MACROINVERTEBRATE ABUNDANCE AND HABITAT ASSOCIATIONS IN THE BIG BEND REGION OF THE RIO GRANDE WITH COMMENTS ON THE LIFE HISTORY OF CORYDALUS CORNUTUS

Presented to the Graduate Council of Texas State University-San Marcos in Partial Fulfillment of the Requirements

for the Degree

Master of SCIENCE

by

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San Marcos, Texas May 2008

ACKNOWLEDGEMENTS

Special thanks to the Rio Grande Crew, Tom Heard, Megan Bean, Preston Bean, Josh Perkin, Dennis Runyan, and Kristin Morrison, for their constant pursuit of invertebrate samples. I would also like to thank my committee members, Tim Bonner, Romi Burks, and Weston Nowlin for their ongoing help throughout the research process. Of course I am thankful for the love and support from my fiancé, Andrew, my mom, Stephanie, my dad, Kurt, and my sister, Jessica.

This manuscript was submitted on December 9, 2007.

ABSTRACT

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Arid aquatic environments represent unique and ecologically distinct systems, ranging from highly stable to highly fluctuating water bodies/streams along expansive discharge, water temperature, and salinity gradients. These environments typically support a diverse macroinvertebrate assemblage with a number of endemic taxa. The primary objective of this study included quantifying monthly occurrence and abundance, longitudinal distribution, and habitat associations of the macroinvertebrate community in the Rio Grande, located in northern Chihuahuan Desert. Leptophlebiidae (Order: Ephemeroptera) comprised the most abundant family (21% in relative abundance), followed by Cheumatopsyche (Order: Trichoptera; 14%) and Simulidae (Order: Diptera; 7%). Macroinvertebrate assemblage changed along a downstream gradient, suggesting that upstream pollution inputs favor dipteran taxa at sites 1 and 2, although generalist species occurred at all sites. As water quality improved longitudinally, downstream assemblages shifted to favor ephemeropteran and trichopteran taxa. As a secondary objective, I studied feeding habits and life history information for a top invertebrate predator, *Corydalus cornutus*, in the Rio Grande and its tributaries to compare feeding and fecundity in the Rio Grande and Devils River. I found no difference in egg number and number of hatched eggs between sites, although lower numbers occurred at the site farthest downstream (San Ygnacio; $68 \pm 26\%$) than at both Dolan (91 \pm 9%) and Kickapoo ($84 \pm 17\%$). Food selection consisted primarily of detritus, followed by Cheumatopsyche and Simulidae. This study will aid biomonitoring efforts by establishing an index of change in assemblage structure.

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CHAPTER I

MACROINVERTEBRATE ABUNDANCE AND HABITAT ASSOCIATIONS IN THE BIG BEND REGION OF THE RIO GRANDE

Introduction

Arid aquatic environments represent unique and ecologically distinct systems ranging from highly stable ones (i.e., endorheic springs) to highly fluctuating (i.e., streams and rivers) along expansive discharge, water temperature, and salinity gradients (Fisher and Gray 1983; Herbst and Bromley 1984; Castleberry and Cech 1986; Stanley et al. 1994; Watson 2006). Wide ranging and fluctuating environmental conditions along with relatively few interconnections within and among drainages collectively influence the speciation of arid aquatic organisms, many of which are endemic to only small geographic regions (Stanley 1994; Poff et al. 1997; Richter et al. 2003; Fritz and Dodds 2005). Unfortunately, these habitats exhibit high susceptibility to anthropogenic perturbation. Dams, excessive surface and groundwater withdrawals, and point and nonpoint source pollution alter a suite of habitat characteristics, including flow regime, channel morphology, sediment transport, substrate components, nutrient availability, and riparian vegetation (Brown and Ford 2002; Bunn and Arthington 2002; Schmidt et al. 2003; Richter et al. 2003; Strayer 2006). Likewise, dams and dewatered sections of streams limit longitudinal connectivity of aquatic taxa (Bunn and Arthington 2002), thus impacting natural source-sink dynamics of metapopulations (Ligon et al. 2006).

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The Rio Grande originates in the southern Rocky Mountains of western North America and meanders 2,800 km to the Gulf of Mexico. The majority of the basin lies within the Chihuahuan Desert ecoregion. Although a small number of studies exist on the macroinvertebrate assemblage in the Big Bend of the Rio Grande, we know little about the spatial and temporal associations of macroinvertebrates along the mainstem of the river. Davis (1980a) reported 83 taxonomic groups, including 14 ephemeropteran genera, 7 odonate genera, and 8 tricopteran genera, taken from eight sites in the Rio Grande between El Paso (TX) to Del Rio (TX).

The Rio Grande historically and currently is threatened with alterations to its biotic integrity, or pristine state, because of anthropogenic modifications. Geomorphology of the river changed dramatically during the past century through flow alteration, reduced sediment transport, invasive taxa, and water pollution (Schmidt et al. 2003). The intensity of flood events is reduced by 76% with the construction of dams for irrigation and recreation since 1915 (Schmidt et al. 2003). Consequently, dams converted the once shallow, braided river system to a single channel with steep banks throughout most of lower reaches of the Rio Grande (Dahm et al. 2005). Establishment of invasive plants, such as salt cedar (Tamarix sp.) and giant reed (Arundo donax), further exacerbate channel incision by stabilizing river banks and helping to prevent the river from changing course (Schmidt et al. 2003). Furthermore, these plants aid in dewatering portions of the river, especially during periods of low flow because of their high rates of evapotranspiration (Shafroth et al. 2005). Slower decomposition rates of the invasive plants additionally alters nutrient processing rates and subsequently macroinvertebrate communities in the river (Bailey et al. 2001; Andersen et al. 2003; Kennedy and Hobbie

2004). Decreased flow combined with a high concentration of people (> 1,000,000) and their wastewater discharge along the Rio Grande degrades water quality. Often, the Texas Commission on Environmental Quality (TCEQ) issues warnings because of the high concentration of coliform bacteria in the Rio Grande in Texas (www.ibwc.state.gov/wad/flowdata.htm) even in remote areas of the Rio Grande such as Big Bend National Park (BBNP). Historical flows have gradually decreased within the Rio Grande during the 1900s (Figure 1). The Big Bend reach of the Rio Grande probably represents the least impacted stretch of the Rio Grande because of its distance from heavily populated areas and because of less obvious alteration of stream flow through the area as compared to those in the New Mexico and lower Rio Grande near the Gulf of Mexico. Therefore, ecological integrity should be the greatest in Big Bend reach (Schmidt et al. 2003).

Rio Grande mainstem macroinvertebrates serve as critical components to the riverine and terrestrial communities. Macroinvertebrates play a vital role in the complex aquatic food webs, but also process nutrients and bacteria in the water, rendering the water more suitable for human use (Wallace and Merritt 1980). Additionally, because emergent macroinvertebrates typically live briefly as adults and die on land (Meffe and Minckley 1987; Gray 1981), these insects provide an important energy and nutrient flux to the surrounding arid terrestrial environment (Grimm 1988). Declines in macroinvertebrate abundance and diversity therefore can affect a multitude of aquatic and terrestrial organisms, by reducing nutrient and bacteria processing and cause losses of species diversity within the macroinvertebrate community (Goodnight 1973).

To develop a current understanding of macroinvertebrate populations within the Big Bend region of the Rio Grande, purposes of this study included describing current status of the macroinvertebrate community and to assess the influence of environment on macroinvertebrate distribution. Specifically, I assessed spatial and temporal patterns in physical and chemical habitat parameters among four sites within the Big Bend reach of the Rio Grande, described spatial and temporal patterns macroinvertebrate occurrence and abundance, and associated macroinvertebrate abundance with spatial and temporal patterns in physical and chemical habitat parameters. Understanding of macroinvertebrate distribution and habitat associations in the Big Bend reach will provide a baseline index for macroinvertebrate diversity within the drainage and to better predict how current and future anthropogenic modifications or restoration efforts will influence changes in the macroinvertebrate assemblage.

Study Sites

The Big Bend region is located in the central stretch of the Rio Grande downstream from the confluence of Rio Conchos in Mexico and upstream from Lake Amistad along the United States/Mexico border (Figure 2). I selected four sites along the mainstem of the Rio Grande in the Big Bend region. Site 1 occurred within Big Bend Ranch State Park near the dry bed of Contrabando Creek. Site 2 can be found at Santa Elena Canyon within Big Bend NP at the confluence of Terlingua Creek. I selected Site 3 at Hot Springs within Big Bend NP at the confluence of Tornillo Creek; and Site 4 ocurred farthest downstream at Texas Parks and Wildlife Department, Black Gap Wildlife Management Area (Black Gap WMA). All four sites have large riffle areas suitable for macroinvertebrate sampling with the use of a Hess sampler to provide comparable and quantitative efforts among sites and through time.

<u>Method</u>

I collected macroinvertebrates monthly from the four sites in 2006 following sampling protocols described by Barbour et al. (1999). At each site, I collected two samples using a 0.086 m² Hess sampler with 363 μ m mesh. I collected all samples from shallow riffle areas with substrates <50 cm in diameter. For each Hess sample, two individuals cleaned rocks for 120 seconds, ensuring that all insects were removed from substrata before discarding. We used an invertebrate kick net (1.0 m x 1.0 m, 0.5 mm mesh) in swifter and deeper habitats. One collector disturbed substrate in a one meter area by continuously kicking for 60 seconds, allowing for invertebrates and debris to catch in the net downstream. After collection, we usually picked specimens from the debris for 15 minutes, or until it took several minutes to find the next invertebrate (Growns et al. 1997). I stored all Hess and kicknet samples in separate containers with 70% ethanol. I then sorted samples in the laboratory and identified macroinvertebrates to the lowest practical taxonomic level, usually Genus (Merritt and Cummings 2005). I then classified macroinvertebrate taxa into functional feeding groups.

After macroinvertebrate collection, I estimated physical and chemical habitat parameters of the site. I established 16 to 30 quadrats in riffle areas within transects spaced 20 meters apart. At each transect, the team took two current velocity measurements (m/s), two depth estimations (cm), and ten substrate points from three to six quadrats (about 5 x 8 m) spaced equi-distance apart and spanning the width of the river, except during high flow conditions. I measured current velocity measurements with a Marsh-McBirney, Inc. Flo-Mate Model 2000. Substrate identification included the proportion of silt, sand, gravel (mean diameter < 11.5 mm), cobble (<33 mm), and boulder (> 70 mm; Parker 1989). Averages of current velocity, depth, and substrate type per quadrat provided monthly estimates of current velocity, depth, and percent substrate estimation per site and by month. I used a YSI Model 600 multiprobe water quality meter to measure temperature (°C), dissolved oxygen (mg/l), specific conductivity (μ S/cm), pH, and turbidity (NTU) at each site. However, I used average annual mean temperature, dissolved oxygen, specific conductivity, and pH measurements from two Texas Commission on Environmental Quality (TCEQ) monitoring stations located Castilon and Rio Grande Village to accurately estimate chemical habitat parameters of the Big Bend reach (http://www.ibwc.state.gov/Water_Data/rtdata.htm). TCEQ measurements did not include turbidity so I used monthly turbidity measurements by site taken with the multiprobe water quality meter.

Statistics

I assessed spatial and temporal patterns in physical habitat parameters with Principal Components Analysis (PCA). Quadrat estimations of mean current velocity, mean depth, and percent substrate type (i.e., silt, sand, gravel, cobble, boulder, and bedrock) by site and month required z-scored transformation (Krebs 1999) and analysis with SAS (Proc Princomp; version 9.1; Cary, NC). Quadrat scores along PC axes 1 through 3 were enveloped to infer site differences along habitat gradients. Chemical habitat parameters were not included in PCA to avoid the influence different sampling times (e.g., diel fluctuations) among sites.

I assessed spatial and temporal patterns in macroinvertebrate abundance with semi-quantitative kicknet samples and Hess samples. I assessed these data with an Analysis of Similarity (ANOSIM α = 0.05; 9,999 permutations) using PRIMER 6.1.6 software package (Clarke 1993; Clarke and Warwick 2001). ANOSIM specifies average rank of matrices of similarity and dissimilarity (Bray and Curtis 1957) using binary data within and between samples (Growns et al. 1997; Clarke and Gorley 2006). Similarity percentage breakdowns (SIMPER) determined the most common taxa between sites as well as the least common taxa between sites. I determined diversity indices among the four sample sites using the Shannon-Wiener Index. This function examines both species richness and evenness to determine the likelihood of an individual selected from a population at random (Margalef 1957; MacArthur and MacArthur 1961).

I used canonical correspondence analysis (CCA; ter Braak 1986) to assess associations between stream habitat (eg. substrate, current velocity, and depth), and macroinvertebrate community collected from quantitative Hess samples. Canonical correspondence analysis is a direct gradient analysis where an ordination of one multivariate matrix is constrained by a multiple linear regression on variables in a second matrix (McCune and Grace 2002).

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<u>Results</u>

The first three axes of PCA explained 66% of the total variation in physical habitat parameters among the four sampling sites (Figure 3). The first PCA axis explained 28% of the total variation and contrasted relatively deeper and swifter habitats with cobble substrate from habitats with slower current velocities, shallower depths, and gravel and substrates. Generally, PCA axis I described habitat differences along a longitudinal gradient with upstream habitats (Sites 1 and 2) having shallower depths and more sand and gravel substrates and downstream habitats (Sites 3 and 4) having greater depths, primarily cobble substrates, and swifter current velocities. The second PCA axis explained 21% of the total variation and described a substrate gradient. Site 1 consisted primarily of bedrock and large boulders, whereas substrates at downstream sites consisted primarily of cobble. The third PCA axis accounted for 16% of total variation, describing a gradient of swift current velocity and sand substrates to habitats with slower current velocities dominated by cobble and bedrock substrate. Sites 1 and 4 overlapped in multivariate space, as both sites experienced swift currents and large substrate size. Sites 2 and 3 also overlapped considerably in multivariate space, as both of these sites experience a strong stream influence.

I collected macroinvertebrates from sites with shallow depths (range of means: 17.4 - 24.9 cm) and swift current velocities (0.39 - 0.64 m/s; Table 1). Cobble was the most abundant substrate among all sites (47.0 - 73.2%), followed by gravel (4.2 - 33.6%) and bedrock (0 - 14.3%). Sand and silt collectively were <6% of the mean substrate composition by site. Annual mean turbidity ranged from 234.8 to 379.1 NTU by site.

Mean (\pm 1 SD) or median chemical parameters, which I obtained from two TCEQ monitoring stations, ranged between 23°C (\pm 6.0) and 26 (\pm 5.2) °C for water temperature, 6.0 (\pm 2.3) and 6.0 (\pm 3.4) mg/l for dissolved oxygen, 1,699 (\pm 555.3) to 1,858 (\pm 1,092.1) μ S/cm for specific conductivity, and 7.0 (range: 6.0 – 8.0) and 7.5 (range: 7.0 – 8.0) for pH.

Overall, I collected 9,505 macroinvertebrates from Site 1 (n = 1,716), Site 2 (n =1,777), Site 3 (n = 1,829), and Site 4 (n = 4,183). The Big Bend region of the Rio Grande macroinvertebrate community differed among sites (ANOSIM; Global R = 0.104, P =(0.01) with the macroinvertebrate community at Site 1 differing significantly (P < 0.01) from those at sites 3 and 4. The Tricopteran genus Cheumatopsyche, which comprised 24% of the total invertebrate assemblage, dominated Site 1, followed by two dipteran families, Chironomidae (27.3%) and Simulidae (21.4%; Table 2). Members of genus Thaurodes dominated Site 2, comprising 23.2% of the invertebrate assemblage of the site, followed by family Simulidae (14.6%), and genera Traverella (12.9%) and Helichus (12.5%). Site 3 was dominated by genus *Traverella* (50.4%) followed by Cheumatopsyche (14.1%), Erpetogomphus (4.8%), and Helichus (4.3%). Site 4 was dominated by family Leptophlebiidae, specifically the genera Traverella (66.91%), Thaurodes (4.81%), Helichus (10.2%), and Cheumatopsyche (4.8%). Total assemblage of the four study sites consisted primarily of filterer and collector-filterer taxa (68.9%), followed by collectors-gatherers (24.1%), predators (6.5%), collectors-detritivores (0.7%), scrapers (0.4%), and shredders (0.2%; Merritt and Cummings 2005). Shannon-Wiener diversity indices suggested low diversity, the highest at Site 2 (1.78), followed by Site 3 (1.77), Site 1 (1.74), and Site 4 (1.31).

Habitat, season, and site explained 43% of the variability within the macroinvertebrate assemblage (Figure 4). Pure effects for season explained 8% of the assemblage variation (P < 0.01), whereas pure site effects explained 11% (P = 0.17) and pure habitat effects explained 14% (P = 0.20). Significance difference among seasons was attributed primarily to the high abundance of some taxa (i.e., Simulidae and Chironomidae) in late fall and winter before spring emergence, and the high abundance of other taxa (i.e., *Traverella* and *Thaurodes*) during the summer before their Fall emergence (Figure 5). Although macroinvertebrate community at Site 1 differed from the remaining three (ANOSIM), site did not explain significant variation in the macroinvertebrate community. However, site and environmental gradients expressed by CCA axes still provide predictors of community structure because of pure and partial (2- and 3-way) interactions among season, site, and habitat.

Environmental factors with the highest positive centroids on CCA axis I were July-Sept (0.96), Black Gap (0.93), and Apr-June (0.76). Habitat factors with the highest negative centroids were Contrabando (-0.76), silt substrate (-0.71), and Jan-Mar (-0.71). Macroinvertebrate species were highly correlated ($r^2 = 0.89$) to the CCA axis I. Biplot scaling score included *Neptopsyche* (1.38), *Neochoroterpes* (1.36), *Hygrotus* (1.13), Haplotaxida (1.08), *Trichocorixa* (0.98), *Gyretes* (0.88), *Corbicula* (0.72), *Stenophysa* (0.52), *Traverella* (0.50), *Helichus* (0.46), *Microcylloepus* (0.21), *Argia* (0.17), *Hetaerina* (0.16), *Erpetogomphus* (0.11), and *Fallceon* (0.09). Taxa negatively associated with axis I included *Protoptila* (-1.00), *Ambryus* (-0.71), *Corydalus* (-0.69), *Callibaetis* (-0.61), Sphaerium (-0.55), Chironomidae (-0.50), *Cryphocricus* (-0.45), Simulidae (-0.44), Tabanidae (-0.43), *Tricorythodes* (-0.33), *Cheumatopsyche* (-0.10), and *Thaurodes* (-0.02).

Discussion

Dominance of ephemeropterans, dipterans, and trichopterans (84% across all sites) in my study measured less than the overall abundance of these three taxonomic groups (93%) in 1975 - 1977 (Davis 1980a). Overall relative abundances increased from 39.6% in 1975 – 1977 to 54.8% in 2006 for ephemeropterans, slightly decreased from 21.5% to 17.7% for dipterans, and decreased from 31.6% to 11.4% for trichopterans. Average diversity indices changed from 2.14 in 1975 to 1.65 in 2006. Shifts in diversity indices are probably a result of taxonomic resolution within Diptera, as the 1975-1977 study classified dipteran taxa to Genus or Species level. Observed differences in macroinvertebrates abundance and diversity through time likely did not relate to differences in water years between time periods because mean daily discharge (±SD) in $1976 (28.8 \pm 33.4 \text{ m}^3/\text{s})$ and $1977 (19.6 \pm 33.4 \text{ m}^3/\text{s})$ feel within the range of mean daily discharge in 2006 (19.1 \pm 31.8 m³/s) and all three measured lower than the mean daily discharge (34.4 ± 75.7) on record (1936 - 2007); International Boundary Water Commission gauging station at Castolon; 08-3750). Instead, I expect observed differences to be a result of improvements in water quality through time. This is illustrated by the increased abundance of ephemeropterans, which are often associated with higher water quality (Baumgardner and Bowles 2005) and decreased abundance of dipterans and *Cheumatopsyche*, which generally are associated with areas of lower water

quality such as water with high levels of nutrients and bacteria (Fuller et al. 1988; Edwards 1987; Wallace and Merritt 1980).

Despite moderate improvements of water quality, the Rio Grande macroinvertebrate community is still impacted by poor water quality. For example, I found higher abundances of dipteran taxa at Site 1, which decreased along a downstream gradient. This was contrasted by lower abundances of ephemeropteran taxa at Site 1, which increased at the downstream sites. Although I found habitat similarities between sites 1 and 4 as well as sites 2 and 3, macroinvertebrate assemblages at these sites instead changed longitudinally. Davis (1980a) found similar assemblage trends in his study, although 87% of the most upstream site consisted of *Corbicula manilensis*,

Homoeoneuria, Hydrobaenus, Oligochaeta, Palpomyia tibialis, and Paraclodopelma. Farther downstream at Lower Presidio, the 71% of the dominant taxa consisted of *Cheumatopsyche, Simulium, Thaulodes, Traverella*, and *Orthocladius*. The five dominant taxa at each site continued to drop in percentages after reaching a small increase in percent dominant taxa at Santa Elena (Site 2 of my study). Both Davis (1980a) and my study suggest that there an upstream bacterial pollution input occurs above the Big Bend region of the Rio Grande that affects macroinvertebrate assemblages. Our studies also illustrate water quality improvement along a downstream gradient as reflected by the dominant taxa.

Furthermore, a 2003 water quality survey by the Texas Clean Waters Program at the International Boundary and Water Commission reported that water from the Rio Grande at Lajitas (in close proximity to Site 1) was unfit for human consumption due to elevated levels of chloride, sulfate, total dissolved solids, and fecal coliform bacteria.

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The survey concludes that ingestion of the bacteria present in the river by humans could lead to gastrointestinal disease. The source of the pollution at this site has not been determined, although there are confirmed pollution inputs upstream at Presidio from wastewater (IBWC 2003). As a result of the large populations of filter feeding macroinvertebrates in the upstream reaches of the study region (Sites 1 and 2), the macroinvertebrate assemblages downstream (Sites 3 and 4) have reduced densities of filter feeding taxa. In this sense, the filter feeding taxa upstream are reducing their own downstream densities by consuming detritus in drift and allowing for a greater diversity of macroinvertebrates at Sites 3 and 4 (Wallace and Merritt 1980).

Although the likely pollution influences are likely a result of nutrient inputs from agricultural or municipal waste, other types of pollution, such as DDT and toxaphene, may pose an additional threat to the ecological integrity of the Rio Grande. Elevated levels of DDE and toxaphene were reported in fish and avians in the early 1980s in the lower Rio Grande Valley (White et al. 1983). Additionally, Davis (1980c) suggested elevated levels of the pesticide residues DDT and DDE in sediments from the Rio Conchos may reduce macroinvertebrate diversity in the areas directly downstream, thus favoring more tolerant taxa. Leptophlebiids, such as the *Traverella*, were also reported to have deformities of the eyes and appendages as a possible result of these pesticide residues (Davis 1980a).

Though impacted, the macroinvertebrate community in the Big Bend reach is similar to those in other rivers within the Rio Grande drainage and in other rivers in more humid areas of the southwest. The Devils River, a relatively unimpacted and spring-flow influenced tributary located downstream from my study area, has at least 26 taxa in

common with the Big Bend region (Davis 1980a). Likewise, the Pecos River, a highly impacted tributary also located downstream for my study area, consists primarily of ephemeropterans (Leptophlebiidae; 30.7%), trichopterans Hydropsychidae (7.3%), and Hydroptilidae (5.4%) and dipterans Chironomidae (6.8%) and Simulidae (1.0%); Davis 1980b), 26 of which were found at my study sites within the Big Bend Region of the Rio Grande. Because these similar taxa are found in both disturbed systems, such as the Rio Grande and Pecos River, as well as less impacted systems such as the Devils River, it can be inferred that these generalist taxa are present as a result of their tolerance to natural stressors from an arid environment. Although human alterations to the environment have increased the number of filter-feeding taxa within the Big Bend region of the Rio Grande in the past thirty years (Davis 1980a) in response to increase nutrient levels (Wallace and Merritt 1980), the diversity of west Texas arid rivers is inherently low. The Rio Grande, although similar to other arid drainages, varies substantially from more temperate regions, such as areas of central Texas. In the Blanco River of central Texas, dipterans Chironomidae and Simulidae are the most abundant macroinvertebrates within the assemblage, the two families comprising approximately 20% of relative abundances (Pendergrass 2006). The Rio Ayuquila of west-central Mexico consists primarily of 79 major taxa, including Corydalidae, Elmidae, Baetidae, Leptophlebiidae, Tricorythidae, and Hydropsychidae (Weigel et al. 2002). Both the Blanco River and the Rio Ayuquila experience fewer flash floods, thus housing fewer generalist taxa than rivers in arid regions, such as the Rio Grande.

Physical water parameters between upstream (Sites 1 and 2) and downstream (Sites 3 and 4) reaches of the Rio Grande was did not differ, indicating the differences in

macroinvertebrate assemblages between study sites can not be attributed to differences in pH, dissolved oxygen, temperature, and conductivity. Habitat characteristics (as seen in PCA) are similar as well, differing only in comparison between Sites 1 and 4. Sites 2 and 3 shared very similar habitat characteristics in the PCA, likely because both are influenced by small desert streams and have very similar substrate. Further alterations in habitat as a result of dams or irrigation may favor more generalist species and dramatically reduce overall species diversity within the system.

Overall, the macroinvertebrate assemblage from the benthic regions of my study sites provided an accurate picture of the aquatic ecosystem of the Rio Grande. Limitations of this study were mainly contributed to primarily included the lack of macroinvertebrate sampling from deep pools, eddies, and side channels. Additionally, further information could be acquired from stream inputs, such as Terlingua Creek and Tornillo Creek; greater macroinvertebrate diversity has been found in small tributaries and ephemeral pools within Big Bend (Bane and Lind 1978). My results could be used to continue monitoring water quality in the Rio Grande. Decreased flows alone may produce a more uniform macroinvertebrate assemblage, consisting primarily of Simulidae and Chironomidae.

Filter feeding macroinvertebrate taxa play an important role in the overall structure and function of the aquatic ecosystem of the Rio Grande – primarily through the reduction in nutrient load and bacterial content in water and sediments. This is of particular importance for human recreational purposes, specifically at Lake Amistad in the middle Rio Grande, downstream from Big Bend National Park. Because this lake supports a large bass population, it attracts numerous recreational anglers from both the United States and Mexico. The ability of the river to repair itself through nutrient processing by macroinvertebrates has allowed for a not only a successful fishery, but also cleaner water availability for downstream municipalities.

CHAPTER II

FOOD HABITS AND REPRODUCTION OF THE DOBSONFLY, *CORYDALUS CORNUTUS* IN THE RIO GRANDE BASIN

Introduction

As a result of limited water resources, arid environments depend on aquatic macroinvertebrates in their roles as secondary producers (Fisher and Gray 1983). Similarly, the life history traits of emergent insects render them an equally important assets to the surrounding terrestrial environment (Jackson and Fisher 1986; Jackson 1988). As larvae, emergent insects accumulate nutrients through feeding on detritus, leaf matter, or other insects. These nutrients are transferred to the terrestrial environment during their synchronous emergence as adults, frequently occurring over a period of several days. The majority of the dominating insect orders, especially Trichoptera and Ephemeroptera, spend most of their lives in an aquatic larval form and emerge briefly as mating adults (Fisher and Gray 1983; Jackson and Fisher 1986; Grimm 1988; Jackson 1988). Understanding the role of emergent insects remains crucial for the functioning of desert ecosystems as these insects provide nutrients not only to the aquatic habitat, but also to the surrounding desert after the insects have completed their life cycle (Jackson and Fisher 1986; Jackson 1988). Members of the family Corydalidae are the largest of the

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emergent freshwater hexapods and exhibit life history traits providing a large nutrientflux. The Dobsonfly, *Corydalus cornutus*, commonly occurs in fast moving streams in the southern United States (Stewart et al. 1972; Epperson and Short 1987). Four other described species of the genus *Corydalus* exist in Texas, although they are not as widely distributed as *C. cornutus* (Contreras-Ramos and Harris 1998).

In its larval form (commonly referred to as hellgrammites), *C. cornutus* lives under large rocks in fast flowing riffle areas in streams (Short et al. 1987). The predaceous larvae have sharp mandibles which they use to feed opportunistically on invertebrates found in drift or in benthic areas (Triplehorn and Johnson 2005). Previous studies on temperate populations in Texas suggest that *C. corydalus* consumes primarily Dipteran and Ephemeropteran prey and have a relatively short digestion period of three to six hours (Stewart et al. 1972)

In Texas, *C. cornutus* spends up to a year in its larval form, after which it emerges and pupates in the soil on river banks or in rotting logs during the summer months. It emerges as an adult to reproduce, rarely feeding, although it has been reported to consume nectar and fruit (Brown and Fitzpatrick 1978). Both male and female *C. cornutus* have an average adult lifespan of only five days. During this period, females typically lay up to 3000 eggs in several eggs masses placed on logs or vegetation surrounding the stream area during the early evening hours. *Corydalus cornutus* differs from other Megaloptera in that they produce an egg case that may help protect against overheating or predation (Mangan 1992). The egg case forms from a sticky secretion emitted by the female after oviposition. Hatchling hellgrammite larvae drop from egg masses into water after a period of two weeks (Stewart et al. 1972; Short et al. 1987; Michael and Culver 1986; Mangan 1992 Kirk and Smock 2000).

Although widespread throughout North America, most research on this family involves those species located in the northern regions of the US and Canada (Brown and Fitzpatrick 1978). Very little research exists on Corydalids in the deserts of the American Southwest. Because arid systems differ from more temperate climates in rainfall, climate, species diversity, and human populations, it is feasible that arid populations of *C. cornutus* will exhibit different life history strategies. For example, *C. cornutus* exhibits a multivoltine life cycle in northern latitudes of North America and typically univoltine life cycle at southern latitudes (Brown and Fitzpatrick 1978). Additionally, food selection as well as hatch success may differ in arid regions from temperate populations. To understand arid populations of *C. cornutus*, my objectives of this study are to compare larval feeding, age structure, and egg hatching success of *C. cornutus* among three rivers within the Rio Grande drainage.

Study Site

To study the diet of *C. cornutus*, I collected larval specimens from four sites within the Big Bend region of the Rio Grande: Contrabando, upstream of Lajitas, Santa Elena, at the confluence of Terlingua Creek, Hot Springs, at the confluence of Tornillo Creek, and Black Gap Wildlife Management Area, just downstream of Big Bend National Park (Fig 2.1). I also selected one site (Quemado) along the lower Rio Grande located outside of Laredo. All five Rio Grande sites are representative of drainages experiencing reduced flow as a result of impoundments and irrigation diversions (Edwards and Contreras-Balderas 1991). For comparison, I selected two less impacted sites: Dolan Falls, on the Devil's River, and NM 95, located on the upper Pecos River in New Mexico. I chose all of these locations because of their relatively high current velocity and high abundance of *Corydalus cornutus*. The substrate at all sites within the study area varies from silt and sand to gravel (mean diameter = 11.5 mm), cobble (mean diameter = 32.9 mm), and boulder (mean diameter > 68.9 mm; Parker 1989). I collected egg masses from two sites along the lower Rio Grande near Laredo: Kickapoo and San Ygnacio, and one site, Dolan Falls, located on the Devils River.

Method

I collected larval *C. cornutus* on a monthly basis from January through December 2006, obtaining all specimens with the use of kick nets or Hess sampler (0.086 m²; 363 μ m mesh). Larval *C. cornutus* were immediately preserved in 70% ethanol. In the laboratory, I measured the widest portion of each specimen's head to the nearest 0.1 mm to determine head capsule width (HCW), as HCW is a good predictor of dry weight (Brown and Fitzpatrick 1978). I created HCW size classes to create a HCW-frequency histogram to determine emergence season and longevity within an arid region. Because the upper portions of the digestive system contain identifiable material, I dissected all specimens to remove only the proventriculus and crop, which I refer to as the gut (Stewart et al. 1973). After identifying the contents of the upper digestive tract to lowest practical taxonomic resolution (usually Genus), I weighed the gut contents to determine the percent gut content of each prey item.

I collected egg masses in July 2007 from two sites along the Laredo region of the Rio Grande (at San Ygnacio Bird Sanctuary and Kickapoo Reservation), and from one site on the Devils River at Dolan Falls. Egg masses occurred in large quantities on maple (*Acer sp.*) and ash tree (*Fraxinus sp.*) species on the northern side of the river. To keep clutches intact without breaking the white protective coating, I trimmed the leaf substrate buy cutting the petiole. In the laboratory, I stored egg masses in Petri dishes and incubated them at ambient temperature (29°C for 36 hours) until hatching. I classified surviving hatchlings as those that emerged from the egg masses, while unhatched hatchlings were those deceased within the egg mass or undeveloped eggs. After the hatch and desiccation period, I counted surviving (those present outside of egg mass) and non-surviving (those remaining inside the egg mass) individuals. I analyzed number hatched, percent hatched, and total egg counts using 1-factor analyses of variance followed by Fisher's LSD to assess between treatment differences ($\alpha = 0.05$).

<u>Results</u>

I collected 89 juvenile *C. cornutus* for gut content analysis from Contrabando (N = 13), Santa Elena (N = 20), Hot Springs (N = 5), Black Gap (N = 21), Quemado (N = 2), Dolan Creek (N = 6), and NM 95 (N = 22; Table 2.1). Upper digestive tract was empty in 45 (51%) of the collected individuals. Among those with gut contents, detritus was the most common and abundant food item consumed. Detritus was found in individuals from all seven sites and comprised 33 to 100% of the gut content weight (Table 2.1). Cheumatopsyche were found in individuals from five sites and comprised 0 to 50% of the gut content weight. Simulids comprised 4 to 50% of the gut content weight at two sites.

Other food items (i.e., Leptophlebiidae, Chironomidae, *Corydalus*, Coengrionidae, and unidentified arthropods) cumulatively comprised <25% of the gut content weight. Among sites, gut contents of individuals taken from Black Gap contained the greatest variety of food items (N = 5 categories). Individuals from Quemado and Dolan Creek were collected in late April, 2007 and were found to consume only detritus.

Head capsule widths ranged between 2.1 and 10.6cm from January 2006 to April 2007. The smallest individuals (2.0 to 2.9cm) were taken during the months July through September. Mean individual size progressively increased October through December (4.6cm), January through March (5.4cm), and April through June (6.9cm) across all sites. Emergent adults were observed in May and June at Black Gap and San Ygnacio.

Total number of eggs per clutch (±SD) did not statistically differ ($F_{3, 43} = 1.21, P = 0.30$) between sites. Mean number of eggs per clutch measured 1,317 (± 459) at Dolan Creek, 1,560 (± 516) at Kickapoo, and 1,321 (± 397) at San Ygnacio. Number of hatched larva per clutch also did not statistically differ ($F_{3, 43} = 2.38, P = 0.1$) between sites. Mean number of hatchlings per clutch included 1,199 (± 488) at Dolan Creek, 1,316 (± 548) at Kickapoo, and 899 (± 491) at San Ygnacio. However, mean percent hatched per clutch was lower ($F_{3, 43} = 5.14, P = 0.01$) at San Ygnacio (68±26.0%) than Dolan Creek (91±8.9%) and Kickapoo (84±17.3%).

Discussion

Diet of *C. cornutus* in the arid region of the Rio Grande differs from those living in more temperate regions of the southern United States. Previous studies report larval *C. cornutus* to consume primarily dipterans, feeding opportunistically on a variety of other macroinvertebrates (Brown and Fitzpatrick 1978; Epperson and Short 1987). My study suggests otherwise, as the majority of larval digestive tracts contained detritus. High amounts of detritus in larval guts have not been reported in previous studies, although related species of alderflies present in disturbed systems have been found to shift from a primarily insectivorous diet to one consisting of large amounts of detritus (Woodrum and Tarter 1971).

In desert watersheds that are typically limited in allochthonous inputs, accumulation of detritus on substrate increases macroinvertebrate populations due to the increased nutrient load (Parker 1989), especially Dipterans as they are known filter feeders (Wallace and Merritt 1980). A 2003 water quality survey by the Texas Clean Waters Program at the International Boundary and Water Commission reported that water from the Rio Grande at Lajitas contained elevated levels of fecal coliform bacteria (IBWC 2003). Although the presence of bacteria at this location renders the water unfit for human consumption, it provides nutrients to detritivores living in benthic regions, thus increasing the populations of dipteran prey items. My findings, however, suggest that scavenging for detritus on rocks and sediment in the benthic areas is more valuable energetically than capturing insect prey items. The role of turbidity may have an additional affect on prey selection of *C. cornutus* as well as other predators (Davis 1980) – consumption of detritus may occur accidentally during predation of benthic invertebrates.

The life cycle of *C. cornutus* was similar to those in other parts of Texas. Although northern populations of *C. cornutus* have been found to have a 2- or 3-year life cycle (Chandler 1956; Brown and Fitzpatrick 1978), warmer climates – including arid regions – typically harbor populations with a 1-year or univoltine life cycle. Emergence occurred in warm months, reaching peak emergence in May and June, as indicated by the abundance of large larval size classes during late spring followed by an immediate drop in size during late summer. Frequent drying events in desert systems reduces the number of bi- or multi- voltine macroinvertebrates present in the assemblage (Gray 1981). Summer drying events may also select for macroinvertebrates to reach peak body size in spring, allowing for most growth to take place during the winter months when drying events are not uncommon (Smith and Wood 2002). This allows for pupation and emergence in summer, which ensures aerial adults and egg masses can recolonize the stream after extreme weather events such as drying periods or flash floods. Interestingly, Dolan Falls, a spring influenced system, had similar life cycle patterns than the Rio Grande and Pecos Rivers, suggesting that emergence is not affected by stable water temperature from spring outflows.

Number of eggs laid per clutch and number of eggs hatched did not vary between sites, however percent hatch success was significantly lower at San Ygnacio. Failure to find differences in reproductive success between the two drainages as a whole is likely a result of the life history of *C. cornutus*; the aerial adult phase has the ability to travel a long distance before it reproduces, meaning that egg masses deposited at study sites may be from individuals from different watersheds.

Because the Rio Grande is a threatened system within a fluctuating climate, understanding the life histories of dominant taxa will aid future conservation efforts. For example, macroinvertebrate taxa, especially *C. cornutus* are an important part of the nutrient dynamics within the system, providing food to insectivorous fish and birds in both their aquatic larval stage and aerial adult stage. Shifts in the reproduction or trophic ecology of certain dominant taxa such as *C. cornutus* will have an affect on the organisms that consume them. Additionally, further research on how macroinvertebrates adapt to pollution in impacted systems may aid management efforts within the Rio Grande.

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APPENDIX 1.1



FIG. 1.1. Mean annual discharge of the Rio Grande at a) Elephant Butte, New Mexico, b) Rio Grande Village, Big Bend National Park, Texas, and c) Brownsville, Texas.

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FIG. 1.2. Locations of sampling sites in Texas and New Mexico along the Rio Grande, Devils River, and Pecos River. In the Big Bend region, we categorized Contrabando as Site 1, Santa Elena as Site 2, Hot Springs as Site 3, and Black Gap as Site 4.



FIG. 1.3. Principal components analysis (PCA) of habitat data collected from Big Bend National Park. Individual sites are contained in ordination space, and habitat loadings of a) PCA I and PCA II and b) PCA I and PCA III are located in the margins.



FIG. 1.4. Canonical components analysis (CCA) of species and habitat data collected from Big Bend National Park, Texas



FIG. 1.5. Monthly relative abundance of a) summer emergent macroinvertebrate taxa versus b) spring emergent macroinvertebrate taxa. Specimens collected from Big Bend National Park, Texas, January through December 2006.

Variable	Site 1	Site 2	Site 3	Site 4
Current Velocity (m/s)	0.39 ± 0.24	0.64 ± 0.10	0.53 ± 0.12	0.56 ± 0.16
Depth (cm)	17.4 ± 7.5	17.8 ± 4.2	17.5 ± 4.0	24.9 ± 4.9
Substrate (%)				
Silt	0.3	0.9	1.7	0.0
Sand	0.0	0.7	4.1	0.0
Gravel	14.4	33.6	19.8	4.2
Cobble	47.0	64.6	73.2	71.8
Boulder	24.0	0.3	1.7	24.0
Bedrock	14.3	0.0	0.0	0.0
Temperature (°C)	18.6 ± 7.3	18.02 ± 7.4	22.54 ± 6.2	20.03 ± 6.3
pH	8.18	8.26	7.98	8.09
Turbidity *	370.66 ± 511.5	234.77 ± 458.6	379.09 ± 563. 8	326 01 ± 531 6

TABLE 1.1. Mean water quality and substrate values of
Big Bend sample sites, January through December,
2006.

* Nephlometric turbidity units

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Class	Order	Family	Genus or Scientific Nam	e Trophic Guild	Percent Abundance				2762
		-		•	Site 1	Site 2	Site 3	Site 4	
Turbellaria		·		Collectors-gatherers			0 11		10129
Bıvalvıa	Pelecypoda	Sphaeriidae	Corbicula	Filterers	0 12		0 05	0.38	
			Sphaerium	Filterers		0 11			
Gastropoda	Limnophila	Physidae	Stenophysa	Scrapers	0 06	0 06	0 38		20
Oligochaeta	Haplotaxıda			Collectors-gatherers	0 87		0 16	0 65	õ
Arachnida	Acarına	Acarı		Predators			0 11		
Insecta	Ephemeroptera	Baetidae	Calıbaetis	Collectors-aatherers	1 77	0 51		1.05	
	• •		Fallceon	U U	1 28	3 83	1 75	0 31	
		Heptageniildae	Neochoroterpes		0 12		0.27		
		Leptophlebiidae	Thaurodes		8.16	23 19	3.94	4 81	
			Traverella	Collectors-filterers	1 81	12 90	50 41	66.91	
		Tricorythidae	Tricorythodes	Collectors-gatherers	3 73	1 97	3 66	0 24	
	Odonata	Coengrionidae	Argia	Predators	0 70	1 18	4 43	1 98	
		Calopterygidae	Hetaerina americana			0.11	0 44	0 22	
		Gomphidae	Brachemorhoga			0 11		0.07	
			Erpetogomphus		0 93	0 96	4 81	0 79	
		Macromiidae	Macromia		0 06				
	Hemiptera	Corixidae	Trichocorixia		0.06		0 16	0 10	
		Naucoridae	Ambryus		0 41	0 17		0.05	
			Cryphocricos		0 12	0.39	0 22		
		Veludae	Helocharus			0 06			
	Megaloptera	Corydalidae	Corydalus cornutus		1 17	0 96	0 22	0 84	
	Trichoptera	Glossomatidae	Protoptila	Scrapers	0 12	0 39	0 11		
		Leptoceridae	Neptopsyche	Shredders			0 05	0 43	
		Hydropsychidae	Cheumatopsyche	Collectors-filterers	23 78	11 42	14 05	4 81	
	Lepidoptera	Pyralidae	Petrophila	Shredders	0 12				
	Coeloptera	Dytiscidae	Hygrotus	Predators			0 05	0 02	
		Elimidae	Microcylloepus	Collectors-detritivores	0 93	0 51	0 77	0 41	
		Dryopidae	Helichus	Scrapers, Collectors-gatherers	4 78	12 49	4.26	10 23	
		Gyrinidae	Gyretes	Predators	0 17	0 73	0.60	0 36	
	Diptera	Simulidae		Collectors-filterers	21 40	14 63	3 32	4 26	
		Tabanıdae		Predators	0 06	0 28	0 16	0 17	
		Chironomidae		Collectors-gatherers, filterers	27 27	11.09	5 47	0 93	
				N	1,716	1,777	1,829	4,183	

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APPENDIX 2.1



FIG. 2.1. Locations of sampling sites in Texas and New Mexico along the Rio Grande, Devils River, and Pecos River. In the Big Bend region, we categorized Contrabando as Site 1, Santa Elena as Site 2, Hot Springs as Site 3, and Black Gap as Site 4.





Diamage	Site	N	Mean total weight (mg)	Empty stomachs (%)	Detritus (%)	Cheumatopsyche (%)	Simulidae (%)	Leptophlebiidae (%)	Chironomidae (%)	Corydalus (%)	Coengrionidae (%)	Unidentified arthropods (%)
R10 Grande	Contrabando	12	>1	67%	50.0 ± 58	38 ± 48	*********		125 ± 25			an a
	Santa Elena	20	>1	60%	50.0 ± 53	12.5 ± 35		125 ± 35				25.0 ± 46
	Hot Springs	5	>1	60%	50.0 ± 71	50.0 ± 71						
	Black Gap	4	>1	80%	33.3 ± 52	16.6 ± 41	28 ± 48			16.7 ± 10	4.7 ± 10	
	Quemado	2	>1	0%	$100\ 0\pm 0$							
Devils River	Dolan Creek	6	>1	0%	$100\ 0\pm 0$							
Pecos River	Highway 95	22	>1	5%	90.0 ± 30	4 7 ± 22	50 0 ± 58					

Table 2.1. Gut content analysis of *Corydalus cornutus* collected from the Rio Grande, Texas, Devils River, Texas, and Pecos River, New Mexico, collected January 2006 through June 2007.

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