ASSESSING THE CONDITION OF HAWAIIAN COASTAL WETLANDS USING A MULTI-SCALED APPROACH

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Sandy Margriter

Thesis Committee:

Gregory L. Bruland, Chairperson

Greg Kudray

Tomoaki Miura

Christopher Lepczyk

ABSTRACT

The U.S. Environmental Protection Agency (EPA) advises using a multi-scaled approach to evaluating the integrity of wetland ecosystems that includes intensive field surveys (Level I), rapid on-site assessments (Level II), and remote or landscape-scale assessments (Level III). Although wetland condition assessment procedures have been developed, validated, and calibrated in the continental U.S., they have not yet been fully developed or field-tested with wetlands in the Pacific Islands. To my knowledge, this project was the first to field test rapid assessment methods (RAMs) in Hawaiian coastal wetlands. Three on-site RAMs, developed for coastal wetlands in the continental U.S. (Florida Wetland Rapid Assessment Procedure, California Rapid Assessment Method, and the Hawai`i Hydrogeomorphic Method), were field tested in 27 coastal Hawaiian wetlands. In addition, three different indicators of landscape condition (landscape development intensity, road density, and forest cover) were used in remote (Level I) condition assessments of the wetland buffers (100m and 1,000m radii) and watersheds. Based on the assumption that condition assessments collected at different levels of detail should provide consistent information, the results from the remote and rapid assessments were compared with detailed (Level III) field data collected on soil and water quality during prior surveys (2007-2009). Results showed that landscape indicators (development intensity and road density) calculated from readily available GIS data, particularly for the 1,000m buffers and watershed basins of each wetland, were significantly correlated with wetland soil parameters (i.e. bulk density [BD], pH, soil organic matter [SOM], total nitrogen [TN], and extractable phosphorus [ExP]) and with water quality parameters (total dissolved nitrogen [TDN], total phosphorus [TP], and $\delta^{15}N$ levels in wetland plant tissue). In addition the high degree of correlation between landscape development intensity (LDI) and road densities (in watersheds; $r = 0.95$, $p < 0.01$)

suggests that both provide comparable indicators of anthropogenic stressors in Hawaiian landscapes that vary from natural, mixed agriculture, to urban. The scores from the rapid assessment methods (RAMs), however, were somewhat weakly correlated with soil parameters (i.e. BD, total carbon [TC], SOM, and TN, ExP), yet strongly correlated with the $\delta^{15}N$ levels in wetland plant tissue, an indicator of human and animal waste. The correlations between RAM scores and landscape indicators (LDI scores and road density), calculated for the wetland buffers (1,000m) and watersheds, were also strong. Among the rapid assessment methods, CRAM showed the strongest correlations with Level I and Level III data, which suggests that although CRAM was developed for condition assessments along the Pacific West coastline (from Mexico to Oregon), with modifications to account for local conditions, the method could be applied to assessing the condition of Hawaiian coastal wetlands.

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Table of Contents

List of Tables

List of Figures

CHAPTER I: INTRODUCTION

Wetlands are among the most productive ecosystems on Earth (Mitsch and Gosselink 2007). In addition to providing unique plant communities and wildlife habitat they also act as a filter to cleanse polluted waters, buffer coastlines from storms, recharge groundwater, and attenuate floods. Wetlands perform these functions with or without the presence of humans, but are nonetheless valued for the services they provide for human welfare. In fact, wetlands have been estimated to provide ecosystem services worth approximately \$20,000 ha^{-1} yr^{-1} (Costanza et al. 1997). In recognition of the valuable ecosystem services wetlands provide, the concept of "no net loss" became a unifying and seemingly simple national goal in the United States during the 1990s. This has led to natural resource protection laws requiring that wetland areas be created or restored when wetlands are degraded or destroyed by development projects. However, few statistics exist on what functions (or ecosystem services) are gained or lost when wetlands are damaged, created, or restored (Mitsch and Gosselink 2007).

Given the complexity and diversity of ecological systems and the desire to establish baseline conditions, it is challenging to develop a suite of reliable indicators for wetlands across the United States. In order to do so effectively, the indicators must be scientifically based (Sutula et. al. 2006, Faber-Langendoen et. al. 2008) and scaled appropriately in order to discern gradients in the condition of wetlands and contributing landscapes (i.e. buffers and watersheds). Thus, in developing performance standards, indicators should be iteratively calibrated and validated with detailed field data (Fennessy et. al. 2004, Sutula et. al. 2006). In addition, data collected at different levels of detail should provide relatively consistent information about ecological integrity (Faber-Langendoen et. al. 2008) and therefore metrics collected at different scalses should be correlated (Sutula et. al. 2006, Collins et al. 2008, Faber-Langendoen et. al.

2008). Furthermore, assessments conducted at different scales can be used for cross-validation, in order to increase confidence in the methods (Collins et al. 2008).

While methods for assessing the condition of wetlands have been developed and documented for other regions such as California (Collins et al. 2008), Florida (Brown and Vivas 2005) and Ohio (Mack 2005), they have not yet been fully developed for wetlands in Hawai`i. The intent of this project is to assist in the development of indicators that can be used for assessing the condition of Hawaiian coastal wetlands at multiple spatial scales. Once wetland assessment methods have been developed, they can then be used to establish baseline conditions and to monitor changes over time.

Human Alteration of Coastal Wetlands in Hawai`i

Alteration of Hawaiian coastal wetlands began no later than 800 AD when Polynesians first colonized the Hawaiian Islands (Vitousek et al. 2004) and introduced non-native plants and animals including wetland tree and shrub species such as *Hibiscus tiliaceus* (Hau), *Thespesia populnea* (Milo), *Cocos nucifera* (Coconut), and *Ludwigia octivalvis* (Primrose willow) (Athens 1997). In addition, Polynesian agricultural practices, such as slash-and-burn techniques, were a major cause of change in vegetation communities, principally in lowland areas below 460m (Cuddihy and Stone 1990). Furthermore, Hawaiians intensively cultivated coastal lowlands through the use of wetland irrigation and rain-fed dryland systems (Kirch 1994). With the expansion of the Hawaiian population, the forests became increasingly impacted by plantings and the foraging of plants and animals for human use, as well as the invasion of rats (*Rattus spp*.) and feral pigs (*Sus scrofa*). As the mountainous slopes were denuded, water resources were diminished through the reduction of dewfall and perhaps rainfall and there was increased siltation into freshwater streams and lagoons (Handy et. al. 1991).

By A.D. 1600, between 80 to 100% of the lowland vegetation below 460m had been altered (Athens 1997) and almost all sizable saline and brackish water bodies were being utilized as fishponds (Kikuchi 1976). In addition, many of the freshwater wetlands, upslope from the fishponds, were used for taro (*Colocasia esculenta)* cultivation. However, these fishponds and taro fields were protected by cultural sanctions and the religious kapu (taboo) system. In fact, all bodies of water, from the smallest pool to the largest fishpond, were considered the home of guardian spirits, mo‘o, and contaminating them with sewage, corpses, etc. was absolutely prohibited (Kikuchi 1976). The expansion of wetlands created by the Hawaiian irrigation system allowed permanent colonization by waterbirds (e.g. *Anus wyvilliana* and *Gallinula chloropus sandwicensis*) and favored some native species of plants and animals at the detriment of others (Kirch 1994).

The arrival of Europeans in 1778 brought about drastic changes to the Hawaiian culture as well as to the landscape. By 1930 most of the Hawaiian fishponds and taro fields had been abandoned (Kikuchi 1976, Stone 1988). Furthermore, wetlands were filled and drained to make way for the expansion of agriculture, such as sugar cane and pineapple plantations. Deforestation in upland areas increased sedimentation into waterways and many coastal wetlands were transformed into harbors, military bases, or urban developments. Introduced plants and animals exacerbated the degradation of wetland ecosystems and by 1940, Hawai`i's six endemic waterbird species were classified as endangered (USFWS 2005). Since then, increasing development for airports, residential subdivisions, and resorts has continued to both directly and indirectly threaten coastal wetland ecosystems. At least 30% of the natural lowland wetlands have been lost, with the remainder seriously degraded from altered hydrology and the invasion of non-native species (Henry 2005, Erickson and Puttock 2006, Bantilan-Smith et al.

2009). In addition, there is growing concern over impacts of nutrient loading into coastal ecosystems from land-use such as agriculture and urban development (Laws et. al. 1994, Bruland and MacKenzie 2010).

Hawaiian Coastal Wetlands: Geography and Ecology

Estimates for total wetland area in Hawai`i range from 1.3 to 2.7% (Henry 2005) to 6% of land area based on National Wetland Inventory data (USFWS 2009). The "Oceania Wetland Inventory" identified over 450 wetlands in Hawai`i, including perennial streams, natural lakes, a few reservoirs, upland bogs, coastal marshes, mangrove swamps, and anchialine pools (Meier et al. 1993). Once historically a much more prominent part of the landscape, estuaries now occupy < 1% of the coastal area of the Hawaiian Islands and are more prevalent on older islands, especially Kauai and O`ahu (Nelson et al. 2007). Most of these estuaries are now highly channelized and rapidly transport runoff directly to the sea (Nelson et al. 2007).

Despite their impaired state, coastal wetlands provide an array of ecosystem services including supporting habitat for unique plant and animal species (Stone 1988, Erickson and Puttock 2006) and filtering polluted waters (Laws et al. 1999). Over 80 species of migratory waterfowl and shorebirds have been recorded in the Hawaiian Islands (Erickson and Puttock 2006). In addition a few native Hawaiian fish and shellfish use wetlands as part of their life cycle, including several species of native shrimp (`opae) and five Hawaiian stream fishes. Furthermore, Hawaiian wetlands support 33 endemic dragonfly and damselfly species and eleven species of aquatic bugs (Heteroptera) (Erickson and Puttock 2006).

Wetland Protection

Federal resource protection laws and policies in the U.S. require that wetlands be created or restored when they are degraded by human development. The federal Water Pollution Act of 1972, and later the Clean Water Act amendments, authorized the Army Corps of Engineers and the U.S. Environmental Protection Agency (EPA) to administer a permit program to protect wetlands and other special aquatic sites. The main goal of these federal programs is to maintain and improve the chemical, physical, and biological integrity of the nation's waters. As a result, projects that could potentially impact aquatic ecosystems, including wetlands, must be reviewed and permitted.

While in many cases these laws have protected wetlands from being developed they have not completely eliminated the degradation of wetlands throughout the State. In addition, the loss of wetlands has only been partially compensated for by the creation and/or restoration of wetlands (Meier et. al. 1993). Furthermore, there is evidence suggesting that created and restored wetlands do not perform certain functions (i.e. nutrient cycling, carbon sequestration) as well as their natural counterparts (Bruland and Richardson 2005, Bantilan-Smith et. al. 2009).

Wetland Condition Assessments

The U.S. EPA advises using a multi-scaled approach to evaluating the integrity of wetland ecosystems that includes remote or landscape-scale assessments (Level I), rapid on-site assessments (Level II), and intensive field surveys (Level III) (Faber-Langendoen et. al. 2008) (Figure 1.1). Intensive field surveys provide detailed information on water and soil quality, and the biotic integrity of vegetation, birds, macroinvertebrates, and fish. Rapid assessment methods, on the other hand, use observable field indicators to assess the overall health or

functional integrity of a wetland. At larger spatial scales, remote assessments use readily available geographic information to assess the natural habitat and degree of human disturbance within the surrounding landscape or watershed. Together these methods provide wetland practitioners with a set of tools for determining whether or not mitigation efforts are improving the ecological condition of wetlands. Furthermore, they can be used to assess the status of wetlands at local as well as regional scales. The ultimate goal of these approaches is to assess the status of wetlands in the U.S., establish baseline conditions, and monitor changes over time (Faber-Langendoen et. al. 2008).

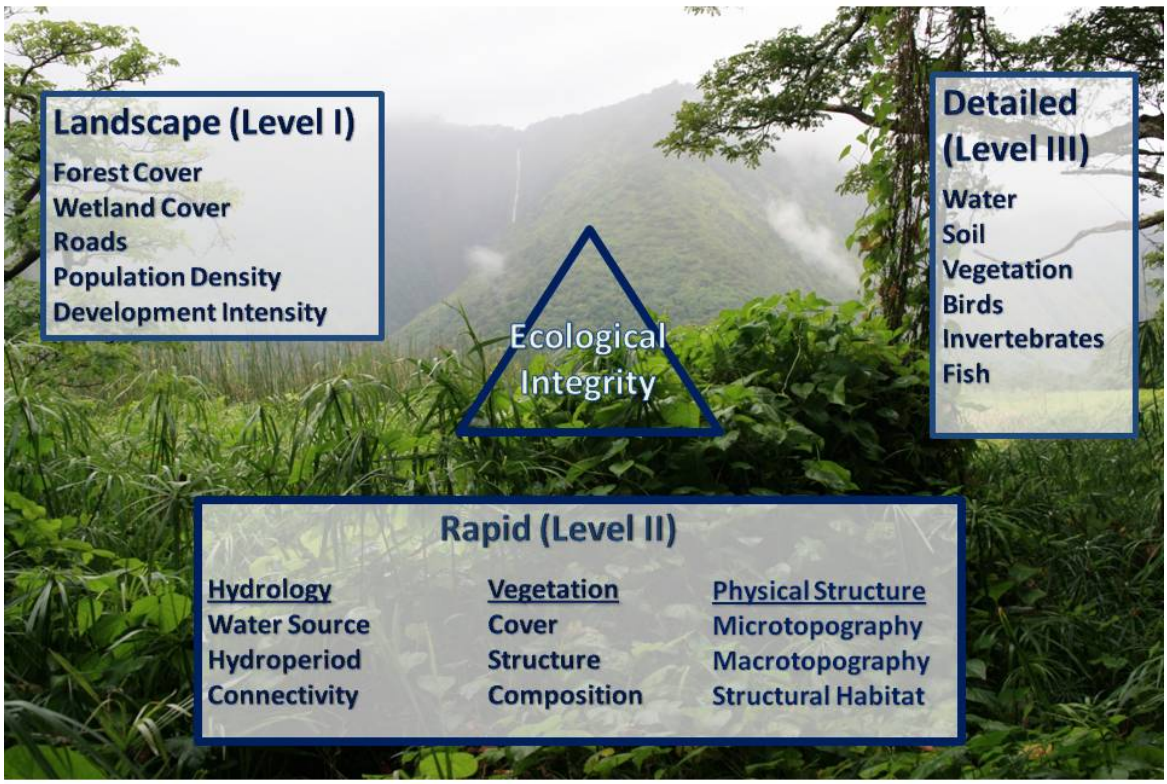

Figure 1.1. A conceptual model of a multi-scaled approach to assessing the ecological integrity of wetlands, as recommended by EPA (modified from Faber-Langendoen et. al. 2008).

Natural wetlands within undisturbed watersheds represent baseline conditions and set the standard for measuring the relative capacity of stressed wetlands to perform various

functions (Smith et al. 1995, Faber-Langendoen et. al. 2008). However, the entire "reference domain" should include wetlands that represent the range of variability within a region and wetland class as a result of natural processes and human disturbances (Smith et al. 1995). Furthermore, the desired range of ecological integrity and function should be clearly specified (Sutula et. al. 2006, Faber-Langendoen et. al. 2008) and metrics should be sensitive to departures from baseline conditions.

Wetland Condition Assessments in Hawai`i

A multi-scaled approach, using a combination of field inventories and geographic information system (GIS) analysis to assess the condition of wetlands, is in the early stages of development in Hawai`i. As part of the National Coastal Assessment (NCA), the Environmental Monitoring and Assessment Program (EMAP) surveyed all Hawaiian estuaries and bays, which included 50 sites, each sampled once between April-October 2002 (Nelson et al. 2007). Their study provided a statistical summary of the physical and biological condition of Hawaiian bays and estuaries and determined that unlike wetlands along the west coast of the continental U.S., these ecosystems showed a more narrow range of temperature, pH, and salinity. Furthermore, the values for chlorophyll *a* were several orders of magnitude below those measured at coastal sites in the continental United States. Similarly, relatively low levels of nitrogen and phosphorus were also recorded in the water column of most of the estuaries and bays (Nelson et al. 2007).

In support of a program to conduct a statewide assessment of the habitat and biotic integrity of Hawaiian streams, Kido (2002, 2008) tested bioassessment protocols and landscape-scale indicators in a preliminary study that included eight streams representing reference conditions in six watersheds on the Northeastern slopes of Hawai`i Island. Preliminary results

indicate a strong positive correlation between watershed health indices, developed from the Hawai`i Gap Analysis Program (GAP) land-cover data, and the biotic integrity of streams (2008).

Between 2007 and 2009 survey crews collected detailed field data for 40 wetlands along the leeward and windward coastlines of five main Hawaiian Islands (Bantilan-Smith et al. 2009, Bruland and DeMent 2009, MacKenzie and Bruland 2009, Bruland and MacKenzie 2010). These sites included created, restored, and natural wetlands in isolated and developed landscapes in various salinity classes (freshwater, brackish, euhaline, hyperhaline: Table 1.1). Results revealed that soil organic matter (SOM), total nitrogen (TN), and clay content were higher in natural wetlands compared to created and restored wetlands (Bantilan-Smith et al. 2009). Furthermore, natural wetlands had lower soil BD than created or restored wetlands. However, the composition of plant communities of all wetland types was reported to be predominantly non-native; only 16 of the 85 plants were identified as native species (Bantilan-Smith et al. 2009). While 27 of the 38 wetlands sampled were dominated by non-native nekton species (MacKenzie and Bruland 2009), due primarily to differences in hydrological connectivity than to differences in wetland type (i.e. created, restored, and natural), the densities of exotic fish were also significantly and positively correlated with TDN and temperature measured in wetland surface water. Furthermore, $\delta^{15}N$ levels in wetland plant tissue, an indicator of human and livestock waste, were positively correlated with the percentage of development within a 1 km radius of the wetlands and to population densities within the watersheds (Bruland and MacKenzie 2010).

Table 1.1. Distribution of natural (NWs), restored (RWs), and created wetlands (CWs) across four salinity classes sampled from 2007-2009 (Bantilan-Smith et al. 2009).

	Freshwater (<0.5 ppt)	Brackish (0.5-30 ppt)	Euhaline (30-40 ppt)	Hyperhaline (> 40 ppt)
NWs	5	9	1	1
RWs	0	6	0	2
CWs	1	1	0	1

Project Objectives

This project builds on previous research and focuses on: 1) determining the effectiveness of different remote (Level I) and rapid assessment methods (Level II) in detecting a gradient in the condition of Hawaiian coastal wetlands and their surrounding landscapes; and 2) determining whether or not the different levels of assessments are correlated with each other. In achieving these objectives, the condition of 27 of the original 40 wetlands on the islands of O`ahu, Maui, and Hawai`i were evaluated using remote and rapid assessment methods (RAMs). Results from this project are expected to aid in the development of a suite of multi-scaled indicators for condition assessments of Hawaiian coastal wetlands.

CHAPTER II: RAPIDLY ASSESSING THE CONDITION OF HAWAIIAN COASTAL WETLANDS

INTRODUCTION

The U.S. Environmental Protection Agency (EPA) advises using a multi-scaled approach to evaluating the integrity of wetland ecosystems that includes intensive field surveys (Level I), rapid on-site assessments (Level II), and remote or landscape-scale assessments (Level III; Faber-Langendoen et al. 2008; Figure 1.1). Intensive field surveys provide detailed information on water and soil quality, and the biotic integrity of vegetation, birds, macroinvertebrates, and fish. Rapid assessment methods, on the other hand, use easily observable field indicators to assess the overall health or functional integrity of a wetland. At larger spatial scales, remote assessments use readily available geographic information to assess the natural habitat and degree of human disturbance within the surrounding landscape or watershed. Together these methods provide wetland practitioners with a set of tools for determining whether or not mitigation efforts are improving the ecological condition of wetlands.

Given the complexity and diversity of ecological systems and the desire to establish baseline conditions, it is challenging to develop a suite of reliable indicators for wetlands across the United States. In order to effectively and reliably assess wetlands, indicators must be scientifically based (Sutula et al. 2006, Faber-Langendoen et al. 2008) and scaled appropriately in order to discern gradients in their condition. In developing performance standards, indicators should be iteratively calibrated and validated against independent measures of ecological condition (Fennessy et al. 2004, Sutula et al. 2006). In addition, information collected at different levels of detail should provide relatively consistent information about ecological

integrity (Faber-Langendoen et al. 2008) and therefore should correlate with one another (Sutula et al. 2006, Faber-Langendoen et al. 2008).

Research has shown that rapid and remote assessment procedures developed for Ohio wetlands were correlated with vegetation field surveys (Mack 2001). Furthermore, Florida Wetland Rapid Assessment Method (WRAP) values were correlated with water quality data (Brown and Vivas 2005) and HGM functional assessments were correlated with land cover in Maryland and Delaware (Weller et al. 2007). In California, Stein et al. (2009) determined that scores assigned to wetlands using the California Rapid Assessment Method (CRAM) were significantly correlated with multiple measures of biological community structure, including benthic macroinvertebrate indices (BMI), avian richness and plant community composition.

Wetland condition assessment methods are still in the early stages of development in Hawai`i. In support of a program to conduct a statewide assessment of the habitat and biotic integrity of Hawaiian streams, Kido tested bioassessment protocols (2002) and landscape-scale indicators (2008) in a preliminary study that included eight streams, representative of reference conditions, in six watersheds on the northeastern slopes of Hawai`i Island. Preliminary results indicate a strong positive correlation between watershed health indices, developed from Hawai`i Gap Analysis Program (GAP) land-cover data, and stream biotic integrity (2008).

As part of the National Coastal Assessment (NCA) of the Environmental Monitoring and Assessment Program (EMAP), Nelson et al. (2007) surveyed all Hawaiian estuaries and bays, which included 50 sampling sites, sampled once between April-October in 2002. Their study provided a statistical summary of the physical and biological condition of Hawaiian bays and estuaries and determined that unlike wetlands along the west coast of the continental U.S., these ecosystems showed a more narrow range of temperature, pH, and salinity. Furthermore,

the values for chlorophyll *a* were several orders of magnitude below those measured at coastal sites in the continental United States. Similarly, low levels of nitrogen (N) and phosphorus (P) were also recorded in the water column of most of the estuaries and bays (Nelson et al. 2007).

In order to compare the condition of natural wetlands with created and restored wetlands, detailed field surveys were conducted between 2006 and 2009 in 40 wetlands along the leeward and windward coastlines of the five main Hawaiian Islands. These 40 wetlands included a wider variety of coastal wetlands such as anchialine pools and coastal depressions in addition to riverine and tidal fringe (or estuarine) wetlands. The sites also included created, restored, and natural wetlands in isolated and developed landscapes in various salinity classes (freshwater, brackish, euhaline, hyperhaline). Notably, the quality of the soils tended to be higher in natural wetlands compared to created and restored wetlands (Bantilan-Smith et al. 2009) and $\delta^{15}N$ levels in wetland plant tissue, an indicator of human and livestock waste, were positively correlated with the percentage of development within a 1 km radius (Bruland and MacKenzie 2010). Furthermore, the composition of plant communities of all wetlands in the study was reported to be predominantly non-native; only 16 of the 85 plants were identified as native species (Bantilan-Smith et al. 2009). The fish communities were also primarily composed of non-native species (Bruland and MacKenzie 2010).

This project evaluates the applicability of rapid assessment methods (RAMs) for coastal wetlands in Hawai`i relative to Level III soil and water quality data collected during prior field surveys. Three rapid assessment methods (RAMs) were selected for the project: Florida Wetland Rapid Assessment Procedures (WRAP; Miller and Gunsalus 1997), California Rapid Assessment Methods (CRAM; Collins et al. 2008), and the draft Hawai`i Hydrogeomorphic Method (HHGM; SAIC 2004). These methods were selected because they are well documented,

they vary in the degree of qualitative and quantitative metrics used as indicators of wetland condition, and they have proven to be sensitive tools for assessing wetland ecosystems in other coastal states (Mack 2005, Fennessy et al. 2004, Brown and Vivas 2005, Collins et al. 2008). In order to determine if these rapid assessments were correlated with more detailed (Level III) condition assessments, the final RAM scores were compared with water (Bruland and MacKenzie 2010) and soil quality (Bantilan-Smith et al. 2009) data collected during prior surveys.

Rapid Assessment Approaches

Wetland rapid assessment methods (RAMs) use easily-identifiable field indicators to assess overall ecosystem integrity or specific functions, relative to intact well-functioning wetland ecosystems (Fennessy et al. 2004, Faber-Langendoen et al. 2008). The intent of these assessments is to rapidly and accurately quantify the extent to which ecosystems depart from full ecological integrity. The best RAMs provide a quantitative measure of wetland condition on a continuum ranging from full ecological integrity to highly impaired (Fennessy et al. 2004). The challenge with RAMs involves developing methods that are both simple and efficient while also accurate and repeatable (Smith et al. 1995, Sutula et al. 2006). In addition, the RAMs should be scientifically based so that they can be used for a variety of purposes such as prioritizing sites for conservation, identifying sites for restoration, and monitoring trends in wetland condition at specific sites or multiple sites over time.

A critical component of condition assessments is the selection of baseline conditions that serve as a reference system or assessment end point(s). For instance, depending on the chosen RAM, a reference wetland can represent culturally-unaltered (natural) wetlands or the "best attainable" condition, given existing constraints on the landscape (Sutula et al. 2006). In either

case, the “reference domain” should include wetlands that represent the range of variability that can occur within a region and wetland class as a result of natural processes and human disturbances (Smith et al. 1995).

Most wetland RAMs are designed to assess condition or function. While condition assessments tend to focus on the overall integrity or health of an ecosystem (Collins et al. 2008), the functional approach focuses on measures (or metrics) of biotic and abiotic processes (Brinson et al. 1995). The term function specifically refers to processes that involve the transformation of energy and materials, but can also be more broadly applied to include the provision of habitat for native species (Brinson et al. 1995). It is possible therefore to assess natural wetland functions without regard to their significance to society (Mitsch and Gosselink 2007). In addition, the condition or functional capacity of a wetland can be estimated using quantitative or qualitative metrics. For example, topographic complexity, and interspersion of vegetation communities are assessed using qualitative narrative descriptions or graphical representations, while species abundance and soil depth are measured quantitatively.

The Hydrogeomorphic (HGM) approach uses a combination of qualitative and quantitative metrics to estimate gains or losses in wetland functions relative to reference conditions. In estimating, a wetland’s capacity to cycle nutrients, the depth of the O and A soil horizon is measured quantitatively, while sediment delivery is rated (low, medium, or high) based on narrative descriptions of observable indicators of features such as rills and gullies. However, in order to accurately apply this metric, the natural range of depths for the O and A soil horizons must first be determined for the reference wetlands in that specific region. The HGM scores are then assigned based on the degree of departure from the reference standard.

In contrast, the CRAM assumes that ecological integrity can be determined by the hydrology, physical structure, biological structure, and landscape context of the wetland (Sutula et al. 2006, Collins et al. 2008). Thus, the indicators for the overall condition of a riverine wetland include water source, hydroperiod, plant community structure and composition, topographical complexity, and landscape connectivity (Collins et al. 2008). The CRAM uses qualitative attributes to estimate the condition of all attributes, except for biological structure, which are described more quantitatively. Furthermore, the CRAM rates wetlands relative to their capacity to support wetland dependent species and therefore can and should be calibrated with field data (Sutula et al. 2006) on biological integrity.

Hydrology

Hydrology is the primary driver of wetland processes (Mitsch and Gosselink 2007). Information on a wetland's hydroperiod (frequency and duration of flooding), hydrologic connectivity, and water source are often used as indicators of overall ecosystem health (Collins et al. 2008) or in evaluating the capacity of a wetland to perform a wide range of functions, from nutrient cycling to habitat support (Smith et al. 1995, Shafer and Yozzo 1998, Hruby 2001, Hauer et al. 2002, SAIC 2004, Faber-Langendoen et al. 2008). Human modifications, such as roads, berms, and channels, constrain the natural flow of water, which can significantly impact natural processes and reduce the ecological integrity of a wetland.

Water Quality

Aside from hydrology, the quality of water entering aquatic ecosystems is affected by human activities in a watershed via point and non-point sources of pollution. Excess nutrients and sediments that drain into wetlands can cause various problems such as the spread of

invasive plants, increased turbidity, and a host of problems associated with eutrophication, including low levels of dissolved oxygen and excessive algal growth (Dunne and Leopold 1978).

Typically, water quality is evaluated primarily on chemical and to a lesser degree on physical characteristics (e.g. turbidity). However, hierarchical sets of indicators, that include land-use stressors and indices of biological integrity, have been suggested for more comprehensive analysis of water quality (Karr and Yoder 2004, Wang 2001). Specifically, bioassessments have been shown to more accurately detect and quantify aquatic life impairments than chemical measurements alone (Karr and Yoder 2004). In addition, stable nitrogen isotopes in wetland vegetation have been used successfully to identify sources of dissolved inorganic nitrogen (DIN). A study in Hawai`i, for example, suggested that higher $\delta^{15}N$ levels in plant tissue of urban coastal wetlands was due to increases in DIN from human waste (Bruland and MacKenzie 2010).

Landscape Indicators

Various landscape indicators such as forest cover, urban land-use cover, agricultural land-use cover, impervious surface cover, and width of riparian buffers have all been linked to variables such as levels of N and P in surface water and the diversity of native fish, macroinvertebrates, and birds in aquatic ecosystems (Allan et al. 1997, Wang et al. 2001, Gergel et al. 2002, Houlahan and Findlay 2008). There is also substantial evidence showing that roads can significantly reduce biodiversity in wetland habitats (Findlay and Bourdages 2000) and adjacent ecosystems for distances up to one km from the roadway (Forman and Deblinger 2000). Roads fragment landscapes, increase surface runoff of toxic chemicals, and provide vectors for invasive species (Millar and Wardrop 2006, Weller 2007). Miller and Wardrop

(2006) found that while quality of wetland habitats corresponded with road density, habitat quality was not correlated with distance to nearest road. Similarly, Weller et al. (2007) found that road variables (density and nearest) were only weakly related to wetland condition. Others have suggested the potential exists for a considerable lag time, on the order of decades, before some taxa show a response to road construction (Forman and Deblinger 2000).

Finally, patch size and landscape connectivity have been suggested as a model for estimating plant species diversity and abundance as well as overall habitat complexity (MacArthur and Wilson 1967, Turner et al. 2001). In theory, large wetlands connected to other habitats that are similar in form support more species since they are better able to provide refuge and alternative habitat patches for metapopulations of wildlife (Shafer et al. 2002, Collins et al. 2008). The complexity of the aquatic edge, native plant cover and abundance, interspersion of patches, and richness of aquatic and terrestrial habitats also influence the number and types of species a wetland can support (Shafer and Yozzo 1998, Hauer et al. 2002, Collins et al. 2008).

Soil Quality Indicators

Soil properties are often described in various levels of detail during condition assessments (Brinson 1993, Bruland and Richardson 2005, Bruland and Richardson 2006, Nelson et al. 2007, Bantilan-Smith et al. 2009). Hydrogeomorphic methods, for example, use quantitative measurements (e.g. soil depth) and qualitative descriptions (e.g. soil color and texture) to assess the quality of wetland soils (Hauer et al. 2002, SAIC 2004). A thinner, lighter colored A horizon (relative to baseline conditions) or an absent A horizon altogether usually represents human land-use activities that have resulted in the removal of top soil, while a thick A horizon can indicate increased deposition of soil eroded from upland areas (Hauer et al. 2002).

Furthermore, properties such as soil organic matter (SOM) play an important role in wetland ecosystems by reducing bulk density and providing a major source of plant nutrients, including phosphorus, nitrogen, and sulfur (Bruland and Richardson 2005, Bruland et al. 2009). Soil organic matter is also a good predictor of denitrification potential, as well as fish and invertebrate species richness (Findlay et al. 2002). In undisturbed sites, the amount of SOM represents the balance between net primary productivity and decomposition, both of which are determined by temperature, hydrology, and nutrient availability (Mitsch and Gosselink 2007). Disturbed sites, on the other hand, tend to have more compacted soils and increased soil bulk density (Bantilan-Smith et al. 2009) with less SOM (Bruland and Richardson 2006, Bantilan-Smith et al. 2009). Older, natural wetlands (of similar type) generally have finer-grained nutrient rich organic soils, while younger or disturbed wetlands tend to have accumulated less organic matter and have fewer well developed nutrient pools (Shafer et al. 2002). In other words, created or restored wetlands tend to have lower SOM than natural unaltered wetlands (Shaffer and Ernst 1999, Bruland and Richardson 2006, Bantilan-Smith et al. 2009).

Vegetation Communities

Vegetation communities are determined by climate, geomorphology, and hydrology, and as with soils, represent longer-term human impacts to the ecological integrity of wetlands. In turn, the type and structure of vegetation drives ecological functions such as nutrient cycling, retention of sediments, dissipation of floods, and supporting ecological diversity in a wetland (Shafer and Yozzo 1998, Hauer et al. 2002, Collins et al. 2008). For example, dense vegetation contributes to surface roughness, which dampens wave or flood energy and helps to retain water after floodwaters recede (Shafer and Yozzo 1998). Overlapping layers of vegetation usually result in greater species diversity and more complex food webs among macroinvertebrates,

fishes, amphibians, and birds (Collins et al. 2008). In addition, multiple vegetation layers enhance hydrological functions, including rainfall interception, reduced evaporation from soils, enhanced water filtration, and dissipation of floodwaters (Collins et al. 2008).

Changes in vegetative cover can represent altered hydrology (e.g. reduced freshwater inputs), nutrient loading, or direct land-use impacts. Mack (2006) found that vegetative indices developed for different types of wetlands in Ohio were strongly correlated with RAM scores and with landscape development intensity (LDI). Similarly, Cohen et al. (2004) developed indices of floristic quality for herbaceous depressional wetlands in Florida, which were strongly correlated with LDI occurring within 100m of the wetlands.

However, it can be challenging to develop a list of native species that characterize reference standards and provide reliable indicators of anthropogenic disturbances (Shafer et al. 2002). Nonetheless, there is general agreement that an increase in the overall cover and abundance of non-native plant species lowers the capacity of a wetland to support habitat for native food webs. Therefore, almost all rapid assessments include metrics (i.e. non-native plant species cover and abundance) as indicators of wetland habitat quality. In addition, the composition, cover, and structure of vegetation develop along salinity gradients (Brinson 1993, Bantilan-Smith et al. 2009) therefore; reference standards should be different for each wetland class (Shafer and Yozzo 1998, Hauer 2002, Collins et al. 2008).

Wetland Classification Systems

Wetland classification systems improve the accuracy of wetland condition assessments and enable more sensitivity in detecting differences between altered and natural wetlands (Smith et al. 1995). Classification systems generally categorize wetlands based on hydrology, vegetation, geomorphology, and to some degree soils. The HGM classification system, for

example, focuses on abiotic features and groups wetlands into hydrogeomorphic classes based on their geomorphic setting, dominant source of water, and hydrodynamics (Brinson 1993). The CRAM classification system is similar to the HGM system, with both using the characteristics of vegetation as indicators of habitat quality, rather than as attributes for classifying wetlands. The Cowardin (1979) classification system, on the other hand, emphasizes vegetative characteristics and associated ecological factors, which facilitates mapping changes in wetland area over time using remotely sensed imagery.

Using the HGM approach, wetlands are grouped into seven hydrogeomorphic subclasses: tidal fringe, depressional, slope, mineral soil flats, organic soil flats, riverine, and lacustrine. Although there is significant overlap between the CRAM and HGM classification systems, CRAM defines estuarines (tidal fringe) as having a distinct inlet and outlet channel (i.e. river) (Collins et al. 2008), which excludes many of the groundwater driven tidal wetlands along the Hawaiian coastlines. Furthermore, while the indicators for assessing tidal fringe, depression, and riverine wetlands have been thoroughly documented in the "National Guidebook for Application of Hydrogeomorphic Assessment to Tidal Fringe Wetlands," (Shafer et al. 1998) the guidelines were limited to wetlands in the continental US. Wetlands in the Hawaiian Islands were intentionally excluded because they "are functionally different enough from those in the more temperate regions of the country" (Shafer et al. 1998). Detailed information on the functions and processes of Hawaiian wetland ecosystems are sorely lacking (Shafer et al. 1998).

Objectives

Although wetland condition assessment procedures have been developed, validated, and calibrated in the continental U.S., they have not yet been fully developed or tested with

wetlands in the Pacific Islands. Furthermore, the relative capacity of Hawaiian coastal wetlands to perform specific functions such as nutrient retention, shoreline protection, and wildlife habitat support is currently unknown. The overall goal of this project was to test RAMs developed for coastal wetlands in the continental U.S. in order to determine their applicability to Hawaiian coastal wetlands. Ultimately, the project will assist in the development of methods that can be reliably used for assessing the ecological condition of coastal wetlands in Hawai`i.

This objective allowed for several questions and hypothesis to be investigated. First are rapid assessment scores that use field observations and landscape indicators of wetland condition correlated with soil and water quality parameters measured during intensive field surveys? Given that rapid field assessment scores should be indicative of ecological integrity I hypothesized that they would be:

- Negatively correlated with nutrient levels measured in wetland surface water: total dissolved nitrogen (TDN), total phosphorus (TP), and phosphates (PO_4); and with $\delta^{15}N$ levels in wetland plant tissue (an indicator of human and animal waste)
- Negatively correlated with soil bulk density (BD), and
- Positively correlated with soil total carbon (TC), total nitrogen (TN) , and extractable phosphorus (ExP)

Secondly I questioned whether or not specific metrics were correlated with rapid assessment method (RAM) scores? Given that a single road can increase invasive species, decrease water quality, and interfere with the natural flow of water, I hypothesized that wetland condition assessment scores would be negatively correlated with distances between the edge of wetlands and the nearest road. Furthermore, I hypothesized that larger wetlands provide greater

habitat diversity and a higher capacity to buffer land-use impacts, and therefore would receive higher condition assessment scores.

METHODS

This study was conducted in 27 coastal wetlands on the islands of O`ahu, Hawai‘i, and Maui (Figure 2.1) and included created, restored, and natural wetlands, representing isolated and developed wetlands across a spectrum of freshwater, brackish, euhaline, and hyperhaline conditions (Bantilan-Smith et al. 2009). In addition, the selected wetlands spanned a gradient of disturbances, within different HGM subclasses and ecoregions.

The wetland assessment areas were determined from existing GIS layers of wetlands (USFWS 2009) and water bodies (USGS 2000) and were based on clear breaks in surface hydrology, sediment supply, and geomorphology per CRAM guidelines (Collins et al. 2008). In most cases these assessment areas followed the wetland areas as delineated by the National Wetland Inventory, which used the Cowardin classification system (1979) to map the wetlands (USFWS 2009). However, some wetlands, such as the Percolation Ditch on O`ahu, were not mapped at all by NWI, while other wetlands, such Nu`u (Maui) and Kamilo (Hawai`i) were poorly delineated; in these instances the wetland assessment areas were delineated using GIS layers of water bodies (USGS 2000) and Quickbird satellite imagery (DigitalGlobe 2008). Because larger wetlands tend to have greater structural complexity (Collins et al. 2008), which results in higher CRAM and HHGM scores, at least two partial assessment areas were used for wetlands greater than 2.5 ha. The partial assessment areas were limited to 1 ha. Scores from all partial assessment areas were averaged to provide a final score for the entire wetland assessment area.

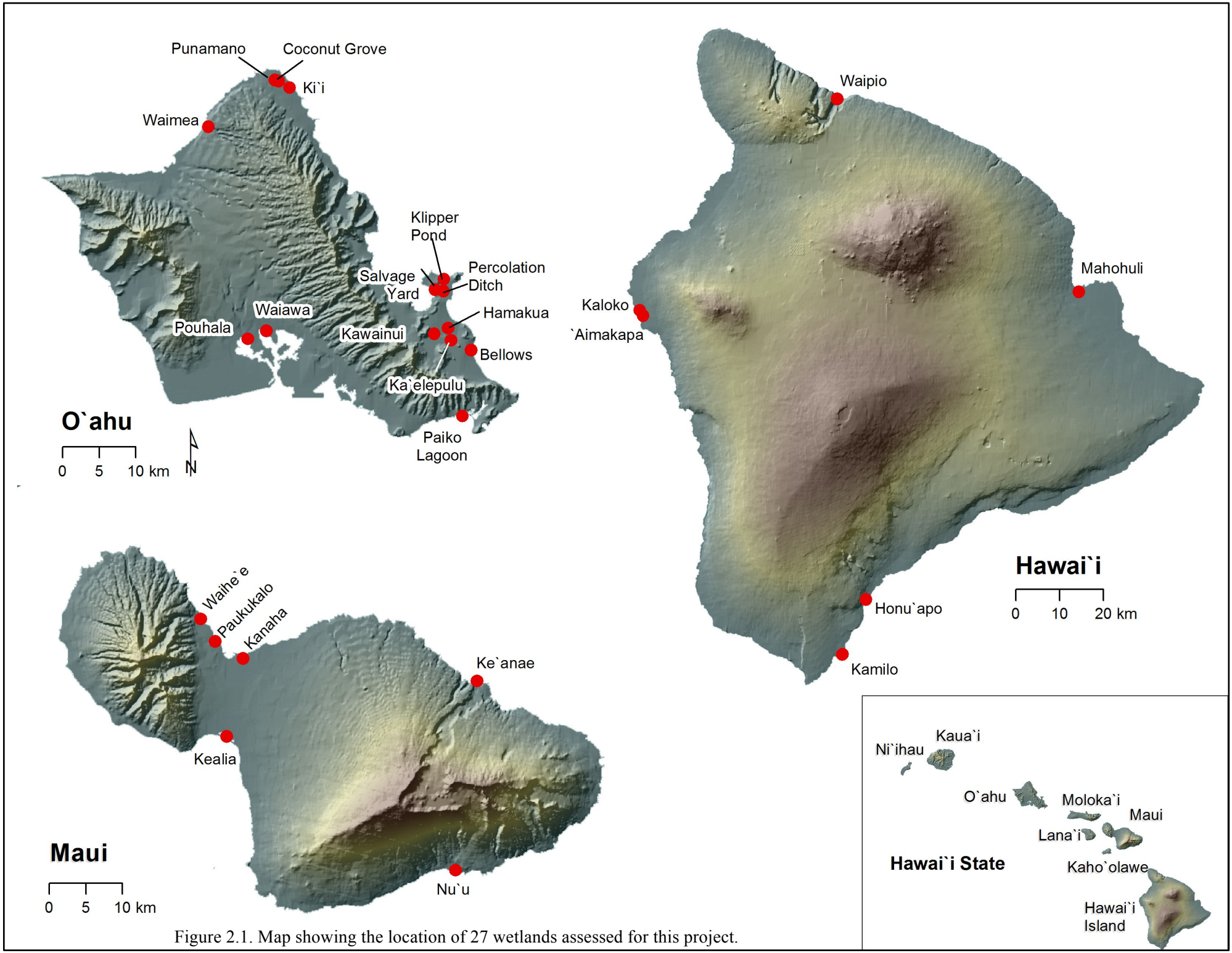

Figure 2.1. Map showing the location of 27 wetlands assessed for this project.

The condition of the wetlands were assessed using three different on-site rapid assessment methods: Florida Wetland Rapid Assessment Procedures (WRAP [Miller and Gunsalus 1997]), California Rapid Assessment Methods (CRAM [Collins et al. 2008]), and the draft Hawai`i Hydrogeomorphic Method (HHGM [SAIC 2004]). These methods were selected because they are well documented, they vary in the degree of qualitative and quantitative metrics used as indicators of wetland condition, and they have proven to be sensitive tools for assessing wetland ecosystems in other coastal states (Mack 2005, Fennessy et al. 2004, Brown and Vivas 2005, Collins et al. 2008). Of the three methods WRAP was the most rapid and least quantitative, whereas the HHGM method was the most detailed and labor intensive. In addition, the WRAP and CRAM scores both provide a measure of the overall health or condition of a wetland, whereas the HHGM method addressed the capacity of wetlands to perform specific functions. Table 2.1 shows the functional integrity categories for assessing estuarine wetlands using the Hawai`i HGM method compared with CRAM and WRAP.

Table 2.1. List of categories used in rapidly assessing the condition and function of coastal wetlands

WRAP	**CRAM**	**Hawai`i HGM (Tidal Fringe Wetland)**
Wildlife Utilization	Landscape Connectivity	Dissipation of Energy
Wetland Canopy	Buffer	Retention of Imported Elements
Wetland Ground Cover	Hydrology	Characteristic Plant Community
Habitat Support Buffer	Physical Structure	Characteristic Invertebrate Food Webs
Hydrology	Biotic Structure	Characteristic Vertebrate Habitats
Water Quality Input		Habitat for Threatened and Endangered Species

Both the CRAM and HHGM use different metrics to rate a wetland, depending on the wetland class. Thus, wetlands were first classified as tidal, riverine, or depressional, according to HGM guidelines, before they were assessed. The final rapid assessment scores were calculated on continuous gradients: 1-3 (WRAP), 1-120 (Hawai‘i HGM), and 1-100 (CRAM). Since the maximum HHGM scores vary for each hydrogeomorphic class (80 for tidal fringe,

120 for riverine, and 100 for depressional), the final HHGM scores were rescaled from 0 to 1 (i.e. final scores were divided by the maximum possible score for each category), with final scores representing overall wetland functional capacity. Furthermore, because one of the goals of the project was to test the HHGM and determine its utility in assessing function rather than values the human value attributes, referred to as opportunity variables, were excluded from the analysis.

With CRAM, a numerical value was assigned to 14 different metrics, determined from alternative narrative descriptions for each metric. While some metrics, such as interspersion of vegetation and hydrologic connectivity were based on qualitative visual observations, other metrics such as biotic patch richness and native plant species richness were more quantitative (Collins et al. 2008). An overall CRAM score that represented the best achievable condition for that wetland type was calculated for each site. The final CRAM score was then calculated as a ratio of the total score divided by the maximum possible score.

The WRAP method was used to evaluate the wetland sites based on six variables noted in Table 2.1. Scores from 0 to 3 were assigned to each variable based on narrative descriptions of indicators observed while walking at least half the wetland perimeter. Although a liberal interpretation of "desirable species" that included plants introduced by Polynesians was used when rating the categories for overstory/shrub canopy and vegetative ground cover, this had relatively little effect on the final WRAP scores. After summing the scores for each category, the final WRAP scores were then divided by the maximum possible score so that they would be comparable to CRAM and HHGM scores.

Statistics on wetland area and distance to nearest road (USGS 2000) were calculated in a GIS (ArcGIS, Version 10.0, ESRI, Redlands, CA). The distances were calculated between the

wetland edges and the nearest road (USGS 2000). The areas were calculated for wetlands, delineated using CRAM guidelines as previously described.

Soil parameters including bulk density (BD), total nitrogen (TN), total carbon (TC), soil organic matter (SOM), and extractable phosphorus (ExP) collected from intermediate and wetter zones of the wetland sites were used in the analysis; results from soils collected in the drier zones were omitted because they were more representative of upland terrestrial soils instead of those most typically found in wetlands. The distributions of many of the soil and water parameters (i.e. TN, TC, SOM, and ExP) were skewed. Therefore, Spearman ranked correlation analysis was used in examining the relationship between the rapid assessment scores and soil and water quality parameters. More specifically, regression analysis (Minitab, Version 15, State College, PA) was used in determining the strength of the relationships among RAM scores and:

- mean soil BD, TC, TN, soil organic matter SOM, ExP (Bantilan-Smith et al. 2009), surface water total dissolved nitrogen (TDN) and total phosphorus (TP) (Bruland and MacKenzie 2010), and $\delta^{15}N$ concentrations in wetland plant tissues (Bruland and MacKenzie 2010)
- distance to nearest road and wetland area

RESULTS

Sixteen of the 27 wetlands were classified as depressional wetlands according to the HGM and CRAM typologies. While three of the six tidal wetlands receive a significant amount of fresh surface or groundwater, the others receive lower proportions of freshwater inputs and were classified as euhaline or hyperhaline by Bantilan-Smith et al. (2009). The wetland areas classified as depressional by the HGM typology tended to be classified as palustrine or

lacustrine by NWI (USFWS 2009), using the Cowardin et al. (1976) classification system; the HGM riverine areas were predominantly palustrine, and tidal fringe areas were classified as estuarine (USFWS 2009). Furthermore, according to the National Wetland Inventory (2009), the areas classified as depressional were predominantly unconsolidated bottom or emergent vegetation, while Riverine wetlands included primarily emergent vegetation. A fairly high proportion of the Tidal Fringe wetlands were classified as forested; this class also included large areas classified as emergent vegetation and unconsolidated bottom and shore (Cowardin et al. 1976).

Qualitative assessments of soils indicated that wetlands closer to the shoreline tended to have higher sand content, while those further inland had higher clay and organic matter content (Figure 2.2). Wetlands invaded by *Rhizophora mangle* (red mangrove) had deep litter layers overlaying organic mucky soils (Figure 2.3) whereas, wetlands invaded by *Battis maritima* (pickleweed) tended to have a thinner sparser litter layer along with a thin organic layer overlaying well-drained substrates. Furthermore, wetlands located on younger volcanic soils had shallower O and A horizons than older wetlands, located on the windward, wetter coastlines.

Riverine wetlands tended to have more trees and plants that are more characteristic of terrestrial ecosystems, but fewer facultative and obligate wetland species (Figure 2.4). In contrast, depressional wetlands tended to have shorter plant layers, but more wetland-dependent species. While short wetland plants such as *B. maritima* dominated some tidal fringe wetlands, such as Kaloko fishpond, *R. Mangle* trees dominated others, such as the edge of Pouhala Marsh for example.

Figure 2.2. Photos of wetland soil depth and texture samples. Top left: thin well-drained soil sample from wetland invaded by *B. maritime*. Top right: sample showing dark clay soils overlain by lighter red clay layer. Bottom left: light-colored compacted clay soil sample. Bottom right: deep organic soil pit.

Figure 2.3. Photos showing examples of litter cover and depth within a Mangrove forest.

Furthermore, the wetlands classified as tidal fringe lacked channels and pools that typify coastal estuaries along the southeast coast of the U.S. However, many of the wetlands in this study did include microtopographic features such as deep soil cracks, rocky terrain with vegetated clumps, pools, and occasional small channels, which increased the CRAM scores for microtopographic complexity (Figure 2.5). While the HHGM included metrics for rating the microtopographic complexity of tidal fringe and riverine wetlands, these metrics were not included for depressional wetlands.

Figure 2.4. Riverine wetlands, such Mahohuli fishpond pictured here (left), tended to have more trees, but fewer facultative and obligate wetland species. Depressional and tidal fringe wetlands (right) tended to have shorter plant layers, but more wetland dependent species. The photo on the right is of `Aimakapa, which is dominated by *P. vaginatum* and *B. maritima,* with patches of *B. maritima* along the water edge.

Figure 2.5. Photos of microtopographic features found in Hawaiian depressional wetlands, which included deep soil cracks, rocky terrain with vegetated clumps, pools, and occasional small channels. Microtopography was used by CRAM to rate the quality of wetland habitats.

Figure 2.6. Photographs of features such as snags, boulders, submerged vegetation, crenulated foreshore, and unvegetated mudflats, which contributed to CRAM structural patch richness scores.

Wetland Condition Assessment Scores

The wetlands in this study received the highest overall mean condition assessment scores from the HHGM, compared with CRAM, and WRAP. Furthermore, the HHGM scores were narrowly distributed whereas WRAP, and especially CRAM detected a wider gradient in wetland condition (Table 2.2; Figure 2.7). Wetlands were scored the lowest overall condition when assessed using WRAP. Some wetlands received scores as low as 0.25; only six wetlands rated higher than 0.50. Despite the difference in the range of rapid assessment scores calculated with the three methods (HGM, CRAM, and WRAP) they were all significantly correlated with each other (Table 2.3).

Table 2.2. Statistics of rapid assessment scores across the three rapid assessment methods.

Variable	Mean	Minimum	Q1	Median	Q3	Maximum
HGM Score	0.62	0.46	0.55	0.62	0.71	0.76
CRAM Score	0.55	0.32	0.44	0.56	0.66	0.79
WRAP Score	0.42	0.25	0.33	0.42	0.50	0.67

Table 2.3. Pearson correlation coefficients among rapid assessment methods (CRAM = California Rapid Assessment Method, HHGM = Hawai`i Hydrogeomorphic Method, WRAP = Florida Wetland Rapid Assessment Procedure, and CRAM = California Rapid Assessment Method).

Variable	CRAM Score	WRAP Score
HHGM Score	0.614	0.761
CRAM Score	*	0.813

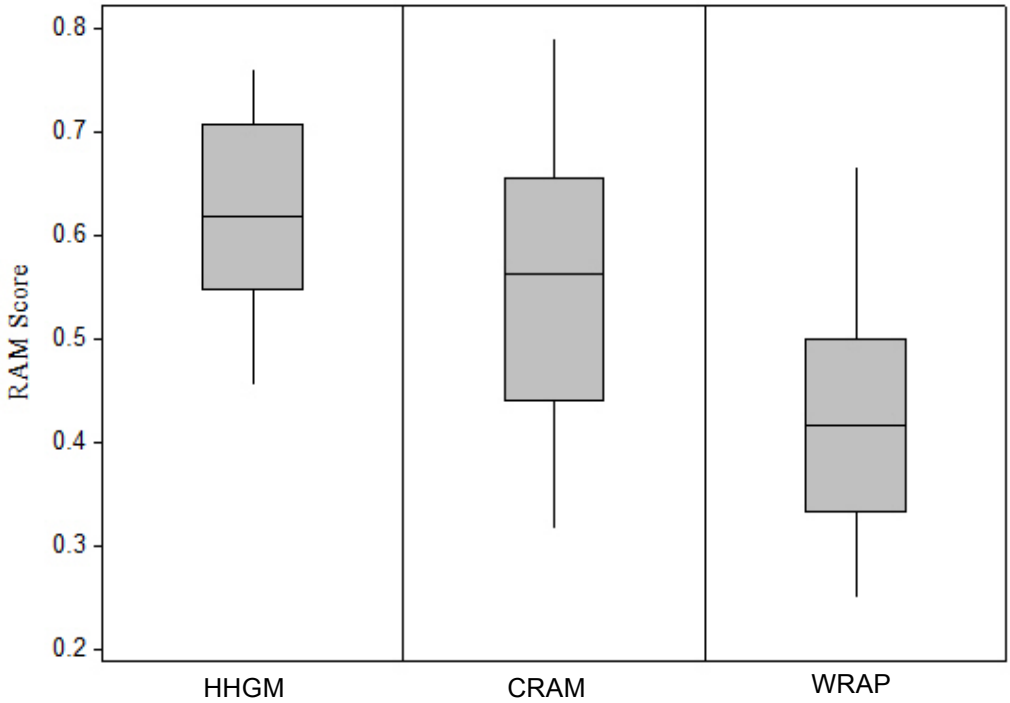

Figure 2.7. Box-and-whisker plots comparing rapid assessment method (RAM) scores for 27 coastal wetlands, rated using guidelines from Hawai`i HGM, CRAM, and Florida WRAP.

Plant Communities

The HHGM appears better at detecting a gradient in the condition of plant communities than CRAM and WRAP (Figure 2.8). In addition to native species cover, the CRAM used metrics for number of plant layers and number of co-dominant species (including non-native

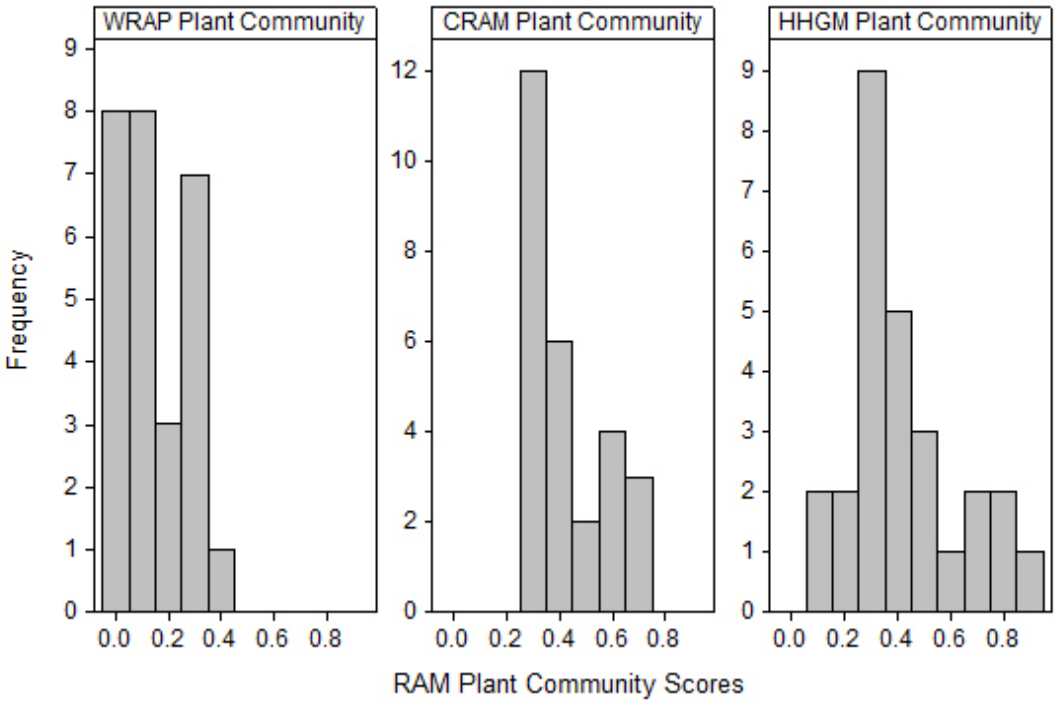

Figure 2.8. Histograms comparing the RAM scores for plant community variables.

species). The HHGM included metrics for total plant cover in addition to cover and ratio of native wetland plants, whereas the Florida WRAP depended solely on the cover of native and invasive plants.

Low WRAP scores were primarily due to the prevalence of non-native or "undesirable" species in the lowland coastal areas, which lowered scores in three of the five WRAP categories: canopy cover, ground cover; and habitat buffer. The dominance of non-native species (cover and abundance) also lowered the CRAM scores for percent invasion, which impacted the scores for biotic structure as well as the overall wetland condition scores. In addition, invasive plant species lowered the HHGM scores in the functional category for characteristic plant communities and habitat support. The tree canopy along the perimeter of most wetlands had greater than 75% non-native species cover and there were no native wetland

tree species (facultative or obligate) recorded in any of the wetlands. In addition, the coastal wetlands generally had more than 50% non-native ground cover.

Some wetlands, such as Klipper Pond and Punamano (on O`ahu) received fairly high CRAM plant community scores, primarily due to high scores in the categories of number of plant layers or number of co-dominant species, which included native and non-native species. With >50% cover of *Batis maritima*, Nu`u (Maui) received the highest HHGM and CRAM scores for native species coverage, but with only three dominant species (*Bolboschoenus maritimus*, *Bacopa monnieri*, and *Sesuvium portulacastrum*), it received a low CRAM score in the category for co-dominant species. Both Klipper Pond (O`ahu) and Kamilo (Hawai`i) received high RAM scores for native plant cover due to > 50% cover of *B. monnieri*.

The most common native ground cover species were *Bolboschoenus maritimus* (kaluha), *Bacopa monnieri* (`ae`ae or water hyssop), *Schoenoplectus tabernaemontani* (`aka`akai or giant bulrush), *Sesuvium portulacastrum* (`akulikuli or sea purslane) and *Cyperus laevigatus* (makaloa or smooth flatsedge), with a few wetlands also providing at least limited habitat for *Cyperus javinicus* (`ahu`awa or java sedge). Non-native *B. maritima* was recorded at all tidal fringe wetlands except Honu`apo, a fishpond located along the dry Ka`u coast of Hawai`i Island. The non-native grasses *Paspalum vaginatum* (seashore paspalum) and *Distichlis spicata* (salt grass) recorded at Honuapo resulted in lower HHGM scores in various functional categories such as energy dissipation, carbon export, and retention of elements and particulates, relative to wetlands with more woody species such as *R. mangle* and *B. maritima*. All of the Riverine wetlands in this study had been substantially channelized and therefore tended to lack facultative and obligate wetland plant species, and were instead dominated by plants more

typical of upland areas, such as *Urochloa mutica* (California grass) and *Pluchea* spp (marsh fleabane).

The cover and abundance of Polynesian trees including *Thespesia populnea* (milo), *Cocos nucifera* (coconut), and *Pandanus tectorius* (lauhala) was also somewhat limited. Thus even when a liberal interpretation of desirable species was used in scoring wetlands with Polynesian trees, the WRAP scores for the canopy cover variable never exceeded 0.17. In some cases managed wetlands received slightly higher WRAP scores if the non-native vegetation provided bird nesting and/or foraging habitat and was considered "desirable" by wetland managers.

Habitat Quality

Wetlands with vegetated buffers, natural hydrology, and native plant cover such as Nu`u, received the highest CRAM and HHGM scores in variables such as connectivity, hydrology, and structural patch richness. The CRAM "structural patch richness," scores were based on the number of patch types such as boulders, algal mats, mud flats, crenulated foreshore, snags, course woody debris, pools, and submerged vegetation (Figure 2.6). A comparable HHGM variable called "number of aquatic and terrestrial habitats," assigned scores based on the number of terrestrial and aquatic habitats; the highest scores were assigned to wetlands with 2 aquatic habitats (i.e. open water, channels, and pools) and > 1 terrestrial habitat (i.e. herbaceous cover, shrubs, mud flats, snags, etc.) and wetlands with no aquatic habitat and only one terrestrial habitat received the lowest scores. Although the HHGM had separate metrics for rating wetland edge complexity the metrics were simplistic binary metrics in which all wetlands were assigned either a score of 1 (regular edge) or 10 (irregular or complex).

Both CRAM and HHGM included a variable for estimating the degree of connectivity to other wetlands in the vicinity. The CRAM involved drawing four 500m lines, from the wetland edge in each cardinal direction, and estimating the percentage of the line segments that intersected with wetland or aquatic habitat of any kind. Rating a wetland's capacity to "maintain habitat interspersion and connectivity," using HHGM involved: 1) estimating the density of all freshwater wetlands within 800m of the wetland edge, and 2) measuring the distance to the nearest wetland of the same HGM class. However, the HHGM did not consider this a function of tidal fringe wetlands.

Interestingly, the WRAP and CRAM did not include variables for rating impacts from non-native animals. Furthermore, the HHGM metrics for this variable, which was based on sighting six or more non-native individuals during an assessment period, resulted in all wetlands receiving the same HHGM score of 0. Given that species such *Acridotheres tristis* (mynah bird), *Paroaria coronata* (brazilian cardinal), *Streptopelia chinensis* (ring neck dove), *Passer domesticus* (house sparrow), *Zosterops japonicas* (white eyes), and *Bubulcus ibis* (cattle egret) are so common in coastal lowlands, it was inevitable that more than six non-native birds would be sighted during an assessment period.

Wetland Buffers

The three RAMs differed in their approach to assessing the quality of a wetland buffer. The WRAP assessed the condition of 100m buffers, whereas CRAM assessed 250m buffers. While all of the RAMs assessed the quality of riparian buffers, the HHGM was the least consistent in how those metrics were applied. For example, land use was used as an indicator in evaluating almost all functions for depressional wetlands, but was rarely used in rating riverine wetlands, and was not used at all in rating tidal fringe wetlands. The HHGM nonetheless was

spatially more refined and characterized the land-use within two different buffer widths: 1) within 25m of the wetlands and 2) between 25-100m of the wetland edge; furthermore, the land-use within the wetland was characterized as well. However, the entire buffer or wetland had to be characterized as only one broad land-use/land-cover type such as urban, residential, pasture, generalized soil disturbance, or undisturbed. The HHGM did, however, include a land-use category for light grazing, which lowered the buffer score for the depressional wetland at Waipio, a wetland surrounded primarily by non-native vegetation that is "lightly grazed" by horses.

Wetlands surrounded by residential, urban, military, and golf courses received the lowest buffer scores in all three RAMs, with Ka`elepulu (O`ahu), Klipper Pond (O`ahu), and Kanaha (Maui) receiving some of the lowest scores for buffer condition. Once again the CRAM and HHGM buffer scores tended to be higher than the WRAP scores (Figure 2.9). Only the condition of terrestrial landscapes was considered when assessing the condition of buffers (i.e. open water was not included in the analysis of wetland buffers).

The WRAP characterized the quality of buffers for all wetlands based on width, native vegetative cover, and land-use intensity (in the water source variable). The CRAM buffer scores also included metrics for width and condition, as well as percent of wetland edge with a buffer. Therefore, the prevalence of non-native vegetation within the wetland buffers lowered both the CRAM and WRAP buffer condition scores. The HHGM buffer variable, however, did not include vegetation community metrics. In addition, the HHGM did include a variable for evaluating the buffers of tidal fringe wetlands.

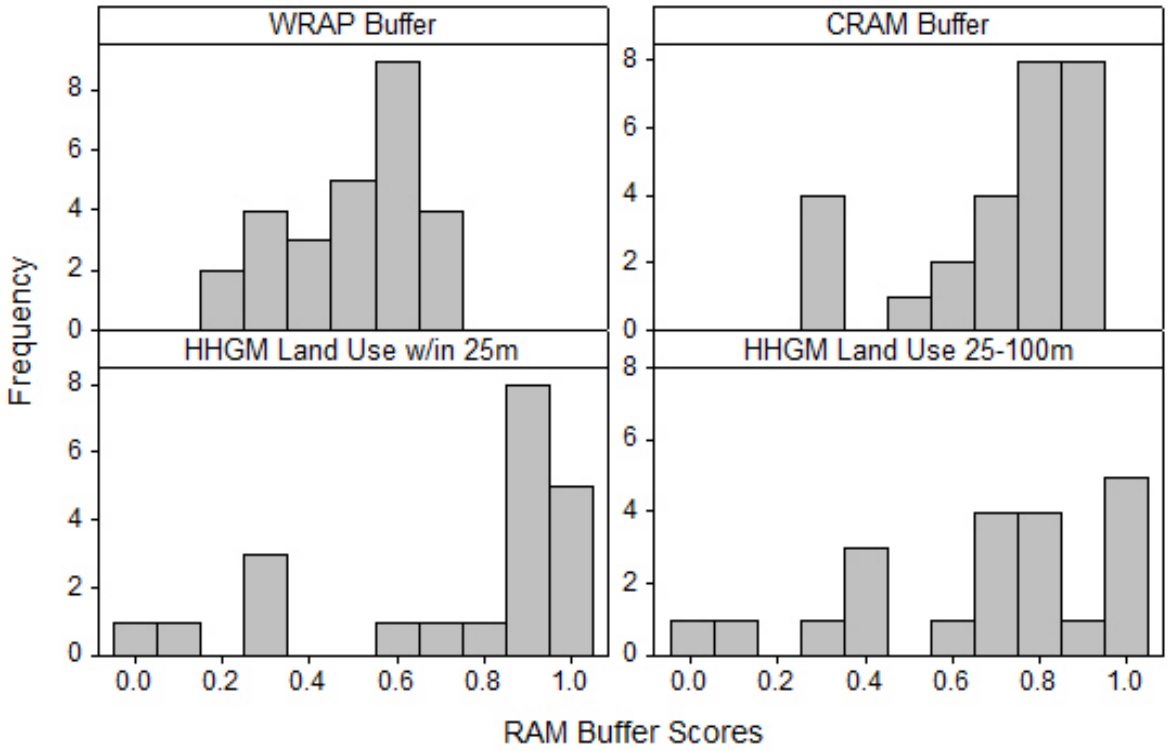

Figure 2.9. Histograms comparing RAM scores for wetland buffer condition; from 0 (worst) to 1 (best).

Water Quality

In addition to evaluating the quality of the habitat surrounding a wetland, the WRAP and CRAM included separate metrics specific to evaluating water quality. The WRAP assessed the extent and intensity of land-use occurring within 100m of the wetland edge while the CRAM took a more landscape-scale approach, which included determining if developed or irrigated land comprised more than 20% of the drainage basin within 2km of the wetland. In addition, CRAM used vegetation (i.e. presence of salt tolerant vegetation) as an indicator of altered hydrology and changes to the quantity of freshwater. This resulted in fairly large discrepancies between the different RAM water quality scores for wetlands such as Waihe`e (Maui), which has at least a 100m vegetated buffer, but also has reduced water flow due to agricultural irrigation that diverts water into an adjacent watershed.

Correlation between RAM Scores and Field Surveys

The only significant correlations among the water quality parameters (NO_3+NO_2, NH_4, TDN, PO_4, TP) were between the final HHGM (r = 0.53, p < 0.01) and WRAP (r = 0.39, p < 0.05) scores and PO_4. Unexpectedly, the PO_4 levels were higher in wetlands with higher condition assessment scores. There were, however, a number of significant correlations between the final RAM scores and soil quality parameters (Table 2.4). The CRAM scores had the best overall fit with soil parameters, which were significantly and positively correlated with soil TN, TC, and negatively correlated with BD. The WRAP scores were only correlated with TN. In addition to BD and TN, the HHGM scores were also positively correlated with ExP.

Table 2.4. Spearman ranked correlation coefficients for rapid assessment scores and soil parameters (excluding measurements from dry soils). No asterisk = p < 0.005, * = p ≤ .01, ** = p ≤ .025, *** = p ≤ 0.05, ns = not significant.

Wet/Intermediate Soil Parameters	**HHGM Score**	**CRAM Score**	**WRAP Score**
BD (g cm^{-3})	ns	-0.379***	ns
TN%	0.360***	0.501*	0.401**
TC%	ns	0.406**	ns
SOM%	ns	0.348***	ns
ExP (μg·g−1)	0.451**	ns	ns

All of the RAM scores were significantly correlated with $\delta^{15}N$ levels measured in wetland plants (Table 2.5), with the CRAM scores providing the best fit (Figure 2.10). Once again, when the CRAM and WRAP water quality scores were independently analyzed, these scores were even more strongly correlated with $\delta^{15}N$ concentrations (r = -0.77 and r = -0.52 respectively, p < 0.01).

Table 2.5. Pearson correlation coefficients between rapid assessment scores and $\delta^{15}N$ levels recorded in vegetation.

	HGM Score	CRAM Score	WRAP Score
$\delta^{15}N$	*-0.386*	*-0.613*	-0.399
p value	*0.047*	*0.001*	0.039

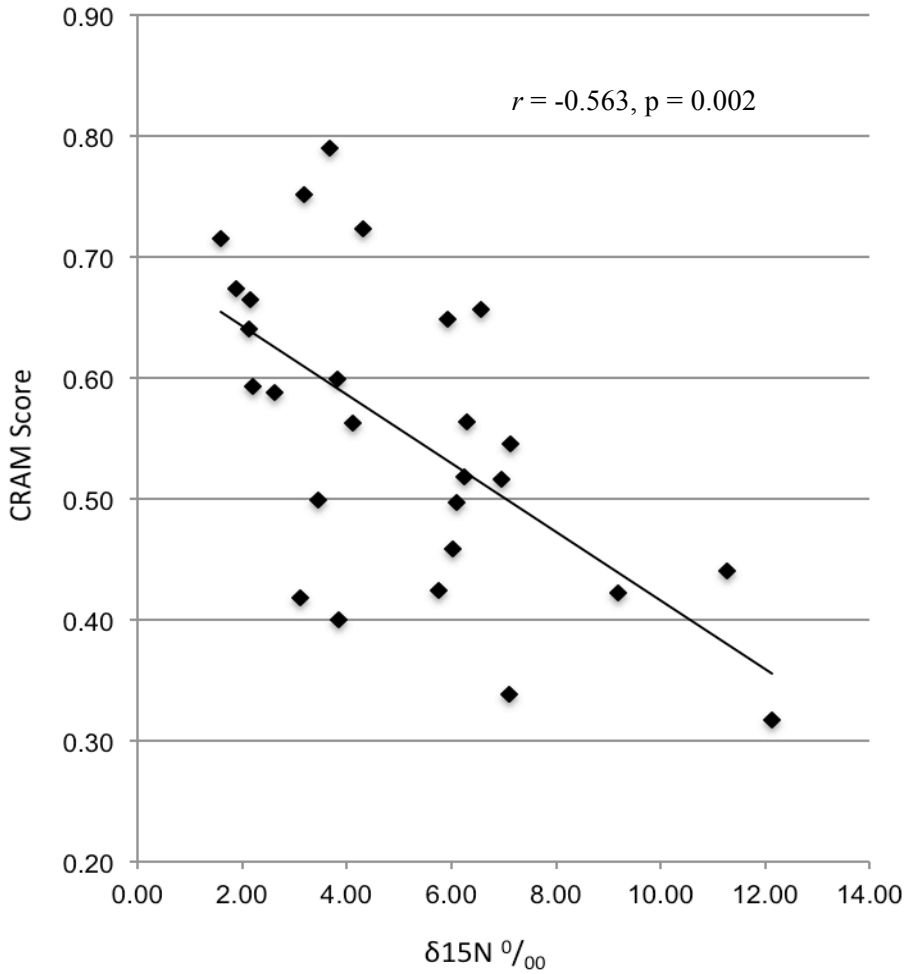

Figure 2.10. Scatterplot between CRAM scores and $\delta^{15}N$ levels measured in wetland plant tissue.

Relationship between Assessment Scores and Individual Metrics

The average size of the wetlands in this study was 16.9 ha; close to 80% of the wetlands were < 14 ha and only five wetlands were > 20 ha. The larger wetlands, such as Kawainui and Kealia, located on the islands of O`ahu and Maui, respectively, skewed the distribution.

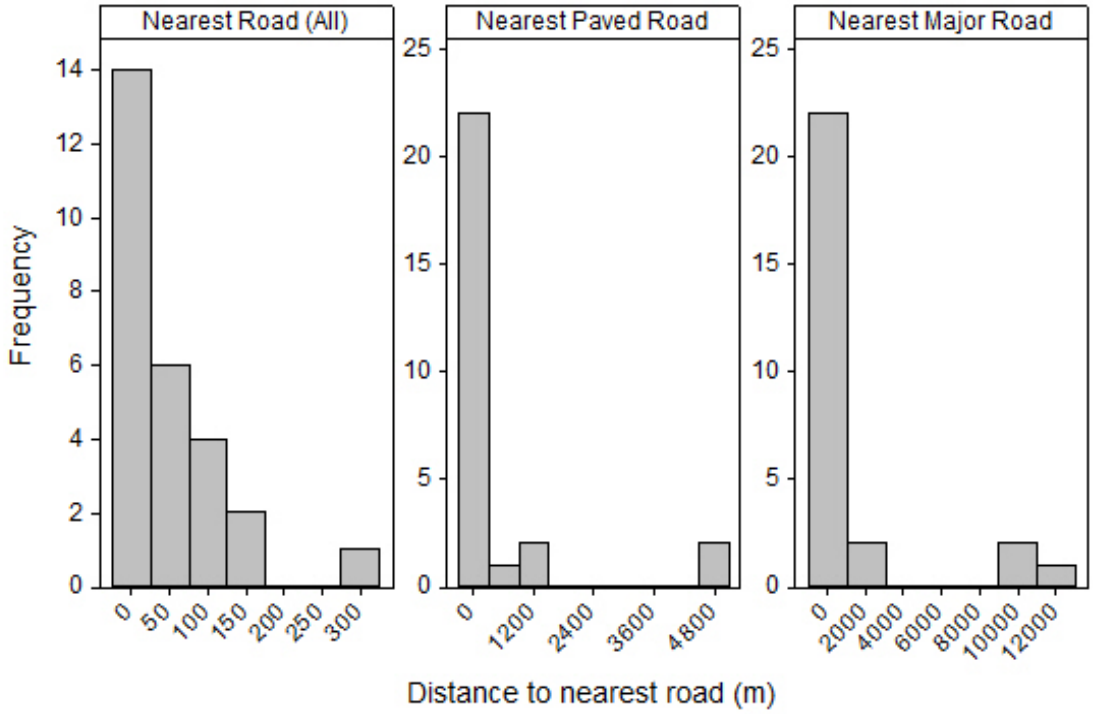

Figure 2.10. Histograms of distances between the edge of 27 wetlands and the nearest road, nearest paved road, and nearest major road (i.e. highway).

Notably, smaller wetlands, such as Nu`u and the ponds at Kamilo, were often located in more remote, natural landscapes. More than 80% of the wetlands were within 110 m of a road, 300 m of a paved road, and 1,000 m of a major highway; Only a few isolated wetlands were located far from paved or major roads, which once again resulted in a skewed distribution. Spearman correlation coefficients indicated no significant relationship between RAM scores and distance to any road, including major roads and 4WD dirt roads. However, the coefficients did indicate a significant relationship between RAM scores and distance to major roads. Nonetheless, these statistical results should be viewed with caution since they were heavily influenced by extreme outliers, which had significant leverage on the outcome. In fact, after removing the three largest outliers the RAM scores were no longer significantly correlated with distance to major roads.

DISCUSSION

This project highlighted differences between three RAMs and analyzed the relationship between assessment scores and the quality of wetland surface water and soils. In general the rapid assessment methods (RAMs) were somewhat weakly correlated with soil parameters (i.e. BD, TC, SOM, TN, and ExP) and strongly correlated with the $\delta^{15}N$ levels in wetland plant tissue, an indicator of human and animal waste. Of the rapid assessment methods, CRAM showed the strongest correlations with Level I and Level III data, which suggests that although CRAM was developed for condition assessments along the Pacific West coastline (from Mexico to Oregon), with modifications to account for local conditions, the method could be applied to assessing the condition of Hawaiian coastal wetlands. For example, the metrics for variables such as number of plant layers and structural patch richness would need to be calibrated with indicators of the biotic integrity of Hawaiian coastal wetlands (e.g. abundance and diversity of native waterbirds). In addition, CRAM was much simpler and faster to perform and therefore falls within the true definition of a rapid assessment method (Fennessy et al. 2004).

Although WRAP was extremely rapid to perform, the scores were only significantly correlated with one soil parameter, TN. On the other end of the spectrum, the HHGM was more quantitative and time consuming, and given that it can easily require > eight hours to evaluate a single wetland, is not necessarily a rapid assessment method. In addition, the final HHGM scores were only correlated with two soil parameters TN and ExP.

Water Quality

Phosphate was the only water quality parameter that was significantly correlated with all three RAM scores: HHGM ($r = 0.529$, $p < 0.01$), CRAM ($r=0.40$, $p<0.05$), and WRAP ($r = 0.386$, $p < 0.05$). However, the relationship was unexpectedly positive; wetlands that received

higher condition assessment scores had higher PO_4 levels. Thus the highest PO_4 levels were measured in Nu`u and Waihe`e (Maui), which also received some of the highest condition assessment scores, while wetlands such as Mahohuli (Hawai`i) and Percolation Ditch (Oahu) had relatively low levels of PO_4 and comparatively low RAM scores. There are many factors such as hydrology, salinity, and wetland status (i.e. natural vs. created) that explain differences in the adsorption of phosphorus in soils and the subsequent release of PO_4 into the water column. In some cases higher PO_4 levels in the water column of wetlands may be due to a combination of nutrient-rich clay soils and increased water retention times. For example, flooded soils that are high in iron tend to release P back into the soil solution, which can lead to higher P in the water column of some wetlands (Bruland and DeMent 2009). Furthermore, natural wetlands tend to have a higher capacity to bind (adsorb) phosphate (PO_4) ions (Bruland and DeMent 2009). When soils become wet, the phosphate becomes solvent and is released into the water column. In addition due to seasonal rainfall and local tidal exchange the quality of water in coastal wetlands is highly variable and can be challenging to characterize.

Indicators such $\delta^{15}N$ concentrations can provide more stable measurements of water quality and the sources of N uptake by wetland plants over time. As predicted, all of the final RAM scores had significant negative correlations with $\delta^{15}N$ levels measured in wetland plant tissue. The strongest correlation was with CRAM scores, which may have at least been partially due to the landscape-scale approach (2 km) used by CRAM in evaluating water sources. Thus, when the CRAM water quality scores were independently analyzed with $\delta^{15}N$, the relationship between the two variables was even stronger ($r = -0.77$, $p < 0.01$). Similarly, research by Bruland and MacKenzie (2010), found that $\delta^{15}N$ levels were positively correlated with urban

development and human population densities within the 1,000m radii of the wetland and within the watersheds of Hawaiian coastal wetlands.

Soil Quality

The least disturbed wetlands, with relatively high condition assessment scores, had the highest levels of soil TC and SOM, as well as TN and ExP. In other words, wetlands with the highest soil TN and ExP values also received the highest rapid condition assessment scores; inversely the wetlands with the lowest assessment scores had the lowest soil nutrient and carbon levels. In general, the wetlands located on the islands with higher levels of human development (i.e. O`ahu and Maui), tended to have the lowest levels of SOM. Some of the highest TC and TN levels were measured in the more remote wetlands such as Waipio, `Aimakapa, Kaloko, Kamilo Point, and Nu`u, which generally had higher overall condition assessment scores. In the opposite extreme the wetland at Klipper Pond, had relatively low soil TN levels (Bantilan-Smith et al. 2009), and because it was a highly-modified wetland located in a golf course, also received low RAM scores especially in the categories for hydrology, buffer condition, and landscape connectivity. Surprisingly, the only scores correlated with soil BD were from CRAM, and even this relationship was somewhat weak. This may have been partially due to the high sand content of some wetland sites, which increased BD, regardless of wetland condition or impacts from humans.

Several factors may explain the positive relationship between RAM scores and soil nutrient content. For instance, created and restored wetlands tend to have less SOM than naturally occurring, unaltered wetlands (Shaffer and Ernst 1999, Bruland and Richardson 2006, Bantilan-Smith et al. 2009). With only one exception (Salvage Yard Pond, O`ahu), the wetlands with the highest SOM and correspondingly high soil nutrient levels tended to be “natural”

wetlands or fishponds located on the least developed Island of Hawai`i. Inversely, the wetlands with the least SOM and lowest soil nutrient levels included primarily created or restored wetlands located on the islands of Maui and O`ahu (Bantilan-Smith et al. 2009). Since created and restored wetlands tend to be located in more developed areas, it is difficult to determine whether or not soil parameters were more directly related to wetland status (created vs. natural) or past and present disturbance levels.

In contrast, N levels accumulate from season to season in undisturbed natural ecosystems, particularly for ecosystems dominated by N-fixing plants, which can easily add more N than is naturally leached from soils (Vitousek et al. 2002). Many of the wetlands in this study were surrounded by N-fixing tress such as *Prosopis pallida* (Kiawe) and *Leucaena leucocephala* (Koa Hoale). The degree to which these plants are contributing to soil and water N levels in Hawaiian coastal wetlands is currently unknown, but is potentially quite high given the abundance of N-fixing tress in and around coastal wetlands.

It is interesting that the HHGM scores were not more strongly correlated with SOM, given that "depth of the O and A soil horizons" was used in scoring the capacity of a wetland to perform nutrient cycling. Results from this project showed however, that HHGM was not correlated with SOM. These results were, nonetheless, not entirely surprising since the "depth of the O and A soil horizons" was one of over 20 variables used in rating the total functional capacity of a wetland (Appendix A).

Results showing that the least disturbed sites had the highest soil TC and SOM contrasted sharply with the EMAP survey in 2002, in which Nelson et al. (2007) found that Hawaiian estuaries in urban areas had higher TC. This may be due to the difference between wetlands that receive surface water and have free exchange with the open ocean (i.e. estuaries)

and depressional wetlands, which lack either an inlet or outlet and are generally dominated by subsurface flow.

Landscape Indicators

While total area is considered an important factor in estimating an ecosystem's potential for wildlife support (MacArthur and Wilson 1967, Turner et al. 2001) and is explicitly defined in HHGM as a metric for assessing the functional capacity of tidal fringe wetlands to support characteristic vertebrate habitats (SAIC 2004), none of the final rapid assessment scores were significantly correlated with wetland area. As described in the methods, many of the variables in the HHGM and CRAM assessments were area weighted, which eliminated the influence that wetland size had on the final assessment score. It is, nonetheless, worth reflecting on the question of quality versus quantity and whether larger, yet degraded, wetlands provide more ecosystem services than smaller intact well functioning wetlands. In other words, it is important that the assessment methods do not inadvertently downgrade wetlands just because they are small; this is particularly true for small wetlands that provide critical habitat for endangered species as does many of the Hawaiian coastal wetlands.

Although roads fragment landscapes, increase surface runoff of toxic chemicals, and provide vectors for invasive species, in this study the distance to the nearest road was not necessarily a predictor of wetland condition. These results are somewhat consistent with findings by Miller and Wardrop (2006), which showed that while the quality of wetland habitats corresponded with road density, it was not correlated with distance to nearest road. Weller et al. (2007) also found that road variables (density and nearest) were weakly related to wetland condition. The lack of correlation between road variables and wetland condition could be due to

a possible temporal lag between road construction and easily identifiable impacts on wetland ecological integrity.

In some cases restoration activities increased RAM scores if they increased topographic complexity, number of plant layers, number of co-dominant species, native species abundance and cover, and habitat interspersion. For example, Klipper Pond received higher plant community scores, due to the prevalence of *B. monnieri* and the presence of *B, maritimus* and Salvage Yard Pond received higher topographic complexity scores as a result of its Mud Ops invasive species control program (http://www.youtube.com/watch?v=e5Q6QUjXnJc). Waihe`e also received higher plant community scores due to extensive out planting of native wetland species. In response to an adaptive wetland management program at the Ki`i and Punamano wetlands in James Campbell National Wildlife Refuge (http://www.fws.gov/jamescampbell/), there is a diverse plant community that is utilized by an abundance of native and migratory waterbirds. Furthermore, it cannot be overlooked that urban wetlands, particularly those undergoing restoration, provide specific functions, such as dissipation of energy, nutrient cycling, and even habitat support, in spite of degraded water and soil quality. Additionally, these urban wetlands are often valued for their educational opportunities such as those provided at Pouhala Marsh and at Honouliuli Marsh (not included as a study site) shown in Figure 2.11.

Figure 2.11. Photos of restoration and educational events at Pouhala Marsh (left) and Honouliuli Marsh (right).

RAM Vegetation and Wildlife Habitat Metrics

While Hawaiian coastal wetlands historically included shrubs and trees such as *Pritchardia* spp., *Columbrina* spp., and *Kanaloa kahoolawensis* (Athens 1997, Athens et al. 2002, Erickson and Puttock 2006), these species were later replaced by Hawaiian introductions such as *Hibiscus tiliaceus* (hau), *Thespesia populnea* (milo), *Cocos nucifera* (coconut), and *Ludwigia octivalvis* (primrose willow) (Athens 1997). These coastal wetlands have since been invaded by more recent introductions of trees and shrubs including *R. Mangle* and *Pluchea* spp. Although these species aggressively compete with native plants, their presence increased HHGM and CRAM scores for variables related to biotic structure. For example, when using the HHGM to assess the condition of tidal and depressional wetlands, the methods assume that variables such as percent plant cover and presence of shrubs and trees increase the capacity of wetlands to perform functions such as nutrient cycling, carbon export, and dissipation of energy. Additionally, invasive plant species also increased CRAM scores in "number of plant layers" and "number of co-dominant species" variables. Nonetheless, managers have observed that native and migratory bird species in Hawai`i often utilize non-native plant species for forage and nesting material (Michael Silbernagle, pers. comm. 2011), the degree to which needs further research in Hawai`i.

The HHGM variables for 1) percent cover of native facultative and obligate plant species, 2) ratio of native to non-native wetland plant species and 3) degree of native species regeneration resulted in a wider range of scores for "characteristic plant community" (Figure 2.10), which seemed to provide the most useful metrics for detecting a gradient in the condition of plant communities in coastal Hawaiian wetlands. Nonetheless, the HHGM metrics did assign higher scores to wetlands with the greatest percent plant cover, which may not be appropriate

for rating the quality of wildlife habitat in Hawai`i, given that many water-birds such as the endangered *H. mexicanu knudsenies* (Hawaiian stilt), utilize unvegetated mud flats for nesting and foraging (USFWS 2005).

Some of the RAM metrics could be improved in order to better detect a gradient in the capacity of coastal wetlands to support wildlife habitat. For example, the WRAP wildlife utilization scores were based on whether or not there was visible evidence of existing wildlife utilization in all taxonomic levels from macroinvertebrates to mammals. Given that there are only two native mammals in Hawaii, it is not surprising that Hawaiian wetlands received relatively low wildlife utilization scores compared with the more taxonomically rich wetland ecosystems in Florida. Furthermore, while the presence of wildlife can be a good indicator of habitat support, some wetlands may be wildlife sinks. For instance, while assessing the condition of some wetlands, dead waterbirds and bird skeletons, as well as dead floating fish were found within the vicinity of some urban ponds. Furthermore, some wildlife species are cryptic (e.g. *Gallinula chloropus sandvicensis*) and may be difficult to detect during a rapid assessment.

The CRAM and HHGM both included variables that more indirectly measured habitat quality such as wetland edge complexity, structural patch richness, and connectivity with other wetland habitats. However, the HHGM metrics for edge complexity, non-native animal use, and number of aquatic and terrestrial habitats were too coarse to discern differences among wetlands. In contrast, the CRAM provided a number of variables for detecting a gradient in habitat complexity, including “structural patch richness, “topographic complexity”, and “horizontal interspersion and connectivity”. Furthermore, CRAM provided a detailed list of

features that should be considered when rating "structural patch richness," whereas HHGM metrics were vague in their description for what constitutes an aquatic or terrestrial habitat.

Nonetheless, there does not appear to be a direct correlation between the condition of the riparian buffer and wildlife utilization by waterbirds. Some of the wetlands most heavily utilized by native waterbirds, such as Kanaha Pond (Maui), Hamakua Marsh (O`ahu), and Pouhala (O`ahu) were within 100m of urban development, while wetlands located in more natural landscapes, but geologically younger island of Hawai`i, are not considered core native waterbird habitat. Furthermore, created wetlands received relatively low CRAM scores due to their altered hydrology, altered hydroperiods and reduced hydrologic connectivity, regardless of whether or not these modifications provided additional habitat for endangered water-birds.

Calibration and Validation of Rapid Assessment Methods

In developing a Hawai`i RAM, it is important to calibrate indicators of ecological integrity with local variables. In particular, the development of accurate HHGM metrics will require establishing reference standards for coastal wetlands that account for natural variation within each wetland class. Although, these metrics are usually based on a ratio of measured values relative to reference conditions, many of the HHGM metrics used in this study were based on the original metrics developed for wetlands in the continental U.S. (Smith et al. 1995, Shafer and Yozzo 1998, Hauer et al. 2002). For example in rating the function for nutrient cycling, tidal fringe and depressional wetlands were assigned higher HHGM scores as the depth of the O and A horizon increased from 0 to 40 cm and lower scores as depths increased from 40 to 70 cm. However, those metrics were based on reference conditions developed for Prairie Potholes, despite the fact that the geomorphology and hydrology of Prairie Potholes is quite different from that of coastal depressions in Hawai`i. While prairie potholes occur in glacial

substrates and are generally closed systems that concentrate nutrients from surface water flowing from adjacent upland areas (Hauer et al. 2002), anchialine pond depressions are located on relatively young, porous substrates that intercept the groundwater table and are open to subsurface tidal exchange with marine waters. Furthermore, these saline or brackish water pools are unique to the Hawaiian Islands and are functionally different from prairie potholes and do not necessarily meet the standard definition of a depressional wetland.

In addition, the HGM classification system may not be entirely appropriate for categorizing coastal wetlands in Hawai‘i. The classification system, presented by Erickson and Puttock (2006) as the “Wetland Analysis Protocol,” was developed specifically for the Hawaiian Islands, which categorizes Hawaiian wetlands by hydrologic source, salinity, geomorphology, and substrate, to yield the following five main wetland types: anchialine pools, coastal flat, bog (groundwater and precipitation driven), estuarine and palustrine (groundwater, precipitation, and surface driven) (Erikson and Puttock 2006). Note that anchialine ponds/pools are recognized as a distinct wetland type. Furthermore, unlike most classification systems, the Hawaiian typology recognizes that the source of water for some tidal wetlands, particularly those along leeward coastlines, is primarily from groundwater flow. Thus, these wetlands technically span two HGM classes: depressional and tidal fringe. Therefore, future research in the development of rapid assessment methods for Hawai`i should consider using the Hawaiian classification system instead of the HGM.

In addition, many of the wetlands in this study received low scores for litter depth based on the Hawai`i HGM criteria. Given the higher decomposition rates in the warm tropics this metric should be rescaled in order to accurately evaluate nutrient cycling in Hawaiian coastal wetlands. Additionally, HHGM included canopy gaps as an indicator for spatial structure and

biodiversity. Since Hawaiian wetlands rarely include native tree canopy, and those with *R. Mangle* rarely have canopy gaps, this metric does not appear to be a valuable indicator for Hawaiian coastal wetlands.

Research on complex relationships between soil depth and nutrient cycling, deposition, and accumulation is sorely needed for Hawaiian coastal wetlands, and could take years to develop before it can be used as an indicator of wetland condition. In addition, establishing reference standards for Hawaiian coastal wetlands may be especially challenging given that historic and current land use has significantly impacted these ecosystems. Nu`u (Maui) and Kamilo Point (Hawai`i Island) may provide the best examples of reference conditions, especially with regard to native species cover (Bantilan-Smith et al. 2009) and habitat quality. It may, however, be more difficult to find unaltered tidal fringe and riverine wetlands along coastlines in Hawai`i. It is likely that the baseline for coastal wetlands in Hawai`i has permanently shifted and that reference wetlands will be more representative of "best attainable" conditions rather than natural, unaltered wetlands.

The underlying assumption of most RAMs, including those used in this study, is that there are direct and indirect links between human land use and the ecological integrity of wetlands. This assumption is difficult to test, however, when assessment scores incorporate both the cause and effect of human land use. It is, thus, questionable whether rapid assessments should include stressors when assessing the condition of a wetland or if these metrics should be recorded and evaluated separately, especially during the calibration and validation phase. In other words, should both the stressor and response variables be averaged to produce a single final score?

Additionally, in rating the capacity of wetlands to perform specific functions, the HHGM assessments included human value attributes, such as flood control and protection of human property as "opportunity" variables. While it is tempting to incorporate human values into a rapid assessment methodology, this technique should be avoided since human values are subject to change with time and culture. For example, the Hawaiian culture values wetlands for food production, but this was not included as an "opportunity" variable in the HHGM method. Furthermore, no assessment technique will likely be robust enough to evaluate both functions and values and therefore functional assessments should be science based (Brinson 1993). Instead, human value attributes could be recorded and analyzed using separate score sheets. In Hawai‘i these score sheets should include values such as opportunities for recreation and education as well as traditional Hawaiian cultural values, including the protection and restoration of fishponds and taro farms for food production.

Since most RAMs include metrics for measuring a wetland's capacity to support wildlife, they should be validated with detailed biological surveys of algae, vegetation, macroinvertebrates, invertebrates, or fish (Fennessy et al. 2004, Sutula et al. 2006). Furthermore, biological assessments have been shown to be more reliable indicators of water quality compared with traditional chemical analysis (Wang et al. 2001). Thus, future research on the development of a Hawai`i RAM should compare indicators of wetland condition with a full suite of biotic indicators. Expansion of the EMAP survey to include all types of coastal wetland ecosystems would also facilitate future attempts to validate and calibrate Hawaiian rapid assessment methods.

CHAPTER III: REMOTELY ASSESSING THE CONDITION OF HAWAIIAN COASTAL WETLANDS

INTRODUCTION

The condition of ecological communities is strongly related to the levels of human activity occurring within a landscape. Land-use, and especially the intensity of the use, affects ecological communities through direct, secondary, and cumulative impacts (Brown and Vivas 2005). The quality of water entering aquatic ecosystems, for instance, is affected by both point and non-point sources of pollution occurring within a watershed. Excess nutrients and sediments draining into a wetland can cause various problems such as the spread of invasive plants, increased turbidity, and a host of problems associated with eutrophication, including low levels of dissolved oxygen and excessive algal growth (Dunne and Leopold 1978). Aquatic ecosystems, especially those at the receiving end of topographic basins, are inextricably linked to the condition of their riparian buffers and watersheds (Allan et al. 1997, Wang et al 2001, Gergel et al. 2002, Collins et al. 2008).

In order to comprehensively assess the functional integrity of wetlands, the U.S. Environmental Protection Agency (EPA) advises using Level I (remote), landscape-scale analysis in addition to Level II on-site rapid assessment methods (RAMs) and detailed field surveys (Level III) (Faber-Langendoen et al. 2008). This project focused on remote, Level I assessments and their ability to provide a quantitative measure of human disturbances in Hawaiian landscapes. In addition, the project investigated the potential impacts of land-use stressors on soil and water chemical parameters as well as the overall condition of wetlands, assessed using Level II methods. The integrity of landscapes were analyzed in a geographical information system (GIS) using existing data sets of roads and land-use/land-cover in order to quantify the land-use indicators at a variety of spatial scales.

Landscape Indicators

Various landscape indicators (e.g. forest cover, urban land-use cover, agricultural land-use cover, impervious surface cover, and width of riparian buffers) have been linked to variables such as levels of nitrogen (N) and phosphorus (P) in surface water and the diversity of native vegetation, fish, macroinvertebrates, and birds in aquatic ecosystems (Allan et al. 1997, Wang et al. 2001, Gergel et al. 2002, Cohen et al. 2004, Houlahan and Findlay 2008). For example, nutrient losses from agricultural watersheds are consistently higher than forested or grassland basins (Allan et al. 1997, Gergel et al. 2002). Urban land-use and the size of riparian buffers have also been linked to increases in N and P levels measured in wetland surface water (Gergel et al. 2002) and to fish species richness (Wang et al. 2001). A study analyzing land-use impacts on fish biotic indicators within 47 watersheds in Wisconsin found that imperviousness was a better predictor of fish community attributes than variables such as urban or agricultural land-use cover (Wang et al. 2001).

While some research has shown a correlation between wetland degradation and land-use impacts at large spatial scales (Roth et al. 1996), others have shown stronger correlations between wetland health indicators and landscape disturbances within smaller (100m) buffer areas (Cohen et al. 2004, Brown and Vivas 2005). Many rapid condition assessment methods characterize the stressors to wetland condition by evaluating the land-use intensity occurring within 100m or 250m from the wetland edge (Miller and Gunsalus 1997, Cohen et al. 2004, SAIC 2004, Collins et al. 2008). Some research suggests that vegetated riparian buffers, as narrow as 15m, significantly reduce sediments and nutrients from being transported from terrestrial areas into aquatic habitats (Osborne and Kovacic 1993, Castelle et al. 1994, Wenger 1999). Others have suggested that the appropriate buffer width should be based on the quality of

the wetland and the intensity of the adjacent land-use (Miller and Gunsalus 1997). Additionally, wildlife may require wider buffers (> 100m) depending on the resource needs of a particular species (Miller and Gunsalus 1997).

Numerous studies have also suggested that wetland ecosystems are negatively impacted by land-use activities occurring in locations distant from the wetland edge (Roth et al. 1996, Allan et al. 1997, Wang et al. 1997, Wang et al. 2001, Houlahan and Findlay 2004, Mack 2006, Bruland and MacKenzie 2010). For example, Bruland and MacKenzie (2010) found that $\delta^{15}N$ concentrations from plant tissue collected in 34 Hawaiian coastal wetlands were positively correlated with urban land-use within 1 km radii and with watershed human population densities. Houlahan and Findlay (2004), in Ontario Canada, found that surface water N and P levels were negatively correlated with forest cover calculated for distances as great as 2,250m and soil P levels were negatively correlated with forest cover calculated at 4 km from the wetland edge. Wang et al. (2001) found that imperviousness in Wisconsin watersheds was more strongly correlated with the health of riverine fish communities than with any other land cover category (including agricultural and woodland) calculated at smaller distances (50m, 100m, 1.6Km, 1.6 – 3.2Km and 3.2Km). Research in Michigan indicated that land-use metrics calculated for watersheds were better predictors of stream biotic integrity than local land-use (Roth et al. 1996, Allan et al. 1997).

Remote condition assessments such as landscape development intensity (LDI) indices represent a fairly new approach to estimating the cumulative impacts from human disturbances at various spatial scales (Cohen et al. 2004, Brown and Vivas 2005, Mack 2006). The advantage of an LDI is that it quantifies human disturbances along a continuous gradient, from one, for natural systems, to ten for central business districts (Brown and Vivas 2005). The calculation of

LDI indices is based on concepts presented in Odum's (1995) emergy analysis, used in accounting for the amount of non-renewable energy required for different land-use activities. The LDI indices have been shown to be good predictors of wetland floristic quality (Cohen et al. 2004, Mack 2006) and condition (Mack 2006). Although LDIs were first developed for landscape assessments in Florida, they have also been shown to be correlated with wetland condition in Montana (Vance 2009) and Ohio (Mack 2006).

Soil Quality Indicators

Due to their importance in ecosystem functioning, soil properties, such as texture and chemistry are frequently measured or described during condition assessments (Brinson 1993, Bruland and Richardson 2005, Bruland and Richardson 2006, Nelson et al. 2007, Bantilan-Smith et al. 2009). High soil bulk density (BD), for example, usually indicates reduced soil porosity, which leads to increased overland surface flow, erosion, and sedimentation. Compacted soils can also affect ecosystem processes by inhibiting root growth, limiting microbial habitat, and reducing hydrological functioning. In general, human land use tends to compact soils and increase BD.

Soil organic matter (SOM), on the other hand, reduces BD, aerates soils, and provides a major source of plant nutrients, including P, N, and sulfur (Bruland and Richardson 2005, Bruland et al. 2009). Furthermore, SOM is the primary source of carbon for heterotrophic soil organisms and has been shown to be a good predictor of denitrification potential, as well as fish and invertebrate species richness (Findlay et al. 2002). The amount of SOM, in undisturbed sites, represents the balance between net primary productivity and decomposition, both of which are determined by temperature, hydrology, and nutrient availability (Mitsch and Gosselink 2007). In addition, research has shown that created and restored wetlands tend to have less

SOM than naturally occurring, unaltered wetlands (Shaffer and Ernst 1999, Bruland and Richardson 2006, Bantilan-Smith et al. 2009).

Water Quality Indicators

Typically, water quality is evaluated based on chemical and to some extent physical characteristics (i.e. turbidity). Hierarchical sets of indicators have been suggested for a more comprehensive analysis of water quality in the U.S. (Wang 2001, Karr and Yoder 2004, Faber-Langendoen et al. 2008) that includes land-use stressors (i.e. development and flow alteration) and biological indicators (Wang 2001). For example, stable nitrogen isotopes have been used successfully to identify sources of dissolved inorganic nitrogen (DIN). A study in Hawai`i showed that $\delta^{15}N$ concentrations measured in plant tissue collected in Hawaiian coastal wetlands were positively correlated with watershed population densities, indicating increases in dissolved inorganic nitrogen (DIN) from human waste (Bruland and MacKenzie 2010).

Rapid Assessments

Rapid assessment methods (RAMs) use easily identifiable field indicators to assess a wetland's overall ecosystem integrity or capacity to perform specific functions, relative to wetlands intact well-functioning ecosystems (Fennessy et al. 2004, Faber-Langendoen et al. 2008). The intent of these assessments is to rapidly and accurately quantify the extent to which ecosystems depart from full ecological integrity or best attainable condition. While some RAMs qualitatively assess the condition of wetlands based on a series of narratives describing different levels of condition, others are more quantitative or use a combination of qualitative and quantitative indicators to assess the capacity of wetlands to perform specific functions (i.e.

water storage, dissipation of energy, nutrient cycling). In addition, RAMs typically include variables for rating a wetland's capacity to support wildlife habitat.

Hawaiian Landscapes

Each of the major islands in Hawai`i is a discrete hydrological system with a relatively large number of small watersheds. The islands include a total of 614 watersheds ranging in size from < 0.04 ha to > 21,448 ha, most of which (97%) are smaller than 2,000 ha (GDSI 1995). The watersheds on the older, more eroded volcanoes (i.e. Kauai and O`ahu) tend to be relatively small with steep slopes. The steep mountainous slopes shed water rapidly, producing short flashy streams that respond quickly to rainstorms with the potential of delivering substantial amounts of nutrients and sediments to coastal waters (Laws et al. 1994, Laws et al. 1999, Ringuet and Mackenzie 2005). It is not uncommon for the volume of water in a stream to increase by a factor of 100 within a single hour (Laws et al. 1999, Oki et al. 2010). Perennial streams are more common on the older volcanic slopes of Kauai and O`ahu, and northeastern slopes of Maui. The younger shield volcanoes on Hawai`i Island have gentler slopes with porous substrates and perennial streams that occur only on the oldest and wettest north-eastern slopes of Mauna Kea and Kohala volcanoes. Perennial streams, however, are rare on the drier leeward sides of all islands.

Along the leeward coasts direct precipitation is a negligible source of water relative to groundwater flow from the mountainous interior, which is a more stable, less seasonally-variable source of water. Except for the wetlands that are hydrologically connected to the ocean, these leeward depressional wetlands can experience water deficits when the water table is low (Oki 1999). Furthermore, long term salinity and temperature studies suggest that salinities in

some coastal wetlands may be increasing due to either a decrease in groundwater or an increase in marine water flow due to sea level rise (Hoover and Gold 2005).

The soils in Hawai`i are primarily volcanic in origin, formed from basaltic lavas and to a lesser degree andesitic lavas, ash, spatter and cinder. In some relatively small areas along the coast, some soils are of coralline origin. The age of the surface varies from many thousands of years to young newly erupted lavas. This range in substrate age combined with the variety of climates found in Hawai`i has produced a remarkable diversity of soils. In fact, eleven of the twelve soil orders in soil taxonomy are found in Hawai`i, with nine occurring on O`ahu alone (Gavenda et al. 1998).

Human Land-use Impacts

Most of the people in the State of Hawai`i reside on the island of O`ahu, with an average population density of 600 people km^{-2}, followed by Maui (82.9), Kaua`i (52.5) Hawai`i (16.0), Moloka`i (12.1), Lana`i (11.7), and Ni`ihau (0.9) (DBEDT 2010). Problems associated with human land-use in Hawai`i include modification of the natural hydrology (DBEDT 2010), increases in runoff of polluted waters (Laws et al. 1999, De Carlo et al. 2007, Bruland and MacKenzie 2010), and the introduction of invasive species (Stone 1988, Bantilan-Smith et al. 2009). Flood control projects result in artificially straightened concrete-lined stream reaches, flat-bottomed channels and reinforced banks, primarily at the mouths of rivers (Parrish et al. 1978, DLNR 1990, Brasher 2003, DBEDT 2010). A survey in 1978 found that 15% of the streams in Hawai`i had been altered, 89% of which were on O`ahu (Parrish et al. 1978). As of 1990 approximately 56% of the 366 perennial streams in Hawai`i had been lined, straightened, or otherwise channelized in some way (DLNR 1990); less than 14% of the streams were determined to be physically pristine (Parrish et al. 1978). The modification of streams in

Hawai`i has resulted in reduced water flow and lower water quality and has degraded the physical habitats of native streams (Brasher 2003).

The nonpoint sources of pollution in Hawai`i include sediments, nutrients, toxic chemicals, pathogens, and acidity. Agriculture, forestry, urban development and hydromodification activities are the primary sources of polluted runoff problems (Laws and Ferntino 2003, De Carlo et al. 2007). Furthermore, researchers have estimated that a total of 1,139,844 tons of sediments are eroded from the main Hawaiian Islands each year (Gavenda et al. 1998) with the highest rates of soil erosion from barren land, cropland, pasture, and rangeland (Dunne and Leopold 1978, Gavenda et al. 1998).

Degradation of landscapes have been shown to impact wetland and aquatic ecosystems in Hawai`i. Research by Bruland and MacKenzie (2010) found that $\delta^{15}N$ isotope levels in wetland plant tissue (an indicator of human and animal waste) were positively correlated with urban land-use within a 1,000-m radius, and with watershed population densities. Kido (2008) in a field reconnaissance study in Hawai`i found strong positive correlations between watershed condition and stream biotic integrity. In addition, land-use stressors may be reducing the diversity of benthic species in Hawaiian estuaries (Nelson et al. 2007).

Objectives

The primary objective of this study was to determine the relationship between Hawaiian landscapes and the condition of coastal wetland ecosystems. I hypothesized that wetlands located in the least developed landscapes would have better soil and water quality and would receive the highest condition assessment scores. Specifically, I hypothesized that LDIs, calculated at three scales (100m, 1,000m, and entire watershed) would be:

- Positively correlated with nutrient levels measured in wetland surface water: total dissolved nitrogen (TDN), nitrite+nitrate (NO_2+NO_3), and total phosphorus (TP), phosphate (PO_4); and with $\delta^{15}N$ levels in wetland plant tissue
- Positively correlated with soil bulk density (BD)
- Negatively correlated with soil total carbon (TC), and total nitrogen (TN), extractable phosphorus (ExP)
- Negatively correlated with wetland rapid assessment scores

METHODS

The wetland assessment areas were determined from existing GIS layers of wetlands (USFWS 2009) and water bodies (USGS 2000) and were based on clear breaks in surface hydrology, sediment supply, and geomorphology per CRAM guidelines (Collins et al. 2008). In most cases these assessment areas closely followed the wetland areas as delineated by the National Wetland Inventory, which used the Cowardin classification system (1979) to map the wetlands (USFWS 2009). However, some wetlands, such as the Percolation Ditch, were not mapped by NWI; these wetlands were delineated using GIS layers of water bodies (USGS 2000) and Quickbird satellite imagery (2005). Because larger wetlands tend to have greater structural complexity (Collins et al. 2008), which results in higher CRAM and HHGM scores, at least two partial assessment areas were used for wetlands greater than 2.5 ha. The partial assessment areas were limited to 1 ha. Scores from all partial assessment areas were averaged to provide a final score for the entire wetland assessment area.

Two overlapping buffers (100 m and 1,000 m wide) were generated around each of the 27 wetlands, using the buffer function in ArcGIS (Version 10.0, ESRI, Redlands, CA). The watershed boundaries (GDSI 1994) were downloaded from the Hawai`i State GIS web site

(http://Hawai`i.gov/dbedt/gis/). In cases where wetlands were bisected by watershed boundaries, both watersheds were used in the analysis. The buffers and watersheds were then used to "clip" the land-use/land-cover data, mapped from satellite imagery (30m resolution, NOAA 2001).

The LDI coefficients were assigned to each land-use/land-cover class (NOAA 2001) within the buffers and watersheds. In the analysis, these values were called LDIs (Table 3.1). A second set of LDI values, called LU-LDIs (Tables 3.2 and 3.3), were calculated using both land cover data (NOAA 2001) and land-use information derived from county level zoning data (2005) and from Maui County Tax Map Parcel data (2009). The detailed county zoning data were not available for Maui; therefore the County Tax Map Parcel PITT Codes (tax rate codes) were used instead (Table 3.3). The LDIs were then calculated for the watersheds and buffer zones (100 and 1,000-m) of 27 coastal wetlands using the following data sources: land cover (NOAA C-CAP 2001), Hawai`i County Zoning (2005), City and County of Honolulu Zoning (2005), and Maui County Tax Map Parcel data.

Table 3.1. LDI coefficients assigned to the land-cover classes mapped by NOAA (2001).

Land Cover (NOAA 2001)	**LDI**
Background	0
Unclassified	0
High Intensity Developed	9.19
Low Intensity Developed	7.55
Cultivated Land	4.54
Grassland	1
Deciduous Forest	1
Evergreen Forest	1
Mixed Forest	1
Scrub/Shrub	1
Palustrine Forested Wetland	1
Palustrine Scrub/Shrub Wetland	1
Palustrine Emergent Wetland	1
Estuarine Forested Wetland	1
Estuarine Scrub/Shrub Wetland	1
Estuarine Emergent Wetland	1
Unconsolidated Shore	1
Bare Land	1
Water	1

Table 3.2. The LU-LDI coefficients assigned to land-use/land-cover classes, derived from combined maps of land-cover (NOAA 2001) and County Zoning data (for Hawai`i and O`ahu).

LDI class (Brown and Vivas 2005)	**Land Cover (NOAA 2001)**	**Hawai`i Zoning / Land-use**	**LU-LDI**
Natural system	Bare land, Shrub/Scrub, Evergreen Forest		1
Natural open water	Open Water		1
Recreational / Open space, low intensity	Grassland		1.83
Woodland Pasture (with livestock)	Scrub/Shrub		2.02
Improved pasture (without livestock)	Grassland, Bare land	Agriculture	2.77
Improved pasture - low intensity (with livestock)	Pasture/Hay		3.41
Improved pasture (high intensity with livestock)	Grassland		3.74
Row crops	Cultivated crops		4.54
Single family residential - low density		Residential, min 10,000 sq’	6.9
Recreational / open space - high intensity	Developed, open space		6.92
Single family residential - medium density	Developed, low intensity	Residential, min 5-7,500 sq’	7.47
Single family residential - high density	Developed, medium intensity	Residential, min 3,500 sq’	7.55
Low-intensity commercial	All	Neighborhood, Community Business, Mixed	8
Institutional	All	Developed Federal and Military lands	8.07
Industrial	All	Limited and Intensive Industrial District	8.32
Multi-family residential (low rise)	All	Low Density Apartment	8.66
High Intensity commercial			9.18
Multi-family residential (high rise)		Medium and High Density Apartment	9.19
Central business district (average 2 stories)	Developed, high intensity		9.42
Central business district (average 4 stories)		Central Business Mixed Use	10

Table 3.3. The LU-LDI coefficients assigned to each land-use/land-cover class, derived from combined maps of land cover (NOAA 2001) and Maui Tax Map Key data.

Land-use (Brown and Vivas 2005)	**Land Cover (NOAA 2001)**	**Maui County Tax Category (PITT Codes)**	**LU-LDI**
Natural system	Wetlands	All	1
Natural system	Bare land, Forest, Shore, Water	Conservation	1
Natural open water	Open Water	All	1
Pine Plantation	NA	NA	1.58
Recreational / Open space, low intensity	Grassland		1.83
Woodland Pasture (with livestock)	Scrub/Shrub, Forest	Agricultural	2.02
Improved pasture (without livestock)	Bare land, Grassland	Agricultural	2.77
Improved pasture - low intensity (with livestock)			3.41
Improved pasture (high intensity with livestock)			3.74
Row crops	Cultivated Land		4.54
Single family residential - low density		Unimproved Residential	6.9
Recreational / open space - high intensity			6.92
Single family residential - medium density	Developed, low intensity	Improved Residential	7.47
Single family residential - high density	Developed, medium intensity		7.55
Low-intensity commercial			8
Instititutional			8.07
Industrial		Industrial	8.32
		Commercial	8.55
Multi-family residential (low rise)		Apartment	8.66
High Intensity commercial			9.18
Multi-family residential (high rise)	Developed, high intensity	Hotel and Resort	9.19
Central business district (average 2 stories)	Developed, high intensity		9.42
Central business district (average 4 stories)			10

The final LDI values, calculated for the landscapes of each wetland, were area-weighted using ArcGIS and the equation: $LDI_{final} = \sum \%LU_i * LDI_i$, where LDI_{final} = LDI ranking for landscape unit (i.e. buffer zone or watershed), $\%LU_i$ = percent of the total area in land-use *i*, and LDI_i = landscape development intensity coefficient for land-use category *i.* For comparison, landscape metrics on percent forest and road density were also calculated for each wetland buffer and watershed. The land cover (NOAA 2001) data were used in calculating percent forest, and digital line graphs, mapped at the 1:24,000-scale (USGS 1998) were used in calculating road density. All road categories, including 4WD dirt roads as well as paved major roads were included in road density calculations (m km^{-2}).

Analysis of correlation (Minitab Version 15, State College, PA) was used in determining the strength of the relationships among landscape metrics (LDI, % forest, and road density) and soil and surface water quality data collected during intensive (Level III) field surveys: mean soil BD, TC, TN, SOM, ExP, and pH (Bantilan-Smith et al. 2009), surface water Temp, Cond, NO_2+NO_3, TDN, PO_4, TP, wetland plant tissue $\delta^{15}N$ values (Bruland and MacKenzie 2010). In addition, correlation coefficients were also calculated between landscape indicators and (Level II) wetland rapid assessment scores. The water quality data were collected between 2007-2009, plant tissue samples for $\delta^{15}N$ measurements were collected between March and April 2007 (Bruland and MacKenzie 2010), and soil parameters were collected in March and April of 2007 (Bantilan-Smith et al. 2009).

Soil samples from intermediate and wetter zones of the 27 wetland sites were used in the analysis; results from drier zones were omitted because they were more representative of upland terrestrial soils than those most typically found in wetlands. The distributions of many of the soil and water parameters were skewed to the right. Therefore, Spearman correlation coefficient

analysis was used in examining the relationship between the LDI values and soil and water quality parameters. All statistical analysis was done using Minitab software (Version 15, State College, PA).

Three different methods were used for rapidly assessing the condition of 27 wetlands: Florida Wetland Rapid Assessment Procedures (WRAP [Miller and Gunsalus 1997]), California Rapid Assessment Methods (CRAM [Collins et al. 2008]), and the draft Hawai`i Hydrogeomorphic Method (HHGM [SAIC 2004]; see Chapter 2). The final rapid assessment scores were calculated on continuous gradients: 1-3 (WRAP), 1-120 (Hawai‘i HGM), and 1-100 (CRAM) and then divided by their maximum possible scores (i.e. rescaled to values from 0 to 1) so that they would be comparable. In order to avoid artificially increasing the correlation between RAMs and landscape development indices (LDI), the metrics for anthropogenic stressors within the wetland buffers were first subtracted from rapid assessment scores before the relationship between these two variables was analyzed. Thus the WRAP water quality variable was excluded from the WRAP scores; the buffer width, length and condition variables were excluded from the CRAM scores; and the buffer land-use intensity variables were excluded from the HHGM scores. In addition, in order for the scores to be more scientific and comparable, the opportunity variables and modifiers were also excluded from the final HHGM scores (see Appendix A).

RESULTS

The study sites included a range of wetlands from those that were relatively isolated and surrounded predominantly by natural vegetation to those surrounded primarily by urban development. The LDI gradient for the watersheds varied from 1.0 to 4.57; the maximum LDI was 5.83 and 4.96 for the 100m and 1,000m buffers respectively (Figure 3.1). The mean LDI

values were relatively low: 2.04 for 100m buffers, 2.53 for 1,000m buffers, and 2.31 for the watersheds. The road densities among the 100m buffers had the widest range, 0 to 14,844 m km^{-2} (100m buffers), compared with 1,045 to 10,166 m km^{-2} for 1,000m buffers, and 776 to 6,641 m km^{-2} for watersheds. Percent forest cover, on the other hand, varied from a minimum of 0 for all landscape scales, to a maximum of 54% for 100m buffers, 44% for 1,000m buffers, and 61% for entire watersheds.

The least developed watersheds were primarily located on the Island of Hawai`i, whereas the more developed landscapes were generally located on O`ahu (Figure 3.2). Based on the LDI values calculated from the land cover data (LDI), 14 of the 27 wetlands were located in "natural" watersheds, with LDI values less than 2. The remaining watersheds received LDI values between 2 and 5, values typically associated with mixed land-use (i.e. natural, agricultural, and urban). None of the watersheds received values greater than 5, the threshold used in defining predominantly urban landscapes; the 100m buffer of only one wetland (Ka`elepulu on O`ahu) exceeded that threshold value.

All of the landscape-scale indicators were most strongly correlated with each other at the watershed scale, while the scores for the 100m buffers were not correlated at all (Table 3.4). The strongest correlation occurred between the LDI scores and road density calculated at the watershed scale (Figure 3.3, Table 3.4). Although forest cover and LDI scores were derived from the same data source (NOAA 2001), they were not as strongly correlated with each other as the LDI scores were with road density. Not surprisingly, forest cover and road density were negatively correlated. However, some leeward dry landscapes, such as those upland of the Nu`u (Maui), Kaloko (Hawai`i), and 'Aimakapa (Hawai`i) sites, had low forest cover as well as low road densities, which reduced the strength of the correlation between the two landscape metrics.

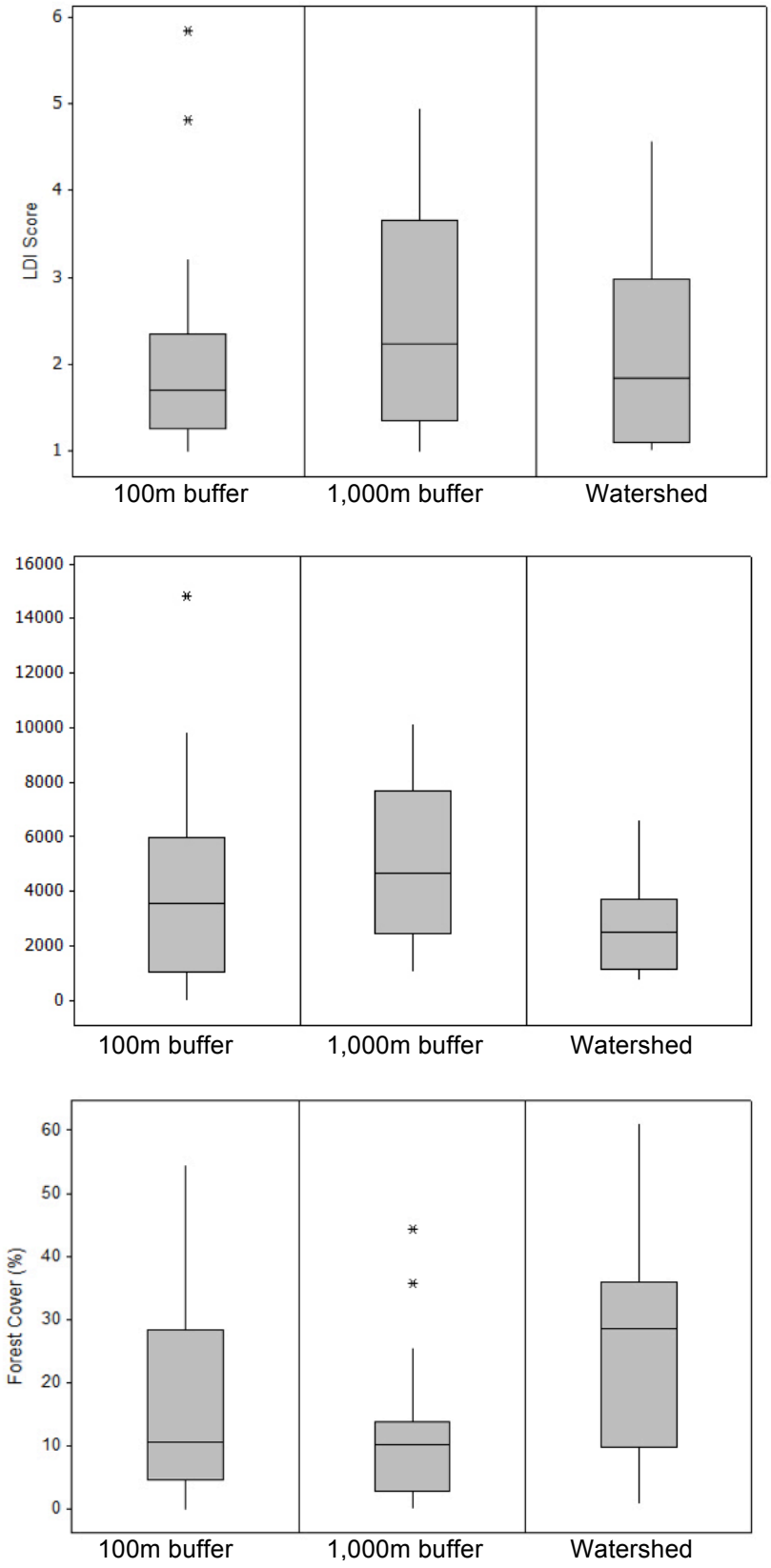

Figure 3.1. Box-and-whisker plots showing the range of LDI scores (top) road densities (middle) and percent forest cover (bottom) calculated for100m buffers, 1,000m buffers, and watersheds of 27 wetlands.

Table 3.4. The Spearman correlation coefficients between the landscape indicators, calculated for the 1,000m buffers and watersheds of 27 wetlands.

	Roads (1000m)	**Forest (1000m)**
Forest (1,000m)	-0.383	
p value	< 0.025	
LDI (1,000m)	0.835	-0.416
p value	< 0.001	< 0.01
	Roads (Watershed)	**Forest (Watershed)**
Forest (Watershed)	-0.658	
p value	< 0.001	
LDI (Watershed)	0.945	-0.702
p value	< 0.001	< 0.001

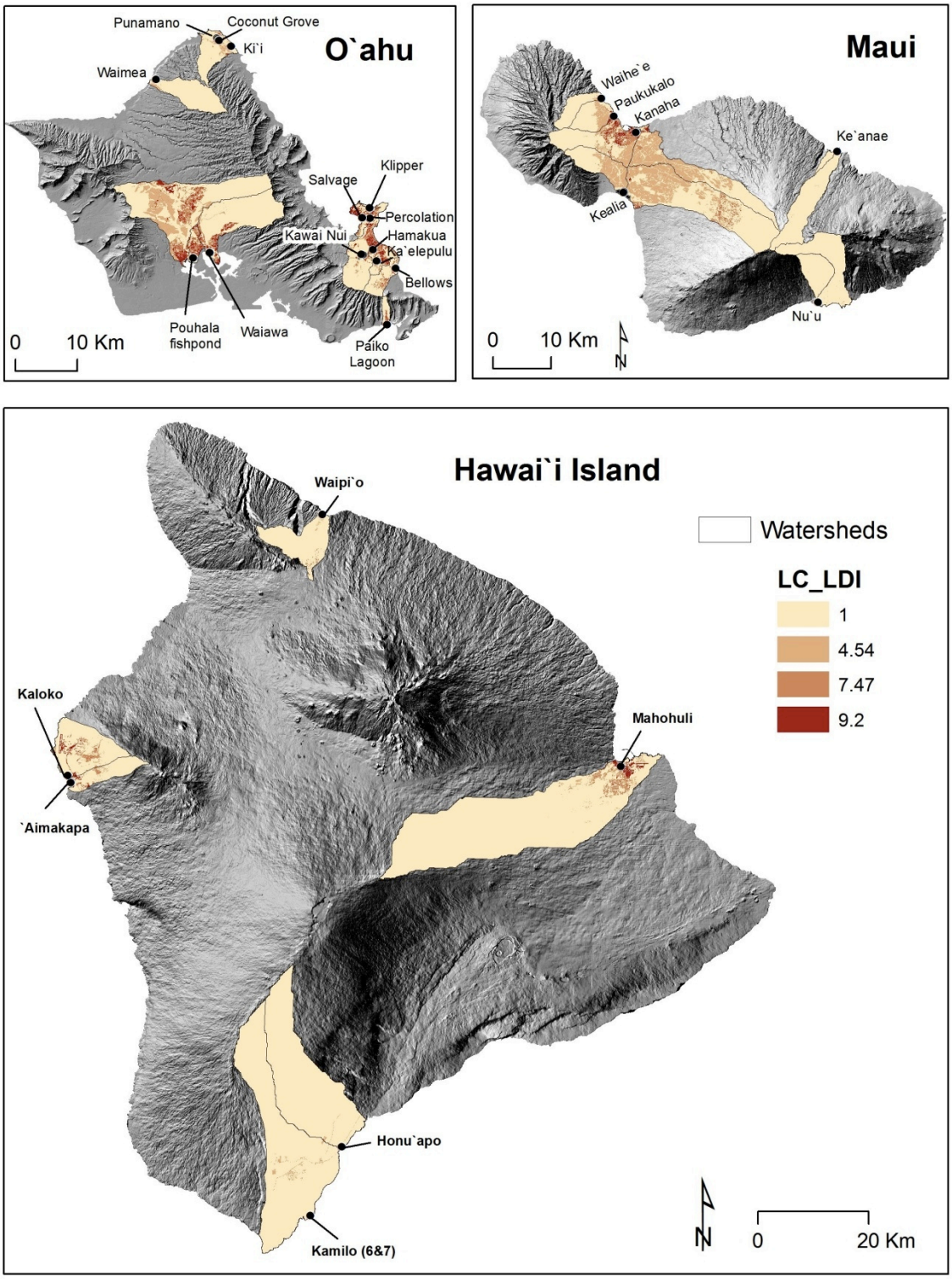

Figure 3.2. Maps showing the LDI values calculated from land-cover maps (NOAA 2001) for the watersheds on the islands of Maui, O`ahu, and Hawai`i. Darker shades represent higher land-use intensity.

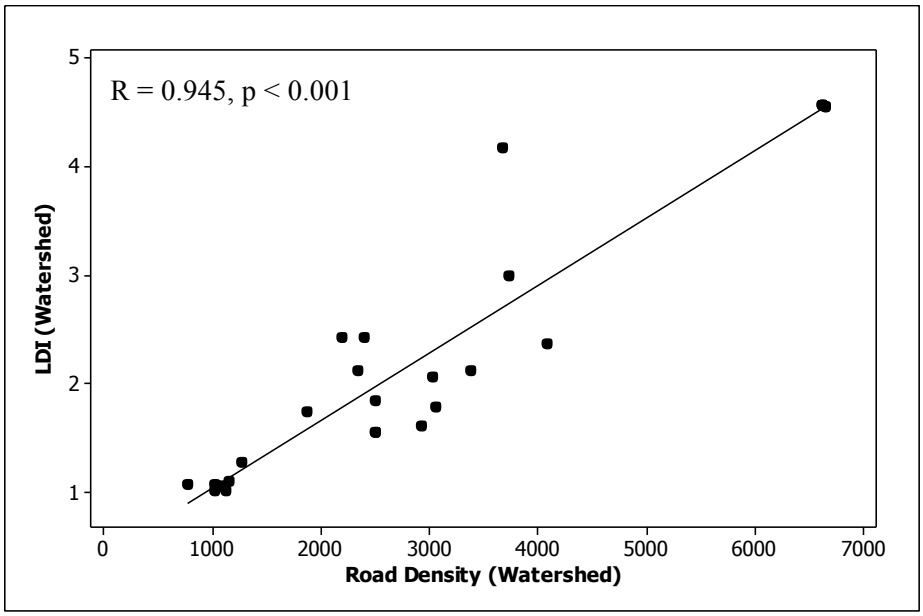

Figure 3.3. Scatterplot showing the relationship between road density and LDI values calculated for the watersheds of 27 wetlands.

The highest road densities (6,641 and 6,618 m/km) and LDI values (4.5, 4.7) were recorded in the Ka`elepulu and Pu`u Hawai`i-loa watersheds, respectively. Five (Ka'elepulu, Hamakua, Salvage Yard Pond, Klipper Pond, and Percolation Ditch) of the 27 wetlands are located on the northeastern side of O`ahu within these two watersheds. The lowest road densities (775.75 m km^{-2}) were calculated for the watersheds draining into the Ke`anae wetland (Nua`ailua and Pinaau), which also received a low average LDI score of 1.07 and the highest forest cover (>61%). Although there was pastureland and residential development in the lower sections of the Hilea and South Point watersheds on Hawai`i Island, these large watersheds were also dominated by forests and barren lava flows, which resulted in low road densities and LDI scores. In addition, the Kaupo watershed on Maui, and the Waimea watershed on O`ahu also had low road densities and LDI scores. Although the Kaupo watershed, where the Nu`u wetland is located, is considered a natural watershed, and received a low LDI score of 1.21,

only 2.4% forest cover was calculated for this relatively dry watershed, of which most (> 59%) was mapped as grassland (Bruland and MacKenzie 2010).

The Kanaha Pond site (central Maui) had the highest road densities as well as the highest forest cover (>54%) within 100m of the wetland edge. The Ka`elepulu site (O`ahu) had the highest LDI scores within the 100m wetland buffers. The 100m buffers surrounding the Paiko and Klipper pond sites (O`au) had the lowest forest cover (0%). Surprisingly, close to 54% of the 100m buffer surrounding one of the ponds at Kamilo was mapped as forest. However, site visits and aerial photos revealed that the area classified as forest, was predominantly *Scaevola sericea* (beach naupaka) with patches of trees closer to the coast. In contrast, the *Prosopis pallida* (kiawe) "forests" surrounding several wetlands, including Nu`u, Waihee, Paiko, `Aimakapa, and Kaloko, were classified as scrubland (NOAA 2001).

Correlation with Field Surveys

Both road density and LDI scores, calculated at landscape scales, were correlated with a number of variables, particularly with soil BD, soil TN, soil pH, $\delta^{15}N$ plant concentrations, surface water TDN and CRAM scores (Table 3.5). In general, the correlations with the detailed Level III data were strongest relative to road density, with the strongest correlation between road density and $\delta^{15}N$ levels in wetland plant tissue ($r = 0.68$, $p < 0.01$; Figure 3.4). Interestingly, soil pH was positively correlated with road density (in 1,000m buffers) and negatively correlated with forest cover (within 1,000m buffers and watersheds; Table 3.5).

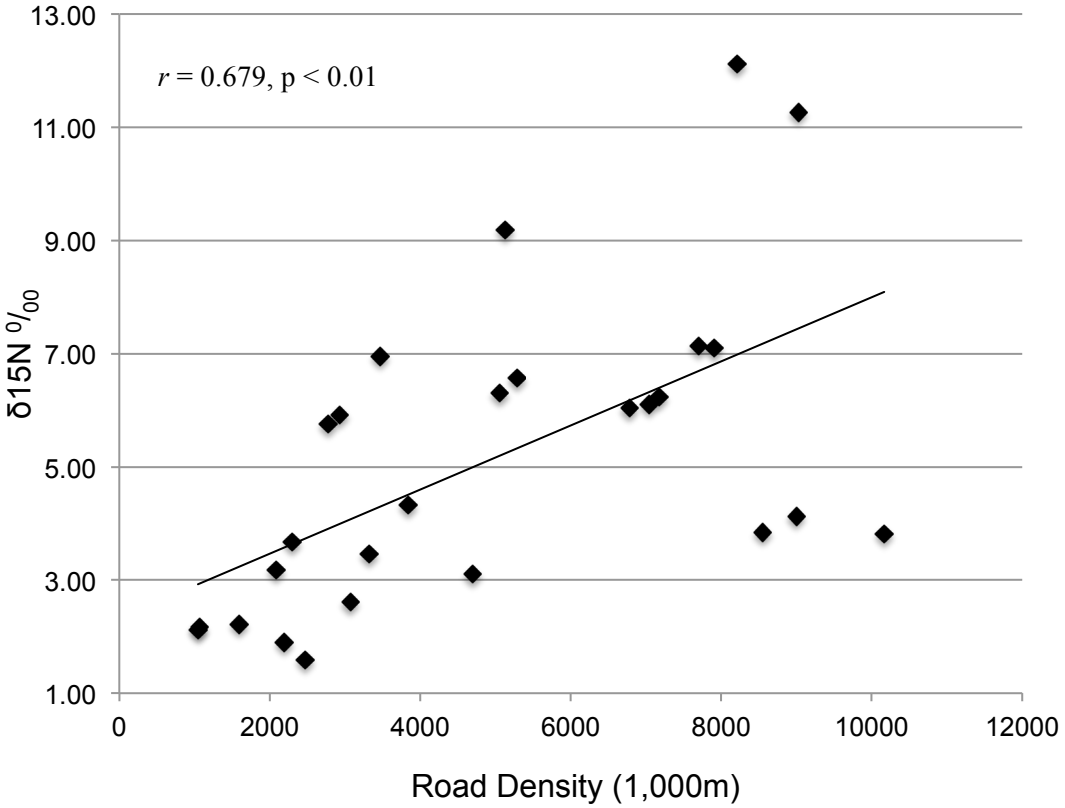

Figure 3.4. Scatterplot showing the relationship between $\delta^{15}N$ levels in wetland plant tissue and road density within the 1,000m buffers of 27 wetlands.

Indicators of landscape condition for the 1,000m buffer and watershed scales were more strongly correlated with wetland water and soil quality parameters than the landscape indicators calculated for the 100m buffers. In general the relationships between field data and landscape metrics calculated for the 100m buffers were weak or non-existent (Table 3.5). With one exception (road density within 100m buffers), the landscape metrics were most strongly correlated with $\delta^{15}N$ levels in wetland plant tissue.

Soil total nitrogen (TN) was negatively correlated with LDIs (at all scales), and with road density within the 1,000m buffers and watersheds. Soil extractable P and SOM were also negatively correlated with LDIs and road densities calculated for the 1,000m buffers and watersheds. Levels of TDN measured in the wetland surface water were positively correlated with LDI values and road density and negatively correlated with percent forest, particularly at

the watershed scale. In general the proportion of forest (in 1,000m buffers and watersheds) was more strongly correlated with water quality (Temp, TDN, and TP), and significantly correlated with only one soil quality parameter (pH).

Table 3.5. Spearman correlation coefficients between landscape indicators and independent soil and water quality data collected during intensive field surveys. No asterisk = $p < 0.005$, * = $p \leq .01$, ** = $p \leq .025$, *** = $p \leq 0.05$, ns = not significant.

	LDI 100m	**LDI 1000m**	**LDI Basin**	**%Forest 100m**	**%Forest 1000m**	**%Forest Basin**	**Road Density 100m**	**Road Density 1,000m**	**Road Density Basin**
Soil Parameters									
BD	ns	0.421**	0.458**	ns	ns	ns	ns	0.501*	ns
pH	ns	0.489*	0.622	ns	-0.464**	-0.459**	ns	0.522*	0.523*
TN	ns	-0.428**	-0.468**	ns	ns	ns	ns	-0.463**	-0.371***
SOM	ns	-0.371***	-0.397**	ns	ns	ns	ns	-0.416**	ns
ExP	ns	-0.41**	ns	ns	ns	ns	ns	-0.355***	-0.347***
Surface Water Parameters									
Temp	ns	ns	ns	ns	ns	-0.384***	ns	ns	ns
TDN	0.381***	0.435**	0.53*	ns	-0.437**	-0.537	ns	0.432**	0.46**
TP	ns	Ns	0.336***	ns	ns	-0.343***	0.452**	ns	ns
PO_4	ns	-0.349***	ns	ns	ns	ns	ns	ns	-0.354***
NO_2-NO_3	ns	ns	ns	0.477*	ns	ns	ns	ns	ns
$\delta^{15}N$ in Wetland Plant Tissue									
$\delta^{15}N$	0.361***	0.634	0.709	-0.425**	-0.451**	-0.538*	ns	0.679	0.620
RAM (w/o Land-use)									
HHGM	ns	-0.433**	ns	ns	ns	ns	ns	-0.426*	ns
CRAM	-0.416**	-0.661	-0.64	ns	ns	0.363***	ns	-0.553	-0.595
WRAP	ns	-0.570	-0.468**	ns	ns	ns	ns	-0.488*	-0.362**

Correlation with Rapid Assessment Scores

The CRAM and WRAP scores were strongly correlated with LDI scores and road densities within the 1,000m buffers and watersheds, particularly between CRAM and LDI scores (Figure 3.5). There was only 1 significant relationship between the landscape variables calculated for 100m buffers: CRAM and LDI. Furthermore, forest cover was not correlated with any of the RAMs, aside from a weak correlation between CRAM and watershed forest cover (Table 3.5).

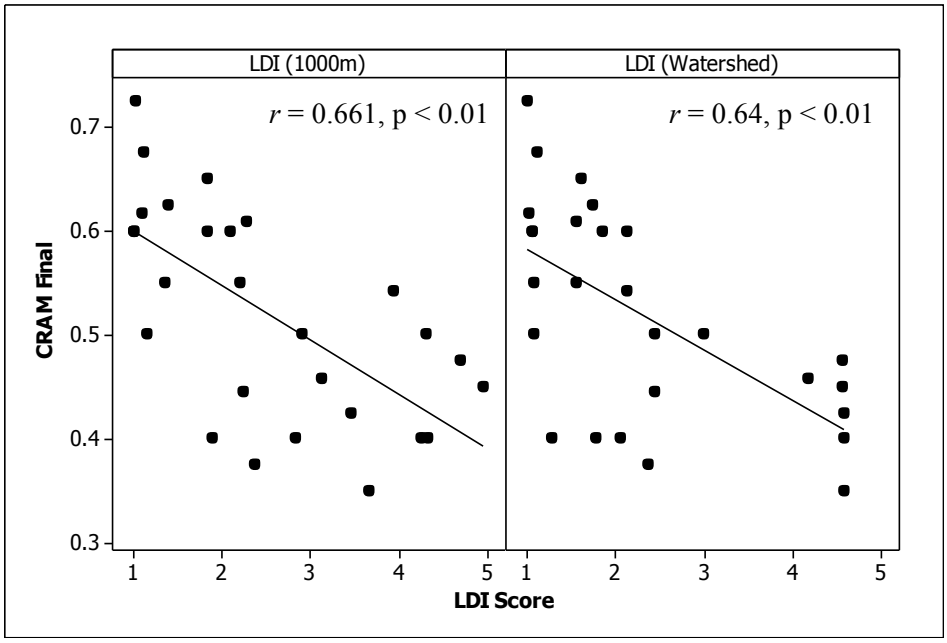

Figure 3.5. Scatterplots showing the relationship between CRAM scores and LDI values calculated for the 1,000m buffers and watersheds of 27 wetlands.

LDI Values, Including "Land-Use"

When the county zoning data were combined with land cover data (NOAA 2001) in calculating LDI scores, the mean LU-LDI values tended to be higher: 3.56 (100m buffers), 3.54 (1,000m buffers) and 3.17 (watersheds). Three watersheds (Ka'elepulu on O`ahu, Kapakahi, Pu`u Hawai`i-loa) had LDI scores greater than five, which is the threshold value for defining urbanized landscapes. Areas zoned for military use received LDI coefficients of 8.07 (Brown and Vivas 2005), which increased the LDI scores for landscapes on O`ahu. Although, the LDI scores calculated from slightly different data sources were correlated with each other (Figure 3.6), the final LU-LDI scores were not significantly correlated with most of the soil and water quality parameters (Table 3.6).

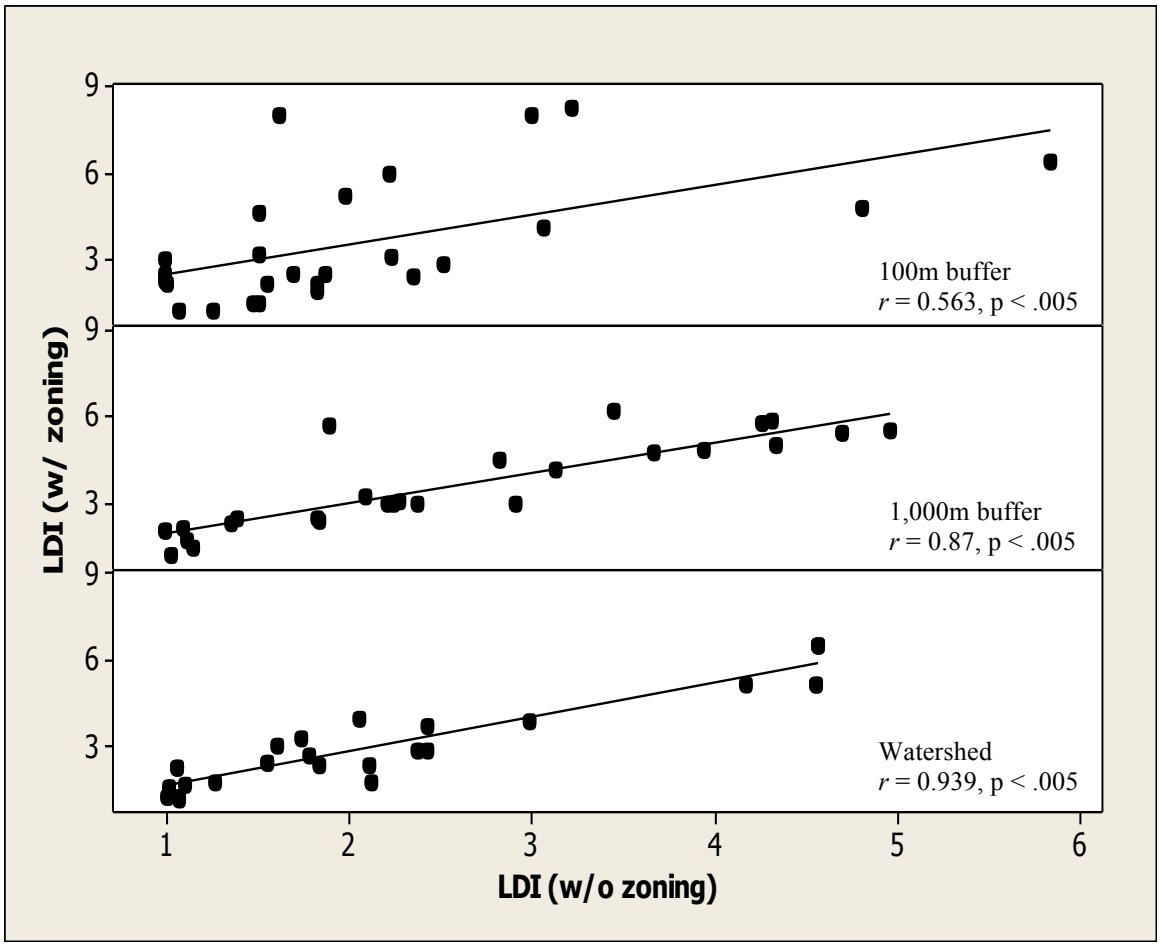

Figure 3.6. Scatterplots showing the relationship between LDIs calculated from 1) land cover data only and 2) land cover data combined with zoning and tax map key data.

Table 3.6. Spearman correlation coefficients among LU-LDIs, calculated for buffers (100m and 1,000m) and watersheds, and independent soil and water quality data collected for 27 wetlands. No asterisk = $p < 0.005$, * = $p \leq .01$, ** = $p \leq .025$, *** = $p \leq 0.05$, ns = not significant.

Parameter	**LU-LDI 100m**	**LU-LDI 1000m**	**LU-LDI Watershed**
Soil			
BD	ns	ns	ns
pH	ns	0.451**	0.481*
TN	ns	ns	ns
SOM	ns	ns	ns
ExP	ns	-0.342***	ns
Water			
Temp	ns	ns	ns
Cond	ns	ns	ns
NO_2-NO_3	ns	ns	ns
TDN	0.351***	0.444**	0.438**
PO_4	ns	-0.343***	ns
TP	ns	ns	ns
$\delta^{15}N$			
plants	0.476*	0.548	0.618
RAM (w/o land-use)			
HHGM	ns	ns	ns
CRAM	-0.541	-0.660	-0.576
WRAP	-0.448**	-0.490*	ns

DISCUSSION

This study investigated the applicability of LDIs in detecting a gradient in the condition of Hawaiian landscapes and the impacts on coastal wetland ecosystems from anthropogenic stressors occurring at various spatial scales (i.e. 100 m and 1,000 m buffers, and watersheds). In addition, the final LDI scores were compared with detailed on-site (Level II) water (Bruland and MacKenzie 2010) and soil quality (Bantilan-Smith et al. 2009) data collected during prior surveys as well as with on-site rapid assessments scores collected for this project. The intent was to determine if remote (Level II) assessments can provide information on the condition of

coastal wetlands in Hawai`i that is consistent with data from Level II (rapid assessments) and Level III (field surveys).

Results showed that, in general, wetlands located in disturbed landscapes with more roads and less forest cover tended to have the highest TDN levels in wetland surface water and the highest $\delta^{15}N$ concentrations in plant tissue. These wetlands also had lower soil nutrient levels and RAM habitat scores. However, while urban wetlands tended to have the highest $\delta^{15}N$ concentrations, some of the highest surface water TDN values were measured in wetlands located in agricultural (Waihe`e), urban (Kanaha and Pouhala) as well as natural (Nu`u) landscapes. Forest cover appeared to also play a role in TDN; surface water as well as groundwater driven wetlands that had less forest cover within the 1000m buffers also tended to have higher levels of TDN.

In addition, the proportion of forest within 100m buffers showed a surprisingly strong, and positive, correlation with surface water NO_2-NO_3. One potential explanation for this is that many of the wetlands in this study are surrounded by N-fixing trees such as *Prosopis pallida* (Kiawe) and *Leucaena leucocephala* (Koa Hoale). However, the NOAA (2001) landcover maps did not always classify those areas as forest; some areas dominated by *P. pallida* were mapped as shrubland while other areas dominated by shrubs such as *S. sericea* were mapped as forest. In addition, while high NO_2-NO_3 levels were recorded in some of the wetlands encroached on by *P. pallida*, other wetlands with high NO_2-NO_3 levels were surrounded by dense stands of *Rhizophora mangle* trees. It is feasible that tree leaf litter is providing substantial sources of N to the surface water of some wetlands. Dense populations of waterbirds may also be responsible for increased N in some wetlands. For example, more than 200 cattle egrets reportedly roost on

a seasonal basis in the trees (*P. Pallida*) nearby Nu`u wetland, where high $\delta^{15}N$ isotope levels were also recorded (Bruland and MacKenzie, 2010).

In general, the results indicate that Hawaiian coastal wetlands located in "natural" landscapes have accumulated more SOM and correspondingly have more nutrient rich soils with lower soil BD than wetlands in developed landscapes. With only one exception (Salvage Yard Pond, O`ahu), the wetlands with the highest SOM and correspondingly high soil nutrient levels tended to be "natural" wetlands or fishponds located on the least developed Island of Hawai`i. Inversely, the wetlands with the least SOM and lowest soil nutrient levels primarily included created or restored wetlands located on the islands of Maui and O`ahu (Bantilan-Smith et al. 2009). Since created and restored wetlands tend to be located in more developed areas, it is difficult to determine whether or not soil parameters are more directly related to wetland status (created vs. natural) or past and present disturbance.

With only one exception, the land-use/land-cover metrics calculated at regional scales (i.e. 1,000m buffer and entire watershed) were better indicators of soil and water quality than the metrics calculated within the immediate vicinity of the wetlands (100m buffers). Even the RAM scores (CRAM and WRAP) were only significantly correlated (with p-values < 0.05) with LDI values and road density calculated for 1,000m buffers and watersheds. These results contrast sharply with research by Cohen et al. (2004) and Brown and Vivas (2005) in which they found that 100m buffers adequately captured the effects of surrounding land use on depressional wetlands located in Florida's relatively flat terrain. When Brown and Vivas (2005) extended the buffers up to 500m, the strength of the relationship between LDI values and qualitative WRAP scores declined. Perhaps their use of detailed land-use maps, with 27 land-use categories, could partially explain why they found strong correlations between wetland

condition and 100m buffers. In contrast, this project used coarse (30m pixel resolution) land cover data (with only 4 LDI classes) and road data (mapped at 1:24,000 scale) in calculating the landscape metrics. While these data may be suitable for analysis at large spatial scales, they probably lack the level of detail needed to adequately characterize small areas such as 100m buffers. On the other hand, it may be that small 100m buffers sufficiently characterize land-use stressors in flat landscapes, whereas wetlands receiving inputs from steeper topographic terrain are impacted by land-use occurring farther away. In addition, the LDI scores calculated for the 100m buffers were lower than those calculated for the 1,000m buffers and watersheds, indicating lower land-use intensity within 100m of the wetlands.

Road density calculated for the 100m buffers, on the other hand, appears to be an especially poor indicator of wetland soil and water quality in these Hawaiian coastal wetlands. In fact, road density within the 100m buffers was not correlated with any of the soil or water quality parameters and surprisingly was negatively correlated with $\delta^{15}N$ levels. Perhaps this is because some wetlands are located within golf courses or are adjacent to large water bodies and have very few roads in the immediate (100m) vicinity, while other wetlands located in relatively natural landscapes are in close proximity to infrequently traveled dirt or paved roads.

In general, the correlations with forest cover were also weak. Misclassification errors between forest and shrubland likely contributed to these weak correlations. In addition, in some cases low forest cover may be more indicative of low rainfall than landscape degradation. For example, Kaupo was among one of the more natural watersheds in this study, but nonetheless had low forest cover due to relatively low levels of precipitation and erodible soils on steep slopes.

While LDI scores may be well suited for discerning gradients in landscape condition in Florida (Cohen et al. 2004, Brown and Vivas 2005), this research indicated that simple metrics such as road density were often as good as LDIs in relation to wetland soil and water quality. The high degree of correlation between LDI scores and road densities ($r = 0.95$, $p < 0.01$) suggests that both provide comparable indicators of landscape stress for Hawaiian landscapes that vary from natural, mixed agriculture, to urban. Furthermore, roads have strong spectral signatures and can therefore be readily mapped from satellite imagery compared with land-use, which is difficult to delineate using traditional image interpretation methods. For example, the NOAA (2001) land-cover maps classified bare disturbed land such as quarries as well as natural cover types such as beaches and lava flows all as bare land. In addition, pasturelands were classified as either grassland or scrubland (NOAA 2001). Thus, when assigning LDI coefficients to land-use/land cover classes, all bare lands, grasslands, and scrublands were assigned an LDI coefficient of 1 (for natural) despite their differences in land-use (i.e. pasture, golf course, urban park, wild grasslands). In addition, LDI coefficients of one were assigned to all water bodies, regardless of whether they were used extensively as harbors for commercial and military purposes or were designated marine protected areas. Needless to say more research is needed in order to determine the relationship between different levels of "water-use" and their potential impacts on coastal wetlands.

The effort to refine broad land cover categories and more accurately map land-use from zoning and permitting data, did not however, reveal stronger correlations between the LDIs (LU-LDIs) and independent field data. The lack of detailed and reliable land-use data for the entire State of Hawai`i may at least partially explain why the LU-LDIs were not as strongly correlated with soil and water quality. For example, the Tax Map Key (TMK) data used in

calculating the LU-LDI values for Maui were much less detailed than the county zoning data used in calculating LU-LDIs for the islands of Hawai`i and O`ahu. Furthermore, as mentioned previously, areas zoned for military use received LDI values of 8.07 (Brown and Vivas 2005) which increased the LDI values for these landscapes on O`ahu. This relatively high LDI coefficient may be too high for military lands in Hawai`i. In addition, zoning, like land cover, does not accurately reflect land-use. For example, there are large tracts of land zoned agricultural that are used for residential purposes. Land cover data, mapped from higher resolution Quickbird imagery (2.4m), are in the process of being developed for the Hawaiian Islands and will likely improve future land-use analysis at finer scales. In addition to higher spatial resolution, the new land-cover data will include a class for pastureland and open developed land (e.g. urban parks). Furthermore, the inclusion of detailed information on land-use as well as land-management, such as implementation of best management practices and the control of invasive species could also improve our understanding of the links between human land-use and the condition of wetland ecosystems.

Although a consistent data set providing high resolution, detailed land-use/land-cover data will likely improve our ability to characterize landscapes at both small and large spatial scales, results from this project indicate that landscape indicators, calculated from readily-available data layers, can provide useful metrics for estimating the impacts of human land-use to wetland ecosystems, particularly with regard to wetland water quality (TDN), plant tissue $\delta^{15}N$ concentrations, and soil parameters (SOM, TN, ExP, BD). For example LDI scores calculated for watershed basins were significantly correlated with TDN in wetland surface water ($r = 0.53$, $p < 0.01$), plant tissue $\delta^{15}N$ concentrations ($r = 0.71$, $p < 0.01$), soil BD ($r = 0.46$, $p < 0.05$), soil TN ($r = -0.47$, $p < 0.05$) and with CRAM condition assessment scores ($r = -0.64$, p <

0.01). Furthermore, GIS analysis of nationally available data sets permit the calculation of landscape metrics for large regions such as for the entire State or Nation. This type of analysis can provide managers with valuable information on how regional stressors may be affecting their efforts to restore wetland ecosystems.

CHAPTER IV: CONCLUSION

The U.S. EPA advises using a multi-scaled approach to evaluating the integrity of wetland ecosystems that includes intensive field surveys (Level I), rapid on-site assessments (Level II), and remote or landscape-scale assessments (Level III) (Faber-Langendoen et. al. 2008) (Figure 1.1). Similar types of data, with increasing levels of detail, are often collected as one proceeds from remote (Level I) assessments to site-specific field surveys (Level III). For instance, information on land-use stressors is typically collected in both rapid (Miller and Gunsalus 1997, SAIC 2004, Collins et. al. 2008) and remote assessments (Brown and Vivas 2005, Mack 2006), while information on vegetation (Miller and Wardrop 2006) and soils are collected during intensive field surveys (Bantilan-Smith et. al. 2009) as well as during rapid assessments (SAIC 2004, Faber-Langendoen et. al. 2008).

In developing assessment procedures, information collected at the three assessment levels should provide relatively consistent information (Faber-Langendoen et. al. 2008) and should therefore be correlated with one another (Sutula et. al. 2006, Faber-Langendoen et. al. 2008). Developing indicators of ecological condition usually involves iteratively testing various levels of assessments against more detailed field data (Sutula et. al. 2006). While wetland condition assessment procedures have been developed, validated, and calibrated in other regions, they have not yet been fully developed or tested for wetlands in the Pacific Islands. Furthermore, the relative capacity of Hawaiian coastal wetlands to perform specific functions such as nutrient retention, shoreline protection, and wildlife habitat support is currently unknown.

The purpose of this project was to assist in the development of indicators for assessing the condition of coastal wetlands in Hawai`i at multiple spatial scales. The project expanded on

previous research on Hawaiian wetland soils (Bantilan-Smith et. al. 2009) and water quality (Bruland and MacKenzie 2010) and evaluated Level I and Level II indicators in assessing the condition of 27 coastal wetlands in Hawai`i. Results revealed that of the three RAMs, CRAM (Collins et. al. 2008) showed the strongest correlations with soil parameters and plant $\delta^{15}N$ levels and was simpler and faster to perform than HHGM (SAIC 2004). This suggest that although CRAM was developed for condition assessments of wetlands along the Pacific West coastline, with modifications the method could be applicable to condition assessments of Hawaiian coastal wetlands.

Surprisingly, the HHGM metrics were often based on the original metrics presented in guidelines developed for other regions. For example, the metrics for assessing the functional capacity of depressional wetlands were nearly identical to those developed for prairie potholes in the Northern Rocky Mountains (Hauer et al. 2002). The HHGM scores were, nonetheless, significantly correlated with two soil parameters, ExP and TN, as well as with $\delta^{15}N$ plant levels. In addition, although WRAP was less quantitative and extremely rapid to perform, the final scores were only significantly correlated with one soil parameter, TN.

Interestingly, all three rapid assessment procedures (CRAM, WRAP, and HHGM) used distances of 100m or 250m from the wetland edge to analyze potential impacts from surrounding land use. However, results from this project suggest that land-use occurring farther from the wetland edge may be impacting the health of wetland ecosystems. The strongest correlations with soil and water parameters occurred between the LDI scores and road densities calculated for the 1,000m buffers and watersheds. On the other hand, the relationships between field data and landscape metrics calculated for the 100m buffers were weak or non-existent.

Thus condition assessments for Hawaiian wetlands should include indicators of land-use stress occurring within distances far from the wetland edge.

However, the land-use stressor metrics should be recorded on separate evaluation sheets, especially during the development phase of condition assessment protocols. The inclusion of land-use indicators in rapid assessment methods makes it difficult to evaluate links between human land use and ecological integrity. As demonstrated in the results from this project, once the land use indicators were excluded from the calculations of condition assessment scores, the RAM scores were not strongly correlated with LDI scores.

In addition, the results suggested that remote assessments of landscape condition, calculated from readily-available data layers, can provide useful metrics for estimating the impacts of human land-use to wetland ecosystems, particularly with regard to wetland water quality (TDN), plant tissue $\delta^{15}N$ levels, and soil parameters (SOM, TN, ExP, BD). Additionally, the results indicated that simple metrics such as road density were at least as good as LDIs in relation to wetland soil and water quality. Thus rather than collecting generalized information on land-use stressors during rapid assessments, these data could instead be remotely analyzed using GIS to calculate land-use/land-cover variables at multiple scales. Rapid methods could then focus on collecting more detailed, site specific land-use stressors such as the presence of ditches, berms, and drainage pipes that directly impact the integrity of wetland ecosystems; as well as mitigation and restoration efforts that control invasive plants and animals, increase native species habitat, and induce the natural flow of water.

Similarly, while it is tempting to incorporate human value attributes into rapid assessment methods, this technique should be avoided since human values are subject to change with time and culture. Furthermore, no assessment technique will likely be robust enough to

evaluate both functions and values (Brinson 1993). Instead, human value attributes should be recorded and analyzed using separate score sheets. In Hawai‘i these score sheets should include values such as opportunities for recreation and education as well as traditional Hawaiian cultural values, including the protection and promotion of fishponds and taro farms.

It is widely recognized that a wetland classification system helps distinguish natural variability within and among types in order to discern both subtle and major differences between wetlands with good integrity and poor integrity (Faber-Langendoen et. al. 2008). Therefore, it is imperative that Hawaiian RAMs be based on a classification system that recognizes the distinct types of wetlands in Hawai`i. For instance, anchialine pools/ponds are considered depressional wetlands by HGM and CRAM. In contrast, the classification system presented by Erickson and Puttock (2006) was developed specifically for the Hawaiian Islands. In addition to recognizing anchialine ponds as a distinct wetland class it also places brackish (groundwater fed) coastal flat wetlands in a separate class. Future research in the development of wetland indicators should consider using the Hawaiian classification system instead of those proposed by HGM or CRAM.

Furthermore, in developing indicators of wetland ecosystem health, it is important to clearly define “endpoints” for each class of wetland. While Nu‘u (Maui) and Kamilo Point (Hawai‘i Island) may provide the best examples of reference conditions for “natural” depressional wetlands, especially with regard to native species cover (Bantilan-Smith et al. 2009), it will be difficult or impossible to establish reference standards for unaltered “natural” tidal fringe and coastal riverine wetland ecosystems given that historic and current land uses have significantly impacted these wetlands along the Hawaiian coastline. In some cases we may want to define desired conditions in terms of specific points in time in Hawaiian history. For

example, functioning or intact Hawaiian fishponds and taro fields could provide reference standards for some wetlands.

More research on the complex ecological relationships between soils, hydroperiod, and nutrient cycling and how these parameters respond to land-use change is needed in Hawai`i. This type of research would facilitate the development of indicators that are sensitive to gradients in wetland condition in response to land-use changes. In addition, research on Hawaiian wetland ecosystem processes and habitat support would result in clearer guidelines for rating the condition of habitats (i.e. presence of rocks, coarse woody debris, snags, etc.), vegetation communities (i.e. percent plant cover and number of layers), soil quality (i.e. depth of O and A horizon), topographic complexity, and hydrology. The development of protocols for Level I and Level II assessments would also benefit from Level III indicators of biological assessments that include algae and invertebrates. In order to be comparable with other protocol development efforts in Hawai`i, future research should consider using techniques that are consistent with data collected during prior surveys such as: water quality (Nelson et al. 2007, Bruland and MacKenzie 2010), soil quality (Nelson et al. 2007, Bantilan-Smith et al. 2009), vegetation (Bantilan-Smith et al. 2009) as well as information on biotic integrity developed for estuarine wetlands by Nelson et al. (2007) and for riverine wetlands by Kido (2002).

Recommendations for Hawai`i RAM

The following observations were made while conducting the rapid condition assessments:

- All of the wetland sites were given an HHGM score of 0 for invasive / non-native animals based on the criteria of sighting more than six individuals during the assessment, making this variable incapable of discerning differences in the presence of specific non-native animals such as cats and dogs vs. the presence of non-native birds (i.e. Myna Birds, Cardinals, Pigeons, and Ducks). Therefore, this indicator lacks the resolution needed to provide useful information on how invasive animals may be impacting coastal wetlands in Hawai`i.
- The metrics recommended by Hauer (2002) for the assessment of Prairie pothole wetlands should be modified based on research conducted on Hawaiian coastal depressions, particularly for anchialine pools.
- In the methods for tidal fringe wetlands, the functional category "Opportunity for Access to the Wetland" should be changed to "Access to Tidally Connected Edge." Furthermore, the instructions are misleading and explicitly call out fishponds as examples of wetlands that lack nekton access. The tidal fringe wetlands in Hawai`i were given low ratings for nekton utilization primarily due to obstructions such as walls which prevent access into the wetland or because they lacked subtidal creeks, channels or ponds due to either channelized flow or the presence of soils with high infiltration rates (i.e. Paiko Lagoon, on O`ahu).
- In rating the water storage capacity of depressional wetlands, the HHGM methods only used the following non-land use related indicators: soil texture and precipitation, whereas the indicators for riverine wetlands included frequency of overbank flow,

microtopographic and macrotopographic complexity, percent vegetative cover, coarse woody debris, and entrenchment ratio in addition to geomorphic modification as the stressor indicator.

- Many of the wetlands received low scores for litter depth based on the Hawai`i HGM criteria. This may be due to the higher rates of decomposition in the tropics. Therefore, this metric should be rescaled in order to accurately evaluate nutrient cycling in tropical wetlands.
- The Hawai`i HGM is inconsistent in applying land use as an indicator for the functional capacity of wetlands. While land use is used as an indicator for almost all functions for depressional wetlands they are rarely used in rating riverine wetlands and are not used at all in rating tidal fringe wetlands.
- Hawai`i HGM includes canopy gaps as an indicator for spatial structure and biodiversity. Wetlands in Hawai`i rarely have a tree canopy, and those with Mangroves rarely have canopy gaps. Therefore, this metric does not appear to be a valuable indicator for Hawaiian coastal wetlands.
- The "Atlas of Natural Hazards In the Hawaiian Coastal Zone" (Fletcher et al. 2002) could be used in scoring the shoreline slope metric for tidal fringe wetlands.
- Landscape development indices and/or metrics such as regional road density could potentially be incorporated as metrics when assessing impacts from human land use.
- It may be worthwhile to include management activities, such as trapping of non-native animals, when rating a wetland's capacity to provide wildlife habitat.

- The Hawai`i RAM should explicitly list features such as boulders, algal mats, submerged vegetation, coarse woody debris, lava cracks, and snags that should be used when rating variables for habitat complexity.
- The current HHGM metrics for rating percent plant cover needs to be reevaluated, given that some native water-birds require un-vegetated mudflats for nesting and foraging.

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APPENDIX A. SCORE CARDS FOR HAWAI`I HGM CONDITION ASSESSMENTS

HAWAI`I HGM SCORE CARD
DEPRESSIONAL WETLAND
STORAGE OF SURFACE WATER
Functional Indicator
Dominant upland use between 25 and 100m of the wetland boundary
Dominant land use within 25m of the wetland boundary
Dominant land use within the wetland
Dominant soil texture in wetland
Estimated duration of flooding in a typical year
Opportunity Variable
Amount of precipitation in an area (see Appendix E)
SCORE (average)
STORAGE OF SUBSURFACE WATER AND MODERATION OF GROUNDWATER FLOW OR DISCHARGE
Functional Indicator
Infiltration rate of dominant soil in wetland
Area of inundation that dries out every year
Presence of surface salts
Maximum depth of annual inundation (i.e. hydraulic head)
Water levels controlled by human activities
Opportunity Variables
Inlet but no outlet
Amount of precipitation in an area
SCORE
RETENTION OF IMPORTED ELEMENTS, COMPOUNDS, AND PARTICULATES
Functional Indicator
Dominant upland use between 25 and 100m of the wetland boundary
Dominant land use within 25m of the wetland boundary
Dominant land use within the wetland
Detention time
Average water depth
Depth of the O and A soil horizons
Dominant soil texture

Rate of sediment delivery into wetland from adjacent uplands
Percent plant cover
RAW SCORE
Opportunity Variables
Watershed category
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
CYCLING OF NUTRIENTS
Functional Indicator
Dominant land use within the wetland
Dominant land use within 25m of the wetland boundary
Texture of C soil horizon
Depth of the O and A soil horizons
Rate of sediment delivery into wetland from adjacent uplands
Percent plant cover
Percent non-native invasive plant species cover
Organic matter comosition
Amount of precipitation in area
Estimated duration of flooding in a typical year
SUM RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
EXPORT OF ORGANIC CARBON
Dominant land use within the wetland
Dominant land use within 25m of the wetland boundary
Percent plant cover
Detritus / Litter depth
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
MAINTAIN HABITAT INTERSPESION AND CONNECTIVITY
Functional Indicator
Dominant upland use between 25 and 100m of the wetland boundary

Dominant land use within 25m of the wetland boundary
Dominant land use within the wetland
Distance to nearest wetland of the same HGM class
Density of all freshwater wetlands within 800m of wetland
Vegetation cover between habitat types (e.g. wetland and upland)
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
MAINTAIN CHARACTERISTIC PLANT COMMUNITY
Functional Indicator
Percent plant cover
Percent cover of native facultative wetland and obligate plant species
Ratio of native to non-native wetland plant species
Degree of native species regeneration
RAW SCORE (average)
Opportunity Variable
Dominant land use within the wetland
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
MAINTAIN CHARACTERISTIC INVERTEBRATE FOOD WEBS
Functional Indicator
Dominant land use within the wetland
Dominant land use within 25m of the wetland boundary
Depth of the O and A soil horizon
Percent plant cover
Number of aquatic and terrestrial habitat types
RAW SCORE (average)
Opportunity Variable
Watershed category
SUM
SCORE (raw score + modifier)
MAINTAIN CHARACTERISTIC VERTEBRATE HABITATS
Functional Indicator
Raw score from maintain interspersion and connectivity function

Raw score from maintain characteristic plant community function
Raw score from maintain characteristic invertebrate food webs function
Step A. Calculate average score for Nos. 1-3 and multiply by (0.75)
Dominant land use in wetland
Groundcover microhabitat
Number of vertical strata
Number of aquatic and terrestrial habitat types
Percent open water
Use by non-native animal species
Wetland edge complexity
Step B. Calculate average score for Nos. 4-10 and opportunity variable and multiply by (0.25)
RAW SCORE (average)
Opportunity Variable
Dominant upland use between 25 and 100m of the wetland boundary
Modifier
Percent non-native invasive plant species cover
SCORE (raw score + modifier)
MAINTAIN HABITAT FOR THREATENED AND ENDANGERED SPECIES
Functional Indicator
Raw score from maintain interspersion and connectivity
Raw score from maintain characteristic plant community function
Raw score from maintain characteristic vertebrate habitats function
Step A. Calulate average score for Nos. 1-3 and multiply by (0.50)
Occurrence of a threatened or endangered species
Occurrence of designated critical habitat for threatened or endangered species
Use by non-native animal species
Percent cover of non-native invasive plant species
Step B. Calulate average score for Nos. 4-7 and multiply by (0.50)
SCORE (sum of A and B)
Total Funcational Score and Functional Capacity
Wetland Function
Storage of surface water
Storage of subsurface water and moderation of groundwater flow or discharge
Retention of imported elements, compounds, and particulates
Cycling of nutrients
Export of organic carbon

Maintain habitat interspersion and connectivity
Maintain charactersistic plant community
Maintain characteristic invertebrate food webs
Maintain characteristic vertebrate habitats
Maintain habitat for Threatened and Endangered species
Average
TOTAL FUNCTIONAL SCORE (sum)
Percent of Potential Total/120

Hawai`i HGM Score Card
Tidal Wetlands
DISSIPATION OF ENERGY
Functional Indicator
Shoreline slope
Exposure (i.e. fetch)
Wetland width
Microtopographic complexity
Vegetation roughness characteristics
RAW SCORE (average)
Opportunity Variables
Dominant land use within 100m of wetland boundary
Raw Score
Modifier
Percent non-native invasive plant species
FINAL SCORE
RETENTION OF IMPORTED ELEMENTS COMPUNDS, AND PARTICULATES
Functional Indicator
Hydrologic regime
Microtopographic complexity
Vegetation surface roughness characteristics
Proximity to source channel
RAW SCORE (average)
Opportunity Variables
Watershed category
Raw Score
Modifier
Percent non-native invasive plant species

FINAL SCORE
EXPORT OF ORGANIC CARBON
Functional Indicator
Hydrologic regime
Percent plant cover
Mean plant height index
Raw Score (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE
MAINTAIN CHARACTERISTIC PLANT COMMUNITY
Functional Indicator
Percent plant cover
Percent cover of native facultative wetland and obligate plant species
Ratio of native to non-native wetland plant species
Degree of native species regeneration
Raw Score (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE
MAINTAIN CHARACTERISTIC INVERTEBRATE FOOD WEBS
Functional Indicator
Hydrologic regime
Length of aquatic edge
Percent plant cover (invertebrates)
Raw Score (average)
Opportunity Variable
Watershed category
Raw Score (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE (raw score + modifier)
MAINTAIN CHARACTERISTIC VERTEBRATE HABITATS
Functional Indicator
Raw score from maintain characteristic plant community function
Raw score from maintain characteristic invertebrate food webs function

Step A: Calculate average score for Nos. 1&2 and multiply by (0.50)
Level of human disturbance in and around wetland
Wetland size
Number of aquatic and terrestrial habitat types
Degree of corridor connectivity
Use by non-native animal species
Length of aquatic edge
Step B: Calculate average score for Nos. 3-8 and multiply by (0.50)
Raw Score (A+B)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE
MAINTAIN HABITAT FOR THREATENED AND ENDANGERED SPECIES
Functional Indicator
Raw score from maintain interspersion and connectivity function
Raw score from maintain characteristic plant community function
Raw score from maintain vertebrate habitat function
Step A: Calculate average score for Nos. 1-3 and multiply by (0.50)
Occurrence of a threatened or endangered species
Occurrence of designated critical habitat for a threatened or endangered species
Use by non-native animal species
Percent cover of non-native invasive plant species
Step B: Calculate average score for Nos. 4-7 and multiply fy (0.50)
FINAL SCORE (A+B)
NEKTON UTILIZATION
Functional Indicator
Hydrologic regime
Length of aquatic edge
Opportunity for access to the wetland
Number of aquatic habitat types
Opportunity Variables
Level of human use along the aquatic edge
FINAL SCORE (average)
TOTAL FUNCTIONAL SCORE AND FUNCTIONAL CAPACITY
Functional Indicator
Dissipation of Energy
Retention of imported elements, compunds, and particulates

Export of organic chemicals
Maintain interspersion and connectivity
Maintain characteristic plant community
Maintain characteristic invertebrate community
Maintain characteristic vertebrate habitat
Maintain habitat for threatened and endangered species
Nekton utilization
FINAL (average)
TOTAL FUNCTIONAL SCORE
Percent of Potential Total/90

Hawai`i HGM Score Card
Tidal Wetlands
STORAGE OF SURFACE WATER
Functional Indicator
Frequency of overbank flow
Microtopographic complexity
Macrotopographic relief
Percent herbaceous cover
Percent shrub/sapling cover
Tree density
Amount of coarse woody debris
Geomorphic modification
Opportunity Variable
Entrenchment ratio
Sum
Raw Score (average)
Modifier
Percent cover by non-native invasive plant species
Final Score (raw + modifier)
STORAGE OF SUBSURFACE WATER AND MODERATION OF GROUNDWAER FLOW OR DISCHARGE
Functional Indicator
Frequency of overbank flow
Infiltration rate and storage capacity of dominant soil in wetland
Macrotopographic relief

Geomorphic modification
Opportunity Variables
Perrennial or Intermittant
Entrenchment ratio
Average Opportunity
FINAL SCORE (average)
DISSIPATION OF ENERGY
Functional Indicator
Microtopographic complexity
Macrotopographic relief
Percent shrub/sapling cover
Tree density
Amount of coarse woody debris
Entrenchment ratio
Frequency of overbank flow
Geomorphic modification
RAW SCORE (average)
Opportunity Variables
Relative position of the wetland to floodable properties
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE (raw + modifier)
RETENTION OF IMPORTED ELEMENTS, COMPOUNDS, AND PARTICLES
Functional Indicator
Frequency of overbank flow
Dominant land use in the wetland
Geomorphic modification
Microtopgoraphic complexity
Percent herbaceous cover
Percent shrub/sapling cover
Tree density
Amount of soil texture in wetland (i.e. soil sorptive properties)
Hydric soil indicators (upper 30 cm)
Percent vegetative litter cover (survaces for microbial activity)
RAW SCORE (average)
W/O Land Use

Opportunity Variables
Watershed category
SUM
RAW SCORE (average)
Modifier
Percent non-native invasive plant species
FINAL SCORE (raw + modifier)
CYCLING OF NUTRIENTS
Functional Indicator
Percent herbaceous cover
Percent shrub/sapling cover
Tree density
Percent vegetative litter cover
A soil horizon biomass
Opportunity Variable
Entrenchment ratio
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE (raw score + modifier)
Export of Organic Carbon
Functional Indicators
Frequency of overbank flow
Macrotopographic relief
Percent vegetative litter cover
Percent plant cover
Opportunity Variable
Entrenchment ratio
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
FINAL Score (raw score + modifier)
SPATIAL STRUCTURE AND BIODIVERSITY OF HABITATS
Functional Indicator
Snag Occurrence
Number of vertical strata

Percent herbaceous cover
Percent shrub/sapling cover
Tree density
Patchiness of habitat
Canopy gaps
RAW SCORE (average)
Opportunity Variable
Dominant land use between 25 and 100 m of the wetland boundary
RAW SCORE
Modifier
Percent non-native invasive plant species cover
FINAL SCORE
HABITAT INTERSPERSION AND CONNECTIVITY
Functional Indicator
Dominant land use between 25 and 100 m of the wetland boundary
Dominant land use within 25 m of the wetland boundary
Dominant land use within the wetland
Distance to nearest wetland of the same HGM class
Density of all freshwater wetlands with 800 m of wetland
Vegetation cover between habitat types (e.g. wetland and upland)
Frequency of overbank flow
Entrenchment ratio
Geomorphic modification
RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE (raw + modifier)
MAINTAIN CHARACTERISTIC PLANT COMMUNITY
Functional Indicator
Percent plant cover
Percent cover of native facultative wetland and obligate plant species
Ratio of native to non-native wetland plant species
Degree of native species regeneration
RAW SCORE (average)
Opportunity variable
Dominant land use within the wetland

RAW SCORE (average)
Modifier
Percent non-native invasive plant species cover
FINAL SCORE
MAINTAIN CHARACTERISTIC INVERTEBRATE FOOD WEBS
Functional Indicator
Dominant land use within the wetland
Dominant land use within 25 m of wetland boundary
Depth of the O and A soil horizon
Percent plant cover
Number of aquatic and terrestrial habitat types
Percent embeddedness of riffle/run channel substrate
RAW SCORE (average)
Opportunity Variable
Watershed category
SUM
RAW SCORE (average)
Modifier
Percent non-native plant species cover
FINAL SCORE (raw + modifier)
MAINTAIN CHARACTERISTIC VERTEBRATE HABITATS
Functional Indicator
Raw score from spatial structure and biodiversity of habitats
Raw score from maintain interspersion and connectivity function
Raw score from maintain characteristic plant community function
Raw score from maintain characteristic invertebrate food webs function
Step A. Calculate average score for Nos. 1-4 and multiply by (0.5)
Dominant land use in wetland
Groundcover microhabitat
Number of vertical strata
Number of aquatic and terrestrial habitat types
Macrotopographic relief
Percent open water
Use by non-native animal species
Wetland edge complexity
Step B. Calculate average score fo Nos. 5-12 and opportunity variable and multiply by (0.50)

RAW SCORE (sum of Steps A and B)
Opportunity Variable
Dominant upland use between 25 and 100 m of wetland
Modifier
Percent non-native invasive plant species cover
FINAL SCORE (raw + modifier)
MAINTAIN HABITAT FOR THREATENED AND ENDANGERED SPECIES
Functional Indicator
Raw score from maintain interspersion and connectivity function
Raw score from maintain characteristic plant community function
Raw score from maintain characteristic vertebrate habitat function
Step A. Calculate average score for Nos. 1-3 and multiply by (0.5)
Occurrence of a threatened or endangered species
Occurrence of designated critical habitat for a threatened or endangered species
Use by non-native animal species
Percent cover of non-native invasive plant species
Step B. Calculate average score fo Nos. 4-7 and opportunity variable and multiply by (0.50)
FINAL SCORE (sum of steps A and B)
Total Funcational Score and Functional Capacity
Wetland Function
Storage of surface water
Storage of subsurface water and moderation of groundwater flow or discharge
Dissipation of energy
Retention of imported elements, compounds, and particulates
Cycling of nutrients
Export of organic carbon
Maintain spatial struture and biodiversity of habitats
Maintain habitat interspersion and connectivity
Maintain charactersistic plant community
Maintain characteristic invertebrate food webs
Maintain characteristic vertebrate habitats
Maintain habitat for Threatened and Endangered species
Average
TOTAL FUNCTIONAL SCORE (sum)
Percent of Potential Total/120

APPENDIX B: CRAM SCORE CARD

CRAM ASSESSMENT
Landscape Connectivity
Buffer submetric A: Percent of AA with Buffer
Buffer submetric B: Average Buffer Width
Buffer submetric C: Buffer Condition
Hydrology
Water Source
Hydroperiod or Channel Stability
Hydrologic Connectivity
Physical Structure
Structural Patch Richness
Topographic Complexity
Biotic Structure
Plant Community submetric A: Number of Plant Layers
Plant Community submetric B: Number of Co-dominant species
Plant Community submetric C: Percent Invasion
Plant Community Metric (average of sumbetrics A-C)
Horizontal Interspersion and Zonation
Vertical Biotic Structure