SAMPLING FISH AND MACROINVERTEBRATE

RESOURCES IN TIDAL WETLANDS

SACRAMENTO-SAN JOAQUIN DELTA

Report on

PHASE IV PILOT MONITORING IN 2018

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# Preface

Much of the tidal wetland restoration in the Sacramento-San Joaquin Delta and Suisun Marsh (the Upper SF Estuary) is being constructed under the premise that wetland restoration will increase the resilience of threatened fish. The Fish Restoration Program Monitoring Team (FRP) is tasked with developing monitoring plans for tidal wetland sites restored pursuant to requirements in the 2008/2009 Biological Opinions for state and federal water project operations (USFWS 2008, NMFS 2009, CDFW 2009). We led the Interagency Ecological Program (IEP) Tidal Wetlands Monitoring Project Work Team (PWT) in developing the Tidalwetland monitoring framework for the upper San Francisco Estuary (hereafter "Framework"; PWT 2017a). The PWT has developed a set of conceptual models and hypotheses for how wetlands benefit fish (Sherman et al. 2017). These were the basis for recommendations for sampling methods to evaluate effectiveness of restoration projects (PWT 2017b). FRP has rigorously tested these sampling methods, but questions remain as to the comparability of FRP data, collected in wetlands, and long-term IEP monitoring data, collected in channels.

The Framework recommends a whole-ecosystem approach for describing the capacity, opportunity, and realized function (sensu Simenstad and Cordell 2000) provided by wetland restoration to at-risk fishes. As such, FRP is collecting data on the physical habitat (water quality, bathymetry, and flow), nutrients, primary producers (phytoplankton and vegetation), invertebrates, and fish that occur on each of our restoration sites. Each restoration site is paired with an existing wetland site as a “reference”, and each wetland will be monitored both before and after restoration. However, measuring all these sampling components required extensive refinement of sampling methods and sampling schema.

Meso- and macroinvertebrates, including amphipods, mysids, insects, copepods and isopods, are important food resources for tidal wetland fish, but are often patchily distributed and highly variable (Slater and Baxter 2014, Whitley and Bollens 2014, Baxter et al. 2015, David et al. 2016). Monitoring change over time requires understanding the level of spatial and temporal replication necessary for statistical validity. Information on meso- and macroinvertebrates is necessary to address Framework hypotheses F2-F5, which were derived from the PWT’s Food Web Conceptual Model (Hartman et al. 2017a) and Chinook Salmon Tidal Wetland Model (Goertler et al. 2017).

Even for established methods, such as zooplankton trawls and nutrient sampling, more research is needed to determine the spatial and temporal extent of inference that can be made for a given metric. Multiple long-term monitoring surveys sample the pelagic realm for meso- and macro-zooplankton, phytoplankton, and nutrients using well-established methods (Hennessy 2009). However, it is unclear the extent to which these constituents differ between the deep channel habitat currently sampled and the wetlands that our program will sample (Bollens et al. 2014; Kimmerer et al. 1998). Understanding differences between channels and wetlands is also necessary to detect exchange between these habitats that is predicted to increase food availability in the channel (Framework hypotheses F8-F10, (Hartman et al. 2017b)).

Fish are also highly variable across the Upper SF Estuary. Although fishes ranging from larvae to adults are sampled regularly, just a few sampling programs focus on small channel/shallow water/vegetation edge habitats. We have tested several different types of gear for catching fish in these habitats in the North Delta (Cache Slough Region). We need a better understanding of the function of these gears across estuarine gradients and how catches compare to long-term IEP monitoring. Fish community sampling in long-term monitoring is needed to determine the presence of listed species (hypotheses P4 and P14), to provide specimens of listed fish for various potential studies of diet, condition, and growth (hypotheses F4-F7), and to understand the potential for predation on and competition with the listed species (hypothesis S4), aspects of “capacity” in the Habitat Attributes tier of the Chinook Salmon Tidal Wetlands conceptual model (Goertler et al. 2017).

## Pilot Monitoring Phases

We conducted a Phase I “gear exploration” from July to October 2015 (Contreras et al. 2016). Based on results from that effort, successful methods were selected for inclusion in the second phase of pilot work. Phase II occurred from February through July 2016, and provided a more rigorous evaluation of gear feasibility during the time of year listed fish are most likely to be using wetlands (Contreras et al. 2017). Phase II included quantitative comparisons of abundance, size, composition, and diversity which led to recommended gear types. The recommended gear types from Phase II were used in the Phase III study that further refined the spatial and temporal sampling schemes necessary to evaluate monitoring hypotheses, and began testing the extent to which IEP’s channel-based monitoring surveys can be used to make inferences on tidal wetlands (Contreras et al. 2018).

After each pilot phase, results were reviewed by the PWT before inclusion in development of plans for the next phase. This report covers Phase IV, which expanded comparisons between FRP and IEP surveys, tested new methods for monitoring fish, SAV, nutrients, and phytoplankton, and began collecting pre-project data at future restoration sites.

## Project Objectives

* Determine the level of spatial and temporal replication necessary to make sampling design recommendations for long-term monitoring.
* Determine the extent to which long-term IEP sampling reflects conditions in nearby shallow-water and wetland habitats.
* Determine whether gear efficiency evaluations are feasible using new sampling technology (i.e., ARIS sonar).
* Begin developing a baseline of biomass, community composition, and fish condition for fish and invertebrates near planned tidal restoration and comparison sites. This will allow us to make pre-and-post-restoration comparisons for evaluating restoration progress.

## 

# Part 1: Phytoplankton and Invertebrate spatial and temporal variability

## Introduction

### Invertebrates

Understanding the variability of zooplankton, and macroinvertebrates will allow us to evaluate appropriate timing and replication of samples for characterizing the food-web support in restored tidal wetlands. Macroinvertebrates associated with vegetation and shallow water habitat, such as amphipods and insect larvae, have been historically under-studied in this system; however, they provide the majority of salmonid diet composition in these areas (Bottom et al. 2011; David et al. 2016a; Maier and Simenstad 2009; Sommer et al. 2001), and are a component of Delta Smelt diets when smelt occur in areas of high macrophyte production (Whitley and Bollens 2014). Because these groups are understudied, it is unclear how variable biomass and community composition are within a site, between sites and between seasons. Because epifaunal invertebrates are a smaller percentage of smelt diets (Slater and Baxter 2014) and are less mobile than zooplankton, it may only be necessary to sample these macroinvertebrates once or twice per year. If sampling is limited, we want to determine what time of year has greatest overlap between listed fish species and their food supply.

We chose spring for our spatially intensive sampling, because this is when there is the highest density of juvenile salmonids rearing in the estuary (Williams 2012), Delta Smelt are spawning in the freshwater Delta (Baxter et al. 2015; Sommer and Mejia 2013), and Longfin Smelt are present and spawning in the marshes surrounding Suisun Bay (Grimaldo et al. 2017). However, the importance of food supply during the fall, while juvenile smelt area rearing, is considered one of the major factors in population resilience (Baxter et al. 2015; Brown et al. 2014). Therefore, chose to characterize potential food supply in areas important to smelt in the fall.

Due to limited resources, we targeted fall sampling only in areas likely to have high smelt abundance. In most years, a majority of the population migrates to the Low Salinity Zone (LSZ), to rear (Baxter et al. 2015). This area generally stretches from Suisun Bay to the Confluence, though exact location depends on water year type (Brown et al. 2014). There is an additional resident population of Delta Smelt which remains in the North Delta year round (Sommer et al. 2009), so we chose to target these areas for additional fall sampling.

### Phytoplankton

Phytoplankton are considered to be one of the key carbon sources in the aquatic food web. However, not all phytoplankton are created equal. Diatoms and green algae are preferred over cyanobacteria and flagellates in most cladocera and copepod species studied in this estuary (Orsi 1995), though more recent studies suggest cyanobacteria make up a large part of copepod diets when they occur in high abundance (Kimmerer et al. 2018a). Furthermore, preferred food differs by species of zooplankton and species of phytoplankton (Bouley and Kimmerer 2006; Cloern and Dufford 2005; Jassby 2008). In the past, most evidence pointed to pelagic phytoplankton as the key driver of the Bay-Delta food web (Canuel et al. 1995; Sobczak et al. 2005; Sobczak et al. 2002), however, more recent work suggests benthic, epiphytic, and wetland-derived carbon may plan a more important role than previously realized (Schroeter et al. 2015).

Given the importance of taxonomic identity when assessing phytoplankton’s role in the food web, it is important to better understand phytoplankton community composition at restoration sites. Once we start regular monitoring of restoration sites, we may decide to collect phytoplankton samples concurrently with monthly zooplankton samples. However, we wanted to first establish a baseline of phytoplankton variability within the wetlands. Therefore, we collected phytoplankton concurrently with the intensive spring sampling of macroinvertebrates.

#### Study questions:

1. How do invertebrate and phytoplankton communities change from year to year?
2. Are there significant differences between channel habitat, managed wetlands (pre-restoration), and tidal wetlands (remnant and/or post-restoration)?
3. What food is available for listed fish species throughout the year?
   1. When during the spring is most important to sample?
   2. How do fall food resources compare to spring food resources?

## Methods

### Sampling Sites

To answer questions 1 and 2, we sampled FRP restoration sites and surrounding wetlands distributed across the Delta and Suisun Marsh FRP (Figure 1, Table 1). These sites incorporate varying salinity and surrounding land use (the “ecoclines” identified in the IEP TWM PWT conceptual models, see (Sherman et al. 2017). Sites for 2018 were an expanded set of sites sampled in 2017, allowing for year-to-year comparisons. Sampling restoration sites (“Impact” sites) before restoration and comparing these sites to existing wetlands (“Control” sites) allows for a Before-After, Control-Impact design, once restoration is completed on some of these sites. While we do not expect our restoration sites to develop exactly like the comparison sites, the network of comparison sites provides a background, or “ambient” condition for the estuary that will allow us to better see the effectiveness of the restoration actions.

Note: We collected samples at Dow Wetlands in the spring of 2018, but we did not process the samples or analyze the data due to changes in priorities recommended by DWR FRP staff.

To answer question 3, we conducted increased sampling at one site (Decker Island) four times throughout the spring, and sampled in the fall at a subset of the locations where Delta Smelt are found most often. An analysis of data from the Fall Midwater Trawl Survey from September and October 2010-2016 showed the majority of the smelt caught where in the Confluence of the Sacramento and San Joaquin Rivers and Suisun Bay (CDFW data available: https://www.wildlife.ca.gov/Conservation/Delta/Fall-Midwater-Trawl). There is an additional, resident population in the Cache Slough Complex (Baxter et al. 2015), so we sampled the subset of wetland sites near these “smelt- hot- spots” during the fall (October 9th – November 1st), at a lower intensity than spring sampling. Before conducting the fall sampling, we assessed the data from the FMWT and the USFWS’s Enhanced Delta Smelt Monitoring (EDSM) survey to determine whether the fall 2018 smelt distribution matched previous years.

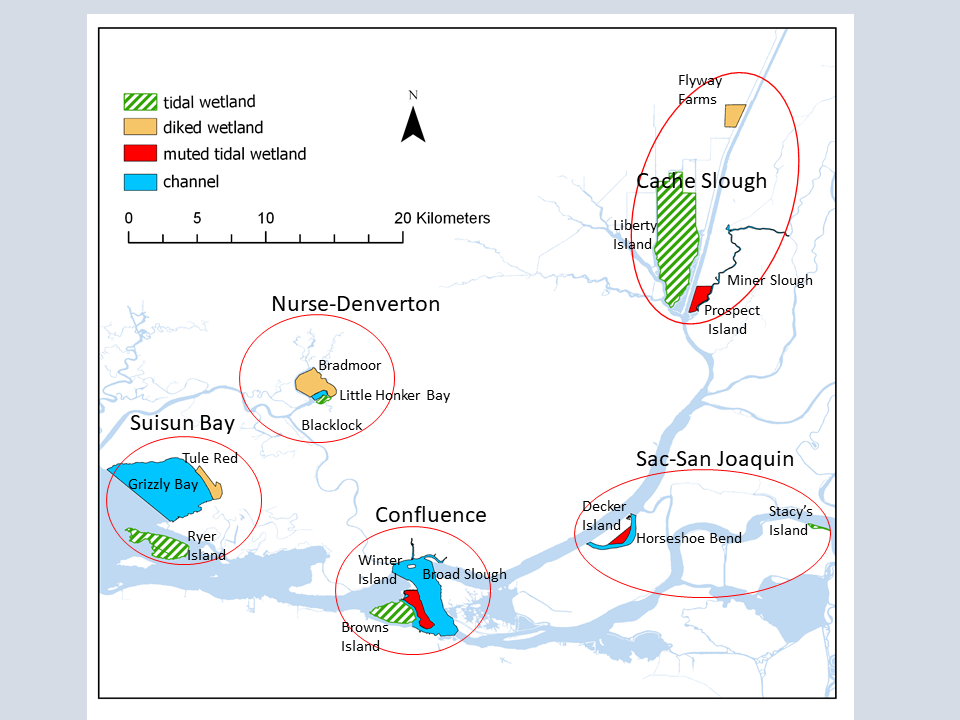


Figure 1 Sites that were sampled during the spring of 2018. Within each region (outlined in red), we compared pre-project data from currently diked or muted tidal wetlands (planned restoration sites), with data from associated channel habitat, and existing tidal wetlands.

Table 1. Sample sizes for spring sampling in 2018. Sample numbers differ based on site size, habitat availability, and logistical constraints.

|  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
| Site | Region | benthic grab | mysids | neuston | phytoplankton | Sweep net | Zooplankton | Total |
| Flyway Farms | Cache Slough Complex | 4 | 0 | 0 | 7 | 6 | 0 | 17 |
| Liberty Island | Cache Slough Complex | 6 | 9 | 6 | 17 | 8 | 9 | 55 |
| Miner Slough | Cache Slough Complex | 4 | 5 | 4 | 4 | 4 | 5 | 26 |
| Prospect Island | Cache Slough Complex | 2 | 3 | 1 | 5 | 9 | 8 | 28 |
| Broad Slough | Confluence | 4 | 5 | 6 | 5 | 5 | 6 | 31 |
| Browns Island | Confluence | 3 | 6 | 6 | 6 | 10 | 6 | 37 |
| Winter Island | Confluence | 9 | 7 | 9 | 6 | 1 | 4 | 38 |
| Grizzly Bay | Suisun Bay | 6 | 8 | 6 | 6 | 0 | 8 | 40 |
| Ryer Island | Suisun Bay | 6 | 6 | 6 | 6 | 12 | 6 | 42 |
| Tule Red | Suisun Bay | 5 | 0 | 4 | 6 | 6 | 6 | 27 |
| Blacklock | Nurse-Denverton | 4 | 4 | 4 | 6 | 4 | 4 | 26 |
| Bradmoor | Nurse-Denverton | 6 | 1 | 1 | 6 | 10 | 6 | 30 |
| Little Honker Bay | Nurse-Denverton | 6 | 8 | 6 | 6 | 8 | 8 | 42 |
| Decker | Sacramento-San Joaquin | 2 | 6 | 5 | 6 | 7 | 9 | 35 |
| Horseshoe Bend | Sacramento-San Joaquin | 6 | 3 | 3 | 3 | 8 | 3 | 26 |
| Stacys Island | Sacramento-San Joaquin | 4 | 5 | 2 | 4 | 4 | 5 | 24 |
| Grand Total |  | 77 | 76 | 69 | 107 | 102 | 92 | 523 |

Table 2 Sample sizes for fall sampling in 2018.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Samples near | region | site type | Sweep net | Mysid trawl | Zoop trawl | **Total** |
| Winter Island | Confluence | diked wetland | 6 | 3 | 3 | **12** |
| Browns Island | Confluence | tidal wetland | 6 | 3 | 3 | **12** |
| Ryer Island | Grizzly Bay | tidal wetland | 5 | 3 | 3 | **11** |
| Prospect Island | Cache Slough | diked wetland | 6 | 3 | 3 | **12** |
|  |  | **Total** | **23** | **12** | **12** | **47** |

### Habitat Types and Sampling gears

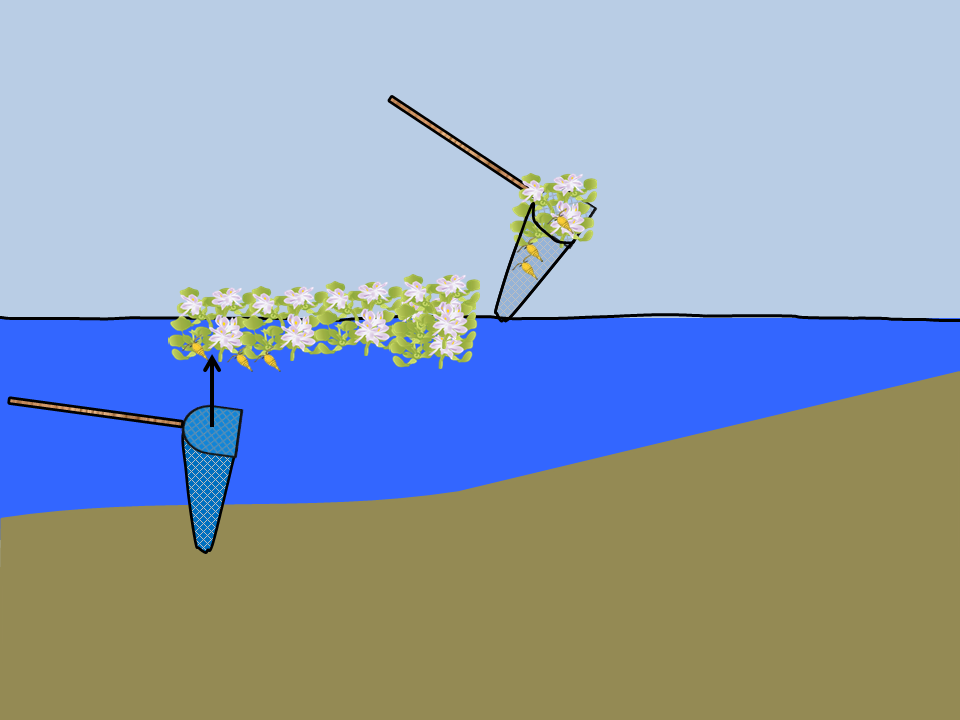
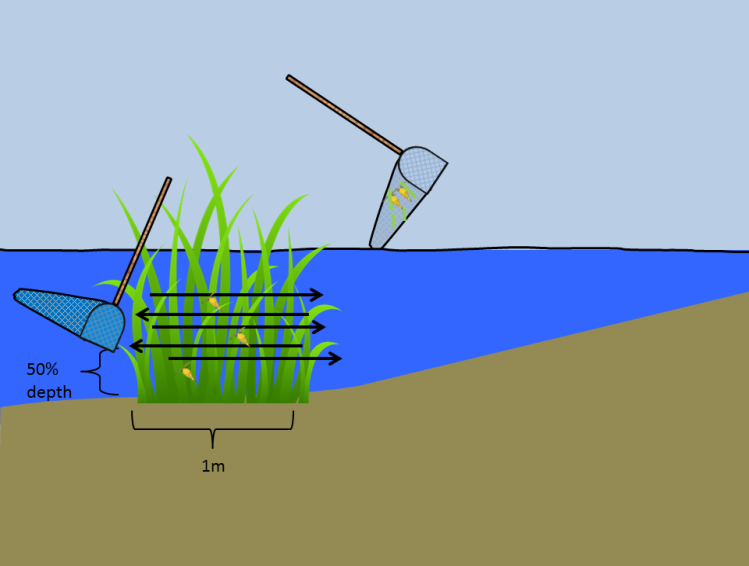
#### Vegetation

Previous studies showed very high replication was necessary to differentiate between vegetation types (2017; 2018; Contreras et al. 2016). Therefore, we randomly distributed our sweep net samples through all vegetation types present on the site in proportion to their abundance, rather than specifically targeting samples from each vegetation type. We haphazardly choose 6-12 sampling locations per site and used sweep nets to sample vegetation.

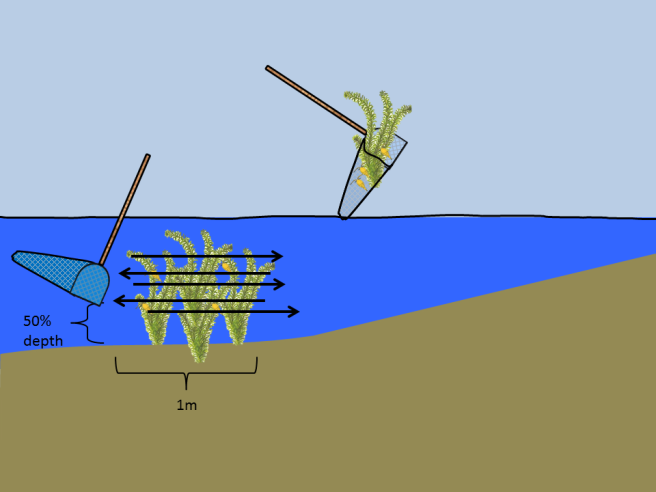
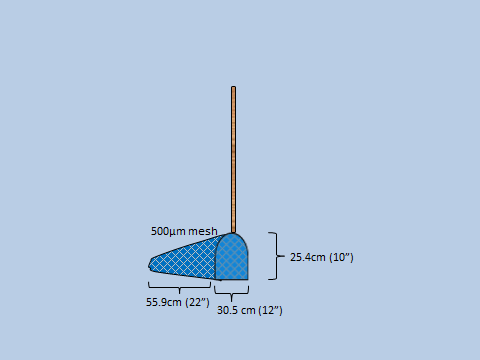
**Sweep nets:** Sweep nets are a simple but effective way to sample the invertebrate community. Sweep nets may capture higher species diversity than many passive methods, though with higher variability in biomass (Turner and Trexler 1997).We will use a 25cm x 30cm D-frame net with 500 micron mesh for all sweep net samples (Figure 2A). The sweep net technique was adapted slightly in different vegetation types.

**EAV:** Emergent vegetation (EAV) samples occurred in the dominant emergent vegetation species, usually either *Schoenoplectus* spp., *Typha* spp., or *Phragmites australis*. We took five, one-meter sweeps through the edge of the vegetation, scraping the vegetation as much as possible to knock invertebrates off the stems (Figure 2B). We then rinse down the net and preserved all invertebrates in ethanol for later identification.

**SAV:** To sample in SAV, we took the same five, 1-meter sweeps as for EAV and collected any vegetation within the border of the net after the sweep is completed (Figure 2D). The sample will be placed on ice for processing in the lab. In the lab, we will rinse the vegetation, remove all invertebrates, and preserve the invertebrates for later identification. Any vegetation captured in the sweep net will be dried to a constant weight to standardize the sample.



B



C

D

A

Figure 2 A) Specifications of the sweep net. B) Use of sweep net in emergent vegetation. C) Use of sweep net in submerged vegetation. D) Use of sweep net in floating vegetation.

To allow us to make inferences for broad-scale invertebrate-vegetation relationships, we collected an SAV rake sample immediately after collecting a sweep net sample at each SAV site (see SAV survey techniques, below).

**FAV:**  Sampling techniques in FAV were dependent on FAV species. For *Eichhornia crassipes*, *Hydrocotyle,* and *Azolla*, we harvested a 25cm x 30cm sample from below using the same d-frame net and severing the connection to surrounding plant material with shears (Figure 2C; (Marineau et al. 2019)). We placed the roots of the plant material and associated invertebrates on ice. Upon return from the field, we will separate the invertebrates from the vegetation and dry the plants to a constant weight. *Ludwigia* spp. is a creeping emergent, and does not form discrete, easy-to-harvest clumps. Therefore, it will be sampled with five, 1-meter sweeps, as for EAV.

#### Open water and channel

Our open water sampling patches were haphazardly distributed across all unvegetated open water and channels > 1.5 m across. Methods used in open-water have a long history of use in monitoring in the Delta, and using these same methods will allow us to compare our measurements in vegetated wetlands to conditions in channels and make comparisons to long-term data sets. Methods included: zooplankton and mysid trawls, PVC cores and ponar grabs.

**Benthic core:** Benthic cores have been used extensively to quantify bivalves and other infauna in tidal wetlands (Howe et al. 2014; Wells 2015). In shallow water (<1.5 m), we took a 4 in (10 cm) diameter benthic core (Figure 3A), hand-deployed to a depth of 20 cm. In deep water >1.5 m, we used a 15.2 cm x 15.2 cm ponar grab modified for use in hard substrates (as per USFWS Liberty Island Monitoring, L. Smith pers. comm, Figure 3B), with three samples at each site. The core was washed and sieved on board the boat to remove the sand/mud and preserve any organic detritus and invertebrates. We calculated effort as catch per surface area of substrate sampled.

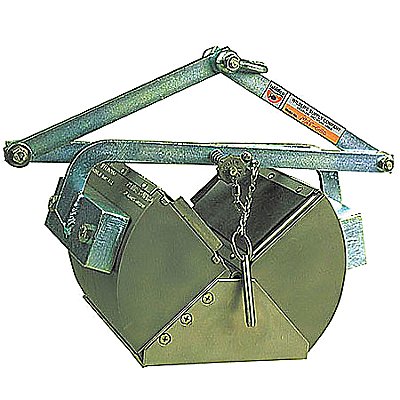
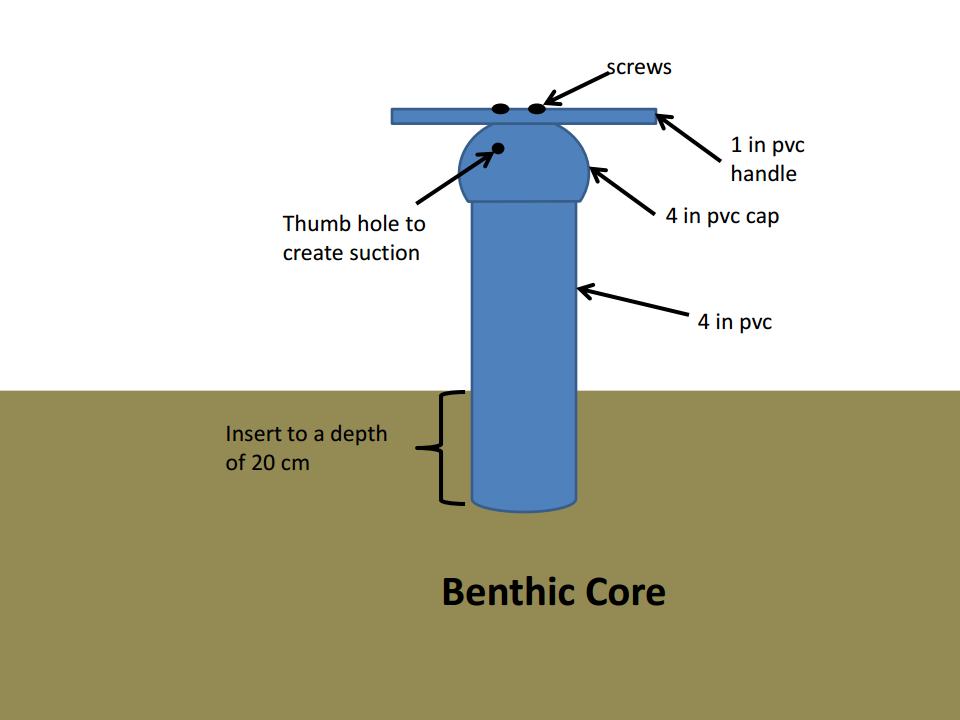


Figure 3. A) Benthic core made of 4” PVC pipe for use in shallow water (<1.5 meters). B) Ponar grab for use in water greater than 1.5 meters.

**Mysid and zooplankton nets:** Macrozooplankton (Mysid) nets have been used extensively to characterize macrozooplankton in the water column. Macrozooplankton includes amphipods and mysids that are large components of fish diets (Feyrer et al. 2003, Slater and Baxter 2014). We sampled macrozooplankton in the water column using a 50 cm mouth diameter (0.500 mm mesh size) mysid net and sampled mesozooplankton with a 14.6 cm diameter (0.150 mm mesh size) zooplankton net attached or held alongside (Figure 4, similar to EMP methods, (Hennessy 2009)). These nets were held approximately 1 m to the site of the bout, 10 cm below the surface of the water and trawled at 1-2 mph for five minutes. A flowmeter mounted in the net measured sample volume, and effort was standardized by catch per liter of water sampled.

Where channels are less than 1.5 m in width, and no open water is present, we placed the net in the channel on an ebb tide and allowed the tidal current to flow through the net for five. After retrieval, the nets were rinsed from the outside to wash down the sample into the cod end. All content collected in a cod end was preserved in 70% ethanol for identification in the lab.

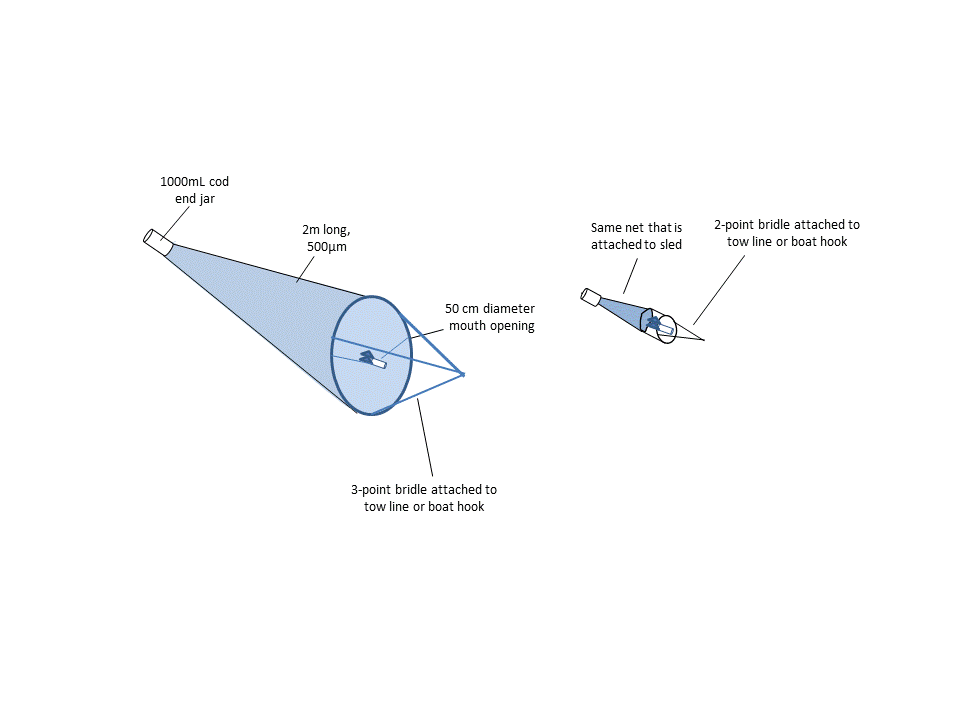
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Figure 4. Set up of mysid and zooplankton nets.

**Neuston tow:** The neuston net is a 45 cm x 30 cm rectangular net, 1 m long with 0.500 mm mesh towed half-way out of the water to sample invertebrates on the surface of the water (Figure 5A, B). We towed the neuston net at the surface of the water from the side of the boat via a boat-hook. In very shallow or narrow channels, we pulled the net along the edge of emergent vegetation by hand (Figure 5C; as in (Howe et al. 2014)). We standardized effort by the distance of the tow calculated by GPS track multiplied by half the mouth area of the net to calculate volume of water sampled. After retrieval, all content collected in a cod end was preserved in 70% ethanol for later ID.

**Phytoplankton:** At each zooplankton trawl site, a single, 150 mL sample of water was from the surface and preserved with Lugol’s iodine solution for identification of phytoplankton community composition.

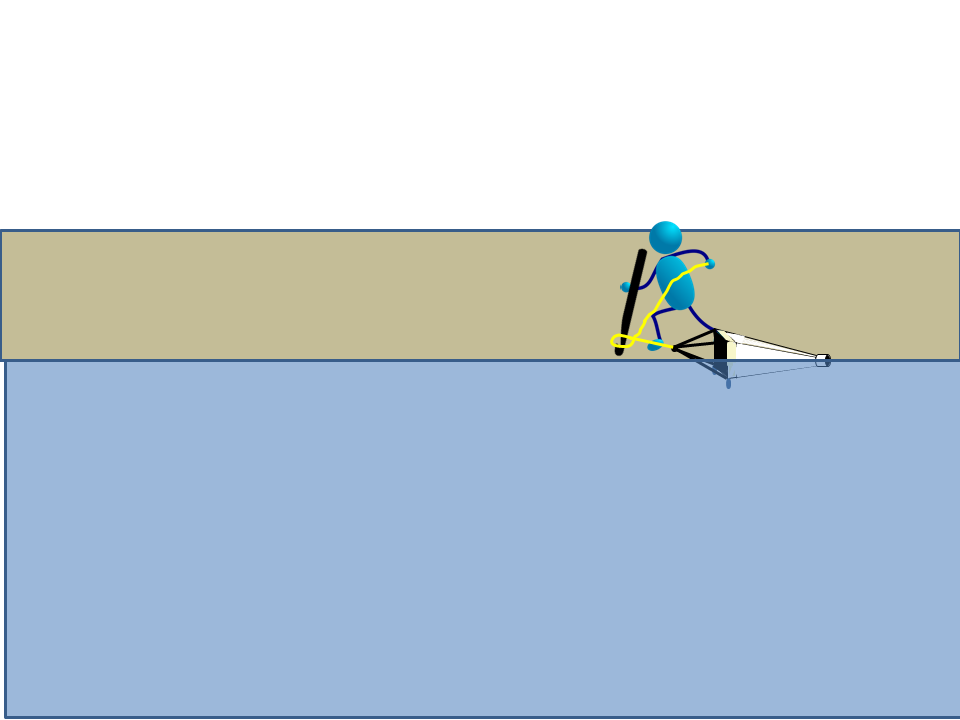
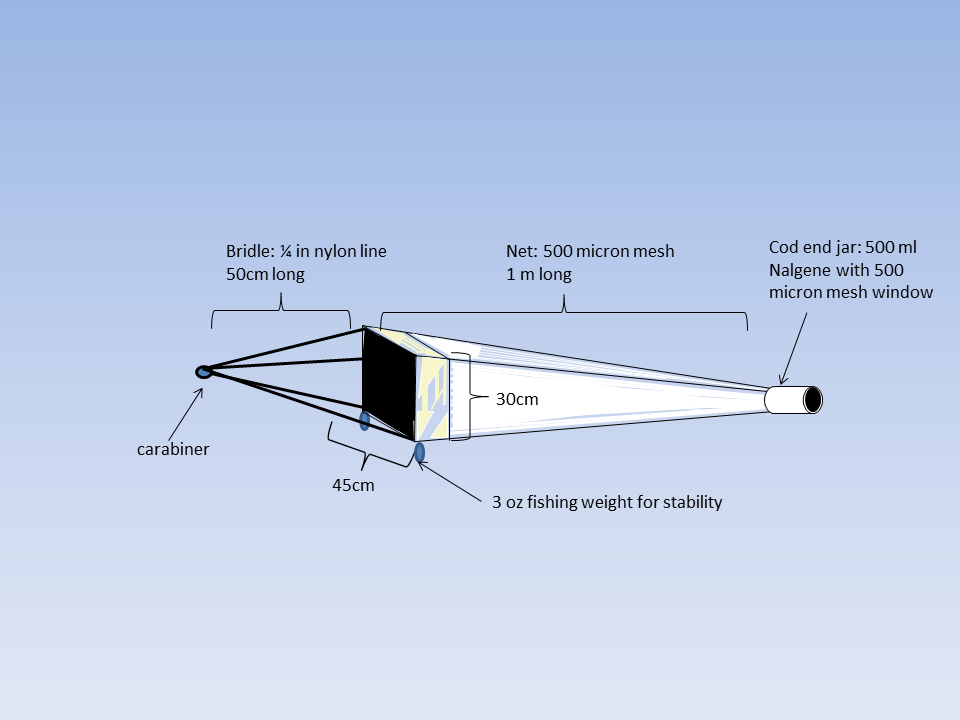
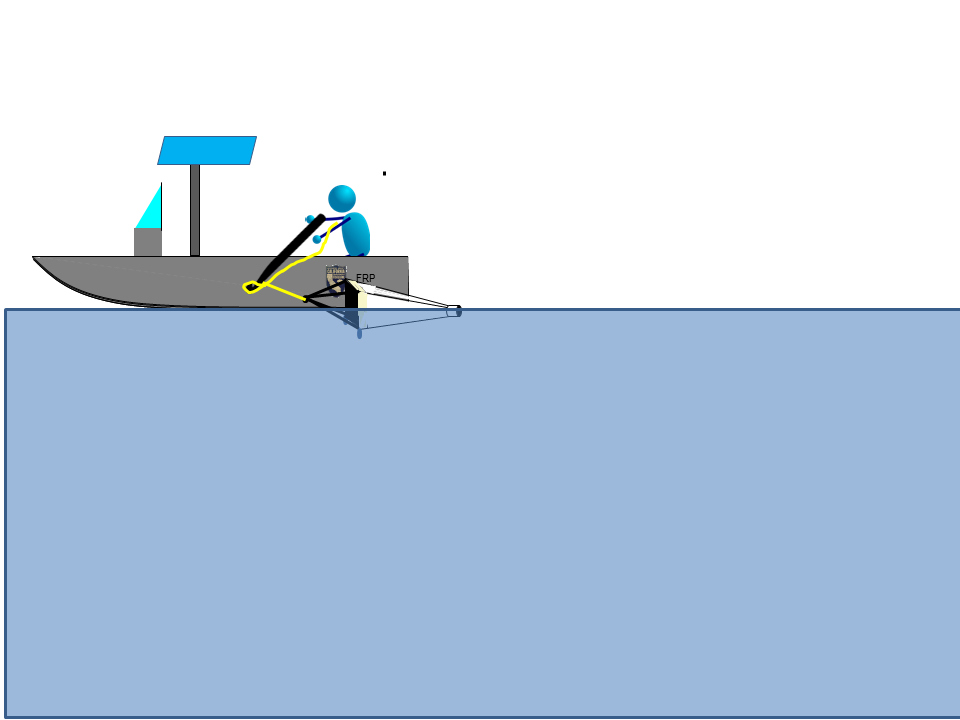


Figure 5. A) Specifications for the neuston net. B) Deploying the neuston net alongside a boat. C) Deploying the neuston net from shore.

### Laboratory Methods

**Taxonomic effort**: Invertebrates were sorted and identified to taxonomic level according to their importance in fish diets (see Table 3). Mysids, isopods and amphipods were identified to Genus or Species; insects were identified to Family. The first twelve individuals of each taxonomic group per sample will be measured to the nearest 0.1mm using an ocular micrometer. Therefore, all bivalves were identified to genus and measured along the longest axis to the nearest mm.

#### Macroinvertebrates

All trawls and sweepnet samples were sorted to extract macroinvertebrates (> 0.500 mm) from plant material and detritus. Benthic infauna (most importantly invasive bivalve grazers *Corbicula* and *Potamocorbula*), are not commonly found in salmon or smelt diets; however, the influence of invasive bivalves on the food web makes them important to predicting availability of production to the pelagic food web (Kimmerer and Lougee 2015; Lucas and Thompson 2012). Therefore, all bivalves were extracted from benthic samples and enumerated. All other benthic infauna were marked as “Present” but not counted. Zooplankton caught incidentally in macroinvertebrate samples were marked as “present” but not counted.

Invertebrates were identified by a Senior Laboratory Assistant (SLA) or Scientific Aide. A subset of samples had identifications checked by an Environmental Scientist for quality assurance. Another subset of samples were checked by an outside lab (EcoAnalysts, Inc.), for external quality assurance.

**Subsampling:** Approximately 400 invertebrates from each sample were identified. If more than 400 invertebrates were present in a sample, or more than four hours was required for processing, they were quantitatively sub-sampled using a grid tray.

Table 3. Levels of taxonomic resolution for each group of taxa commonly found in invertebrate samples.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| Phylum | Subphylum | Class | Order | Level of ID |
| Annelida |  | all | all | Class |
| Arthropoda | Chelicerata | Arachnida | all | Class |
| Arthropoda | Crustacea | Maxillopoda: Copepoda | all | Genus |
| Arthropoda | Crustacea | Malacostraca | Amphipoda | Genus |
| Arthropoda | Crustacea | Malacostraca | Cumacea | Class |
| Arthropoda | Crustacea | Malacostraca | Decapoda | Species |
| Arthropoda | Crustacea | Malacostraca | Isopoda | Genus |
| Arthropoda | Crustacea | Malacostraca | Mysidea | Species |
| Arthropoda | Crustacea | Branchiopoda | Cladocera | Genus |
| Arthropoda | Crustacea | Ostracoda | Podocopida | Order |
| Anthropoda | Hexapoda | Collembola | All | Class |
| Anthropoda | Hexapoda | Insecta | All | Family |
| Mollusca |  | Bivalvia | All | Genus |
| Mollusca |  | Gastropoda | All | Family |
| Nematoda |  | All | All | Phylum |
| Platyhelminthes |  | All | All | Phylum |

#### Zooplankton

Most zooplankton samples were processed by CDFW staff at the Stockton laboratory, but 50 samples were processed by EcoAnalysts, Inc. (Moscow, ID). Samples were processed in the same manner by both laboratories. First, samples were filtered and washed in a 0.150 mm mesh sieve. Filtered zooplankton were diluted to a set volume depending on the concentration of zooplankton and/or detritus. One-mL subsamples were then placed on a Sedgewick-Rafter cell glass slide. All organisms were identified to the taxonomic resolution identified in Table 3. At least 5 slides, but no more than 20 slides were processed for each sample, targeting at least 400 organisms. This subsample was then extrapolated to calculate the total number of organisms in the sample. A subset of samples was checked by a second taxonomist for quality assurance.

#### Phytoplankton

All laboratory analysis of algal samples was conducted by EcoAnalysts, Inc. (Moscow, ID), using the Utermöhl microscopic method (Utermöhl 1958) and APHA Standard Methods (APHA 2017). In brief: At least 400 total algal units and 100 units of the dominant taxon or taxa (genus or species level) were counted at appropriate levels of magnification for the cell size. Final counts were expanded to account for subsampling.

### Analysis

To answer Questions 1 and 2 on the variation in phytoplankton, zooplankton, and macroinvertebrate abundance, we compared samples from across sites in 2017 and 2018. We used mean log-transformed CPUE to compare invertebrate abundance across sites and between years using generalized linear mixed models (GLMMs) with site as an error term and the predictor variables listed in Table 4. These different wetland sites will provide the “Control-Impact” blocks for our BACI design in future analyses. We tested the fit of all possible models using Akaike’s Information Criterion corrected for small sample sizes (AICc) (Anderson 2008, Gotelli and Ellison 2012). Since we only have two years of data per site, we were not able to differentiate between variance due to water year type and variance due to other inter-annual factors, however the same analyses can be conducted in future years when we have more data per water year type. This data will also provide the “Before” of our Before-After Control-Impact design. All GLMMs used the “lme4” package in R (Bates et al. 2016).

To detect differences in community composition, we used permutational multivariate analysis of variance (PERMANOVA) and non-metric multidimensional scaling (NMDS) on the Bray-Curtis dissimilarities to assess overall differences in communities. These analyses used the R package “vegan”(Oksanen et al. 2016). To see whether some organisms at as “indicators” for a particular wetland type, were performed a multi-level pattern analysis using the multipat function from the “indicspecies” R package (Cáceres and Jansen 2016).

Table 4 Predictor variables for GLMMs of log-transformed invertebrate CPUE.

|  |  |  |
| --- | --- | --- |
| **Variable** | **Variable type** | **Description** |
| Region | Categorical | Region of the estuary as shown on Figure 1 |
| Site type | Categorical | Depth and water management regime (diked wetland, tidal wetland, shallow open water, or channel) |
| Habitat type | Categorical | Depth of water and presence of vegetation (Emergent wetland, SAV, FAV, open-water, or benthic) |
| Site (error term) | Categorical | Identity of wetland site, sued as an error term to prevent pseudoreplication. |

To answer Question 3, we will analyze the four sampling events from Decker Island to see when CPUE and BPUE of fish food invertebrates are maximal. We will test the fit of linear and quadratic equations of catch versus date to see when abundance peaks. We also tested the fit of a linear model of catch versus Sacramento river flow to see whether flow was a better predictor of abundance than date (using DAYFLOW calculations: <https://water.ca.gov/Programs/Environmental-Services/Compliance-Monitoring-And-Assessment/Dayflow-Data>). We graphically compared the trend in macroinvertebrate catch to the catch of Chinook Salmon smolts caught by the USFWS Chipps Island survey (USFWS data available <https://www.fws.gov/lodi/juvenile_fish_monitoring_program/jfmp_index.htm>) and the catch of Delta Smelt caught by the Spring Kodiak Trawl (CDFW data available <https://www.wildlife.ca.gov/Conservation/Delta/Spring-Kodiak-Trawl>).

We analyzed the data from the subset of stations with both Spring and Fall sampling by analyzing log-transformed CPUE of macroinvertebrate abundance using GLMMs similar to those used for the spring dataset. To detect differences in community composition, we used PERMANOVA and NMDS to assess overall differences in communities. We then qualitatively compared the differences in communities to organisms found in Delta Smelt diets for the life stage at observed in proximity to the sites using published diet studies (Slater and Baxter 2014).

## Results

FMWT caught no Delta Smelt during 2018. EDSM caught 89 Delta Smelt in September and October, most of which were either in the Sacramento Deep Water Ship Channel, or the Lower Sacramento River (Table 5). This was similar to the distribution predicted in our workplan, so our fall sampling proceeded as planned, targeting the Confluence and Cache Slough Complex.

Table 5. EDSM Delta Smelt catch from September and October, 2018 (USFWS, data available <https://www.fws.gov/lodi/juvenile_fish_monitoring_program/jfmp_index.htm>) .

|  |  |  |
| --- | --- | --- |
| **EDSM Stratum** | **September** | **October** |
| Lower Sacramento | 17 | 6 |
| Sac DW Ship Channel | 38 | 26 |
| Suisun Marsh | 2 | 0 |
| **Grand Total** | **57** | **32** |

Macroinvertebrate abundance, zooplankton abundance, and chlorophyll during the spring sampling period were all highly variable, both within sites and between sites (Figure 6, Table 10). There was significantly higher catch in the mysid net, zooplankton net, and sweep nets in diked wetlands than any other habitat type. There was also significantly higher chlorophyll in diked wetlands (Figure 9, Table 6). However, there was significantly lower clam abundance in diked wetlands than other site types (Table 6, Figure 7). There was significantly higher catch of clams and mysids in 2018 than 2017, but no other significant differences in abundance between years. Region of the estuary was included in the models for zooplankton and clam abundance, however there were no significant differences between regions for either parameter (Table 6). Within the sweep net samples, EAV had the lowest average CPUE, FAV had significantly higher CPUE, and SAV had the highest CPUE (Table 6). Model selection for Neuston tows did not select any of the potential explanatory variables, choosing the intercept-only model instead.

There were more clear patterns in community composition than CPUE. There were significant differences in Region, Year, and Site Type for all the parameters tested (Figure 11, Figure 10, Figure 12, Table 9). NMDS plots indicate some differences between communities by site type, however the separation between the hulls varies by gear type, and most of the hulls overlap (Figure 13, Figure 14, Figure 15, Figure 16). This indicates that the communities share some components but also have some unique taxa or unique combinations of taxa. The indicator species analysis found 13 indicator taxa for diked wetlands, 10 indicator taxa for muted tidal wetlands, three taxa for tidal wetlands, and no indicator taxa for channel habitat (Table 8).

The GLM of macroinvertebrate CPUE at Decker island over the course of the spring found a significant interaction between Day of the Year and Year, such that invertebrate CPUE increased over the spring in 2017 but decreased slightly in 2018 (Table 12). Without the interaction term, there was a slight, non-significant, positive effect of Day of the Year. Data from EMP’s Zooplankton Study Mysid data show a similar pattern (Figure 18). When flow in the Sacramento River was added to the model, there was a clear negative relationship between flow and CPUE, and flow was a better predictor than Day of the Year when all possible models were ranked with AICc (Table 13, Figure 19). Catch of adult Delta Smelt in the Spring Kodiak Trawl peaks in January, whereas catch of Chinook Salmon smolts peaks in May (Figure 18).

Four sites had fall macroinvertebrate sampling events in addition to the spring sampling event. The GLM found no significant difference between total invertebrate CPUE between the spring and fall sampling events, however Prospect Island had significantly higher catch than the other sites, and the sweep net had significantly higher catch than the mysid net (Table 13). There were larger differences in community composition. The PERMANOVA found a highly significant effect of season, with more cumaceans, collembola, and larval fish in the spring, and more amphipods and isopods in the fall (Table 14, Figure 21). The NMDS plot shows that hulls for season overlap somewhat, but have some unique NMDS space by season (Figure 22). There were also significant differences in community composition between sites, with Prospect Island having particularly low overlap in the NMDS plot (Figure 22, Table 14).

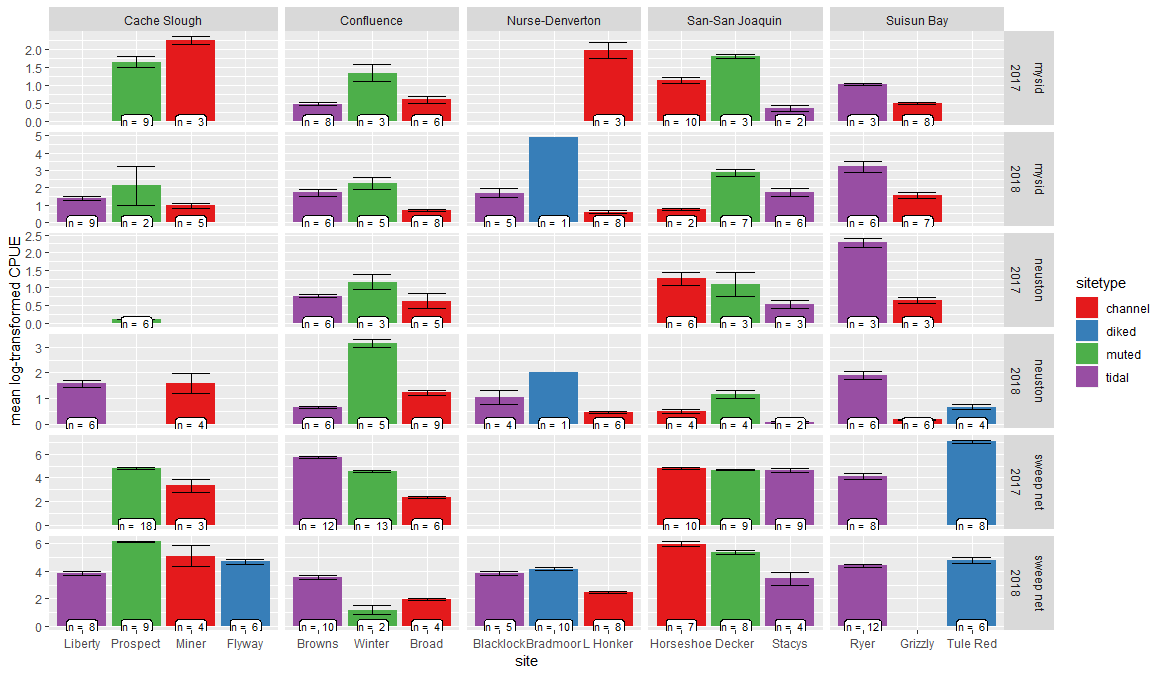


Figure 6 – Log-transformed CPUE of macroinvertebrates collected during the spatially intensive spring sampling events of 2017 and 2018, +/- one standard error. Data are separated by site, year, and gear type.

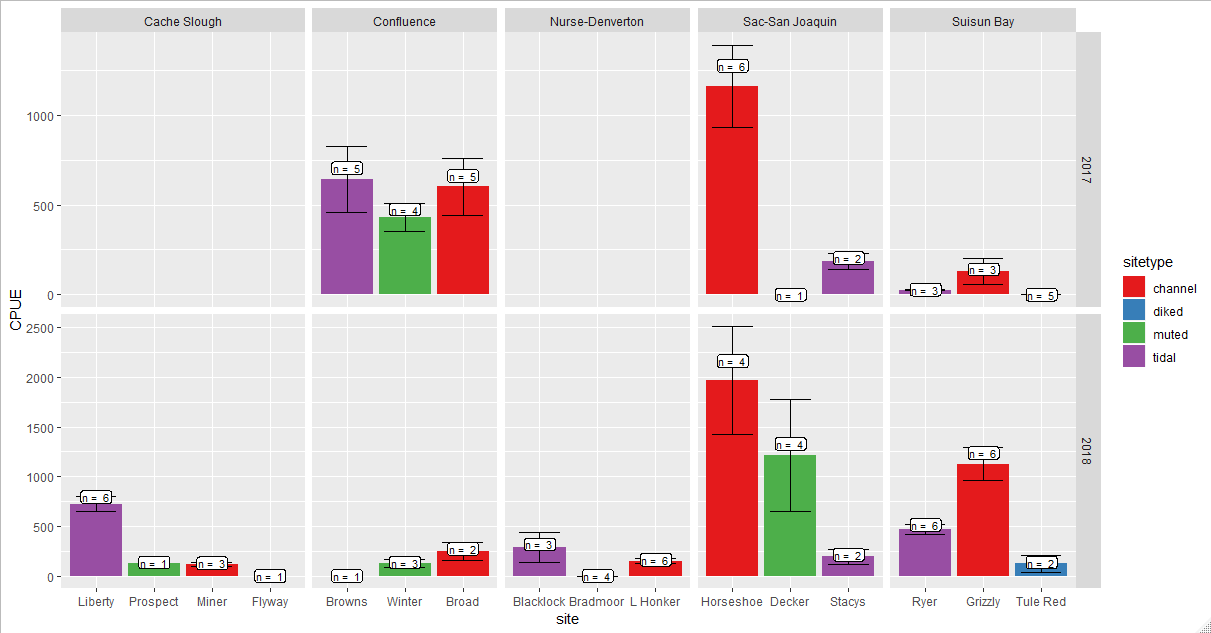


Figure 7 - Catch of clams per square meter of substrate in 2017 and 2018.

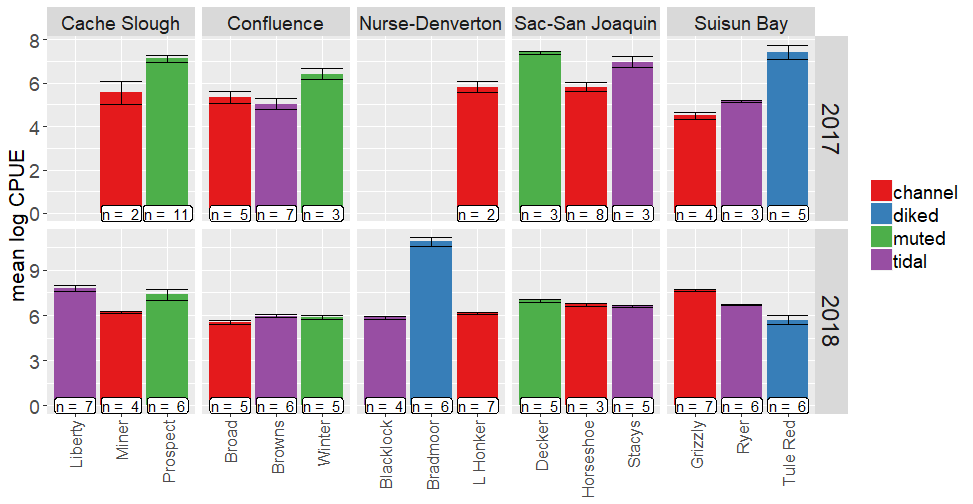


Figure 8 - Mean log-transformed CPUE of zooplankton catch collected during the intensive spring sampling period of 2017 and 2018.



Figure 9 - mean log-transformed concentration of Chlorophyll (ug/L) measured by florescence during spring of 2018.

Table 6 - GLMM for each ecosystem component. Predictor variables were chosen via AICc selection and site was included as an error term in each model. Response variables were log-transformed to conform with model assumptions, where necessary.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| Mysid catch per m3 |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept: channel, 2017) | 0.737 | 0.207 | 12.371 | 3.561 | 0.004 | \*\* |
| site type: muted | 1.079 | 0.307 | 9.365 | 3.519 | 0.006 | \*\* |
| site type: tidal | 0.427 | 0.272 | 9.640 | 1.570 | 0.149 |  |
| Year: 2018 | 0.517 | 0.204 | 122.198 | 2.532 | 0.013 | \* |
|  |  |  |  |  |  |  |
| Sweep net catch per m3 |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept: EAV, channel) | 3.123 | 0.327 | 11.886 | 9.551 | <0.0001 | \*\*\* |
| Veg type: FAV | 1.423 | 0.257 | 192.226 | 5.536 | <0.0001 | \*\*\* |
| Veg type: SAV | 2.197 | 0.252 | 191.961 | 8.729 | <0.0001 | \*\*\* |
| site type: diked | 1.548 | 0.495 | 11.386 | 3.128 | 0.009 | \*\* |
| site type: muted | 0.890 | 0.471 | 9.432 | 1.891 | 0.090 | . |
| site type: tidal | 0.693 | 0.428 | 10.515 | 1.621 | 0.135 |  |
|  |  |  |  |  |  |  |
| Neuston net catch per m3 |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept) | 1.056 | 0.193 | 12.040 | 5.482 | <0.0001 | \*\*\* |
|  |  |  |  |  |  |  |
| Zooplankton catch per m3 |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept: Cache, Channel, 2017) | 6.345 | 0.788 | 7.414 | 8.057 | 0.000 | \*\*\* |
| Region: Confluence | -1.281 | 0.888 | 6.462 | -1.444 | 0.196 |  |
| Region: Nurse-Denverton | -0.102 | 1.005 | 7.256 | -0.101 | 0.922 |  |
| Region: Sac-San Joaquin | -0.283 | 0.892 | 6.584 | -0.317 | 0.761 |  |
| Region: Suisun Bay | -1.200 | 0.983 | 6.651 | -1.221 | 0.264 |  |
| Site type: diked | 2.560 | 1.009 | 6.623 | 2.537 | 0.041 | \* |
| Site type: muted | 0.737 | 0.841 | 6.456 | 0.876 | 0.412 |  |
| Site type: tidal | 0.330 | 0.695 | 6.739 | 0.475 | 0.6501 |  |
| year: 2018 | 0.451 | 0.245 | 124.870 | 1.839 | 0.0682 | . |
|  |  |  |  |  |  |  |
| Clam catch per m2 |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept: Cache, Channel, 2017) | 4.791 | 1.028 | 23.387 | 4.662 | 0.000 | \*\*\* |
| Region: Confluence | -0.225 | 1.071 | 14.446 | -0.210 | 0.837 |  |
| Region: Nurse-Denverton | -1.499 | 1.039 | 12.275 | -1.444 | 0.174 |  |
| Region: Sac-San Joaquin | 0.464 | 1.028 | 11.876 | 0.451 | 0.660 |  |
| Region: Suisun Bay | -0.018 | 0.963 | 9.218 | -0.019 | 0.985 |  |
| Site type: diked | -4.609 | 0.925 | 7.982 | -4.980 | 0.001 | \*\* |
| Site type: muted | -1.642 | 0.905 | 7.409 | -1.815 | 0.110 |  |
| Site type: tidal | -0.891 | 0.673 | 6.502 | -1.323 | 0.230 |  |
| Year: 2018 | 1.276 | 0.582 | 75.740 | 2.193 | 0.031 | \* |
|  |  |  |  |  |  |  |
| Chlorophyll fluorescence in ug/L |  |  |  |  |  |  |
|  | Estimate | Std. Error | DF | t value | P value |  |
| (Intercept: channel) | 1.577 | 0.260 | 12.062 | 6.073 | 0.000 | \*\*\* |
| Site type: diked | 1.475 | 0.426 | 12.292 | 3.462 | 0.005 | \*\* |
| Site type: muted | -0.381 | 0.426 | 12.232 | -0.894 | 0.388 |  |
| Site type: tidal | -0.207 | 0.370 | 12.341 | -0.559 | 0.586 |  |

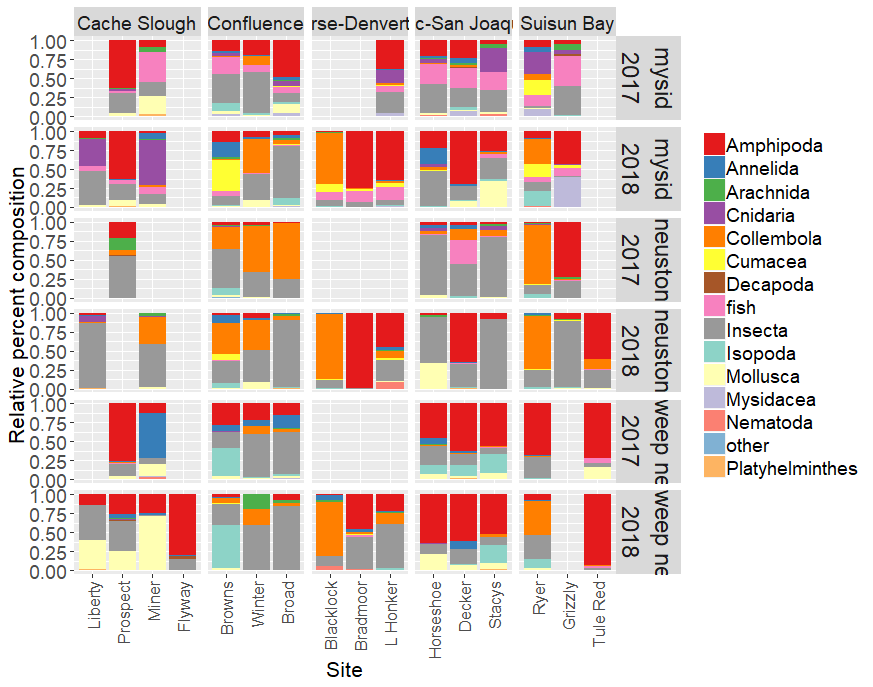


Figure 10. Relative composition of macroinvertebrate samples separated by gear type, site, and year for sampling during the intensive spring sampling period in 2017 and 2018.

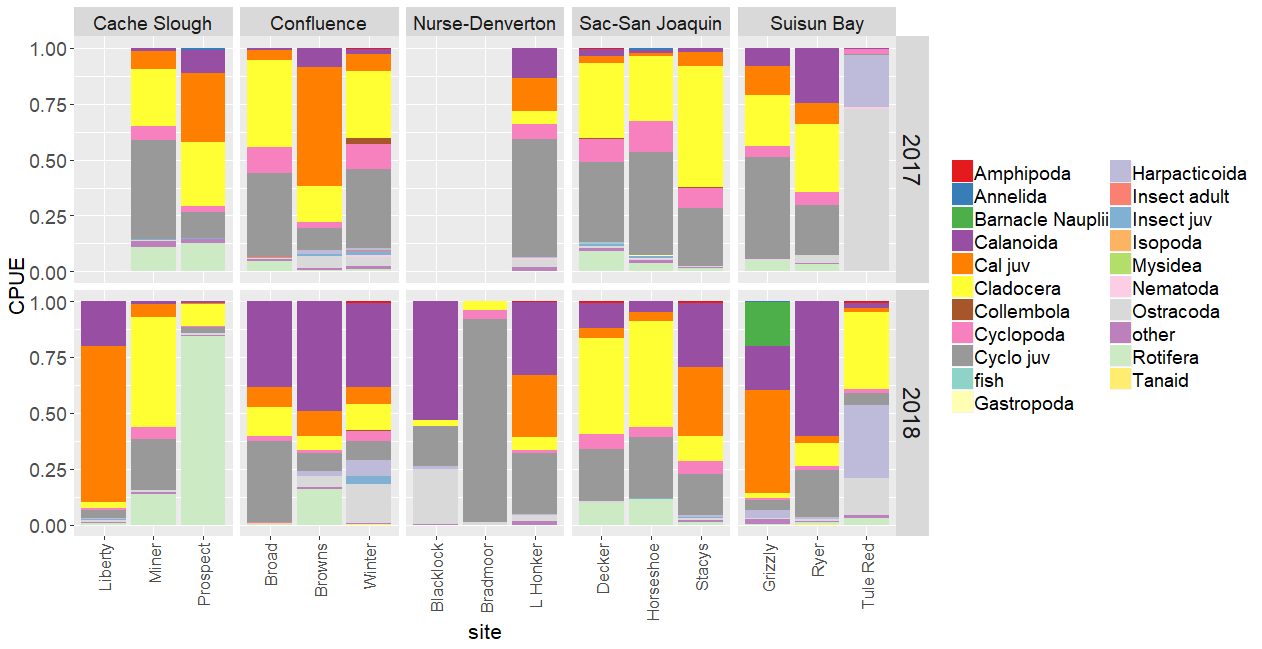


Figure 11 - Relative composition of zooplankton samples separated by site, and year for sampling during the intensive spring sampling period in 2017 and 2018.



Figure 12 - Relative composition of phytoplankton samples separated by site and year for sampling during the intensive spring sampling period in 2017 and 2018.

Table 7 - Multilevel pattern analysis of invertebrate associations with wetlands of differing types.

|  |  |  |  |
| --- | --- | --- | --- |
| **Diked Wetlands** |  |  |  |
|  | **Stat** | **P value** |  |
| *Eogammarus* spp. | 0.361 | 0.001 | \*\*\* |
| Dytiscidae | 0.306 | 0.001 | \*\*\* |
| Mosquitofish | 0.272 | 0.001 | \*\*\* |
| Hydrophilidae | 0.272 | 0.001 | \*\*\* |
| Corixidae | 0.259 | 0.001 | \*\*\* |
| Threespine Stickleback | 0.244 | 0.001 | \*\*\* |
| *Procambarus clarkii* | 0.233 | 0.001 | \*\*\* |
| Prickly Sculpin | 0.226 | 0.001 | \*\*\* |
| *Americorophium spinicorne* | 0.218 | 0.001 | \*\*\* |
| Notonectidae | 0.195 | 0.004 | \*\* |
| Hydrobiidae | 0.159 | 0.003 | \*\* |
| Hydroscaphidae | 0.152 | 0.007 | \*\* |
| Hemiptera Other | 0.137 | 0.041 | \* |
|  |  |  |  |
| **Muted Tidal Wetlands** |  |  |  |
|  | **Stat** | **P value** |  |
| Mesoveliidae | 0.259 | 0.001 | \*\*\* |
| *Palaemonetes* spp. | 0.203 | 0.002 | \*\* |
| Hymenoptera adult | 0.168 | 0.015 | \* |
| Libellulidae larvae | 0.162 | 0.02 | \* |
| Coenagrionidae larvae | 0.157 | 0.016 | \* |
| Diptera adult | 0.153 | 0.029 | \* |
| *Ferrissia* spp. | 0.146 | 0.034 | \* |
| Ceratopogonidae larvae | 0.146 | 0.021 | \* |
| Tipulidae larvae | 0.146 | 0.026 | \* |
| Vellidae adult | 0.127 | 0.028 | \* |
|  |  |  |  |
| **Tidal Wetlands** |  |  |  |
|  | **Stat** | **P value** |  |
| *Gnorimosphaeroma* | 0.22 | 0.001 | \*\*\* |
| Collembola | 0.151 | 0.026 | \* |
| Tanaidacea | 0.131 | 0.041 | \* |



Figure 13 - Mysid NDMS of bray-curtis dissimilarities, stress = 0.2276. Two convergent solutions found after 100 tries.



Figure 14 – NMDS of bray-curtis dissimilarities for neuston tow samples. Stress = 0.122. Two convergent solutions found after 63 tries.



Figure 15 - NMDS of Bray-Curtis dissimilarities for sweep net data (2018 only). Stress = 0.153. Two convergent solutions found fter 624 tries.



Figure 16 - NMDS of of Bray-Curtis dissimilarities for phytoplankton taxa scaling using Bray-Curtis Dissimilarity index. Stress = 0.227. Two convergent solutions after 99 tries.

Table 8 – PerMANOVA of relative percent composition of taxa for each ecosystem component.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Mysids** |  |  |  |  |  |  |  |
|  | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| Site type | 2 | 2.210 | 1.105 | 5.753 | 0.068 | 0.001 | \*\*\* |
| Year | 1 | 0.767 | 0.767 | 3.995 | 0.024 | 0.004 | \*\* |
| Region | 4 | 3.085 | 0.771 | 4.016 | 0.096 | 0.004 | \*\* |
| Site | 6 | 3.160 | 0.527 | 2.742 | 0.098 | 0.001 | \*\*\* |
| Residuals | 120 | 23.046 | 0.192 |  | 0.714 |  |  |
| Total | 133 | 32.268 |  |  | 1.000 |  |  |
|  |  |  |  |  |  |  |  |
| **Sweepnets** |  |  |  |  |  |  |  |
|  | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| Site type | 3 | 5.022 | 1.674 | 13.546 | 0.121 | 0.001 | \*\*\* |
| Year | 1 | 1.210 | 1.210 | 9.789 | 0.029 | 0.001 | \*\*\* |
| Vegetation type | 2 | 3.627 | 1.813 | 14.672 | 0.088 | 0.001 | \*\*\* |
| Region | 4 | 4.867 | 1.217 | 9.845 | 0.117 | 0.001 | \*\*\* |
| Site | 7 | 4.587 | 0.655 | 5.303 | 0.111 | 0.001 | \*\*\* |
| Residuals | 179 | 22.122 | 0.124 |  | 0.534 |  |  |
| Total | 196 | 41.436 |  |  | 1.000 |  |  |
|  |  |  |  |  |  |  |  |
| **Neuston** |  |  |  |  |  |  |  |
|  | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| site type | 3 | 2.517 | 0.839 | 6.923 | 0.139 | 0.001 | \*\*\* |
| Year | 1 | 0.172 | 0.172 | 1.420 | 0.010 | 0.263 |  |
| Region | 4 | 2.380 | 0.595 | 4.910 | 0.132 | 0.001 | \*\*\* |
| Site | 7 | 2.589 | 0.370 | 3.052 | 0.143 | 0.001 | \*\*\* |
| Residuals | 86 | 10.422 | 0.121 |  | 0.576 |  |  |
| Total | 101 | 18.079 |  |  | 1.000 |  |  |
|  |  |  |  |  |  |  |  |
| **Zooplankton** |  |  |  |  |  |  |  |
|  | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| Region | 4 | 4.687 | 1.172 | 9.303 | 0.157 | 0.001 | \*\*\* |
| Site type | 3 | 4.355 | 1.452 | 11.526 | 0.146 | 0.001 | \*\*\* |
| year | 1 | 1.571 | 1.571 | 12.469 | 0.052 | 0.001 | \*\*\* |
| site | 7 | 3.951 | 0.564 | 4.481 | 0.132 | 0.001 | \*\*\* |
| Residuals | 122 | 15.367 | 0.126 |  | 0.513 |  |  |
| Total | 137 | 29.930 |  |  | 1.000 |  |  |
|  |  |  |  |  |  |  |  |
| **Phytoplankton** |  |  |  |  |  |  |  |
|  | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| Site type | 3 | 5.127 | 1.709 | 7.830 | 0.109 | 0.001 | \*\*\* |
| year | 1 | 3.050 | 3.050 | 13.976 | 0.065 | 0.001 | \*\*\* |
| Region | 4 | 5.920 | 1.480 | 6.782 | 0.126 | 0.001 | \*\*\* |
| site | 7 | 6.510 | 0.930 | 4.262 | 0.138 | 0.001 | \*\*\* |
| Residuals | 121 | 26.408 | 0.218 |  | 0.562 |  |  |
| Total | 136 | 47.015 |  |  | 1.000 |  |  |

Table 9 - coefficients of variation for each sample type in each year.

|  |  |  |  |
| --- | --- | --- | --- |
| **Sample Type** | **Year** | **Mean within-site CV** | **Mean between-site CV** |
| benthic | 2017 | 0.826577536 | 1.894361 |
| benthic | 2018 | 1.015171266 | 1.212494 |
| mysid | 2017 | 0.740843281 | 0.922856 |
| mysid | 2018 | 1.367034748 | 1.550785 |
| neuston | 2017 | 0.970689418 | 0.995882 |
| neuston | 2018 | 0.933620472 | 1.445109 |
| sweep net | 2017 | 1.125644042 | 1.47165 |
| sweep net | 2018 | 1.039019521 | 1.416646 |
| zooplankton | 2017 | 0.9318 | 1.4391 |
| zooplankton | 2018 | 0.7345 | 3.2851 |



Figure 17 – CPUE of macroinvertebrates at Decker Island over the course of the spring.

Table 10 - GLMM of log total CPUE of macroinvertebrate samples collected at Decker over the course of the spring of 2017 and 2018. Preliminary analyses found no significant effect of Julian Day unless the interaction term was included.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |  |
| (Intercept – mysids, 2017) | 0.284 | 0.481 | 0.591 | 0.557 |  |
| Gear - neuston | -1.174 | 0.351 | -3.343 | 0.002 | \*\* |
| Gear – EAV sweep net | 2.042 | 0.341 | 5.985 | 0.000 | \*\*\* |
| Julian Day | 0.018 | 0.006 | 3.114 | 0.003 | \*\* |
| Year - 2018 | 2.458 | 0.619 | 3.968 | 0.000 | \*\*\* |
| Julian Day \*Year | -0.022 | 0.007 | -3.186 | 0.003 | \*\* |

Table 11 - GLM of log total CPUE of macroinvertebrate samples at Decker. AICc model selection found Sacramento river flow to be a better predictor than Julian Day.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |  |
| (Intercept: mysid) | 2.742 | 0.336 | 8.163 | 0.000 | \*\*\* |
| Gear: neuston | -1.030 | 0.363 | -2.836 | 0.007 | \*\* |
| Gear: sweep net | 1.960 | 0.358 | 5.484 | 0.000 | \*\*\* |
| Saramento River Flow (CFS) | -1.816E-05 | 5.501E-06 | -3.302 | 0.002 | \*\* |

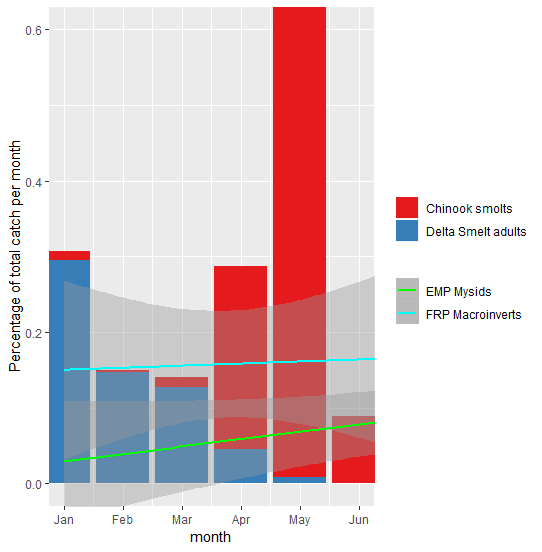


Figure 18 - Distribution of catch per month for adult Delta Smelt (SKT survey, 2002-2018), Chinook Salmon smolts (chipps island survey, 2002-2018) and macroinvertebrates at Decker Island (FRP data, 2017-2018)

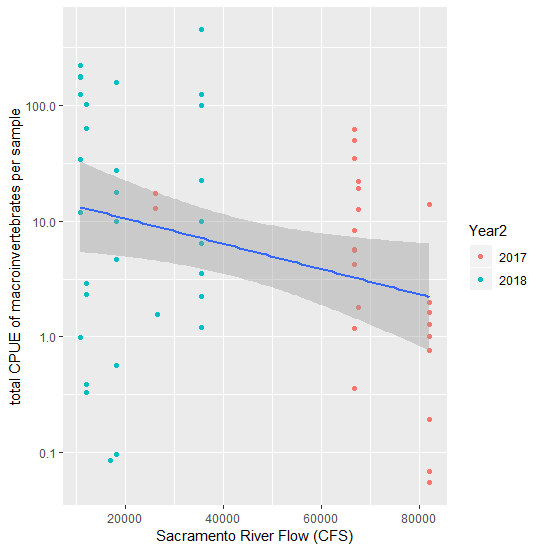


Figure 19 - Macroinvertebrate catch versus Sacramento River flow (CFS) for samples collected at Decker Island in spring of 2017 and 2018. Flow from CDWR’s Dayflow calculations.



Figure 20 - Mean log-transformed CPUE of mysid and sweep net samples in the fall versus spring of 2018. +/- 1 SEM.

Table 12 - GLMM of log-transformed CPUE of invertebrates collected during the spring sampling period versus the fall sampling period.

|  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |  |
| (Intercept – mysid, Ryer, spring) | 1.563 | 0.528 | 2.959 | 0.004 | \*\* |
| Site: Browns | -0.999 | 0.589 | -1.698 | 0.093 | . |
| Site: Winter | -1.157 | 0.674 | -1.718 | 0.090 | . |
| Site: Prospect | 1.748 | 0.630 | 2.775 | 0.007 | \*\* |
| Gear: sweep net | 4.250 | 0.470 | 9.037 | <0.0001 | \*\*\* |
| Season: fall | 0.471 | 0.466 | 1.011 | 0.315 |  |

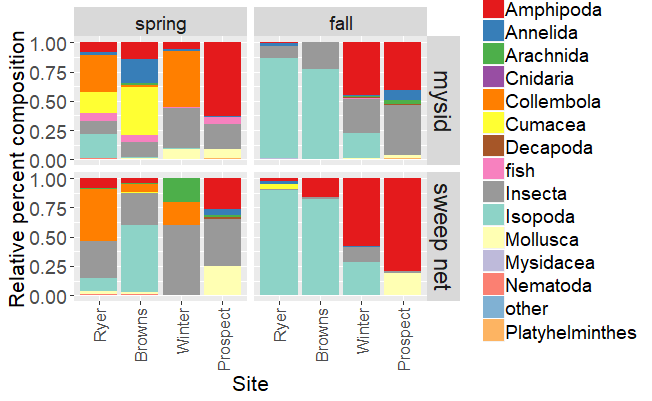


Figure 21 - relative percent composition of spring verses fall macroinvertebrates.

Table 13 - PerMANOVA comparing site, geartype, and season for macroinvertebrate samplig in 2018

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Factor** | **DF** | **SumsOfSqs** | **MeanSqs** | **F value** | **R2** | **P value** |  |
| Site | 3 | 5.6911 | 1.897 | 11.1194 | 0.24275 | 0.001 | \*\*\* |
| Gear | 1 | 0.7332 | 0.7332 | 4.2978 | 0.03128 | 0.002 | \*\* |
| season | 1 | 3.2007 | 3.2007 | 18.7607 | 0.13652 | 0.001 | \*\*\* |
| Residuals | 81 | 13.8191 | 0.1706 |  | 0.58945 |  |  |
| Total | 86 | 23.4442 |  |  | 1 |  |  |

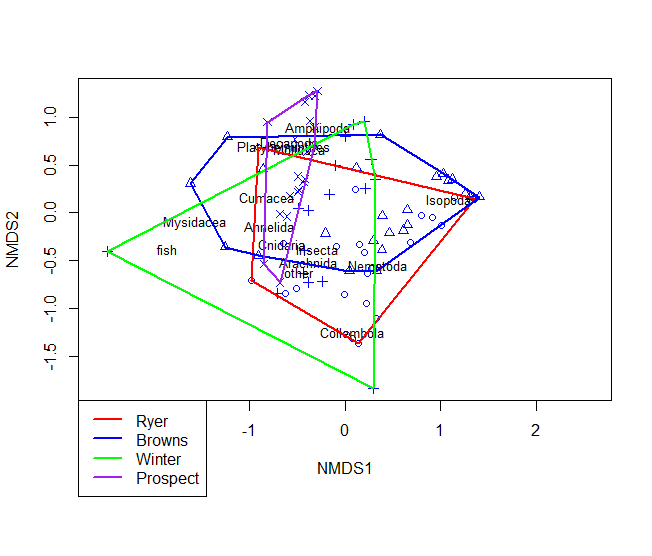
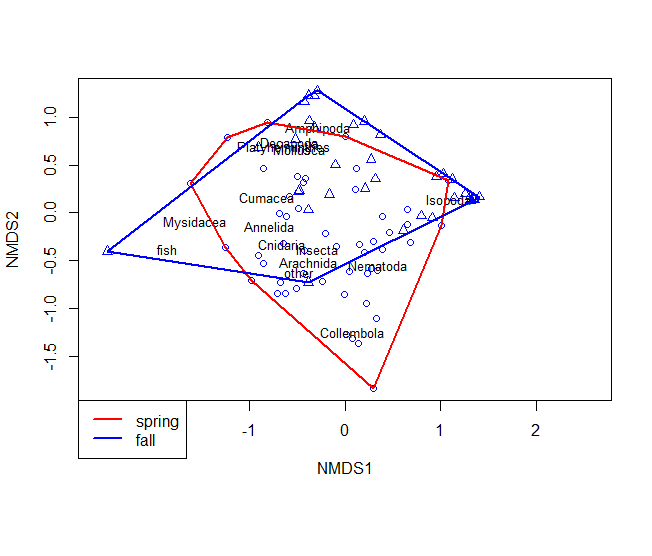


Figure 22. NMDS plot of Bray-Curtis dissimilarity indices of community composition of invertebrate data. Stress = 0.182, two convergent solutions found after 50 tries. With A) Hulls are drawn around samples from spring versus fall. and B) Hulls drawn around samples from different sites.

## Discussion

### Inter-annual differences

There were some differences in invertebrate and phytoplankton communities in 2018 versus 2017, most likely driven by the difference in water flow. There was significantly higher catch overall in mysid nets and benthic cores during the intensive spring sampling event (Table 6), and most ecosystem components also had significantly different community compositions (Table 8). The data from Decker Island provides a potential explanation for this difference. While there was a pattern of increased catch over the course of the spring in 2017, there was no similar increase in 2018, and the total catch was highly correlated with flow in the Sacramento River (Figure 18). Flow during 2017 was much higher than 2018 throughout the spring, and high river flows may serve to dilute existing productivity, while short residence times and cooler temperatures may slow growth of phytoplankton and zooplankton (Downing et al. 2016; Glibert et al. 2014). This pattern has been described many times throughout the estuary, though it is dependent on the study species and region (Cloern et al. 2010; Kimmerer 2002; Kimmerer et al. 2018b; Sommer et al. 2004).

Overall, a greater proportion of the macroinvertebrate catch was larval fish and amphipods in 2017, whereas more cumaceans and isopods were caught in 2018 (Figure 9), but the differences in community composition tended to describe a small proportion of the variance (less than 10%, Table 8). This is similar to research by Howe et al (Howe et al. 2014), who also found sampling year to describe around 5% of variation in invertebrate communities. Broad-scale classification of benthic invertebrates also show communities tend to be fairly similar year-to-year, with greater differences between geographic regions or substrate types (Thompson et al. 2013). We will be able to make a more substantial analysis of the sources of year-to-year variation once more years of data have been collected.

### Differences between site types

Total catch of macroinvertebrates and chlorophyll was higher in diked wetlands than any other site type (Figure 6, Table 6), though diked wetlands also had significantly lower clam abundance than channel habitat (Table 6). Similar to inter-annual differences in catch, the difference in total catch of between site types may also be explained by water residence times. Diked wetlands, with long residence times, have ample opportunity to grow large standing stocks of organic material, phytoplankton, and zooplankton, though extremely long residence times may deplete nutrients (Brown et al. 2016; Herbold et al. 2014), and low dissolved oxygen conditions may dominate during the summer and fall (Moyle et al. 2014). Similar patterns of increased chlorophyll in disconnected wetlands have also been found in floodplains in the region (Ahearn et al. 2006). Despite the high concentrations of phytoplankton and zooplankton, production in diked wetlands may not benefit pelagic fish species if fish are excluded from the site. Export of production from the site only occurs when the site is being drained, usually over the span of a week or two during the spring (Moyle et al. 2014). Also, even though chlorophyll concentrations in diked wetlands were high relative to other site types, they were still less than the 10 ug/L considered necessary for high zooplankton production (Brown et al. 2016).

Tidal and muted-tidal wetlands have also demonstrated greater concentrations of organic material and chlorophyll than open-water sites in other studies (Lehman et al. 2015; Muller-Solger et al. 2002; Strong 2015), however, in many tidal systems benthic grazing may deplete phytoplankton biomass (Lucas and Thompson 2012). In our study, diked wetlands had high abundance of invertebrates and high concentrations of chlorophyll, but also the lowest abundance of invasive clams in benthic samples (Figure 7). Therefore, the high chlorophyll content is likely a combination of high productivity and low benthic grazing rates. We did not see higher invertebrate abundance in tidal wetlands than diked wetlands or channel habitat, in contrast to results from 2017 (Contreras et al. 2018). This may be because the data were more variable in 2018 than 2017 (Table 11), making it more difficult to see trends, or it may be due to inter-annual differences in weather and hydrology (as seen in Cloern et al. 2010). More years of data may better elucidate these trends.

Wetlands also had significantly different community composition from the channel habitat. Zooplankton communities were very unusual at the diked wetland of Tule Red in particular, where large numbers of ostracods and harpacticoid copepods dominated the samples. Harpacticoid copepods occur occasionally in Delta Smelt diets (Slater and Baxter 2014), and may more common in the diets of other fishes (Howe et al. 2014), but are not consumed as frequently as calanoid copepods. Ostracods are also rare in smelt diets, but may occur in salmon diets in greater frequency than in the surrounding habitat (Katz et al. 2017). Diked wetlands, also had a greater relative abundance of amphipods than other sites, while tidal wetlands had a greater relative abundance of collembola and isopods. Amphipods and isopods are generally epibenthic or epiphytic, and while they occur less frequently in smelt diets than copepods, they are increasing in dietary importance (Brown et al. 2016; Slater and Baxter 2014). Collembola and isopods are found frequently in salmon diets, and Collembola, along with other terrestrial fall-out invertebrates, may be a significant portion of salmon diets (David et al. 2016a; David et al. 2016b).

The phytoplankton communities were also unique in diked wetlands. While most tidal and channel habitats had large proportions of pennate and centric diatoms, diked wetlands had more variable communities, including some samples with large numbers of dinoflagellates (see Bradmoor in 2018), or chrysophytes (see Tule Red in 2017). The large proportion of diatoms in all the samples was somewhat unexpected, since the relative concentration and absolute abundance of diatoms has been in decline since the introduction of the invasive clam, *Potamocorbula*, in the 1980s (Lehman 2000; Winder and Jassby 2011). Dinoflagellates, chrysophytes, and cyanobacteria that dominated diked wetlands are not thought to be preferred zooplankton food (Kimmerer et al. 2018a), though there is insufficient research on zooplankton feeding.

The large number of indicator macroinvertebrate species identified for diked wetlands most likely resulted from the lack of connection to the surrounding area and difference in water quality components. Diked wetlands may experience low dissolved oxygen conditions, larger swings in temperatures, and lower pH than surrounding tidal waters (Brown et al. 2016; Moyle et al. 2013). Therefore, taxa that thrive in tidal may not be able to thrive in impounded water and vise versa. Tidal wetlands share a direct connection with the surrounding sloughs, so only two taxa were identified as indicators.

### Intra-annual differences

#### Timing of spring sampling

During 2017, total macroinvertebrate biomass increased monotonically over the course of the spring. This trend led us to select April as the month with the greatest overlap between adult Delta Smelt, juvenile Chinook Salmon, and macroinvertebrates (Contreras et al. 2018). However, in 2018 we found the opposite trend between day of year and invertebrate abundance, with a slight decrease over the course of the spring (Table 10). AICc model selection indicated that flow in the Sacramento River was a better predictor than Julian Day when data from both years were analyzed together (Table 11). There was a clear trend towards lower invertebrate biomass during high flows (Figure 16). As noted above, many studies have seen a similar inverse relationship between invertebrate abundance and flow (Kimmerer 2002; Kimmerer et al. 2018b; Sommer et al. 2004). While flow did decrease over the course of the spring in 2018, the maximum flows were never as high as in 2017 (Figure 19), so the slight decreasing trend in invertebrate abundance is most likely an artifact of the high variability of invertebrate biomass.

Besides the differences in flow, some of the difference in invertebrate catch may have been due to the presence or absence of aquatic vegetation in the samples. We specifically excluded SAV sweep nets from the analysis because they were not collected in all months, and SAV sweep nets tended to have an order of magnitude higher catch than some of the other gear types (Figure 6), and many researchers have found similar high invertebrate productivity in SAV (Boyer et al. 2013; Young et al. 2018). However, the mysid net and neuston net would also occasionally hit patches of aquatic vegetation, resulting in much higher total invertebrate CPUE. We did not collect quantitative data on abundance of vegetation in 2017 or 2018, but it has been added to our sampling plan in 2019 and 2020 to attempt to refine this problem.

Other researchers with larger data sets support an increase in overall abundance over the course of the spring, similar to our results in 2017. The IEP Zooplankton Study shows an overall increase in zooplankton and macroinvertebrate abundance during the spring, peaking in July (Figure 17) (Hennessy and Enderlein 2013). Studies of floodplains and wetland habitat also show increases in abundance of zooplankton during the spring, and associated with lower flows (Sommer et al. 2004). Even benthic and drift invertebrates often have summer peaks in abundance (Howe et al. 2014). Therefore, negative trend for macroinvertebrate abundance in 2018 is most likely anomalous.

Fish trends are complicated. Salmon smolt abundance in the Delta usually peaks in May (Brandes and McLain 2000), but number of fry versus smolts depends on water year type. Total production of juvenile salmonids is higher in wet years, but more of them enter the estuary as fry rather than smolts, and may move through the Delta faster (Brandes and McLain 2000). Juvenile survival is also higher in wet years versus dry years, especially along the Sacramento River(Michel et al. 2015). However, higher survival may be because smolts move quickly down the channelized river, exposing themselves to less predation but also spending less time in wetlands. Invertebrate production in 2017 was low during the high flows in January and February, but by the time salmon outmigration peaked in May, invertebrate production had increased and was similar to 2018 (Figure 19).

The complicated relationship between invertebrate biomass, time of year, and flow, warrants more investigation. We only planned a single spring sampling event in 2019 because we had not analyzed the 2018 data before developing our workplan. We expected 2018 data to be similar to 2017, with increasing biomass over the course of the spring. Given the difference in temporal trends between 2017 and 2018, we are planning to repeat the series of four sampling events in 2020.

#### Spring versus fall

Overall invertebrate abundance was similar in spring and fall, but the species composition was significantly different. High abundances of collembola and cumaceans in spring were replaced by isopods in the fall, though insects and amphipods remained abundant throughout. These differences in invertebrate composition may result in different use of wetland invertebrates by different life stages of at-risk fishes.

Fall invertebrate abundance is particularly important for rearing Delta Smelt, since fall food resources have been identified as a limiting factor in population growth (Brown et al. 2014). We found amphipods and isopods to be especially abundant in the fall (Figure 21), and Delta Smelt diet studies found higher rates of consumption of amphipods in late summer and fall versus spring (Slater and Baxter 2014). Rearing Delta Smelt are most common in the Low Salinity Zone (1-6 PSU), which is geographically located in Suisun or the Confluence region in the fall, depending on water flow (Brown et al. 2014). However, the high abundance of the invasive clam *Potamocorbula amurensis* in Suisun means zooplankton abundance is generally low (Kimmerer and Lougee 2015). Therefore, wetlands, and wetland-derived invertebrates (such as amphipods) may be particularly important to Delta Smelt when in this region (as suggested by Hammock et al. 2019).

Total juvenile salmon abundance and biomass peaks in the spring, generally Mar-June, however spring out-migrants are mostly fall-run Chinook. Late-fall run, winter-run, and some spring-run Chinook smolts out-migrate November-April, so may be accessing the isopods, insects, and amphipods common in wetlands during the fall. (Yoshiyama et al. 1998). Chinook diets will shift rapidly based on the prey available in the surrounding environment, and they can consume large numbers of insects, amphipods, collembola, and isopods (Busby and Barnhart 1995; Duffy et al. 2010; Goertler et al. 2018). Insects and amphipods, in particular, are more energy-dense than zooplankton (Duffy et al. 2010; Tiffan et al. 2014), so provide valuable food resources for fish large enough to eat them.

### A note on neuston:

In 2017, we combined data from neuston tows with data from mysids tows and sweep nets to test for differences between site types. Power analysis showed poor ability for CPUE of neuston tows to differentiate between site types or regions even at increased sample size (Contreras et al. 2018). This year, we decided to conduct separate analyses on each sampling type, and the GLMM of neuston tow data failed to find any differences between years, regions, or site types (Table 6). However, neuston tows were able to show differences in community composition between site types in both 2017 and 2018. Fall-out invertebrates and terrestrial drift invertebrates have been identified as important components of salmon diets (David et al. 2014; Duffy et al. 2010), so while neuston tows may not be the most effective means of differentiating between restoration sites and surrounding channels, they may still be important in assessing availability of surface invertebrates for fish diets.

# Part 2: Channel-Shoal Gear Comparison

## Introduction

### Nutrients

### Zooplankton

Mesozooplankton are recognized as the largest component of Delta Smelt diets (Slater and Baxter 2014) and a significant component of salmon diets (Sommer et al. 2001). Our conceptual models postulate that tidal wetland restoration sites will have higher production and availability of zooplankton when compared with existing channel habitat and pre-project conditions (Hartman et al. 2017a). In order to support this hypothesis, we must compare zooplankton we collect within the wetland to zooplankton collected from the channels. We will leverage existing datasets from long-term monitoring programs currently sampling pelagic and channel habitat whenever possible, but we need a better understanding of how these samples compare to samples taken concurrently from adjacent wetlands. Water depth, substrate, presence of vegetation, presence of benthic grazers (clams), and differences in fish community may alter the zooplankton community (Bollens et al. 2014; Kimmerer and Thompson 2014). Furthermore, changes to the physical environment will affect the efficiency of our sampling gear.

By sampling wetlands concurrently with existing channel sampling, we can characterize some of these sources of variation. During Phase III sampling we conducted sampling of wetland habitat adjacent to eight of the long-term stations sampled by 20mm, however the extremely high water year of 2017 meant that results from that study may not be applicable to all years. Therefore, we repeated our channel-shallow comparisons using stations sampled by either 20mm, FMWT, or the Environmental Monitoring Program (EMP) (see Table 14). We tested for differences in mesozooplankton, macrozooplankton (mysids), and nutrients between channel habitats in which IEP samples and the shallow littoral habitats in which we sample. This will also give us a better understanding of the spatial variability in zooplankton in wetlands across the estuary.

Mesozooplankton questions:

1. How do mesozooplankton and macrozooplankton communities in the littoral and wetland habitat compare to open water habitat?
2. How do these communities change over the course of the year?
3. How do these communities change along the salinity gradient?

Nutrient questions:

1. Are there differences in nutrients, chlorophyll, and organic carbon concentrations between the wetland and the exterior channel?
2. Are nutrients limiting phytoplankton production?
3. Are excess nutrients a causal factor for harmful algal blooms on our sites?

### Fish

The extent to which at-risk fish species will benefit from tidal wetland restoration in the San Francisco Estuary is unknown (Brown 2003; Herbold et al. 2014). However, restored wetlands in other areas have shown to be productive food sources and provide refuge from predation (Gray et al. 2002; Shreffler et al. 1992; Simenstad and Cordell 2000). The FRP Monitoring Team was established to monitor the benefits of tidal wetland restoration to at-risk fish species in the San Francisco Estuary. Comparing fish communities and their condition pre- and post-construction can inform how at-risk fish benefit from tidal wetland restoration.

Littoral habitat provides benefits to at-risk fish species, such as salmon, which rear in littoral areas, and Delta Smelt, which inhabit the littoral zone to maintain their position during ebb tides when migrating (Bennett and Burau 2015; McLain and Castillo 2009). However, many of CDFW’s long term monitoring studies sample open water habitat due to gear size, boat size, and absence of vegetation. Sampling littoral and open water habitat simultaneously can provide insights into how fish species utilize different habitats. FRP sampled littoral habitat near planned tidal wetlands concurrently with mid-channel sampling by the IEP Summer Townet and Fall Midwater Trawl surveys.

## Methods

### IEP Surveys

The EMP survey monitors water quality, phytoplankton, meso-/marco-zooplankton, and benthic invertebrates in the upper estuary throughout the year. Zooplankton is collected using a steel sled with paired mesozooplankton (0.160 mm mesh) and macrozooplankton (0.500 mm mesh) nets. Phytoplankton is sampled using water collected from submersible pump.

The 20mm Survey monitors Delta Smelt distribution throughout their historical spring range in the Sacramento-San Joaquin Delta and San Francisco Estuary during the spring. The 20mm survey targets Delta Smelt in the post-larval and juvenile life stage, at lengths >20mm. The net is a cone shaped plankton net 5.1 meters in length with an opening circumference of 4.9 meters (1.5 cubic meters). Zooplankton is collected concurrently with a 0.160 mm mesh modified Clarke-Bumpus net mounted on the frame with its own flowmeter. The survey samples at 40 stations throughout the estuary and completes three 10-minute tows at each station. Zooplankton are only sampled at the first of these tows (Damon, 2015).

The Summer Townet Survey is composed of 12.7 mm stretch, knotted, nylon mesh 1.8 m long, tapering down to an additional 0.6 m "fyke". This "fyke" fits entirely within the second section, a 2.2 m section of woven mesh with approximately 8 holes per 24.7 mm. The net measures approximately 4.6 m in total, and is lashed directly to a fixed metal "D" frame. The townet was deployed in channel habitat from the stern of a boat. Before each tow, the cod end was tossed into the water when the boat was traveling at an idle speed. The front skis were lifted off the back deck until the net slid off the stern. Once the net and frame had been deployed it free-spooled based on site depth. Once the desired net depth was achieved, hydraulics were engaged and a 10-minute stepped oblique tow began. At the end of the 10-minute tow, the net was brought onboard and fish were released into a tub filed with water. Three tows were conducted if any fish were captured during the first two tows, except in the Sacramento Deep Water Shipping Channel where only two tows were completed. Fork lengths were measured to the nearest millimeter for all Striped Bass and Delta Smelt, and for the first 50 fish per tow for all other species.

The Fall Midwater Trawl Survey net is approximately 17.7 m long with mouth dimensions of 3.7 m x 3.7 m when stretched taught, but mouth dimensions are smaller when under tension during a tow. Net mesh sizes graduate in nine sections from 203.2 mm at the mouth to 12.7 mm stretch-mesh at the cod-end. The midwater trawl was deployed from the stern of the boat when the boat moved at an idle speed. As the rest of the net was deployed, two crew members each grabbed two planing doors. Each crewmember dropped a pair of doors into the water once the entire net was deployed. The net was then free-spooled into the water until the site depth was reached, at which a 12-minute continuous tow retrieval began. Once the net was approximately 7.6 m from the stern of the boat, it was brought onboard and all caught fish were released into a tub filled with water. Fork lengths were measured to the nearest millimeter for all Striped Bass and Delta Smelt, and for the first 50 fish per tow for all other species.Beginning in 2010, meso- and macro-zooplankton sampling was added at 32 of the 122 regular FMWT fish sampling sites. Zooplankton is collected after fish trawling is complete, using a same sized EMP steel sled with paired mesozooplankton (0.160 mm mesh) and macrozooplankton (0.500 mm mesh) nets. We will sample near five EMP sites, five 20mm sites, two FMWT sites, in adjacent tidal channels or fringing marsh (Table 5, Figure 8), using a paired mysid and zooplankton net as described in Chapter I (Figure 6). These sampling sites were chosen based on their proximity to future FRP restoration sites or comparison wetlands. Thus, we will be able to use these stations to establish a pre-project baseline for zooplankton production, and determine to what extent the effect of restoration is detectable in nearby long-term survey monitoring. Note that while we will discontinue the 2017 macroinvertebrate sampling in Lindsey Slough, we will continue to survey zooplankton alongside 20mm to increase our power to compare data between years.

### FRP Sampling

#### Zooplankton

FRP gear and methods are easily comparable to IEP’s methods. The most important difference is our gear was trawled at the surface of the water for five minutes instead of ten minutes, to reduce potential for take of listed fishes. Where tidal channels or marsh habitat was too short to take a full five-minute tow, the tow time was reduced. In some cases, the gear was held in the mouth of a tidal channel to sample water flowing out of the channel on an ebb tide instead of being trawled.

We sampled monthly from March-June and September - December, in wetlands nearby the long-term surveys as close to the same time as possible. When it was not possible to sample at the same time as the long-term surveys, we sampled the following day at the same point in the tidal cycle.

#### Nutrients

Nutrient sampling methods followed methods used by EMP as closely as possible. At each FRP sampling site, we collected two to three nutrient samples from three different sampling locations:

1. Deep within the wetland as possible, where the water will have the greatest influence from the wetland and least influence from the channel (when possible);
2. At breach/outlet of the site where water is actively moving in or out of the site, or the location of the future breach at pre=restoration sites; and
3. Approximately 100 m outside the site, where we expect some influence of the wetland on water quality in the surrounding channel.

At Tule Red it was infeasible to sample within the wetland on the same date as exterior samples, so only exterior samples were collected.

For each sample we collected two liters of water from just below the surface and transported them back to the lab on ice. We measured chlorophyll florescence from 10 cm below the water’s surface using a YSI 6600 sonde.

In the laboratory, we processed the water as required by DWR’s Bryte laboratory (Wong 2012). For dissolved nutrient samples, we filtered 200 mL water using a 0.45 micron nitrocellulose membrane filter and saved the filtrate on ice; for total nutrients, we filled the sample bottle with unfiltered water and placed the sample on ice. We collected one chlorophyll sample per day to calibrate chlorophyll florescence readings. For each chlorophyll sample, we collected one liter of surface water and transported the water to the lab. In the lab, we filtered 500 mL of water onto a 0.7 micron GF/F glass fiber filter and froze the sample immediately.

#### Fish

The beach seine is a shallow water gear type that is deployed from shore by crewmembers. It measures 15 m long x 1.2 m high with 1.2 m³ cod end bag and is composed of 3.2 mm delta square mesh. One crewmember walked perpendicular from shore into the water holding one end of the net until a depth appropriate for proper seining was reached. A second crewmember followed the path of the first crewmember to minimize site disturbance and positioned their seine pole upon reaching the first crewmember. The first crewmember then turned parallel with the shore and continued walking until the seine was fully opened. Water depth and seine lengths were recorded before both crewmembers pulled the seine towards the beach at a similar speed until only the cod end bag remained in the water. The crew filled a tub with water and placed the cod end in the tub along with any fish caught in the wings of the seine. Fork length of thirty individuals of each fish species was measured to the nearest mm and all remaining fish were plus counted. One to three beach seines were completed at Decker Island or Prospect Island outside the tidal wetland restoration area.

The lampara net is a tapered net measuring 36.5 m long x 3.7 m high. The cod end is composed of 9.5 mm stretch mesh and connects to two wings composed of variable stretched mesh (69.9, 146.1, and 88.9 mm). This net was deployed in shallow and channel habitat from a boat where the tip of the wing was tossed into the water attached to a buoy and sea anchor. Crewmembers deployed the net from the bow of the boat as the boat moved in a circular fashion back to the buoy and sea anchor. One crewmember then brought the buoy and sea anchor onboard and hooked both ends of the net onto the front cleat. The boat then went backwards and caused the net to impinge on itself to prevent fish from escaping through the bottom of the net. Once the net was “folded in half”, each crewmember grabbed one side of the net and brought it onboard. Once the cod end was reached, it was placed in a tub filled with water. Fork length of thirty individuals of each fish species was measured to the nearest mm and all remaining fish were plus counted. Three to four lampara samples were completed outside Prospect Island, Winter Island, Bradmoor Island/Arnold Slough, and Tule Red.

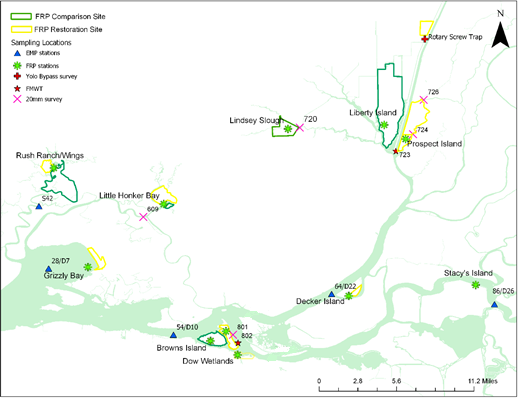


Figure 23. Zooplankton sampling stations. We collected zooplankton, macroinvertebrates, and nutrients samples from shallow wetland habitats (green stars) in proximity to established 20mm sampling sites (pink Xs), established FMWT zooplankton sites (red stars), and EMP sites (blue triangles).

Table 14. Sample numbers for meso- and macro-zooplankton trawls and nutrient samples taken concurrently with other monitoring programs.

|  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- |
| **Survey** | **Months** | | **FRP Site** | **Long Term Site** | **Mysid Trawl** | **Zoop Trawl** | **Nutrient Water Sample** | **Total** |
| 20mm | March-June | | Little Honker/Blacklock | 609 | 4 | 4 | 24 | 32 |
| 20mm | March-June | | Lindsey Slough | 720 | 4 | 4 | 12 | 20 |
| 20mm | March-June | | Prospect Island | 724 | 4 | 4 | 12 | 20 |
| 20mm | March-June | | Liberty Island | 723 | 4 | 4 | 12 | 20 |
| 20mm | March-June | | Winter, Dow | 801 | 8 | 8 | 24 | 40 |
| FMWT | Sept-Dec | | Prospect, Liberty | 723 | 8 | 8 | 24 | 40 |
| FMWT | Sept-Dec | | Winter, Dow | 802 | 8 | 8 | 24 | 40 |
| EMP | March-Dec | | Grizzly Bay | 28/D7 | 8 | 8 | 16 | 32 |
| EMP | March-Dec | | Rush Ranch/Wings | S42 | 8 | 8 | 24 | 40 |
| EMP | March-Dec | | Decker Island | 64/D22 | 8 | 8 | 24 | 40 |
| EMP | March-Dec | | Stacy’s Island | 86/D26 | 8 | 8 | 24 | 40 |
| EMP | Sept-Dec | | Browns | 54/D10 | 4 | 4 | 12 | 20 |
|  |  | Total | |  | 76 | 76 | 232 | 384 |

### Lab Methods

Zooplankton were processed using similar methods to IEP’s 20mm Survey, FMWT survey and EMP survey. In brief:

All samples were filtered and washed in a 0.150 mm mesh sieve. Filtered zooplankton were diluted to a set volume depending on the concentration of zooplankton and/or detritus. One mL subsamples were placed on a Sedgewick-Rafter cell glass slide. All organisms were identified to the taxonomic resolution identified in Table 3. At least 5 slides, but no more than 20 slides were processed for each sample, targeting 400 organisms. This subsample was then extrapolated to calculate the total number of organisms in the sample in individuals per cubic meter of water sampled.

Most samples were processed by a trained Senior Laboratory Assistant (SLA) at the CDFW office in Stockton, California. A subset of samples had identifications checked by a second SLA or Environmental Scientist for quality assurance. Some samples were processed by EcoAnalysists, Inc (Moscow, ID).

Macrozooplankton will be processed using the same methods as described in Chapter 1. Because the FMWT survey and EMP survey only enumerate mysids and amphipods in their samples, we could only compare catches of these taxa, and use catch of other taxa for separate analyses of spatial variability.

Nutrient samples are processed at DWR’s Bryte laboratory using standard methods (Wong 2012).

### Analysis

As of July, 2019, processing of FRP zooplankton, FMWT mysids and zooplankton, and 20mm survey zooplankton was not complete. Therefore, we will not present analysis of that data at this time. The 2018 data will be presented along with a full analysis of 2017 data when sample processing is complete.

To test differences in nutrients and carbon between the wetland and the channel, we graphed chlorophyll, nutrients, and organic carbon versus sampling location (inside, breach, outside, far outside), over time at each site. A generalized linear model will be attempt compare these values statistically, but it is unlikely any detection will occur with one year of data. To determine whether nutrients are limiting phytoplankton production, we will compare nutrient and chlorophyll concentrations in the wetland to published literature values for nutrient concentrations and ratios in the Delta. A GLM of all the chlorophyll and nitrogen data will be best modeled by a parabolic curve or show a threshold of chlorophyll above which nitrogen decreases. To answer causal factors for algal blooms, we will compare visual reports of *Microcystis* and other cyanobacteria with nitrogen concentrations.

Table 15. Predictor variables for models of spatial and intra-annual variation in zooplankton catch and biomass

|  |  |  |
| --- | --- | --- |
| **Variable** | **Variable type** | **Description** |
| Julian day | Continuous | Day of year sample was collected |
| Collection group | Categorical | EMP, 20mm survey, FMWT survey, or FRP survey |
| Habitat type | Categorical | Deep channel, tidal channel, or shoals. |
| Distance from GG | Continuous | Distance from the Golden Gate, in Km. |

This work will help us determine the future level of replication necessary to support assessment of restoration project goals. Data used to answer our questions on timing and replication of samples will also provide pre-project data for evaluating the food web benefits of restoration sites.

Fish

Three components of data were compared between the sampling gears in each habitat type: fish catch per unit effort (CPUE), fork lengths, and species composition in 2017 and 2018. Fish CPUE was calculated using the number of fish caught per volume water sampled (standardized to 10,000 m3) using the following equation: (fish catch/water volume sampled)\*10000.

Data were tested for normality using a Wilks-Shapiro test and the appropriate statistical tests were run using Past3 3.25 software (Hammer et al. 2001). Based on results from the Wilks-Shapiro test, a paired t-test or Wilcoxon rank sum test was used to compare CPUEs between the beach seine or lampara to the townet and midwater trawl. Gear comparisons were made at each sampling location (e.g., Decker Island, Winter Island, etc). If CPUE differences were detected during the paired t-test or Wilcoxon rank sum test, a GLM was run using R 3.5.3 (R\_Core\_Team 2018) to tease apart the potential factors causing CPUE differences between gear types in 2017 and 2018 at each site. The predictor variables chosen were year, tide, temperature, turbidity, and specific conductance. An auto correlation was run on temperature, turbidity, and specific conductance to help choose the best predictor variables for the GLM model for each site in Past3 3.25 software (Hammer et al. 2001).

A permutational multivariate analysis of variance (PerMANOVA, (Anderson 2001)) using a Bray-Curtis similarity index was used to test for fish species composition differences between gear types at each sampling location (Table 7, Table 11). Each fish species CPUE was transformed into a percent catch based on the total CPUE for each net deployment. Any fish not identified to species was removed from this analysis except for *Tridentiger spp*. This genus was not removed because identification would have been similar for all gear types. Using the percent catch of each species caught per tow, a PerMANOVA was run using the adonis feature in the vegan package of R (Oksanen et al. 2016) to determine whether differences in fish communities occurred among gear types using the covariates year, month, temperature, specific conductance, and turbidity. All samples that caught no fish were removed for the PerMANOVA.

A two sample Kolmogorov-Smirnov (K-S) test was used to analyze whether fish size distribution differed between gear types. Three pair-wise comparisons were made independently for the three gear types in the summer and fall. Fish fork lengths were graphed to indicate each gear type’s common length ranges and used to set the maximum fork length values to be evaluated. The common size ranges selected represented 91-99% of all fish measured and provided a good representation of each gear’s target fish size ranges. Fish sizes larger than the commonly caught size ranges were considered outliers and excluded from analysis. Graphs were made using R 3.5.3 with the ggplot2 and plyr packages (R\_Core\_Team 2018; Wickham 2011; Wickham 2016).

Significance was determined at α = 0.05 for all comparisons.

## Results

Fish – Summer

A total of 11,242 fish and 32 fish species were collected in the summer of 2017-2018 during this study (Table 16). When comparing CPUE in shallow vs channel habitats, catch differed between habitat types and was higher in shallow habitat outside Bradmoor Island, Decker Island, and Prospect Island (Figure 24, Table 17). CPUE was also higher in the lampara net compared to the townet in shallow water habitat outside Tule Red in 2017 (Figure 24, Table 17). The lampara net and townet had similar CPUEs at Browns Island, Tule Red in 2018, and when both gears sampled in channel water habitat outside Winter Island (Figure 24, Table 17).

Table 16. Catch & CPUE of fish species caught by each gear type for this study during jun-aug. A star next to a species name indicates native fish species.

|  |  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | **Shallow Water Gear Types** | | | | | | | | **Channel Water Gear Types** | | | | |
|  | **2017 Beach Seine** | | **2018 Beach Seine** | | **2017 Lampara** | | **2018 Lampara** | | | **2017 Townet** | | **2018 Townet** | |
| **Fish Species** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** | | **Catch** | **CPUE** | **Catch** | **CPUE** |
| American Shad | 42 | 16048.0 | 52 | 8689.5 | 157 | 14541.3 | 23 | 1368.7 | | 303 | 3471.0 | 2 | 25.3 |
| Bigscale Logperch | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 40.9 | | 0 | 0 | 0 | 0 |
| Black Bass | 15 | 5444.2 | 20 | 5321.2 | 0 | 0 | 4 | 311.8 | | 0 | 0.0 | 0 | 0 |
| Black Crappie | 1 | 317.5 | 0 | 0 | 3 | 349.7 | 1 | 62.7 | | 2 | 19.9 | 0 | 0 |
| Bluegill | 3 | 779.4 | 2 | 401.2 | 0 | 0 | 2 | 105.2 | | 0 | 0 | 0 | 0 |
| Centrarchid Spp. | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | 1 | 11.2 | 0 | 0 |
| Delta Smelt\* | 0 | 0 | 0 | 0 | 2 | 311.0 | 0 | 0 | | 24 | 283.2 | 0 | 0 |
| Golden Shiner | 0 | 0 | 83 | 16140.4 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Herring (Unid) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | 12 | 151.2 | 0 | 0 |
| Hitch\* | 0 | 0 | 1 | 367.6 | 0 | 0 | 4 | 198.7 | | 0 | 0 | 0 | 0 |
| Lamprey Spp.\* | 0 | 0 | 1 | 1470.6 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Largemouth Bass | 11 | 2450.0 | 24 | 7130.8 | 0 | 0 | 5 | 263.0 | | 0 | 0 | 0 | 0 |
| Longfin Smelt\* | 0 | 0 | 0 | 0 | 1 | 71.4 | 0 | 0 | | 0 | 0 | 1 | 11.2 |
| Mississippi Silverside | 1217 | 431386.6 | 4124 | 1254025.2 | 197 | 26751.6 | 83 | 8994.0 | | 32 | 366.4 | 0 | 0 |
| Mosquitofish | 1 | 210.5 | 5 | 1285.0 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Northern Anchovy\* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | 6 | 72.0 | 12 | 146.6 |
| Prickly Sculpin\* | 16 | 4643.0 | 1 | 429.2 | 6 | 299.2 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Rainwater Killifish | 83 | 23649.5 | 22 | 6773.8 | 0 | 0 | 2 | 68.8 | | 0 | 0 | 0 | 0 |
| Red Shiner | 2 | 2040.8 | 0 | 0 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Sac. Pikeminnow\* | 15 | 7270.4 | 0 | 0 | 0 | 0 | 1 | 48.9 | | 0 | 0 | 0 | 0 |
| Sac. Sucker\* | 76 | 30078.8 | 0 | 0 | 0 | 0 | 2 | 72.8 | | 0 | 0 | 0 | 0 |
| Shimofuri Goby | 80 | 21498.7 | 62 | 19269.9 | 12 | 1980.5 | 0 | 0 | | 13 | 145.4 | 1 | 11.2 |
| Shokihaze Goby | 0 | 0 | 0 | 0 | 1 | 122.7 | 0 | 0 | | 103 | 1139.4 | 2 | 21.7 |
| Smallmouth Bass | 0 | 0 | 6 | 1743.1 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Splittail\* | 437 | 99370.3 | 1 | 192.3 | 10 | 1047.1 | 22 | 2206.7 | | 2 | 23.7 | 1 | 11.3 |
| Spotted Bass | 0 | 0 | 1 | 232.0 | 0 | 0 | 0 | 0 | | 0 | 0 | 0 | 0 |
| Starry Flounder\* | 0 | 0 | 0 | 0 | 2 | 295.8 | 5 | 606.4 | | 2 | 26.5 | 1 | 11.4 |
| Striped Bass | 6 | 684.2 | 3 | 629.3 | 209 | 27752.0 | 117 | 11822.5 | | 728 | 8951.9 | 69 | 769.3 |
| Sunfish | 3 | 1554.4 | 0 | 0 | 0 | 0 | 0 | 0 | | 0 | 0.0 | 0 | 0 |
| Threadfin Shad | 282 | 72678.7 | 0 | 0 | 284 | 36073.8 | 69 | 4530.0 | | 160 | 1848.5 | 5 | 55.8 |
| 3-spine Stickleback\* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | | 13 | 128.6 | 0 | 0 |
| Tridentiger spp. | 0 | 0 | 0 | 0 | 1 | 155.5 | 0 | 0 | | 811 | 8439.1 | 633 | 7165.4 |
| Tule Perch\* | 3 | 649.1 | 0 | 0 | 25 | 1396.1 | 12 | 773.4 | | 1 | 11.5 | 0 | 0 |
| Wakasagi | 0 | 0 | 0 | 0 | 3 | 224.1 | 2 | 140.6 | | 6 | 83.7 | 1 | 11.5 |
| White Catfish | 0 | 0 | 3 | 948.7 | 0 | 0 | 0 | 0 | | 18 | 267.8 | 1 | 11.3 |
| White Sturgeon\* | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 48.9 | | 0 | 0.0 | 0 | 0 |
| Yellowfin Goby | 92 | 25441.6 | 84 | 32485.5 | 61 | 8841.7 | 56 | 5642.5 | | 5 | 62.8 | 5 | 60.5 |

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Figure 24. CPUE boxplots of the gear types in various wetlands during jun-aug of 2017-2018.

Table 17. CPUE comparison values for gear types in various habitats during jun-aug of 2017 & 2018.

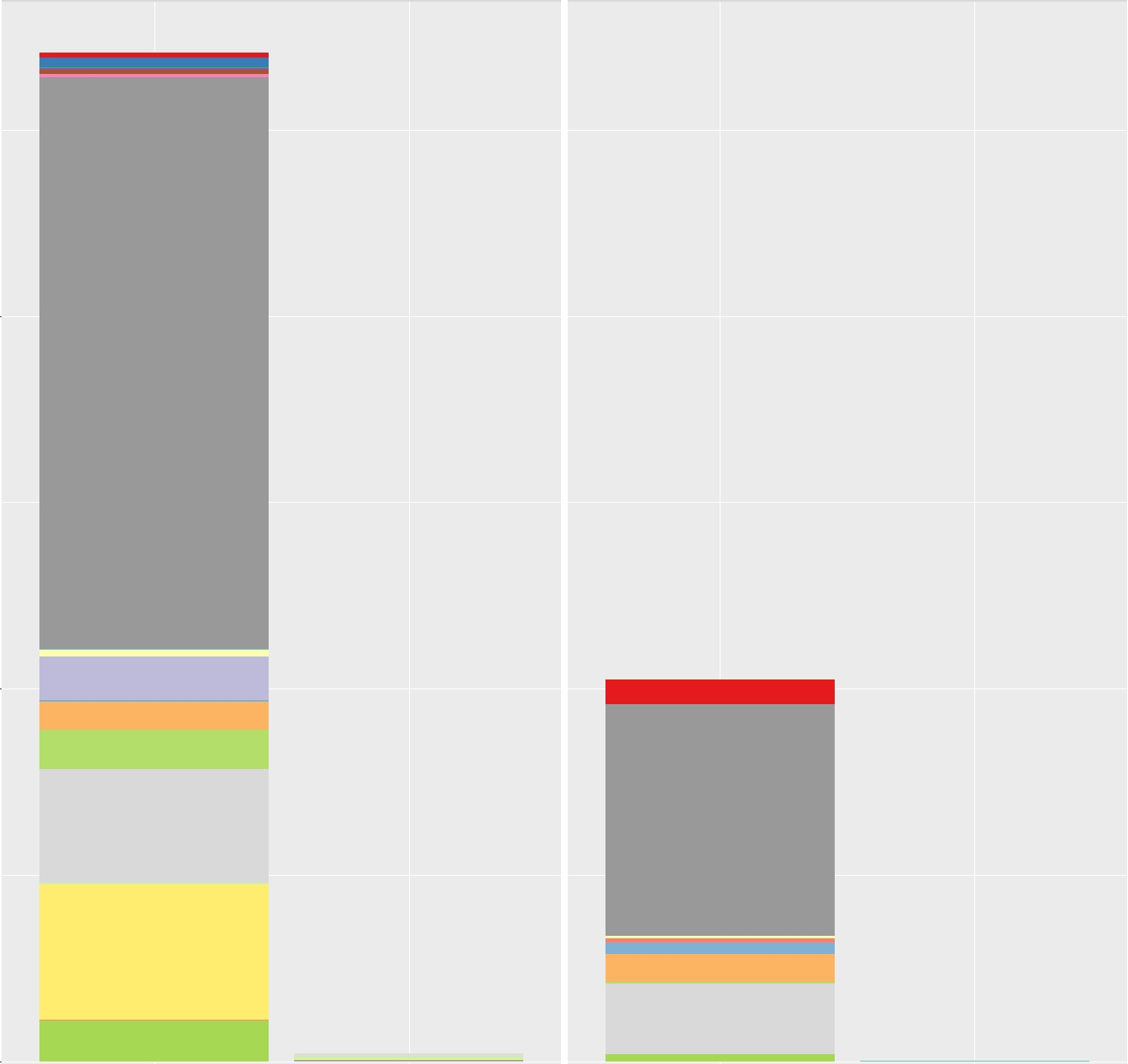
|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Shallow vs Channel Habitat Comparisons** | | | | | | |
| **Bradmoor Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Lampara | 18 | 5010.8 | 1200.3 |  | 29.1 | <0.001 \* |
| 2017 Townet | 18 | 818.7 | 165.2 |  |
| 2018 Lampara | 18 | 1191.5 | 249.7 |  |
| 2018 Townet | 18 | 348.6 | 103.9 |  |
| **Browns Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Wilcoxon Paired T-Test | |
|  | Z | p |
| 2018 Lampara | 11 | 146.45 | 89.3 |  | 1.6 | 0.12 |
| 2018 Townet | 11 | 17.573 | 5.1 |  |
| **Decker Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Beach Seine | 15 | 36094.3 | 15665.1 |  | 38.4 | <0.001 \* |
| 2017 Townet | 15 | 285.3 | 90.4 |  |
| 2018 Beach Seine | 9 | 54426.1 | 17356.5 |  |
| 2018 Townet | 9 | 2.2 | 1.5 |  |
| **Prospect Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Beach Seine | 9 | 22762.0 | 7931.3 |  | 15.4 | 0.001 \* |
| 2017 Townet | 9 | 28.9 | 21.7 |  |
| 2018 Beach Seine | 7 | 91638.3 | 66101.5 |  |
| 2018 Townet | 7 | 5.1 | 2.5 |  |
| **Shallow vs Shallow Habitat Comparisons** | | | | | | |
| **Tule Red** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Lampara | 18 | 1373.0 | 384.5 |  | 23.3 | <0.001 \* |
| 2017 Townet | 18 | 306.3 | 67.9 |  |
| 2018 Lampara | 13 | 141.2 | 55.5 |  |
| 2018 Townet | 13 | 75.8 | 21.2 |  |
| **Channel vs Channel Habitat Comparisons** | | | | | | |
| **Winter Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Lampara | 18 | 294.7 | 109.2 |  | 3.4 | 0.33 |
| 2017 Townet | 18 | 84.3 | 43.0 |  |
| 2018 Lampara | 17 | 134.3 | 52.1 |  |
| 2018 Townet | 17 | 38.1 | 9.8 |  |

The GLM of fish CPUE at Bradmoor Island found significant interactions between gear types, year, specific conductance, and turbidity, where CPUE was higher for the lampara net in 2017 (Table 18). The GLM of fish CPUE at Decker Island found significant interactions between gear types, specific conductance, and temperature, such that CPUE was higher for beach seines (Table 18). A similar GLM pattern was found at Prospect where the significant interaction between CPUE and gear type, where CPUE was for beach seines (Table 18). At Tule Red, the GLM of fish CPUE found significant interactions with specific conductance, where the specific conductance was lower in 2017 (Table 18). GLMs were not run for Browns Island or Winter Island since CPUE did not differ with a Wilcoxon Paired T-Test and Kruskal-Wallis Test, respectively (Table 17).

Table 18. GLMMs of log total CPUE of fish samples collected at various wetland locations during jun-aug of 2017 and 2018. GLMMs only run for locations where cpue differed between gear types.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Bradmoor Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |
| (Intercept) | 4.41E+03 | 7.86E+02 | 5.602 | 4.45e-07 \* |
| GearTownet | -1.67E+00 | 2.77E-01 | -6.051 | 7.56e-08 \* |
| Year | -2.18E+00 | 3.90E-01 | -5.605 | 4.41e-07 \* |
| Temp | 3.24E-01 | 1.71E-01 | 1.89 | 0.0632 |
| SpC | 1.88E-04 | 8.27E-05 | 2.273 | 0.0263 \* |
| Turb | 1.82E-02 | 8.47E-03 | 2.15 | 0.0352 \* |
|  |  |  |  |  |
| **Decker Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |
| (Intercept) | 4.87E+03 | 1.02E+03 | 4.792 | 2.00e-05 \* |
| GearTownet | -6.47E+00 | 4.75E-01 | -13.625 | < 2e-16 \* |
| SpC | 2.41E-03 | 6.52E-04 | 3.7 | 0.000609 \* |
| Temp | 3.75E-01 | 1.58E-01 | 2.37 | 0.022331 \* |
| Year | -2.41E+00 | 5.04E-01 | -4.786 | 2.03e-05 \* |
|  |  |  |  |  |
| **Prospect Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |
| (Intercept) | 9.2455 | 0.67 | 13.799 | 1.79e-13 \* |
| GearTownet | -7.7311 | 0.9475 | -8.159 | 1.22e-08 \* |
|  |  |  |  |  |
| **Tule Red** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **P value** |
| (Intercept) | 2.54E+03 | 1.44E+03 | 1.766 | 0.0827 |
| Year | -1.25E+00 | 7.12E-01 | -1.762 | 0.0834 |
| SpC | -1.61E-04 | 7.46E-05 | -2.161 | 0.0348 \* |
| Turb | 7.33E-03 | 4.79E-03 | 1.53 | 0.1315 |

Each gear type caught a multitude of fish species (Table 16) and all habitat comparisons resulted in differences in fish composition due to gear type (Table 19). However, the time of catch (i.e., covariate – “year” and/or “month”) was also a predictor of fish composition differences observed between gear types at a multitude of wetland sites (Table 19). In general, the townet caught a higher abundance of Striped Bass and Tridentiger goby species (Figure 25). The beach seine caught a higher abundance of Mississippi Silversides and Splittail (Figure 25). The lampara net generally caught higher abundances of Mississippi Silverside, Striped Bass, and Threadfin Shad (Figure 25).



Decker Island

Prospect Island

Beach

Seine

Beach

Seine

Townet

Townet

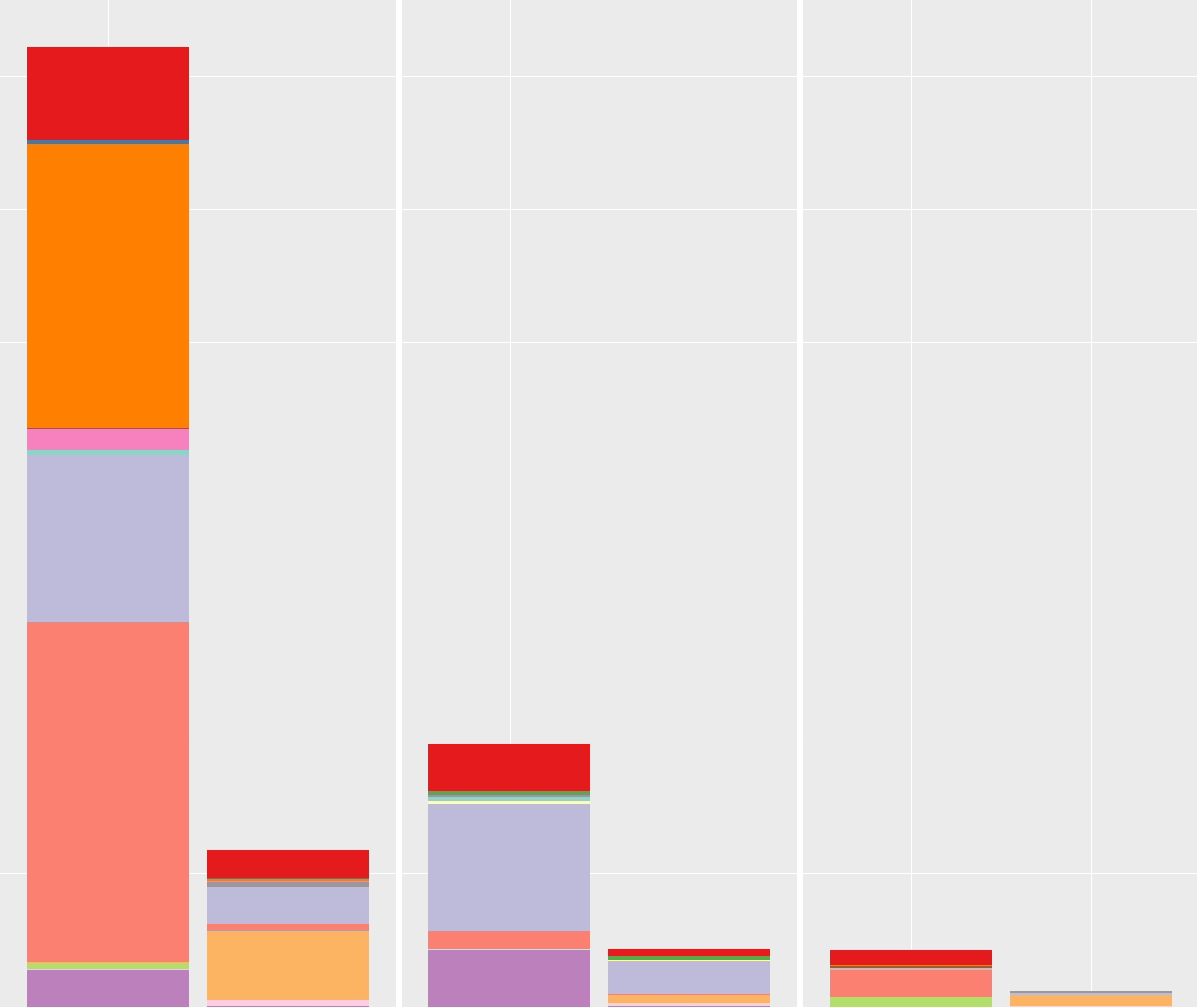


0

200000

400000

Total CPUE



Tule Red

Winter Island

Lampara

Lampara

Lampara

Townet

Townet

Townet



Bradmoor Island

0

25000

50000

75000

Total CPUE

Table 19. Results of PerMANOVAs on relative abundance of fish species during the Summer Townet Survey. All habitat comparisons show differences of fish composition influenced by gear type at each sampling location.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Browns Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 0.27387 | 0.27387 | 2.7872 | 0.09291 | 0.005 \* |
| Month | 1 | 0.23356 | 0.23356 | 2.377 | 0.07924 | 0.037 \* |
| Temp | 1 | 0.06091 | 0.06091 | 0.6199 | 0.02067 | 0.633 |
| SpC | 1 | 0.11422 | 0.11422 | 1.1624 | 0.03875 | 0.356 |
| Turbidity | 1 | 0.49633 | 0.49633 | 5.0513 | 0.16839 | 0.003 \* |
| Residuals | 18 | 1.76865 | 0.09826 |  | 0.60004 |  |
| Total | 23 | 2.94755 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Bradmoor Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 5.5707 | 5.5707 | 46.527 | 0.28343 | 0.001 \* |
| Year | 1 | 1.6474 | 1.6474 | 13.759 | 0.08382 | 0.001 \* |
| Month | 1 | 3.2597 | 3.2597 | 27.225 | 0.16585 | 0.001 \* |
| Temp | 1 | 0.4023 | 0.4023 | 3.36 | 0.02047 | 0.018 \* |
| SpC | 1 | 0.5302 | 0.5302 | 4.428 | 0.02698 | 0.007 \* |
| Turbidity | 1 | 0.7014 | 0.7014 | 5.858 | 0.03568 | 0.002 \* |
| Residuals | 63 | 7.5431 | 0.1197 |  | 0.38378 |  |
| Total | 69 | 19.6548 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Decker Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 4.6462 | 4.6462 | 24.5225 | 0.34986 | 0.001 \* |
| Year | 1 | 0.4537 | 0.4537 | 2.3947 | 0.03417 | 0.047 \* |
| Month | 1 | 0.9548 | 0.9548 | 5.0392 | 0.07189 | 0.003 \* |
| Temp | 1 | 0.4882 | 0.4882 | 2.5766 | 0.03676 | 0.030 \* |
| SpC | 1 | 0.1863 | 0.1863 | 0.9832 | 0.01403 | 0.41 |
| Turbidity | 1 | 0.2985 | 0.2985 | 1.5756 | 0.02248 | 0.168 |
| Residuals | 33 | 6.2524 | 0.1895 |  | 0.47081 |  |
| Total | 39 | 13.2802 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Prospect Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 1.7182 | 1.7182 | 5.1698 | 0.2045 | 0.001 \* |
| Year | 1 | 0.5789 | 0.57891 | 1.7419 | 0.0689 | 0.095 |
| Month | 1 | 0.609 | 0.60905 | 1.8325 | 0.07249 | 0.08 |
| Temp | 1 | 0.4082 | 0.40818 | 1.2282 | 0.04858 | 0.268 |
| SpC | 1 | 0.2568 | 0.2568 | 0.7727 | 0.03056 | 0.622 |
| Turbidity | 1 | 0.1778 | 0.17785 | 0.5351 | 0.02117 | 0.849 |
| Residuals | 14 | 4.6529 | 0.33235 |  | 0.55379 |  |
| Total | 20 | 8.4019 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Tule Red** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear | 1 | 5.2779 | 5.2779 | 23.9257 | 0.27281 | 0.001 \* |
| Year | 1 | 1.4141 | 1.4141 | 6.4103 | 0.07309 | 0.001 \* |
| Month | 1 | 0.8726 | 0.8726 | 3.9556 | 0.0451 | 0.004 \* |
| Temp | 1 | 0.9395 | 0.9395 | 4.259 | 0.04856 | 0.006 \* |
| SpC | 1 | 0.2593 | 0.2593 | 1.1756 | 0.0134 | 0.314 |
| Turbidity | 1 | 0.4355 | 0.4355 | 1.974 | 0.02251 | 0.096 |
| Residuals | 46 | 10.1473 | 0.2206 |  | 0.52451 |  |
| Total | 52 | 19.3462 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Winter Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 2.0206 | 2.02058 | 10.7422 | 0.12875 | 0.001 \* |
| Year | 1 | 1.6077 | 1.6077 | 8.5471 | 0.10244 | 0.001 \* |
| Month | 1 | 2.3271 | 2.32709 | 12.3717 | 0.14828 | 0.001 \* |
| Temp | 1 | 0.4217 | 0.4217 | 2.2419 | 0.02687 | 0.062 |
| SpC | 1 | 0.3692 | 0.36921 | 1.9629 | 0.02353 | 0.105 |
| Turbidity | 1 | 0.1067 | 0.10669 | 0.5672 | 0.0068 | 0.719 |
| Residuals | 47 | 8.8406 | 0.1881 |  | 0.56333 |  |
| Total | 53 | 15.6936 |  |  | 1 |  |

Figure 25. The percent relative cpue (# of fish species/10,000 m³) of fish species in and around wetlands during jun-aug of 2017-2018.

All three Kolmogorov-Smirnov tests showed significant differences of fish lengths between all three gear types (Table 20), however the townet consistently caught smaller fish primarily comprised of *Tridentiger spp*. and Striped Bass (Figure 26) under 25 mm. The beach seine caught a greater proportion of fish between 20-50mm, while the lampara caught a greater proportion of fish between 25-60mm (Figure 26).

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Figure 26. The proportion of fork length frequency caught by each gear type during jun-aug of 2017 and 2018. Fish greater than 78 mm were not used for length comparisons between gear types, but shown here for additional information.

Table 20. **Kolmogorov-Smirnov comparisons of fish sizes between gear types during jun-aug of 2017 and 2018.**

|  |  |  |  |
| --- | --- | --- | --- |
| **Kolmogorov-Smirnov test for equal distributions** | | | |
| **Beach Seine** | | **Lampara** | |
| N: | 568 | N: | 240 |
| D : | 0.3022 | P value | <0.001 \* |
| **Beach Seine** | | **Townet** | |
| N: | 568 | N: | 514 |
| D : | 0.7825 | P value | <0.001 \* |
| **Lampara** | | **Townet** | |
| N: | 240 | N: | 514 |
| D : | 0.7896 | P value | <0.001 \* |

Fish – Fall

A total of 8,135 fish and 29 fish species were collected during the fall of 2017 and 2018 during this study (Table 21). When comparing CPUE in shallow vs channel habitats, catch differed between habitat types and was higher in shallow habitat outside Decker Island, Prospect Island, and Ryer Island (Figure 28, Table 22). Similar to our summer comparisons, CPUE was also higher in the lampara net compared to the townet in shallow water habitat outside Tule Red in 2017 (Figure 28, Table 22). The lampara net and townet had similar CPUEs at Browns Island and when both gears sampled in channel water habitat outside Winter Island (Figure 28, Table 22).

Table 21. Catch & CPUE of fish species caught by each gear type for this study during Sep-Dec of 2017 and 2018. A star next to a species name indicates native fish species.

|  |  |  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | **Shallow Water Gear Types** | | | | | | | | **Channel Water Gear Types** | | | |
|  | **2017 Beach Seine** | | **2018 Beach Seine** | | **2017 Lampara** | | **2018 Lampara** | | **2017 Midwater Trawl** | | **2018 Midwater Trawl** | |
| **Fish Species** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** | **Catch** | **CPUE** |
| American Shad | 0 | 0 | 1 | 236.4 | 49 | 2745.9 | 10 | 453.0 | 427 | 753.0 | 124 | 240.7 |
| Black Bass | 0 | 0 | 1 | 246.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Bluefin Killifish | 0 | 0 | 1 | 416.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Bluegill | 2 | 476.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| California Halibut\* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.3 |
| Chinook Salmon\* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.7 |
| Delta Smelt\* | 0 | 0 | 0 | 0 | 3 | 190.8 | 0 | 0 | 0 | 0 | 0 | 0 |
| Golden Shiner | 0 | 0 | 3 | 416.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Jacksmelt | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.8 | 0 | 0 |
| Largemouth Bass | 3 | 628.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Longfin Smelt\* | 0 | 0 | 0 | 0 | 0 | 0.0 | 0 | 0 | 11 | 20.0 | 3 | 5.1 |
| Mississippi Silverside | 2082 | 384734.5 | 4695 | 1325655.8 | 23 | 1024.6 | 75 | 2283.1 | 0 | 0 | 0 | 0 |
| Mosquitofish | 2 | 227.8 | 6 | 1623.5 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Northern Anchovy\* | 0 | 0 | 0 | 0 | 0 | 0 | 4 | 177.5 | 13 | 21.9 | 20 | 36.2 |
| Prickly Sculpin\* | 1 | 238.1 | 1 | 641.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Rainwater Killifish | 9 | 1499.4 | 143 | 78123.2 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Sac. Pikeminnow\* | 1 | 98.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Shimofuri Goby | 8 | 1750.2 | 45 | 18683.4 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Shokihaze Goby | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2.3 |
| Smallmouth Bass | 0 | 0 | 1 | 246.9 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Splittail\* | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 115.3 | 1 | 1.9 | 0 | 0 |
| Spotted Bass | 0 | 0 | 4 | 987.7 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.0 |
| Starry Flounder | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.6 |
| Striped Bass | 0 | 0 | 0 | 0 | 8 | 471.6 | 1 | 40.9 | 80 | 137.4 | 8 | 16.2 |
| Threadfin Shad | 12 | 1352.9 | 4 | 845.7 | 108 | 5950.4 | 10 | 519.0 | 45 | 86.1 | 31 | 60.0 |
| Tule Perch\* | 0 | 0 | 1 | 111.1 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Wakasagi | 0 | 0 | 0 | 0 | 2 | 123.8 | 0 | 0 | 0 | 0 | 0 | 0 |
| White Catfish | 0 | 0 | 2 | 663.6 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 2.3 |
| White Sturgeon\* | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 1.5 | 0 | 0 |
| Yellowfin Goby | 10 | 1653.9 | 30 | 7384.7 | 2 | 123.8 | 0 | 0 | 0 | 0 | 0 | 0 |

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Figure 27. CPUE boxplots of the gear types in various wetlands during sep-dec of 2017-2018.

Table 22. CPUE comparison values for gear types in various habitats during jun-aug of 2017-2018.

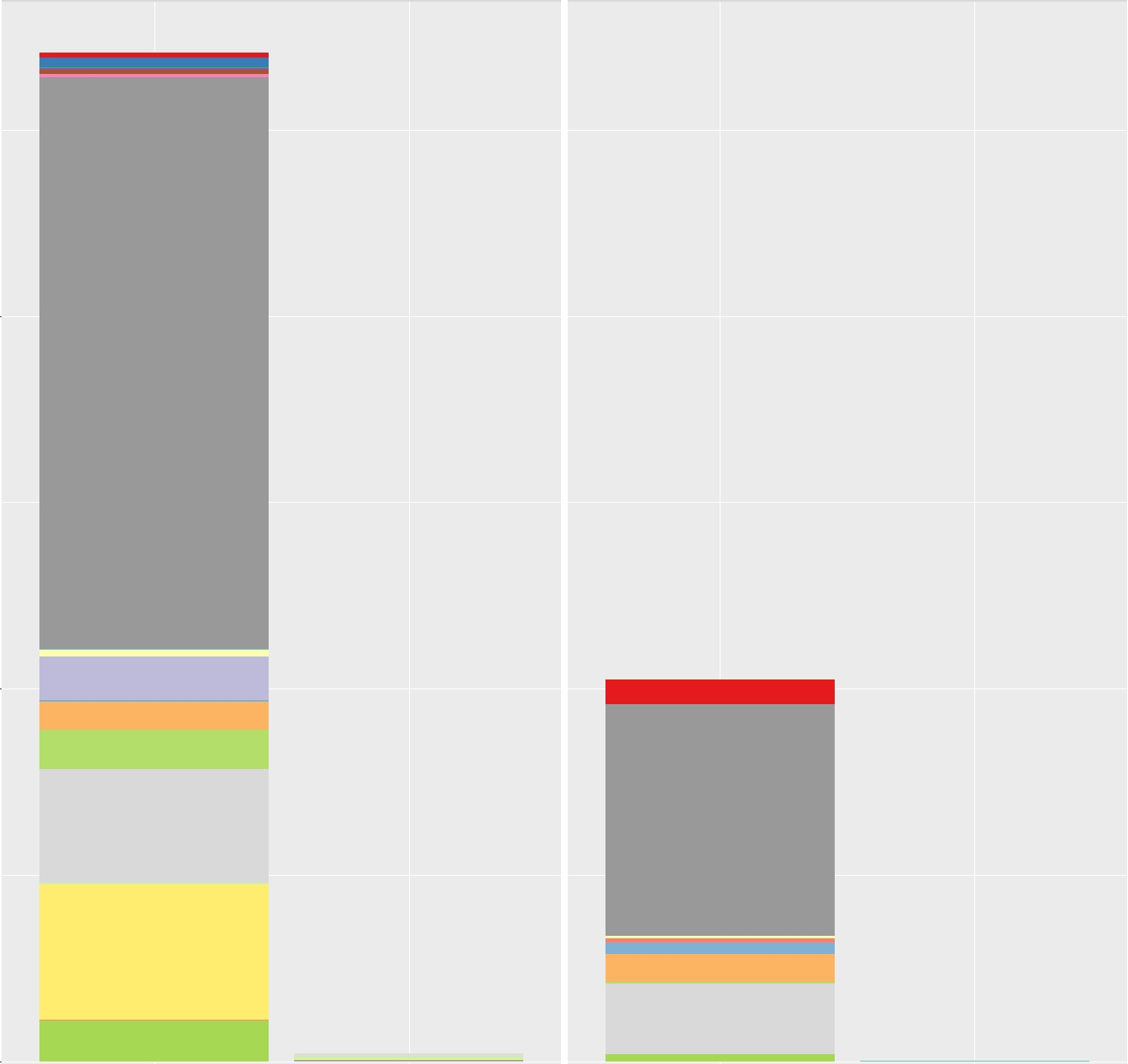
|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Shallow vs Channel Habitat Comparisons** | | | | | | |
| **Browns Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Wilcoxon Paired T-Test | |
|  | Z | p |
| 2018 Lampara | 8 | 63.485 | 46 |  | 0.68 | 0.5 |
| 2018 Midwater Trawl | 8 | 7.7 | 2.3 |  |
| **Decker Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Beach Seine | 9 | 36208.1 | 16123.1 |  | 27.9 | <0.001 \* |
| 2017 Midwater Trawl | 9 | 48.3 | 21.7 |  |
| 2018 Beach Seine | 12 | 82865.6 | 22981.9 |  |
| 2018 Midwater Trawl | 12 | 17.7 | 5.8 |  |
| **Prospect Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Beach Seine | 3 | 22262.7 | 15406.2 |  | 9.9 | 0.015 \* |
| 2017 Midwater Trawl | 3 | 0.6 | 0.6 |  |
| 2018 Beach Seine | 4 | 109794.9 | 75306.4 |  |
| 2018 Midwater Trawl | 4 | 5.7 | 5.7 |  |
| **Ryer Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Wilcoxon Paired T-Test | |
|  | Z | p |
| 2018 Lampara | 6 | 334.9 | 213.0 |  | 2 | 0.046 \* |
| 2018 Midwater Trawl | 6 | 2.7 | 0.8 |  |
| **Shallow vs Shallow Habitat Comparisons** | | | | | | |
| **Tule Red** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Lampara | 16 | 525.2 | 183.5 |  | 12.85 | 0.005 \* |
| 2017 Midwater Trawl | 16 | 33.6 | 7.7 |  |
| 2018 Lampara | 9 | 77.4 | 51.9 |  |
| 2018 Midwater Trawl | 9 | 4.8 | 1.6 |  |
| **Channel vs Channel Habitat Comparisons** | | | | | | |
| **Winter Island** | | | | | | |
| Gear Type | n | Mean | Std. Error |  | Kruskal-Wallis Test | |
|  | H | p |
| 2017 Lampara | 3 | 401.3 | 329.0 |  | 4.0 | 0.192 |
| 2017 Midwater Trawl | 3 | 16.3 | 14.3 |  |
| 2018 Lampara | 3 | 0 | 0 |  |
| 2018 Midwater Trawl | 3 | 3.6 | 3.6 |  |

The GLM of fish CPUE at Decker Island found significant interactions between gear type, specific conductance, and tide where CPUE was higher in the beach seine, especially during low slack and ebb tides and slightly higher specific conductance (Table 23). The GLM of fish CPUE at Prospect Island found significant interactions between gear types where the beach seine had higher CPUE (Figure 28, Table 23). At Ryer Island the GLM of fish CPUE was found to interact with turbidity, where turbid waters increased CPUE (Table 23). At Tule Red, the GLM of fish CPUE found significant interactions with gear type, tide, turbidity, and year (Table 23). At site Tule Red, fish CPUE was higher in 2017 for both gear types and ebb tides. CPUE was Similar to what occurred in the summer, the 2017 lampara GLMs were not run for Browns Island or Winter Island since CPUE did not differ with a Wilcoxon Paired T-Test and Kruskal-Wallis Test, respectively (Table 17).

Table 23. GLMMs of log total CPUE of fish samples collected at various wetland locations during jun-aug of 2017 and 2018. GLMMs only run for locations where cpue differed between gear types.

|  |  |  |  |  |
| --- | --- | --- | --- | --- |
| **Decker Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **Pr(>|t|)** |
| (Intercept) | 1.11E+01 | 1.07E+00 | 10.417 | 1.08e-12 \* |
| GearMidwaterTrawl | -7.77E+00 | 6.48E-01 | -12.003 | 1.69e-14 \* |
| SpC | 1.59E-03 | 7.95E-04 | 2.006 | 0.052 |
| Tide | 9.43E-01 | 3.52E-01 | -2.142 | 0.0387 \* |
|  |  |  |  |  |
| **Prospect Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **Pr(>|t|)** |
| (Intercept) | 8.966 | 1.164 | 7.7 | 5.55e-06 \* |
| GearMidwater | -8.387 | 1.647 | -5.093 | 0.000265\* |
|  |  |  |  |  |
| **Ryer Island** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **Pr(>|t|)** |
| (Intercept) | 9.6245168 | 4.3290162 | 2.223 | 0.0569 |
| Turb | -0.0004685 | 0.0002582 | -1.814 | 0.1072 |
|  |  |  |  |  |
| **Tule Red** | | | | |
| **Factor** | **Estimate** | **Std. Error** | **t value** | **Pr(>|t|)** |
| (Intercept) | 5.53E+03 | 9.97E+02 | 5.543 | 1.32e-06 \* |
| GearMidwater | -1.64E+00 | 4.66E-01 | -3.522 | 0.000964 \* |
| Tide | -7.05E-01 | 2.61E-01 | -2.697 | 0.009683 \* |
| Turb | -0.0308 | 0.01393 | -2.21 | 0.031976 \* |
| Year | -2.73597 | 0.49437 | -5.534 | 1.35e-06 \* |

Each gear type caught a multitude of fish species (Table 21) and all habitat comparisons resulted in differences in fish composition due to gear types (Table 24). However, the time of catch (i.e., covariate – “year” and/or “month”) was also a predictor of fish composition differences observed between gear types at a multitude of wetland sites (Table 24). In general, the midwater trawl caught a higher abundance of American Shad, Striped Bass and Threadfin Shad (Figure 25). The beach seine caught mostly Mississippi Silversides, but in 2018 caught numerous Rainwater Killifish at Decker Island (Figure 25). The lampara net generally caught higher abundances of American Shad, Mississippi Silverside, and Threadfin Shad (Figure 25).



Decker Island

Prospect Island

Beach

Seine

Beach

Seine

Townet

Townet

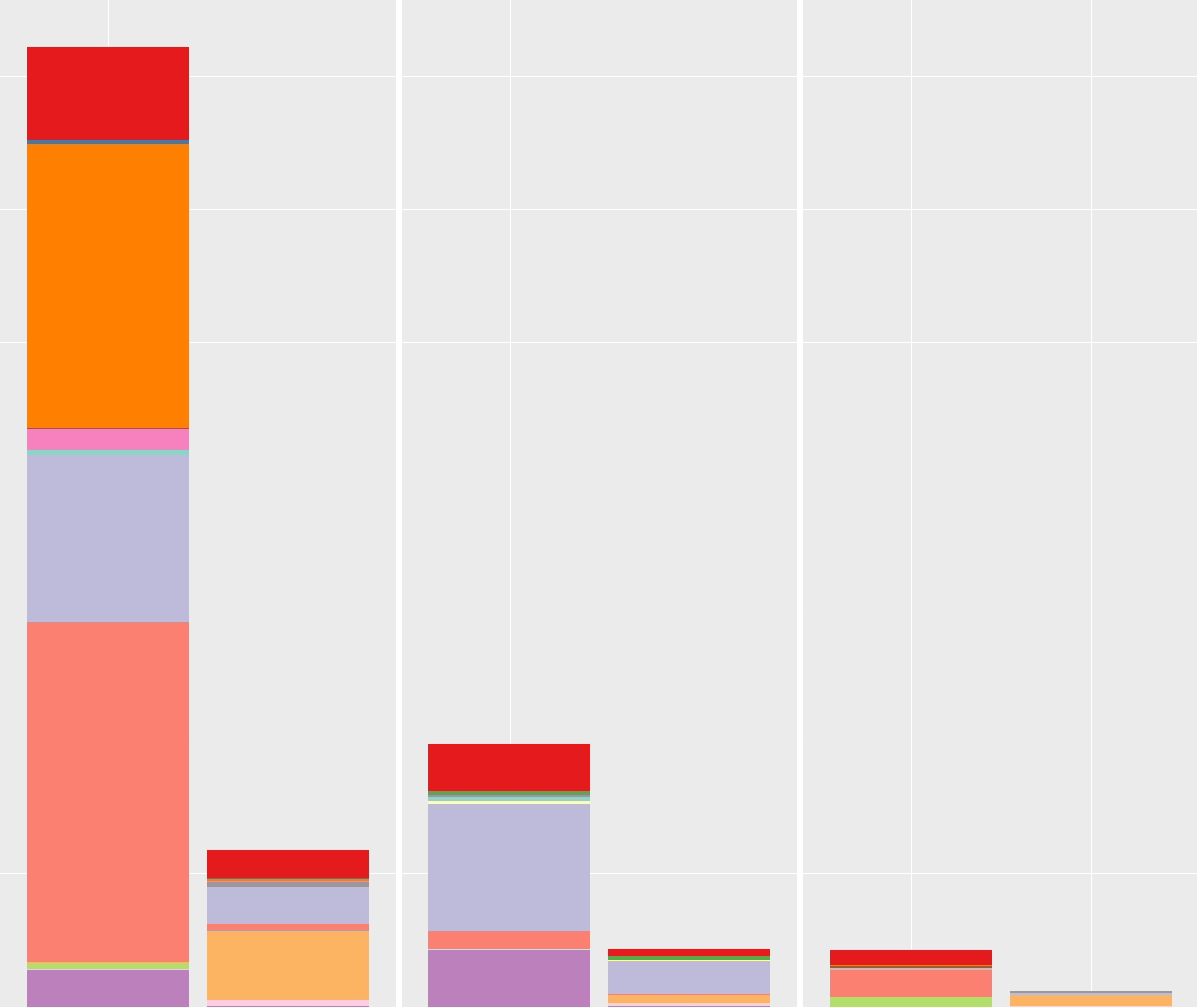


0

200000

400000

Total CPUE



Tule Red

Winter Island

Lampara

Lampara

Lampara

Townet

Townet

Townet



Bradmoor Island

0

25000

50000

75000

Total CPUE

Table 24. Results of PerMANOVAs on relative abundance of fish species during the Fall Midwater trawl Survey.

|  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- |
| **Browns Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **P value** |
| Gear.Type | 1 | 0.71074 | 0.71074 | 17.6134 | 0.2706 | 0.007 \* |
| Month | 1 | 0.81973 | 0.81973 | 20.3143 | 0.31209 | 0.001 \* |
| Temp | 1 | 0.43157 | 0.43157 | 10.695 | 0.16431 | 0.009 \* |
| SpC | 1 | 0.51848 | 0.51848 | 12.8488 | 0.1974 | 0.010 \* |
| Turbidity | 1 | 0.02499 | 0.02499 | 0.6192 | 0.00951 | 0.605 |
| Residuals | 3 | 0.12106 | 0.04035 |  | 0.04609 |  |
| Total | 8 | 2.62656 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Decker Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **P value** |
| Gear.Type | 1 | 6.3021 | 6.3021 | 46.099 | 0.54763 | 0.001 \* |
| Year | 1 | 0.117 | 0.117 | 0.856 | 0.01017 | 0.413 |
| Month | 1 | 0.1211 | 0.1211 | 0.886 | 0.01052 | 0.429 |
| Temp | 1 | 0.211 | 0.211 | 1.544 | 0.01834 | 0.168 |
| SpC | 1 | 0.3048 | 0.3048 | 2.23 | 0.02649 | 0.099 |
| Turbidity | 1 | 0.3508 | 0.3508 | 2.566 | 0.03048 | 0.069 |
| Residuals | 30 | 4.1013 | 0.1367 | 0.35638 |  |  |
| Total | 36 | 11.5081 |  | 1 |  |  |
|  |  |  |  |  |  |  |
| **Prospect Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **P value** |
| Gear.Type | 1 | 1.37662 | 1.37662 | 7.6029 | 0.55599 | 0.019 \* |
| Year | 1 | 0.11641 | 0.11641 | 0.6429 | 0.04702 | 0.554 |
| Month | 1 | 0.1652 | 0.1652 | 0.9124 | 0.06672 | 0.472 |
| Temp | 1 | 0.11508 | 0.11508 | 0.6356 | 0.04648 | 0.578 |
| SpC | 1 | 0.00771 | 0.00771 | 0.0426 | 0.00311 | 0.985 |
| Turbidity | 1 | 0.15177 | 0.15177 | 0.8382 | 0.0613 | 0.466 |
| Residuals | 3 | 0.5432 | 0.18107 |  | 0.21938 |  |
| Total | 9 | 2.47599 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Ryer Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **P value** |
| Gear.Type | 1 | 1.5687 | 1.56865 | 4.8067 | 0.30078 | 0.002 \* |
| Month | 1 | 0.5417 | 0.54167 | 1.6598 | 0.10386 | 0.158 |
| Temp | 1 | 0.3447 | 0.34466 | 1.0561 | 0.06609 | 0.402 |
| SpC | 1 | 0.0223 | 0.02228 | 0.0683 | 0.00427 | 0.997 |
| Turbidity | 1 | 0.1273 | 0.12729 | 0.3901 | 0.02441 | 0.874 |
| Residuals | 8 | 2.6108 | 0.32635 |  | 0.5006 |  |
| Total | 13 | 5.2153 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Tule Red** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **Pr(>F)** |
| Gear.Type | 1 | 0.8228 | 0.82275 | 3.7462 | 0.07803 | 0.006 \* |
| Year | 1 | 0.808 | 0.80797 | 3.6789 | 0.07663 | 0.009 \* |
| Month | 1 | 0.4475 | 0.44755 | 2.0378 | 0.04245 | 0.085 |
| Temp | 1 | 0.707 | 0.70696 | 3.2189 | 0.06705 | 0.012 \* |
| SpC | 1 | 0.1537 | 0.15368 | 0.6998 | 0.01458 | 0.614 |
| Turbidity | 1 | 0.3569 | 0.35691 | 1.6251 | 0.03385 | 0.142 |
| Residuals | 33 | 7.2476 | 0.21962 |  | 0.68741 |  |
| Total | 39 | 10.5434 |  |  | 1 |  |
|  |  |  |  |  |  |  |
| **Winter Island** | | | | | | |
|  | **Df** | **SumsOfSqs** | **MeanSqs** | **F.Model** | **R2** | **P value** |
| Gear.Type | 1 | 1.4993 | 1.4993 | 7.3208 | 0.39037 | 0.004 \* |
| Month | 1 | 0.221 | 0.22098 | 1.079 | 0.05754 | 0.365 |
| Temp | 1 | 0.7839 | 0.78393 | 3.8278 | 0.20411 | 0.046 \* |
| SpC | 1 | 0.0949 | 0.09492 | 0.4635 | 0.02471 | 0.714 |
| Turbidity | 1 | 0.0128 | 0.01282 | 0.0626 | 0.00334 | 0.972 |
| Residuals | 6 | 1.2288 | 0.2048 |  | 0.31994 |  |
| Total | 11 | 3.8408 |  |  | 1 |  |

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Figure 28. The percent relative cpue (# of fish species/10,000 m³) of fish species in and around wetlands during Sep-Dec of 2017-2018.

All three Kolmogorov-Smirnov tests showed significant differences of fish lengths between all three gear types (Table 25), however the midwater consistently caught larger fish between 50-90 mm than the other two gear types (Figure 29). The beach seine caught a greater proportion of fish between 20-70mm, while the lampara caught a greater proportion of fish between 40-80mm (Figure 29).

A close up of a map

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Figure 29. The proportion of fork length frequency caught by each gear type during sep-dec of 2017 and 2018. Fish greater than 100 mm were not used for length comparisons between gear types, but shown here for additional information.

Table 25. **Kolmogorov-Smirnov comparisons of fish sizes between gear types during jun-aug of 2017 and 2018 for fish ≤100mm.**

|  |  |  |  |
| --- | --- | --- | --- |
| **Kolmogorov-Smirnov test for equal distributions** | | | |
| **Beach Seine** | | **Lampara** | |
| N: | 590 | N: | 72 |
|  |  |  |  |
| D : | 0.48948 | P value | <0.001 \* |
| **Beach Seine** | | **Midwater Trawl** | |
| N: | 590 | N: | 172 |
|  |  |  |  |
| D : | 0.54821 | P value | <0.001 \* |
| **Lampara** | | **Midwater Trawl** | |
| N: | 72 | N: | 172 |
|  |  |  |  |
| D : | 0.33607 | P value | <0.001 \* |

## Discussion

After comparing the relative abundance of fish between shallow water gear types to the Summer Townet and Fall Midwater Trawl surveys for two years, shallow water gear types typically resulted in higher catch. In particular, beach seines had the highest CPUE of all gear types tested and may suggest that shallow water habitat is better rearing habitat for certain fish species (Grimaldo et al. 2004; Sommer et al. 2001). However, fish species American Shad, Striped Bass, Threadfin Shad, and *Tridentiger spp*. were more readily found in channel habitat and may suggest they rear in this habitat (Figure 25, Figure 28).

During the summer, fish abundance, composition and size were different when making comparisons between shallow water and channel gear types, except at Browns Island and Winter Island, where abundances were the same. For the most part fish abundance was higher in 2017 for lampara and townet and may possibly due to the wet water year type (in contrast to the below average water year type in 2018) (Figure 24). Fish have been shown to respond favorably to wet water year types in the past by providing favorable habitat, rearing, and foraging opportunities (Kratville 2008; Moyle 2002; Sommer et al. 2004; Sommer et al. 2001). Conversely, CPUEs were higher for the beach seines in 2018 and mainly driven by Mississippi Silversides both years (Figure 24). However, native fish Splittail were more abundant in the beach seine during 2017 likely due to floodplains being inundated providing ample spawning grounds and rearing opportunities (Kratville 2008). The townet also has smaller cod end mesh that allowed many *Tridentiger spp*. and Striped Bass to be retained, as opposed to the lampara net. In contrast, the lampara caught more and larger pelagic fishes such as American Shad and Threadfin Shad. It’s likely that the differences observed in fish compositions between gear types is attributed to site depth, gear deployment, gear mesh size, water year type, and fish size. Overall, the townet collected smaller fish than the beach seine and lampara net in 2017 and 2018.

Similar to the summer data, fall fish abundance, composition and size were different when making comparisons between shallow water and channel gear types, except at Browns Island and Winter Island, where abundances were the same. Once again, the greatest CPUE catches were with beach seines and caused by the high number of Mississippi Silversides captured. Fish compositions differed between the lampara and midwater trawl appear to be driven by the more frequent catches of American Shad and Striped Bass caught by the midwater trawl which may be more frequent in channel habitat. Longfin Smelt were exclusively caught by the midwater trawl at various sites through the estuary at both shallow and channel sites. It may be that the sparseness of Longfin Smelt make it difficult to detect since the lampara samples such a small volume of water compared to the midwater trawl.

After a two year comprehensive gear evaluation between shallow water gears and Summer Townet and Fall Midwater Trawl, we recommend continued sampling of shallow water habitat outside future tidal wetland restoration sites. In particular, sampling shallow water habitat in June and July can provide great snap shots of rearing opportunities by juvenile cyprinids and the Sacramento Sucker. There is currently a great interest to study cyprinids and other native fishes of the Delta not routinely studied. Fish of interest Delta Smelt, Longfin Smelt, and Chinook Salmon were not frequently caught, so it may be best to rely on smelt data from IEP surveys such as the Townet, Fall Midwater Trawl, Enhanced Delta Smelt studies to know where the smelt are located. Chinook Salmon fry (fish < 70mm) were not captured by our shallow water gear types and likely due to the time of year we sample as they are more abundant in between March-May (Brandes and McLain 2000). If the future, it may be beneficial to sample shallow water habitat in April and May to capture Chinook Salmon fry.

# CHAPTER 3: Methods Development

Project Component Lead: Dave Contreras

## Introduction

### Submerged Aquatic Vegetation

### ARIS Sonar

ESA take is getting more difficult to obtain as many of our listed fish species are in imperil. Traditional means of assessing the relative fish abundance and composition involves dragging a net through the water to collect fishes. This causes stress and may harm to listed fish species as contact with the net can pin fish against the net and cause skin abrasions. Gill nets also cause skin abrasions and sometimes suffocate fish if left in the water for extended periods of time (Nielson and Johnson 1983).

ARIS sonar imaging is a modern sampling technique, where no fish are harmed or taken out of the water for monitoring. However, the imagery projected on screen may have some limitations as its unknown whether fish identification down to species and fish lengths can be accurately recorded. Since gill nets are stationary, it may be possible to capture fish movement with sonar imagery and identify fish entangled in the net after the net has been retrieved. In addition, it is also possible to capture images of fish behavior around gill nets to see how effective this sampling technique may be.

### Algae/Phytoplankton

In the past, most evidence pointed to pelagic phytoplankton as the key driver of the Bay-Delta food web (Canuel et al. 1995; Sobczak et al. 2005; Sobczak et al. 2002), however, more recent work suggests benthic, epiphytic, and wetland-derived carbon may play a more important role than previously recognized (Schroeter et al. 2015). Due to the lack of information on benthic and epiphytic algae in the SFE, it is unclear whether algae found in the water column of wetlands were produced in the water column, or were produced in other microhabitats and washed into the water column. Learning how algae are distributed within a wetland will be important to supporting Framework Hypothesis F3: Form and magnitude of primary production, along with site and landscape attributes, will drive the form and magnitude of secondary production.

Study Questions

1. Can the ARIS sonar be used as a tool to determine gill net and electrofishing efficiency?
2. Can ARIS sonar be used to monitor fish use of wetlands with decreased take of listed species?
3. What is the relative contribution of different types of algae to the phytoplankton in wetland channels versus major channels?

## Methods

### SAV survey techniques

Rake collections

Sonar w/Biobase

Table 26 Number of SAV rake samples collected at each site.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| **Samples near** | **Region** | **Site type** | **Mar, 2018** | **Aug, 2018** | **Oct, 2018** | **Jan, 2019** | **Total** |
| Liberty Island | Cache Slough | Tidal Wetland | 62 | 0 | 0 | 0 | 62 |
| Prospect Island | Cache Slough | Muted tidal Wetland | 119 | 60 | 0 | 0 | 179 |
| Winter Island | Confluence | Muted Tidal wetland | 86 | 84 | 73 | 74 | 317 |
| Browns Island | Confluence | Tidal wetland | 82 | 74 | 65 | 62 | 283 |
|  |  | **Total** | 349 | 218 | 138 | 136 | 841 |

### ARIS Evaluation of a Gill Net

The ARIS was attached to an ARIS AR2 rotating device that allows the camera to be pivoted under water (Figure 30). The AR2 device was mounted on a aluminum pole attached to the side of the boat that allowed the camera to be taken in and out of the boat (Figure 30). With collaboration with the USGS, gill nets were set at various sites in Wildlands. As the net sat stationary across the channel, the boat was anchored to shore and recorded a section of the gill net.



Figure 30. ARIS camera mounted to a USGS boat ready to capture gill net footage.

### Algae/Phytoplankton

To supplement routine monitoring of phytoplankton communities at all FRP sites, and to better differentiate between the sources of phytoplankton in the water column, we intensively sampled a wide variety of habitats at one site (Liberty Island). Collection methods were be based on standard benthic algae methods developed by the California State Water Resources Control Board Surface Water Ambient Monitoring Program (Ode et al. 2016). At Liberty Island, we collected algae from four microhabitats within the wetland (also see Table 2):

* 3 SAV samples – algae were scraped from a 10-cm section of the *Stuckenia pectinata* or dominant form of submerged vegetation.
* 4 EAV samples – algae were scraped from a 10-cm section of *Schoenoplectus acutus*, or dominant form of emergent vegetation.
* 3 Benthic samples (Epipelic or Episammic) – algae will be rinsed from the mud/sand collected by a 10-cm PVC core, or petite ponar grab.
* 6 Pelagic – phytoplankton were collected as per “Water Quality Grab Samples SOP” (PWT 2017).
* 1 Filamentous algae sample.

All samples were preserved in Lugol’s Iodine solution and shipped to EcoAnalysts, Inc. for analysis. EcoAnalysts measured and counted all taxa of algae within the sample using the Utermöhl microscopic method (Utermöhl 1958) and APHA Standard Methods (APHA 2017), as described in section 2.2.3. Laboratory methods, above.

## Analysis

**Algae:** We compared the community composition of algae samples from different habitat types with PERMANOVA and visualized these differences using NMDS with functions from the R package vegan (Oksanen et al. 2016). We then performed multiple pattern analysis to see whether some taxa were associated with particular habitat types using the R package indicspecies (Cáceres and Jansen 2016).

## Results

**ARIS:** Due to procurement issues once again the ARIS sonar device was obtained later in the year. However, we briefly worked with the USGS observing how fish behave around gill nets in Wildlands. From the video shot that day, we observed many fish swimming up to the net and turn back to swim away. Other fish appeared to swim through the net or were able to avoid being tangled likely due to the selective mesh sizes of the gill net. During one instance, we were able to see a larger fish entangled within the gill net and were able to identify the fish species once the gill net was retrieved (Figure 30).

A screen shot of a computer

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Figure 31. A Channel Catfish caught on ARIS sonar imaging (left) and idenitfied later in the gill net (right).

**Algae:** There were significant differences in algal communities collected in different microhabitats at Liberty Island (Table 16). In particular, pelagic samples had more *Microcystis*, benthic samples had more *Oscillatoria*, and filamentous algae had more *Oedogonium* (Figure 24). This was apparent from the separation in the NMDS plot (Figure 24), however the sample sizes were relatively small. There were surprisingly few taxa pulled out as indicator species. Only five taxa were indicated as being associated with open water, and two species with benthic/epiphytic samples (Table 17). We attempted to identify indicator species for SAV, EAV, and benthic samples separately, but none were selected until we combined the groups into to groupings: “Pelagic” and “Benthic/epiphytic”.



Figure 32 - community composition of algal samples from different microhabitats in Liberty Island.

Table 27 – Results of a PERMANOVA comparing algae collected in different habitat types at Liberty Island in March of 2018.

|  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- |
| Factor | DF | SumsOfSqs | MeanSqs | F value | R2 | P value |  |
| Habitat | 4 | 1.3518 | 0.33795 | 2.4844 | 0.45299 | 0.003 | \*\* |
| Residuals | 12 | 1.6324 | 0.13603 |  | 0.54701 |  |  |
| Total | 16 | 2.9842 |  |  | 1 |  |  |



Figure 33. NMDS plot of algal samples taken from different habitat types at Liberty Island. The sample labled “other” is a sample of filamentous algae.

Table 28. Multilevel pattern analysis of invertebrate associations with wetlands of differing types. A significant p value indicates the particular taxa was positively associated with a particular habitat type. Habitat types were combined so that there were only two groups, “Pelagic” and “Benthic/epiphytic”.

|  |  |  |  |
| --- | --- | --- | --- |
| **Pelagic** |  |  |  |
|  | **stat** | **P value** |  |
| *Cyclotella* | 0.988 | 0.005 | \*\* |
| *Teleaulax* | 0.983 | 0.005 | \*\* |
| *Monoraphidium* | 0.924 | 0.005 | \*\* |
| *Cryptomonas* | 0.896 | 0.010 | \* |
| *Achnanthidium* | 0.754 | 0.020 | \* |
|  |  |  |  |
|  |  |  |  |
| **Benthic/epiphytic** |  |  |  |
|  | **stat** | **P value** |  |
| *Melosira* | 0.895 | 0.025 | \* |
| *Oscillatoria* | 0.858 | 0.030 | \* |
|  |  |  |  |

## Discussion

## ARIS

Observing fish behavior around the gill net did seem probable. Fishes of various sizes were seen actively swimming away from, through, and ensnared in the net. Although the ARIS will not be able to image smaller fish, those should not affect the efficiency of the gill net as the mesh sizes do not target those smaller sized fishes. Based on our brief use of the ARIS imaging software, identifying fish to species may prove difficult and will require identifying fish based on movement patterns and body shape. However, other patterns such as the flux of fish in and out of tidal wetlands may prove useful to understand how at which times of the years wetlands are being utilized.

## Algae

We expected to observe significantly different communities exiting the wetland than are observed in the exterior channel, and that differences will be driven by higher abundances of epibenthic and epiphytic taxa in the wetland channels. Samples from the surface of vegetation and benthic substrates within the wetland will have greater overlap in community composition when compared to tidal channels than with exterior channels. However, our sample size during 2018 was too small to make any firm conclusions about taxa that can be labeled as definitively “benthic” or “pelagic”.

There were some significant differences in communities between microhabitats, but the degree of overlap between hulls on the NMDS is difficult to interpret with only a few samples per group. It does seem that the benthic, epiphytic, and filamentous samples cluster together, while the pelagic samples form a separate grouping (Figure 25). There were also few species identified as “indicators” for of pelagic versus benthic/epiphytic habitats. Of the indicator species that were identified, *Cyclotella* is a centric diatom considered to be good zooplankton food and historically common in the LSZ (Lehman 2000). *Teleaulax* is a generalist mixotorophic cryptophyte that can withstand varying salinities, light availability, and temperatures (Cloern and Dufford 2005). *Monoraphidium* is a green algae found to be abundant in the central Delta and considered to be good zooplankton food (Lehman et al. 2010). *Cryptomonas* is a small cryptophyte considered to be highly nutritious (Burns et al. 2011), but rarely consumed by zooplankton (Kimmerer et al. 2018a) and found to be associated with the toxic cyanobacteria *Microcystis* (Lehman et al. 2010), though no *Microcystis* was found in these samples. *Achanathidium* is a genus of pennate diatoms that are more often benthic than pelagic (Potapova and Hamilton 2007), so it was somewhat surprising to see it associated with the “pelagic” samples. This may have been an artifact of our low sample size, or may be due to the high wind-wave resuspention of benthic sediments common on Liberty Island. The indicators for benthic/epiphytic habitats were *Melosira*, a chain-forming centric diatom, and *Oscillatoria,* a chain-forming cyanobacteria that can produce toxins (Paerl and Otten 2013).

We repeated our sampling of microhabitats in 2019 at a difference site (Little Honker Bay), and will re-sample some of these sites in 2020. With an increased sample size at a wider variety of sites, we may be able to make inferences as to differences in epibenthic versus pelagic contributions to phytoplankton at various wetland sites.

# Endangered Species Act Take

Table 29. Take of listed fish species in all FRP sampling during 2018. Gears not listed had zero take.

|  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  | **Listed fish species take** | | | | | | | | |
| **Gear Type** | **Chinook Salmon** | **Steelhead** | **Delta Smelt (larval)** | **Delta Smelt (juv)** | **Delta Smelt (adult)** | **Longfin Smelt (larval)** | **Longfin Smelt (juv)** | **Longfin Smelt (adult)** | **Green Sturgeon** |
| Mysid Trawls | 0 | 0 | 2 | 0 | 0 | 25 | 0 | 0 | 0 |
| Beach Seine | 2 (fall-run) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| **Total** | **2** | **0** | **2** | **0** | **0** | **25** | **0** | **0** | **0** |

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