate drifted over a hotspot, moving three to four inches a year. Image source: TASA Graphic Arts, Inc. (2009)

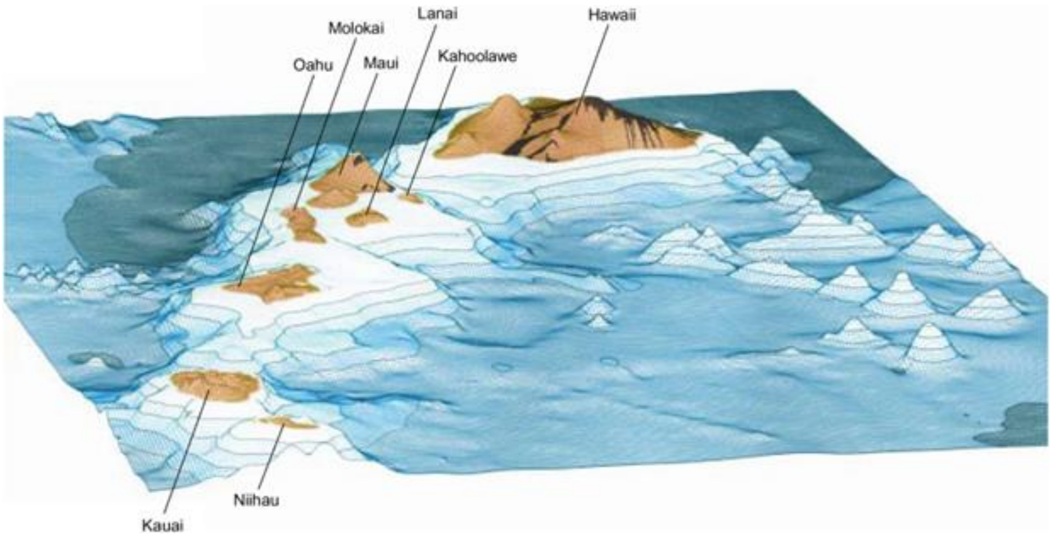

Figure 6. The Hawaiian Islands gradually diminish in height from southeast to northwest, with the newest islands being the tallest. Only the biggest island is still volcanic and building in elevation (Olsen 2004)

Islands like Molokai and Oahu are now eroding at an average rate of 0.1 to 0.2 m per year (COEMAP 2015). Coastal erosion results from natural processes such as hurricanes, storm surges, flooding, tsunamis, wave action, and wind (NOAA 2017). In Hawaii, beaches naturally erode and shift their position landward, releasing a supply of sand to the adjoining beach. The beach then remains wide and healthy even as it moves with the eroding coastline (COEMAP 2015). Local beach erosion results mainly from sand loss—if sand is not available to a beach on a chronically eroding coast, then erosion will continue, leading to narrowing and eventually beach loss (COEMAP 2015). The majority of beach loss is a result of poor land use and management related to industry, not because of vegetation like mangroves. Beach loss is a legitimate concern for locals and the tourism economy, but since *Rhizophora mangle* grows mainly in brackish waters at the mouths of streams and estuaries, they are not considered contributors to beach loss but coastal elevation gain.

Rhizophora Mangle Accretion Rates in Hawaii

At present, coastal researchers are not able to isolate the effects of long-term sea-level rise (SLR) on the stability and dynamics of the Hawaiian shoreline (NOAA). However, it seems coastal erosion rates observed on large portions of Oahu are consistent with sea level rise as predicted by the Bruun Rule, which uses local tide gauge trends to relate beach retreat to sea-level rise rates (Bruun 1962; COEMAP 2015). Vertical sediment accretion by *Rhizophora mangle* outpaces current SLR rates in Hawaii; SLR and sediment deficiencies caused by human activities are now reducing shorelines at an increased rate of 1.41 ± 0.22 to 2.95 ± 0.31 mm $year^{-1}$ (NOAA 2017), whereas *Rhizophora mangle* on the island of Molokai has been shown to accrete sediment at a rate of 7.0 to 18.0 mm $year^{-1}$ (Soper et al. 2019).

Unlike many Hawaiian native plants, mangroves like *Rhizophora mangle* may have the plasticity required to survive increased sea levels and corresponding increases in soil salinity (Goldberg & Heine 2017). Paleoenvironmental records of mangrove sediments show positive adjustments to past sea level changes (Woodroffe et al. 2016). Their persistence during rapid SLR events (5 to15 mm $year^{-1}$) over the past 18,000 years is a result of their ability to both migrate upland and to accrete their own soil using below-ground organic matter (Alongi 2008). This resilience is a focus of many climate mitigation projects which aim to restore mangrove populations along shorelines worldwide (Siikamäki et al. 2012; Murdiyarso et al. 2015).

Water Quality

Rhizophora mangle are primarily located at the mouth of streams and estuaries where intertidal wetlands historically contained no vascular plants (Allen 1998). Before human occupation, the intertidal area was dominated by algae, fungi, and some herbaceous plants. A few of these include salt-tolerant freshwater plants like *Ruppia maritima* (widgeon grass), *Sesuvium portulacastrum* (sea purslane in the ice plant family), *Heliotropium curassacvicum* (a flowering plant), and *Lycium sandwicense* (Hawaii desert thorn, a sprawling shrub) (Bryan 1915). Although *Rhizophora mangle* stores large amounts of carbon and has an impressive ability to vertically accrete sediment, it also blocks water from flowing unlike these natives.

Large scale removal efforts by the government of Hawaii are mainly focused on water quality and flow rates of invaded areas like commercially relevant harbors and historical fishponds, many of which are protected by the US National Register of Historic Places (Case 2017; Agustin 2019). Most *Rhizophora mangle* stands are found in heavily disturbed harbors and canals (Figure 7).

The DLNR-Division of Aquatic Resources (DAR) outlined the damage caused by *Rhizophora mangle* in an official Request for Bids (No. IFB-2019-02) on April 26, 2019 to remove mangroves and other non-native vegetation along 0.5 miles of navigable waterways along the Honouliuli stream through the West Loch Golf Course and stream mouth where it enters Pearl Harbor on the island of Oahu. The removal justification generally outlines *Rhizophora mangles* contribution to decreased water quality by its ability to physically restrict flow rates, increasing the amounts of organic matter and decomposing detritus in the water—which absorb oxygen, kill fish, and promote algal blooms. The restricted water flow rates also increase flood risk during significant rainfall events. Water quality issues associated with *Rhizophora mangle* pose a significant ecological tradeoff when considering management instead of full eradication. Management of mangroves in these areas will likely be indefinite and stakeholders will need to budget for their maintenance like other harbor chores such as dredging.

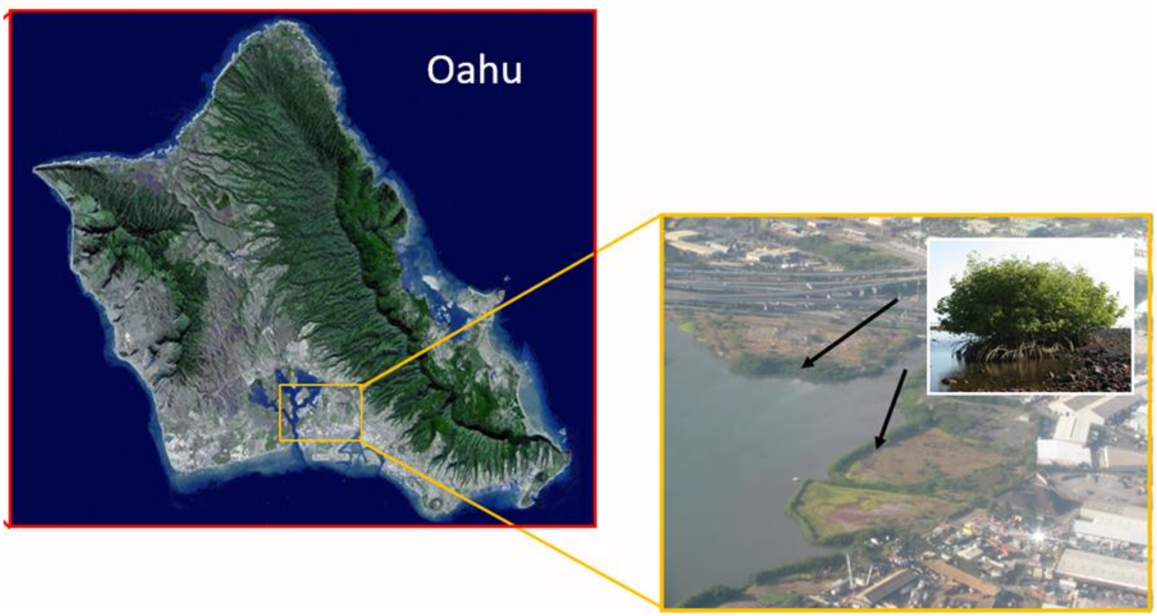

Figure 7. *Rhizophora mangle* stands border shorelines in Pearl Harbor on the island of Oahu. Satellite Image of Oahu: NASA/GSFC/METI/ERSDAC/JAROS, and U.S./Japan ASTER Science Team; Aerial Photo of Pearl Harbor © Forest and Kim Starr; Inset of *Rhizophora mangle* tree © Forest and Kim Starr

Anchialine Pools

Anchialine pools are unique inland ecosystems formed from groundwater and subterranean ocean water (Figure 8). More than half of the world's known anchialine pools are found on the Hawaiian Islands (NOAA 2020). The organisms that live here are sensitive to changes in salinity and temperature, making them particularly vulnerable to global climate change. Mangroves like *Rhizophora mangle* have colonized many of these pools, increasing the organic detritus and underwater sediment which reduce the water quality (NOAA). In the case of anchialine pools, *Rhizophora mangle* will be one of many nonnative species where removal and maintenance are necessary to restore balance. However, the introduction of Oreochromis spp. (tilapia fish), poecilids (a family of freshwater fish that includes mollies and guppies), *Gambusia affinis* (mosquitofish), and invasive prawns are thought to be the primary cause of anchialine pool degradation in Hawaii (Allen 1998).

Figure 8. An anchialine pool near Akahu Kaimu Bay, Hawaii. Photo © Allan Cressler

Anchialine pool degradation is now compounded by SLR and resulting pool inundation. As the sea level rises, pools will become more connected to the ocean, pushing groundwater higher. Approximately 1,000 new pools are projected to emerge in low-lying areas by 2050, which will likely increase the prevalence of *Rhizophora mangle* and the need for continued removal and maintenance (Allen 1998).

Plant and Wildlife Impacts

Endemic Bird Nesting Habitat

It is widely stated in gray literature and state government documents that nonnative mangroves in Hawaii physically prevent endemic birds from nesting. Nevertheless, it is challenging to find peer-reviewed studies on the subject likely because mangroves often occur with other nonnative species like *Batis maritima* (Pickleweed) that may also negatively impact endemic bird habitat (Rauzon & Drigot 2002).

In one formal study in Nu'upia Ponds WMA, biologists performed spring counts of endangered *Himantopus mexicanus knudseni* (Hawaiian stilt) nesting in fishpond margins from 1976 to 2001 and suspect that rat trappings and *Batis maritima* (pickleweed) removal made a larger impact than *Rhizophora mangle* removal (Table 1). Widespread removal and management of *B. maritima* was completed in 1986 whereas *Rhizophora mangle* removal occurred in 1999. The mean value of *H. m. knudseni* counts increased between 1986 and 1990 and remained relatively high compared to 1986. It is also noteworthy that *H. m. knudseni* did not increase immediately after removal, although data on counts after 2001 are not published.

The variables affecting native bird populations are complex and extensive. Of these variables, food and habitat loss is the most serious concern—33 of Hawaii's 44 endemic birds are listed under the Endangered Species Act and 11 more will likely be added soon (Kawasaki et al. 2019). Any efforts to restore native wetland features from urban development will significantly improve Hawaii's native bird populations.

Table 1. Hawaiian Stilt Census in Nu'upia Ponds WMA on the island of Oahu in Kailua from 1978-2001 (Table from Rauzon & Drigot 2002)

Years	Mean	Counts	Range	S.D.
1976-80	88	15	50-124	24.2
1981-85	66	14	38-109	17.8
1986-90	117	12	50-162	30.2
1991-93	106	12	75-137	19.2
1994	129	41	89-162	15.2
1995	146	18	124-187	18.2
1996	135	21	118-164	14.2
1997	129	7	107-161	20.2
1998	129	2	119-139	14.1
1999	122	2	116-127	7.8
2000	113	1	113	0
2001	129	2	112-146	0

Soft-Sediment Biodiversity

Studies on macrofauna found living among root structures of unspecified mangrove species on the islands of Molokai and Oahu showed soft-sediment biodiversity resembling that of native mangrove forests in their native regions, with a predominance of oligochaetes, polychaetes, and amphipods (Demopoulos & Smith 2010). Introduced species like *Chthamalus proteus* (a barnacle), Oreochromis spp. (tilapia fish) and *Scylla serrata* (Samoan crab) were also found, leading researchers to conclude mangroves both enhance local species richness but also facilitate the establishment of opportunistic exotics (Demopoulos & Smith 2010).

Competition with Natives

Rhizophora mangle propagules are robust and fast-growing. They express higher plasticity to variations in salinity compared to native vegetation (Goldberg & Heine 2017; Fazlioglu & Chen 2020). These traits may increase their ability to outcompete natives as SLR increases. *Rhizophora mangle* trees form thick monocultures, reducing vegetative biodiversity (Cox & Jokiel 1996). Some biologists postulate dense *Rhizophora mangle* stands provide protection for juvenile fish and wildlife as they do in their native ranges, while others maintain *Rhizophora mangle* stands foster niche expansion for other nonnative species (Cox & Jokiel 1996; Allen 1998).

COST OF ERADICATION

Financial Cost of Removal

In areas like Pearl Harbor, where up to 70% of the 240 ha of *Rhizophora mangle* stands on the island of Oahu are located, eradicating only 1.5 ha of mangrove in the Honouliuli stream cost the State of Hawaii $260,000 in 2019 (Agustin 2019). Hypothetically, if eradication costs were consistent throughout the island using these figures, it would cost over $41M to fully remove mangroves on the island of Oahu alone. This estimate does not include maintenance like reestablishing native vegetation or monitoring and removing future mangrove propagules, which would likely continue indefinitely (Allen and Krauss 2006).

Naturally, each removal and restoration project are unique. Removal sites are in water saturated soil conditions making the use of technology delicate. *Rhizophora mangle* stands are also found in areas with both endangered species and archeological features, greatly increasing the cost (Allen 1998). In the case of Nu’upia Ponds WMA, removing 8 ha of invasive mangrove and other nonnatives cost $2.5 M in 1999 (Drigot 2002). This is twice the cost of current removal in the Honouliuli stream and would likely be higher with economic inflation.

Ecological Cost of Removal at Site

The environmental impacts of *Rhizophora mangle* removal methods vary by site. Most mangrove eradication projects are completed by outside contractors who follow removal criteria outlined in a contract bid from the state (DLNR 2019). A contract to remove 1.5 ha in Honouliuli Stream and West Pearl Harbor in Oahu outlines manual labor and the use of machinery to cut trees at the water line while using a tractor to haul out debris. It also approves manual application of an herbicide triclopyr when vegetation cannot be cut below the average water level for all exposed roots, stumps, and branches (DLNR 2019).

Rhizophora mangle removal in Nu’upia Ponds WMA used machinery like a tracked bucket excavator to clear mangroves from dredge spoil islands and to reduce the surface of the islands below water level to reduce future colonization (Allen 1998; Rauzon & Drigot 2002). To manage nonnative plant growth, Nu’upia Ponds WMA now hosts an annual “Mud Ops”, where amphibious water vehicles from the adjacent U.S. Marine Corp base plow the ponds in a checkerboard fashion as a training exercise.

Wildlife managers perceive this as a benefit to both interested parties, where the Marines get to train in realistic conditions and the ponds are cleared of invasive regrowth (Rauzon & Drigot 2002; Hurley 2015).

Issues with Propagules and Regrowth.

Rhizophora mangle regrowth is a challenge for restoration projects. Propagules can be transported with tides and are viable up to a year in water (Allen and Krauss 2006). Once established, *Rhizophora mangle* seedlings grow faster than other mangrove species under a wide variety of light and salinity conditions, often outcompeting native vegetation in high salinity environments. They can establish under forest canopy in low-light conditions and can propagate throughout the year (Goldberg & Heine 2017).

DISCUSSION

In Hawaii, efforts to remove *Rhizophora mangle* by various organizations fall under two main justifications: ecological and societal. The ecological justifications aim to restore native habitat, particularly nesting habitat, for endemic species. They also focus on the water quality of streams, estuaries, and manmade structures like harbors and canals important for commerce. Societal justifications include recreational facilities like golf courses, public access to coastlines, and historical and culturally valuable structures like Polynesian fishponds and sacred burial sites. In the case of mangroves in Hawaii, often generalizations about negative ecological impacts like a reduction in bird habitat are used to justify removal for societal issues like clogging canals and harbors. By separating the justifications, land managers can more accurately weigh site-based issues with potential services.

Degradation of pre-colonial fishponds represents an important societal issue mangrove management must address. Culturally important fishponds represent a valued part of Hawaiian history and are formally protected by the state's constitution. Before Polynesian settlers transformed the area into fishponds, shallow open channels and embankments with coastal dunes threaded the region. Restoring fishponds, as opposed to reinstating coastal channels where water flows freely, will always require continued human intervention and maintenance. Strategically managing mangroves as an accepted novel part of this maintenance may also reduce financial strain on restoration budgets.

The crisis of endemic species loss in Hawaii is unquestionably due to habitat loss and land use practices. Further research is needed to determine the extent *Rhizophora mangle* negatively effects native species and endemic bird habitat; anecdotally, *H. knudseni* (native Hawaiian stilt) also seem to frequent adjacent protected wetlands behind the military police department, the Klipper Golf Course, and the percolation ditch behind the Combat Logistics Battalion 3 motor pool compound (Lodder 2011).

Mangroves continue to grow in Nu'upia Ponds WMA today. It is debatable whether mowing the wetlands with amphibious military vehicles each year responsibly manages *Rhizophora mangle.* It most certainly destroys invasive pickleweed but cannot discriminate between nonnatives slated for removal and important wetland vegetation. It also damages sensitive hydric soils vital to nutrient storage and cycling which may take decades to mature (Wang et al. 2014). The restoration strategies by land managers at Nu'upia Ponds WMA demonstrate creative problem solving on limited budgets and the necessity for more research into cost-effective coastal wetland restoration specific to Hawaii. *Rhizophora mangle* perseverance in Nu'upia Ponds WMA demonstrates a great need to reevaluate the ecological tradeoffs of *Rhizophora mangle* eradication with the limited financial resources available. As Nu'upia Ponds WMA and many other fishpond preservation and restoration sites are located along the coastline, climate induced SLR will further exasperate efforts.

The ecological impacts *Rhizophora mangle* growth has in Hawaii highly depend on location. In the case of Pearl Harbor, where 70% of Oahu's *Rhizophora mangle* populations grow, it is clear eradication is not feasible. Freshwater streams feed into the harbor creating ideal saltwater conditions for *Rhizophora mangle* while traffic from commercial vessels consistently disperse propagules. As propagules live up to one year in water, eradication projects would have to cover the entire area of the harbor—33 km^2—at once. Because Pearl Harbor will remain a highly disturbed ecosystem due to U.S. military occupancy, it is a likely candidate for a modern conservation approach using this framework.

CONCLUSION

Conservation efforts in Hawaii should take an adaptive approach to control some invasive species, like mangroves, without the intention of fully eliminating them (Hobbs et al. 2013). In a future where returning to historical natural states is not possible due to global climate change, reevaluating ecological services mangroves provide despite their evolutionary origin is necessary. Mangroves remove prodigious amounts of carbon dioxide from the atmosphere and protect coastline from the effects of rapid SLR.

Rhizophora mangle stands currently store up to eight times more carbon than their native counterpart (mudflats). They are currently sequestering 0.3% of Hawaii's 2.61

MMT GHG reduction goal for 2025. Over a period of five years, 4 km^2 of mangrove stands in Oahu and Molokai sequester the yearly net GHG emissions from approximately 1,769 passenger cars in the US. If removed, they would release ecosystem stocks of 211,770 Mg C back into the atmosphere. This translates to mangroves returning 8% of Hawaii's GHG reduction goal back into the atmosphere from an area of plants occupying less than 0.02% (4 km^2) of the islands (28,311 km^2). Finally, *Rhizophora mangle* stands vertically accrete shoreline sediment at a rate that significantly outpaces coastal erosion and SLR rates locally.

Modern conservation management in the form of shoreline mangrove "gardening" may now be possible with the advancement of GIS and remote sensing tools. Advancements in classification methods allow ecologists to discriminate mangroves from surrounding terrestrial vegetation using satellite imagery (D'iorio et. al 2007; Wang et al. 2019). Maps that provide an island-wide overview of mangrove locations can more accurately monitor expansion and identify potential candidates for removal. Stand "pruning" by selective removal could use mangrove distribution maps in conjunction with satellite and coastal buoy data on surface currents, water turbidity, salinity, temperature, and algal bloom patterns. Modern techniques for mapping mangrove distributions in tropical coastal environments will continue advancing with mangrove restoration efforts worldwide (D'iorio et. al 2007).

Embracing the value and services of novel ecosystems by using new approaches, or traditional approaches applied in new ways, will shape the development of a better management framework addressing rapidly changing ecosystems in a way that benefits the well-being of both humans *and* other species.

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APPENDIX A

EPA Carbon Storage Calculation Sheet for U.S. Forests

The *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2017* (EPA 2019) provides data on the net change in forest carbon stocks and forest area. Net changes in carbon attributed to harvested wood products are not included in the calculation.

Annual Net Change in Carbon Stocks per Area in Year t = (Carbon Stocks$_{(t+1)}$ - Carbon Stocks$_t$)/Area of land remaining in the same land-use category

Step 1: Determine the carbon stock change between years by subtracting carbon stocks in year t from carbon stocks in year (t+1). This calculation, also found in the *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2017* (EPA 2019), uses the USDA Forest Service estimates of carbon stocks in 2018 minus carbon stocks in 2017. (This calculation includes carbon stocks in the aboveground biomass, belowground biomass, dead wood, litter, and soil organic and mineral carbon pools.)

Annual Net Change in Carbon Stocks in Year 2017 = 57,687 MMT C – 57,546 MMT C = **141 MMT C**

Step 2: Determine the annual net change in carbon stocks (i.e., sequestration) per area by dividing the carbon stock change in U.S. forests from Step 1 by the total area of U.S. forests remaining in forests in year t (i.e., the area of land that did not change land-use categories between the time periods).

Applying the Step 2 calculation to data developed by the USDA Forest Service for the *Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990–2017* yields a result of 210 metric tons of carbon per hectare for the carbon stock density of U.S. forests in 2017, with an annual net change in carbon stock per area in 2017 of 0.52 metric tons of carbon sequestered per hectare per year (or 0.21 metric tons of carbon sequestered per acre per year).

Note: Due to rounding, performing the calculations given in the equations below may not return the exact results shown.

Carbon Stock Density in Year 2017 = (57,546 MMT C × 10^6) / (273,623 thou. hectares × 10^3) = **210 metric tons of carbon stored per hectare**

Annual Net Change in Carbon Stock per Area in Year 2017 = (-141.2 MMT C × 10^6) / (273,623 thou. hectares × 10^3) = **-0.52 metric tons of carbon sequestered per hectare per year***

*Negative values indicate carbon sequestration.

From 2007 to 2017, the average annual sequestration of carbon per area was 0.53 metric tons C/hectare/year (or 0.21 metric tons C/acre/year) in the United States, with a minimum value of 0.49 metric tons C/hectare/year (or 0.20 metric tons C/acre/year) in 2014, and a maximum value of 0.55 metric tons C/hectare/year (or 0.22 metric tons C/acre/year) in 2011.

*These values include carbon in the five forest pools: aboveground biomass, belowground biomass, dead wood, litter, and soil organic and mineral carbon, and are based on state-level Forest Inventory and Analysis (FIA) data. Forest carbon stocks and carbon stock change are based on the stock difference methodology and algorithms described by Smith, Heath, and Nichols (2010).

APPENDIX B

EPA Greenhouse Gases Equivalencies Calculator

Passenger Vehicles Per Year

Passenger vehicles are defined as 2-axle 4-tire vehicles, including passenger cars, vans, pickup trucks, and sport/utility vehicles.

In 2017, the weighted average combined fuel economy of cars and light trucks was 22.3 miles per gallon (FHWA 2019). The average vehicle miles traveled (VMT) in 2017 was 11,484 miles per year (FHWA 2019).

In 2017, the ratio of carbon dioxide emissions to total greenhouse gas emissions (including carbon dioxide, methane, and nitrous oxide, all expressed as carbon dioxide equivalents) for passenger vehicles was 0.989 (EPA 2019).

The amount of carbon dioxide emitted per gallon of motor gasoline burned is 8.89×10^{-3} metric tons, as calculated in the "gallons of gasoline consumed" section above.

To determine annual greenhouse gas emissions per passenger vehicle, the following methodology was used: VMT was divided by average gas mileage to determine gallons of gasoline consumed per vehicle per year. Gallons of gasoline consumed was multiplied by carbon dioxide per gallon of gasoline to determine carbon dioxide emitted per vehicle per year. Carbon dioxide emissions were then divided by the ratio of carbon dioxide emissions to total vehicle greenhouse gas emissions to account for vehicle methane and nitrous oxide emissions.

Calculation

Note: Due to rounding, performing the calculations given in the equations below may not return the exact results shown.

8.89×10^{-3} metric tons CO_2/gallon gasoline × 11,484 VMT $_{\text{car/truck average}}$ × 1/22.3 miles per gallon $_{\text{car/truck average}}$ × 1 CO_2, CH_4, and N_2O/0.989 CO_2 = **4.63 metric tons CO_2E/vehicle /year**

*A note on global warming potentials (GWPs): Some of the equivalencies in the calculator are reported as CO2 equivalents (CO2E). These are calculated using GWPs from the Intergovernmental Panel on Climate Change's Fourth Assessment Report.A