12-2016

Developing a multimetric index to assess resaca ecosystem health

Buford J. Lessley
The University of Texas Rio Grande Valley

DEVELOPING A MULTIMETRIC INDEX TO

ASSESS RESACA ECOSYSTEM

HEALTH

A Thesis

by

BUFORD J. LESSLEY

Submitted to the Graduate College of
The University of Texas Rio Grande Valley
In partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

December 2016

Major Subject: Biology

DEVELOPING A MULTIMETRIC INDEX TO

ASSESS RESACA ECOSYSTEM

HEALTH

A Thesis
by
BUFORD J. LESSLEY

COMMITTEE MEMBERS

Dr. Alejandro Fierro-Cabo
Chair of Committee

Dr. Jude Benavides
Committee Member

Dr. Richard Kline
Committee Member

December 2016

ABSTRACT

Lessley, Buford, Developing a Multimetric Index to Assess Resaca Ecosystem Health. Master of Science (MS), December, 2016, 78pp., 17 tables, 16 figures, references, 121 titles.

As the only freshwater ecosystem in the lower Rio Grande Valley aside from the Rio itself, resacas are critical habitat for many species of flora and fauna. Old distributaries of the Rio Grande, resacas provided conveyance routes moving floodwater to the Laguna Madre. Today these wetlands are novel ecosystems and are artificially maintained. Urbanization and agriculture have lead to sedimentation, habitat loss, contaminants, poor water quality, and invasive species. The objective of this study was to assess and monitor resaca pools and to compose the Resaca Health Index (RHI) from selected indicators of ecosystem structure and function including leaf litter decomposition rates, a riparian habitat assessment index, the trophic state index, and a fish community index. Overall, the RHI adequately discriminated among resacas in various ecosystem statuses, and can be a valuable management tool allowing for more accurate assessment and monitoring of ecosystem health in urban and suburban resacas.

DEDICATION

I dedicate this thesis to the loving memory of my grandmother

Linda Walker-Green

ACKNOWLEDGMENTS

I would like to thank the University of Texas Rio Grande Valley for providing the opportunity to continue my education and gain valuable experience conducting field research. I would like to thank my advisors Dr. Alejandro Fierro-Cabo and Dr. Jude Benavides along with Dr. Richard Kline for sharing their knowledge and support throughout this process. I would also like to thank the National Oceanic and Atmospheric Administration Bays Watershed Education and Training for financial support throughout my research.

Thank you to the many people who have assisted with field and lab work throughout the course of my research: Dr. Elizabeth Heise, Dr. Boris Ermolinsky, Cesar Ayala, David LecusayJr. Guadalupe Garcia III, Monica Delgado, Brenda Bazan, Alex Mendoza, Manuel Saucedo,Stefany Salinas, Mario Marquez, Ethan Getz, Leticia Contreras, Samantha Moore, Shelby Bessette, Tiffany Cogan, Austin Greene, Samantha Silvestri, Evan Bare, Al Alder, Catheline Froehlich, Guillermo Aguilar and anyone else I might have missed.

A very special thank you to my parents for always supporting me (especially financially) and to my loving wife Amanda as she has supported me from the beginning of this journey.

TABLE OF CONTENTS

Page

ABSTRACT iii

DEDICATION iv

ACKNOWLEDGMENTS v

TABLE OF CONTENTS vi

LIST OF TABLES ix

LIST OF FIGURES xi

CHAPTER I. INTRODUCTION 1

Wetlands 1

Ecosystem Health 2

Resacas 3

Purpose 5

Hypotheses 6

Objectives 6

CHAPTER II. METHODOLOGY 7

Site Descriptions 7

Town Resaca System 7

Resaca de la Palma System 8

Fort Brown System 9

Water Quality Measurements 10
Biotic Community Assessment 11
Decomposition 12
Sediment Analysis 13
Riparian Assessment 13
Data Analysis 14
Resaca Health Index 16
CHAPTER III. RESULTS 18
Sediment Analysis 18
Water Quality Parameters 18
Fish Community 19
Riparian Assessment 21
Decomposition 22
Resaca Health Index 22
CHAPTER IV. DISCUSSION 25
Resacas 25
Sediments 25
Water Parameters 26
Fish Community 26
Riparian Habitat 30
Decomposition 33
Resaca Health Index 34
CHAPTER V. CONCLUSION 39

REFERENCES 41

APPENDIX..52

BIOGRAPHICAL SKETCH 78

LIST OF TABLES

Page

Table 1. Geographic Coordinates Of Study Sites In Brownsville, Tx. Coordinates Are Located At Middle Points Of Resaca Pools 53

Table 2. Trophic State Classification Based On Carlson's Trophic State Index 53

Table 3. Criteria For The Riparian Index (RI) Used For Resaca Riparian Assessment 54

Table 4. Critera For Classifying Riparian Quality Based On RI Scores. Adapted From Munne et al., 2003 54

Table 5. Mean Annual Water Parameters (±SE) Based On Monthly Sampling In Six Resaca Pools From September 2015 to August 2016 55

Table 6. Seasonal Carlson's Trophic Sate Index In Six Resaca Pools Derived From Secchi Depths From September 2015 To August 2016 55

Table 7. Principle Component Analysis Results Of Monthly Water Parameters Recorded From September 2015 to August 2016. Expressed As (a) Eigenvalues And (b) Eigenvectors. Principal Component (PC) ; Cumulative Percent Variation (Cum% Variation) 56

Table 8. Seasonal And Total Fish Abundances From The Six Resaca Pools 57

Table 9. Fish Species Found In Each Of The Six Resaca Pools 58

Table 10. Species Richness, Number of Piscivourous Speices, And Dominace By Pool And Season 59

Table 11. Species Evenness Indices. Shannon-Weiner, Simpson, Pielou, And Margalef....................60

Table 12. Analysis Of Similarity (ANOSIM) Results Based On Communities And Abundances 60

Table 13. Similarity Percentage Analysis (SIMPER) Results Based On Community Abundances 61

Table 14. Riparian Index (RI) Scores For The Six Resaca Pools Based On Criteria Specified In Table 2 61

Table 15. Seasonal Decay Constants (K) For The Six Resaca Pools 61

Table 16. Fish Community Index RHI Scores; Fall (a); Winter (b); Summer (c) 62

Table 17. Resaca Health Index Scores And Normalized Values Of Its Components For Six Resaca Pools: Fall (a); Winter (b); Summer (c) 63

LIST OF FIGURES

Page

Figure 1. Freshwater Found Within Cameron County, TX Including The Arroyo Colorado Five Distributary Systems, Numerous Oxbows, and The Rio Grande 64

Figure 2. Resaca Systems Within Cameron County, TX, Systems Used Fro This Study Included Resaca de la Palma, Town Resaca, And Fort Brown And Lozano Banco 65

Figure 3. Sample Resaca Pools Located In Brownsville, TX. FTB-Fort Brown; LB-Lozano Banco; DP-Dean Porter; BM- Billy Mitchell; And VICC-Valley International County Club 66

Figure 4. Drone Imagery Of Resaca Blvd 67

Figure 5. Drone Imagery Of Dean Porter 68

Figure 6. Drone Imagery Of VICC 69

Figure 7. Drone Imagery Of Billy Mitchell 70

Figure 8. Drone Imagery Of Fort Brown 71

Figure 9. Drone Imagery Of Lozano Banco 72

Figure 10. Mean Sediment Particle Size (% Dry Weight) Found Via Stratified Sampling In The Six Sample Pools. Samples Collected In August 2016, Brownsville, TX 73

Figure 11. Mean Sediment Organic Matter Content From Stratified Sampling In The Six Sample Pools. Samples Collected In August 2016, Brownsville, Tx. 73

Figure 12. Principle Component Of Monthly Environmental Variables. Three Principal Components Were Required To Account For 75% Of The Variation. Samples Collected Between September 2015 and August 2016, Brownsville, TX 74

Figure 13. Multi-Dimensional Scaling (MDS) Of Seasonal Sampling Events (√Transformed) For Fish Communities Of The Six Sample Pools. Samples Collected During October 2015, February 2016, and June 2016. 74

Figure 14. Average Seasonal Decomposition Rates By Pool..75

Figure 15. Resaca Health Index Scores Based On Resaca Pool Location.....................................76

Figure 16. Average Seasonal Decomposition Rates Compared To Average Seasonal Temperatures..77

CHAPTER I

INTRODUCTION

Wetlands

As we see continued urbanization and land use changes around the world, many ecosystems are plagued with the effects of fragmentation and are at an increased risk of degradation (Roach et al, 2008). Wetlands are just one of the many systems that face drastic changes due to anthropogenic activities resulting in increased development, nonpoint source pollution, and climate change (Erwin, 2008). It has been noted that over half the original wetlands found in the United States as of 1700 were lost, according to a 2002 estimate by the Environmental Protection Agency (EPA, 2002; Dudgeon, 2010). In Texas alone, 61,126 hectares of palustrine wetlands were lost from 1950 to 1990 (Moulton et al, 1997). Stressors from local and landscape (i.e. watershed) scales constantly affect most wetlands as pollutants, nutrients, and sediments are carried far downstream (Haberl et al., 2003). These stressors combined with the presence of invasive species, altered water regimes, channelization, and habitat modification continue to challenge the resiliency of these systems (Van Dam et al., 1998).

Wetlands are defined as environments that are exposed to periodic or permanent flooding and are classified based on depth, hydrology, soils, and vegetation (Tiner, 2000, Harberl et al., 2003). Though a broad term, wetlands include swamps, tidal marshes, and shallow water bodies such as bayous, which create the transitional zones between open water and dry land (Tiner, 1984). These systems maintain valuable ecosystem services such as improving water quality, providing habitat for many species, sequestering carbon and nutrients such as nitrogen and phosphorus, and holding floodwaters (EPA, 2002; Boyer and Polasky, 2004; Woodward and Wui, 2001;Kayranli et al., 2010). Wetlands also provide economic benefits through commercial and recreational fishing, hunting, bird watching, and timber (Woodward and Wui, 2001, Boyer and Polasky, 2004). The recognition of wetlands as critical ecosystems needed to sustain life and for human wellbeing has triggered the need to find integrative and reliable ways to assess their degradation and recovery.

Ecosystem Health Assessment

Ecosystem health is seen as the desired endpoint for environmental management and is defined by three major aspects: productivity and ecosystem function, organization or structure, and resilience (Costanza, 2012). Initially proposed by Rapport et al. (1985), the concept has been expanded to incorporate both qualitative and quantitative measurements are for its assessment (Sun et al., 2016). This concept has been applied to a variety of ecosystems including terrestrial systems, such as forests, and aquatic systems (both marine and freshwater) all over the world (Dahms and Geils, 1997; Bonde et al., 2004; Rombouts et al., 2013; Xu et al., 2011). Ecosystem health assessments are greatly based on the measurement of structural and functional attributes (Spencer et al., 1998; Rapport, 1989; Hering et al., 2006). Structural indicators for wetlands are most commonly used as they allow for rapid assessment and are derived from biotic communities such as fish, macroinvertebrates, plankton, and vegetation in terms of diversity, complexity, and or individual indicator species (Blackburn and Mazzacano, 2012; Uzarski et al., 2005; Schiemer, 2000; Reiss, 2006). Metrics that are used to assess structure can include species richness, presence of indicator species, abundance, evenness, and dominance (Gibson et al., 2000). Functional indicators are derived and based on ecosystem processes such as leaf litter decomposition, net ecosystem metabolism, mineralization and primary production (Young et al., 2008; Young et al., 2004).

The use of multimetric indices is a well-accepted integrative approach to accurately diagnose the health of the selected systems. Several of these indices have been used extensively such as the index of biological integrity (IBI) (Brinson and Rheinhardt, 1996), Florida Wetland Condition Index (FWCI) (Reiss, 2006), Stream-Wetland-Riparian Index (SWR) (Brooks et al., 2009), Benthic Index of Biotic Integrity (B-IBI)(Van Dolah et al., 1999). These assessments mostly based on structural indicators, allow for ecosystem health determinations and identify systems that are usually classified as degraded, recovering, or healthy. Other indices are derived from the use of the hydrogeomorphic approach (HGM) that use functional indicators to make health determinations (Hauer and Smith, 1998; Brinson, 1996). While many indices are based on either structural or functional indicators, very few integrating both (Clapcott et al., 2012; Riipinen et al., 2009).

Resacas

Many fragmented wetlands meander through Cameron County, Texas and the lower Rio Grande Valley (Figure 1). Known locally as resacas, the term encompasses distributaries and oxbows or bancos of the Rio Grande. Generally speaking the term resaca can be used to refer to the five distributary systems, the numerous oxbow systems, or the pools that make up those systems (Figure 2). These wetlands formed naturally as floodwaters topped riverbanks and found alternate routes to the Laguna Madre and the Gulf of Mexico (Mathis and Matisoff, 2004). Since the construction of multiple dams along the Rio Grande and its tributaries, downstream river flow has been tightly controlled leading to a loss of the natural flood cycle which both created and maintained these resacas (Burks-Copes and Webb, 2012). Over time and due to the absence of recurrent flooding, resacas were naturally cut off from the river and these once flood dependent systems experienced a reduction in biological carrying capacity due to loss of nutrients and water recharge (Jones, 2009). Due to reduced volumetric exchange of water and extended residence times, sediments accumulated causing drastic decreases in overall depths (Jones, 2009). Today, with the exclusion of floodwaters, some resacas are only supplied by rainfall and runoff and function as ephemeral wetlands (Jones, 2009). Other resacas have been converted for use as reservoirs and are maintained in a flooded state with water pumped through canals from the Rio Grande (Robinson, 2010). Regardless of how resacas are managed, in many cases they provide the only freshwater habitat for many species of migratory birds, fish, amphibians, and reptiles in the semi-arid environment of the Rio Grande Valley (Burkes-Copes et al., 2005). Along with the numerous plant and animal species known to inhabit resacas, they are important habitat for several state listed species such as the south Texas siren (*Siren intermediz texana*), black spotted newt (*Notophthalmus meridionalis*), and white-faced ibis (*Plegadis chihi*) (TPWD, 2016).

Another critical component of resacas is the riparian zone found along their banks. The majority of resacas located within the urban environment are plagued with erosion caused by wind driven waves or flooding (Burkes-Cope and Webb, 2012). As a consequence, retaining walls are built that greatly diminish the riparian-water interface (Dahm et al., 2002; Gegory et al., 1991). In addition, much of the habitat needed by birds, reptiles, fish, and amphibians is lost by the removal of native trees and shrubs. The common practice of removing native riparian vegetation, such as sabal palm (*Sabal mexicana*), black willow (*Salix nigra*) and montezuma cypress (*Taxodium mucronatum*), facilitates the installation of retaining walls, altering the local runoff hydrology. Following the installationof retaining walls, many properties are converted to turf grass and mowed to the water's edge (Burkes-Cope et al., 2005). With the native vegetation gone in less manicured resaca banks, invasive species such as Chinese tallow (*Triadica sebifera*), giant river cane (*Arundo donax*), salt cedar (*Tamarix spp.)* and Brazilian pepper (*Schinus terebinthifolius*) can easily dominate open niches (Seawright et al., 2009; Hunter et al., 1988; Burkes et al., 2005, Esparza-Diaz et al., 2011). The conversion from the original riparian habitat eliminates many important ecosystem processes such as sediment and nutrient filtering (Gregory et al., 1991).

Despite the general recognition of the important ecosystem services are provided by resacas, a continuing trend of degradation is seen due to anthropogenic causes (Woodward and Yui, 2000). Urban resacas in the lower Rio Grande Valley, which are the focus of this study, are plagued with high sedimentation rates, nonpoint source pollution, degraded water quality, illegal dumping of solid waste, invasive species, and riparian modification. Currently, attempts are being made to improve the quality of the aquatic habitat and public perception of resacas through a restoration project involving dredging, bank stabilization, public awareness, and environmental education. As relevant as this ecosystem is to the area, little ecological research has been conducted on resacas. The lack of basic ecological information and understanding is a fundamental impediment in monitoring, maintaining, and restoring these valuable wetlands.

Purpose

The purpose of this study is to establish and evaluate a health index for local resacas based on easily obtainable and repeatable metrics. Limited data on both structural and functional aspects of the resaca ecosystem are currently available. In gathering baseline data on these aspects we aim to assemble a Resaca Health Index (RHI) that will ideally help fill the gap in our understanding of these regionally relevant aquatic systems. If used to its potential, measured indicators and the composed multimeric index will be employed as an assessment tool in monitoring programs to determine degradation or recovery of resacas. The RHi is particularly needed to evaluate the effectiveness of restoration efforts currently occurring in Brownsville.

Hypotheses

1) Selected functional and structural metrics will discriminate resacas with various levels of degradation or recovery.
2) Resaca pools have differences in decomposition rates, fish communities, and trophic state.
3) A combination of structural and functional metrics will allow for the creation of a multimetirc index to assess ecosystem health.
4) Resacas across distributary systems have a different score of Resaca Ecosystem Health Index (RHI).

Objectives

1) Periodically collect data over 12 months on several potential indicators of structure and function in six resaca pools with a priori different ecosystem health status.
2) Assess the ecosystem health status of resacas from three river distributary and oxbow systems.
3) Assemble an index using least redundant but most site discriminating metrics collected, and evaluate the six resaca pools in this study.

CHAPTER II

METHODOLOGY

Site Description

Six resaca pools located within the city of Brownsville, TX (Table 1)(Figure 3) were monitored and assessed monthly between September 2015 and August 2016 for this study. Sites were selected through preliminary observations of degradation and existing information based on distributary systems, location of pool within system, and presumed stressors. The Brownsville area has three resaca systems covering 70 plus river miles, two of which were included in this study: Town Resaca system and Resaca De La Palma system. The area also includes numerous oxbow systems, one of which is included in this study: Fort Brown-Lozano Banco system. No sites were selected in the Rancho Viejo system (Figure 3). Two pools from each system were selected, one each on the upstream and downstream ends of the system. Pools were also selected based on differentiating ecological states. All systems were heavily influenced by urbanization having differing degrees of riparian degradation, water depth, and anthropogenic impacts.

Town Resaca System

Town Resaca system (Figure 2) contained 17 pools and flowed through central Brownsville. This system is extensively affected by urbanization and the majority of it is designated for flood control. The pools selected within the Town Resaca system included Resaca Boulevard (RB) on the upstream end and Dean Porter (DP) on the downstream end. Both sites were part of the Brownsville Public Utilities Board (PUB) dredging project along with Cemetery Resaca, which lies between the two study sites. The dredging phase of the Resaca Restoration Plan (RRP) was completed for DP in the summer of 2015. Resaca Boulevard was still being dredged at the time of writing this thesis. In addition to the dredging, other restoration efforts such as bank stabilization and revegetation and public education are being implemented; many of which may have significant impacts on the metrics discussed in this study.

Resaca Boulevard pool (Figure 4) sits at the upstream end of the Town Resaca system and is one of the first pools in this system to receive pumped water from the Rio Grande. Pool size is 42,274 m^2. The pool flows north to south with the east bank being comprised of residential neighborhoods and the west bank being mostly covered by unmanaged vegetation dominated by invasive species. The centerline is shallow, averaging 46 cm, and contains several islands in various states of erosion due to wind waves.

Dean Porter pool (Figure 5) is surrounded by residential neighborhoods along the north and west bank forming a horseshoe. Dean Porter Park and Cummings Middle School occupy the interior. Pool size is 46,977 m^2 with an average centerline depth of 162 cm. Dredging targeted reaching maximum depth at the centerline via a gradual slope from the shallows. Pumped water flows through the Gladys Porter Zoo into the DP pool and out via drainage ditches towards the Brownsville ship channel. The DP pool also receives storm water runoff from downtown via a storm/sewer system bypassing overland flow.

Resaca de la Palma System

The Resaca De La Palma system (Figure 2) consisted of 28 pools and was the longest system studied. The De La Palma system flowed from northwest to southeast while transecting Brownsville many residential areas of Brownsville. This system received pumped water from the Rio from both BPUB (northwest side) as well as the Brownsville Irrigation District (southeast side). Pools selected for the study included Valley International Country Club (VICC) and Billy Mitchell (BM).

The Valley International Country Club pool (Figure 6) sits between residential areas along the southern bank and the golf course on the northern bank. Pool size is 40,618 m^2 with an average centerline depth of 81 cm with the pool reaching shallower depths of 52 cm downstream. Waters flows through several pools within the country club before reaching the study pool. This pool also was a primary receiving point for pumped water from the Rio Grande.

The BM pool (Figure 7) sits on the far downstream end of the De La Palma system and is currently one of the longest non-fragmented pools in Brownsville. Road crossing over this pool is made via bridges instead of culverts. Pool size was 67,981 m^2 with an average centerline depth of 164 cm. Urbanization around this pool is light due to zoning restrictions and landowner decisions to maintain vegetated shore lines. The pool forks on the upstream end with one section flowing west while the other flows south. The bank west of the fork presents evidence of the well-preserved native riparian habitat and is believed to be the original bank from before the resacas were cut off from the Rio Grande.

Fort Brown-Lozano Banco System

The Fort Brown-Lozano Banco system (Figure 2) is believed to be the most recently formed system as it sits alongside the current Rio Grande Channel and is made of five relatively smaller pools that together form a classic, oxbow lake. Since hurricane Dolly in the summer of 2010 destroyed the pumping station, this system receives no pumped water and relies solely on rainwater and runoff from surrounding areas, leading to greater variability of water levels. Prior to 2010, water was pumped into Fort Brown resaca (FTB) and flowed through a culvert to Lozano Banco (LB). Much of the land surrounding the system was converted and used by the military during the Mexican-American War and Civil War. Currently, a university, junior college, and high school occupy the adjourning lands. Pools selected within the Fort Brown system were FTB and LB as they were the only ones that continuously contain water.

The FB pool (Figure 8) was the largest pool studies and it formes a horseshoe. Pool size was 118,540 m^2 with centerline depths ranging for 30 cm to 170 cm throughout the year. The pool was cut off from the river and currently has no flow. The majority of the pool is surrounded by retaining walls and riparian vegetation is lacking. The pool contained three islands; one upstream, one down stream, and one in the middle of the northern stretch of the horseshoe.

The LB Resaca (Figure 9) is the eastern portion of the Fort Brown System and has a pool size of 35,261 m^2 and an average centerline depth of 177 cm. The resaca is bordered by retaining walls in two small sections, supporting a wood bridge, but has large tracks of aquatic vegetation growing along the edges due to prolonged times with low water levels. This resaca shares similarities with a majority of the resacas outside of the urban area and may provide indications of how they function.

Water Quality Measurements

Water column parameters were collected monthly from September 2015 to August 2016 at four predetermined sampling stations within each pool. A multi-parameter sonde (Hach HQ40d) was utilized to measure dissolved oxygen, pH, temperature, specific conductivity, and salinity. Water clarity was determined using a secchi disk at each point along with total depth. Due to equipment failure pH was not recorded for RB during September 2015 and secchi depth during December 2015 at DP.

Biotic Community Assessment

Assessment of the biotic community was initially planned to focus on benthic macroinvertebrates to derive structural indicators, as described in McIntosh (2014). Preliminary results from study sites indicated high variability in species richness and abundances within the same pool. This high variability is believed to be due to considerable differences in depth and sediment composition. Based on these preliminary results a decision was made to replace the benthic community assessment with a fish community assessment in order to derive more consistent structural indicators (Barbour et al., 2003).

Fish communities were sampled in October 2015 for fall, February 2016 for winter, and June 2016 for summer using experimental gillnets and cast nets. Experimental gill nets were 38.1 m in length, with five 7.62-meter panels of mesh size: 2.54, 3.81, 5.08, 6.35, and 7.62 cm (Memphis Net and Twine, Memphis, TN: Hubert et al., 2012). Cast nets had a 121.9 cm radius with a mesh size of 0.95 cm (The Fitec Group, Memphis, TN).

On each sampling date, experimental gill nets were set for 30 minutes to allow for rapid assessment and to minimize by-catch of turtles and diving birds (Siesennop, 1997), which were noted near the set during every event. Two gill net sets were conducted at each pool on the same day. In addition, ten cast net throws were conducted during the gill net sampling to collect smaller fish (Emmanuel et al., 2008). Nets were deployed randomly but perpendicular to the shoreline throughout the sampled pool and sampling events. Nets were always set with the smallest mesh size closest to shore to increase the likelihood of capturing smaller fish. Fish captured from both methods were placed in aerated livewells, weighed to the nearest gram, measured to the nearest ±1mm, and released. All fish were handled in accordance with State regulations under permit SPR-0913-125.

Decomposition

As a measure of ecosystem functioning decomposition rates using litterbags were estimated in the water column of the six pools. A total of eight bags, two at each of four stations, were deployed for 60 days (Chadwick et al., 2006). Methods used were based on a previous study conducted to assess functional recovery of local aquatic systems (Marquez et al., 2016). Mature sabal palm (*Sabal mexicana*) fronds were used as the decomposition substrate and were collected from a local palm grove. Fronds were cut into similar sized pieces (3 cm long), rinsed to remove debris, and dried at 60 °C for a minimum of 72 hours leading to a constant weight. Litterbags were 15 centimeter by 15 centimeter square nylon mesh envelops with a mesh size of 1 mm. Bags were filled with 10 grams of dried frond pieces, along with an identification tag before being stapled shut. Litterbags were deployed between September-November 2015, January-March 2016, and May-July 2016.

Each sampling station was marked with a 2.54 x 205 cm PVC pipe driven into the sediment. These stations remained the same throughout the study. Care was taken when attaching litterbags to avoid contact with the sediment and ensuring continuous submersion of the bags. At each station, both bags were tied together in the upper left corner and attached to the post with multiple zip ties. After 60±1 days, bags were retrieved and placed in individual plastic bags and on ice for transport. Once returned to the lab, bags were rinsed to remove accumulated sediment and biofouling organisms. Following rinsing, bags were placed in the refrigerator overnight at 4 °C to remove excess water. After refrigeration, bags were dried at 60 °C for a minimum of 72 to 96 hours. Once dry, bags were opened and the material removed by hand, weighed and recorded as final weight. The remaining biomass was ground and homogenized using a coffee grinder and a subsample placed in a vial to be incinerated. Two grams of homogenized subsample were weighed into crucibles, dried at 105 °C for 24 hours and reweighed to determine percent humidity. After weighing the crucibles were placed in a muffle furnace (Thermo Scientific F6010) and incinerated at 500 °C for 6 hours. After cooling the crucibles were weighed again in order to denote the mass of ash remaining. Ash free dry weight was calculated and used to correct for sediment infiltration.

Sediment Analysis

Sediment samples were collected from each resaca pool in August 2016 using an Ekman dredge for basic site characterization (particle size distribution and total organic matter). Transects were determined using Esri ArcMap 10 Fishnets and transferred to Google Maps for use in the field. Dredge samples were collected following transects and subsamples that were similarly colored and textured were homogenized for analysis. Particle size distribution was performed using a wet sieve protocol (Folk, 1974). Sediments were wet sieved through a 63-micron sieve with the remaining mass dried at 60 °C and dry sieved using a mechanical shaker. The remaining mass was classified as pebble (4mm), granule (2mm), very coarse sand (1mm), coarse sand (500 µm), medium sand (250 µm), fine sand (125 µm) and very fine sand (63 µm). Material passing through the 63 µm wet sieve was classified as silt and clay. Total organic matter content was determined by drying a subsample at 105 °C for 24 hours and incinerating in the muffle furnace (Thermo Scientific F6010) at 500 °C for two hours to obtain the ash-free dry weight remaining.

Riparian Assessment

During September 2016, riparian assessments were conducted using a drone imaging system (DJI Phantom 3 Professional 12.74 megapixel). Drone technology was utilized due to issues with private property access and the need for better imagery than what was available from satellite imagery (50 cm pixels). Images were processed using Harris Envi software which georeferenced all images and allowed for accurate measurements to be taken using processing tools in Esri ArcMap 10. To evaluate the riparian zone, 10 m buffers were placed around resaca pools and divided into equal transects, based on square meters, allowing images to be analyzed for vegetation cover, vegetation structure, and channel alteration using an adaptation of the Qualitat del Bosc de Ribera (QBR) index created by Munne et al (2003).

Data Analysis

Data from gillnet and cast net samples were combined to form one sampling event for analysis. Species abundance and biomass were determined from monthly sampling and overall for the study period. Community structure between sties was analyzed for species richness, diversity, dominance and evenness using Shannon-Wiener diversity ($H' = - \Sigma\ pi \ln (Pi)$), Simpson dominance ($D=1/ \Sigma\ pi^2$), Pielou's evenness ($J'=H'/\log_e$), and Margalef diversity ($d=(s-1)/\ln N$) where s=number of species, N=total number of individuals per total, n=number of individuals per species, and pi=n/N. Dominance ratio was calculated using dominance ratio N_{max}/N where N_{max}=abundance of most abundant taxa. Fish community index (FCI) was composed of scores obtained from Shannon diversity index, richness, dominance, and number of piscivorous species (Fausch et al., 1990).

Differences in community structure between seasons and sample pools were analyzed using PRIMER v6. Multidimensional scaling (MDS) analysis was used to visually assess if there were differences between fish communities by configuring samples based on similarities. The MDS was supported by a one-way analysis of similarity (ANOSIM) on square root transformed data to identify differences in fish communities among pools. Similarity percentages (SIMPER) analysis was used to determine which species were driving dissimilarities between sites and similarities within sites using untransformed data (Clark and Gorley, 2006). All multivariate analyses were derived from Bray-Curtis similarities.

Environmental data was analyzed separately from biotic data. A principal component analysis (PCA) was used to investigate if environmental factors were driving community differences between the six study sites. Secchi disk depths were used to assign a trophic state index value to each resaca following Carlson's Trophic State Index (TSI; Carlson, 1977) using the following equation:

$$TSI\ (SD) = 10 * (6 - (Ln(Secchi\ Depth)/Ln(2)))$$

Preliminary determinations showed that TSI and trophic state classifications of resacas were similar when obtained with chlorophyll α concentrations or secchi depth (Table 2). A two-way ANOVA was computed to determine if there was a significant difference between site and season and its interaction.

Riparian habitat was assessed using a modification of the QBR, which allowed for observations to be made with limited plant identification skills (Table 3). Resaca bank sections were scored in 3 categories for a maximum of 100 points: total vegetation cover (0-40 points), vegetation structure (0-30 points), and channel alteration (0-30 points). The criteria for each category was adapted to better reflect the conditions of urban resacas. When total vegetation cover was comprised of more than 50% natives species, an adjustment of +10 points was made. When more than 20% of the surface was impervious non-vegetated surface, the adjustment was -5 points. Vegetation structure was adjusted with -10 points when more than 50% of the surface was bare ground or had mowed herb beneath the canopy. The channel alteration score was adjusted with -5 points when width-constraining structures such as bridges, weirs, or culverts were present. Scores derived from riparian habitat were used to classify sites as excellent, good, fair, poor, or very poor (Table 4) (Munne et al., 2003).

Decay constants (K) were obtained from a single component exponential model assuming that leaf litter decomposed at the same rate throughout the 60 days of litterbag deployment. K was calculated as follows (Petersen and Cummins, 1974):

$$K=\ln(W(t_f)/W(t_i))/(t_f-t_i)$$

where $W(t_i)$ is initial weight (ash free dry weight), $W(t_f)$ is final weight remaining (ash free dry weight), and (t) is the time in days of the litterbags being submerged (60 or 61 days).

Percent humidity was calculated using weights obtained at 105 °C; ash content was the weight remaining after incineration at 500 °C and used to calculate percent ash needed to determine ash free dry weight (AFDW).

Decay rates from all 8 bags were averaged to obtain the seasonal K value. A two-way ANOVA was run to determine a relationship between sites and seasons.

Resaca Health Index (RHI)

The Resaca Health Index was derived using obtained metrics for decomposition, fish community, riparian habitat, and trophic state of each water body. The specific metrics were decay constant (K), Riparian Index (RI), Trophic State Index (TSI), and the Fish Community Index (FCI). RHI is therefore a multimetric index calculated as:

$$\mathbf{RHI = K+RI+TSI+FCI}$$

Due to each metric exhibiting values on different scales, RHI scores were calculated using normalized data, allowing each metric to be weighted equally based on an unit-less scale (Brooks et al., 2007; Fano et al., 2003; Blocksom, 2003; Baptista et al, 2011; Hering et al., 2006). Each metric was scaled to 25 points with possible scores ranging from 0 to 25. FCI was calculated based on: Shannon diversity, species richness, dominance ratio, and number of piscivorous species with each being worth 6.25 points (6.25x4=25).

Reference conditions were determined a posteriori as the best obtained value for each metric in a given season regardless of pool (Brooks et al., 2009; Gibson et al., 1996; Reynoldson et al., 1997).

CHAPTER III

RESULTS

Sediment Analysis

Sediments for all six sites were primarily composed of silt and clay (62.87-84.47%). LB had the highest amount of silt and clay at 84.47%, followed by RB, VICC, BM, FTB, and DP (Figure 10). Remaining fractions ranged from pebble to very fine sand. Mean organic content of sediments ranged from 5.91 to 8.03%, and was highest at BM followed by LB, FTB, VICC, DP, and RB (Figure 11).

Water Quality Parameters

Water temperatures ranged from 18.6 °C in January to 32.5 °C in July with an average annual temperature of 26.3 °C for all pools (Table 5). Monthly conductivity ranged from 589 at FTB to 3120 at DP (µs/cm) with a mean annual conductivity of 1501 µs/cm among all sites (Table 5). DP exhibited a large variation in conductivity starting the study at 1279 µs/cm during September 2015, highest recorded at 5900 µs/cm during March 2016, lowest of 915 µs/cm during June, and recording 4090 µs/cm at the end of the study. It should be noted that water levels also fluctuated Monthly pH ranged from 8.6 to 9.2 with a mean annual pH of 8.7. Dissolved oxygen ranged from 2.5 to 21.0 mg/L (Table 5). Mean annual dissolved oxygen was 9.3 mg/L with the highest monthly average recorded at LB followed by DP, RB, FTB, VICC, and BM (Table 5). Secchi depth ranged from 10 cm to 75 cm with an annual average of 29.25 cm (Table 5).

Total depth ranged from 52 cm at RB to 177 cm at LB (Table 5). Based on Carlson's TSI using secchi depth, all sites were classified as hypereutrophic or eutrophic but significant differences in TSI scores were observed among sites ($F_{df=5}$=172.290; p= >0.001) and seasons ($F_{df=2}$=28.843; p=>0.001)(Table 6)

Principal Component Analysis ordination determined that secchi depth, dissolved oxygen, and conductivity were strongly driving differences among the six pools (Figure 11). Three principal components accounted for 78% of the variation (Table 7). PC1 accounted for 33.2% of the variation with secchi depth, pH, total depth, and dissolved oxygen contributing the most. PC2 accounted for 25.2% of the variation with dissolved oxygen, total depth, and secchi depth. PC3 accounted for 19.8% of the variation and was strongly driven by conductivity and pH.

Fish Community

Fish were collected from all six resaca pools on three occasions representing 18 species, with a total of 254 individuals captured in fall, 276 individuals captured in winter and 238 individuals captured in summer (Table 8 and 9). DP accounted for 26.7% of the total individuals from 12 species including four invasive species (Table 8). VICC accounted for 22.9% of total individuals from 11 species including two invasive species (Table 8). LB accounted for 17.0% of total individuals from 11 species including three invasive species (Table 8). RB accounted for 16.4% of total individuals from nine species including two invasive species (Table 8). FTB accounted for 8.7% of individuals from six species including two invasive species (Table 8). BM accounted for 8.2% of individuals from 11 species including two invasive species (Table 8).

The species with the greatest abundances were *Dorosoma cepedianum*, *Dorosoma petenense*, *Pterygoplichtys disjunctivus*, *Ictiobus bubalus*, and *Oreochromis aurea* combining for 662 individuals (Table 8). Native species accounted for 594 individuals (77.3%) from 14 species and invasive species accounted for 174 individuals (22.7%) from four species. Species richness varied among sites ranging from four species at FTB to nine species in BM and RB (Table 10). The number of piscivorous species ranged from zero at both Fort Brown and RB in all sampled months to three at BM and VICC in fall (Table 10). Dominance ratio ranged from 0.21 at BM to 0.71 at RB in winter (Table 10). Shannon diversity index ranged from 0.92 at FTB in winter to 1.99 at BM in summer (Table 11). Pielou's evenness ranged from 0.52 at DP in June to 0.93 at BM in February (Table 11). Simpson index ranged from 1.10 at FTB during winter to 6.69 at DP during summer (Table 11). Margelef's richness ranged from 0.92 at FTB during winter to 2.55 at BM during summer (Table 11).

Multivariate community analysis with square root transformed fish abundances showed no significant difference between months (ANOSIM: R=-0.053; p=0.914) but significant differences among pools (R=0.288; p=.0001) (Table 12). Post hoc analysis showed (Tukey HSD) no significant differences (p>0.05) between (Table 12). Pools showing significant differences included: FTB-LB, FTB-DP, FTB-BM, FTB-VICC, LB-VICC, DP-BM, DP-VICC, RB-BM, RB-VICC, and BM-VICC. Pools showing no significant differences included: FTB-RB, LB-DP, LB-RB, LB-BM, and DP-RB. Results were visually represented using a multi-dimensional scaling (MDS) plot showing groupings; however, there was a large amount of overlap (Figure 12). SIMPER analysis showed *D. cepedianum* was the leading species accounting for similarity within pools (Table 13). No single combination of communities was responsible for differences between the six pools. Cluster analysis showed that regardless of factor, no significant groupings were discovered.

Riparian Assessment

Riparian habitat condition was assessed using a riparian index, with observed scores ranging from 38 to 99 overall with an average of 62 (Table 14). No sites were classified as very poor quality but three sites, DP (38.44), FTB (38.65), and VICC (48.67) were scored as poor quality. RB (73.33) was scored as fair quality, BM (77.08) was scored as good quality with some disturbance and LB (99.44) scored as excellent quality.

Riparian index scores varied greatly between sites in the vegetation structure component, (% tree and shrub) ranging from 2.0 to 28.3 out of 30 points primarily due to surrounding land use (Table 14). The highest scoring resaca in the vegetation structure component was LB having an almost continuous woody riparian area (Figure 9). BM and RB, which scored second and third, exhibited excellent cover, fair to good structure but had large areas of channel alteration (Figure 7 and 4). VICC scored lowest largely due to the northern bank being manicured turf grass utilized by the country club for golfing (Figure 6). Other low scoring resacas include FTB and DP. FTB riparian area has been cleared largely of woody vegetation except for small areas on each end and along the southern bank (Figure 8). Due to its location and surrounding land use, much of the area has been converted to turf grass for ease of maintenance. DP's riparian area on the inside bank of the bend has been largely cleared for access to the water from the park (Figure 5). Many low scoring sites received a -10 point adjustment due to having large amounts of manicured turf grass below the woody canopy.

Vegetation cover had the highest weight in this evaluation having a potential to yield up to 50 points (Table 3). Scores for this component of the riparian assessment ranged from 28.4 to

46.6 with LB scoring highest and DP scoring lowest. Scores in this category were greatly influenced by the presence of turf grasses. Other factors influencing cover scores included a -5 point adjustment for greater than 20% impervious non-vegetated surfaces and a +10 point for greater than 50 % of native species cover.

Channel alteration scores ranged from 4.0 to 25.6 out of 30 and was influenced most by the presence of rigid structures (i.e. retaining walls) (Table 14). LB scored highest in the channel alteration category having the largest intact riparian-water interface with FTB scoring lowest. All resacas except LB had a -5 point adjustment due to the pool having flow restrictions/hydraulic strucutre in place (i.e. culverts, bridges).

Decomposition

Decomposition rates from all litterbags were averaged to determine the seasonal decay constant (Table 15)(Figure 14). For this study decay constants showed significant difference based on site ($F_{df=5}$=20.677, p= >0.001) and season ($F_{df=2}$=115.526, p=>0.001). Post Hoc Tests (Tukey HSD) determined that all sites were significant different, as well as all seasons. During the fall season, DP exhibited the fastest decomposition rate (K=0.0120) with the slowest occurring at RB (K=0.0078) on average (Table 15). Winter decomposition rates were fastest at BM (K=0.0078) and slowest again at RB (0.0042) on average (Table 15). Summer decomposition rates were fastest at DP (K=0.0204) and slowest once again at RB (0.0084) (Table 15).

Resaca Health Index

The proposed RHI was calculated seasonally for all sites using the normalized values derived from the riparian index (RI); the trophic state index (TSI); the fish community index

(FCI); and the decay constant (K). Riparian index scores were obtained only once (August 2016), and used for each season, as riparian conditions did not change throughout the study period.

The FCI was calculated seasonally using normalized values for Shannon diversity, species richness, dominance, and number of piscivorous species as a measure of community complexity. Scores ranged from 8.0 to 25.0 (Table 16). During the fall, VICC exhibited the highest overall FCI score, recording the best values in three of the four metrics (Shannon, richness, piscivorous species) (Table 16A). LB recorded the best value in the fourth category having the lowest dominance. Winter sampling resulted in BM recording the highest overall FCI score having the best values in all four categories (Table 16B). BM again recorded the best-observed values in all four categories yielding the highest FCI during summer (Table 16C). FTB scored worst in all seasons primary due to low species richness and the absence of piscivorous species.

Normalized RI scores were determined based on site averages from transects and overall ranged (after normalization) from 9.7 at DP to 25.0 at LB (Table 17).

Trophic state index scores were calculated using, twelve secchi depth observations (four from each month of the season) and were scored based on the lowest trophic index being awarded the maximum of 25 points. LB recorded the best trophic state score being awarded 25 points and RB recorded the worst in all three seasons (Table 17).

Normalized decomposition scores for the RHI were determined based on setting the reference score (25 points) to that on LB, which was assumed to have the least altered decomposition process, based on having the best riparian conditions and water quality (Table 17). Decomposition scores had the largest variation among sites during winter (16.0-25.0) and the smallest variation during fall (19.57-25). Lowest scoring sites for this component of the RHI were DP in fall, RB in winter, and RB in summer.

RHI final values were determined by adding normalized values of RI, TSI, FCI, and K for a potential maximum score of 100 (Table 17)(Figure 15). Overall, LB ranked the best during fall and winter. BM, which ranked second in fall and winter, ranked first during the summer. FTB consistently scored as the worst pool throughout the study. Fall scores ranged from 92.93 at LB to 65.08 at FTB with a seasonal average of 77.69 for all six sites (Table 17). Winter scores ranged from 94.47 at LB to 62.80 at FTB with an average of 74.02 for all six sites (Table 17). Summer scores ranged from 91.77 at BM to 64.28 at FTB with an average of 77.07 for all six sites (Table 17).

CHAPTER IV

DISCUSSION

Resacas

Historically, rescas experienced a regime of periodic flooding with high river levels followed byprogressive drying and filling in over time. Due to urbanization and controlled water use, resacas no longer functioning in this fashion. These systems are now artificially maintained in a suspended state of succession, thus creating a novel ecosystem. This conversion and altered hydrology have led to changes in riparian vegetation, fish and invertebrate communities, along with increased the impacts of urbanization such as turbity and sedimentation.

Sediments

Sediment particle size showed all pools in this study were dominated (>60%) by silt and clay. Results were consistent with McIntosh (2014) findings of silt and clay percentages in rural resacas that remained permanently flooded. DP displayed a noticeably lower percentage (63%) of silt and clay, perhaps as an effect of recent dredging. Sediment organic matter (SOM) was highest at LB and BM, which corresponds to the report that more urbanized wetlands have shown decreased SOM (Meyer et al., 2005). This trend is possibly a reflection of more plant residues entering these two resacas from the more vegetated banks compared to the other pools. Pools had similar SOM contents to results observed in McIntosh (2014) with FTB and VICC being lower than reported values. The lowest SOM contents were recorded at DP and RB with both pools SOM was likely as a consequence of the dredging project. Both our results and those from McIntosh (2014) differed from results found by Stanley and Randazzo (2001) in relation to particle size and SOM.

Water Parameters

Among recorded water parameters, conductivity varied the most between pools. DP exhibited highly variable conductivity based on monthly averages. Currently no explanation can be constructed as to the range observed; however it is believed that water level fluctuations due to on-going dredging operations upstream could have factored in the variation.

LB appeared to have the highest dissolved oxygen average concentration, as well as the highest water clarity, averaging nearly double the recorded values for the other resaca pools. Tropic state index was lowest for LB over the three sampling seasons. This pool has abundant emergent aquatic vegetation on its shores, which may limit phytoplankton abundance and suspended sediments, thus increasing secchi depth (Qui et al., 2001). In terms of water quality, this resaca could be considered as a reference site within the context of sites found in the Brownsville urban area. Pools recording high TSI values had little to no aquatic vegetation, however the presence of vegetation was not enough to change site trophic classification, which remained eutrophic or hypreutrophic for all pools. It should be noted that during the course of this study the dredging impacted RB and DP water quality.

Fish Community

Throughout the six study pools all fish species found have been previously documented in the Rio Grande or the resaca systems (Mora et al., 2001; Edwards and Contreras-Balderas, 1991). However, limited data currently exists to compare species richness and abundances therefore this study is baseline for this aspect of the aquatic habitat. Texas Parks and Wildlife (TPWD) created a regional index for biotic integrity for use in Texas Ecoregion 34 (Western

Gulf Coast Plain), which includes Brownsville, however, no sampling efforts occurred in the lower Rio Grande Valley to validate this index (Linam et al., 2002). It is important to note that unlike many indices, including the TPWD index, that penalize scores when invasives are present, the FCI considers invasive species equal to native species in all community metrics primary due to the novelness of the ecosystem and the assumption that such system is not pristine. Total abundances were dominated by *Dorosoma cepedianum*, which accounted for 39.8% of all individuals captured in all sites combined. Studies have shown that in warm water systems with high turbidity and mud bottoms that are similar to resacas, *D. cepedianum* have shown the ability to rapidly dominate communities and limit game fish production (i.e. *Micropterus salmoides*, *Lepomis spp., Promoxis annularis* (Bennett, 1943; Aday et al., 2002; Miranda, 1983). This dominance and high trophic level production limitation could explain the low observed abundances of these piscivorous species. Due to rapid growth and the lack of piscivorous species found in many recasas, *D. cepedianum* can influence phytoplankton biomass, as well as nitrogen and phosphorus dynamics due to feeding methods (Schaus et al., 1997; Mundahl and Wissing, 1986). This fast transfer of nutrients from detritus to the phytoplankton community was reflected in the TSI used for the calculation of the RHI.

Other species documented in high abundances (>50 individuals) included *Dorosoma petenense Pterygoplichtys disjunctivus, Ictiobus bubalus*, and *Oreochromis aurea*. *D. petenense* was found at all sites however abundances on average were 34.5% lower than that of *D. cepedianum*. *D. petenense* has been shown to be opportunistic feeders feeding on algae and invertebrate eggs (Miller, 2011; Gerdes and McConnell, 1963). The high abundances of *D. cepedianum* and D. *petenense* observed are indicators of their overall high numbers but may also reflect their schooling habits. *Pterygoplichtys disjunctivus* and *Oreochromis aurea,* both invasive species in the region accounted for 11.1% and 9.8% of the total abundances. Due to the urban resaca ecosystem being novel, the presence of non-native species is expected and integrated into all diversity and abundance metrics (Hermoso and Clavero, 2013). *P. disjuctivus* were observed in all study pools and their presence is currently assumed in all resacas. This species was likely introduced into the resacas via aquarium releases just as was determined for other Texas water bodies (Pound et al., 2009). Little is known about their impacts on native fish communities; however, their feeding habits indicate they could be direct competitors of *D. cepedianum* and potential predators of native species eggs (Hoover et al., 2004; Pound et al., 2009). Along with the potential competition with native species, *P. disjuctivus* create borrows for breeding and survival in adverse conditions (Hoover et al., 2004). The excavation of borrows leads to bank destabilization, increased erosion, and increased sediment loads (Nico and Martin, 2001). *Oreochromis spp.* were observed in all study pools except BM and have been observed in the Rio Grande as far back as 1980 (Edwards and Contreras-Balderas, 1991). *Oreochromis spp.* have shown to have a similar diet to those of D. *cepedianum* and D. *petenense* filter feeding on algae and detritus (Lu et al., 2006). *I. bubalus* accounted for 10.53% of all individuals and were observed in all sample pools. I. bubalus, *Cyprinus carpio* and *Aplodinotus grunniens* all benthivorous species, can degrade water quality when they occur in elevated numbers, primarily by resuspending solids through feeding activity (Lougheed et al., 1998; Goetz et al., 2014).

Fifty-four piscivorous individuals from four species including *M. salmoides*, *P. annularis*, *Lepisosteus oculatus* and *Ictalurus punctatus* were captured in all resacas combined. Thirty-four of these were *L. oculatus* captured in BM, VICC, LB, and DP pools. It should be noted that *L. oculatus* was documented in Fort Brown during an unrelated sampling event and was not included in any analysis. These highly opportunistic predators have been well documented in backwaters and represent the highest trophic level in this and other nekton communities (O'Connell et al., 2007). P. *annularis* were observed at BM and VICC possibly indicating a lack of habitat requirements in the other pools (Edwards et al., 1982). M. *salmoides* were observed in VICC, DP, and BM. The presence of *M. salmoides* and *L. oculatus* showed multiple apex predators can be found in resaca pools. *Lepomis spp*. were documented in all study pools except FTB pointing multiple trophic levels observed in forage species. *I. punctatus* were observed in BM, VICC, and LB. It should be noted that Texas Parks and Wildlife (TPWD) has stocked *I. punctatus* and *M. salmoides* in DP and *I. Punctatus* FTB, and BM. No *I. punctatus* were recorded at DP or FTB during sampling for this study but were observed being taken with hook and line. Currently it is unknown what effects recreational and subsistence fishing have on the population of these highly sought sport fish.

Other species collected in low abundances included *Herichtys cyanoguttatus*, *Astyanax mexicanus*, *Menidia beryllina*, *Gobiosoma bosc*, and *Poecilia formosa*. These species were captured with cast net (Medeiros et al., 2008). The use of the cast net was deemed suitable for sampling the smaller members of the fish community, however overall numbers and species may not accurately reflect other species present. This bias was produced due to the net mesh size may not be small enough to capture smaller fish such as *Gambusia affinis*, which were seen at all sites but none captured. Also, adding to the bias was the presence of submerged structures and aquatic vegetation in which the net would snag many have sometimes limited its efficacy. Seines would not be effective for capturing smaller fish in resaca pools because of varying water depths and thick layers of soft sediments making such sampling impractical in most of the sites.

The use of gillnets provided a representative sample of larger fish for our purposes, however the use of this passive technique can add sampling time while limiting the sampling effort. The use of an active technique, such as electrofishing, alone or in combination with other methods, could allow for a more representative sampling, and the possibility of observing species that a single sampling method may not be effective for (Eros et al., 2009). An important factor in choosing gillnets was their weight and ease of movement due to the limited access to many sites as the boat was frequently launched off bridges or over retaining walls. Other factors influencing the decision to use nets included high conductivity (>1,000 us.cm) in many pools and shallow water depth restricting the use of larger boats needed for effective electrofishing (Larimore, 1961).

Overall the fish community sampled exhibited more species than were expected and was different among pools making the fish community metrics potentially good structural indicators to be included in the RHI. Potential factors affecting fish species diversity and abundance in resacas include habitat structure, presence of birds such as *Phalacrocorax auritus*, *Pelecanus spp.*, and members of *Ardeidae*, resaca water use, fishing pressure, and potentially the abundance of invasive species. Documenting and quantifying these causes and effects will lead to possible management options for resaca fish communities and the overall aquatic habitat management.

Riparian Habitat

Due to the importance that riparian vegetation exhibit on water quality it is necessary to understand the makeup of resaca riparian buffers (Naiman and Decamps, 1997; Anbumozhi et al., 2005). Using a modified version of the QBR index allowed for rapid evaluation of the riparian areas surrounding the resaca pools using an established method (Sirombra and Mesa, 2012; Cornell et al., 2008; Colwell and Hix, 2008). The original index was designed to be utilized from stream and river crossings and determined a score based on what riparian vegetation was visible from these limited viewing locations; using this viewing approach would greatly limit the amount of riparian area that could be assessed along resacas, and therefore considerably limiting the accuracy and representativeness of the evaluation.

The use of a drone instead of visual surveys at ground level, made it possible to capture high definition images of the entire riparian zone of each pool within a short amount of time (less than one day for all sites) and process them at a later date. The drone also provided additional capability to analyze areas of the reasaca no readily accessible due to private property concerns.

Drone technology is relativity new and limited information has been gathered on its use for these purposes (Bonin et al., 2014). Images were captured from an altitude of 100 meters allowing for both banks to be included in the same image for most sites. Images captured can serve as excellent references to be able to monitor riparian change over time. Due to the location and the impacts of urbanization, the ideal riparian width of 30 m was determined to be unachievable (Klapproth and Johnson, 2009). A maximum resaca buffer width to be evaluated was determined to be 10 m for the present study allowing the analysis of several important attributes of riparian zones (Osborne and Kovacic, 1993; Hawes and Smith, 2005).

Category scores for varied between sites, but differences in riparian index scores were driven primarily by vegetation structure and channel alteration. As expected in an urban environment, all pools exhibited sections restricted and retaining walls. DP and FTB were almost exclusively bordered by retaining wall eliminating the ecologically important littoral zone and decreasing habitat diversity needed by many species (Strayer and Findlay, 2010; Brauns et al., 2011).

All sampled pools exhibited scores greater than 70% in the vegetation cover category. The presence of turf grass within the buffer translated to higher than expected scores in the cover category. Unlike the original QBR, which excluded herbaceous grasses because many were annuals, our modified index included them because the climate of south Texas allows them to be perennials and several studies noted their benefit in trapping sediments and sequestering nutrients (Osborne and Kovacic, 1993). Considerable points in this category were lost at some pools due to the presence of nonvegetated surfaces located within the 10 meter wide buffers. An adjustment was performed if nonvegetated surfaces were impervious similar to Chadwick et al., 2006. These impervious areas represented more than 90% of all nonvegeteated surfaces and included walkways, patios, swimming pools, and homes. Vegetation cover scores were also adjusted based on the proportion of native species found within each transect. Very few transects were awarded adjustment points based on native species due to much of the riparian zone being located on private property and covered with ornamental non-native species. Several studies looking at riparian diversity found that nonnative species dominated urban riparian areas (Burton et al., 2005; Maskell et al., 2006)

Vegetation structure points based on proportions of tree and shrub cover were relatively low with three pools scoring close to 30%. DP, VICC, and FTB all had limited tree and shrub cover due to land use surrounding the resacas. Each pool scoring low had small areas that exhibited good vegetation structure but overall areas of limited or no vegetation structure dominated the riparian areas of these pools eliminating ecosystem services such as reducing sediment and nutrient inputs as reported by Thawait and Chauhan (2014). This lack of tree/shrub structure drastically reduces buffer function and impacts water quality and ultimately the aquatic habitat (Lee et al., 2003; Mankin et al., 2007). A five point deduction was also applied if the area under the canopy was mowed herbaceous vegetation allowing points to be awarded for areas with tree structure but lacking shrub structure. This adjustment also took into consideration the release of much of the stored nutrients (Doosskey et al., 2010). Similar to what was observed by

Dutcher et al. (2004) it is believed that the importance of riparian vegetation is no fully understood by landowners. Overall, the RI appears to be a critical component of the RHI as it directly assesses the habitat quality.

Decomposition

Leaf litter decomposition rates varied between sites and seasons identifying differences in ecosystem processes such as organic matter turnover and nutrient recycling among sites. Currently no data exists for systems similar to resacas but similar results have been documented in many degraded systems worldwide (Gessner and Chauvet, 2002; Brinson et al. 1981). The seasonal effect of decomposition was noticeable as all pools experienced higher decomposition during summer and lower decomposition during winter pointing to a temperature affect, which has been suggested as the most important variable in sites with similar dissolved oxygen (Figure 16) (Brinson et al., 1981). Overall, dissolved oxygen concentrations observed in all sites in the present study could be considered adequate for aquatic life including microbial decomposers; however all measurements were obtained during day light hours (Young et al., 2008). Leaf litter in RB had the slowest decomposition in all three seasons, which could possibly be attributed to an increase in suspended solids from the dredging project that eventually entered the litterbags, partially covering and physically protecting decaying leaf litter from microbial attack similar to that reported in Fierro et al., 2000. Indeed, ash content of remaining litter averaged 41% in Resaca Blvd, which was twice as much as all other sites except VICC (35%), which also likely experienced similar effects as RB. DP experienced a large decrease in decay rates during winter indicating the presence of an unknown inhibitor to the microbial community. During fall, DP exhibited the highest decay rate (0.0120), dropped below all sites except Resaca Blvd during winter (0.0045), and returned to the highest rates during summer (0.0204). This was not seen in other metrics, and shows the importance of using multiple functional and structural metrics for a more accurate assessment of ecosystem health. The increased rates observed during fall and summer could possibly be driven by DP pool's location just downstream of the Glady's Porter Zoo. Water flowing through the Town Resaca system toward Dean DP passes through a series of concrete canals inside the zoo that receive runoff for multiple animal exhibits thus increasing nutrient loads. The potential for nutrient loading along with the input of storm water via sewers from the highly urbanized areas surrounding the pool make DP a pool where decomposition process is likely altered (Carpenter and Adams, 1979). Fort Brown and LB exhibited the smallest seasonal changes and were the pools that most resembled the original fluctuations of periods of high and low water levels but were dependent on rainfall instead of flooding. Decay rates at LB dropped during winter when compared to fall but returned to similar decay rates during summer. LB recorded lower decay rates than FTB during fall and winter but higher rates during summer indicating the potential presence of a decay accelerant. BM and VICC had relatively similar K constants during fall and winter but VICC jumped drastically during the summer sampling period. This increase was possibly due to nitrogen and phosphorus runoff from increased fertilizing of the golf course at the country club (Graves et al., 2004). BM had comparable rates to LB showing both pools function similarly in respect to this important ecosystem process.

Resaca Health Index (RHI)

As urbanization and residential development continue to increase around Brownsville's resacas, the need for an ecosystem assessment tool is critical, especially in view of the multiyear resaca restoration program currently under way (Ehrenfeld, 2000). With the emergence of increased anthropogenic impacts on a novel ecosystem, the need for monitoring is paramount in determining if management is adequate or if improvement is needed (Dale and Beyeler, 2001). The overall objective of the RHI to be a reliable tool for accurate assessment and monitoring of resaca ecosystem health based on metrics derived from aspects of ecosystem structure and function (Meador and Goldstein, 2003). Structural metrics were derived from the fish community and riparian buffers, whereas the riparian index is an indicator of habitat structure and quality within riparian buffers. Functional metrics included trophic state index and decay constant. Both identified LB as the healthiest pool in terms of function, but is important to remember that the best K was assigned a priori LB based on factors known to affect decomposition. If any selected metric was used separately to access ecosystem health rankings for resaca pools, it would be slightly different with LB scoring highest in TSI, K, and riparian but VICC and BM ranking highest in FCI depending on season. Each selected metric represents a different component of the resaca ecosystem and are not redundant; therefore a more accurate assessment should be expected when all are included in the RHI.

The major issue that arose in the creation of the RHI was determining the reference conditions or the condition representative of a minimally disturbed site, even if it is a novel ecosystem and not pristine (Reynoldson et al., 1997). If urban, managed resacas are recognized as novel aquatic ecosystems, then no true reference site exists that can be used in determining reference conditions. Based on the lack of a true reference site the best values obtained for RI, TSI, and FCI should logically constitute the reference conditions as used in this study. Resacas with an ecosystem function depending on periodic or sporadic flooding recurrence originating from the Rio Grande are now extinct as river flows are heavily controlled, effectively preventing the use of historical observations. (Brooks et al., 2009). As more resaca pools are evaluated over time with the proposed metrics, reference conditions will emerge and can be adjusted in accordance. In other words, reference conditions for the use of the RHI should be considered subject to change. Data collected from the six resaca pools included in this study pointed at Lozano Banco representing reference conditions for riparian condition, water quality, and TSI but several sites contributed "reference" conditions for the FCI. As for the reference value (i.e. highest score) for K, the decay constant obtained at LB was selected. It was neither the highest nor lowest K that constituted the references condition, but rather a value at the site with the best conditions (i.e. riparian, dissolved oxygen, trophic state) to maintain the least altered ecosystem processes such decomposition (Cole, 2002).

RHI scores resulted in similar classifications to what was expected at the beginning of this study for LB, BM, and Fort Brown. LB placed first during the fall (92.9) and winter (94.5) and Billy Mitchell scored highest during summer (91.8). The difference in ranking BM and LB based on RHI scores during the summer sampling was caused by LB recording the worst FCI values. However, LB appeared to have the best RHI on an annual basis (Table). LB is located on the campus of the University of Texas Rio Grande Valley Brownsville campus and has had limited impacts to water quality and riparian habitats. Based on historical maps and imagery it was determined that LB is the youngest resaca studied, having been part of the Rio Grande main channel in 1933. In contrast, FTB resaca, part of the same oxbow system as LB and located on the campus of Texas Southmost College recorded the lowest RHI score in all seasons. FTB, unlike LB has had extensive modification to the riparian habitat and receives large amounts of storm water runoff from downtown Brownsville during rain events. It should be noted that FTB has had a long record of anthropogenic impacts as land surrounding the resaca was occupied by a military fortification in the 1800's followed by the establishment of academic institutions found there today (Walton, 2011).

Dredged pools (DP and RB) scored lower than expected during all three seasons. The disturbance due to dredging and the unfinished riparian restoration likely caused these low scores. Over time, it is expected that these scores will increase and stabilize with a value near that of the high scoring resacas. VICC placed third in each season with scores of 78.1, 74.0, and 75.9 and is speculated that these scores most closely reflect the majority of the resacas found within Brownsville.

This evaluation of ecosystem health improves the understanding of novel ecosystems like resacas, and has implications on the potential management and ongoing restoration of the resacas. The RHI is an attempt to characterize the health of the resaca ecosystem using new baseline data. Over time, the determination and integration of other structural and functional metrics could better define and diagnose resaca ecosystem health. In any case, the validation of these metrics in additional resaca pools is necessary before recommending the use of the RHI for broad monitoring and assessment.

Other important aspects that were suggested by Rapport et al. (1998) to be examined and possibly integrated are based on human health, social value, and cultural views (Tzoulas et al., 2007; Rapport, 1989; Rapport et al., 1998). These aspects look at the ecosystem as not just important to the many species of fish, bird, reptiles, and amphibians that depend them for habitat, but also how they affect the people who use and benefit from the services they provide (Rapport et al., 1998). The understanding of how these views and values affect the system determines the desired endpoint of ecosystem health from a local perspective (Karr, 1999). The view of an endpoint however, can be influenced by awareness and education leading to changes over time in the importance placed on an ecosystem and health criteria. Currently several education and outreach programs such as "*I♥resacas"* and "*Resaca Rangers"* have been created in an effort to improve awareness and education.

Though the goal of the RHI was to be an assessment tool for urban resacas it has potential to be useful with agricultural or ephemeral (i.e. rain dependent) resacas. The effectiveness of the RHI in resacas outside the urban context however is unknown at this time, but the potential is clear if adequate reference conditions are determined. To accurately assess resacas different from urban ones, metric substitutions may need to be investigated such as using macroinvertebrate communities instead of fish in shallower resacas or aquatic vegetation communities in ephemeral wetlands.

CHAPTER V

CONCLUSION

Overall the development and testing of the Resaca Health Index showed to successfully discriminate between urban resaca pools with a priori different ecosystem status, based on the selected indicators. The RHI has the ability to be an integral part of the evaluation of the resaca restoration multiannual project; and has the potential to be used for assessment and monitoring of other resacas outside the urban context. Through the creation of the RHI it was established that no single indicator could accurately differentiate ecosystem health in these novel systems but the use of the multimetric approach increased the understanding ad evaluation accuracy. Each indicator studied is relevant either structurally or functionally, to the resaca aquatic ecosystem.

The used of structural indicators allowed for rapid assessment and gathering of baseline data on reasaca fish communities. Using the fish community allowed for the assessment of multiple trophic levels easily, with the ability to see long-term effects. Riparian habitat data was collected via drone technology and allows for not only present day assessment but the also the ability to monitor long term changes through images. Functional indicators allowed for an understanding of important ecosystem processes that are required to fully assess ecosystem health. The use of decomposition assays and primary productivity allowed again for baseline data to be collected, but also underlying factors such as nutrient or pollutant loading that influence resaca ecosystem function to be seen.

In conclusion the RHI is a pertinent and integrative index developed for assessing and monitoring a novel ecosystem, which serves regional importance to many. Through the knowledge gained it is now possible to evaluate resaca systems or pools in terms of ecosystem health in various states including healthy, restoring, or degrading.

REFERENCES

Aday, D.D., Joxmeier, R.J.H., and Wahl, D.H. (2003). Direct and Indirect Effects of Gizzard Shad on Bluegill Growth and Population Size Structure. *Transactions of the American Fisheries Society*, 132: 47-56.

Anbumozhi, V., Radhakrishnan, J., and Yamaji, E. (2005). Impact of Riparian Buffer Zones on Water Quality and Associated Management Considerations. *Ecological Engineering*, 24: 517-523.

Baptista, D.F., de Souza, R.S.G., Vieira, C.A., Mugnai, R., Souza, A.S., and de Oliveira, R.B.S. (2011). Multimetric Index for Assessing Ecological Condition of Running Waters in the Upper Reaches of the Piabanha-Paquequer-Preto Basin, Rio de Janeiro, Brazil. *Zoologia*, 28: 619-628.

Babour, M.T., Gerritsen, B.D., Snyder, B.D., and Stribling, J.B. (1999). Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macoinvertebrates, and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water, Washington, D.C.

Bennett, G.W. (1943). Management of Small Artificial Lakes. *Bull. Illinois Nat. Hist. Sur*v, 22: 357-376.

Blackburn, M., and Mazzacano C. (2012). Using Aquatic Macroinvertebrates as streamflow duration. Washington and Idaho Indicators. *The Xerces Society for Invertebrate Conservation*, 18.

Blocksom, K.A. (2003). A Performance Comparison of Metric Scoring Methods for a Multimetric Index for Mid-Atlantic Highlands Streams. *Environmental Management*, 31: 670-682.

Bonde, R.K., Aguirre, A.A., and Powell, J. (2004). Manatees as Sentinels of Marine Ecosystem Health: Are They the 2000-Pound Canaries?. *EcoHealth*, 1:255-262.

Bonin L., Proulx, R., and Rheault G. (2014). A Digitial Photography Protocol for the Rapid Assessment of Herbaceous Communities in Riparian Buffers. *Riparian Ecol. Conserv.*, 2: 25-44.

Boyer, T. and Polasky, S. (2004). Valuing Urban Wetlands: A Review of Non-market valuation studies. *Wetlands*, 24 744-755.

Brauns, M., Gucker, B., Wagner, C., Garcia, X.F., Walz, N., and Pusch, M.T. (2011). Human Lakeshore Development Alters the Structure and Trophic Basis of Littoral Food Webs. *Journal of Applied Ecology*, 48: 916-925.

Brinson, M. (1996) Assessing Wetland Functions Using HGM. *National Wetland's Newsletter*.

Brinson, M.M., Lugo, A.E., and Brown, S. (1981). Primary Productivity, Decomposition, and Consumer Activity in Freshwater Wetlands. *Annual Review of Ecology and Systematics*, 12: 123-161.

Brinson, M.M, and Rheinhardt, R. (1996). The Role of Reference Wetlands in Functional Assessment and Mitigation. *Ecological Applications*, 6, 69-76.

Brooks, R., McKenney-Easterling, M., Brinson, M., Rheinhardt, R., Havens, K., O'Brien, D., Bishop, J., Rubbo, J., Armstrong, B., and Hite, J. (2009). A Stream-Wetland-Riparian (SWR) Index for Assessing Condition of Aquatic Ecosystem in Small Watersheds along the Atlantic Slope of the Eastern U.S. Environ. *Monit. Assess*., 150: 101-117.

Burkes-Copes, K.A. and Webb, A.C. (2012). *A Community-based Ecosystem Response Model for the Resacas (Oxbow Lakes) of the Lower Rio Grande. U.S Army Corps of Engineers*. Environmental Laboratory Vicksburg, Mississippi. ERDC/EL TR-12-31.

Burks-Copes, K.A., Webb, A.C., Henderson, J. (2005). *City of Brownsville Resacas Ecosystem Restoration Study Cameron County, Texas Ecosystem Restoration Habitat Assessments Using HEP. ERDC/ITL SR-00-01*. US Army corps of Engineers Engineer Research and Development Center.

Burton, M.L., Samuelson, L.J., and Pan, S. (2005). Riparian Woody Plant Diversity and Forest Structure Along an Urban-Rural Gradient. *Urban Ecosystems*. 8: 93-106

Carlson, R. E. (1977). A trophic state index for lakes. *Limnol. Ocean,* 22, 361–369.

Carpenter, S.R. and Adams, M.S. 1979. Effects of Nutrients and Temperature on Decomposition of *Myriophyllum spicatum L.* in a hard-water Eutropic Lake. Limnol. *Oceanogr*., 24: 520-528.

Chadwick, M.A., Dobberfuhl, D.R., Benke, A.C., Hyryn, A.D., Suberkropp, K. and Thiele, J.E. (2006). Urbanization Affects Stream Ecosystem Function by Altering Hydrology, Chemistry, and Biotic Richness. *Ecological Applications*, 16: 1796-1807.

Clarke, K.R., Gorley, R.N. (2006) *PRIMER V6: user manual/tutorial*. PRIMER-E, ‘ Plymouth, p 192.

Clapcott, J.E., Collier, K.J., Death, R.G., Goodwin, E.O., Harding, J.S., Kelly, D., Leatherwick, J.R., and Young, R.G. (2012). Quantifying Relationships Between Land-use Gradients and Structural and Functional Indicators of Stream Ecological Intgerity. *Freshwater Biology*, 57: 74-90.

Cole, C.A. 2002. The Assessment of Herbaceous Plant Cover in Wetlands as an Indicator of Function. *Ecological Indicators*, 2: 287-293,

Colwell, S.R. and Hix, D.M. (2008) Adaption of the QBR Index for use in Riparian Forests of Central Ohio. *Proceedings of the 16th Central Hardwoods Forest Conference.*

Cornell, J.E., Gutierrez, M., Wait, D.A., and Rubio-Arias, H.O. (2008). Ecological Characterization of a Riparian Corridor along the Rio Conchos, Chihuahua, Mexico. *The Southwestern Naturalist*, 53: 96-100.

Costanza, R. (2012). *Ecosystem health and ecological engineering*. Institute of Sustainable Solutions Publications. Paper. 70.

Dale, V.H. and Beyeler, S.C. (2001). Challenges in the Development and use of Ecological Indicators. *Ecological Indicators*, 1: 3-10.

Dahm, C.N., Cleverly, J.R., Allred Coonrod, J.E., Thibault, J.R., McDonnell, D.E., and Gilroy, D.J. (2002). Evapotranspiration at the Land/water Interface in a Semi-arid Drainage Basin. *Freshwater Biology*, 47: 831-843.

Dahms, C.W.; Geils, B.W., tech. eds. (1997). *An assessment of forest ecosystem health In the Southwest. General Technical Report RM-GTR-295*. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station. 97 p.

Dudgeon, D. Prospects for Sustaining Freshwater Biodiversity in the 21st Century: Linking Ecosystem Structure and Function. *Environmental Sustainability*, 2: 422- 430.

Dutcher, D.D., Finley, J.C, Luloff, A.E., Johnson, J. (2004) Landowner Perceptions of Protectingand Establishing Riparian Forests: A Qualitative Analysis. *Society and NaturalResources*. 17: 329-342.

Edwards, E.A., Krieger, D.A., Gebhart, G. and Maughan, O.E. (1982). *Habitat Suitability.Index Models: White Crappie*. U.S.D.I. Fish and Wildlife Service. FWSOBS-82/10.7. 22. pp.

Edwards, R.J. and Contreras-Balderas. (1991). Historical Changes in the Icthyofauna of the Lower Rio Grande (Rio Bravo del Norte), Texas and Mexico. *The Southwestern Naturalist*, 36: 201-212.

Ehrenfeld, J.G. 2000. Evaluating Wetlands within an Urban Context. *Ecological Engineering*, 15: 253-265.

Emmanuel, B.E., Chukwu, L.O., and Azeez, L.O. 2008. Cast Net Design Characteristics, Catch Composition and Selectivity in Tropical Open Lagoon. *African Journal of Biotechnology*, 7: 2081-2089.

Eros, T., Specziar, A., and Biro, P. (2009). Assessing Fish Assemblages in Reed Habitats of a Larage Shallow Lake-A Comparison Between Gillnetting and Electric Fishing. *Fisheries Research*, 96: 70-76.

Erwrin, K.L. (2008). Wetlands and Global Climate Change: the Role of Wetland Restoration in a Changing World. *Wetlands Ecol. Manage*., 17: 71-84.

Esparza-Diaz, G., Armstrong, S., and Villanueva, R.T. (2011). Detection and Identification of Philephedra tuberculosa on Cotton in South Texas. *Southwestern Entomologist*, 36:4.

Fano, E.A., Mistri, M., and Rossi, R. (2003). The Ecofunctional Quality Index (EQI)" a New Tool for Assessing Lagoonal Ecosystem Impairment. *Estuarine, Coastal and Shelf Science*, 56: 709-716.

Fausch, K.D., Lyons, J., Karr, J.R., and Angermeier, P.L. 1990. Fish Communities as Indicators of Environmental Degradation. *American Fisheries Society Symposium*, 8: 123-144.

Fierro, A. Angers, D.A., Beauchamp, C.J. 2000. Decomposition of Paper De-inking Sludge in aSandpit Minesoil During its Revegetation. *Soil Biology and Biochemistry*, 32: 143-150.

Folk, R.L. (1974). *Petrology of sedimentary rocks*. Hemphill Publishing Company, Austin, TX.

Gerdes, J.H. and McConnell, W.M.J. (1963). Food Habits and Spawning of the Threadfin Shad in a Small, Desert Impoundment. *Journal of the Arizona Academy of Science*, 2: 113-116.

Gessner, M.O. and Chauvet, E. 2002. A Case for Using Litter Breakdown to Assess Functional Stream Integrity. *Ecological Applications*, 12: 498-510.

Gibson, G.R., Barbour, M.T., Stribling, J.B., Gerritsen, J. and Karr, J.R. (1996). *Biological Criteria Technical Guidance for Streams and Small Rivers (revised edition). EPA 822-B-96-001*. U.S. Environmental Protection Agency. Office of Water. Washington, DC.

Gibson, G.R., Bowman, M.L., Gerritsen, J., and Snyder, B.D. (2000). *Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance. EPA 822-B-00-024*. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Goetz, D. Kroger, R., and Miranda, L.E. (2014). Effects of Smallmouth Buffalo *Ictiobus bubalus* Biomass on Water Transparency, Nutrients, and Productivity in Shallow Experimental Ponds. *Bull Environ Contam Toxicol*., 92: 503-508.

Graves, G.A., Wan, Yongshan, and Fike., D.L. (2004). Water Quality Characteristics of Storm Water from Major Land Uses in South Florida. *Journal of the American Water Resources Association*. 40: 1405-1419.

Gregory, S.V., Swanson, F.J., McKee, W.A., and Cummins, K.W. (1991). An Ecosystem Perspective of Riparian Zones. *BioScience*, 41: 540-551.

Harberl, R., Grego, S., Langergraber, G., Kadlec, R.H., Cicalini, A., Dias, S.M., Novais, J.M., Subert, S., Gerth, A., Thomas, H. and Hebner, A. (2003). Constructed Wetlands for the Treatment of Organic Pollutants. *J. Soils & Sediments*, 3:109-124.

Hauer and Smith. (1998). The hydrogeomorphic approach to functional assessment of riparian wetlands: evaluating impacts and mitigation on river floodplains in the U.S.A. *Freshwater Biology*, 40: 517-530.

Hawes, E. and Smith M. (2005). *Riparian Buffer Zones: Functions and Recommended Widths*. Yale School of Forestry and Environmental Studies.

Hering, D., Feld, C.K., Moog, O., and Ofenbock, T. (2006). Cook Book for the Development of a Multimetric Index for Biological Condition of Aquatic Ecosystems: Experiences from European AQEM and STAR Projects and Related Initiatives. *Hydrobiologia*, 566: 311-324.

Hermoso, V. and Clavero, M. (2013). Revisiting Ecological Integrity 30 Years Later: Non-native Species and the Misdiagnosis of Freshwater Ecosystem Health. *Fishes and Fisheries*, 14: 416-423.

Hoover, J.J., Killgore, K.J., and Cofrancesco, A.F. (2004). Suckermouth Catfishes: Threats to Aquatic Ecosystems of the United States?. *ANSRP Bulletin*, 4: 1-13.

Hubert, W.A., Pope, K.L., and Dettmers, J.M. (2012). *Passive Capture Techniques*. Nebraska Cooperative Fish and Wildlife Reseach Unit. Staff Publications. Paper 111.

Hunter, W.C., Ohmart, R.D., and Anderson, B.W. (1988). Use of Exotic Saltcedar (Tamarix Chinensis) by Birds in Arid Riparian Systems. *The Condor*, 90: 113-123.

Jones, S. (2009). *Project Review Plan Resacas at Brownsville, Texas Ecosystem Restortaion Draft Feasibility Study*. U.S. Army Corps of Engineers Galveston Distract.

Karr, J.R. Defining and Mesuring River Health. (1999). *Freshwater Biology*, 41: 221-234.

Kayranli, B., Scholz, M., Mustafa, A., and Hedmark, A. (2010). Carbon Storage and Fluxes within Freshwater Wetland: a Critical Review. *Wetlands*, 30: 111-124.

Klapproth J.C., and Johnson, J.E. (2009). *Understanding the Science Behind Riparian Forest Buffer: Effects on Water Quality*. College of Agriculture and Life Sciences. Virginia Polytechnic Institute and Sate University. 420-151.

Larimore, R.W. (1961). Fish Population and Electrofishing Success in a Warm-Water Stream. *The Journal of Wildlife Management*. 25(1): 1-12.

Lee, K.H., Isenahrt, T.M., and Schultz, R.C. (2003). Sediment and Nutrient Removal an Established Mulit-Species Riparian Buffer. *Journal of Soil and Water Conservation*, 58: 1-8.

Linam, G. W., Kleinsasser, L.J., and Mayes, K.B. (2002). *Regionalization of the Index of Biotic Integrity for Texas Streams*. Resource Protection Division. Texas Parks and Wildlife Department, Austin, TX. River Studies Report No. 17.

Lougheed, V.L., Crosbie, B., and Chow-Fraser. (1998). Predictions on the Effects of Common Carp (*Cyprinus carpio*) exclusion on water quality, zooplankton, and submergent macrophytes in a Great Lakes Wetland. *Can. J. Fish. Aquat. Sci*., 55: 1189-1197.

Lu, K., Jin, C., Dong, S., Gu, B., and Bowen, S.H. (2006). Feeding and Control of Blue-Green Algal Blooms by Tilapia (*Oreochromis noloticus*). *Hydrobiologia*, 568: 111-120.

Mankin, K.R., Ngandu, D.M., Barden, C.J., Hutchinson, S.L., and Geyer, W.A. (2007). Grass-Shurb Riparian Buffer Removal of Sediment, Phosphorus, and Nitrogen From Simulated Runoff. *Journal of the American Water Resources Association*, 43: 1108-1116.

Marquez, M.A., Fierro-Cabo, A., Cintra-Buenrostro, C.E. (2016). Can ecosystem functional recovery be traced to decomposition and nitrogen dynamics in estuaries of the Lower Laguna Madre, Texas?. *Restoration Ecology*. In press

Maskell, L.C., Bullock, J.M., Smart, S.M., Thompson, K., and Hulme, P.E. The Distribution and Habitat Associations of Non-Native Plant Species in Urban Riparian Habitats. *Journa l of Vegetation Science*. 17: 499-508.

Mathis, M. and Matisoff, D. (2004). *A Characterization of Ecotourism in the Texas Lower Rio Grande Valley. VNT-04-01*. Houston Advanced Research Center.

McIntosh, L.M. (2014). Resaca ecosystem development: colonization and succession of the macroinvertebrate community (Masters Thesis). University of Texas Brownsville. Brownsville, Texas.

Meador, M.R. and Goldstein, R.M. (2003) Assessing Water Quality at Large Geographic Scales: Relations Among Land Use, Water Physicochemistry, Riparian Condition, and Fish Community Structure. *Environmental Management*, 31: 504-517.

Medeiros, E.S.F., Silva, M.J., Figueiredo, B.R.S., Ramos, T.P.A., and Ramos, R.T.C. (2008). Effects of Fishing Techniques on Assessing Species Composition in Aquatic Systems in Semi-arid Brazil. *Braz. J. Biol*., 2: 255-262.

Meyer, J.L., Paul, M.J., Taulbee, W.K. (2005). Stream Ecosystem Function in Urbanizing Landscapes. *J. N. Am. Benthol*., Soc. 24: 602-612.

Miller, R.R. (2011). *Systematics and Biology of the Gizzard Shad (Dorosoma Cepedianum) and Related Fishes. Fish and Wildlife Service*. U.S. Department of the Interior. Washington D.C. 60: 371-392.

Miranda, L.E. (1983). Average ichthyomass in Texas large impoundments. *Proc. Tex. Chap. Am. Fish*., Soc. 6: 58-67.

Mora, M.A., Papoulias, D., Nava, I., Buckler, D.R. (2001). A comparative assessment of contaminants in fish from four resacas of the Texas, USA-Tamaulipas, Mexico border region. *Environmental International*, 27:15-20.

Moulton, D.W., Dahl, T.E., and Dall, D.M. (1997). *Texas Coastal Wetlands Status and Trends, Mid 1950's to Early 1990s.* Fish and Wildlife Service, U.S. Department of the Interior. Albuquerque, New Mexcio.

Mundahl, N.D. and Wissing, T.E. (1987). Nutritional Importance of Detritivory in the Growth and Condition of Gizzard Shad in an Ohio Reservoir. *Environmental Biology of Fishes*, 20: 129-142.

Munne A., Prat, N., Sola, C., Bonada, N., and Rieradevall, M. (2003). A simple method for assessing the ecological quality of riparian habitat in river and streams:QBR index. *Aqatic Conserv: Mar. Freshw. Ecosyst*., 13: 147-163.

Naiman, R.J. and Decamps, H. (1997). The Ecology of Interfaces: Riparian Zones. *Annual Review of Ecology and Systematics*, 28: 621-658.

Nico, L.G. and Martin, R.T. (2001). The South American Suckermouth Armored Catfish. Pterygoplichthys anisitsi (Pisces:Loricaridae, in Texas, with Comments on Foreign Fish Introductions in the American Southwest. *The Southwestern Naturalist*, 46: 98-104.

O'Connell, M.T., Shepherd, T.D., O'Connell, A.M.U., and Myers, R.A. (2007). Long-Term Declines in Two Apex Predators, Bull Sharks (Carcharhinus leucas) and Alligator Gar (Atractosteus spatula), in Lake Pontchartrain, an Oligohaline Estuary in Southeastern Louisiana. *Estuaries and Coasts*, 30: 567-574.

Osborne, L.L. and Kovacic, D.A. (1993). Riparian Vegetated Buffer Strips in Water-Quality Restoration and Stream Management. *Freshwater Biology*, 29:243-258.

Petersen, R.C. and Cummins, K.W. (1974). Leaf Processing in a Woodland Stream. *Freshwater Biology*, 4: 343-368.

Pound, K.L., Nowlin, W.H., Huffman, D.G., and Bonner, T.H. (2009). Trophic Ecology of a Nonnative population of Suckermouth Catfish (Hypostomus plecostomus) in a Central Texas Spring-fed Stream. *Environ. Biol Fish*, 90: 277-285.

Qiu, D., Wu, Z., Liu, B., Jiaqi Deng, L., Fu, G., and He, F. (2001). The Restoration of Aquatic Macrophytes for Improving Water Quality in a Hypertrophic Shallow Lake in Hubei Province, China. *Ecological Engineering*, 18: 147-156.

Rapport, D.J. (1989). What Constitutes Ecosystem Health?. *Perspectives in Biology and Medicine*, 33, 120-132.

Rapport, D.J., Costanza, R., and McMichael, A. (1998). Assessing Ecosystem Health. *Trends Ecol. Evol.*, 13: 397-402.

Rapport, D.J., Regier, H., and Hutchinson, T. (1985). Ecosystem Behavior Under Stress. *Trends Ecol. Evol.*, 13: 397-402.

Reiss, K.C., (2006). Florida Wetland Condition Index for depressional forested wetlands. *Ecological Indicators*, 6:337-352.

Reynoldson, T.B., Norris, R.H., Resh, V.H., Day, K.E., and Rosenberg, D.M. (1997). The Reference Condition: A Comparison of Multimetric and Multivariate Approaches to Assess Water-Quality Impairmen Using Benthic Macroinvertebrates. *J.N. Am. Benthol. Soc.,* 16: 833-852.

Riipinen, M.P., Davy-Bowker, J., and Dobson, M. (2009). Comparison of Structural and Functional Stream Assessment Methods to Detect Changes in Riparian Vegetation and Water pH. *Freshwater Biology*, 54: 2127-2138.

Roach, W.J, Heffernan, J.B., Grimm, N.B., Arrowsmith, J.M., Eisinger, C., Rychener, T. (2008). Unintended Consequences of Urbanization for Aquatic Ecosystems: A Case Study from the Arizona Desert. *BioScience*, 58: 8 715-727.

Robinson, C.M. (2010). Resaca. Handbook of Texas Online.

Rombouts, I., Beaugrand, G., Artigas, L.F., Dauvin, J.C., Gevaert, F., Goberville, E., Kopp, D., Lefebvre, S., Spilmont, N., Travers-Trolet, M., Villanueva, M.C., and Kirby, R.R. (2013). Evaluating Marine Ecosystem Health: Case Studies of Indicators Using Direct Observations and Modeling Methods. *Ecological Indicators*, 24: 353-365.

Schaus, M.H., Vanni, M.J., Wissing, T.E., Bremigan, M.T., Garvey, J.E., and Stein, R.A. (1997). Nitrogen and Phosphorus Excretion by Detritivorous Gizzard Shad in a Reservoir Ecosystem. *Limnology and Oceanography*, 42: 1386-1397.

Schiemer, F. (2000). Fish as indicators for the assessment of the ecological integrity of large rivers. *Developments in Hydrobiology*, 149:271-278.

Seawright, E.K., Rister, M.E., Lacewell, R.D., McCorkle, D.A., Sturdivant, A.W., Yang, C., and Gooisby, J.A. (2009). Economic Implications for the Biological Control of *Arundo donoax*: Rio Grande Basin. *Southwestern Entomologist*, 34:4:377-394.

Siesennop, G.D. (1997). *Evaluation of Lake Trout Index Netting Methods in Ten Northeastern Minnesota Lakes*. Minnesota Department of Natural Resources. I Investigational Report. 461.

Sirombra, M.G. and Mesa, L.M. (2012). A Method for Assessing the Ecological Quality of Riparian Forests in Subtropical Andean Streams: QBRy Index. *Ecological Indicators*, 20: 324-331.

Strayer, D.L. and Findlay, S.E.G. (2010). Ecology of Freshwater Shore Zones. *Aquat. Sci.,* 72: 127-163.

Spencer, C., Robertson, A.I., and Curtis, A. (1998). Development and testing of a rapid appraisal wetland condition index in south-eastern Australia. *Journal of Environmental Management*, 54, 143-159.

Stanley, J.D. and Randazzo, G. (2001) Petrologic Database to Define the Human-Reworked, Sediment-Deficient Plain of the Rio Grande Delta, Texas. *Environmental Geology*. 41: 37-53.

Sun T., Lin, W., Chen, G, Guo, P., and Zeng, Y. (2016). Wetland ecosystem health assessment through integrating remote sensing and inventory data with an assessment model for the Hangzhou Bay China. *Science of the Total Environment*, 556-557: 627-640.

Thawait, A.K. and Chauhan, M.S. (2014). Riparian Vegetated Buffer Strips for Stream Water Quality Restoration. *Journal of Environmental Science and Sustainability*, 2:1-6.

Texas Parks and Wildlife. (2016). *TPWD Wildlife Diversity TXNDD Tracked Animals*. Online.

Tiner, R.W. (2000). An Overview of Wetland Identification and Delineation Techniques,With Recommendations for Improvement. *Wetland Journal*, 12: 15-22.

Tiner, R.W. (1984). *Wetlands of the United States: Current Status and Recent Trends*. Fish and Wildlife Service, U.S. Department of the Interior. Washington D.C.

Tzoulas, K., Kopela, K., Venn, S., Yli-Pelkonen, Kazmierczak, A., Niemela, J., and James, P. (2007). Promoting Ecosystem and Human Health in Urban Areas using Green Infrastructure: A Literature Review. *Landscape and Urban Planning*, 81: 167-178.

U.S. EPA. (2002). *Methods for Evaluating Wetland Condition: Introduction to Wetland Biological Assessment*. Office of Water, U.S. Environmental Protection Agency, Washington, DC. EPA-822-R-02-014.

U.S. EPA. (2000). *Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance*. Office of Water, U.S. Environmental Protection Agency. Washington D.C. EPA 822-B-00-024.

Uzarski, D.G., Burton, T.M., Cooper, M.J., Ingram, J.W., and Timmermans, S.T.A. (2005). Fish Habitat Use Within and Across Wetland Classes in Coastal Wetlands of the Five Great Lakes: Development of a Fish-based Index of Biotic Integrity. *J. Great lakes Res*., 31:171-187.

Van Dam, R.A., Camilleri, C., and Finlayson, C.M. (1998). The Potential of Rapid Assessment Techniques as Early Warning Indicators of Wetland Degradation: A Review. *Environmental Toxicology*, 13:4: 297-312.

Van Dolah, R.F., Hyland, J.L., Holland. A.F., Rosen. J.S., and Snoots, T.R. (1999). A benthic index of biological integrity for assessing habitat quality in estguaries of the southeastern USA. *Marine Environmental Research*, 48: 269-283.

Waldon, J. (2011). *Health & Human Performance Department History: 83 Years and Still Going Strong*. Health and Human Performance Department. University of Texas Rio Grande Valley.

Woodward, R.T. and Wui, Y. (2001). The Economic Value of Wetland Services: a Meta-analysis. *Ecological Economics*, 37: 257-270.

Xu, F., Yang, Z.F., Chen, B., and Zhao, Y.W. (2011). Ecosystem Health Assessment of the Plant-Dominated Baiyangdian Lake Based on Eco-Exergy. *Ecological Modeling*. 222: 201-209.

Young, R.G., Townsend C.R., Matthaei, C.D., (2004). *Functional indicators of river ecosystem health-an interim guide for use in New Zealand*. Cawthorn Report 870.

Young, R.G., Matthaei, C.D., and Townsend, C.R. (2008). Organic Matter Breakdown and Ecosystem Metabolism: Functional Indicators for Assessing River Ecosystem Health. *J.N. Am. Benthol. Soc.,* 27: 605-62.

APPENDIX

APPENDIX

TABLES

Table 1. Geographic coordinates of study sites in Brownsville, Tx. Coordinates are located at middle points of resaca pools.

Pool	Latitude	Longitude
Fort Brown (FTB)	25.897435	-97.493933
Lozano Banco (LB)	25.894972	-97.487461
Dean Porter (DP)	25.917118	-97.494763
Resaca Blvd (RB)	25.919079	-97.507282
Billy Mitchell (BM)	25.914348	-97.457415
VICC (VICC)	25.942219	-97.512141

Table 2. Trophic state classifications based on Carlson's Trophic Index (Carlson, 1977).

Secchi Depth (M)	Chlorophyll α	Trophic Index	Trophic Classification
4->8	0-2.6	<30-40	Oligotrophic
2-4	2.6-20	40-50	Mesotrophic
0.5-2	20-56	50-70	Eutrophic
<0.25-0.5	56-+155	70-+100	Hypereutrophic

Table 3. Criteria for the Riparian Index (RI) used for resaca riparian assessment.

Total Vegetation Cover	
Points	**Criteria**
40	>80% cover
20	50-80% cover
10	10-50% cover
0	< 10% cover or less than 3m width
	Adjustment criteria
+10	If vegetation is dominated by native species (>50%)
-5	If more than 20% of non-vegetated surface is impervious

Vegetation Structure	
Points	**Criteria**
30	>75% woody vegetation (trees and shrubs)
15	45-75% woody vegetation (trees and shrubs)
5	10-45% woody vegetation (trees and shrubs)
0	<10% woody vegetation (trees and shrubs)
	Adjustment criteria
-10	If bare ground or mowed herb beneath the canopy (>50%)

Channel alteration	
Points	**Criteria**
30	Unmodified channel
15	>50% margins without rigid structure
5	20-50% margins without rigid structure
0	<20 margins without rigid structures
	Adjustment criteria
-5	If width-constraining structures exist

Table 4. Criteria for classifying riparian quality based on RI scores. Adapted from Munne et al., 2003.

Riparian Habitat Class	RI score
Riparian habitat in natural condition, excellent quality	≥95
Some disturbance, good quality	75-90
Disturbance important, fair quality	55-70
Strong alteration, poor quality	30-50
Extreme degradation, very poor quality	≤25

Table 5. Mean annual water parameters (± SE) based on monthly sampling in six resaca pools from September 2015 to August 2016 (N=48).

Pool	Temperature (°C)	Conductivity (us/cm)	Dissolved Oxygen (mg/L)	pH	Secchi Depth (cm)	Total Depth (cm)
FB	26.2±0.7 (17.6-33.9)	589±23 (354-981)	9.6±0.3 (4.4-12.1)	9.7±0.04 (8.5-9.7)	20±0.8 (10-30)	86±4.7 (30-170)
LB	26.6±0.7 (17.2-33.7)	775±21 (553-1004)	11.58±2.66 (3.4-14.3)	8.6 ±0.04 (8.1-9.2)	53±1.2 (30-75)	177±1.9 (120-195)
DP	26.2±0.7 (17.9-32.7)	3258±218 (818-6580)	11.2±0.5 (45.0-21.0)	8.6 ±0.08 (8.2-9.1)	28±1.4 (15-45)	162±4.6 (65-215)
RB	26.7±0.6 (18.9-32.8)	1509±65 (732-2260)	9.7±0.4 (3.8-14.8)	8.9 ±0.04 (8.5-9.6)	15±0.6 (10-25)	52±3.2 (20-120)
BM	26.1±0.6 (17.4-32.5)	1469±49 (830-2188)	8.3±0.3 (4.9-11.5)	8.6±0.04 (7.6-9.1)	35±1.1 (25-55)	163±3.6 (110-215)
VICC	26.27±0.6 (18.2-32.8)	1392±47 (472-1839)	8.7±0.4 (2.5-13.3)	8.6±0.04 (7.7-9.1)	24±0.9 (15-40)	81±3.5 (30-130)

Table 6. Seasonal Carlson's Trophic State index in six resaca pools derived from secchi depths from September 2015 to August 2016.

Pool	Season	Carlson's TSI	Trophic Classification
FTB	Fall	85.6	Hypereutrophic
	Winter	82.4	Hypereutrophic
	Summer	83.3	Hypereutrophic
LB	Fall	69.6	Eutrophic
	Winter	71.5	Hypereutrophic
	Summer	67.9	Eutrophic
RB	Fall	86.2	Hypereutrophic
	Winter	92.7	Hypereutrophic
	Summer	85.0	Hypereutrophic
DP	Fall	74.8	Hypereutrophic
	Winter	84.5	Hypereutrophic
	Summer	77.7	Hypereutrophic
VICC	Fall	83.7	Hypereutrophic
	Winter	83.5	Hypereutrophic
	Summer	79.9	Hypereutrophic
BM	Fall	75.9	Hypereutrophic
	Winter	75.6	Hypereutrophic
	Summer	74.2	Hypereutrophic

Table 7. Principal Component Analysis results of monthly water parameters recorded from September 2015 to August 2016. Expressed as (A) Eigenvalues and (B) Eigenvectors. Principal Component (PC); Cumulative percent variation (Cum.% Variation).

(A)

PC	Eigenvalues	%Variation	Cum.%Variation
1	1.99	33.2	33.2
2	1.51	25.2	58.4
3	1.19	19.8	78.2
4	0.831	13.9	92.1
5	0.279	4.6	96.7

(B)

Variable	PC1	PC2	PC3	PC4	PC5
Temperature (C)	-0.253	0.47	-0.053	0.785	0.215
Conductivity (us/cm)	0.09	-0.273	0.773	0.39	-0.126
Dissolved Oxygen (mg/L)	0.423	-0.563	-0.091	0.212	0.58
pH	0.469	-0.1	-0.528	0.397	-0.484
Secchi Depth (cm)	-0.548	-0.329	-0.332	0.072	0.348
Total Depth (cm)	-0.479	-0.519	-0.043	0.153	-0.496

Table 8. Seasonal and total fish abundances by species from the six resaca pools.

	FTB			LB			DP			RB			BM			VICC			Species totals
Genus species	F	W	S	F	W	S	F	W	S	F	W	S	F	W	S	F	W	S	
Promoxis annularis													2			4		1	7
Ictalurus punctatus					1								2					2	5
Micropterus salmoides								1	1				1	1	3	1			8
Lepisosteus oculatus				3	1		1		2					4	2	7	14		34
Aplodinotus grunniens														1			2		3
Lepomis macrochirus					1	4			3	1		2	1	3	3	2	1	3	24
Herichthys cyanoguttatus			1					1	2	1					1				6
Dorosoma cepedianum	9	15	10	5	31	19	18	31	28	20	24	20	10	4	7	19	22	14	306
Dorosoma petenense	5	1	1	6	12	17	35	12	2	8	3	4	1	3	2	4			116
Pterygoplichtys disjunctivus	1		4	3	2	3	21	21			1	5	3	2	1	7	4	7	85
Oreochromis aurea	2	2	5	7		12	9	7	3	4	2	11				4	3	4	75
Ictiobus bubalus	1	8	2		1		2		2	1	4	5	1		3	12	26	12	80
Cyprinus carpio				1				1		7		1		1	1			1	13
Ctenopharyngodon idella							1												1
Gobiosoma bosc												1							1
Menidia beryllina							1												1
Poecilia formosa												1							1
Astyanax mexicanus					2														2
Monthly totals	18	26	23	25	51	55	88	74	43	42	34	50	21	19	23	60	72	44	768
Pool totals	67			131			205			126			63			176			

Table 9. Fish species found in each of the six resaca pools.

Fort Brown	Lozano Banco	Resaca Blvd	Dean Porter	VICC	Billy Mitchell
Dorosoma cepedianum	*Dorosoma cepedianum*	*Dorosoma cepedianum*	*Dorosoma cepedianum*	*Dorosoma cepedianum*	*Dorosoma cepedianum*
Dorosoma petenense	*Dorosoma petenense*	*Dorosoma petenense*	*Dorosoma petenense*	*Dorosoma petenense*	*Dorosoma petenense*
Pterygoplichtys disjunctivus	*Pterygoplichtys disjunctivus*	*Pterygoplichtys disjunctivus*	*Pterygoplichtys disjunctivus.*	*Pterygoplichtys disjunctivus.*	*Pterygoplichtys disjunctivus*
Ictiobus bubalus	*Ictiobus bubalus*	*Ictiobus bubalus*	*Ictiobus bubalus*	*Ictiobus bubalus*	*Ictiobus bubalus*
Oreochromis aurea	*Oreochromis aurea*	*Oreochromis aurea*	*Oreochromis aurea*	*Oreochromis aurea*	
	Lepisosteus oculatus		*Lepisosteus oculatus*	*Lepisosteus oculatus*	*Lepisosteus oculatus*
	Lepomis spp.	*Lepomis spp.*	*Lepomis spp.*	*Lepomis spp.*	*Lepomis spp.*
Herichthys cyanoguttatus		*Herichthys cyanoguttatus*	*Herichthys cyanoguttatus*		*Herichthys cyanoguttatus*
			Micropterus salmoides	*Micropterus salmoides*	*Micropterus salmoides*
	Cyprinus carpio	*Cyprinus carpio*	*Cyprinus carpio*	*Cyprinus carpio*	*Cyprinus carpio*
				Promoxis annularis	*Promoxis annularis*
	Ictalurus punctatus			*Ictalurus punctatus*	*Ictalurus punctatus*
				Aplodinotus grunniens	*Aplodinotus grunniens*
			Ctenopharyngodon idella		
		Gobiosoma bosc			
			Menidia beryllina		
		Poecilia formosa			
	Astyanax mexicanus				

Table 10. Species richness, number of piscivourous species, and dominance ratio by site and season.

Pool	Season	Richness	Piscivorous	Dominance
FTB	Fall	5	0	0.50
	Winter	4	0	0.58
	Summer	6	0	0.43
LB	Fall	6	1	0.28
	Winter	8	2	0.61
	Summer	5	0	0.35
RB	Fall	7	0	0.48
	Winter	5	0	0.71
	Summer	9	0	0.40
DP	Fall	8	1	0.40
	Winter	7	1	0.42
	Summer	8	2	0.65
RB	Fall	9	3	0.32
	Winter	7	1	0.36
	Summer	8	2	0.32
BM	Fall	8	3	0.48
	Winter	9	2	0.21
	Summer	9	2	0.30

Table 11. Species evenness indicies: Shannon-Wiener, Simpson, Pielou, and Margalef.

Site	Season	Shannon	Pielou's	Margalef's	Simpson
FTB	Fall	1.27	0.79	1.38	2.89
	Winter	0.92	0.66	0.92	1.10
	Summer	1.48	0.83	1.59	3.60
LB	Fall	1.66	0.93	1.55	4.84
	Winter	1.21	0.58	1.78	2.33
	Summer	1.41	0.88	1.00	3.69
RB	Fall	1.46	0.75	1.61	3.32
	Winter	0.98	0.61	1.13	1.91
	Summer	1.73	0.79	2.04	4.21
DP	Fall	1.51	0.72	1.56	3.73
	Winter	1.41	0.73	1.39	3.43
	Summer	1.08	0.52	1.86	6.69
VICC	Fall	1.91	0.87	1.95	5.49
	Winter	1.50	0.77	1.40	3.74
	Summer	1.72	0.83	1.85	4.61
BM	Fall	1.66	0.80	2.30	3.64
	Winter	1.94	0.93	2.38	6.33
	Summer	1.99	0.91	2.55	6.08

Table 12. Analysis of similarity (ANOSIM) results based on communities and abundances.

Global Test

Sample statistic (Global R): 0.288

Significance level of sample statistic: 0.0001

Groups	R Statistic	Significance
Fort Brown, Lozano Banco	0.306	0.024
Fort Brown, Dean Porter	0.398	0.015
Fort Brown, Resaca Blvd	0.113	0.134
Fort Brown, Billy Mitchell	0.444	0.004
Fort Brown, VICC	0.452	0.002
Lozano Banco, Dean Porter	-0.081	0.775
Lozano Banco, Resaca Blvd	0.172	0.08
Lozano Banco, Billy Mitchell	0.163	0.126
Lozano Banco, VICC	0.309	0.026
Dean Porter, Resaca Blvd	0.144	0.128
Dean Porter, Billy Mitchell	0.552	0.002
Dean Porter, VICC	0.313	0.024
Resaca Blvd, Billy Mitchell	0.493	0.002
Resaca Blvd, VICC	0.361	0.004
Billy Mitchell, VICC	0.307	0.022

Table 13. Similarity percentage analysis (SIMPER) results based on community abundances. Average similarity reports similarity of samples within each pools. All other values report percent contribution of individual species to reported sample similarities within pools.

Species	FTB	LB	RB	DP	BM	VICC
Average Similarity %	50.6	35.7	55.5	53.0	41.7	42.7
Dorosoma cepedianum	64.3	42.7	74.8	60.4	38.9	55.2
Dorosom. petenense		25.4	7.8	15.8		10.2
Oreochromis aurea	13.1	14.2				
Pterygoplichtys disjunctivus					16.3	9.8
Ictiobus bubalus					22.3	

Table 14. Riparian Index (RI) scores for the six resaca pools based on criteria specified in Table 2.

Site	Vegetation Cover	Vegetation Structure	Channel	Score
Fort Brown	28	6	4	39
Lozano Banco	46	28	26	99
Resaca Blvd	39	20	15	73
Dean Porter	28	4	6	38
VICC	31	2	15	49
Billy Mitchell	41	21	15	77

Table 15. Seasonal decay constants (K) for the six resaca pools.

Pool	Fall K	Winter K	Summer K
Fort Brown	0.0098	0.0079	0.0098
Lozano Banco	0.0087	0.0066	0.0123
Resaca Blvd	0.0078	0.0042	0.0084
Dean Porter	0.0120	0.0045	0.0204
VICC	0.0114	0.0073	0.0186
Billy Mitchell	0.0116	0.0078	0.0120

Table 16. Normalized fish community index (FCI) scores. fall (A); winter (B); summer (C).

(A)

Pool	Shannon	Dominance	Richness	Piscivorous	FCI
FTB	4.15	3.50	3.47	0.00	11.12
LB	5.43	6.25	4.17	2.08	17.93
DP	4.92	4.40	5.56	2.08	16.96
RB	4.77	3.68	4.86	0.00	13.31
BM	5.43	3.67	5.56	6.25	20.91
VICC	6.25	5.53	6.25	6.25	24.28

(B)

Pool	Shannon	Dominance	Richness	Piscivorous	FCI
FTB	2.95	2.28	2.78	0.00	8.01
LB	3.88	2.16	5.56	6.25	17.85
DP	4.55	3.14	4.86	3.13	15.68
RB	3.16	1.86	3.47	0.00	8.50
BM	6.25	6.25	6.25	6.25	25.00
VICC	4.83	3.64	4.86	3.13	16.46

(C)

Pool	Shannon	Dominance	Richness	Piscivorous	FCI
FTB	4.65	4.38	4.17	0.00	13.19
LB	4.43	5.51	3.47	0.00	13.41
DP	3.38	2.92	5.56	6.25	18.10
RB	5.41	4.76	6.25	0.00	16.42
BM	6.25	6.25	6.25	6.25	25.00
VICC	5.41	5.98	5.56	6.25	23.19

Table 17. Resaca Health Index scores and normalized values of its components for six resaca pools. fall (A); winter (B); summer (C).

(A)

Fall					
Site	RI	TSI	K	FCI	RHI
LB	25.0	25.0	25.0	17.9	92.9
BM	19.4	23.1	20.0	21.7	84.1
VICC	12.2	21.4	20.1	24.4	78.1
RB	18.4	21.0	22.4	14.1	75.9
DP	9.7	23.4	19.6	17.4	70.0
FTB	9.7	21.1	22.3	12.0	65.1

(B)

Winter					
Site	RI	TSI	K	FCI	RHI
LB	25.0	25.0	25.0	19.5	94.5
BM	19.4	23.7	21.6	25.0	89.7
VICC	12.2	21.9	22.7	17.2	74.0
DP	9.7	21.7	17.2	16.7	65.2
RB	18.4	20.4	16.0	10.3	65.1
FTB	9.7	22.1	21.4	9.6	62.8

(C)

Summer					
Site	RI	TSI	K	FCI	RHI
BM	19.4	23.0	24.4	25.0	91.8
LB	25.0	25.0	25.0	13.5	88.5
VICC	12.2	21.7	18.7	23.2	75.9
RB	18.4	20.8	17.1	16.7	73.0
DP	9.7	22.2	17.9	19.3	69.0
FTB	9.7	21.2	19.8	13.6	64.3

FIGURES

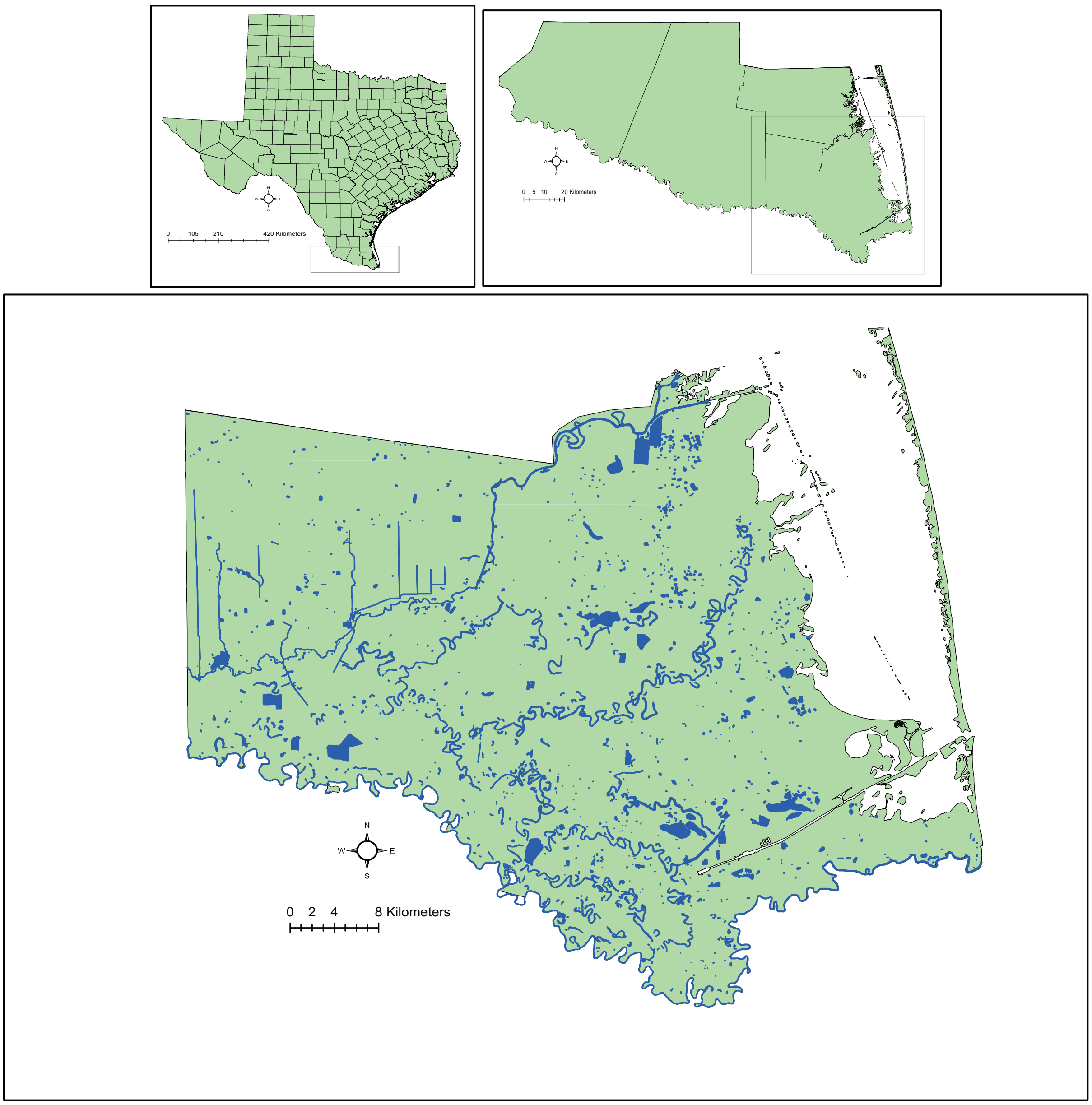

Figure 1. Freshwater found within Cameron County, TX including the Arroyo Colorado, five distributary systems, numerous oxbows, and the Rio Grande.

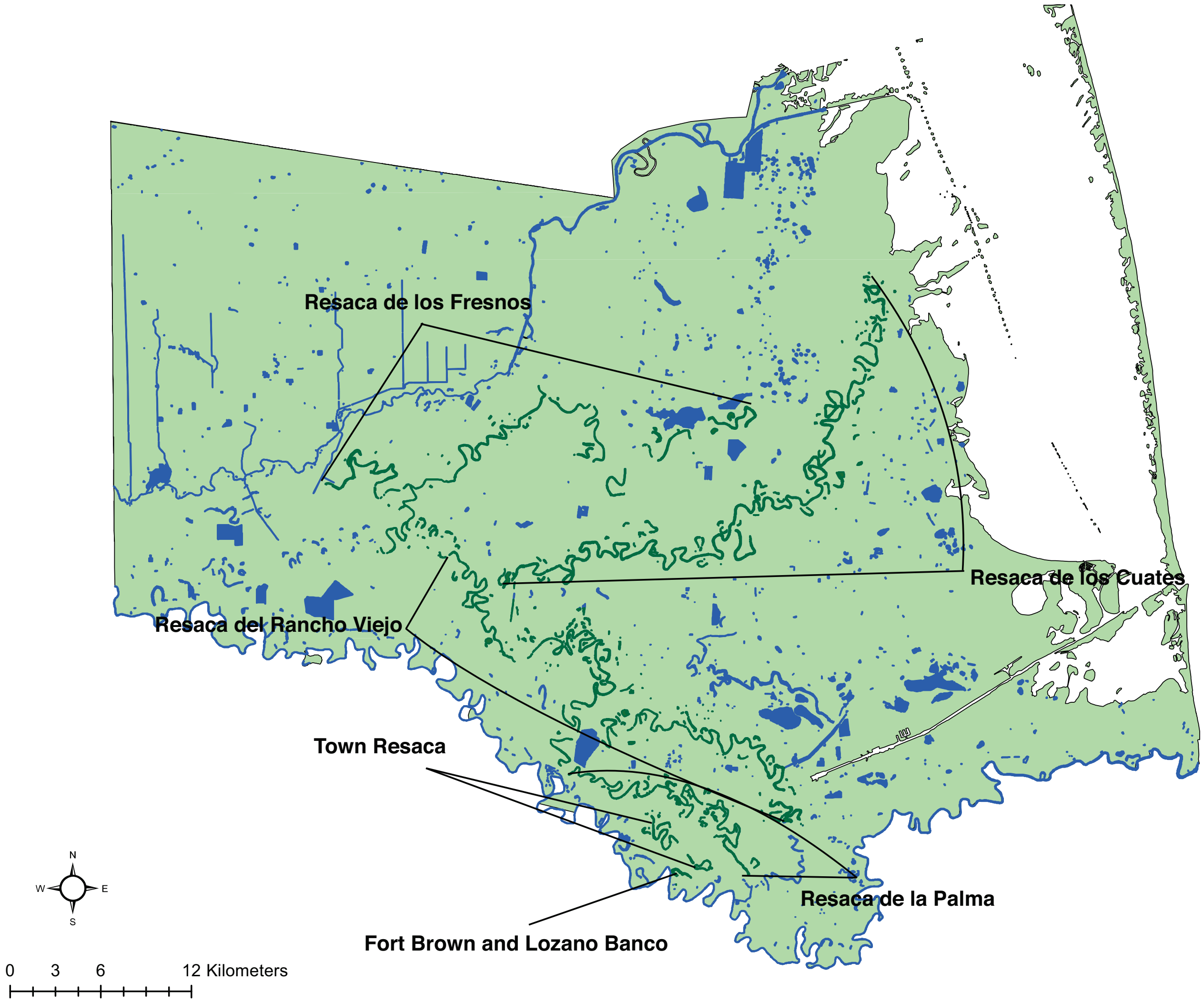

Figure 2. Resaca systems within Cameron County, systems used for this study included Resaca de la Palma, Town Resaca and Fort Brown and Locano Banco.

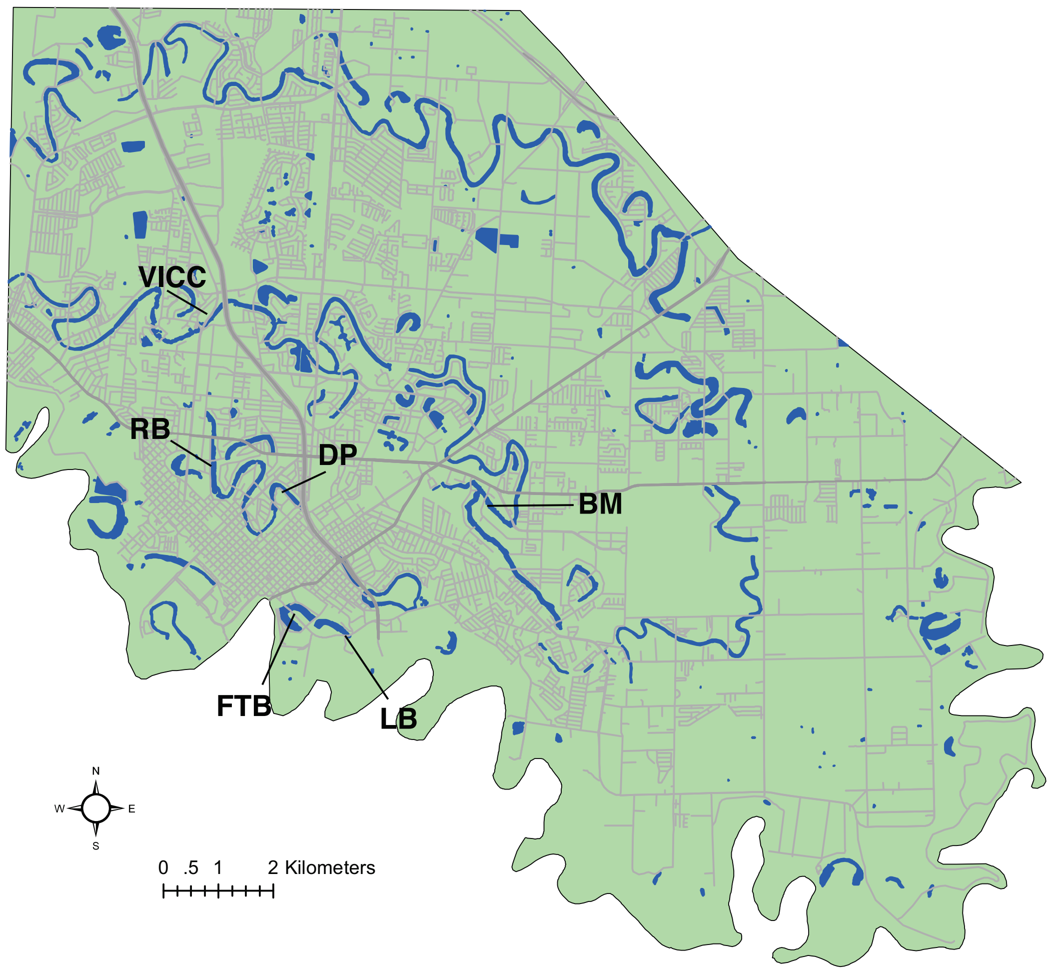

Figure 3. Sample pools located in Brownsville, TX. VICC-Valley International Country Club; BM- Billy Mitchell; RB- Resaca Blvd; DP- Dean Porter; FTB- Fort Brown; LB- Lozano Banco.

Figure 4. Drone Imagery of Resaca Blvd.

Figure 5. Drone Imagery of Dean Porter.

Figure 6. Drone Imagery for VICC.

Figure 7. Drone Imagery for Billy Mitchell.

Figure 8. Drone Imagery for Fort Brown.

Figure 9. Drone Imagery for Lozano Banco.

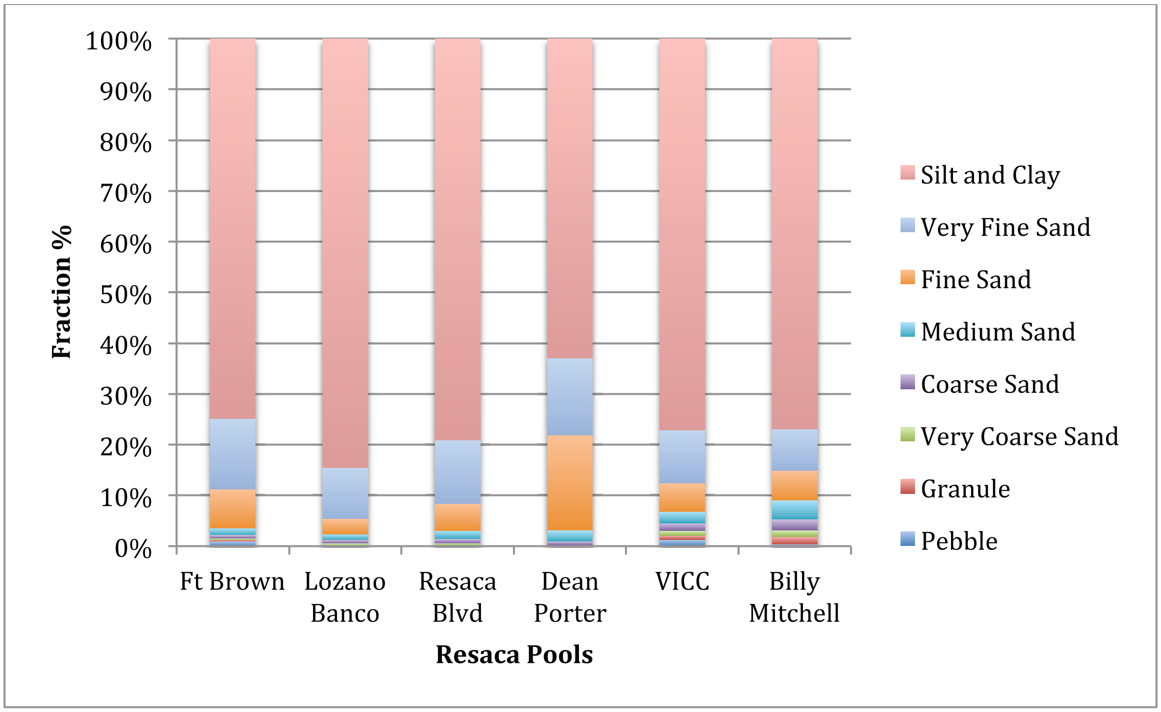

Figure 10. Mean sediment particle size (% dry weight) found via stratified sampling in the six sample pools. Samples collected in August 2016, Brownsville, Texas.

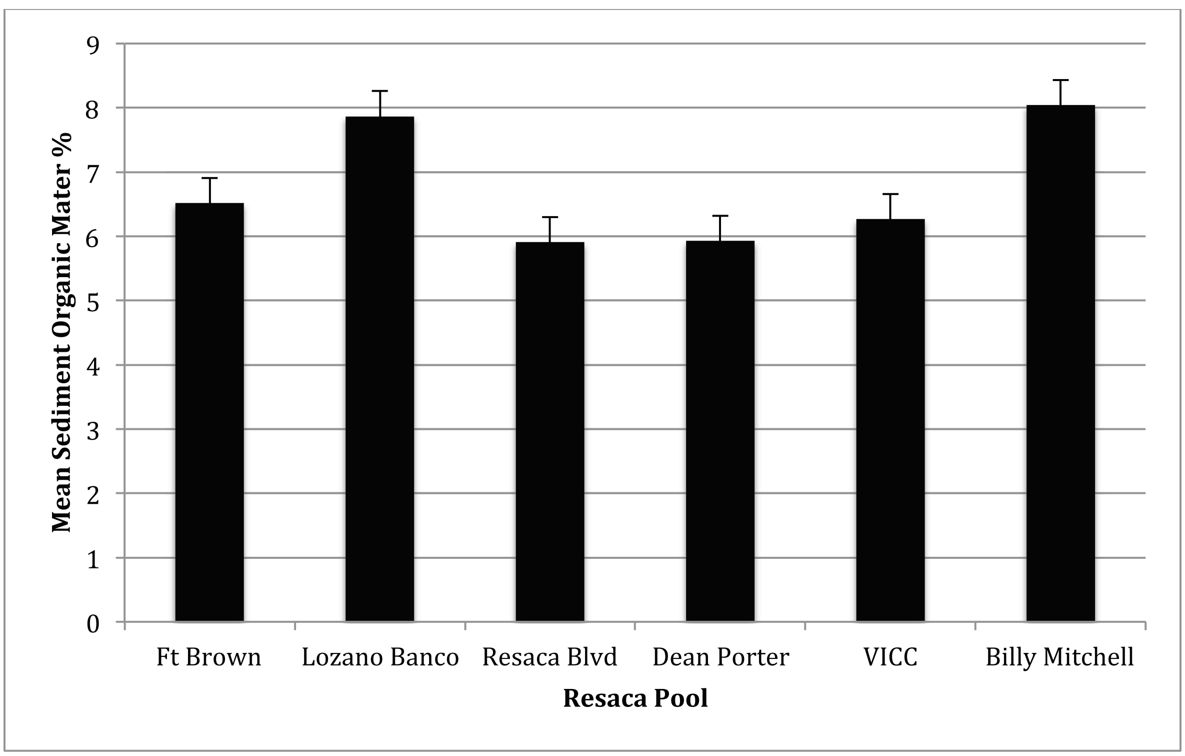

Figure 11. Mean sediment organic matter content from stratified sampling in the six sample pools. Samples collected in August 2016, Brownsville, Texas.

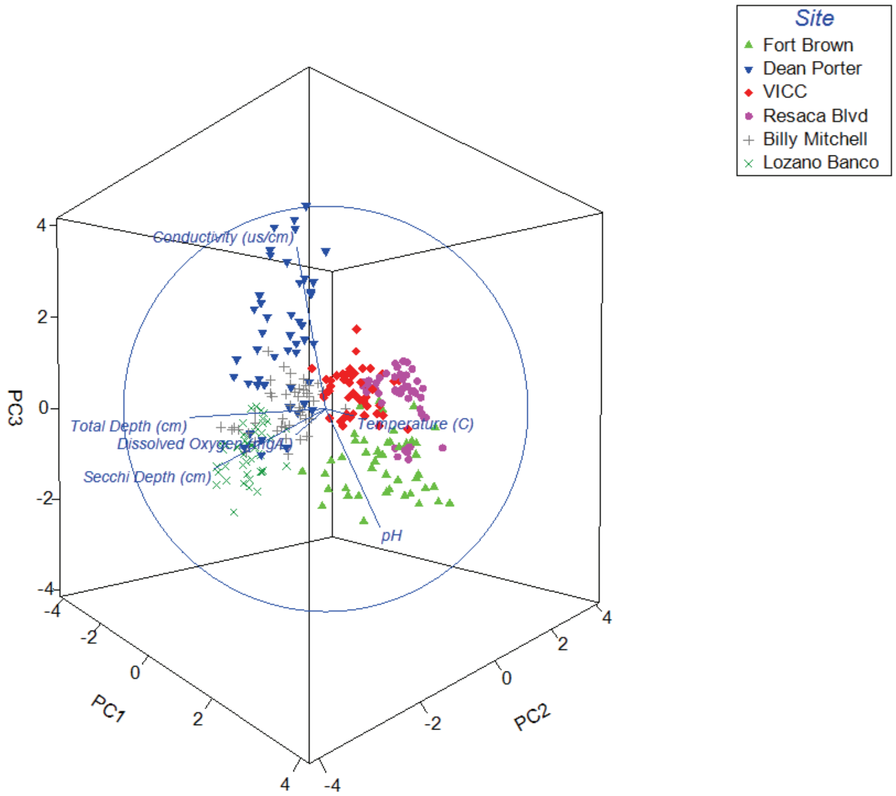

Figure 12. Principle Component of monthly environmental variables. Three principal components were required to account for 75% of the variation. Samples collected between September 2015 and August 2016, Brownsville, Texas.

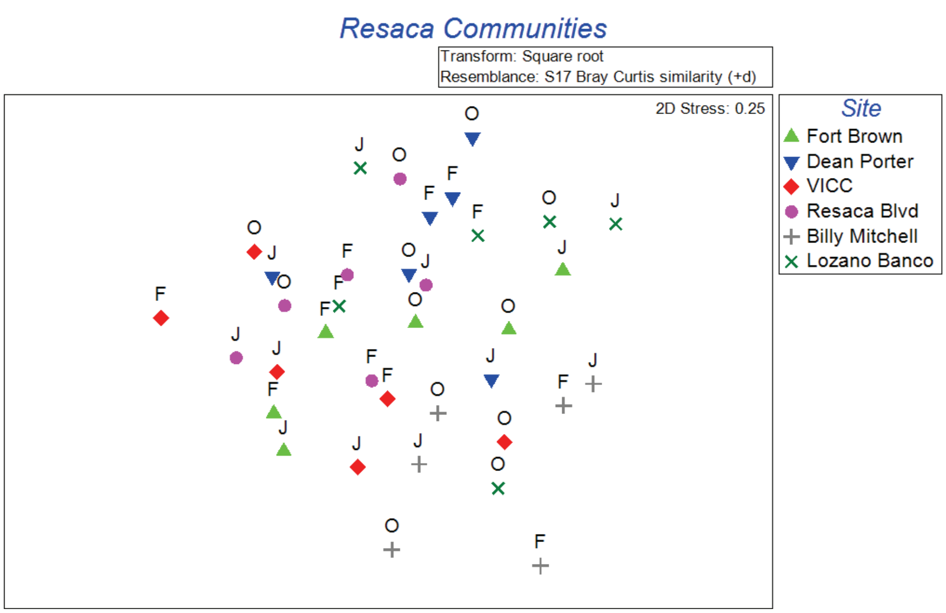

Figure 13. Multi-dimensional scaling (MDS) of seasonally sampling events (√ transformed) for fish communities of the six sample pools. Samples collected during October 2015 (fall), February 2016 (winter), and June 2016 (summer).

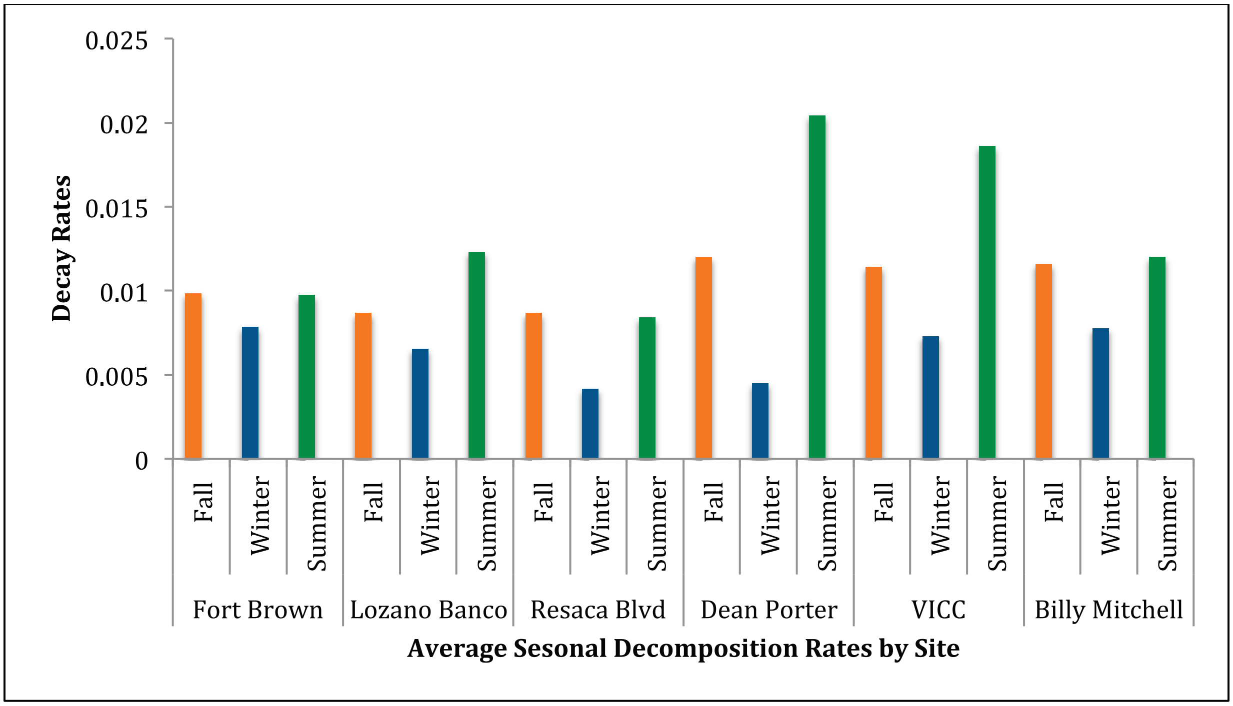

Figure 14. Average monthly decomposition rates by pool.

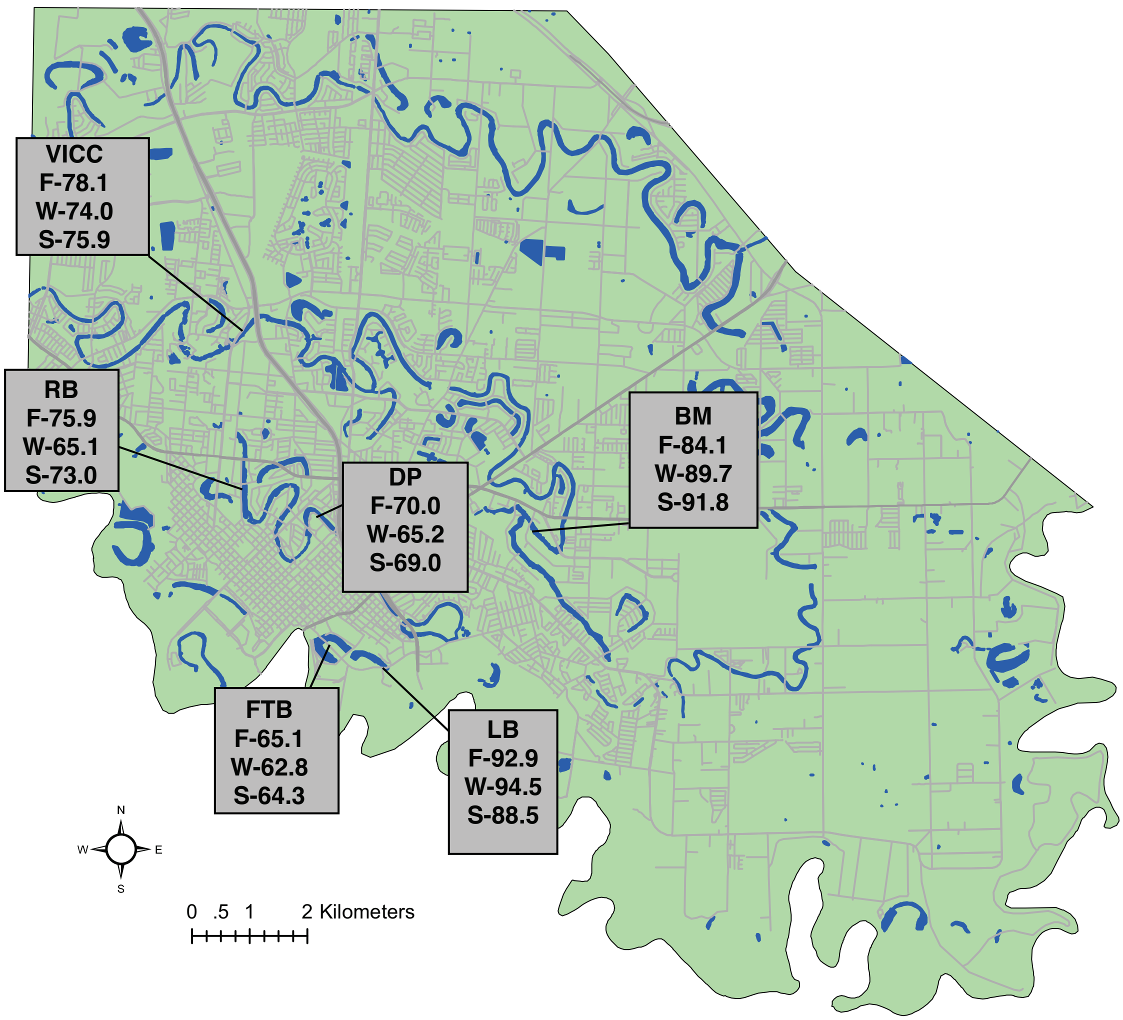

Figure 15. Resaca Health Index scores based on resaca pool location.

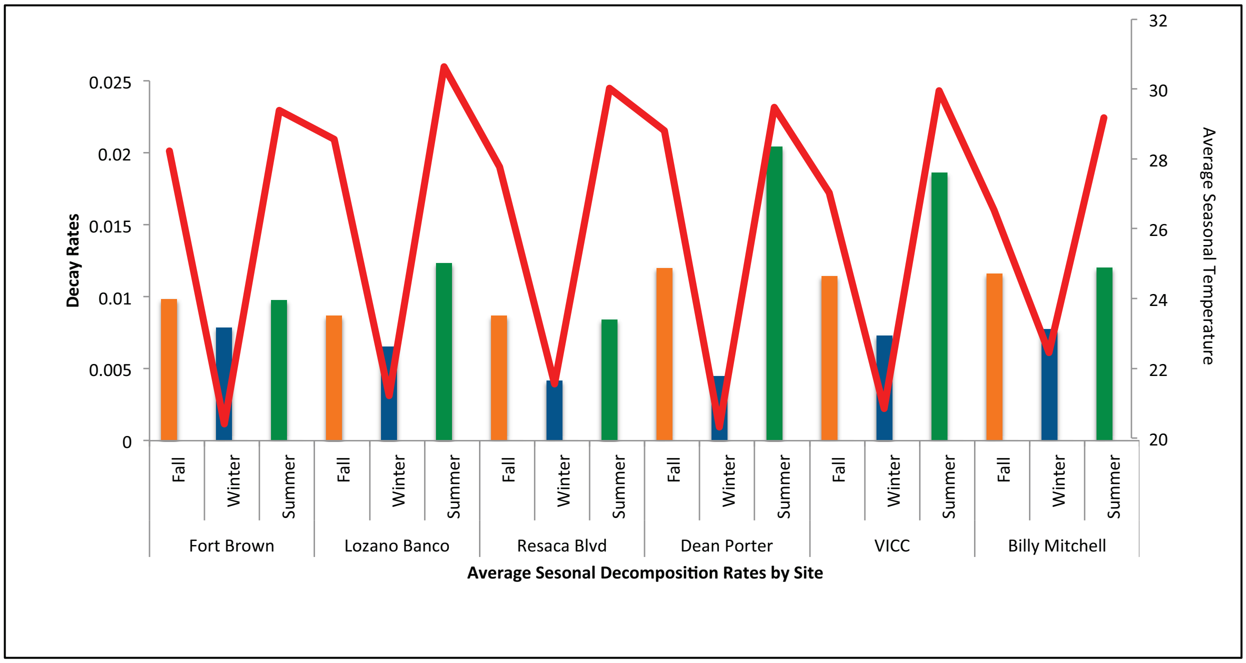

Figure 16. Average seasonal decomposition rates compared to average seasonal temperatures.

BIOGRAPHICAL SKETCH

Buford J. Lessley earned his B.S. in wildlife, fisheries, and aquaculture with a concentration in wildlife-agriculture conservation from Mississippi State University in 2010. After his graduation from the University of Texas Rio Grande Valley in 2016 with a M.S. in biology Buford plans to work as a wildlife and fisheries biologist. He can be contacted at buford.lessley@gmail.com.