UNIVERSITY OF CALIFORNIA
SANTA CRUZ

COMPENSATORY MITIGATION AND HABITAT RESTORATION IN COASTAL CALIFORNIA

A dissertation submitted in partial satisfaction
of the requirements for the degree of

DOCTOR OF PHILOSOPHY

in

ECOLOGY AND EVOLUTIONARY BIOLOGY
with an emphasis in COASTAL SCIENCE AND POLICY

by

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June 2023

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Abstract

Compensatory mitigation and habitat restoration in California

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Habitat loss is one of the main threats to global biodiversity and ecosystem services. Human development, resource extraction, and climate change all contribute to the degradation or loss of ecosystems worldwide and in California. To counter these impacts, development may be regulated by following a general mitigation hierarchy where the priority is to first avoid injury to natural resources, then minimize remaining impacts, and finally, compensate for any unavoidable losses. This last step, known as compensatory mitigation, can offset environmental impacts through the restoration, enhancement, preservation, or creation (i.e., establishment) of habitat. My research focuses on three aspects of this process: the quantification of impacts to habitat, the cumulative effects of the permitted development, and strategies to improve outcomes of restoration projects.

In Chapter 1, I review the cumulative impacts of development permitted by the California Coastal Commission between 2010 and 2018. Wetland habitat was the most frequently impacted, and mitigation usually took the form of restoration and was predominantly on-site and in-kind. I also provide recommendations for improving the compensatory mitigation process. In Chapter 2, I review metrics and tools that quantify impacts to seagrasses, kelps, and other macroalgae. I provide a list and flow chart to identify tools best suited for ecological valuation and equivalency analysis for

mitigation. In Chapter 3, I demonstrate restoration actions that can establish cover and survivorship of a dominant species at a restored tidal marsh in central California. Irrigation and larger plants provided the most cost-effective strategies in the short-term, when compensatory mitigation projects are likely to be held to performance standards.

The topics of this thesis are varied but all relate back to the process of compensatory mitigation, the goal of which is to ensure the persistence of natural resources for present and future generations.

This body of work is dedicated to

Frederick Pausch,

who had a genuine interest in every project I was working on.

Thank you for getting me here and for all the memories that keep me laughing.

Acknowledgements

There are so many people who helped me get to the finish line. Thank you to my advisor, Pete Raimondi, who held my spot in the program and proved to be an exceptional mentor in his style of questioning while simultaneously encouraging. That is a fine line to walk, and I have learned so much from you. Thank you for being so available to your students, answering the same stats questions multiple times, and enthusiastically supporting my policy and leadership interests.

Thank you to Kerstin Wasson, who convinced to me to move my research landward and start counting things there. I have learned so much about tidal marshes and never in my wildest dreams could’ve imagined a more dynamic system. Thank you for your dedicated mentorship in applying ecological theory and your truly fantastic editing. Your investment in your graduate students is a gift to each and every one of us. Thank you.

Thank you to Don Croll, who kept me coming back to revisit why the casual reader should keep reading. Thank you to Rich Ambrose, whose vast experience in both academia and compensatory mitigation led to incredibly helpful edits. It’s hard to believe we’ve never actually met in person, but I thank you for all the Zoom hours.

Thank you to my final committee member, Lauren Garske-Garcia. Along with the rest of the Ecology team, you welcomed me into your regulatory world and invested so much in our project and education. I am excited to continue my career working with you and the incredible team.

Thank you to my coauthors on Chapter 2. Tommy, I so appreciate you opening your project to Beth and me. Beth, thank you for hanging in there with me to get this paper out. Our weekly check-ins kept me grounded during lockdown and after.

I must thank the team at Elkhorn Slough National Estuarine Research Reserve. Monique, Rikke, Andrea, John, Charlie, Susie, Kari, Mary, and Alex: thank you for all the lessons learned at the Slough and Hester.

Thank you to all of those on the Submerged Aquatic Vegetation team at NOAA. I truly appreciate your investment in the project. A special thank you to my coauthor, Jessie, who became one of the few people I could talk with about the finer points of equivalency analysis.

Thank you to the UCSC Greenhouses, Sylvie and Laura, for guiding me in rooftop science in the middle of COVID. Thank you to Judy Straub, Stephanie Zakarian, and Nicci Hack for taking care of all the things that slipped through the cracks and being such a strong support system for grads and TAs.

Thank you to the other faculty in EEB, who helped with questions outside my areas of expertise and collaborated with me as a graduate student representative. Ingrid, Suzanne, and Mark, I appreciate all of the time you've dedicated to grads.

Thank you to the RC lab and the 2018 cohort, whose support meant so much over the last five years. You are the reason Santa Cruz started to feel like home and your support has gotten me through the challenges of this process. Thank you to Alissa, Stephen, Jacqueline, and Joyce, who cheered me on from afar and then flew all the way to California for the final celebration. Current friends from the “before” times are a true gift, and your friendship means the world to me. Thank you to my mother for always picking up the phone and being happy to listen to the recaps of good and bad days. And finally, thank you to Sevag (and Fly), who has kept me smiling and enjoying life with our family on the Westside.

This work was funded by the National Science Foundation’s Graduate Research Fellowship Program, the California Native Plant Society, the Myers Oceanographic and Marine Biology Trust, and the UCSC Ecology and Evolutionary Biology Department.

Chapter 1 is a version of the following manuscript in prep:

Pausch, RE, Alexander, T, Howard, E. On-site and in-kind: compensatory mitigation of California Coastal Zone habitat impacts between 2010 and 2018

Chapter 2 is a version of the following manuscript in prep:

Pausch, RE, Hale, J et al. Quantity and quality: A review of ecological valuation and equivalency analysis methods for temperate nearshore submerged aquatic vegetation habitat

Chapter 3 is a version of the following manuscript in prep:

Pausch, RE. Testing strategies to enhance transplant success under stressful conditions at a tidal marsh restoration

Introduction

California has the highest plant and animal species richness of any state in the United States (Mooney & Zavaleta 2016). Its coast stretches over 1300 km and contains one of five Mediterranean-type regions in the world. This mosaic of habitats includes the California Floristic Province, a globally recognized hotspot of biodiversity (Bellard et al. 2014). Coastal dunes and estuaries buffer inland areas from flooding and sea level rise (Martínez et al. 2013). Eelgrass beds and estuaries sequester carbon and sediment, improving water quality. Salt marshes, seagrass, and kelp beds serve as species' nursery habitat and operate optimally when connectivity is maintained (Olson et al. 2019). Despite these habitats' importance, 23% of California coastal county land is developed, with over 500 square kilometers developed between 2001 and 2011 (Theobald et al. 2021). Threats to these systems include resource exploitation, increasing human disturbance from development and recreation, invasive species cover and a changing climate (Mooney & Zavaleta 2016). Warm water events can decimate kelp forests (Beas-Luna et al. 2020) and successful wetland vegetation establishment may be relatively dependent on large rain events (Allison 1992; Zedler et al. 2003).

To minimize continued loss of coastal habitat, point sources of human disturbance are highly managed in coastal California. Federal and state agencies enforce several regulations, including the National Environmental Policy Act, Endangered Species Act, Magnuson–Stevens Fishery Conservation and Management

Act, Clean Water Act, California Coastal Act, and California Environmental Quality Act. Regulators set a hierarchy for mitigation of development impacts to natural resources:

1) Avoid impact (e.g., siting a project away from sensitive habitat)

2) Minimize impact (e.g., using turbidity curtains to reduce sedimentation during construction)

3) Compensate for any remaining impacts (e.g., restoring a wetland or eelgrass bed)

This third step is known as compensatory mitigation, or the offset of lost resources through the restoration, enhancement, creation, or preservation of additional resources. In practice, this last step often involves a regulatory agency and the quantification of adverse impacts and subsequent mitigation action.

Successful compensatory mitigation can be examined through both an ecological and regulatory lens. The process has been most well-studied for wetlands. Only a portion of past wetland projects met compensatory mitigation permit requirements and even fewer met functioning criteria (Turner et al. 2001; Sudol & Ambrose 2002; Ambrose et al. 2007; Alexander 2020). Other projects failed to reduce cumulative development impacts (Stein & Ambrose 1998; Zedler 2004; Swenson & Ambrose 2007). Regardless, compensatory mitigation remains an important strategy to offset habitat loss. My thesis examines three aspects of this process in understudied coastal habitats: quantifying impacts and mitigation, the development and compensatory mitigation trends of a regulatory agency, and field strategies for improving restoration performance.

To better understand the development impacts and commensurate mitigation for various habitats, Chapter 1 reviewed publicly available CCC staff reports for all projects approved from 2010 to 2018 that impacted coastal habitats and required some sort of compensatory mitigation. I examined patterns in project size, habitat type impacted, type of impact, review the subsequent mitigation actions required for those impacts, identify habitat-specific trends related to development and compensatory mitigation, and provided recommendations intended to improve the mitigation process.

Chapter 2 reviewed peer-reviewed literature and ‘white-paper’ reports on habitat quantification tools historically used with, or applicable to, temperate nearshore habitats containing submerged aquatic vegetation, namely seagrass and kelp. I summarized the valuation and equivalency models available, their institutional origins, specific habitat focus, common input metrics or derived indicators, and strengths. I also included a checklist for future tools and identify further research to benefit the compensatory mitigation of these important ecosystems. My intent was to clarify the state of the habitat quantification tool landscape, provide managers with a list of tools and appropriate applications, and highlight specific gaps where development resources for existing and additional tools should be focused.

Not all coastal restoration projects are mandated by compensatory mitigation, but rather are motivated by agency initiatives, community efforts, or climate resilience. Regardless of motivation, identifying best practices for restoration is of broad interest. Chapter 3 tested multiple strategies to enhance restoration success in persistently bare areas of a tidal marsh restoration site. I transplanted a dominant halophyte species (*Salicornia pacifica*; perennial pickleweed) into areas of a restored tidal marsh in central California that had remained bare almost three years following a sediment addition. I tested four factors known to influence transplant success: salt hardening of plants prior to transplanting, irrigation, the size of initial plants, and planting configurations affecting potential facilitation. Due to *a priori* observations and hypothesized differences in sediment source and stress at the west and east sites of the project (Thomsen et al. 2022), I also investigated site differences and associated sediment properties. The findings of this study can be used to accelerate the success of other tidal marsh restoration projects, which will become more common as coastal managers develop and implement strategies for climate resilience.

While the topics of this thesis are varied, they each fit under the umbrella of compensatory mitigation and restoration of coastal habitats and directly address agency priorities and management concerns. Each chapter contributes to a better understanding of loss, mitigation, or specifically restoration of, coastal habitat, through both policy implementation review and experimental design.

Chapter 1

On-site and in-kind: compensatory mitigation of California Coastal Zone habitat impacts between 2010 and 2018

Abstract

Planning development while minimizing impacts to sensitive habitats poses a challenge for global natural resource management. After impacts from development are avoided and minimized to the maximum extent possible, remaining adverse impacts may be offset using compensatory mitigation. Along the California coast, the California Coastal Commission (CCC) regulates development and subsequent mitigation for allowable impacts. We reviewed publicly available CCC staff reports for approved projects that impacted coastal habitats and required compensatory mitigation from 2010 to 2018. The median project size was approximately 728 square meters and almost all permanent impacts were mitigated at a >1:1 ratio, with regional and habitat-specific planning regulations driving some variation across the state. We found that wetlands were the most frequently impacted and had higher mitigation ratios. Temporary impacts were almost always mitigated at a 1:1 ratio. While most mitigation was on-site and in-kind, mitigation that did occur off-site had a median distance of 4.7 km from the site of impact. Restoration was the most frequent mitigation action, over creation, enhancement, or preservation, but proportions of each action varied across habitat types. While our findings highlight no net loss of habitat area along the coast, the net change in ecosystem function is wholly dependent

on the performance of the mitigation projects. This review is only the first step in evaluating the success of compensatory mitigation along California's coast.

Introduction

Habitat loss is one of the main threats to global biodiversity and ecosystem services (Airoldi et al. 2008; Gonçalves-Souza et al. 2020). Human development, resource extraction, and climate change all contribute to the degradation or loss of ecosystems worldwide (Airoldi & Beck 2007; Mantyka-Pringle et al. 2012). To counter these impacts, development may be regulated by following a general mitigation hierarchy where the priority is to first avoid injury to natural resources (Phalan et al. 2018), then minimize remaining impacts, and finally, compensate for any unavoidable losses. This last step, known as compensatory mitigation, can offset environmental impacts through the restoration, enhancement, preservation, or creation (i.e., establishment) of habitat.

California (USA) is recognized as a global center for biodiversity with dozens of distinct marine and terrestrial habitat types (Heady et al. 2018) and over a thousand rare and endemic coastal species (Loarie et al. 2008; Baldwin et al. 2012; California Department of Fish and Wildlife 2022). While the total historical habitat loss in California is difficult to quantify, southern California has lost 75% of vegetated estuarine habitat since 1850 (Stein et al. 2020) and 49% of shrublands since the 1930s (Talluto & Suding 2008). Significant climate-related stressors such as sea level rise (Garner et al. 2015; Kaplanis et al. 2020), rising land and sea surface temperatures, loss of the coastal fog belt (Johnstone & Dawson 2010), ocean acidification (Kroeker et al. 2013), and wildfire (Schwartz & Syphard 2021) threaten remaining undeveloped areas. Planning development while minimizing impacts to sensitive habitat inevitably poses challenges for coastal California and global natural resource management.

Loss of coastal resources and public access to the coast under development pressure motivated the passage of the California Coastal Act of 1976 and subsequently, the establishment of the California Coastal Commission (CCC). The CCC is unique in that its establishment began with a 1972 voter initiative and it was granted broad authority to regulate development along approximately 1100 miles (1770 km) of California's coastline (see Figure A1.i. for details). The CCC's mission is to "preserve and enhance California's coast for present and future generations" (CCC 2019) within the California Coastal Zone. The Coastal Zone is an area identified by a politically drawn boundary that extends from the outer extent of state waters located three nautical miles (5.5 km) offshore, to a variable inland edge ranging from hundreds of feet (~60 m) in developed areas to more than five miles (~8 km) in more rural regions. Due to the robust framework of the Coastal Act and the economic value and resources of the California coast, some have called the CCC "the single most powerful land use authority in the United States" (Steinhauer 2008).

Among its many policies, the Coastal Act prohibits development in environmentally sensitive habitat areas [ESHA; habitat that is rare or especially valuable and easily disturbed or degraded; Public Resources Code (PRC) §30107.5] and wetlands with limited exceptions, in which case it still requires minimization of impacts and compensatory mitigation for those that cannot be avoided (see PRC §30107.5, §30240 and §30233 for definitions). The definition of "development" under the Coastal Act is expansive and includes not only activities such as construction, landform alteration, and land divisions, but also changes in density and land use intensity, discharge of various materials, and the harvest of major vegetation among other things (PRC §30106).

Implementation of the Coastal Act often occurs through local jurisdictions that have a Commission-certified Local Coastal Program (LCP) or other planning document. Once certified as consistent with the Coastal Act, an LCP and its associated policies are used by the local jurisdiction to guide permitting for development proposed within its Coastal Zone. In areas without a Commission-certified planning document and in certain other situations, the CCC retains jurisdiction and issues coastal development permits (CDPs) for approved development projects. When the Commission is the permitting authority, project applications undergo review by staff, often including a staff ecologist, to determine if the proposed project may adversely impact natural resources. In instances where impacts may be unavoidable but permitted, they are minimized to the maximum extent feasible and required to be fully mitigated. A staff report and recommendations are presented at a monthly CCC meeting, open to public comment, at which twelve appointed Commissioners vote to deny or approve the projects, potentially with special conditions (e.g., best management practices or monitoring requirements). If the Commissioners approve the project, a CDP is issued consistent with any approved conditions.

During the review process, the magnitude of impacts that would result from the proposed development and mitigation must be provided, or estimated until final impacts and mitigation obligations can be verified based upon actual construction. Functional assessments, which measure ecosystem structure or function, are widely cited as the most comprehensive approach to calculating impact and mitigation requirements (Tallis et al. 2015). However, these assessments can be difficult to standardize across California's numerous habitat types and often require site-specific investigations. Instead, the CCC uses an area-based mitigation ratio relying on the spatial area of impact rather than a measure of function of that area. The direct losses and gains in area from development and compensating mitigation, respectively, can be translated into a mitigation ratio that theoretically replaces adversely impacted resources. For instance, a 4:1 mitigation ratio would require four acres of mitigation for every one acre of impact. Mitigation ratios used by the CCC can be based on ESHA designation, the type of habitat, type of impact (e.g., temporary vs. permanent), mode of mitigation (e.g., creation, significant restoration, enhancement, or preservation), and specific considerations such as confidence in recovery trajectories or other guidance.

Regardless of habitat type, the Commission often requires mitigation ratios for permanent impacts greater than 1:1 to account for the inherent uncertainty of project success. Higher ratios also account for the temporal loss of resources, or the loss of ecosystem function and services from an area between the time of impact and the completion of mitigation (Quétier & Lavorel 2011). Temporal loss can be minimized using advance mitigation (i.e., mitigation completed before impact), which may occur through a regulated area of habitat that has been preemptively restored to balance other adverse impacts or a mitigation bank. In-lieu fees, or money paid to another party to fulfill mitigation requirements, can also replace permittee-responsible mitigation and instead channel project funding to select efforts. Regardless of the project funding and timeline, the CCC follows a set of guidelines generally prioritizing on-site and in-kind mitigation, meaning the habitat impact is mitigated in the immediate area with actions involving the same type of habitat(s). Out-of-kind mitigation can be justified when a nexus to impact exists (e.g., removing marine debris to compensate for impacts to a rocky reef).

The literature surrounding compensatory mitigation in California largely focuses on wetlands and aquatic habitat, or on habitat supporting specific protected species and highlights the overall mixed success of compensatory mitigation. For example, previous studies have found only a portion of wetland projects met permit requirements and even fewer met functioning criteria (Turner et al. 2001; Sudol & Ambrose 2002; Ambrose et al. 2007; Alexander 2020). Other projects failed to reduce cumulative wetland and riparian impacts (Stein & Ambrose 1998; Zedler 2004; Swenson & Ambrose 2007). When compensatory mitigation was found to be successful overall, nearby intact wetland habitat was cited as a driving factor (Breaux et al. 2005). With the historical mixed success of compensatory mitigation, it is crucial that the mitigation requirements mandated by regulators facilitate mitigation project success to replace lost resources.

The CCC has the challenge of regulating adverse development impacts across dozens of distinct habitat types, resulting in a compensatory mitigation program that is broader than the well-studied realms of aquatic and species-specific conservation banking (McKenney & Kiesecker 2010). To better understand the development impacts and commensurate mitigation for various habitats, we reviewed publicly available CCC staff reports for all projects approved from 2010 to 2018 that impacted coastal habitats and required some sort of compensatory mitigation. We examined patterns in project size, habitat type impacted, type of impact, subsequent mitigation actions required for those impacts, and habitat-specific trends related to development and compensatory mitigation. We also provided recommendations intended to improve the mitigation process.

Methods

We reviewed staff reports and noted the impacts of and compensatory mitigation required for projects approved by the CCC between 2010 and 2018 (N=433). We limited our investigation to certain project categories, excluding voluntary habitat restoration projects. We also excluded projects approved under LCPs and other certified planning documents unless the project had been appealed to the CCC or represented a consolidated permit where a project's area included both a local and CCC jurisdiction.

To better understand the habitat types impacted by development, we sorted projects into 21 habitat types (Figure 1) within six broad habitat categories: coastal dune/strand/bluff, marine/open water, oak/coastal sage scrub/chaparral, other upland, riparian, and wetlands. We used permit language to characterize habitat type. "Riparian" included vegetation types often near a stream or river. Similarly, the "other upland" category was used for non-wetland habitats that were not more specifically described (e.g., identified as oak woodland, chaparral, etc.) in the permit language. Impacted habitat described solely as "upland" was often ruderal, and non-native; restoration required in these areas was often considered best management practice. Some projects included more than one habitat type; thus, we noted impact and mitigation details both at the permit scale and distinct habitat scale. Areas of impacted and mitigated habitat were often expressed in acres and so we present acres here with metric conversions in tables.

To provide context for the extent of habitats included in the area applicable to our review (the California Coastal Zone, excluding areas with CCC-certified planning documents), we selected datasets that mapped habitats approximately aligned with some of the habitats in Figure 1. We clipped datasets (ESRI, 2021) for various ecosystems within areas of the Coastal Zone that did not have certified LCPs as of 2010 (California Coastal Commission, 2010; State Lands Commission, 2021). While maps for each habitat type were not available, we tallied the total areas of emergent tidal wetlands, emergent freshwater (palustrine) wetlands, and riparian habitat using the National Wetlands Inventory (USFWS, 2022). It should be noted that the habitat maps used offer extremely coarse estimates and did not always follow the exact categories as the CCC staff reports.

We sorted applicants as either public or private and by type: individuals, universities, military, parks (regional and state), public works/harbor districts/regional transportation (e.g., public works departments, airports, California Department of Transportation [Caltrans]), utilities (e.g., public water districts, Pacific Gas and Electric Company), and the remaining non-utility businesses.

Based on the permit language, we categorized impacts as temporary or permanent. “Temporary” generally meant the impact would last less than a year, or not significantly longer than the construction phase, with minimal ground disturbance (e.g., vegetation disturbance needed for a crew to access a work site). Longer or more severe impacts were considered “permanent”; however, the staff reports did not cite a formal definition or framework during 2010-2018. Other project attributes included the mode of mitigation (i.e., creation, restoration, enhancement, or preservation) and in- or out-of-kind mitigation, which we characterized based on whether the habitat type impacted was the same as the mitigation habitat (i.e., in-kind). For in-kind mitigation, we calculated the mitigation ratio by dividing the mitigation acreage by the impact acreage. To summarize mitigation ratios prescribed for the specific habitats, we separately analyzed projects that mentioned the use of advance mitigation (i.e., mitigation completed before impact occurred; n=16) in the staff report due to the theoretical lack of temporal loss. We followed permit language to determine whether mitigation was on- or off-site. In most cases, “on-site” referred to mitigation implemented on or directly adjacent to the parcel where the impact occurred. We also noted when mitigation occurred off-site as part of mitigation banks or in-lieu fees rather than being implemented by the applicant.

While mitigation approaches are defined differently by various agencies and literature, we followed permit language, which drew from past CCC decisions and customs. “Creation” is the establishment of a habitat type where it did not exist historically. In some permits, habitat restored by the removal of decades-old development (e.g., the removal of a seawall) was referred to as creation. Other agencies may use the term “establishment” to refer to creation; it can also involve the conversion of highly impacted habitat (e.g., a parking area) to another habitat of interest, like a wetland. “Restoration” often involves a complete revival of ecosystem functions that have been significantly degraded or lost from a location where they have historically existed. “Enhancement” improves targeted characteristics or functions and is often less comprehensive than restoration. “Preservation” does not improve habitat quality but provides an existing habitat area legal protections from specified uses.

Certain projects were approved at CCC hearings with possible impacts estimated but not finalized, generally due to shifting habitat (e.g., dynamic eelgrass patches) or the uncertain existence or extent of development impacts (e.g., final size of a construction staging area). In these cases, estimated areas of potential impact were typically mapped somewhere within the supporting documentation of the staff report. Commonly, permit conditions would require exact area calculations before a permit was issued post-hearing, or after the permit was issued and construction completed so that the impact estimates could be validated. Due to the uncertainty surrounding projects with mapped estimates but lacking finalized acreages at the time of the staff report, these were not included in our area calculations. We refer to these as “TBD impact” projects. Finally, we researched the status of permits as of December 2021; those recorded as issued suggested that a project’s pre-issuance requirements had been met.

Results

Who are project applicants in California and how large are their impacts?

Thirty-six to 58 projects were approved each year from 2010 to 2018, with 50% of approved applications submitted by private applicants, and 50% by public entities. Applicant groups and median project size are presented in Table 1. Individual applicants had the smallest median project size (0.02 acre; 81 m^2), while military projects were the largest (median 2.14 acres; 0.87 hectares) but very infrequent (<1% of projects). Regional transportation, harbor districts, and public works were approved for the greatest number of projects (n=152).

Which habitats are being most impacted?

Wetland habitat was the most commonly impacted category by frequency (Figure 2) when tallying impacts by habitat type rather than permit, given that one permit may impact multiple habitat types. However, wetlands were only the second largest category by area, with 262 acres (106 hectares) impacted compared to 565 acres (229 hectares) of marine and open water habitats. Seventy-seven percent (437 acres; 177 hectares) of the total marine/open water area impacted came from the operational footprint of a single aquaculture project. Other upland had the largest median impact size (0.72 acres; 0.29 hectares; for impact size distributions of habitat types, see Figure A1.ii.). Impacts were often very small on a project-by-project basis; 69% of projects adversely impacted <1 acre (~4000 m^2, Figure 3) with a median permit project size of 0.18 acres (0.07 hectares). The smallest projects (<10 ft^2; <1 m^2) often involved soft bottom habitat impacted by seawall and dock repairs. The largest impacts were larger than 40 acres (16.2 hectares) and included coastal dune (campground project development), soft bottom (aquaculture), eelgrass (aquaculture and dredging), coastal prairie (highway realignment), and tidal wetland (power station and campground) habitats.

Cumulative impacts led to a total of 161.0 (65.2), 62.8 (25.4), and 23.9 (9.7) acres (hectares) of tidal wetlands, freshwater, and riparian habitat, respectively. These acreages represented 4.6%, 1.1%, and 0.5% of the same habitat in existence in areas without certified LCPs (United States Fish and Wildlife Service 2022). Impacts to tidal wetlands represented 0.3% of the total tidal wetland area within the entire Coastal Zone (regardless of LCP status) mapped by the California Aquatic Resources Inventory between 2009 and 2016 (San Francisco Estuary Institute 2017). These estimates are coarse and based on similar, but not necessarily identical, habitat delineations.

How are impacted habitats mitigated?

Forty percent of the projects were approved at a CCC hearing with estimated acreages to be finalized, mainly due to dynamic habitat footprints (e.g., eelgrass) that require pre-construction surveys shortly before impacts would occur, or the uncertain existence or extent of development impacts (e.g., exact size of a construction staging area; see methods) that would depend on final project plans and post-construction surveys. Of the projects for which the extent of impacts was finalized at the time the permit was issued (60% of all projects), 32% had both permanent and temporary impacts, 49% had only permanent, and 19% had only temporary. Some habitat type alterations (e.g., tree removal) are permanent by nature, and thus resulted in projects without temporary impacts.

Ratios for permanent impacts varied both across habitats and within habitat types. Temporary impacts were often mitigated at a 1:1 mitigation ratio while permanent ratios were higher, most often 4:1 for wetlands and 3:1 for other habitat types (Figure 4), with a few exceptions. The nearly 9:1 coastal dune average ratio was primarily driven by individual home development in the Asilomar Dunes area in Pacific Grove, CA, which was guided by an uncertified, partial LCP mandating a large portion of a parcel be restored concurrently with other development. The average mitigation ratio excluding those projects was 2:1. Eelgrass ratios were driven by the California Eelgrass Mitigation Policy (National Marine Fisheries Service 2014). Oak woodlands had one project with a ratio lower than 1:1, but this project also included out-of-kind mitigation in place of a higher ratio. Some staff reports included justification for the use of ratios different from the averages shown in Figure 4. This included distinctions such as that between coastal sage scrub buffer habitat surrounding ESHA vs. threatened coastal California gnatcatcher habitat qualifying as ESHA. Other examples included increased mitigation ratios to replace non-native eucalyptus trees that supported raptor nesting, to prevent "permanent loss of potential raptor nesting habitat." Higher ratios were mandated for some off-site creation and restoration. There were also instances of deviations to a lower ratio, including a tidal wetland that used salvaged marsh soils for the restoration and was located at a historic marsh, surrounded by extant habitat.

Of 16 advance mitigation projects, >90% were required to mitigate adverse impacts at a 1:1 ratio pending credit availability at applicant-specific mitigation 'banks.' The most common applicants to utilize advance mitigation were government agencies, including Caltrans and the San Diego Association of Governments (SANDAG) for highway and rail projects, most often to compensate for tidal wetland and coastal sage scrub impacts.

Across all habitats, restoration was the most frequent mitigation action (compared to creation, enhancement, and preservation; Figure 5). Wetlands and marine/open water had the highest percentage of projects involving habitat creation. This was commonly achieved through the grading of adjacent upland, the removal of levees, and establishment of new areas of eelgrass. Preservation through conservation easements was more common than creation for certain habitats, such as dunes. Both preservation and creation were almost exclusively used to mitigate for permanent impacts, while enhancement accounted for a larger proportion of mitigation actions for temporary impacts. Within staff reports, restoration actions included invasive species control and native vegetation plantings, often citing a final restoration plan with additional actions. The median ratio prescribed by habitat and mode of mitigation is given in Table 2.

Most of the compensating mitigation included an on-site or in-kind component: 77% of permits included on-site mitigation, and 89% involved in-kind mitigation at the time of the staff report (Figure 6). Coastal sage scrub and tidal wetlands had the highest number of off-site mitigation projects (n=18 and 16, respectively). However, marine habitats, especially rocky reefs ("other benthic") and open water, had the largest proportion of total mitigation off-site and out-of-kind, respectively (e.g., marine debris removal as mitigation for cable laid in proximity to rocky subtidal reefs). Off-site mitigation occurred with projects that relied on mitigation bank or advance credits (i.e., actions that are commonly off-site; n=16 advance projects, and an additional 12 bank projects) or where limited availability of suitable habitat at the impact site precluded on-site mitigation. Off-site mitigation occurred at a median distance of 2.9 miles (4.7 km) from the site of impact (for projects with known mitigation locations at the time the staff report was prepared; 82% of off-site projects).

Over 94% percent of staff reports that cited habitat impacts specifically mentioned monitoring requirements or referenced a forthcoming mitigation plan that would include monitoring performance standards for consideration and approval before the permit could be issued. Monitoring requirements included in staff reports typically referenced a certain percentage of vegetation cover within one, three, and five years, or in the case of eelgrass, followed the guidelines set forth by the California Eelgrass Mitigation Policy (National Marine Fisheries Service 2014), however a full review of monitoring requirements across habitats is outside the scope of this study.

In 2021, three years after the last permit application we included, 75% of CDPs had progressed to the point where permits had been issued, indicating that all post-hearing “prior to issuance” permit requirements had been met (e.g., an approved restoration plan and detailed description of expected impacts). Twenty-five percent of projects showed a notice of intent to issue a permit, but no record of the permit being issued.

Discussion

Are mitigation ratios high enough to replace lost resources?

The majority of adverse impacts permitted came with some form of in-kind and on-site compensatory mitigation requirement, which has historically been the CCC’s preferred mitigation approach (CCC 1995), and often makes use of available habitat within the same parcel as the permitted development. Virtually all habitats were mitigated at a ratio higher than 1:1 for permanent impacts, effectively replacing lost habitat area. The performance of those mitigation projects is outside the scope of this paper (though wetland mitigation performance is reviewed in Alexander 2020), which precludes a conclusive answer to whether the prescribed ratios were sufficient in offsetting functional loss of the developed habitat.

McKenney and Kiesecker (2010) listed considerations that guide mitigation ratios across agencies: location and mode of mitigation, habitat rarity, the time lag between impact and complete mitigation (i.e., full ecosystem functioning of mitigated area), and uncertainty of project success. The uncertainty of project success can result from partial or complete failure of restoration efforts, both in terms of permit requirements (e.g., cover) and ecosystem functioning (Maron et al. 2012; Dixon 2018; Gibbons et al. 2018). Bradford (2017) examined freshwater habitat for fish (equivalent to our “riparian” category) and found that area multipliers between 1.5 and 2.5 accounted for uncertainty in project success. Alternatively, Moilanen et al. (2009) demonstrate that ratios higher than 8:1 may be needed to sufficiently mitigate areas with high conservation value. Other research regarding appropriate mitigation ratios is largely focused on wetlands and taxon-specific mitigation projects (Theis et al. 2022). Matthews and Endress (2008) reported that of a subset of projects with a realized average ratio of 1.1:1 (lower than permitted ratios in their study), only 30% met all success criteria. Appropriate ratios also depend on the definition of project success. Quigley and Harper (2006) found that while a 1.1:1 ratio resulted in no net loss for Canadian riparian habitat, at least 4.8:1 was actually needed to achieve gains in habitat productivity. Relatively low required mitigation ratios may achieve net land conservation, but higher ratios may be needed to achieve equivalent functioning (Pickett et al. 2013; Moilanen & Kotiaho 2021).

The CCC ratios for permanent impacts generally fall in between the values reported here from the literature, suggesting they are high enough to avoid net land loss, but without further monitoring data, could be too low to consistently conserve habitat function. While over 94% of the analyzed staff reports referenced a forthcoming finalized monitoring plan or success criteria, most commonly related to vegetation cover, multiple studies cite cases where the full extent of prescribed mitigation area, independent of quality, is never realized (Sudol & Ambrose 2002; Matthews & Endress 2008; Griffin & Dahl 2016). This highlights the importance not only of using the “right” ratio, but of allotting agency resources to tracking and enforcing project compliance.

While the mitigation ratios used for different mitigation actions (i.e., creation, enhancement, preservation, and restoration) within our broad habitat categories did not dramatically vary, enhancement generally had higher mitigation ratios than other actions for individual habitats. The US Army Corps of Engineers (USACE) states that restoration will lead to greater functional gains than enhancement or preservation (33 CFR § 332.3(a)(2)), supporting the reasoning behind higher ratios for enhancement. Creation should, in theory, provide the greatest functional gain but is cited as the least successful (33 CFR § 332.3(a)(2)). Preservation would not result in any net gain of function or area in the present day, but rather prevent future loss in targeted areas (Grimm 2020). Following this reasoning, one would expect to see the highest mitigation ratios required for preservation and enhancement, with lower ratios required for restoration and creation, assuming project success. While we did not observe this universally for the projects we reviewed, similar guideline ratios have been recently suggested by the Coastal Commission (Garske-Garcia, 2020).

In contrast to the permanent impact ratios, temporary impacts were often mitigated at a 1:1 ratio achieved through revegetating disturbed areas (e.g., construction staging areas, trails, etc.). Revegetation is often crucial to the recovery of these temporarily impacted areas. Wagner (2021) reviewed a subset of compensatory wetland mitigation projects permitted under the Clean Water Act through the USACE and found that up to 40% of projects did not fully recover to a pre-impact state. Fewer than 60% of Wagner’s (2021) projects included specific permit language requiring active restoration of these areas. We found the CCC included a special condition for active restoration of temporarily impacted areas in a majority of its staff reports. However, the 1:1 ratio prescribed in the CCC staff reports does not account for the temporal loss between the start of construction and completion of restoration. This is a widely recognized source of temporal loss that could be addressed with additional mitigation (Moilanen & Kotiaho, 2021). The CCC has also recently provided examples of mitigation for longer-term temporary versus permanent impacts (Garske-Garcia 2020) to recognize the increased impacts sustained from longer construction periods.

Some staff reports highlighted correspondence in amending project construction plans to follow the standard mitigation hierarchy of “avoid, and then, only if allowable, minimize, then compensate” impacts to habitat that was permitted (e.g., decreasing the area of a dock from the original proposal). This is encouraging, as multiple authors have highlighted the widespread failure of regulatory entities to enforce sufficient avoidance and minimization efforts before assigning mitigation ratios (Phalan et al. 2018; Bigard et al. 2020; Barbé & Frascaria-Lacoste 2021). Additionally, the Coastal Act (and the certified LCPs and planning documents that provide local jurisdictions with implementing authority) is relatively unique in specifying limits on allowable uses in California’s coastal resource areas.

<u>Are habitats impacted differently?</u>

Wetland habitats were the most commonly impacted habitat by frequency. This is likely because the Coastal Act gives greater allowance for impacts to wetlands (PRC §30233) that do not rise to the level of ESHA (PRC §30240). The CCC also retains jurisdiction over LCPs for all areas between the most seaward road and the mean high water line, where many tidal wetlands are located.

Projects that involved out-of-kind mitigation often described the nexus between impact and mitigation. Most common was the conversion between freshwater and tidal wetlands and riparian areas, usually occurring on-site because of site elevation recontouring for habitat restoration. Other out-of-kind actions included invasive *Spartina densiflora* removal as mitigation for entrainment impacts of seawater intake, and debris removal for impacts to rocky subtidal habitat. There is evidence that out-of-kind projects do not always achieve production equal to the production impacted (Burton 2002; Bull et al. 2015), but overall out-of-kind mitigation is largely understudied.

Some habitats are more dynamic over time than others. There was a high incidence of "TBD impact" acreages for eelgrass due to standards that eelgrass acreages be finalized just before construction, accounting for the species' seasonally dynamic nature and limited growing periods. Additionally, a statewide management plan (California Eelgrass Mitigation Policy; NMFS, 2014) recommends region-specific mitigation implementation ratios based on historical restoration success across California. It establishes a shared goal of final mitigation achievement at 1.2:1. Other acreages unquantified at the time of the staff report, like those associated with coastal sage scrub and chaparral habitats in the Santa Monica Mountains, were driven by permit language for final areas impacted to be reported following construction completion but similarly, set mitigation ratios to calculate final obligations from. As a result, these habitats were likely underrepresented in our area-based analyses.

Off-site mitigation and alternatives to permittee-responsible mitigation

Permit applications included approximately half private and half public applicants. While public applicants tended to have larger median projects and make more use of advance mitigation, previous studies in other areas have noted no significant difference in permit compliance between different applicant groups (Hill et al. 2013).

The largest median impacts by area were associated with projects proposed by government and utility applicants (e.g., Caltrans, San Diego Association of Governments). They also represented 88% of instances of advance mitigation, which theoretically reduces temporal loss by providing compensation before the impact occurs and is often considered preferable to mitigation occurring after impact (USACE & EPA 2008; Sciara et al. 2015, 2017). Some of these projects included project-specific banks, especially for construction occurring in phases over multiple years.

In California, mitigation banking is common for endangered species habitat, usually operating under the federal Endangered Species Act and California Endangered Species Act mandates (Grimm 2020). In some areas, mitigation and conservation banks are managed by businesses and private landowners that sell credits to a variety of applicants. These types of banks, open to various applicants rather than "applicant-specific" projects, as seen with Caltrans and Southern California Edison, are relatively uncommon in coastal California, possibly due to the price of land. However, once operating, public and applicant-specific banks can provide a streamlined experience for the regulators, applicants, and consultants implementing and monitoring mitigation. Following the large initial administrative task of establishing the bank, banking could offer the CCC reduced administrative effort for small projects, which currently constitute many permit applications and require considerable staff time to ensure permit requirements are met. Additionally, credits established before impact can add ecosystem function and services that may accrue benefits before an off-site impact occurs. With an ecologically appropriate credit and debit system (Stein et al. 2000), siting, and performance, banks can offset habitat loss. However, it should be noted that mitigation banks may not fulfill multiple agencies' mandates or required mitigation. The Coastal Commission requires mitigation for ESHA and other sensitive habitat, which may differ from other agency requirements. Creating banks, regardless of habitat type or policy, requires high administrative costs and usually takes years of multi-agency efforts to develop.

Banks, other advance mitigation, and permittee-responsible mitigation (i.e., individual projects managed by applicant-hired consulting companies) may occur off-site from the impacted area due to physical space limitation and suitable habitat availability. Thirteen percent of CCC projects within the study period included mitigation exclusively off-site, but mitigation sites listed in staff reports were often less than 5 km away from the site of impact. This bodes well for efforts to minimize local habitat loss or redistribution within the Coastal Zone. In contrast, BenDor and Brozović (2007) found an average distance of 27.5 km for off-site mitigation for Chicago wetlands. While banking may streamline mitigation and facilitate better performance, it is important to consider landscape context when deciding where off-site mitigation may take place (Tallis et al. 2015; Accatino et al. 2018; Bigard et al. 2020), as there is risk of transfer of ecosystem function and services to other areas, especially outside of the Coastal Zone, and loss of habitat corridors (Bowler 2000).

Regional programs and landscapes can affect which habitats are impacted and how those impacts are mitigated

Local district office operations, consolidated permits with LCP requirement considerations, and variation in habitat distribution along the coast contributed to variation in mitigation actions and outcomes. For instance, scrub habitat was much more commonly impacted in southern California; the small number of scrub-impacting projects in northern California is most likely an artifact of its limited occurrence and/or qualification as ESHA north of the San Francisco Bay (Westman 1981).

While we omitted projects approved under LCPs, we did include project decisions that were informed by partially certified LCPs. Each certified LCP contains two parts: the land use plan (LUP) and a subsequent implementation plan (IP). If only the LUP is approved, the CCC retains jurisdiction, but will often use the document as guidance when prescribing mitigation. An example of this is from the Asilomar Dunes area in Pacific Grove, which between 2010 and 2018 had only a certified LUP, which mandated development be limited to no more than 20% of any given lot area. Additionally, the LUP guidance required applicants to restore native dune habitat equivalent to a minimum of 80% of their lot size. Additionally, off-site mitigation at 2:1 for development impacts, executed with an in-lieu fee, was also required. As a result, the ratios for dunes in our data were relatively high, which highlights the importance of regional influence on compensatory mitigation for development.

Staff reports also documented an in-lieu fee program for projects located in the Santa Monica Mountains wherein applicants could pay into a fund managed by the Santa Monica Mountains Conservancy. Applicants could choose to pay an amount per acre to offset impacts to chaparral and coastal sage scrub rather than undertake permittee-responsible mitigation. The local Coastal Commission district office reported that most applicants (>90%) chose the in-lieu fee option (South Central Coast District Office, personal communication, 2021). The local popularity of this option over permittee-responsible mitigation options suggests that in-lieu fees might become even more popular if program availability increased statewide. This necessitates the use of an appropriate fee that ensures mitigation funded by these programs is in fact compensatory for its associated impacts. However, if fees had to be increased for sufficient mitigation of lost resources, then popularity with applicants may erode to some extent.

Recommendations and future areas for investigation

Our recommendations to ensure effectiveness of the permitting process and the "compensatory" nature of mitigation echo those listed by Alexander (2020) and others. We broadly suggest the use of alternatives to very small permittee-responsible projects, increased resource allocation to evaluation and enforcement, and periodic programmatic review of cumulative impact.

Because we can only address the information available at the time of the staff report, we recommend that considerable agency resources and attention be allocated to the monitoring of permit requirements and mitigation performance, especially with the prevalence of projects that required pre- and post-construction surveys to ascertain the final extent of impacts. A full analysis of specific permit requirements, operational definitions of success, metrics used, or whether these mitigation projects achieved the prescribed performance criteria is outside the scope of this paper, and we encourage future studies to examine mitigation performance in the style of Alexander's (2020) work on the CCC's permits issued for development in wetlands. Further studies of mitigation performance could also elucidate preferred indicators to quantify project success and net gains in ecosystem function.

The mitigation ratios used for the projects we reviewed varied based on type of impact, mitigation mode, habitat types, and in some cases, unique characteristics of the impact or mitigation site. Providing additional guidance on when ratios may be adjusted, such as Garske-Garcia's 2020 memo on mitigation approach, or other options (e.g., ILFs, out-of-kind, advance mitigation) would aid applicants in planning mitigation options. Additionally, periodic regional review of cumulative impacts and gains from mitigation could help guide appropriate mitigation requirements moving forward. We noted a high number of projects with impact acreages to be finalized, and 25% of permits that had not been issued as of 2021, implying impacts had not yet occurred. Periodic program review with final quantifications of impact and mitigation could give better landscape context to mitigation ratios, actions, and in-lieu fee or advance mitigation programs.

Additionally, more research is needed to better understand the connectivity of habitat and impacts of concentrating mitigation areas off-site from impacted habitats, possibly redistributing habitat and ecosystem services. At the very least, continued attention should be paid to the displacement from the site of impact. Increased mapping of specific habitats statewide could also contribute to incorporating landscape context into permitting decisions made by the Commission as well as aid in identifying mitigation opportunities.

The majority of permitted impacts were small, leading to cumulative administrative costs from numerous projects. Streamlining permitting processes and programs to consolidate administrative efforts can benefit both the applicant and agency staff. Smaller projects are often more expensive and harder to implement (Li & Gornish 2020), and permittee-responsible mitigation projects often do not have the same staff allocation that ILFs and banks receive. This suggests that once established, in-lieu fee programs and mitigation banks, or even applicant-specific projects that can supply advance credits, may provide logistical and performance advantages that can benefit both regulators and applicants. However, these programs are not without their challenges, such as cost, lengthy initial review processes, differing agency mandates, and failure to replace ecosystem functioning (Kihslinger et al. 2020). These programs are only preferable over permittee-responsible mitigation with regular reporting on in-lieu fee project performance to ensure functional equivalency, and consideration of redistributing resources if mitigation occurs off-site.

High mitigation ratios required in the Asilomar Dunes showed that LCPs could impose specific requirements for mitigation above and beyond what the Coastal Commission may require. We urge cities and counties to incorporate special areas of interest within LCPs to prioritize the conservation and restoration of these areas. Regularly updating LCPs can also help incorporate best available science and other ecological considerations (e.g., range shifts due to climate change).

Temporary impacts were mitigated at 1:1 for the majority of projects. While this results in no net loss of area at the conclusion of construction, it does not account for temporal loss of resources. We recommend that the CCC require ratios greater than 1:1 to account for this temporal loss from temporary impacts and have seen evidence of this practice in projects permitted after 2018.

In this study, we have presented a summary of statewide development impacts and associated compensatory mitigation within the California Coastal Zone. Although our findings are encouraging—such as the general use of mitigation ratios greater than 1:1 for permanent impacts, the small percentages of total habitat impacted versus available, the high percentages of in-kind mitigation, and localized use of off-site mitigation—these conclusions are only valid if project performance standards are met following permit issuance. The California Coastal Commission is uniquely positioned to manage a diversity of habitats in an ecologically and economically important region. We see the results presented above as only the first step in evaluating the success of compensatory mitigation along California's coast.

Table 1. Applicant types from all 433 projects, with median project size shown for projects with impact area determined.

Type of applicant	Percent of total applicants	Median project impact size (acres)
Public works/Harbor districts/Regional transportation	<1% (private) 35% (public)	0.27 (0.11 hectare)
Individuals	24% (private)	0.02 (0.008 hectare)
Businesses (non-utility)	16% (private)	0.13 (0.05 hectare)
Utilities	8% (private) 2% (public)	0.29 (0.12 hectare)
Parks	1% (private) 7% (public)	0.07 (0.03 hectare)
Universities	1% (private) 5% (public)	0.14 (0.06 hectare)
Military	1% (public)	2.14 (0.87 hectare)

Table 2. Median mitigation ratios (X:1) across habitat by mitigation action. Only projects with permanent impacts, determined acreage, and no advance mitigation were included. Number of projects is included at the bottom and to the right.

	Creation	Enhancement	Preservation	Restoration	N
Coastal dune/strand/ bluff	3.0		2.6	3.1	24
Marine/Open water	1.2	3.0		3.0	31
Oak/CSS/ Chaparral	2.0		1.7	2.0	22
Upland				1.0	5
Riparian	3.0	5.8		3.0	18
Wetlands	4.0	5.6		3.8	45
N	29	5	2	109	145

Figure 1. Specific habitat types were assigned based on permit language and sorted into six broader habitat categories.

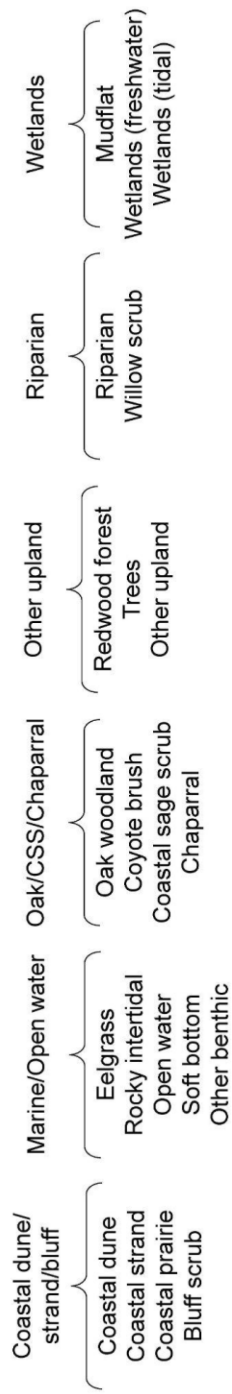


Figure 2. Frequency of known habitats impacted (top). Median project size (middle) and expected cumulative acreage impacted (bottom) were calculated using projects with acreages identified, n=30, 53, 52, 15, 33, 106 from left to right.

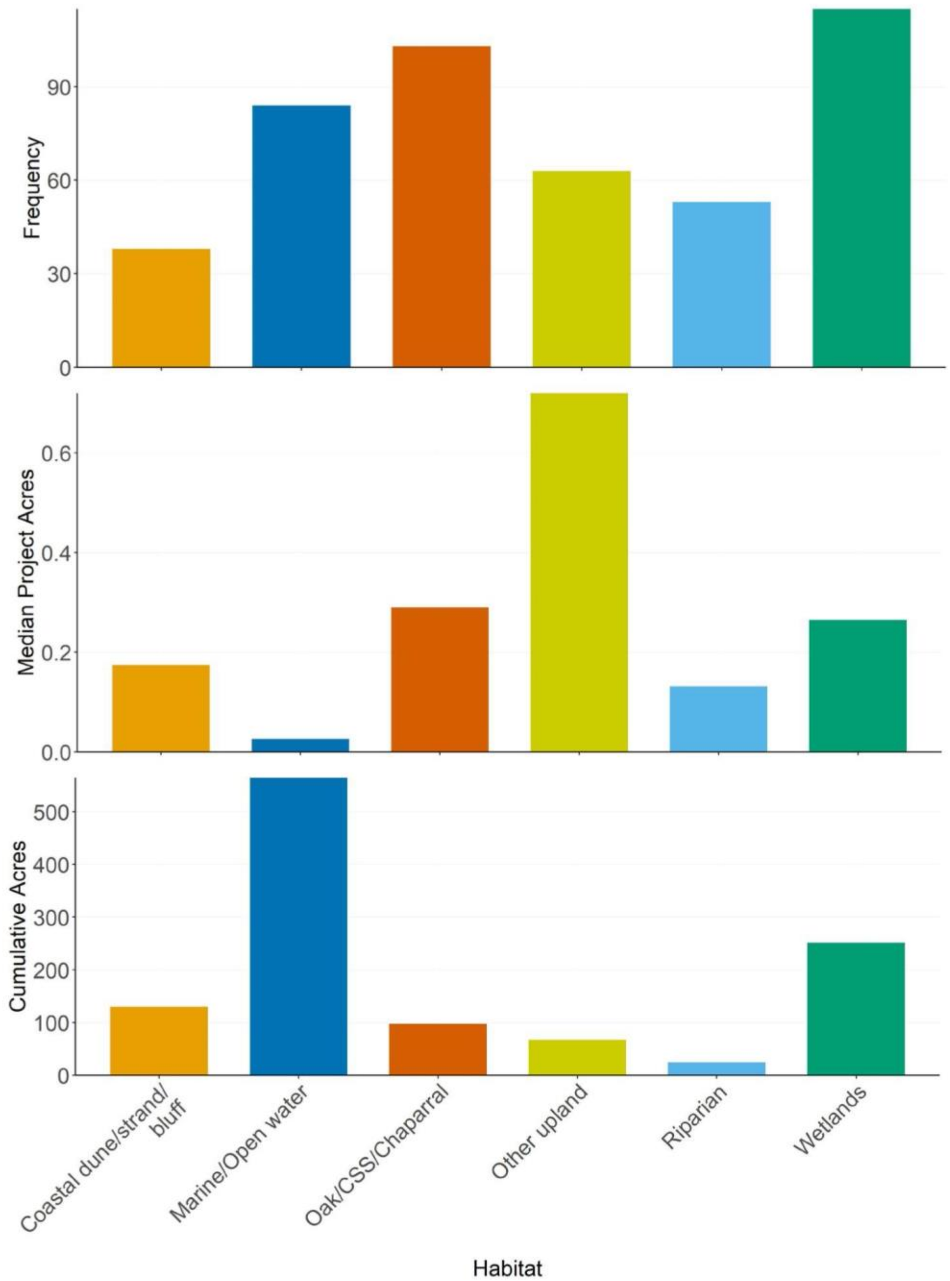


Figure 3. Frequency of permit application impact sizes to individual habitat types. Note log axis; line demarcates an impact size of 1 acre (0.40 hectares).

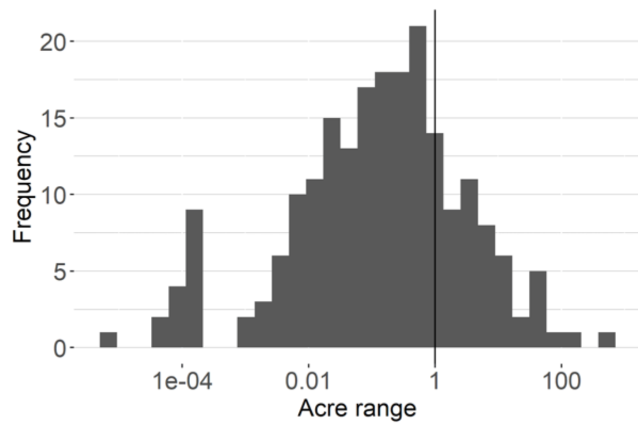


Figure 4. Average in-kind mitigation ratios for permanent and temporary impacts. Ratios were calculated by dividing the mitigation area by the impact area by habitat type; error bars show standard error. Number of impacts per habitat type shown above bar. Projects utilizing advance mitigation, often mitigated at 1:1, were excluded.

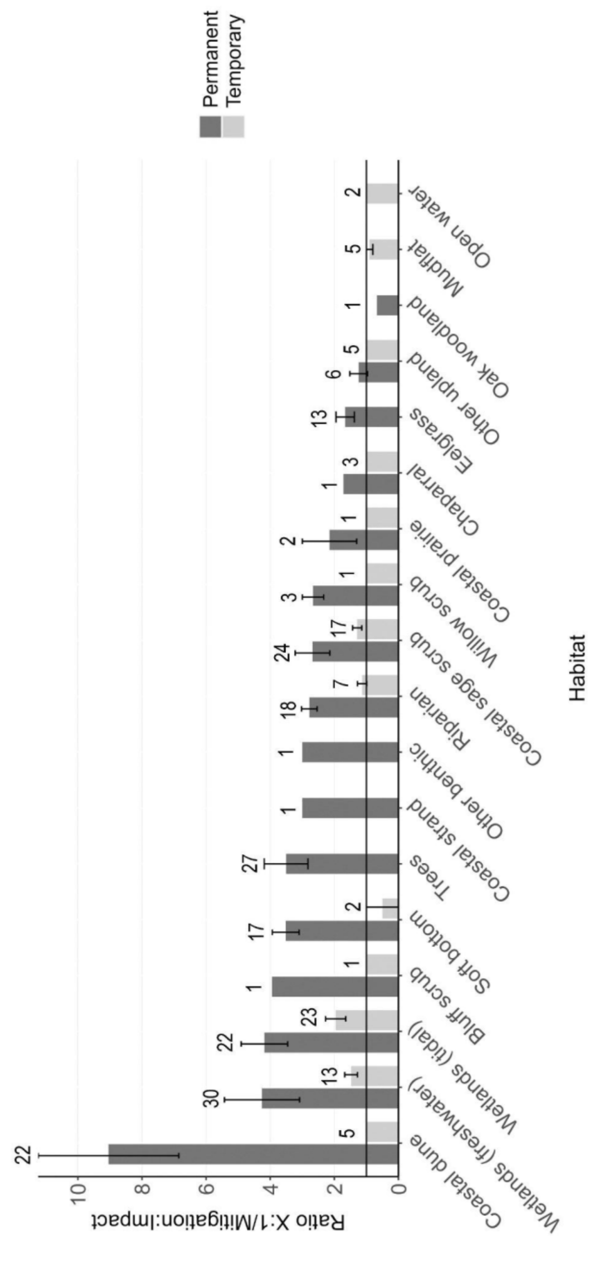


Figure 5. Frequency of various mitigation actions used across habitats for both permanent and temporary impacts for all projects, including if acreage was not finalized at the time of the staff report. TBD refers to the mitigation action that had yet to be finalized at the time of the staff report, pending pre-construction surveys, mitigation credit availability, etc. Numbers denote the number of mitigation actions by habitat.

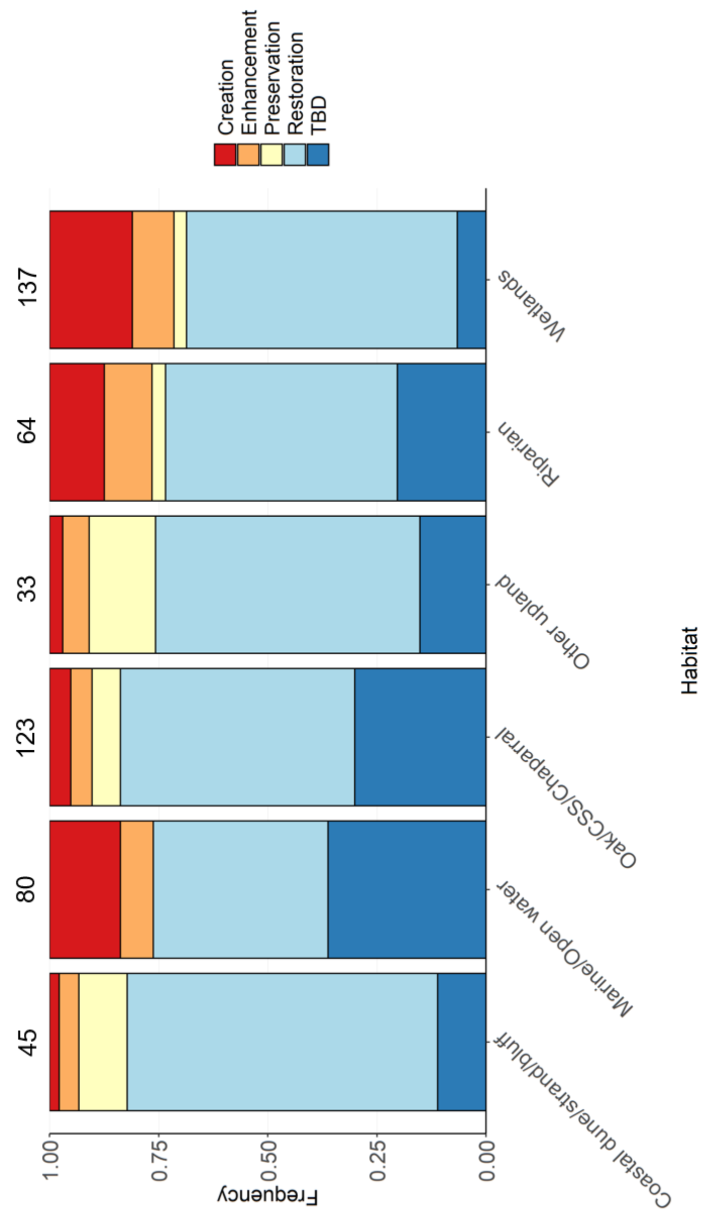
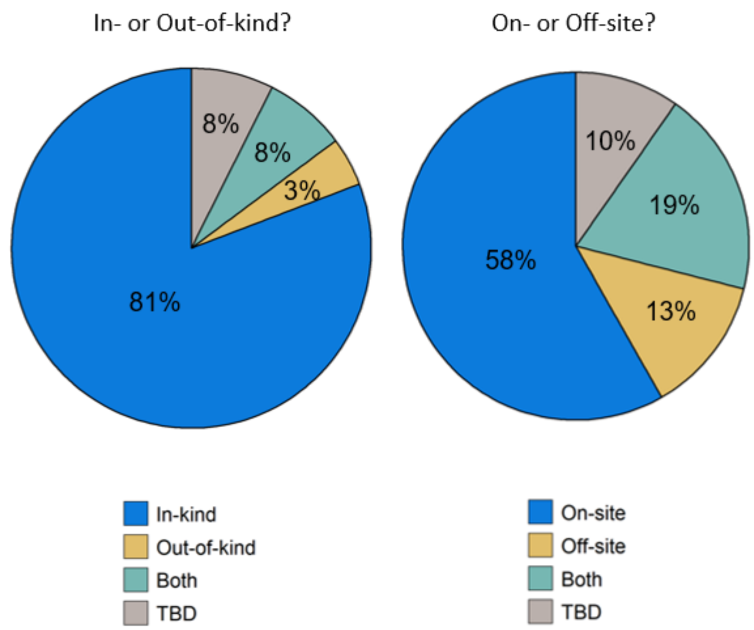


Figure 6. Percent of permits with in-/out-of-kind and on-/off-site mitigation (out of 297 permits with finalized impacts). Mitigation for TBD projects had not been finalized at the time of the staff report.



Chapter 2

Quantity and quality: A review of ecological valuation and equivalency analysis methods for temperate nearshore submerged aquatic vegetation habitat

Abstract

Nearshore seagrass, kelp, and other macroalgae (submerged aquatic vegetation; SAV) are productive and important ecosystems. Development and other impacts to these habitats require tools to quantify their ecological value and the debits and credits of impact and mitigation. To summarize and clarify the state of SAV habitat quantification, we searched peer-reviewed literature and other agency documents for tools that either assigned ecological value to SAV habitats or calculated equivalencies between impact and mitigation. Overall, there were more valuation tools available for seagrasses than kelps or other macroalgae. Eighty-eight percent of tools were developed for agencies or with agency funding. Common categories included Habitat Equivalency Analysis-related tools for oil spills in the United States, and valuation tools that scored habitats as a ratio in relation to reference or ideal conditions, including models designed for the European Union's Water Framework Directive. We provide a flow chart for decision-makers to identify tools that may be applicable to their own management needs. Tool input metrics spanned three spatial scales: individual shoots or stipes, 'bed' or site, and landscape or region. The most common metrics used were cover, density, area, and tissue content. The broad use of similar tool inputs highlights the need for further research

investigating relationships between these metrics and ecological functioning over time and space.

Introduction

Temperate nearshore marine habitats, from subtidal rocky reefs to intertidal estuaries, are some of the most productive ecosystems in the world and provide important ecological and commercial services (Wilson & Liu 2008; Hynes et al. 2021). Kelp forests and seagrass uptake nutrients, improve water quality (Orth et al. 2006), trap sediment, and sequester carbon (Fourqurean et al. 2012; Krause-Jensen & Duarte 2016). Large areas of these habitats can act as buffering systems for both chemistry changes (Nielsen et al. 2018; Hirsh et al. 2020) and wave action (Pinsky et al. 2013). Seagrasses and kelps are also foundation species, and form meadows and beds that serve as critical nursery habitats for harvested species (Toft et al. 2015; McDevitt-Irwin et al. 2016; Kennedy et al. 2018).

Despite the importance of these habitats to both humans and the nearshore marine environment (Filbee-Dexter & Wernberg 2018), over 65% of worldwide coastal wetland and seagrass area has been lost (Lotze et al. 2006), and almost 40% of the world's kelp forests are in decline (Krumhansl et al. 2016). Threats to these habitats include resource exploitation, increasing human disturbance from development and industry, subsequent trophic imbalances, invasive species, and a changing climate (Steneck et al. 2002; Mooney & Zavaleta 2016; Beas-Luna et al. 2020). While efforts to regulate or restore nearshore habitats and ecosystem functioning are prevalent in many regions, these efforts are often challenged by adequately quantifying pre-impact baselines and restoration outcomes.

To minimize continued degradation and loss of seagrass beds (used here interchangeably with "meadows"), kelp beds, and other coastal habitats, impacts to resources are highly managed worldwide. Agencies enforce laws and regulations managing development, habitat, and species protection, including the Clean Water Act (US), Endangered Species Act (US), Magnuson–Stevens Fishery Conservation and Management Act (US), Environment Protection and Biodiversity Conservation Act (Australia), and the Water Framework Directive (EU), as well as other state and regional equivalents and complements. These regulations can require review of development and impacts to associated habitats, often with some sort of compensatory action for lost resources. To reduce degradation and loss of habitat utilized by managed species, a general hierarchy for mitigation (40 CFR §1508.20; 33 CFR §332.1(c); (IUCN 2016; NOAA 2022) of development impacts exists across multiple continents to:

1) Avoid impact (e.g., siting a project away from sensitive resources)
2) Minimize unavoidable impacts (e.g., using grated material for docks to allow the passage of sunlight)
3) Compensate for any remaining impacts (e.g., restoring a nearby seagrass bed).

The last step is known as compensatory mitigation, or the offset of lost resources through the restoration, enhancement, creation, or preservation of habitat (USACE & EPA 2008). In cases of accidental human impacts, like oil spills and unregulated discharges into waterways, compensatory mitigation is often the only action possible within the mitigation hierarchy. Successful compensatory mitigation is assumed to be truly compensatory, where all lost habitat value or resources are replaced following an impact, usually with preference for replacing the same resources locally when feasible (McKenney & Kiesecker 2010). Evaluating the equivalency of this loss and gain requires accurate quantification of baseline status, impact to the habitat, and subsequent benefit of the compensation action and time to recovery.

While there are myriad options for quantifying impact, regulators commonly require some sort of measurement of the area to be affected by the action, sometimes paired with a measure of habitat quality or value with respect to one or more specific traits. The field of 'habitat valuation' often refers to economic valuation, where ecosystem services are converted to some present-day monetary value (Shaw & Wlodarz 2013; Dewsbury et al. 2016; Hynes et al. 2021). For the purposes of this review, we omit purely economic methods and instead focus on ecological habitat valuation, or the process of prescribing a rank, score, or index to a defined area of habitat based on metrics linked to ecological function. While prescribing an ecological value score undoubtedly oversimplifies the functions and services of a complex habitat, resource managers require practical tools for decision-making in the face of incomplete understanding and ecological complexity. When data to inform the development of such scores is limited, or missing for one of several priority services, the precautionary principle may be applied as a form of bet hedging to reduce anticipated impacts in the face of an incorrect decision (e.g., Cooney 2004).

A model that captures the complex functioning of a system will likely depend on simplified, representative metrics or proxies for ecosystem condition (Smit et al. 2021). The decision of which metrics to use may be influenced by policy mandates, the species of interest, project goals, or data availability. Metrics may be measured in the field or remotely (e.g., cover, biomass), and are often assumed to be related to functional or ecological value. Common proxies for ecosystem function include single species traits or abundance (e.g., density of kelp stipes, the stem-like part of kelp that provides structural support; Krumhansl et al. 2016), abiotic measurements (e.g., turbidity), productivity, and community structure (e.g., associated invertebrate community).

Habitat valuation models and their metrics can be used to assess both the impact and mitigation site, before and after an alteration or mitigation action. For example, what is the value of an existing seagrass meadow where a dock is proposed to be built? If the dock development requires mitigation in the form of planting seagrass at a nearby site, what is the value of a formerly bare area that now supports planted seagrass? Equivalence assessment methods (Bezombes et al. 2017) can then integrate these values over time and/or space, ultimately providing a credit and debit system for impact and mitigation and identifying the net change of habitat resources (e.g., commonly with the objective of 'no net loss'; Moilanen et al. 2009; Maron et al. 2018).

Habitat valuation and equivalence assessment tools collectively comprise what we refer to here as "habitat quantification tools" (Figure 1; Chiavacci & Pindilli 2018). Due to the varied terminologies used in published literature, we provide relevant definitions and related terms in Table 1. For the purposes of this paper, and because of the variety of terms used across applications, we use "tools" and "models" interchangeably (Figure 1).

Of all marine or coastal habitats, the practice of habitat valuation and ecosystem equivalencies is most developed for vegetated wetlands (Strange et al. 2002), which commonly includes sites with erect, emergent vegetation. Other nearshore marine systems, like eelgrass meadows and kelp forests, referred to here as submerged aquatic vegetation (SAV; macroalgae/kelp and seagrass), are under-represented in quantification tools (Jacob et al. 2018). Less than 10% of the methods identified by Chiavacci and Pindilli's (2020) review of quantification tools were for marine species or habitats; the majority of those focused on salmonid habitat. Published literature on temperate marine system quantification has largely focused on individual organisms, such as seabirds, or bycatch rather than habitat, especially since quantifying organisms can be operationally simpler than quantifying habitat (but see Levrel et al. 2012; Dewsbury et al. 2016; Jacob et al. 2018). Those tools that do exist for marine habitat often focus on in-kind mitigation, where the habitat impacted is the same habitat mitigated, leaving little guidance for when in-kind mitigation is not feasible and the impacted and mitigation habitat differ (i.e., out-of-kind mitigation).

Nearshore systems pose special challenges when attempting to choose metrics to assess ecological value. Marine systems are biologically dynamic, with open populations, migratory species, annual species, and shifting biomass (Munsch et al. 2023) with strong temporal patterns (Stephens et al. 2015; Hamilton et al. 2022). Tides, wave action, storms, cyclic oceanographic phenomena and land-sea connectivity also contribute to variable abiotic conditions constantly in flux across scales. Quantifying dynamic habitat attributes over time is often not possible due to site accessibility, project timelines, funding, and limitations of remote methods. This complexity and the underrepresentation of SAV habitat quantification methods leaves regulators with fewer tools to quantify impact or assign compensatory mitigation for these economically and ecologically important ecosystems.

Our paper reviews peer-reviewed literature and ‘white-paper’ reports on habitat quantification tools historically used with, or applicable to, temperate nearshore habitats containing SAV. We summarize the valuation and equivalency models available, their institutional origins, specific habitat focus, common input metrics or derived indicators, applications, and strengths. We also provide an example model output comparison, include a checklist for future tools, and identify further research to benefit the compensatory mitigation of these important ecosystems. Our intent is to summarize and clarify the state of the habitat quantification tool landscape to draw attention to specific gaps where development resources for existing and additional tools should be focused.

Methods

We searched published literature using Web of Science and Google Scholar, as well as white paper reports available online for SAV habitats (“kelp” OR “macroalgae” OR “SAV” OR “seagrass” OR “submerged aquatic vegetation”) associated with the terms “habitat valuation”, “habitat evaluation”, “ecosystem equivalency”, “mitigation ratio”, “habitat quality”, “intrinsic value”, “habitat suitability index”, “HSI”, “biocentric value”, “functional assessment”, “metric”, “index”, and “score.” We also searched incidents from NOAA’s Damage Assessment, Remediation, and Restoration Program for unpermitted impacts to macroalgae or seagrass to note the metrics used in the valuation portion of Habitat Equivalency Analysis, a commonly used equivalency method in oil spill damage assessments.

We chose models that either related specifically to temperate marine or estuarine SAV (i.e., excluded non-applicable stream, riparian, and palustrine/lacustrine/inland wetland tools) or were generic across habitats and could reasonably be applied to SAV. We included models that either measured some aspect of SAV directly or valued broader habitats that included SAV. These broader habitat models were used to assess a particular habitat type (e.g., an estuary) or an area for a particular species (e.g., a habitat suitability index for shrimp). We did not include models that predicted occurrence of SAV based on environmental factors, such as species distribution models, due to the difference between measuring *in situ* habitat value versus predicting occurrence (Stephens et al. 2015). To keep our search relevant to methods being used today with best available science, we only included models utilized between 2000 and 2022. If a method or paper was developed or written before 2000 but cited as still in use after 1999, it was included.

Some models assigned a score or index for SAV habitat itself (e.g., a seagrass meadow or kelp bed) based on one or more metrics. Others quantified broader habitats that include SAV, such as estuaries, or areas that support species of interest, as seen in habitat suitability indices. Tools for these broader habitats were included only if they measured one or more attributes specifically related to SAV. We classified models as habitat valuation tools, equivalency tools, or both (Figure 1). We noted a tool's inclusion of temporal variability in assigning habitat value, reference sites, landscape context, a factor for uncertainty in assigning mitigation, and user complexity. Complexity was based on the number of model inputs and effort or expertise required to obtain the input data. For example, a visual, rapid method to measure seagrass cover in the field was considered "basic," while a tool that used multiple GIS layers or knowledge of an area's metapopulation was considered "complex." We also noted the affiliation of the first author of the citation as a proxy for the institution or entity responsible for the tool and the acknowledged funding sources.

Once we identified available tools, we chose two models that used similar scales of metrics to examine variation in valuation outputs. We compared a hypothetical $20m^2$x$1m^2$ swath of eelgrass (Figure A2.i) and a hypothetical reference site using the Seagrass Quality Index (Neto et al. 2013), which weighted species richness, area with greater than 5% cover, and the 90% percentile of shoot density. We also calculated the Habitat Structure Index (Irving et al. 2013) using model-specific scores for richness, measures of patch continuity, cover, and area. We modified its output to provide a ratio to a reference site to match the Seagrass Quality Index ratio and compared the two final scores.

Results

Figure 2 categorizes identified tools by their intended use. Ultimately, tools quantified broader habitat that contained SAV (either generally or for a specific species) or specific areas of predominantly SAV. For each of these categories, tools either quantified quality, ecological value, or function, or translated this value into equivalency assessments. In some cases, tools could be used to value habitats and create maps identifying particularly sensitive or important areas (Figure 2).

Model categories

Upon selection of the 33 models that met our criteria (Table A2.i), we found that both valuation and equivalency models fell into three general categories: 1) Habitat Equivalency Analysis (HEA; both valuation and equivalency) or related; 2) Habitat Suitability Index (HSI; valuation)/Habitat Evaluation Procedure (HEP; equivalency); and 3) SAV valuation models built under the European Union’s (EU) Water Framework Directive (WFD) to calculate Ecological Quality Ratios (EQR).

We classified the remaining tools as a diverse ‘other’ category that included both valuation and equivalency models from around the globe.

Habitat Equivalency Analysis

Six of the identified tools are either based off, or designed to be used with, HEA (NOAA 1995), which includes both a valuation and equivalency component. This method assumes that lost services can be offset, and that lost service-years can be compensated for by providing additional acres of habitat. The HEA framework includes a habitat quality value or percentage loss of services, the timeline of impact and mitigation, area impacted and mitigated, and a discounting rate that values future habitat less than present habitat. HEA was widely referenced in the literature, most likely due to its frequent use in Natural Resource Damage Assessments (NRDA; commonly used for oil spills). Previous research has lent suggestions for improving HEA (Thur 2007; Shaw & Wlodarz 2013), and highlights assumptions that may be violated in practice, including the proportional scale of services with increasing area and consistent ecological value of habitat over time (Dunford et al. 2004; Desvousges et al. 2018). New methods developed incorporate aspects of HEA and other habitat quantification tools [e.g., Baker et al.’s (2020) HaBREM model, which incorporates biomass per unit of area; Bas et al. 2016; Cabral et al. 2016]. While HEA poses some challenges in its application, there is precedent for its use in courts and across agencies in the United States, with global applications (Kim et al. 2017).

Habitat Suitability Index (HSI) and Habitat Evaluation Procedure (HEP)

HSIs are a category of valuation tool that combine multiple metrics or indices, divide by the score of an ideal habitat, and result in a rating of 0-1 for habitat quality, with 1 representing ideal conditions. Ideal habitats are defined either by using reference sites or as an amalgamation of metrics' optimal ranges for a focal species (e.g., high SAV cover, temperature range). This score can then be used with the HEP equivalency model (USFWS 1980), incorporating the area impacted or mitigated (known as Habitat Units) and average annual function to identify mitigation. We included the generic HSI and HEP procedures as well as three specific applications of HSIs in our HSI/HEP category. HSIs have historically been most common for species of birds and fish (Terrell & Carpenter 1997), as the US Fish and Wildlife Service implements the tool for managing endangered species.

Ecological Quality Ratios (EQR)

The WFD guides the EU's goal toward aquatic ecological quality objectives and has resulted in the development of EQR (Gamito 2008) for seagrass and macroalgae species. Like HSIs, habitats are assigned a score between 0 and 1, with 1 representing an ideal reference site. The WFD directs that these tools should include reference conditions and measures of physio-chemical elements for seagrass. We identified six tools in this category that have been developed for or applied to seagrass species in Europe, and one for macroalgae. Additional monitoring programs have been referenced in the literature, though we did not find specific literature on their associated tools (Marbà et al. 2013).

Other tools

We included all other models or tools (n=15) in an 'other' category. Valuation tools in this category included tools like the Braun-Blanquet scoring method for seagrass (modified from Braun-Blanquet 1932) and the California Rapid Assessment Method (Collins & Stein 2018), which considers SAV presence when calculating the patch structure richness of a particular wetland assessment area. The simplest tool was an area-based mitigation ratio to calculate equivalency, where an area of required mitigation was presented per unit of area impacted. Within this category, mitigation ratios could be an output for models either specifically calculating equivalency based on area, or tools that incorporate valuation as well.

Types of tools and origins

Eighty-eight percent of our tools (n=31) included a valuation component and 33% had some sort of equivalency analysis (n=11; Figure 3). There were tools that included both, like the Puget Sound Nearshore Conservation Calculator (Ehinger et al. 2023), which used its own model to value nearshore salmonid habitat based on SAV density and other landscape attributes, and then fed into a HEA-based debit and credit calculator. Nearly half of the tools (n=16) were authored by a member of a regulatory agency or government research group, while the majority of the rest had first authors primarily affiliated with academic institutions (Figure 4). However, 88% of total tools (n=31) acknowledged a natural resource-related government source of funding. Twelve percent of tools (n=4) were written/developed by consultants, but in each case the tool was affiliated with an agency. Most complex tools that required either a large number of inputs or specialized input data often came from academia.

Common SAV metrics

There were notably fewer tools adapted for macroalgae than seagrass. Tools that could be used for kelp and other algae were most frequently based on some combination of area and cover. Cover was also the most common metric input into seagrass and habitat tools, followed by density, most commonly shoot (i.e., turion) density of seagrass (Figure 5). Tissue content (carbon, nitrogen, etc.) was also a frequent metric for seagrass; this was primarily used in the EU EQR tools. Seagrass metrics utilized a variety of spatial scales, from sampling individual blades for tissue content, epiphytes, or morphology, to site scale metrics that sampled along transects and described larger areas. While some models included landscape-scale inputs, the only metric that related directly to some aspect of SAV was regional rarity.

The number of SAV-specific metrics, meaning some aspect of SAV was measured (e.g., cover), rather than a broader site metric like wave exposure, varied across tools. Many of the tools used three or less inputs in their models, usually some sort of area, cover, or density (Figure 5). Alternatively, the EQR tools contained between 5 and 14 inputs that varied in spatial scale (Figure 6).

We found one instance of a direct SAV metric used as input in an impact assessment following an oil spill. *Phyllospadix torreyi* (surfgrass), as well as *Egregia menziesii* and *Fucus distichus* (macroalgae), were used in a HEA for the 2015 Refugio oil spill in Santa Barbara, CA (Witting and Sullivan 2019). The impact assessment translated baseline oiled SAV cover (54%) adjusted by habitat depth and distance to shore to a percent reduction in 'services' (i.e., function) in the most oiled intertidal and subtidal areas. An annual discount rate of 3% was used.

Regional and temporal context

Perhaps due to widespread prioritization of ease of implementation and minimal sampling, models that incorporated repeated measurements or metrics over time were rare (7%). Repeated measurements included in models did not specifically relate to SAV, but rather other parameters in broader habitat models and were often parameters that could be collected quickly or remotely. For example, the brown and white shrimp HSI (Turner & Brody 1983, Smith et al. 2010) used mean water temperature over time. While rare, there were instances of repeated, field-collected data; Bond et al (1999) developed a method for estimating fish habitat value, including rocky reefs with kelp beds, by using fish density, fidelity, and mean size across multiple time points.

Both broader habitat and SAV-specific models commonly define preferred SAV survey times during the summer growing season to capture the maximum extent of distribution and allow for inter-annual comparison of surveys. For instance, the

California Eelgrass Mitigation Policy (CEMP; NMFS 2014) requires surveys to be done during the summer growing season. Similarly, the CARLIT macroalgae model (Ballesteros et al. 2007) specifies an April-June timeframe. Biological value map scores (Derous et al. 2007) are calculated based on population maximums throughout the year.

Reference sites or conditions were used for the development of many models (43%). As stated above, reference conditions provide the highest possible score for EQR, where another measured site would be presented as a proportion in relation to that reference site. HSIs also present a score between 0-1; in some cases a "1" represents conditions at a reference site, but in other models a "1" is an amalgamation of ideal conditions based on various studies for the focal species.

Forty-eight percent of tools were developed using metrics or weightings designed for a specific region, making half of our identified tools applicable to an extremely small area globally. Thirty-nine percent of the tools included some sort of broad landscape or seascape context. Some models include population connectivity or special weighting or scoring for areas near key features, like pocket or natal estuaries for salmonids (Puget Sound Nearshore Conservation Calculator; Ehinger et al., 2023). In other cases, and related directly to SAV rather than the broader habitat, the model incorporated a metric for 'rarity' of a habitat by looking at regional occurrences of the same habitat (biological valuation map model; Derous et al. 2007). "Regional" was defined differently for different models; it may refer to a regulatory district, or a physical boundary, like a watershed. The USACE Mitigation Ratio Checklist

(USACE 2021) equivalency tool allows for lower mitigation ratios to be used when a mitigation action converts a more common habitat type to a rarer, and ecologically more valuable, habitat. The Five-Step Wetland Mitigation Ratio Calculator (King & Price 2004), which was adapted for seagrass in CEMP (NMFS, 2014), permits adjustments in input parameters based on differences in landscape context between the impact and mitigation areas.

Additional tool attributes

Models' ease of implementation was generally correlated with the number of SAV and broader habitat parameters. Habitat metrics included salinity, temperature, and densities or attributes of other species' populations in the area. Ten percent of the models in our database utilized inputs from an existing database or mapping effort. For example, the Nearshore Assessment Tool for southeast Alaska (Adamus & Harris 2018) uses the NOAA ShoreZone Program's mapping inventory (Harper & Morris 2014) for inputs of wave exposure and slope.

Overall, we observed that models meant for rapid assessment utilized *in situ* sampling that could be completed in less than a day, and credits and debits could be calculated using a published framework or spreadsheet. Most models required moderate user knowledge of biological surveying and working knowledge of a user interface, usually translating to a professional with some knowledge of regional databases, maps, Excel, and field survey methods. Other methods, like mitigation ratios, were much more straightforward, only requiring a survey of the project area.

Some complex tools had accompanying templates, spreadsheets, or interfaces, simplifying the user experience. However, some tools were considered complex due to their intensive laboratory analysis required (e.g., isotope data).

Model comparison

A comparison of seagrass beds using the Habitat Structure Index (see Irving et al. 2013 for methods) and Seagrass Quality Index (see Neto et al. 2013 for methods) produced ratios using the quotient of the 'sampled site' and 'reference site' values for each model (see calculations in Figure A2.i.). The final outputs for the same site using the Habitat Structure Index and Seagrass Quality Index were 0.78 and 0.80, respectively, representing a 2% difference.

Discussion

The majority of habitat quantification tools we identified were designed for seagrass or broader habitats with SAV, and included cover, density, area, and in cases of models driven by water quality policy, tissue content metrics.

Strengths and challenges of metrics

Measures of cover and density were likely most common due to their ubiquity across seagrass species and ability to standardize these measures when collected under specific protocols. While a full discussion of the nexus between metrics and ecosystem functioning is outside the scope of this review, in practice models assume high SAV cover and density positively correlate with ecosystem functioning. However, the details of this relationship warrant further research. We list some of the logistical strengths and challenges of these and other commonly used metrics in Table 2. It should be noted that the effectiveness or appropriateness of these metrics as proxies for ecosystem functioning is highly dependent on survey and statistical methods, as well as interpretation.

For permitting projects, logistically simple metrics are preferred by applicants, but are less ideal for assessing complex ecosystem functions like carbon sequestration and nutrient cycling. Cover, area, and density (Marshall et al. 2019) are not without their own nuance. In the field, the buoyant medium of intertidal habitats presents challenges when attempting to quantify individual shoots or stipes, especially when shoots or stipes may be fully exposed and lie flat at lower tides. Area without a density measurement may be less informative of ecosystem function due to the variation in biomass a set area may encompass. Some protocols overcome this by multiplying area by cover or prescribing a density threshold. While density may be more correlated with biomass (Vieira et al. 2018), biomass is also influenced by shoot/stipe height and life stage. Integrating both density and height can provide more accurate estimations of habitat structure, which is correlated with overall habitat function. Models would benefit from explicit descriptions of how to survey habitat metrics. CEMP presents a strong example of specific survey standards (NMFS 2014).

Some valuation tools can integrate multiple metrics into a single indicator or value that can then be used with an equivalency tool. However, multiple metrics could feed directly into an equivalency tool without first being combined. For example, Baker et al.'s Habitat-Based Resource Equivalency Method (2020) used multiple metrics to produce multiple calculations of an injury. However, only the mitigation that compensated for the largest injury, or slowest to recover, was then used as the required mitigation action. This "limiting factor" approach allows for comparison across many metrics but uses only one that theoretically encompasses the mitigation of many functions lost, minimizing the uncertainty in undetected impacts.

Our almost identical outputs from the Seagrass Quality Index and Habitat Structure Index were likely due to similar metrics taken at a similar spatial scale (i.e., all seagrass patch metrics rather than physio-chemical or landscape metrics). This could suggest that tools operating at the same spatial scale with similar input metrics may provide relatively consistent outputs across tools.

<u>Addressing temporal variability and impacts and mitigation over time</u>

The ephemeral nature of some species of kelp and seagrass, both spatially and temporally, poses a challenge to quantifying impact and appropriate mitigation. The edges of seagrass meadows may migrate up to tens of meters annually (Munsch et al. 2023), suggesting that currently unoccupied habitat near an existing bed should be acknowledged in mitigation plans. Rhizomatic growth in seagrasses and shifting substrates with SAV can complicate mapping efforts, as well as lead to patchiness and landscape heterogeneity that should be acknowledged in models. The ecological value of small-perimeter benthic patches of seagrass versus continuous patches is not fully understood. While resistance to invasion and patch stability has been shown to increase with kelp and seagrass patch size (Cunha & Santos 2009, Layton et al. 2019, Reeves et al. 2022), small patches facilitate connectivity and can help recover areas of dieback (Greve et al. 2005, Cavanaugh et al. 2015). Temporal variability also exists due to some SAV's annual growth cycles and vulnerability to extreme events like storms and marine heatwaves (Hamilton et al. 2022). Consequently, most regulators assign specific time periods for surveying to minimize seasonal variability across years (NMFS 2014, Calloway et al. 2020). Annual surveying across multiple spatial scales can help detect changes in both perennial and annual populations. Tools with documentation of these specific protocols provide better context for their outputs.

Metric choice is relevant for accurate determination of the response time to impact and restoration. Beheshti and Ward (2021) and Roca et al. (2016) found varying timelines of seagrass response to stressors and recovery, and varying degrees of response across types of indicators. For instance, physiological measures showed less response to stressors than structural and demographic indicators. However, physiological measures were much faster to show indications of recovery than demographics. This could influence calculations of impact and mitigation if not planned for in advance.

Time may be incorporated into equivalency assessments differently depending on the method and nature of impact. Time can be included by considering: 1) the duration of impact (e.g., the lifespan of a structure); 2) the time until full recovery from an impact, or the time for initial mitigation action to reach a full-functioning condition; 3) the duration of mitigation (usually incorporated via monitoring requirements); 4) the delay between impact and mitigation; or 5) a discounting factor to adjust debits/credits for present and future value. HEA-related tools most commonly addressed these factors, but other equivalency methods, like the USACE checklist (USACE 2021), increased mitigation ratios based on the number of months full functioning of the mitigation was delayed. CEMP also included factors for discounting, the time between impact and start of mitigation, and time until full functioning was achieved. This consideration of time itself, rather than just the temporal fluctuations of a system, can be important to achieve full compensatory mitigation.

Regional considerations

In addition to seasonal changes, nearshore habitat functions can be affected by site- and landscape-scale factors. Protected areas like pocket estuaries can increase growth and survival in juvenile salmonids (Beamer et al. 2003; Hodgson et al. 2020). SAV function is also affected by landscape (or for subtidal SAV, “seascape”) context, supporting higher diversity when included in a connected mosaic of various habitats (Olds et al. 2016, McAfee et al. 2022). Our identified tools addressed this by including an input for landscape context, which can refer to the spatial arrangement of habitats (Henderson et al. 2017) and connectivity with other non-SAV habitat types (Swadling et al. 2019). Terrestrial influences and connectivity between areas may influence ecological value (Yeager et al. 2020). For example, a culvert limiting fish access to a stream or proximity to a sewer outfall would reduce value. Alternatively, the sole eelgrass habitat in a bay may serve as the only herring spawning habitat, thus increasing its importance.

Many of these models were valuation procedures specific to certain regions and policy frameworks. Our flow chart (Figure 2) shows tools sorted by management scenario, but it should be noted that some valuation tools are highly specialized for certain regions and are more useful as examples rather than applicable to all areas. Meanwhile, equivalency methods were geographically broad, with a few notable exceptions (i.e., the region-specific planting ratios of CEMP), but could be used with more region-specific valuation. This diversity in tool region and policies presents a barrier for the widespread use of specialized methods, but the general framework and strengths could be incorporated into region-specific models.

Addressing uncertainty

A large challenge with equivalency assessments, in practice, is the uncertainty surrounding restored habitat performance. There are multiple sources of uncertainty associated with habitat quantification tools and the compensatory mitigation process:

1) Uncertainty that the habitat functions quantified and accounted for are the most ecologically relevant and "important" to model and mitigate
2) Uncertainty in accurately quantifying change in habitat value or equivalency, both when following and violating the assumptions of the model

3) Uncertainty in the performance or functioning of the mitigation action (e.g., restoration)
4) Uncertainty in calculating how much area or which mitigation action is needed to compensate for the impact (due to the previous sources of uncertainty)

We have discussed the first of these sources, connecting metrics to function, and suggest this as an area in need of further research. The second concerns uncertainty in the quantification of impact and mitigation when the assumptions of a valuation model are violated. Many tools from our database listed model assumptions, while others made implicit assumptions that were not specifically articulated. Ample documentation exists for the assumptions of HEA: the type and quality of impacted and restored services need to be similar, the value of impacted and restored services is assumed to be constant over the assessed time, and impacted services should be limited to relatively marginal changes in ecosystem function (Strange et al. 2002, Dunford et al. 2004, 2019, English et al. 2009, Ray 2009, Shaw & Wlodarz 2013, Desvousges et al. 2018). These assumptions are not always met in applications of HEA. However, imperfect use of an established framework may be preferred over discrete decisions made solely on best professional judgment.

Many of the models we identified, although not all, relied on the assumption that restoration function will operate at the expected capacity within a certain time horizon. However, restoration is rarely successful without continued monitoring and subsequent remediation measures for underperformance (Beheshti & Ward 2021, Eger et al. 2022), and the failure of an area to support recruitment or biomass at the calculated capacity has been observed across SAV types, including seagrass (Bayraktarov et al. 2016) and restored kelp reefs (Reed et al. 2004). A common assumption is the linear recovery of an impacted site or increase in performance of a mitigated site. Linear recovery is rarely the case (Fong 2015), however King and Price (2004) showed that the shape of the recovery curve is less important than the estimation of time to restore full functioning habitat, provided additional manipulation of the mitigation site is not implemented in response before this time horizon is reached.

To buffer against multiple sources of uncertainty, many regulators utilize mitigation ratios at ratios higher than 1:1 for mitigation:impact. Pilot studies and reviews of past projects were used to calculate a percent likelihood of mitigation failure for different regions in the California Eelgrass Mitigation Policy. CEMP utilized this percent likelihood of failure metric within the underlying mitigation calculator tool (King and Price, 2006) to generate a higher starting mitigation ratio (as high as 4.82:1 in northern California) to provide greater assurances that the ultimate performance requirement (i.e., standard mitigation requirement is 1.2:1) is achieved. Uncertainty analyses can also be incorporated into models (Zajac et al. 2015) to increase the probability of providing sufficient mitigation.

A small percentage of the models we identified included some sort of uncertainty factor addressing one of these sources in their calculations. The Uniform Mitigation Assessment Method (UMAM; Florida DEP 2005, Levrel et al. 2012, Stantec 2016) and its hybrids (Bas et al. 2016) include a risk factor for the failure of mitigation actions. This risk factor was chosen by the user and ranged from 1-3, encompassing multiple sources of uncertainty, from time to completion and restoration method. The CEMP mitigation calculator (King & Price 2004; NMFS, 2014) incorporates probability of restoration failure by using regional reviews of past seagrass restoration success. The USACE Standard Operating Procedure for Determining Mitigation Ratios allows for a mitigation ratio adjustment to account for mitigation failure or underperformance due to permittee-responsible mitigation (rather than mitigation banks), modified or artificial hydrology, difficult to replace resources, and more (USACE 2016, USACE 2021). Ratio adjustment factors range from +0.1 to +0.3, resulting in a sum including multiple sources of uncertainty.

Another source of uncertainty in compensatory mitigation is out-of-kind mitigation, i.e., when an impact is mitigated for using a different habitat type. This may include the restoration of a rocky reef for impacts to soft bottom habitat; covering rocky reef with sediment may not be ecologically desirable or feasible. The CEMP calculator (King & Price 2004, NMFS 2014), Habitat Equivalency Procedure (USFWS 1980), and USACE Mitigation Ratio Checklist (USACE 2021) all provide options to adjust mitigation amounts due to out-of-kind mitigation.

Of course, ensuring equivalency relies on sound habitat valuation models in conjunction with rigorous monitoring and compliance efforts (Race & Fonseca 1996, Hough & Harrington 2019). Adequate remediation and performance criteria attention relies on proper funding and resource allocation to compliance and enforcement divisions of regulatory agencies.

Regulatory mandates can drive model framing and inputs

Just over half of the tools we identified were written or commissioned by government agencies (Figure 4) to serve specific regulatory needs associated with the protection and conservation of managed species and/or habitats. Many of the remaining models developed by academic institutions, especially from the European Union, were written to directly meet the requirements of the Water Framework Directive (WFD), which sets aquatic resource standards, including marine waters up to one nautical mile from shore. This significant policy influence over model design specifications can shape tool outputs and perhaps limits utility across agencies. For example, HSIs are commonly used to manage listed species that utilize SAV. While they may comprehensively guide decisions concerning a single species, they may not be the right tool for an agency more holistically concerned with a larger community.

Some models relied on "best professional judgment" to adjust ratios or other outputs depending on a variety of factors (e.g., habitat rarity, out-of-kind mitigation). Those adjustments, while bounded, were often left to project managers or analysts. Best professional judgment, while sometimes the only resource available for final management decisions, can lead to varying interpretations across professionals (Murray et al. 2016). This allows for mitigation decisions to incorporate unique details of the project but can also lead to inconsistent decisions and project applicants being unable to estimate mitigation requirements at the start of projects, effectively slowing the compensatory mitigation process (Kihslinger et al. 2020). However, well-documented best professional judgment decisions informing overall model design, including metric selection and weightings, can enhance consistency as compared to reliance on best professional judgment for every habitat evaluation or mitigation decision. In either case, carefully documenting the justifications for how metrics are selected, weighted, and used in model calculations can ameliorate lack of transparency, but may also add considerable time to the decision timeline.

Looking forward

No model is perfect, but if targeted to address the question at hand, such methods can help resource managers make decisions when faced with incomplete information. We identified the following strengths in tools from our database that could be incorporated into new or existing methods:

- Clear description of the tool's goal, objective, and scale
- Clear description of the 'ideal' or highest scoring habitat attributes, or guidance on reference site selection
- Detailed monitoring protocols that include best practices for survey conditions (time of year, tidal considerations, etc.)
- Description of model assumptions and consequences of assumption violations

- Transparency in weighting of metrics in valuation (i.e., if ‘best professional judgment’ was used, description of logic behind weightings)
- Options for metric inputs over time to capture temporal variation
- Landscape or seascape context, and/or region-specific features
- Incorporation or adjustments for sources of uncertainty

For practical application, the ‘ideal tool’ will depend greatly on the goal of the project, the scale of interest, the best science available to inform that goal, and all relevant regulations motivating the tool use.

HSIs, when not based on reference systems, relied on identifying optimal conditions. If models do not rely on reference conditions and, rather, choose pre-determined optimal values, there is a continued need to identify those ranges and provide the context under which they may occur. There is also a need for identifying thresholds that will play key roles in determining habitat changes resulting from anthropogenic climate change. Assigning mitigation should also include possible climate impacts that may influence the success of restoration or habitat quality in the future (Abelson et al. 2020).

Our review of the habitat quantification literature found a large gap in kelp valuation versus available tools for seagrass. With an increasing shift towards large scale monitoring with aerial and satellite imagery (Finger et al. 2021, McPherson & Kudela 2022; McPherson et al. 2022) and sonar technologies (Phinn et al. 2018), additional research connecting bed and canopy cover, other metrics, or valuation scores to SAV habitat functioning is also needed. However, basing value off just a few surface metrics may overlook important subcanopy and other community measurements.

We chose to exclude models that economically valued SAV, but there is a wealth of models based on non-market valuation and examining the value of ecosystem services, or functions, that directly benefit human society. In the 1980s and early 1990s, impacts to the environment were commonly assessed with methods to elucidate revealed preferences, which attempted to quantify how much money one was willing to pay to use or travel to a resource (Boyer & Polasky 2004). Some of these methods were replaced by the Habitat Equivalency Analysis (HEA; NOAA, 2000), but this should not discount the importance of including socio-economic impacts in injury and mitigation determinations (van Teeffelen et al. 2014; Unsworth et al. 2018).

While we identified over 24 metrics, further research is needed to connect metrics and habitat functioning to guide metric selection. Identifying metrics that can be measured rapidly and easily by consultants or agency staff and predictably relate to habitat functioning, will be integral in identifying feasible and ecologically meaningful models moving forward. A region-specific understanding of the relationship between metrics and an area's ecological value will continue to refine how complex SAV habitat, including systems that support managed species, can be valued.

Table 1. Definitions and related terms in ecological habitat valuation and quantification literature.

Term	**Definition (as used here)**	**Synonyms or related terms from literature**
Ecosystem function	An ecosystem’s energy and materials (e.g., biomass), fluxes of energy or material processing (e.g., productivity, decomposition), or stability over time (e.g. rates or stocks)	Sometimes included with “ecosystem services” (below)
Ecosystem services	The processes through which natural ecosystems, and the species that make them up, benefit society (e.g., storm protection; Daily 1997)	Sometimes included with “ecosystem function” (above)
Metric	A measurement or unit of an ecosystem (e.g., shoot density)	Measurement, trait, parameter
Indicator	One or more combined metrics that relate to some aspect of ecosystem functioning	Index, ecological valuation method
Habitat	The area where species or a community exists, uses resources, and interacts with other organisms	Habitats can be described at different scales: ecosystem, system, site
Habitat quantification tool	Methods for quantifying impact or mitigation for habitats (Chiavacci & Pindilli 2020)	Includes both valuation and equivalence methods (below)
Ecological valuation method	Procedure to assign a value to habitat, representing ecological quality or functioning (e.g., habitat suitability indices)	Ecological condition, functional assessment, suitability assessment, biological valuation, biocentric value, suitability index
Habitat equivalence assessment method	Procedure to evaluate losses and gains within ecosystems (Quétier & Lavorel 2011, Bezombes et al. 2017; e.g., mitigation ratios)	Ecological equivalency models, biodiversity offsets, scaling

Table 2. Strengths and weaknesses of the commonly used submerged aquatic vegetation (SAV) metrics in ecological valuation models considered here.

Metric (with context)	Strength	Weakness
Area Spatial extent of SAV (to be paired with some density metric to define extent)	● Can be mapped with satellite, aerial, sonar or field survey methods (Hossain et al. 2015, D'Archino & Piazzi 2021)	● Can be subjective if no density criteria (Norris et al. 1997) ● Naturally varies greatly over time (Munsch et al. 2023) ● Sensitive to abiotic factors ● May not consider patch and edge effects
Cover Percent of a defined area covered by tissue	● Quick measurement	● May be dependent on tide height
Density Number of stipes, shoots, holdfasts, etc. per area	● Clear metric (shoots or individuals per area) ● Can be used to collect concurrent reproductive data	● More time intensive than cover
Biomass Amount of tissue per area or volume	● More accurate description of abundance than area, density, or cover	● Possibly destructive sampling method ● Time consuming or reliant on coarse proxies ● May be skewed by life stages
Presence/absence Binary measurement of occurrence	● Quick measurement	● Not necessarily functionally relevant without some metric of relative occurrence
Diversity Species richness, evenness, beta diversity, etc.	● Can be described at the site or landscape level ● Can be described as richness or include measures of abundance	● Species identification can be time intensive (D'Archino & Piazzi 2021) ● Less functionally relevant for monospecific patches
Tissue content Concentration of elements or nutrients within tissue	● Responds to changes in relatively short time frames (Lee et al. 2004)	● Possibly destructive sampling method ● Chemical analysis can be costly and time consuming
Epiphyte cover Percent of tissue covered with epiphytes	● Can inform water quality (Giovannetti et al. 2010)	● Lack of consensus in literature on threshold of detrimental levels of cover (Borowitzka et al. 2006)

Figure 1. The three components of assessing impact and assigning compensatory mitigation. Metrics may be incorporated into a habitat valuation model to determine a score or indicator value for submerged aquatic vegetation (SAV; i.e., seagrasses and macroalgae), or a broader habitat that includes SAV (e.g., an estuary). Metrics can also feed directly into equivalency models, where the habitat value is not quantitatively defined.

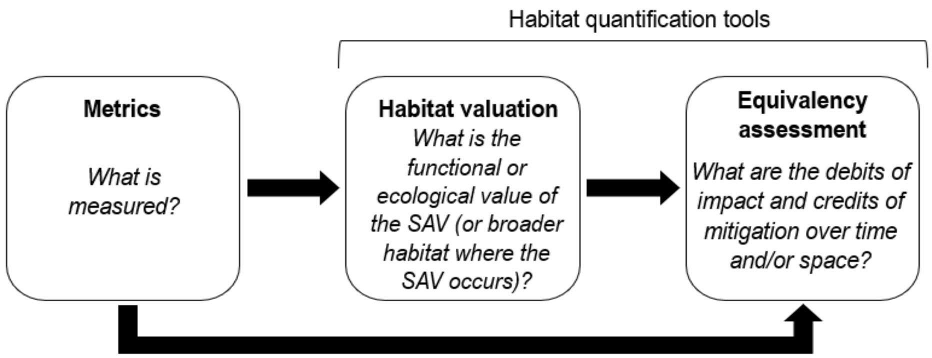


Figure 2. Flow chart to identify tools for specific management needs. Tools included methods for assessing specific SAV habitat or broader habitats, like wetlands, or habitat through the lens of a specific species like salmonids. Additional details for the models are available in Table A2.i. Note that the tools to calculate offsets for broader habitat could, in most cases, be applied to SAV (e.g., seagrass or kelp beds).

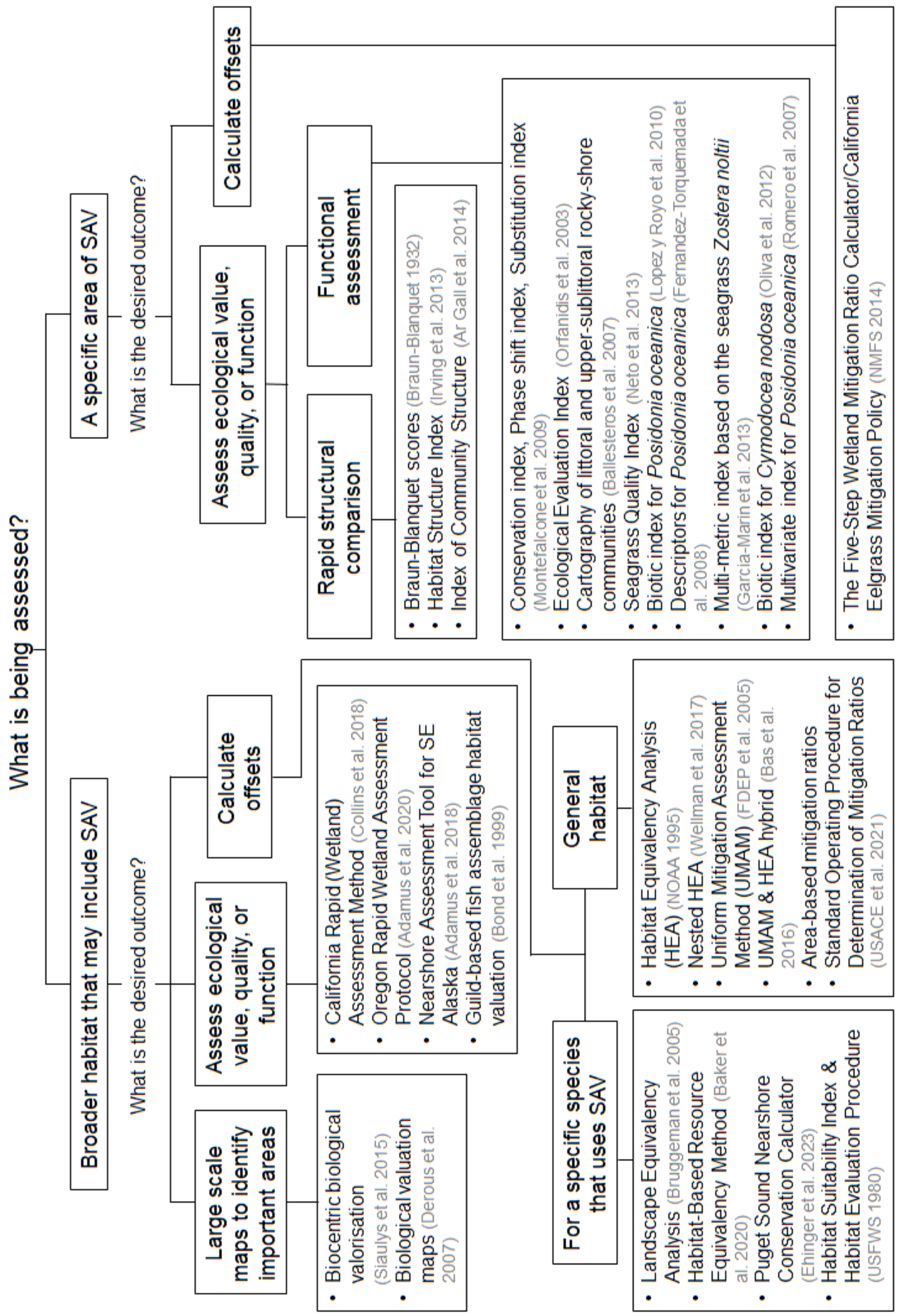


Figure 3. Percentages of tools that scored ecological valuation or calculated equivalency between impact and mitigation, or both.

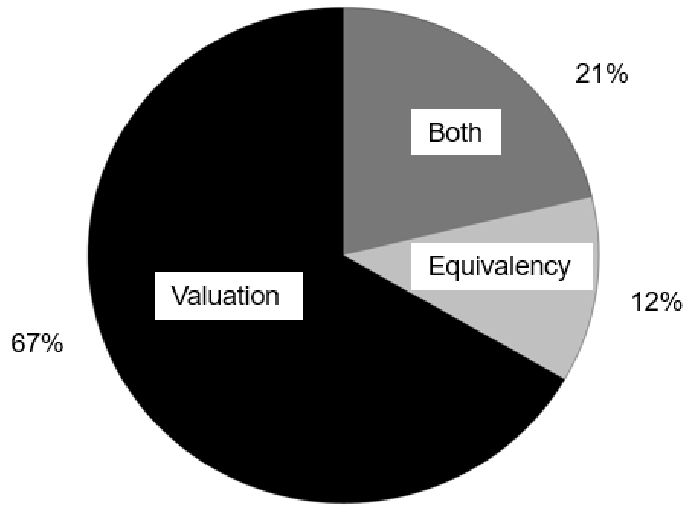


Figure 4. Distribution of the affiliation of the first author on model citations.

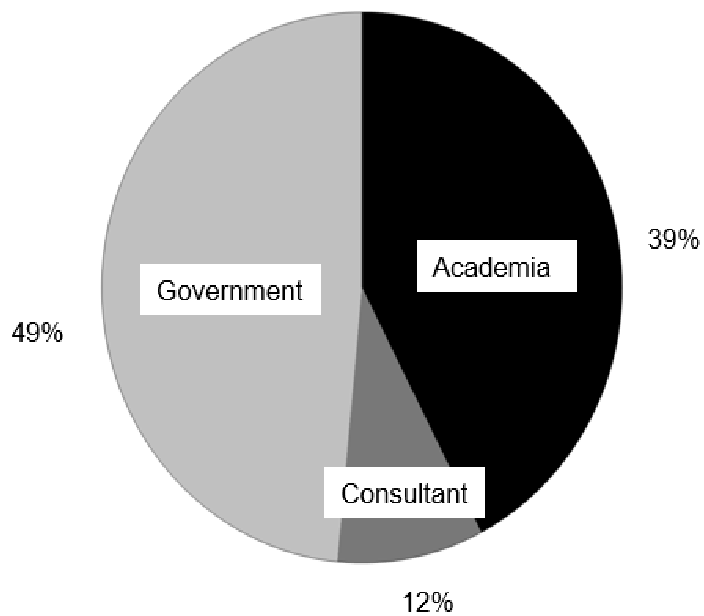


Figure 5. Frequency of SAV metrics used in broader habitat-wide (e.g., estuary and non-SAV species models) and SAV-specific tools.

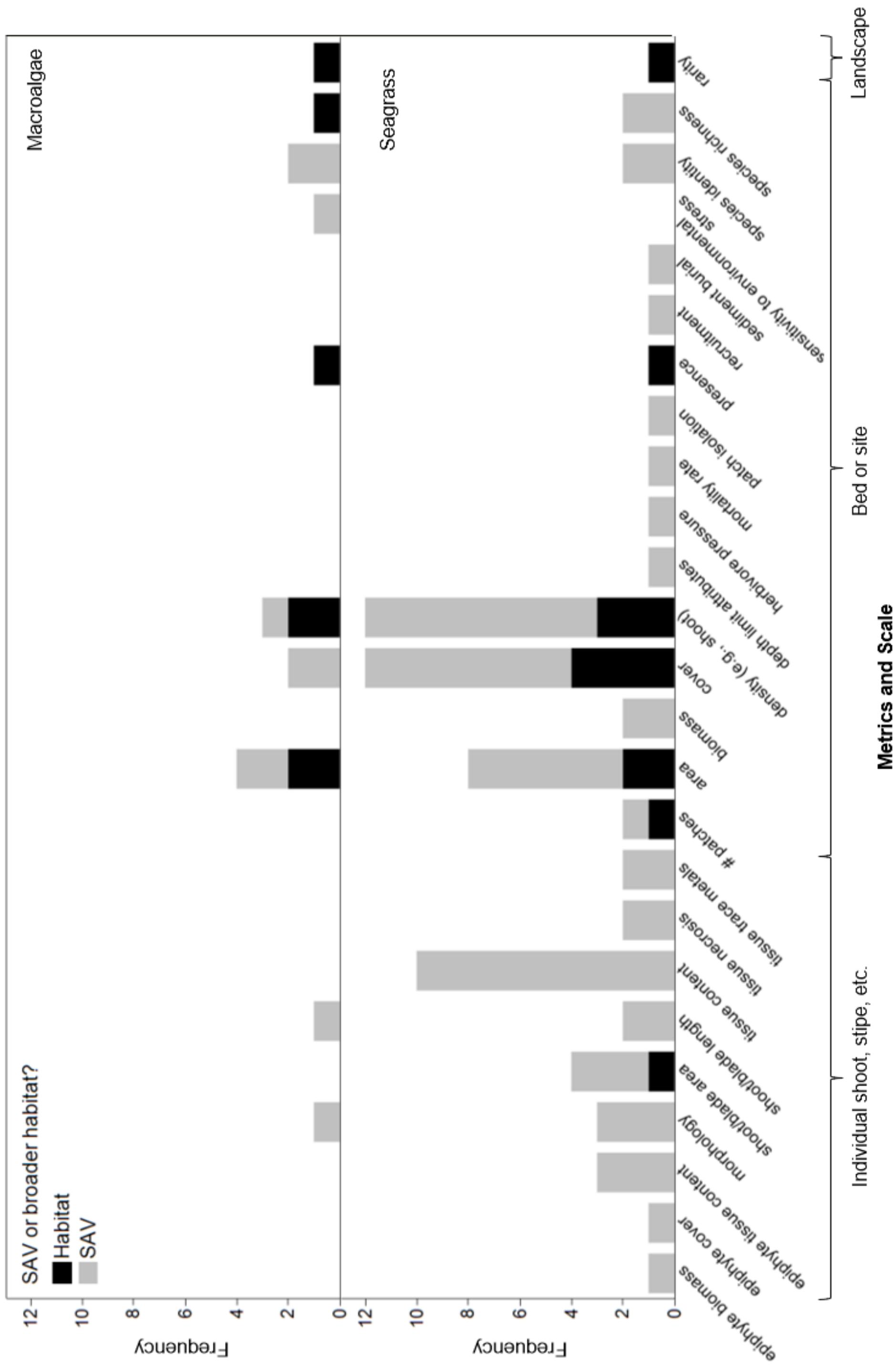
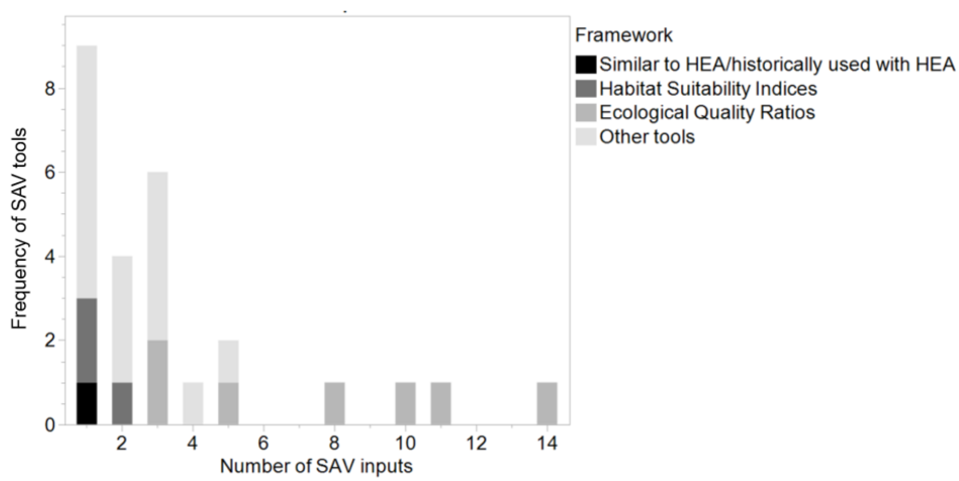


Figure 6. Number of SAV metrics used in a single tool within four identified major tool categories.



Chapter 3

Testing strategies to enhance transplant success under stressful conditions at a tidal marsh restoration

Abstract

Tidal marsh restoration is becoming an increasingly common tool to plan for future sea level rise. Understanding key actions to increase vegetation cover at restored, elevated marshes, and especially areas that remain persistently bare following construction, is a critical component of a project's long-term success. Dominant species can shape ecosystem function, as well as ameliorate stressful environments. We transplanted the dominant species, *Salicornia pacifica*, into bare areas of a restored tidal marsh in central California, USA, three years following a sediment addition. We tested salt hardening, targeted irrigation, transplant size, and planting configuration to identify management actions that could help vegetation persist in the most stressful areas of the high marsh. Weekly targeted irrigation until the first rains began was critical for small plant survivorship. We found that larger plants had increased survivorship and contributed higher amounts of growth and cover, but did not facilitate the performance of nearby smaller plants. After two years, we determined that using lone, larger plants was more cost effective than multiple smaller plants at our tidal marsh. However, performance was highly site-specific with dramatically less growth at a drier site with sandier soil. Our results highlight the importance of identifying site-specific restoration strategies that either ameliorate or

help plants tolerate stressful conditions, contributing to the continued success of tidal marsh restoration for climate resilience.

Introduction

Dominant species can play critical roles in shaping ecosystem functioning (Avolio et al. 2019) by serving as a foundation for the establishment of other organisms (Ellison 2019), significantly contributing to productivity (Smith & Knapp 2003) or reducing invasions by competitively dominating space or reducing allelopathic interactions (Hillebrand et al. 2008). Dominant species, especially those that become established early in succession, can also play a role in ameliorating environmental stressors (Grime 1977). The stress gradient hypothesis (SGH) predicts increased positive interactions and facilitation among plants in areas of high stress such as saline patches of soil, benthic habitats with low nutrients, highly grazed areas, etc. (Bertness & Callaway 1994). While the SGH has been demonstrated to be most applicable for plants with specific life histories and in harsh environments (Maestre et al. 2009; He et al. 2013), the core concepts have been leveraged by the restoration ecology community through the use of cover species and nurse plants (Gómez-Aparicio et al. 2004; Padilla & Pugnaire 2006; Silliman et al. 2015). Improved understanding of intraspecific facilitation via a dominant species and the manipulation of abiotic conditions could enhance habitat restoration and aid in the remediation of underperforming restored areas.

Tidal marshes present a stressful environment for plants but often support high levels of productivity driven by the cover of dominant halophyte species (*Salicornia* spp., *Spartina* spp., etc.). They are dynamic habitats, affected by tides, freshwater inputs, and sediment deposition. They can serve as nurseries for marine and estuarine animals, carbon sinks, and essential habitat for rare species (Barbier et al. 2011; Himes-Cornell et al. 2018). They contribute to water quality and buffer inland areas from coastal hazards (Möller et al. 2014; Adams et al. 2021). Despite the services provided, 85% of vegetated tidal wetlands have been lost on the west coast of the United States (Brophy et al. 2019). Most loss is attributed to hydrologic alterations, including wetland fill, diking, and draining for agricultural or extractive uses (Gedan et al. 2009). The subsequent decrease in soil moisture leads to nearly irreversible compaction and subsidence of the soil, even if tidal waters are reintroduced (Van Putte et al. 2020). Marshes degraded due to fill or subsidence result in artificially raised or lowered marsh plains. Areas elevated beyond tidal influence can be overtaken by ruderal upland species that outcompete native halophyte species, while areas that have subsided and flooded may convert to barren mudflats. Restoring marsh vegetation often requires the restoration of the appropriate elevation to support these species.

Tidal marsh restoration to reverse habitat loss has been attempted since the 1970s (Callaway et al. 2011). Marsh elevation can be actively restored by removing sediment in elevated areas or adding sediment to subsided areas. Both processes often involve recontouring of the area, leaving large areas of exposed bare earth. The available seed bank within the new marsh plain varies depending upon whether sediment was added or excavated. Previous studies have not found remnant seeds to be a significant source of emergent seedlings (Morzaria-Luna & Zedler 2007). While it is possible to rely on the natural establishment of vegetation from nearby populations, the succession of bare earth to a fully functioning tidal marsh can take decades, if not longer (Callaway 2005; Garbutt & Wolters 2008; Burden et al. 2019).

The slow trajectory of tidal marsh recovery from bare earth presents multiple challenges. From a policy perspective, restoration undertaken as compensatory mitigation for development impacts is often held to performance standards on the scale of years rather than decades. Ecologically, active restoration can represent a large area of severe disturbance. If cover is not established rapidly, overtopping tides and subsequent evaporation can result in a feedback loop leading to increasingly stressful conditions. Such degradation can result in closing a likely "window of opportunity" for natural recruitment (Suding et al. 2004; Balke et al. 2014). To address this problem, practitioners often plant into bare areas (Rabinowitz et al. 2022) rather than waiting for the natural deposition of seeds or vegetative fragments to establish. This same strategy is also utilized in the remediation and adaptive management of persistently bare restored areas.

The success of planting in bare restoration areas can potentially be enhanced by a variety of applications of ecological theory. One strategy is to apply the SGH when designing spatial planting patterns. Neighborhood effects and clustering designs have been examined for both interspecific and intraspecific facilitation between plants (Silliman et al. 2015; Duggan-Edwards et al. 2020; Shaw et al. 2020), showing clustering can improve performance in stressful areas. Actively ameliorating localized stress with other actions like short-term irrigation has shown mixed success in facilitating the growth of transplanted plants (Porensky et al. 2014; Xie et al. 2019; Beheshti et al. 2023). As an alternative to ameliorating stress, practitioners can also utilize plants that have a higher tolerance for stress, targeting specific species (Armitage et al. 2006) or larger planting container sizes (Herriman et al. 2016).

The goal of this study was to develop and test strategies to enhance restoration success in persistently bare areas of a tidal marsh restoration. We transplanted a dominant halophyte species (*Salicornia pacifica*; perennial pickleweed) into areas of a tidal marsh in central California that had remained bare for almost three years following a sediment addition. We tested four strategies to enhance salt marsh restoration success: salt hardening of plants prior to transplanting, varying the size of initial plants, irrigation, and using planting configurations affecting potential facilitation. We also investigated how restoration success varied across site differences and associated sediment properties. The findings of this study can be used to accelerate the success of other tidal marsh restoration projects, which will become more common as coastal managers look to tidal marshes as a tool for coastal resilience.

Methods

Site

We transplanted a dominant perennial species, *Salicornia pacifica* (pickleweed), into Hester Marsh, located approximately 2 km inland of the mouth of Elkhorn Slough in Monterey County, California (Figure 1). Decades of diking, agricultural practices, and lack of sediment input had resulted in the subsidence of the marsh plain and conversion to mudflat with a proliferation of algae. In 2018, 61 acres (25 hectares) were elevated with sediment sourced from a local river dredge project and material from surrounding uplands, with regrading of the existing marsh to expand and raise the marsh plain an average of 69 cm. As of 2020, the marsh plain averaged 1.89m NAVD88 in elevation, considerably higher than Mean Higher High Water (MHHW) which is at about 1.75m NAVD88 locally (Fountain 2020). This elevation was chosen to be higher than surrounding marshes to account for post-construction compaction of sediment and projected sea level rise. The high elevation resulted in the marsh plain being inundated 1-3% of the year (Thomsen 2020), representing a drier environment than nearby reference marshes. Due to *a priori* observations and hypothesized differences in sediment source and stressors (Thomsen et al. 2022), the study area was divided by a main north-south channel into "east" and "west" areas ("sites"; Figure 1). Three years following the restoration project, some areas away from tidal channels remained bare for poorly understood reasons (but see Thomsen 2020). The east site was especially devoid of vegetation after transplants and natural recruits failed to establish, indicating a more stressful environment. We chose bare areas in the east and west sites for our transplant experiments, and blocks were evenly divided across the two sites at a constant elevation to facilitate the comparison of site differences.

Transplanting design

In July 2020, pickleweed plants were uprooted from an adjacent, subsided area planned for a future sediment addition project (Figure 1). They were rinsed of marsh soil and transplanted into different size containers with potting soil. Smaller plants with at least one rhizome node were planted into 13 cm tall x 3.8 cm diameter conical planting plugs, from here referred to as “plugs.” Other larger plants, with more tissue and multiple rhizome nodes, were transplanted into gallon pots, measuring 16 cm in diameter and 17.5 cm tall (“pots”). Plants were kept in a greenhouse over the fall and winter until roots established within the pots. They were watered with freshwater until about two months before transplanting, when they were watered with increasingly saline water (using Instant Ocean; Spectrum Brands, Blacksburg, VA), up to 35 ppt. All plants were transplanted to the site in March 2021. The average pickleweed tissue volume of plugs at the time of transplanting was 922 cm^3 and pots averaged 16,193 cm^3. July 2020-June 2021 was a drought year, with only 70% of the previous 9 years’ average rainfall. The July 2021-June 2022 year was an average rainfall year, and winter 2022-2023 was especially wet.

Salt hardening

We kept 24 plug replicates watered with freshwater, rather than salt hardening them. We then planted the non-salt hardened plugs and an equal number of salt hardened plugs at both the east and west sites of the marsh at similar elevations (1.8m-1.9m NAVD88; Figure 1).

Size and planting configuration

In a separate transplant size comparison experiment, eighteen blocks were evenly divided across the two sites, at elevations between 1.8m and 1.9m NAVD88. Each block contained three different planting configurations of plugs and pots (i.e., 3 "plots"): a single pot ("lone pot"), three plugs (planted far enough away to be considered "lone plugs"), and a combination plot with a single pot closely surrounded by three plugs ("pot with plugs"; total study N=144 plants; Figure 2). For some analyses, we analyzed individual plants and used treatments based on both planting configuration and size, dividing the combination plots into the individual plugs ("plugs with pot") and the pot plant ("pot with plugs").

Each of these plants was subsequently watered with freshwater from the initial planting (March 2021) until the first rain of the fall began (October 2021). Plants were watered with a water can with approximately 0.6 L of fresh water twice a week unless the tide had recently overtopped the marsh plain and the soil was still moist.

Sediment sampling

After watering had ceased for a full week in October 2021, we sampled the sediment directly adjacent to plants. At each plot we extracted a 13 cm-deep core, avoiding potting soil in the plot. We calculated the gravimetric water content and bulk density of soil between 8 and 13 cm depth, thought to be characteristic of rooting conditions. Gravimetric water content was calculated by drying samples for 48 hours in a drying oven at 110°C and dividing the weight of the lost moisture by the dry weight. The bulk density was calculated by dividing the dry sample weight by the volume of the core. For a subsample of plants (pots), we analyzed the grain size fractions across sand, silt, and clay content of the rooting depth sediment samples (8-13 cm) using the hydrometer method (California Department of Transportation 2008).

Plant performance

We tracked plot cover, individual plant survivorship, and plant dimensions. For dimensions, we measured a perpendicular maximum length and width, as well as a perpendicular maximum height, and considered their product a measure of individual plant volume. We analyzed the growth rate as the change in plant volume over 30 days. Over the summer of 2021, we noted the presence of flowers as an index of sexual reproduction as well as insect feeding scars as an index of herbivory. Mammal herbivory seemed to be minimal and was not measured. Individual plants were monitored from March 2021 to October 2022, and plot cover for each configuration was monitored through March 2023.

We measured the area of pickleweed cover added to plots after one (March 2022) and two years (March 2023) to provide a cost comparison of each planting treatment, relative to area output. Area was calculated by measuring cover in a $0.25m^2$ quadrat, which contained the plants except for a few rhizomes that extended beyond the border of the quadrat in October 2022 and March 2023. Dead plants were included in survivorship and cover calculations but excluded from individual plant growth calculations.

Statistics

Statistics were run in R version 4.3.0 (R Core Team 2023). Survival curves were compared across watering, configuration treatments, and site using a mixed effects Cox regression model (coxme R package, v. 2.2.18.1; Therneau 2022) with plot as a random factor. Pairwise comparisons were made using Holm pairwise comparisons. Survival curves were created with the survminer R package (Kassambara et al. 2021).

The change in plant volume was analyzed over time across size classes, planting configuration, and site with a linear mixed model (lme4 R package, v. 1.1.32; Bates et al. 2015). Block and individual plants were designated as random factors. Plot cover was compared similarly, but without blocks, and using a generalized mixed model using a beta distribution. Binary presence or absence of herbivory and reproduction were compared using a generalized mixed model (lme4) with a binomial distribution and logit link.

Results

Salt hardening and watering

Watering dramatically increased the survivorship of salt hardened plugs ($p < 0.001$; Figure 3A; Table A3.i); site was not a significant factor ($p=0.160$). Watered, salt hardened plugs showed 65% survivorship after 18 months. Non-watered plugs had 0% survivorship within less than 100 days, regardless of whether they were salt hardened or not ($p=0.920$). Plants that were continuously watered through the summer showed variable signs of excess salt accumulation on the soil surface, but this did not seem to be correlated with survivorship.

Plant sizes and configuration: survivorship

Survivorship for pots was greater overall than for plugs (Figure 3B; Table A3.ii). Pots with plugs did not have significantly greater survivorship than lone pots ($p=0.683$). Similarly, plugs near pot plants did not have significantly higher survivorship than lone plugs ($p=0.683$; $p=0.350$ between March and October 2021, while watering was ongoing; Figure 3B). There was slightly higher survivorship at the west site ($p=0.076$).

Plant sizes and configuration: individual growth

There were significantly higher rates of growth for both pots and plugs at the west site than the east site (pot: $X^2=4.31$, $p=0.038$; plug: $X^2=9.18$, $p=0.002$; Figure 4).

Growth rates did not vary between plugs that were grown with a pot plant versus plugs grown alone (X^2=0.343, p=0.558). There was also no evidence that pot growth with plugs varied from lone pots (X^2=0.617, p=0.432). Growth rates did vary over time (plug; X^2=6.23, p=0.013), with less growth over the winter.

Plant sizes and configuration: plot cover

One year after planting, west plots with 3 lone plugs had added an average of 156 cm^2 of vegetation (6% of plot ± 3.9% SE). Pots with plugs grew an average of 608 cm^2 (24 ± 5.7%) and lone pots added an average of 799 cm^2 (32 ± 7.1%). After two years the total growth averaged 451 cm^2 (18% of plot ± 7.9% SE), 694 cm^2 (28 ± 7.9%), and 937 cm^2 (38 ± 9.7%) for lone plugs, pots with plugs, and lone pots, respectively. Because each treatment began with plantings, their total cover was higher than growth alone (see Figure 5; cover before March 2021 was 0%). Overall, the west site had greater plot cover ($X^2(1)$=5.27, p=0.022; Figure 5), while the east site exhibited very little change in cover over 24 months (Figure 5), in some cases only maintaining the initial cover from plantings. Planting configurations differed in cover as well ($X^2(2)$=206.0, p<0.001). However, there was no significant difference between pots and pots with plugs (p=0.748). The interaction between site and time was significant (p<0.001), with more growth during the summers at the west site.

Cost vs. area estimates

Table 1 shows the cover of pots with plugs and lone pots as factors of the lone plugs' cover at each site (i.e., multiples of area after one and two years). This table can be used to determine if using a certain treatment is cost-effective. For instance, after two years, if a single pot plant does not cost more than 2.1 times the cost of 3 plugs, it is still more cost-efficient to use pots, as the cover added to the plot was 2.1 times that of the lone plugs. In 2022 we contacted a wholesale nursery that routinely supplies plugs and pots for restoration. They provided an estimate of $3 per plug and $4.50 per pot. Because the pot with plugs treatment did not produce significantly more cover than the lone pot, the lone pot treatment was the most cost-effective for increasing cover.

Reproduction & Herbivory

Planting configuration was a significant factor in the presence of flowers (as an index of reproduction) across plugs and pots, ($Z=2.31$, $p=0.02$; $Z=-3.21$, $p=0.001$, respectively). At the west site, plugs planted with a pot plant showed a greater presence of flowers than lone plugs. Lone pots flowered more than pots with plugs on the east side. Site was also a significant factor for the presence of flowers for pot plants ($Z=2.13$, $p=0.03$). Flowering peaked at the west site in late August and early September, but not a single flower was observed on plugs on the east side between July and October. The presence of herbivory scars varied over time, with periodic increases throughout the summer, but was not significantly different across treatments or sites.

Site sediment differences

There was greater soil moisture in the west side plots, even after watering both sites ($t(49)=-4.69$, $p<0.001$; Figure 6). The mean soil bulk density was 1.77 g/cm^3, indicating highly compacted soil conditions, but this was not significantly different across sites. The percentage of sand fraction of sediment samples was greater on the east side (71.6% vs. 57.8%; ($t(17)=-4.13$, $p=0.001$; Figure 6). Clay and silt fractions were less variable than sand fraction across sites (Figure A3.i).

Discussion

Tidal marsh restoration can create stressful, bare environments, and continual overtopping tides in exposed areas can create a positive feedback loop that maintains those conditions. Our east site seemed to be especially stressful for plants, but we did not find clear evidence for the SGH predictions that clustered plants would perform better. Spacing out larger pot plants of our dominant species and using targeted irrigation led to increased cover and survivorship over smaller plants, and was more cost-effective than pots surrounded by plugs.

Role of facilitation

Positive interactions and facilitation via clustered planting patterns have been proposed to improve restoration outcomes in tidal marshes and across ecosystems (He et al. 2013, Renzi et al. 2019). Silliman et al. (2015) also found increased growth and stem densities of clumped configurations of plants in the low marsh, which was attributed to “oxygen leak” in ameliorating soil anoxia, and erosion protection.

Reijers et al. (2019) associated another clustered marsh species' persistence in high salinity areas with increased root oxygenation, as well as increased rain infiltration at higher plant densities. While we did see a facilitative effect in the flowering of smaller plug plants, we did not find significantly increased growth of plugs when planted with a pot versus alone. There did seem to be slightly increased survivorship for plugs with pots early on in 2021, but this did not persist. One explanation for this may be that pot plants helped to retain the targeted irrigation better than lone plugs leading to enhanced initial survival, but this was only advantageous while watering was ongoing during dry intervals between overtopping tides.

Cases where lone pots grew more than pots surrounded by plugs could suggest competition induced by clustering plants. In the ecotone directly adjacent to our study site, Tanner et al. (2022) reported hindered growth of clustered plants, rather than positive facilitation effects. Due to the increased cost of planting additional plugs around a pot, the pots provided a clear best option for cost-effective cover. However, our conclusions rely on the assumption that our costs presented are the true expenses of such planting treatments. They did not include the cost of installation or transportation. Thus, we provided Table 1 which shows performance advantages as multiples of cover gained. Another assumption is that two years is a meaningful timeline for comparison. For compensatory mitigation restoration projects, this may be true, as projects may be held to strict performance standards over short time periods (Matthews & Endress 2008). However, if a project is planned so that monitoring will occur over the first five years rather than two, it may be that the west site's lone plugs would eventually surpass the cost-benefit advantages of the lone pots. We highlight, though, that the east side's particularly stagnant performance seems unlikely to improve over time, pending a major rain event or other disturbance (Allison 1992).

Transplant size

Plant size played a significant role in survivorship of individual plants. Plots with pots had higher survivorship, grew more, and were more cost-effective in establishing cover after two years. We suggest possible mechanisms at work here. First, larger plants were more resistant to the stresses of the bare marsh plain. Larger plants provide more shade to the soil below, reducing localized heat stress more than smaller plants (Jagadish et al. 2021). Other possible mechanisms could be that the greater volume of pots led to more decompaction when transplanted, or the potting soil supplied more amenable rooting conditions than the marsh plain soil. Soil translocation from reference sites has had mixed, site-dependent success (Gerrits et al. 2023). One could assume that potting soil would provide as ideal conditions as possible for a transplant, as well as increased water retention capabilities. Considering that the placed sediment at our restoration site was largely sandy and both sites had high soil bulk densities, additional potting soil could have led to greater transplant success. This could be tested in a future study by adding a greater volume of potting soil with plug plants, essentially serving as a soil amendment action.

Additional physical factors affecting plant restoration

Restoration success can largely depend on restoring appropriate physical conditions for the establishment of target species. Moisture played a key role at our elevated marsh. Our tidal marsh represented a uniquely elevated design for future sea level rise, and thus exhibited drier conditions than nearby areas and the low marshes of other studies. While we did not test non-salt hardened plants in our irrigation study, the lack of irrigation proved to be impossible for plants to overcome, with 100% mortality in just under 3 months. Targeted irrigation has historically aided restoration efforts in high tidal marshes (Beheshti et al. 2023) and other stressful habitats: deserts (Abella et al. 2015), mine exploration areas (Elliott & Turner 2021), seasonally dry oak forests (Badano et al. 2009), etc. Based on the significant differences in performance with watering and the higher moisture at the west site, our plant performance may be limited by available moisture, or a stressor ameliorated with increased moisture, such as salinity. This idea is supported by the previous work at our site by Thomsen et al. (2022), who found slightly higher soil salinity in poorly vegetated areas.

Scaling irrigation presents logistical challenges and resource conflicts, as water is often highly managed and can be scarce in the landscapes where it is most needed. Some restoration efforts have looked to increasing tidal creek density (Van Putte et al. 2022). Others have attempted seawater irrigation input using diversion canals from natural channels, but seawater often does not ameliorate stress as effectively as freshwater (Xie et al. 2019). While we realize that irrigation is rarely a feasible solution at certain scales, our study highlights two key aspects. First, watering through the first year significantly decreased plant mortality (100% of unwatered plants died), and mortality the second year without watering was less or equal to the first year for all but one of the treatment groups (plugs with pots). Considering the transplants were planted during an especially dry period, this highlights that one season of irrigation can serve as an intervention during periods of intense stress. Second, identifying water as a limiting factor to establishing cover can suggest actions for future pilot studies that could help increase soil moisture content. For instance, increasing surface heterogeneity at a restoration site can help retain moisture (Moser et al. 2009), as can adding organic soil amendments to sandy soils (O'Brien & Zedler 2006, Ozores-Hampton et al. 2011), a strategy that could have contributed to the success of the larger plants in our study (i.e., more potting soil). Further analysis of other abiotic factors, organic matter, and microbiome conditions is needed within these bare areas, especially at the east site, which seemed to be especially stressful based on the lack of plant growth and absence of flowering. Additionally, we did not test a combination of watered, non-salt hardened plants, and it is possible that salt hardening is not necessary when paired with irrigation.

In addition to lower moisture, the east site soil also consisted of higher sand content, where some plants failed to grow or reproduce. This could be correlated with different sediment sources, as river-dredged sediment was used for a larger portion of the east site, and adjacent hillside sediment was used for a large portion of the west site. However, the composition is extremely heterogeneous, as hillside soil was placed by the truckload (Thomsen et al. 2022). Sandier soils are often correlated with lower soil moisture, higher permeability, and low organic content (Usowicz & Lipiec 2017) and slow the establishment of functioning at other tidal marsh restoration projects (McAtee et al. 2020). Thus, it is plausible that the dredged sediment resulted in a more stressful environment. Although the bulk density was not significantly different across our sites, the average bulk density of our site (1.77 g/cm3) was significantly higher than many other published bulk density values published for similar systems (O'Brien & Zedler 2006, Beheshti et al. 2023). This suggests that a common stressor in these bare areas may be soil compaction, and warrants future investigation of decompacting measures, either through soil amendments or disturbance (Beheshti et al. 2023).

Management implications

Overall, we were able to establish cover in persistently bare areas, although this was limited on the east side where conditions seemed more stressful, perhaps due to sediment differences. We suggest the following management actions to increase the likelihood of project success:

1) Conduct pilot studies or phased plantings across areas with observable differences to inform future performance.

2) Irrigation can be used to improve survivorship during especially dry years; if irrigation is not feasible, look for ways to naturally retain existing moisture.

3) If attempting to increase cover within stressful environments during the first years following a restoration project, use spaced, larger plants.

Our results do suggest that irrigation and larger plants can increase survivorship and cover in the short-term. Identifying site-specific restoration strategies that either ameliorate, or help plants tolerate, stressful conditions, will contribute to the continued success of tidal marsh restoration for climate resilience.

Table 1. The amount of cover produced by planting configurations as a factor of the plot with a triad of lone plugs after one and two years. A quoted cost for each treatment is provided. If the multiple of cover provided exceeds the multiple of cost, it is more cost-efficient to use that treatment. For example, at the end of two years, the west site pots averaged 1.8x the cover as the lone plugs, but cost 0.5x as much, so it is more cost-effective to plant one lone large plant (pot) than three small plugs.

	Cost	West site		East site	
		Year 0-1	Year 0-2	Year 0-1	Year 0-2
Lone plugs (3)	$9.00 (1.0x)	1.0x	1.0x	1.0x	1.0x
Pot	$4.50 (0.5x)	3.5x	2.1x	4.0x	2.4x
Pot with plugs (3)	$13.50 (1.5x)	3.0x	1.8x	4.3x	2.8x

Figure 1. Hester Marsh in 2020, six months before transplanting. Blocks were distributed between the east and west sites, divided by the main north-south channel. Transplanted pickleweed was originally harvested from the northwest corner of the west site, which was later covered in sediment in a subsequent phase of restoration.

Figure 2. Three configuration treatments (plots) transplanted at both the east and west sites. Smaller plugs were 13 cm tall and larger pots were 17.5 cm tall. The average pickleweed volume of plugs at the time of transplanting was 922 cm^3 and pots averaged 16,193 cm^3.

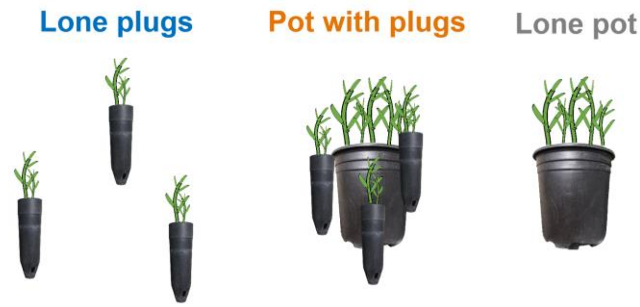


Figure 3. Survival curve for salt hardening and watering treatment plugs over time (A). Letters denote pairwise comparisons between treatments. Survival curve of individual plants within various planting configurations (B), including lone plugs and pots, and pots surrounded by plugs. P values and 95% confidence intervals are provided. Letters denote significance (p<0.05) across pairwise comparisons between treatments.

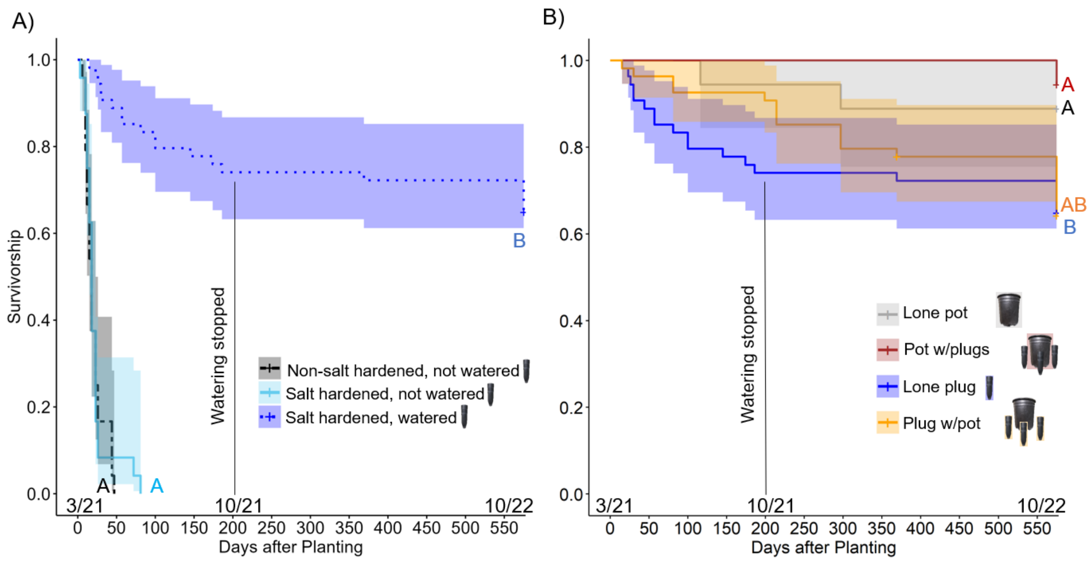


Figure 4. Growth (volume change per 30 days) for all plugs (left) and pots (right) over time and across site. Planting configuration was not significant, and so plugs and pots were pooled. Growth among the same size treatments (pots and plugs) was significantly different across sites (p<0.05).

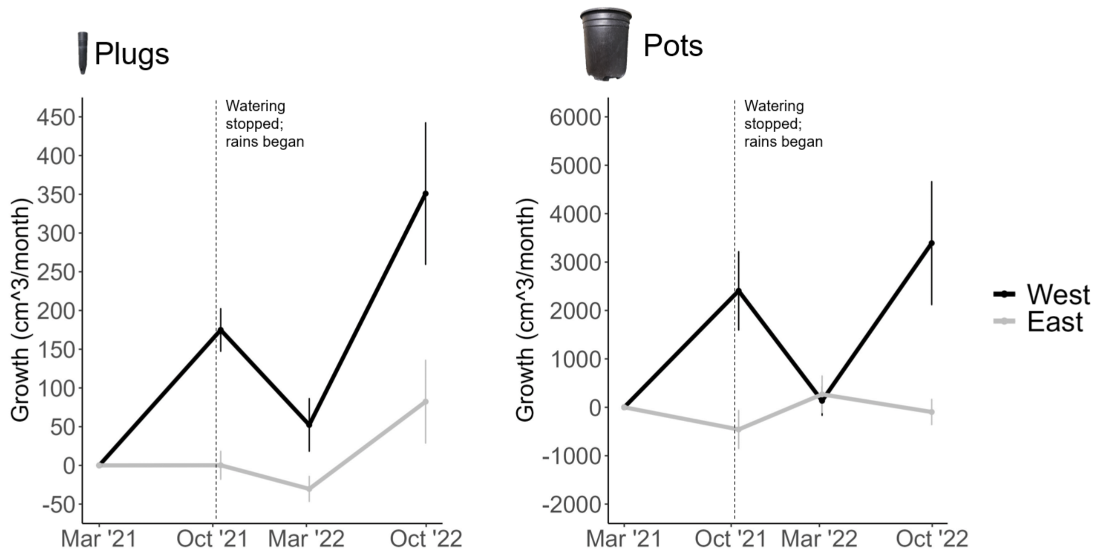


Figure 5. Plot percent cover through time for each planting configuration at west and east sites. Cover was 0% before March 2021. All treatments were significantly different ($p<0.05$) except for the lone pot and lone pot with plugs treatments at the east site.

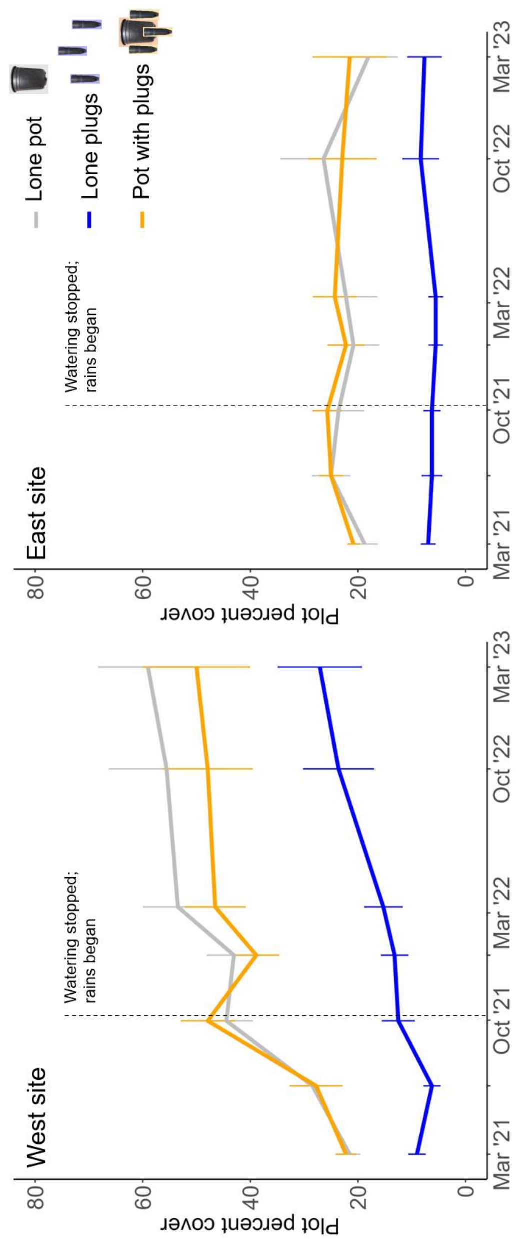
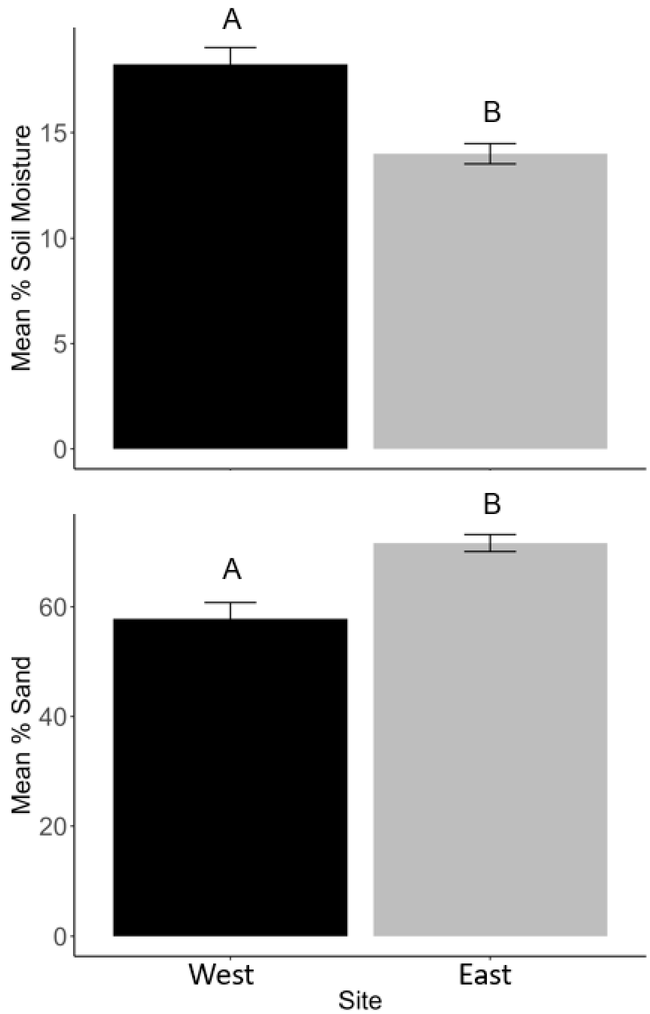


Figure 6. Mean gravimetric soil moisture content (top) and mean percent sand fraction (bottom) of sediment samples taken at 8-13 cm. Letters denote significance.



Conclusion

The mitigation hierarchy (Arlidge et al. 2018) clearly prioritizes avoidance and minimization of impacts before the implementation of compensatory mitigation. However, the reality of continued development along the coast highlights the importance of understanding the cumulative impacts of compensatory mitigation on the landscape, methods of quantifying impact and mitigation, and ways to accelerate restoration performance, especially those projects that do not meet performance standards following construction. I have examined all three of these topics with common threads that unite each chapter.

One aspect relating these three topics is the power of cumulative impacts. While projects on the scale of tenths of square meters may appear to have negligible impacts, the review of the Coastal Commission's compensatory mitigation programs showed that small projects, no matter how they are quantified, can add up to large impacts over time (Stein & Ambrose 1998). Likewise, small mitigation projects add up over time, contributing to overall habitat conservation (Thorne et al. 2014), and ideally, increased ecosystem functioning and resilience.

While ecological principles can be leveraged to improve restoration outcomes (Palmer et al. 1997; Dickens & Suding 2013; Sommer et al. 2023), performance of mitigation and other restoration projects is highly site-specific (Zedler et al. 2003). Whether through the application of site-specific submerged aquatic vegetation (SAV) valuation models, regulatory decisions regarding appropriate mitigation ratios, or restoration actions to jump start vegetation cover, site-specific factors can heavily influence success or relevance of certain methods. While this certainly presents a challenge, it also highlights the importance of pilot studies for restoration, as well as finding robust metrics and indicators that can be applied across a range of habitat values.

My restoration work at Hester Marsh demonstrates the capability of dominant species to establish in even the most stressful areas of restored tidal marshes. Dominant species can play critical roles in shaping ecosystem functioning (Avolio et al. 2019) and by serving as a foundation for the establishment of other organisms (Ellison 2019). While I did not study dominant species specifically in my other chapters, important foundational species are often used to define habitat types, making them critical to the study of habitat mitigation.

My work aimed to take a lens to maximize efficiency or cost effectiveness for restoration and mitigation actions. My work on the Coastal Commission's mitigation program showed that applicants preferred in-lieu fees, and transparent frameworks for mitigation decisions (i.e., area-based mitigation ratios) can streamline decisions for large programs spanning the entire state coastline. SAV valuation models can measure over a dozen metrics, but ultimately models that used just a few metrics were more common, and simpler models have been cited as more effective (Gregr et al. 2019).

The work presented here highlights several avenues of future research that would benefit the compensatory mitigation community. First, there is a dire need for additional studies connecting habitat metrics to function (Vieira et al. 2018). While cover and area were common metrics, I found few studies describing the relationship between those structural components and ecosystem functioning or a habitat's ability to support a managed species. Additionally, there is a need for continued attention for understudied ecosystems in the compensatory mitigation field (e.g., kelp) to identify the most efficient quantification tools and restoration methods.

Further work is needed to study the performance of restoration projects, not only during the common 5 year timeline for mitigation, but for the long-term performance of restoration projects (Fong et al. 2017), especially for those built for climate resilience, like Hester Marsh at Elkhorn Slough. Identifying adaptive management actions like those I identified for establishing vegetation in the bare areas of Hester Marsh will be key in ensuring that restoration projects reach full functioning and are fully compensatory when serving as mitigation.

Improving the compensatory mitigation process and effectively planning mitigation for a changing climate will be crucial in guaranteeing that lost resources replaced by mitigation are in fact conserved over time for present and future generations.

Appendix 1
Supplementary figures for Chapter 1

Figure A1.i. The California Coastal Zone consists of approximately 1100 miles (1770 kilometers) of coastline, including the coasts of the Channel Islands. The San Francisco Bay falls under the jurisdiction of the San Francisco Bay Conservation and Development Commission (established in 1965 before the California Coastal Act) and is not included in this length estimate.

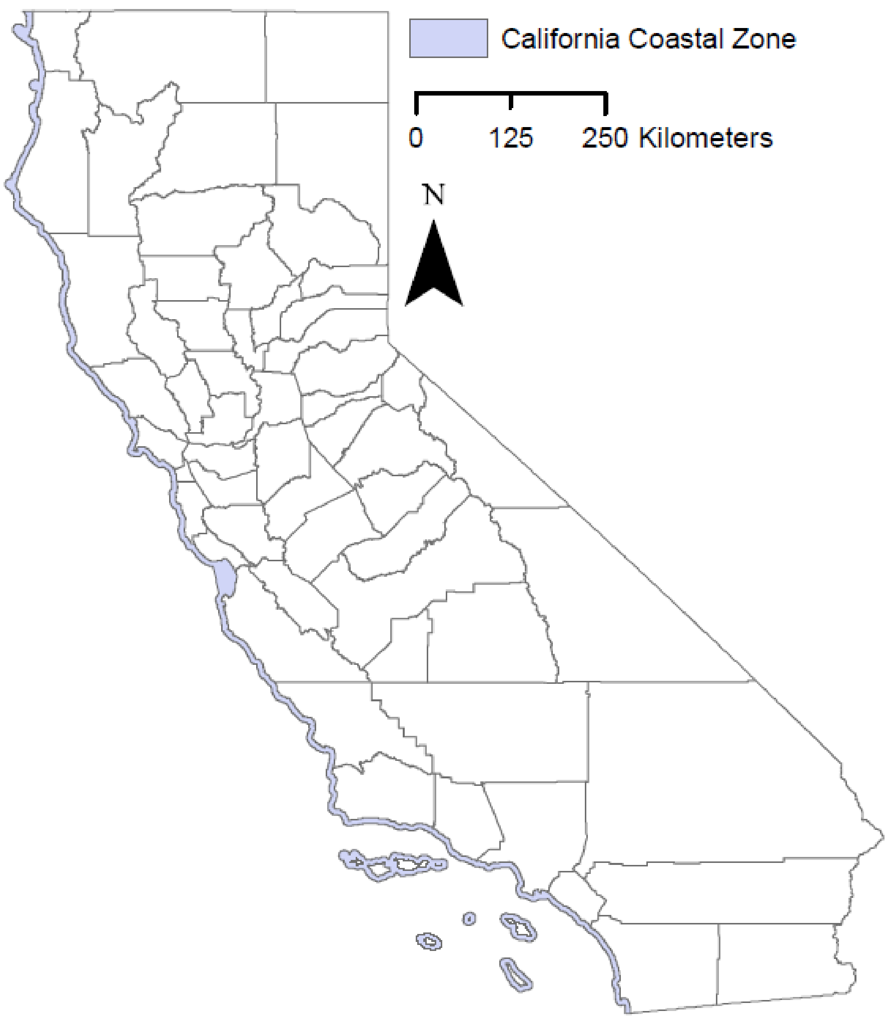
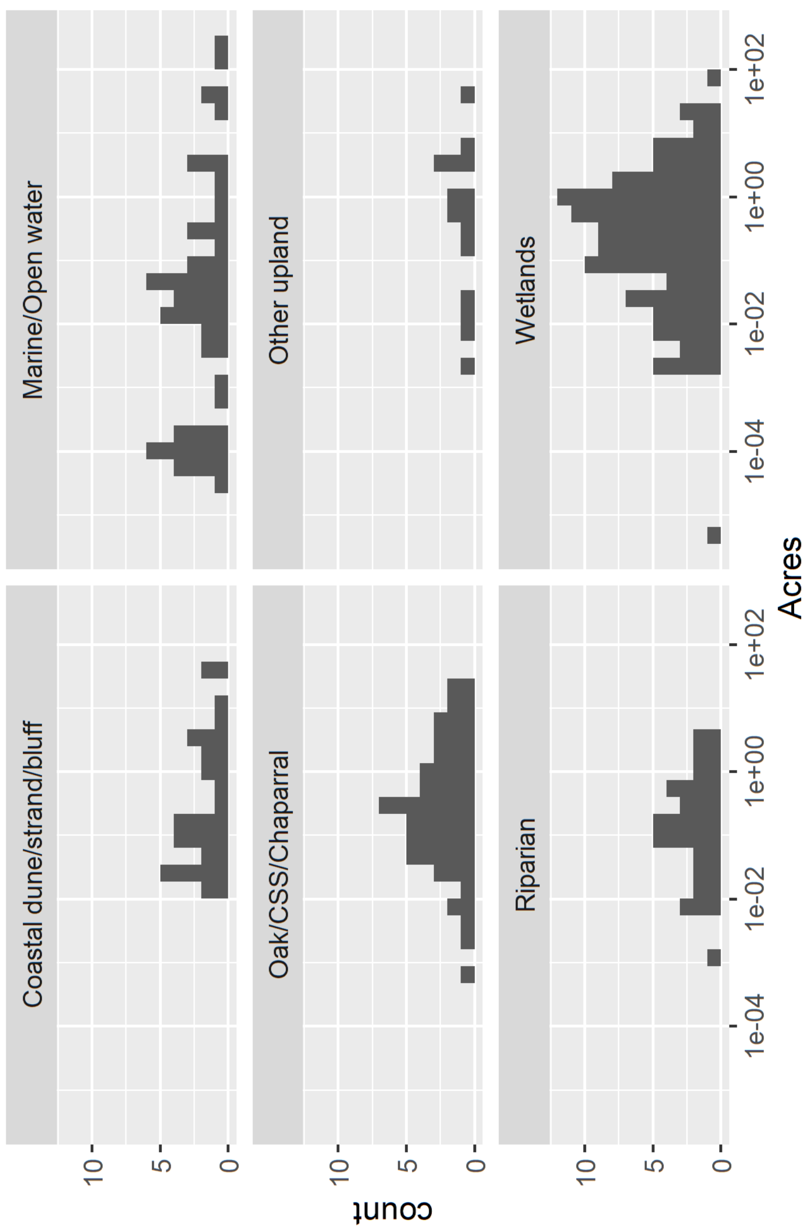


Figure A1.ii. Number of projects across (logarithmic) impact sizes for broad habitat categories.



Appendix 2
Supplementary tables and figures for Chapter 2

Table A2.i. Identified habitat quantification tools. “Purpose of tool” column gives direct quotes from each source on the purpose or motivation for the tool. See Figure 2 for citations. “Guiding policy” gives the applicable policy for which a tool could be used. The “Assesses” column denotes if tool was developed specifically for seagrass or macroalgae, for a broader habitat with SAV (e.g., an estuary), or for the habitat of a specific non-SAV species (e.g., salmonids). “Flexible” tools were designed to be adapted for more than one of these categories. The “Valuation or equivalency?” column identifies tools as a valuation method or equivalency assessment, or both. The following four columns provide ‘yes’ or ‘no’ answers to whether a tool addressed temporal variability (changes in performance across a year), a reference site comparison, regional or landscape context, and uncertainty in assigning mitigation. The last column assigned user complexity based on the number of inputs and effort or expertise require to obtain inputs.

Tool category	Name	Purpose of tool	Guiding policy	Assesses: seagrass, macroalgae, broader habitat (non-SAV species), flexible	Valuation or equivalency?	Includes temporal variability?	Includes reference site comparison?	Includes landscape context inputs?	Includes factor for uncertainty in equivalency?	User complexity (basic, moderate, complex)
Similar to HEA/historically used with HEA	Habitat Equivalency Analysis (HEA)	"…determine compensation for such resource injuries."	CERCLA, CWA, NMSA, OPA (USA)	Flexible	Both	No	No	No	No	Varies
	Landscape Equivalency Analysis	"…a biodiversity credit system for trading endangered species habitat designed to minimize and reverse the negative effects of habitat loss and fragmentation."	ESA (USA)	Broader habitat	Both	No	No	Yes	No	Complex
	Habitat-Based Resource Equivalency Method	"…how much, and what type of environmental restoration will compensate for injuries to natural resources that result from releases of hazardous	CERCLA, OPA (USA)	Broader habitat	Both	No	No	Yes	No	Complex

		substances or oil spills."								
	Uniform Mitigation Assessment Method (UMAM) & HEA hybrid	"propose a standardized, operational approach regardless of the development project and the ecosystem impacted that (i) enhances avoidance and reduction efforts and (ii) assesses biodiversity offset needs based on data available in Environmental Impact Assessments."	HD, WFD, MSFD (EU)	Broader habitat	Both	No	No	Yes	Yes	Varies
	Nested HEA	"By adapting the HEA approach with a nested HEA, NRDA could quantify direct ecosystem services losses as well as additional cross-service flows between habitats."	OPA (USA)	Broader habitat	Both	No	No	Yes	No	Moderate
	Puget Sound Nearshore Conservation Calculator	"Quantify the habitat impacts relevant for Puget Sound (PS) Chinook salmon and	ESA (USA)	Broader habitat (salmonids)	Both	No	No	Yes	No	Moderate

		Hood Canal summer-run chum…by quantifying habitat impacts from proposed project actions (construction, repair, replacement, mitigation)."								
Habitat Suitability Indices	Habitat Evaluation Procedure	"used [for]...1) wildlife habitat assessments, including both baseline and future conditions; 2) trade-off analyses; and 3) compensation analyses"; used with HSI	FWCA, ESA, NEPA (USA)	Flexible	Equivalency	No	No	No	No	Varies
	Habitat Suitability Index (HSI)	"used [for]...1) wildlife habitat assessments, including both baseline and future conditions; 2) trade-off analyses; and 3) compensation analyses"; used with HEP	FWCA, ESA, NEPA (USA)	Flexible	Valuation	No	No	Possible	NA	Varies
	HSI: Northern Gulf of Mexico brown shrimp and white shrimp	"Impact assessment and habitat assessment"	FWCA, ESA, NEPA (USA)	Broader habitat (brown & white shrimp)	Valuation	Yes	No	No	NA	Basic

	HSI: Pink shrimp	Impact assessment and habitat management	FWCA, ESA, NEPA (USA)	Broader habitat (pink shrimp)	Valuation	No	No	No	NA	Basic
	HSI: Eelgrass and oyster aquaculture	"…a biotic index referred herein as the Habitat Suitability Index (HSI), to objectively assess habitat suitability of shellfish aquaculture for critical species of fish and invertebrates"	MSFA (USA)	Broader habitat (salmonids, English sole, Dungeness crab)	Valuation	No	No	No	NA	Moderate
Ecological Quality Ratios	Biotic index for *Posidonia oceanica*	"…develop a biotic index based on P. oceanica (BiPo), focusing on: (i) the necessity of an index that may be applied over the largest geographical extent possible, (ii) the necessity of a tool for a baseline evaluation of P. oceanica status in the Mediterranean, (iii) the compliance with WFD requirements, (iv) the efficiency of the method in	WFD (EU)	Seagrass (*Posidonia oceanica*)	Valuation	No	Yes	No	NA	Moderate

		terms of reliability and cost."								
	Multi-metric index based on the seagrass *Zostera noltii*	"…an ecological quality index based on the seagrass Zostera noltii (ZoNI) according to the WFD requirements"	WFD (EU)	Seagrass (*Zostera noltii*)	Valuation	No	Yes	No	NA	Complex
	Biotic index for *Cymodocea nodosa*	"this index can be an adequate alternative for ecological status assessment in water bodies where other species are absent and, specifically, in transitional waters"	WFD (EU)	Seagrass (*Cymodocea nodosa*)	Valuation	No	Yes	No	NA	Complex
	Multivariate index for *Posidonia oceanica*	"…a multivariate index based on structural and functional attributes of the Posidonia oceanica ecosystem to assess the ecological status of coastal waters following WFD requirements."	WFD (EU)	Seagrass (*Posidonia oceanica*)	Valuation	No	Yes	No	NA	Complex

	Descriptors for *Posidonia oceanica*	"…evaluate some of those potential descriptors with a view to selecting appropriate indicators from the Posidonia ecosystem to use in implementing the WFD."	WFD (EU)	Seagrass (*Posidonia oceanica*)	Valuation	No	Yes	No	NA	Moderate
	Cartography of littoral and upper-sublittoral rocky-shore communities	"… a methodology for monitoring water quality based on the cartography of littoral and upper-sublittoral rocky-shore communities"	WFD (EU)	Macroalgae	Valuation	No	Yes	No	NA	Moderate
	Seagrass Quality Index	"…assessing the ecological quality of intertidal seagrass in estuaries and coastal systems, the Seagrass Quality Index (SQI). The design of the SQI aims to fulfil the Water Framework Directive requirements in terms of compliance"	WFD (EU)	Seagrass	Valuation	No	Yes	No	NA	Basic

Other tools	Guild-based fish assemblage habitat valuation	"…a robust, objective technique for the valuation of marine habitats that makes use of data that are commonly gathered in surveys of marine fish populations: density, fidelity, and mean size."	None	Broader habitat	Valuation	Yes	Yes	No	NA	Moderate
	Biological valuation maps	"baseline maps for future spatial planning at sea… marine biological valuation which is based on a literature review of existing valuation criteria and the consensus reached by a discussion group of experts"	None	Broader habitat	Valuation	No	No	Yes	NA	Complex
	Area-based mitigation ratios	Assigning mitigation	NA	Flexible	Equivalency	No	No	No	No	Basic
	Biocentric biological valorisation	"…biocentric approach for intrinsic valuation of the biodiversity of habitats...The obtained map of biological	None	Broader habitat	Valuation	No	No	Yes	No	Complex

		valuation can provide useful information for ecosystem-based management in the studied area indicating conflict zones and protection areas; it could also be adjusted to sensitivity assessment of benthic habitats.”								
	Nearshore Assessment Tool for Alaska: Southeast	“… a standardized protocol for rapidly assessing the habitat and functions of a particular marine or estuarine shore segment (intertidal zone and immediately adjoining upland) in Alaska's temperate ‘panhandle.’”	None	Broader habitat	Valuation	No	Yes	Yes	NA	Varies
	Five-Step Wetland Mitigation Ratio Calculator/ California Eelgrass Mitigation Policy	“…serve as the guidance for staff and managers within NMFS for developing recommendations concerning eelgrass issues	MSFA, FWCA, NEPA (USA)	Seagrass, flexible	Equivalency	No	No	Yes	Yes	Moderate

		through EFH and FWCA consultations and NEPA reviews throughout California”								
	California Rapid Assessment Method-Wetlands	“CRAM can be used to quickly assess the condition of any wetland relative to its performance standards (for mitigation and restoration projects) or relative to regional reference conditions for wetlands of a similar kind and setting”	CWA (USA)	Broader habitat (wetlands)	Valuation	No	Yes	Yes	NA	Moderate
	Oregon Rapid Wetland Assessment Protocol	“The purposes may include assessing all wetlands within a city for land use planning; assessing wetlands within a watershed; assessing individual wetlands or portions of wetlands for purposes of state and federal permitting and compensatory wetland	Oregon’s Removal Fill Law, CWA (USA)	Broader habitat (wetlands)	Valuation	No	Yes	Yes	NA	Moderate

		mitigation; and evaluating success of voluntary wetland restoration or enhancement projects."								
	Habitat Structure Index	"We present the habitat structure index (HSI), which enables rapid assessment and direct comparison of seagrass habitat structure using scores of 0 (poor) to 100 (excellent) based on integrating five habitat variables"	None	Seagrass	Valuation	No	No	No	NA	Moderate
	Braun-Blanquet scores	"A visual survey technique, modified from the original Braun-Blanquet (BB) scale…chosen to qualitatively assess the change in overall plant cover over time"	None	Seagrass	Valuation	No	No	No	NA	Basic
	Conservation index, Phase shift index, Substitution index	"The application of recently introduced approaches	WFD (EU)	Seagrass (*Posidonia oceanica*)	Valuation	No	Yes	No	NA	Moderate

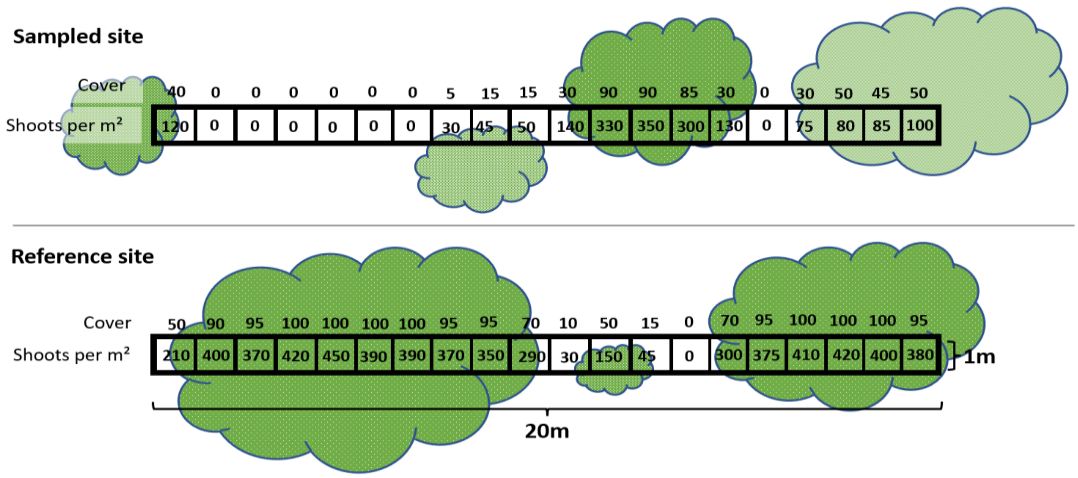
		based on a set of synthetic ecological indices, namely the Conservation Index (CI), the Substitution Index (SI) and the Phase Shift Index (PSI), is also reviewed focusing on their effectiveness in relation to the ecosystem health assessment and to the requirements of the WFD”								
	USACE (South Pacific Division) Standard Operating Procedure for Determination of Mitigation Ratios (Before-After-Mitigation-Impact)	“The purpose of this document is to outline the process for determining compensatory mitigation requirements as required for processing of Department of the Army (DA) permits under Section 404 of the Clean Water Act, Section 10 of the Rivers and Harbors Act, and Section 103 of the Marine	CWA, RHA (USA)	Flexible	Equivalency	No	Yes	Yes	Yes	Complex

		Protection, Research, and Sanctuaries Act."									
	Ecological Evaluation Index	The ecological evaluation index (EEI) was designed to estimate the ecological status of transitional and coastal waters. Marine benthic macrophytes (seaweeds, seagrasses) were used as bioindicators of ecosystem shifts due to anthropogenic stress	WFD (EU)	Seagrass	Valuation	No	No	No	NA	Moderate
	Uniform Mitigation Assessment Method (UMAM)	The Uniform Mitigation Assessment Method (UMAM) was established to fulfill the mandate of subsection 373.414(18), F.S., which requires the establishment of a uniform mitigation assessment method to determine the amount of mitigation needed to offset adverse	373.414(18), Florida Statute (USA)	Flexible	Both	No	Yes	Yes	Yes	Moderate

		impacts to wetlands and other surface waters and to award and deduct mitigation bank credits.								
	Index of Community Structure	A quality index Ics (= index of community structure) has been developed as a single numeric descriptor to assess the structural state of macroalgal communities and to evaluate their relative development on rocky shores.	MSFD (EU)	Macroalgae	Valuation	Yes	No	No	NA	Moderate

CERCLA: Comprehensive Environmental Response, Compensation, and Liability Act (1980; USA)
CWA: Clean Water Act (1972; USA)
FWCA: Fish and Wildlife Coordination Act (1934; USA)
MSFA: Magnuson-Stevens Fisheries Act (1976; USA)
MSFD: Marine Strategy Framework Directive (2008; EU)
NEPA: National Environmental Policy Act (1970; USA)
NMSA: National Marine Sanctuaries Act (1972; USA)
OPA: Oil Pollution Act (1990; USA)
RHA: Rivers and Harbors Act (1899; USA)
WFD: Water Framework Directive (2000; EU)

Figure A2.i. Example calculations and habitat surveyed for the Habitat Structure Index model (Irving et al. 2013) and the Seagrass Quality Index (Neto et al. 2013).



	A	B	C
1	**Habitat Structure Index (see Irving et al. 2013 for methods)**	**Reference**	**Sampling site**
2	% Area (m^2) with seagrass	=15.3/20*100	=5.75/20*100
3	Proximity (distance score between patches)	=(19-(7))/(19-1)*100	=(19-(2))/(19-1)*100
4	Continuity (scores for how many patches)	=(19-3)/(13-1)*100	=(13-4)/(13-1)*100
5	Cover (scores using cover bins)*	=(((3*1)+(4*2)+(13*3))/20)/3*100	=(((13*1)+(5*2)+(2*3))/20)/3*100
6	Species Identity (richness score)	=1/1*100	=1/1*100
7			
8	Habitat Structure Index:	=(B2^2+B3^2+B4^2+B5^2+B6^2)^0.5	=(C2^2+C3^2+C4^2+C5^2+C6^2)^0.5
9	Ratio between sampled and reference site:		=C8/B8
10			**=0.78**
11			
12	**Seagrass Quality Index (see Neto et al. 2013 for methods)**		
13	Species richness	1	1
14	Area with greater than 5% cover (m^2)	19	13
15	90th % of shoots (shoots/m^2)	420	330
16			
17	Seagrass Quality Index		=((C13/B13)*0.2)+((C14/B14)*0.3)+((C15/B15)*0.5)
18			**=0.8**
19			
20			
21	*Habitat Structure Index cover bins		
22	>=90	3	
23	40-89	2	
24	<40	1	

Appendix 3
Supplementary tables and figures for Chapter 3

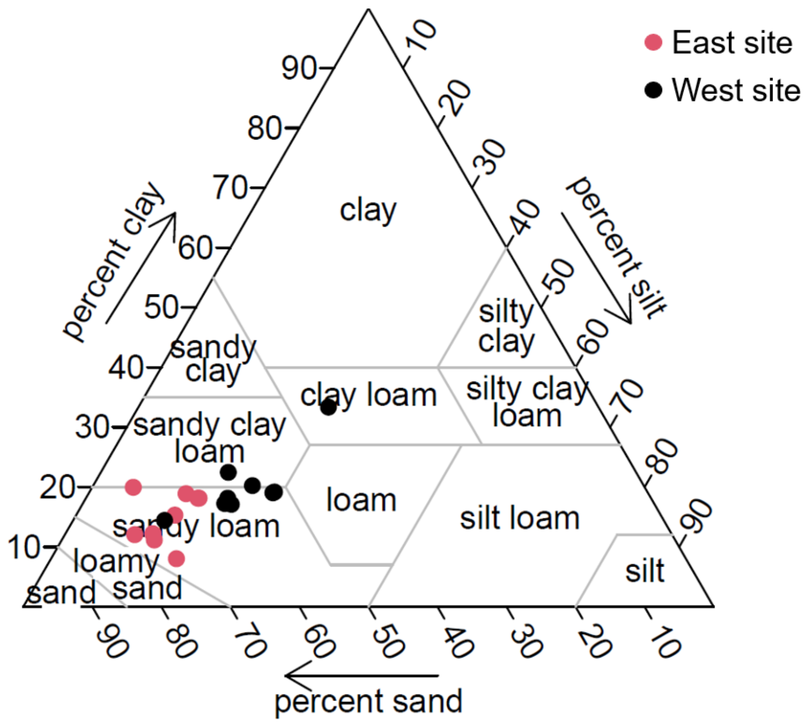
Table A3.i. Results of mixed effects Cox proportional hazards model for survivorship of watering treatments across sites. Table presented includes Watering: Non-salt hardened and not watered and Site: West as comparisons.

	β	**exp(β)**	**SE (β)**	**z**	**p**
Watering: Salt hardened and not watered	-0.033	0.967	0.322	-0.10	0.92
Watering: Salt hardened and watered	-4.392	0.012	0.775	-5.67	<0.001
Site: East	0.897	2.451	0.644	1.39	0.16

Table A3.ii. Results of mixed effects Cox proportional hazards model for survivorship of planting configurations and sizes across sites. Table presented includes Configuration: Lone pot and Site: West as comparisons.

	β	**exp(β)**	**SE (β)**	**z**	**p**
Configuration: Pot w/plugs	-1.046	0.351	1.230	-0.85	0.400
Configuration: Lone plug	1.659	5.252	0.753	2.20	0.028
Configuration: Plug w/pot	1.322	3.752	0.746	1.77	0.076
Site: East	1.255	3.509	0.707	1.78	0.076

Figure A3.i. Soil texture triangle (R package "plotrix"; Lemon 2006) with the east and west sites' sand, silt, and clay fraction. The east site had higher sand fractions and lower silt fractions than the west site.



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