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A COMPARISON OF THE VEGETATION AND SOILS OF NATURAL, RESTORED, AND CREATED COASTAL LOWLAND WETLANDS IN HAWAI'I

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Abstract: The loss of coastal wetlands throughout the Hawaiian Islands has increased the numbers of created (CW) and restored (RW) wetlands. An assessment of these wetlands has yet to occur, and it has not been determined whether CWs and RWs provide the same functions as natural wetlands (NWs). To address these concerns, vegetation and soil characteristics of 35 wetlands were compared within sites along hydrologic gradients and among sites with different surface water salinity and status (i.e., CW, RW, NW). Only 16 of 85 plant species identified were native and three of the four most abundant species were exotic. Vegetative characteristics differed primarily across salinity classes, then along hydrologic zones, and to a lesser extent among CWs, RWs, and NWs. Soil properties exhibited fewer differences across salinity classes and along hydrologic zones and greater differences among CWs, RWs, and NWs. The dominant presence of invasive species in coastal Hawaiian wetlands suggests that it will be difficult to locate reference sites that can be used as restoration targets. Differences in edaphic characteristics suggested that RWs/CWs do not exhibit the same functions as NWs. Future restoration and creation should include planting of native vegetation, controlling invasive vegetation, and alleviating inadequate soil conditions.

Key Words: creation, hydrologic gradient, invasive species, mitigation, restoration

INTRODUCTION

The Hawaiian archipelago is one of the most isolated island groups in the world. As such, coastal lowland wetlands (CLWs) in Hawai'i support unique assemblages of flora and fauna (Cuiddihy and Stone 1990). Hawaiian CLWs play an essential role in maintaining water quality in near shore environments and protecting seagrass beds and coral reefs from sediments, nutrients, and pulses of freshwater during heavy rains (Stedman and Hanson 2007, Bruland 2008). Unfortunately, Hawai'i's CLWs have been extensively altered for agriculture, aquaculture, grazing, and urban development (Rauzon and Drigot 2002). As a result, it is estimated that Hawai'i has lost approximately 31% of its CLWs due to human activities (Dahl 1990, Kosaka 1990). The most recent survey estimated that only 9,095 ha of CLW habitat remain in the state (Dahl 1990).

Increased scientific knowledge of the functional importance of wetland ecosystems has led to changes in federal policies aimed at protecting wetlands. These policies include Section 404 of the Clean Water Act and the "no net loss" policy. The "no net loss" policy sought to replace lost wetland habitat with new habitat by restoring and/or creating wetlands, and is now the cornerstone of wetland conservation in the U.S. (Mitsch and Gosselink 2007). As a result, numerous federal and state agencies, non-governmental organizations, and private landowners are engaged in wetland restoration and creation across the continental U.S. and in Hawai'i.

Despite the increase in restoration and creation projects, the ability of created (CWs) and restored wetlands (RWs) to functionally replace natural wetlands (NWs) has become a topic of considerable debate (Zedler 1996, Zedler and Callaway 1999, Kentula 2000). Consequently, numerous methods exist to evaluate the success of wetland mitigation projects. Today many researchers use NWs as reference sites to assess success of CWs and RWs (Bishel-Machung et al. 1996, Moore et al. 1999, Balcombe et al. 2005). The use of reference wetlands is based on the underlying assumptions that intact NWs exhibit high ecological function, and wetlands sharing similar hydrologic characteristics, vegetation communities, and soil properties will function similarly (Brinson and Rheinhardt 1996, Stolt et al. 2000, Zampella and Laidig 2003). Kentula (2000) recommended comparing identical ecological parameters among populations of CWs, RWs, and NWs within a region to extrapolate results beyond site-specific studies. Two of these parameters are vegetation and soil characteristics, which are relatively easy to sample and are involved in complex interactions that contribute to wetland function (Craft et al. 1988, Shaffer and Ernst 1999, Bruland and Richardson 2006).

Vegetation of CWs and RWs often differs from NWs in species richness (SR), total cover, and species composition. These differences are typically attributed to the young age of the restored or created sites, and differences in landscape position and hydrology (Heaven et al. 2003). Creation or restoration of wetlands usually involves the use of heavy machinery to scrape the surface and excavate the topsoil layer (Bruland and Richardson 2005), including the vegetation and seed bank. Newly created or restored sites tend to support a suite of species that are adapted to disturbed environments (Balcombe et al. 2005). With time the total cover and species composition of CWs and RWs are expected to approach that of NWs (Reinartz and Warne 1993, Balcombe et al. 2005). The presence of exotic invasive vegetation in RWs and CWs can degrade habitat quality, decrease biodiversity, and impact hydrologic and nutrient cycling functions (Zedler and Kercher 2004).

Soil properties of CWs and RWs have also been shown to differ from those of NWs (Bishel-Machung et al. 1996, Bruland et al. 2003, Bruland and Richardson 2005). This is problematic as soils are the physical foundation for every wetland ecosystem and both plants and animals depend on wetland soils for growth and survival (Stolt et al. 2000, Bruland and Richardson 2004). These edaphic differences are the result of a variety of factors. First, the removal of topsoil during creation or restoration of a wetland results in disturbance of soils (Shaffer and Ernst 1999) and can lower concentrations of soil organic matter (SOM). Differences in SOM can significantly affect many other soil properties, such as total-percent nitrogen, bulk density (BD), and pH (Bishel-Machung et al. 1996). Second, the use of heavy machinery results in soil compaction, increasing BD in CWs and RWs. Finally, differences in hydrology between CWs, RWs, and NWs can also affect soil properties (Craft et al. 2002). When the hydroperiod is lengthened, anaerobic conditions slow decomposition rates and allow for organic matter to accumulate in the soil, decreasing BD and pH (Craft et al. 2002).

Despite the large number of CW and RW projects that are currently underway in the state of Hawai'i, it remains unclear how these ecosystems function compared to NWs. Furthermore, no studies have attempted to conduct a comprehensive assessment of the vegetative and edaphic attributes of coastal lowlands wetlands in Hawai'i. An evaluation of these attributes will further our knowledge of Hawaiian wetland ecosystem function and provide recommendations that are applicable to wetland creation and restoration throughout the state. Thus, the objectives of this study were to: 1) document the vegetative and edaphic attributes of coastal lowland Hawaiian wetlands, and 2) compare these attributes within sites across hydrologic gradients, and among sites of different salinity classes and wetland status (i.e., CWs, RWs, and NWs).

METHODS

Study Sites and Sampling Design

In order to restrict our focus to CLWs on the main Hawaiian Islands, we used the following criteria: 1) sites were located between 0–100 m in elevation; 2) sites were located on one of the five major Hawaiian Islands (Hawai'i, Kaua'i, Maui, Moloka'i, and O'ahu); and 3) sites were available for sampling. The elevation criterion effectively excluded all mountain bogs, which comprise a significant component of Hawai'i's wetland area but are considerably different in terms of structure and function from CLWs. As a number of the CLW sites

Table 1. Distribution of natural (NWs), restored (RWs), and created wetlands (CWs) across 4 salinity classes.

	Freshwater (< 0.5 ppt)	Brackish (0.5–30 ppt)	Euhaline (30–40 ppt)	Hyperhaline (> 40 ppt)
NWs	5	10	1	1
RWs	2	6	1	2
CWs	1	4	1	1

are located on private property or military land with restricted access, a random sampling of all CLWs was impossible. Efforts were made to achieve balance between isolated and riparian/estuarine sites; across freshwater, brackish, and euhaline sites; and among natural, restored, and created sites. In using 35 wetlands, we are confident that our sites represented the overall population of CLWs in Hawai'i. In total, 17 NWs, 11 RWs, and 7 CWs were sampled in March and April 2007 (Table 1). Site ownership and management varied from federal (i.e., Fish and Wildlife Refuges), state (Department of Land and Natural Resources), county (Hawai'i County), non-governmental organizations (Maui Coastal Land Trust, etc.), and private lands.

Wetlands sampled were classified as either hydrologically isolated from surface water bodies or hydrologically connected to stream or tidal creeks, and spanned a range of salinities (0.4–120‰). For isolated wetlands, two transects radiated outward from wetter zones towards drier zones in random directions. For riparian/estuarine wetlands, transects ran perpendicular to the stream or tidal creek. Each transect was stratified into three zones representing the wetter, intermediate, and drier sections of the site. The three zones were identified using ecological indicators including vegetation, hydrology, and elevation. Wetter zones were characterized by obligate wetland plant species and water tables that were generally at or near the soil surface. Intermediate zones were characterized by a mixture of obligate and facultative wetland species and water tables generally near the soil surface. Drier zones were characterized by a mixture of facultative and upland plant species and water tables that were generally below the soil surface. Surface water grab samples were collected from each wet zone and salinity was measured using a handheld YSI Inc. Model 556 Multi probe system (Yellow Springs, Ohio). Due to limited information regarding surface water salinities of the 35 sites, we classified them after sampling into 4 salinity classes: freshwater (< 0.5‰), brackish (0.5–30‰), euhaline (30–40‰), and hyperhaline (> 40‰) (Cowardin et al. 1979) based on our surface water measurements. Thus, the sites represent a conceptual gradient from fresh to hyperhaline salinity but they do not represent a geographic gradient in any particular watershed or estuary. The distribution of NWs, RWs, and CWs within each of the salinity categories is shown in Table 1. As the distribution of sites varied across the salinity and status classes, we used a general linear model (GLM) analysis of variance (ANOVA) to quantify the effects of hydrologic zone, salinity class, and wetland status on the vegetative and soil characteristics as this model is robust with unbalanced sampling designs.

Vegetation Sampling

Vegetative cover was examined by randomly placing a 1-m^2 quadrat in each of the three hydrologic zones of the two transects established at each site ($n = 6$ quadrats per site). Within these quadrats, each plant species present was systematically identified to the species level using species keys and plant identification guides (Whistler 1994, Erickson and Puttock 2006). For vegetation that could not be identified in the field, voucher specimens were collected and identified at the Bishop Museum Herbarium (Honolulu, HI). Visual estimates of % cover were made by a minimum of 2 observers for each unique species, exposed substrate, standing water, and litter. Estimates were averaged among observers. Plant species were then determined to be native or exotic (Erickson and Puttock 2006, Star and Star 2007) and categorized as obligate wetland (OBL), facultative wetland (FACW), facultative (FAC), facultative upland (FACU), or upland plants (UPL) (Environmental Laboratory 1987, USDA NRCS 2008).

Species richness per site was determined by calculating the average SR among the six 1-m^2 sample quadrats. Likewise, SR per zone was calculated by averaging the transect data across hydrologic zones within each site (Kent and Coker 1992). For each quadrat, % total cover was determined by visually summing the cover of all plant species, % exotic cover by summing the cover of exotic species, and % total wetland vegetation cover by summing the cover of species from each of the three wetland plant categories (e.g., OBL, FACW, and FAC). Mean values were then determined for each of the hydrological zones within each wetland as well as for each wetland salinity class and type.

Soil Sampling and Laboratory Analysis

Once vegetation was recorded, soil cores were collected in the center of each randomly placed 1-m^2 quadrat ($n = 6$ samples per site, $n = 208$ total samples). Cores were collected from the upper 0–

20 cm of the soil profile using a stainless-steel piston corer with a circular plastic sleeve insert of 4.8 cm diameter. Samples were capped and stored on ice until laboratory analysis was conducted at the University of Hawai'i Mānoa, Soil and Water Conservation Laboratory.

Soil cores were removed from their plastic sleeves and sliced in half vertically with a sharp knife. Half of the core was oven dried at 105°C for 24 hours to determine moisture and BD (Bruland and Richardson 2004, Wilke 2005). Dried soil was passed through a 2-mm sieve to remove rock fragments and large organic debris. Percent soil organic matter content (SOM) was measured by loss on ignition at 450°C for 4 hours (Bruland and Richardson 2006). Soil texture (% sand, % silt, % clay) was only determined for soils with SOM content $< 30\%$ using the pipette method (Tan 1996). Soils with SOM $> 30\%$ were considered organic.

Representative 20 g soil sub-samples of the 208 dried cores were analyzed at the University of Hawai'i Agricultural Diagnostic Service Center. Sub-samples were analyzed for pH using a Beckman pH meter and electrical conductivity (EC) using a conductivity bridge (Hue et al. 2000). Extractable phosphorus (ExP) was measured using the Olsen (1982) extraction method and total carbon (TC) and total nitrogen (TN) were measured using a LECO CN2000^{TM} combustion gas analyzer (AOAC International 1997).

Statistical Analyses

Vegetation. One-way GLM ANOVAs were used to evaluate the effects of wetland type (isolated versus connected) and salinity class on the vegetative characteristics with Minitab Statistical Software (Windows Version 15, Minitab Inc., State College, PA) and SAS (Windows Version 9.1, SAS Institute, Cary, NC). Natural wetlands, RWs, and CWs were compared by calculating mean SR, % cover of exotic species, % cover of wetland plant species (OBL, FACW, FAC), and total cover for each hydrologic zone at each site. Two-way GLM ANOVAs were conducted with the vegetative characteristics (except mean SR) to evaluate the effects of hydrologic zone (wetter, intermediate, drier), wetland status (NW, RW, CW), and their interaction. A 1-way GLM ANOVA was conducted to evaluate the effects of hydrologic zone and wetland status on mean SR per site (averaged across hydrologic zones within each site). Assumptions of normality were evaluated and Levene's Test was used for homogeneity of variances. Post-hoc analyses were conducted using the Tukey test.

To examine and identify the similarities and/or differences in vegetative composition as well as to summarize the vegetative data, a Principle Components Analysis (PCA) was conducted using PC-ORD (MJM Software, Gleneden Beach, OR). The species cover data was placed into a species*quadrat matrix, then arcsine-square root transformed. To rescale the data to range between 0 and 1, it was multiplied by $2/\pi$. As recommended by McCune and Grace (2002), all rare species (species that occurred in $< 5\%$ of the sample units) were removed from the analysis. Points close together in ordination space represent quadrats with similar species composition, whereas points farther apart represent quadrats sharing few species.

Soils. Independent factors for 1-way GLM ANOVAs included wetland type and salinity class. Two-way GLM ANOVA was used to evaluate the effects of hydrologic zone, wetland status, and their interaction. Response variables included moisture, BD, SOM, pH, EC, TC, TN, ExP, clay, silt, and sand. Transect data were averaged across hydrologic zones within each site prior to analysis. Soil organic matter, TN, EC, and TC and ExP did not meet the assumptions of normality; the former 2 parameters were square-root transformed while EC, TC and ExP were log-transformed. Levene's Test was used to test for homogeneity of variances. Relationships among the 11 soil properties were examined using a Spearman's correlation analysis in SAS.

To examine the similarities and/or differences in soil data between CWs, RWs, and NWs, a PCA was conducted using PC-ORD. Soil data were placed into a properties*soil core matrix. The PCA ordination was run with all soil properties, except silt content (which was not independent of sand and clay), and for only those cores for which SOM $< 30\%$. For all analyses in this study, $\alpha = 0.05$.

RESULTS

Vegetation

Eighty-five plants were positively identified to species across the 35 wetland sites. Ten plants could not be identified beyond the family level, of which seven were Poaceae, two were Convolvulaceae, and one was Chenopodiaceae. Of the 85 identified species, only 16 (19%) were native to Hawai'i. The most frequently observed species were *Urochloa mutica* (California grass), *Batis maritima* (pickelweed), *Paspalum vaginatum* (seashore paspalum), and *Bacopa monnieri* (water hyssop); *U. mutica*, *B. maritima*, and *P. vaginatum* are exotic and considered highly-invasive (Erickson and Puttock 2006),

Table 2. Mean values (± 1 standard error in parenthesis) for the vegetative characteristics that exhibited significant differences among the salinity class, hydrologic zone, and wetland status categories (ANOVA, $p < 0.05$). Mean values with different letters are significantly different ($p < 0.05$) according to the post-hoc least squares means test. Characteristics that did not exhibit significant differences ($p > 0.05$) across these categories included total wetland vegetation and zone or status, exotic cover and salinity or status, and species richness and status. There were no significant zone*status interactions for any of the vegetative characteristics (zone*status interaction was not tested for species richness).

Salinity Class†				
Vegetative Characteristics	Fresh (n = 8)	Brackish (n = 20)	Euhaline (n = 3)	Hyperhaline (n = 4)
Total Wetland Veg. (%)	68.6 (5.0) b	68.4 (4.9) b	62.8 (19.1) ab	32.6 (7.9) a
Total Cover (%)	76.0 (6.3) b	76.6 (3.6) b	63.6 (7.5) ab	33.5 (7.5) a
Species Richness‡	3.2 (0.31) c	2.2 (0.18) b	0.94 (0.1) a	1.2 (0.20) ab

Hydrologic Zone††			
Vegetative Characteristics	Wetter (n = 35)	Intermediate (n = 35)	Drier (n = 35)
Exotic Cover (%)	40.5 (6.6) a	52.4 (5.8) ab	68.1 (5.0) b
Total Cover (%)	56.2 (6.2) a	75.1 (5.1) b	79.9 (3.5) b
Species Richness‡§	1.5 (0.2) a	2.3 (0.3) b	2.9 (0.2) b

Wetland Status††			
Vegetative Characteristics	Natural Wetlands (n = 17)	Restored Wetlands (n = 11)	Created Wetlands (n = 7)
Total Cover (%)	76.2 (3.8) b	59.1 (6.2) a	74.1 (6.7) b

† Sample size = 105 and error degrees of freedom (df) = 31 for salinity ANOVAs.
‡ Species richness was measured as the mean number of species per 1-m^2 quadrat.
†† Sample size = 105 and error df = 100 for zone and status ANOVAs.
§ Sample size = 105 and error df = 31 for species richness among zone ANOVAs.

and *B. monneri* is native. Thirteen species were OBL, 11 were FACW, 17 were FAC, 23 were FACU, three were UPL, and the remaining 18 were undesignated (Erickson and Puttock 2006, USDA NRCS 2008).

Wetland type (isolated versus connected) did not account for a significant proportion of the variance in any of the measured vegetation parameters. Salinity, however, accounted for a significant proportion of the variability in total wetland vegetation cover, total cover, and SR (Table 2). Freshwater and brackish sites had greater total wetland vegetation and total cover than hyperhaline sites (Table 2). Mean SR was also greater in freshwater sites than in the other three salinity classes, while SR in brackish sites was greater than in euhaline sites (Table 2).

Total wetland vegetation cover was similar across hydrologic zones (Table 2). Exotic cover differed across hydrologic zones with lower values in drier than in wetter zones (Table 2). Total vegetative cover was greater in both intermediate and drier zones than in wetter zones. The number of species increased from wetter to drier zones, with 28 species in the wetter zone, 47 species in the intermediate zone, and 63 species in the drier zone. Mean SR was lower in wetter zones than in intermediate or drier zones (Table 2).

Neither total wetland vegetation cover nor exotic cover differed among wetland status categories (Table 2). Total vegetative cover, however, was greater in NWs and CWs than in RWs (Tables 2). Natural wetlands had the most species observed with 59, of which 28 (6 native) were only found in NWs; 49 species were observed in RWs and 31were observed in CWs. Overall, 46 exotic species were found in NWs, 28 in RWs, and 13 in CWs. However, mean SR among CWs, RWs, and NWs was similar (Table 2).

In PCA ordination (Figure 1), axes 1 and 2 accounted for 14% (Eignevalue = 1.45) and 13% (1.37) of total variance, respectively. Species with relatively strong loadings on axis 1 were *U. mutica* (Eigenvector = 0.50), *P. vaginatum* (−0.48), and *B. maritimus* (−0.41). *Batis maritima* (−0.70) had a strong negative loading on axis 2, and *Sesuvium portulacastrum* (sea purslane) (0.67) and *Pluchea indica* (Indian fleabane) (0.49) had strong positive loadings on axis 3. Quadrats from freshwater sites grouped together at the high end of axis 1 and axis 2, indicating greater cover of *U. mutica*. Additionally, quadrats from brackish sites tended to group together at the low end of axis 1 indicating high % cover of *B. maritimus*, *P. vaginatum*, and *B. monnieri*. Euhaline and hyperhaline quadrats were dominated by invasive *B. maritima*.

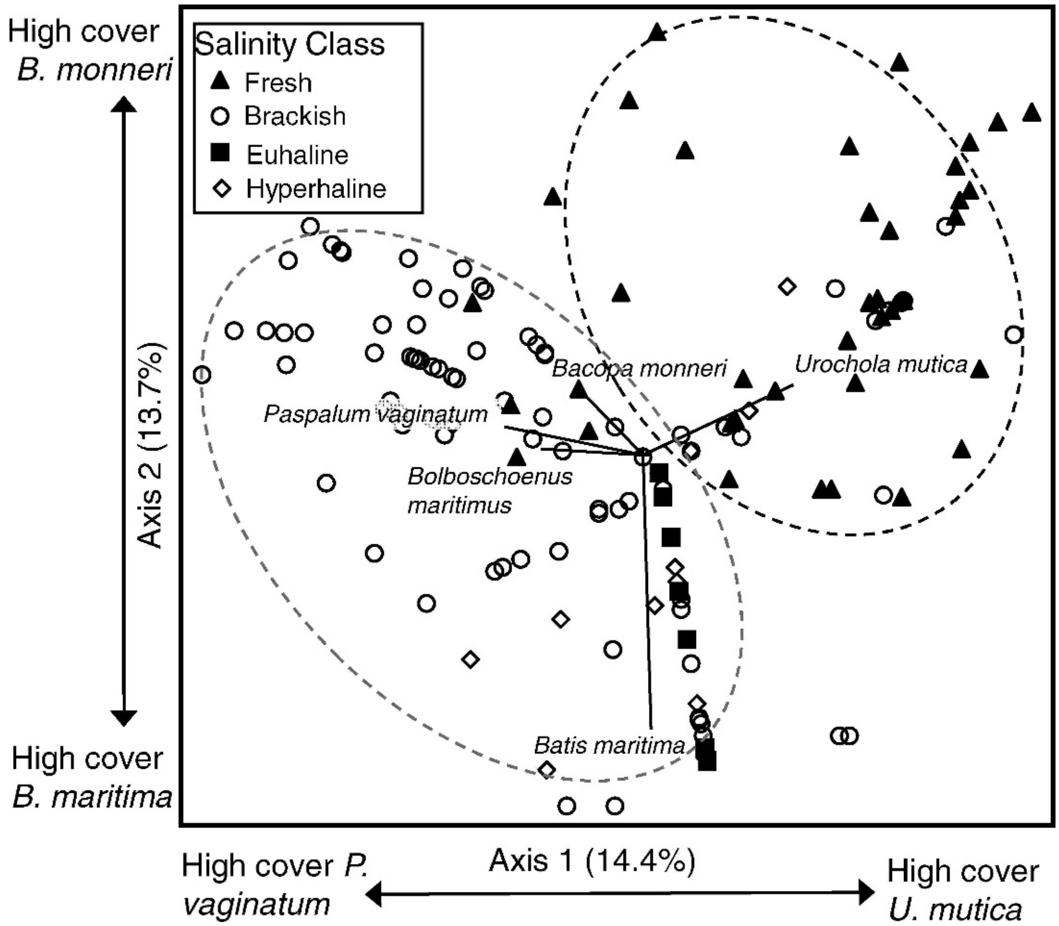

Figure 1. Principle components analysis (PCA) of the species composition of coastal lowland Hawaiian wetlands. Vectors represent the magnitude of correlation with axis 1 or 2. Axes 1, 2, and 3 accounted for 14%, 14%, and 12% of the total variance, respectively.

Soil Properties

Soil properties were similar in isolated and connected sites, and EC, as expected, differed among salinity classes (Table 3). Moisture, BD, and EC differed across the hydrologic zones (Table 3). Moisture was greater in the wetter and intermediate zones than in the drier zones, BD was greater in the drier zones than the intermediate and wetter zones, and EC was greater in the wetter than the intermediate and drier zones (Table 3). Bulk density was the only property to exhibit differences across both hydrologic zones and wetland status categories. It was also higher in RWs and CWs than in NWs. Neither hydrologic zone nor wetland status accounted for a significant proportion of the variance in ExP (Table 3).

While SOM, pH, TC, TN, and particle size distribution (percent sand, silt, clay) did not differ across hydrologic zones, they varied significantly among wetland status categories (Table 3). Both SOM and clay content (Table 3) were greater in NWs than CWs or RWs. Likewise TC was greater in NWs than RWs, while TN was greater in NWs than either RWs or CWs. Soil pH was greater in CWs than in NWs or RWs (Table 3). Restored wetlands contained significantly more silt on average than NWs or CWs. In terms of sand content, while CWs contained significantly more sand than RWs, they did not differ significantly from NWs (Table 3).

Spearman correlation analyses revealed that moisture had significant positive correlations with SOM, EC, TN, clay, and silt and negative correlations with BD, pH, and sand (Table 4). Bulk density had positive correlations with pH and sand content, and negative correlations with SOM, EC, TN, TC, ExP, clay, and silt. Soil organic matter was positively correlated with every soil property except pH and sand, for which correlations were negative. Soil pH was positively correlated with sand and negatively correlated with TN, clay, and silt. Total

Table 3. Mean values (± 1 standard error in parenthesis) for soil properties that exhibited significant differences among the salinity class, hydrologic zone, and wetland status categories. Mean values with different letters are significantly different at the $p < 0.05$ level according to the post-hoc least squares means test. Soil properties that did not exhibit significant differences ($p > 0.05$) among these categories included moisture and salinity or status, bulk density and salinity, soil organic matter and salinity or zone, pH and salinity or zone, electrical conductivity and status, total carbon and salinity or zone, total nitrogen and salinity or zone, extractable P and salinity, zone, or status, and clay, silt and sand and salinity or zone. None of the soil properties exhibited zone*status interactions.

Salinity Class†				
Soil Properties	Fresh (n = 8)	Brackish (n = 20)	Euhaline (n = 3)	Hyperhaline (n = 4)
EC (µS cm^{-1}) §	3.7 (0.90) a	23.9 (5.1) b	37.4 (3.5) bc	74.5 (24.7) c

Hydrologic Zone††			
Soil Properties	Wetter (n = 35)	Intermediate (n = 35)	Drier (n = 35)
Moisture (g cm^{-3})	0.45 (0.02) b	0.42 (0.02) b	0.32 (0.03) a
BD (g cm^{-3})	0.50 (0.05) a	0.60 (0.05) a	0.75 (0.05) b
EC (mmho cm^{-1}) §	37.7 (11.4) b	28.0 (5.14) b	11.8 (2.28) a

Wetland Status††			
Soil Properties	Natural Wetlands (n = 17)	Restored Wetlands (n = 11)	Created Wetlands (n = 7)
BD (g cm^{-3})	0.45 (0.04) a	0.71 (0.04) b	0.69 (0.06) b
SOM (%)‡	24.9 (3.99) b	15.7 (1.07) a	13.3 (2.67) a
pH	6.73 (0.17) a	6.98 (0.19) a	7.77 (0.16) b
TC (%)§	11.27 (1.43) b	5.00 (0.47) b	8.39 (1.37) b
TN (%)‡	0.63 (0.09) b	0.25 (0.03) a	0.21 (0.05) a
Clay (%)‡‡	36.4 (2.90) b	39.6 (2.62) b	25.1 (3.28) a
Silt (%)‡‡	26.9 (2.44) a	36.1 (2.51) b	22.9 (4.75) a
Sand (%)‡‡	36.7 (4.45) ab	24.3 (4.25) a	51.9 (7.83) b

† Sample size = 35 and error degrees of freedom (df) = 31 for salinity ANOVAs.
‡ These soil properties were square-root transformed before the analysis.
†† Sample size = 105 and error df = 96 for zone and status ANOVAs.
§ This soil property was log transformed before the analysis.
‡‡ Sample size = 96 and error df = 87 for clay, silt, and sand as particle size was not determined on samples with > 30% SOM.

N had significant positive correlations with TC, ExP, and clay as well as significant negative correlations with sand. Total carbon was positively correlated with ExP and sand and negatively correlated with clay and silt. Extractable P was positively correlated with sand and negatively correlated with clay and silt. Clay was positively correlated with silt and negatively correlated with sand; silt and sand were negatively correlated.

For the PCA of the soil properties (Figure 2), axes 1 and 2 accounted for 42.5% (Eignevalue = 4.25) and 16% (Eignevalue = 1.57) of the total variance in

Table 4. Significant Spearman rank correlations ($p < 0.05$) among the soil properties measured at the 35 coastal lowland wetland sites in Hawai‘i. (ns = not significant).

	BD	SOM	pH	EC	TN	TC	ExP	Clay	Silt	Sand
Moist	−0.23	0.41	−0.31	0.50	0.32	ns	ns	0.35	0.43	−0.43
BD		−0.79	0.52	−0.43	−0.80	−0.50	−0.25	−0.48	−0.26	0.40
SOM			−0.65	0.50	0.90	0.42	0.34	0.63	0.48	−0.60
pH				ns	−0.61	ns	ns	−0.49	−0.51	0.52
EC					0.36	0.26	ns	0.22	0.23	−0.27
TN						0.54	0.46	0.50	0.30	−0.43
TC							0.17	−0.24	−0.61	0.46
ExP								ns	ns	ns
Clay									0.58	−0.87
Silt										−0.87

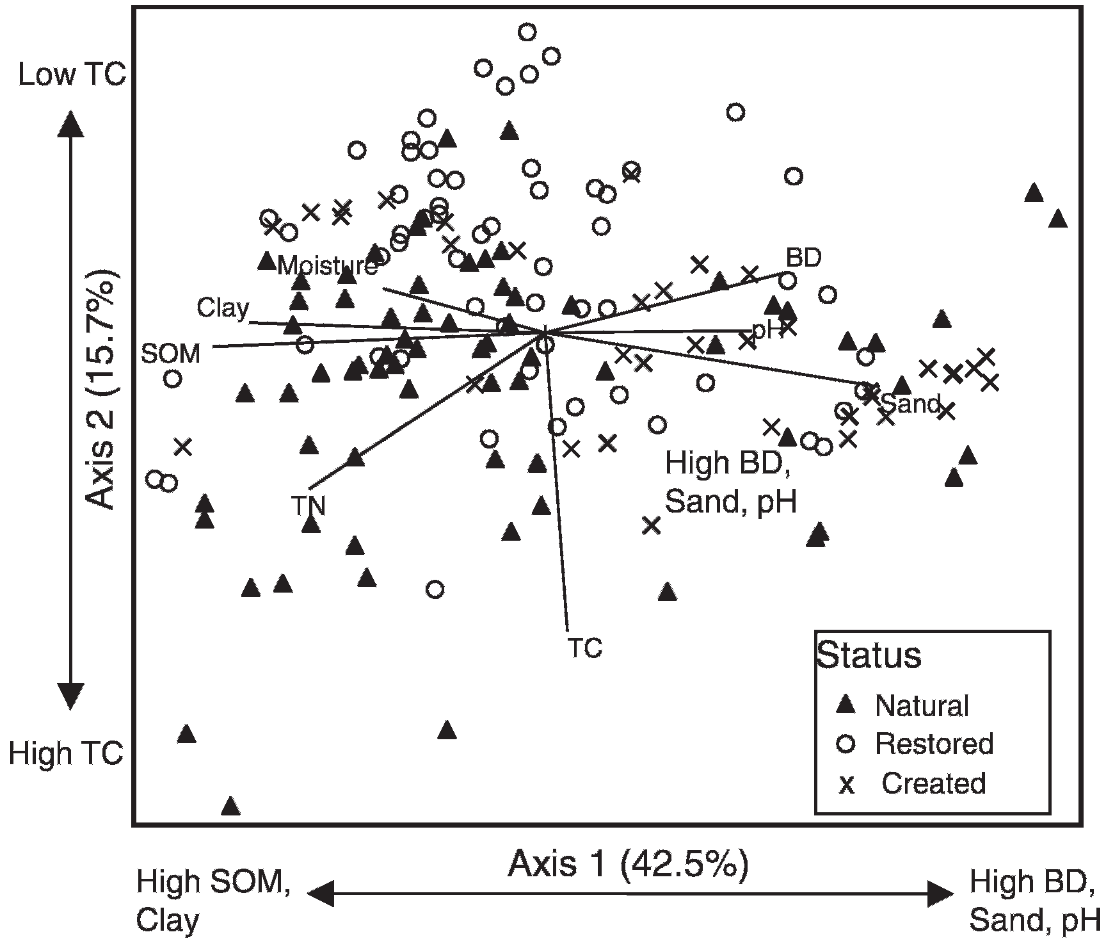

Figure 2. Principle components analysis (PCA) of the soil properties of coastal lowland Hawaiian wetlands. Includes sites with mineral soils only (SOM < 30%). Vectors represent the magnitude of correlation with axis 1 or 2. Axes 1 and 2 accounted for 42.5% and 15.7% of the total variance, respectively.

the data. Soil properties with strong loadings on axis 1 were SOM (Eignvector = −0.44), sand (0.42), clay (−0.39), pH (0.33), and moisture (−0.29). Total C had a strong loading on axis 2 (Eigenvector = −0.65) along with TN (−0.47), while ExP (−0.74) had strong loading on axis 3. Natural wetlands tended to group together at the lower end of axis 1, indicating high SOM and clay content. Created wetlands tended to group together at the higher end of axis 1 indicating high sand content, pH, and BD. Finally, RWs tended to group at intermediate values of axis 1 and at the higher end of axis 2, indicating intermediate texture (i.e., not all sand or all clay) but lower in TC.

DISCUSSION

Vegetation

The lack of significant differences in vegetative characteristics among wetland types (i.e., isolated versus riparian/estuarine) may be due to the alteration of vegetation communities resulting from prior land-use practices (i.e., agriculture and grazing) and the encroachment of invasive plant species from both terrestrial and marine environments. The dominant presence of exotic and invasive species such as *U. mutica*, which occurred across a wide range of isolated and riparian/estuarine sites, supported this finding. Significant differences in total wetland vegetation cover, total cover, and mean SR across the different salinity classes were likely the result of differences in salinity tolerances among wetland plants. Salinity affects vegetative abundance and distribution (Howard and Mendelssohn 1999, Frazer et al. 2006). Increased salinity also results in decreased seed germination (Baldwin et al. 1996), which may account for the lower total wetland vegetation cover, total cover, and SR observed in euhaline and hyperhaline sites compared to fresh and brackish sites.

The effect of hydrologic zone was more pronounced than the effect of wetland status in terms of vegetative characteristics. Increased exotic cover, total cover, and SR from wetter to drier plots across the hydrologic zones were likely due to the major stresses that occur in wetland ecosystems as a result of inundation (oxygen depletion, changes in pH, anoxia). Many plants are unable to survive in these conditions, thus limiting the number of species found within the wetter portions of wetland ecosystems (Mitsch and Gosselink 2007).

While total vegetative cover was the only vegetative parameter that differed among NWs, RWs, and CWs, the relative strength of this effect was less than that observed across the hydrologic zones (Table 2). The lower total cover observed in RWs may be explained by the fact that many RWs in Hawai'i are managed for waterfowl habitat which requires greater amounts of open water and bare mud flats.

In previous studies, CWs and RWs have been shown to have greater exotic cover (Fennessy and Roehrs 1997, Spieles et al. 2005) and proportions of upland plant species than NWs (Heaven et. al. 2003). Typically during creation or restoration, CWs and RWs go through a period of bare ground that often facilitates an influx of species adapted to disturbed environments (Fennessy and Roehrs 1997, Balcombe et al. 2005, Ervin et al. 2006). Thus, it is important to minimize the amount of time a RW or CW operates at an inhibited level of ecological function, and this is especially true in Hawai'i where invasive species encroachment can happen over the course of a single growing season.

In contrast to previous studies (Fennessy and Roehrs 1997, Spieles et al. 2005), we found similar exotic and wetland vegetation cover among the NWs, RWs, and CWs. This may have been a result of their low elevational position in the coastal landscape. Such locations are more susceptible to invasion by exotic species as they serve as landscape "sinks" which accumulate material (e.g., excess water, sediment, nutrients) from upland land-use changes and disturbances (Zedler and Kercher 2004). Furthermore, the presence of roads and other impervious surfaces can alter wetland hydrology by increasing the amount of water and associated contaminants flowing into downstream wetlands, leading to the invasion of weedy, exotic species (Kentula et al. 2004).

The PCA ordination revealed that the orientation of quadrats in ordination space appeared to be driven more by salinity than hydrologic zone or wetland status (Figure 1). However, the relatively low recovered total variance (40%) and the lack of significance of the principal component axes suggested that there may be additional environmental variables driving the vegetation composition in coastal lowland Hawaiian wetlands, including landscape position, site age, site size, and adjacent land-use (Heaven et al. 2003, Seabloom and van der Valk 2003). Alternatively, differences in environmental conditions that exist among the different islands (e.g., substrate age) as well as the location on an island (e.g., leeward vs. windward) may influence vegetative species composition.

The overwhelming number of exotic species (69 of 85) that were observed in this study was a major difference compared to other temperate and continental wetlands where invasive species are present but often not as dominant. In addition, the exotic species in Hawaiian CLWs are different than those found in continental or temperate systems (Zedler and Kercher 2004). For example, Hawai'i is one of the few places where *Rhizophora mangle* (red mangrove) is considered exotic and invasive (Allen 1998). Hawaiian CLWs appear to be similar to wetlands in Australia where 4 of the worst invasive species occur in wetlands and a dozen other wetland species are problematic (Zedler and Kercher 2004).

Our findings suggest that the vegetation of CLWs in Hawai'i needs more intensive management and invasive species control. These results also suggest that it may be difficult to use vegetation to locate "reference" sites for CLWs in Hawai'i due to the pervasive nature of invasive species regardless of wetland status. From a management perspective, however, there are a few sites (e.g., Lawai Kai, Nu'u, Kamilo Point) that have mostly native vegetation and could be considered the least-impacted. These sites are useful as they represent targets for current and future RWs and CWs and also give managers an idea of which sites and species are most resilient to current stressors such as land-use change and invasive species.

The dominant presence of exotic species in Hawaiian CLWs needs to be considered in the management plans for NWs, RWs, and CWs. Attempts should be made to control invasive vegetation with mechanical and chemical means, and to establish native vegetation. Manual seeding or planting can jump-start the growth and production of native vegetation in a disturbed environment that normally favors exotic species (Reinartz and Warne 1993). Additionally, if implemented immediately after restoration or creation, an invasive species control program may result in greater cover of desirable native species. This has been demonstrated at the Marine Corps Base Hawai'i wetland site, Klipper Pond, where managers and volunteers are regularly involved in invasive species removal activities. Native vegetation can also be transplanted to RWs or CWs from wetland sites slated for impacts.

Soil Properties

The 11 soil properties measured in this study provide a wealth of information about the physical (moisture, BD, SOM), chemical (EC, pH, TC, TN, ExP) and textural (clay, silt, sand) status of the soils in these CLWs. A number of previous studies of NWs, RWs, and CWs have shown significant differences in these properties among NWs, RWs, and CWs (Bishel-Machung et al. 1996, Stolt et al. 2000, Bruland and Richardson 2005). Such differences have important implications for the development and functionality of RWs and CWs. For example, as soils form the foundation for these ecosystems, inadequate soil conditions can be detrimental to the growth of wetland vegetation and the establishment of wetland hydrology. In addition, soils are the medium for important biogeochemical process such as denitrification and P sorption that retain and transform nutrients (Bruland and Richardson 2004). Differences in soil properties between NWs and CWs/RWs, suggests that CWs/RWs may not be performing such processes or only performing them at a suboptimal level.

Few studies have compared soil EC across NWs, RWs, and CWs as most studies of this type have involved either all freshwater wetlands or all tidal wetlands. In our study, which included a wide range of salinities, it was important to quantify soil EC as it has been shown to be an important driver of vegetation composition in coastal wetlands (Mitsch and Gosselink 2007). Furthermore, little, if any, information about this parameter exists for coastal Hawaiian wetlands. While there were no significant differences in soil EC among wetlands of different types or status, there were significant differences across hydrologic zones, with values significantly decreasing from wetter to drier zones. There are some important implications of this result with projected increases in sea level under global climate change scenarios. With sea level rise, we expect that soil EC values will increase in intermediate and drier zones of Hawaiian coastal wetlands, thus affecting plant communities and decreasing processes such as P sorption.

In terms of restoration, the hyperhaline sites are interesting as they do not fill the same functional role as fresh or brackish water wetlands. Originally these sites may have been mudflat systems without much emergent vegetation. These mudflats provide habitat for invertebrate communities adapted to these hypersaline conditions, and which serve as an important food source for waterbirds. Thus, those involved in restoration should be clear about the target vegetative community (i.e., coastal marsh versus mudflat) and how soil EC will effect vegetative establishment and development.

The effect of wetland status was more pronounced than the effect of hydrologic zone for the soil properties. Results of this study are consistent with previous studies that reported lower BD and pH, and higher SOM, silt, TN, and TC in NWs than in CWs and RWs (Bishel-Machung et al. 1996, Craft et al. 2002, Bruland and Richardson 2005). The higher BDs and lower SOM observed in CWs and RWs compared to NWs can be explained by several factors. First, the use of heavy machinery in project construction results in the compaction of soils, resulting in the higher BDs in CWs and RWs compared to NWs (Campbell et al. 2002). Second, higher BD and lower SOM in CWs and RWs may be an artifact of excavation into subsurface soil horizons that are compacted and have lower organic matter content (Shaffer and Ernst 1999, Bruland and Richardson 2005). Third, organic matter accumulation is a function of time, established vegetation, and hydrology. Organic matter accumulation is favored in NWs due to the inhibition of decomposition caused by the long-term anaerobic conditions typical of NWs (Craft et al. 2002, Hogan et al. 2004). In contrast, soils of CWs have not been under the same environmental conditions as NWs, thus soils of CWs may resemble typical of terrestrial soils with lower SOM (Hogan et al. 2004).

The low pH and high TC and TN values observed in the NWs were consistent with previous studies (Langis et al. 1991, Bishel-Machung et al. 1996, Fearnley 2008) and were likely the result of saturated soils, low oxygen levels, and subsequent inhibition of organic matter decomposition. When decomposition is inhibited, TC and TN accumulate in surface soil layers. This is coupled with the build up of carbonic acid that leads to the domination of hydrogen ions and lower pH values (Mitsch and Gosselink 2007).

Given the significant differences in clay content and SOM between NWs, RWs, and CWs, it would be expected that ExP would also differ (Hogan and Walbridge 2007). However, this was not the case. The similarity in ExP in NWs, RWs, and CWs may indicate that organic matter is not an important source of P in these Hawaiian wetlands or that P is a taken up fairly uniformly by plants across all 35 sites that were sampled in this project.

In previous studies that measured textural characteristics, NWs were shown to have higher silt and clay than RWs and CWs (Bishel-Machung et al. 1996), while CWs had higher sand (Bruland and Richardson 2005). The trend of higher sand and lower clay in CWs and RWs is the result of the removal of fine-textured surface soils during site excavation (Bishel-Machung et al. 1996). Our study found that NWs and RWs had higher clay content than CWs, but also that NWs had lower silt content than RWs. This may be due to differences in location or parent material. Further investigation of soil type, distance to the coast, and distance to rivers or streams should be conducted as these variables may account for the textural differences among NWs, RWs, and CWs. Textural differences have important implications for the function of wetland restoration and creation projects as fine-textured soils have greater moisture and nutrient retention capacities (Bruland and Richardson 2005, Fearnley 2008).

According to the Spearman correlation analysis, the majority of the soil properties were significantly correlated. Soil organic matter had significant associations with all other soil properties. This highlighted the importance of this parameter as an indicator of soil quality and development. This correlation structure was similar to that observed in a study of RWs and CWs in Pennsylvania (Bishel-Machung et al. 1996) and of CWs in Virginia (Bruland and Richardson 2004). In future studies of Hawaiian wetland soils, if budget or logistical constraints limit the number of soil properties to be analyzed, we suggest that SOM should not be omitted as it is relatively simple and inexpensive to quantify and provides much information about soil quality due to its correlations with many other soil properties.

The soil PCA generally supported the results obtained using univariate procedures. Sand (0.41) and clay (−0.39) had strong loadings on axis 1, suggesting that axis 1 represented a distribution of sites along a texture gradient (Figure 2). The strong loading of TC (−0.65) and TN (−0.47) on axis 2 suggests that this axis represented a nutrient gradient. The location of NWs sites across the majority of the biplot suggested that soils of NWs in Hawai'i are highly variable, although the majority of the NW sites were located in the central left portion of the biplot indicating soils with higher TC and finer textures. The location of CWs on axis 1 was indicative of soils with higher BDs and coarser textures than NWs or RWs. The cluster of RWs located in the center of axis 1 and slightly skewed towards NWs, suggested that soil properties of RWs are initially more similar to NWs than to CWs. Thus soils of RWs are more likely to support wetland functions than soils of CWs.

Several steps can be taken for mitigation to be more effective in replacing wetland structure and function as related to soils. An effective method for reducing soil compaction of CWs and RWs is to use a chisel plow to mechanically rip both the topsoil and subsoil layers, prior to planting, to alleviate soil compaction. Additionally, amendments such as compost, mulch, or other organic material have proven to be effective methods for increasing soil moisture, C and N, and decreasing BD at RWs and CWs (e.g., Stauffer and Brooks 1997, Bruland and Richardson 2004). Although an effective method elsewhere, the use of organic amendments in Hawaiian coastal wetlands needs further evaluation. Specifically, more research is needed to determine whether organic amendments will inhibit or promote the establishment of exotic and invasive plants. Finally, due to the greater similarities between NWs and RWs soil properties (Table 3, Figure 2) it can be inferred that the time for RWs to develop wetland functions characteristic of NWs will be less than that of CWs. Thus, given the option between restoration and creation, restoration should be the preferred mitigation option in Hawai'i.

Long-term monitoring of the development of vegetation and soil properties of RWs and CWs should also be conducted to provide much-needed data about the time requirements for RWs and CWs to develop ecological characteristics comparable to their natural counterparts. Further research is also warranted to examine the interactions between hydrology, vegetation, soils, and wildlife to provide a comprehensive assessment of the functionality of CLWs in Hawai'i. Finally, the cooperation among academia, government agencies, non-governmental organizations, and private landowners is encouraged, as it will result in the acquisition of more baseline and detailed data on wetland status and functionality. A better understanding of these wetland ecosystems will result in improved restoration/creation design techniques, construction methods, adaptive management activities, and long-term sustainability of these vital resources.

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