11-10-2022

Causal Mechanisms of Food Web Structure and Function - A Tale of Three Drivers: Spatiotemporal Variation, Ecosystem Engineering, and Invasive Species

Peter J. Flood
pfloo001@fiu.edu

Recommended Citation

Flood, Peter J., "Causal Mechanisms of Food Web Structure and Function - A Tale of Three Drivers: Spatiotemporal Variation, Ecosystem Engineering, and Invasive Species" (2022). *FIU Electronic Theses and Dissertations*. 5167.
https://digitalcommons.fiu.edu/etd/5167

FLORIDA INTERNATIONAL UNIVERSITY

Miami, Florida

CAUSAL MECHANISMS OF FOOD WEB STRUCTURE AND FUNCTION – A TALE OF THREE DRIVERS: SPATIOTEMPORAL VARIATION, ECOSYSTEM ENGINEERING, AND INVASIVE SPECIES

A dissertation submitted in partial fulfillment of
the requirements for the degree of
DOCTOR OF PHILOSPHY
in
BIOLOGY
by
Peter Joshua Flood

2022

To: Dean Michael R. Heithaus
College of Arts, Sciences and Education

This dissertation, written by Peter Joshua Flood, and entitled Causal Mechanisms of Food Web Structure and Function – A Tale of Three Drivers: Spatiotemporal Variation, Ecosystem Engineering, and Invasive Species, having been approved in respect to style and intellectual content, is referred to you for judgment.

We have read this dissertation and recommend that it be approved.

William T. Anderson

Evelyn E. Gaiser

Alastair R. Harborne

Daniel Simberloff

Joel C. Trexler, Major Professor

Date of Defense: November 10, 2022

The dissertation of Peter Joshua Flood is approved.

Dean Michael R. Heithaus
College of Arts, Sciences and Education

Andrés G. Gil
Vice President for Research and Economic Development
and Dean of the University Graduate School

Florida International University, 2022

DEDICATION

To my life partner, Lanie, whose playfulness and love keep me grounded and inspire me. To my parents, Janice and Andrew, and to my grandparents, Elizabeth and Harold Flood and Jane and John Fox, whose love, support, and dedication to my education made all of this possible – I stand on the shoulders of giants.

ACKNOWLEDGMENTS

I am forever grateful for the work of my advisor, Joel Trexler, who has instilled in me so much knowledge about ecology, the Everglades, and how to be a scientist. He has been a steadfast role model and an encouraging mentor throughout my dissertation journey. The support of my committee was integral to completing this research. Daniel Simberloff is an invaluable resource, a veritable encyclopedia, who was always available and provided quick, insightful feedback. Evelyn Gaiser has always supported my research, both in this dissertation and in other pursuits while at FIU, and introduced me to algal priming. If you have not already, go watch Evelyn's TEDxFIU presentation. Alastair Harborne and his wealth of knowledge about community ecology, trophic ecology, and trait-mediated effects has made valuable contributions to my work. For the past couple of years, Alastair has also been the only committee member who works at the same campus. I have greatly benefited from Bill Anderson's expertise on stable isotopes, particularly when I was concerned some samples may have been contaminated (they were not).

I would like to acknowledge the University Graduate School for providing me with a tuition waiver for calendar year 2022. This allowed me to focus on writing my dissertation these past months. I am also grateful to the Everglades Foundation who funded me as an FIU ForEverglades Scholar which facilitated development and writing of my dissertation. Most of the research in this dissertation was graciously funded by a grant that Joel received from the U. S. National Park Service Critical Ecosystem Studies Initiative (CESI). This research was done in collaboration with and also supported by grants from the National Science Foundation through the Florida Coastal Everglades

Long-Term Ecological Research program and the FIU CREST Center for Aquatic Chemistry and Environment. It is a privilege and an honor to be a part of these two teams of dedicated scientists. Their leaders, Todd Crowl, John Kominoski, and Evelyn Gaiser are unwavering mentors and unrelenting advocates for graduate students.

I want to thank numerous folks for their time, energy, and support in the field, lab, classroom and beyond. This research began with Bill Loftus in the 1970s, long before I was born. Bill’s data provided an extremely rare opportunity to study biological invasions. I am incredibly grateful that Bill and Joel entrusted me with those data that Bill worked so hard to collect. I would like to thank Jeff Kline who has set a great example as a scientist and been generous with his time and support. Bill Chamberlain, Rafael Gonzalez, SERC, Institute of Environment, FIU, and Everglades National Park staff have been irreplaceable in their logistical support. Neither this project nor who I am as a scientist would be the same without the friendship and mentorship of Bradley Strickland. Sarah Sisco, Maria Sabando, Venus Garcia, Ariana Jonas, and Isabella Lopez and countless other technicians and volunteers made this work possible by assisting with field and laboratory work. It was serendipitous that I was part of the Fall 2017 cohort, without their friendship graduate school would not have been the same, especially: Sarah Luongo, David Kochan, Laura Garcia Barcia, Jessica Quinlan, and Nicole Strickland. I am thankful for all of my friends in the academic community and lab mates at FIU, in particular Alan Mock, Matt Lodato, and Marco Fernandez.

I am so grateful to be part of an incredibly loving and supportive family. My parents, Andrew and Janice, worked hard and always prioritized the education of their children. They instilled in me a deep love and passion for learning, taught me how to work hard and persevere, and encouraged me to be who I am and be that well. I am thankful for the close friendship and support of my siblings, Elisabeth, Christopher, and Joseph and their spouses, Phillip and Kayla. I am continuously inspired and rejuvenated by the curiosity of my nieces, Anna, Aletheia, and Abigail, and my nephews, Mason and Micah. I am honored to be marrying into a caring and supportive family. Finally, I want to thank my partner and fiancé, Lanie Esch. She has supported me every step of the way from long nights of analyses and writing to early mornings for field work. This dissertation is what it is, and I am who I am, because of you.

ABSTRACT OF THE DISSERTATION

CAUSAL MECHANISMS OF FOOD WEB STRUCTURE AND FUNCTION - A TALE OF THREE DRIVERS: SPATIOTEMPORAL VARIATION, ECOSYSTEM ENGINEERING, AND INVASIVE SPECIES

by

Peter Joshua Flood

Florida International University, 2022

Miami, Florida

Professor Joel C. Trexler, Major Professor

Food webs are natural networks that provide a comprehensive framework for understanding how ecosystems function. Structure and function of food webs are predicated on resource availability, which is in turn driven by fluctuations in abiotic conditions that vary in space and time. Ecosystem engineers, such as American alligators (*Alligator mississippiensis*), which create and maintain novel habitats, and invasive species, such as African Jewelfish (*Hemichromis letourneuxi*), further influence trophic dynamics and therein ecosystem function. In my study system, Everglades National Park, African Jewelfish have recently undergone a drastic increase in density with associated declines in native fauna. In this dissertation, I quantified trophic dynamics between seasons and among habitats prior to African Jewelfish invasion. I found that consumers often underwent spatiotemporal shifts in diet and trophic niche, but that flexible omnivory facilitated relatively constant trophic positions. I used a contemporary study of alligator-engineered habitats to quantify effects of habitat modification on trophic dynamics and to test the Stress Gradient Hypothesis (SHG). I found that many consumers underwent dietary shifts in engineered habitats, that trophic niches based on stomach contents increased during the dry season, and that trophic niches based on stable isotopes had the opposite trend. The stomach content results suggested decreased competition in the dry season based on the Niche Variation Hypothesis and supported the SGH, while the stable isotope results suggested that other drivers such as consumption may play a more important role than competition in structuring these communities over longer time intervals. I compared shared habitats from the previous two chapters (ponds and marshes) to examine trophic effects of African Jewelfish and test the Trophic Disruption Hypothesis (TDH). Post-invasion, I found widespread trophic displacement and dispersion, increased reliance on autotrophic energy, that spatiotemporal trophic dynamics had a greater magnitude, and a fundamental shift in energy fluxes through the food web that supported the TDH. Energy fluxes that had previously traveled through small, abundant fishes were rerouted to larger fishes, including invasive Cichlids such as African Jewelfish. This dissertation emphasizes the importance of spatiotemporal variation, ecosystem engineers, and invasive species on trophic dynamics and ecosystem function.

TABLE OF CONTENTS

LIST OF TABLES

TABLE PAGE

CHAPTER II

CHAPTER III

CHAPTER IV

LIST OF FIGURES

FIGURE PAGE

CHAPTER II

CHAPTER III

CHAPTER IV

CHAPTER I

GENERAL INTRODUCTION

Food webs are natural networks that provide a comprehensive framework for understanding how ecosystems function (Elton 1927). Processes such as biogeochemical cycling, primary production, and secondary production are ecosystem functions dictated by trophic interactions present in food webs (Lindeman 1942). Similarly, ecosystem stability and resiliency are driven by food-web structure (Rooney and McCann 2012). Food-web network structure (e.g., interaction strength, diversity, and connectance) provides insight into the ecological mechanisms driving ecosystem level patterns (Montoya et al. 2006, Ings et al. 2009). This allows food webs to unite the fields of community ecology and ecosystem ecology (Wang and Brose 2018). As a result, understanding and describing food webs is fundamentally important to ecologists.

Ecosystems are often defined by fluctuations in abiotic conditions such as precipitation, temperature, nutrients, and light that generate variation in resource availability in time and space. It is generally thought that this variation causes temporal and spatial dynamics of species composition, consumer-resource interactions, and emergent food-web properties (Winemiller 1990). Seasonality is considered the most common temporal driver of trophic dynamics, however there is relatively little empirical work on how food webs respond to seasonal variation (Paine 1988, McMeans et al. 2015). Almost a century ago, Elton (1927) pointed out the importance of seasonal dynamics for animal communities and contemporary research demonstrates that spatial variation is a source of stability for food webs (Winemiller 1990, McCann et al. 2005). Nonetheless, most food webs have been treated as static (Poisot et al. 2015). While empirical evidence of the effects of seasonality on food webs is lacking, several examples demonstrate its influence on temporal shifts in trophic position and network structure

(Winemiller 1990, Olesen et al. 2008, Carnicer et al. 2009, McMeans et al. 2019). Recently, both terrestrial and aquatic ecologists have stated the need to explicitly consider spatiotemporal variation in food-web studies (Hampton et al. 2017, CaraDonna et al. 2017, McMeans et al. 2019).

Biotic interactions such as those caused by ecosystem engineers and invasive species further influence food-web structure and function (Sanders et al. 2014, David et al. 2017, Flood et al. 2020, Wainright et al. 2021). Facilitation by ecosystem engineers transcends trophic levels usually enhancing species density, diversity, and food-web structure (Sanders and van Veen 2011, van der Zee et al. 2016). Conversely, invasive species cause trophic disruption that often diminishes food-web structure and function and biodiversity (Wainright et al. 2021). These effects from invasive species can have critical, often unanticipated, impacts on ecosystems that may be difficult to detect (Simberloff 2011, Simberloff et al. 2013). Theoretically and empirically, the effects of both ecosystem engineers and invasive species on food webs and the consequences for ecosystems are not well understood (Sanders and van Veen 2011, Sanders et al. 2014, Flood et al. 2020).

Herein, I examine how trophic dynamics change among habitats and between seasons (spatiotemporal variation) in a historic, pre-invasion condition, how ecosystem engineers create and maintain novel habitats that influence trophic dynamics, and how the invasion of African Jewelfish (*Hemicrhomis letourneuxi*) has impacted trophic dynamics and energy flow relative to a historic baseline. In Chapter 2, I begin by quantifying trophic dynamics among habitats and between seasons in a historic condition of my study system, Everglades National Park (ENP), to establish a pre-invasion baseline. The Everglades is an oligotrophic wetland characterized by three major habitats: a slough/prairie system (spikerush-dominated marsh) comprising ~35% of the wetland area, sawgrass marsh (dense cover of *Cladium jamaicense*) that covers ~50% of the landscape, and alligator ponds (< 0.1 % coverage) (Loftus and Kushlan 1987, Gunderson and Loftus 1993, Davis et al. 1994). The Everglades also experiences marked wet and dry seasons and is governed by the flood pulse (Junk et al. 1989, Gaiser et al. 2012). I use complementary datasets of stomach contents and stable isotopes of carbon and nitrogen to describe consumers' diets, trophic niche, trophic position, and basal energy use and network structure in these habitats in both the wet and dry seasons.

Ecosystem engineering, through direct and trait-mediated indirect effects, interacts with environmental variability to influence heterogeneity in trophic dynamics (Wetzel et al. 2016). In the Everglades, American alligators (*Alligator misissppiensis*) dig and maintain alligator ponds through removal of vegetation and redistribution of sediment (Kushlan 1974, Campbell and Mazzotti 2004, Palmer and Mazzotti 2004). This results in two distinct habitats: the alligator pond and a ring of dense vegetation immediately surrounding the pond (Campbell and Mazzotti 2004, Palmer and Mazzotti 2004). These habitats have distinct floral and faunal communities and engineered habitats have elevated phosphorus levels (Strickland 2020). For Chapter 3, I investigate how alligator-engineering influences trophic dynamics and test the Stress Gradient Hypothesis (SGH) using complementary datasets of stomach contents and stable isotopes of carbon and nitrogen. The SGH states that the intensity of facilitative interactions (e.g., ecosystem engineering) and competitive interactions will vary inversely across a gradient of environmental stress (Bertness and Callaway 1994). I use trophic niches as a proxy for competition via the Niche Variation Hypothesis, which posits that niche area and competition are inversely related (van Valen 1965, Bolnick et al. 2007), and model the direction and magnitude of shifts in trophic niche size among these habitats and along an environmental stress gradient (elevated stress in the dry season).

Recent efforts have sought to assess ecosystem vulnerability and prioritize management through our ability to understand how biological invasions alter trophic dynamics (Catford et al. 2012, McDonald-Madden et al. 2016, Strassburg et al. 2020). Over the past several decades, over fifty species of fishes have been introduced into the freshwaters of Florida, seventeen of which have become established in ENP (Kline et al. 2013). Several years ago, after being established in ENP for over a decade, African Jewelfish underwent a drastic increase in density associated with sharp declines in native taxa (Pintar, Dorn, Trexler, pers. com.). In Chapter 4, I use complementary datasets of stomach contents and stable isotopes of carbon and nitrogen to test the Trophic Disruption Hypothesis (TDH). The TDH states that invasive species will cause native species to undergo trophic dispersion (altered diet variability – niche size) and trophic displacement (diet switching), eventually leading to altered food-web structure and biodiversity loss (Wainright et al. 2021). I test for post-invasion trophic disruption in diets, trophic niches, trophic positions, basal energy use, and energy fluxes to assess impacts on food-web and ecosystem function.

I conclude in Chapter 5 with a discussion of the implications of my work for understanding the effects of spatiotemporal variation, ecosystem engineers, and invasive species on food webs and ecosystem function. I provide insight from this work for managing invasive species, especially in regularly perturbed ecosystems, and I end with future research directions that my increase our understanding of these processes impact food webs and ecosystems.

Literature Cited

Bertness, M. D., and R. Callaway. 1994. Positive interactions in communities. Trends in Ecology and Evolution 9:191–193.

Bolnick, D. I., R. Svanbä, S. Araú Jo §, and L. Persson. 2007. Comparative support for the niche variation hypothesis that more generalized populations also are more heterogeneous.

Campbell, M. R., and F. J. Mazzotti. 2004. Characterization of natural and artificial alligator holes. Southeastern Naturalist 3:583–594.

CaraDonna, P. J., W. K. Petry, R. M. Brennan, J. L. Cunningham, J. L. Bronstein, N. M. Waser, and N. J. Sanders. 2017, March 1. Interaction rewiring and the rapid turnover of plant–pollinator networks. Blackwell Publishing Ltd.

Carnicer, J., P. Jordano, C. J. Melia´n, and M. Melia´n. 2009. The temporal dynamics of resource use by frugivorous birds: a network approach. Page Ecology.

Catford, J. A., P. A. Vesk, D. M. Richardson, and P. Pyšek. 2012, January. Quantifying levels of biological invasion: Towards the objective classification of invaded and invasible ecosystems.

David, P., E. Thébault, O. Anneville, P. F. Duyck, E. Chapuis, and N. Loeuille. 2017. Impacts of Invasive Species on Food Webs: A Review of Empirical Data. Advances in Ecological Research 56:1–60.

Davis, S. M., L. H. Gunderson, W. A. Park, J. R. Richardson, and J. E. Mattson. 1994. Landscape dimension, composition, and function in a changing Everglades ecosystem. Pages 419–444 *in* S. M. Davis and J. C. Ogden, editors. Everglades: the ecosystem and its restoration. St. Lucie Press, Delray Beach, FL.

Elton, C. 1927. Animal ecology. Macmillan, New York, New York, USA.

Flood, P. J., A. Duran, M. Barton, A. E. Mercado-Molina, and J. C. Trexler. 2020. Invasion impacts on functions and services of aquatic ecosystems. Hydrobiologia 847:1571–1586.

Gaiser, E. E., J. C. Trexler, and P. R. Wetzel. 2012. The Florida Everglades. Pages 231–252 *in* D. P. Batzer and A. H. Baldwin, editors. The Florida Everglades. University of California Press, Berkeley, CA.

Gunderson, L. H., and W. F. Loftus. 1993. The Everglades. Pages 199–255 *in* W. H. Martin, S. G. Boyce, and E. G. Echternacht, editors. Biodiversity of the Southeastern United States 2. John Wiley and Sons Inc., New York, New York, USA.

Hampton, S. E., A. W. E. Galloway, S. M. Powers, T. Ozersky, K. H. Woo, R. D. Batt, S. G. Labou, C. M. O'Reilly, S. Sharma, N. R. Lottig, E. H. Stanley, R. L. North, J. D. Stockwell, R. Adrian, G. A. Weyhenmeyer, L. Arvola, H. M. Baulch, I. Bertani, L. L. Bowman, C. C. Carey, J. Catalan, W. Colom-Montero, L. M. Domine, M. Felip, I. Granados, C. Gries, H. P. Grossart, J. Haberman, M. Haldna, B. Hayden, S. N. Higgins, J. C. Jolley, K. K. Kahilainen, E. Kaup, M. J. Kehoe, S. MacIntyre, A. W. Mackay, H. L. Mariash, R. M. McKay, B. Nixdorf, P. Nõges, T. Nõges, M. Palmer, D. C. Pierson, D. M. Post, M. J. Pruett, M. Rautio, J. S. Read, S. L. Roberts, J. Rücker, S. Sadro, E. A. Silow, D. E. Smith, R. W. Sterner, G. E. A. Swann, M. A. Timofeyev, M. Toro, M. R. Twiss, R. J. Vogt, S. B. Watson, E. J. Whiteford, and M. A. Xenopoulos. 2017, January 1. Ecology under lake ice. Blackwell Publishing Ltd.

Ings, T. C., J. M. Montoya, J. Bascompte, N. Blüthgen, L. Brown, C. F. Dormann, F. Edwards, D. Figueroa, U. Jacob, J. I. Jones, R. B. Lauridsen, M. E. Ledger, H. M. Lewis, J. M. Olesen, F. J. F. van Veen, P. H. Warren, and G. Woodward. 2009. Ecological networks - Beyond food webs. Blackwell Publishing Ltd.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The Flood Pulse Concept in River-Floodplain Systems. Canadian Special Publication of Fisheries and Aquatic Sciences Honey Harbour, Ontario, Canada 7:110–127.

Kline, J. L., W. F. Loftus, K. Kotun, J. C. Trexler, J. S. Rehage, J. J. Lorenz, and M. Robinson. 2013. Recent fish introductions into Everglades National Park: An unforeseen consequence of water management? Wetlands 34:S175–S187.

Kushlan, J. A. 1974. Observations on the role of the American Alligator in the Southern Florida Wetlands. Copeia 4:993–996.

Lindeman, R. L. 1942. The Trophic-Dynamic Aspect of Ecology. Ecology 23:399–417.

Loftus, W. F., and J. A. Kushlan. 1987. Freshwater fishes of southern Florida. Bulletin of the Florida State Museum, Biological Sciences 31:147–344.

McCann, K. S., J. B. Rasmussen, and J. Umbanhowar. 2005. The dynamics of spatially coupled food webs. Ecology Letters 8:513–523.

McDonald-Madden, E., R. Sabbadin, E. T. Game, P. W. J. Baxter, I. Chadès, and H. P. Possingham. 2016. Using food-web theory to conserve ecosystems. Nature Communications 7.

McMeans, B. C., T. Kadoya, T. K. Pool, G. W. Holtgrieve, S. Lek, H. Kong, K. Winemiller, V. Elliott, N. Rooney, P. Laffaille, and K. S. Mccann. 2019. Consumer trophic positions respond variably to seasonally fluctuating environments. Ecology 100:1–10.

McMeans, B. C., K. S. McCann, M. Humphries, N. Rooney, and A. T. Fisk. 2015. Food web structure in temporally-forced ecosystems. Trends in Ecology and Evolution 30:662–672.

Montoya, J. M., S. L. Pimm, and R. v. Solé. 2006. Ecological networks and their fragility. Nature 442:259–264.

Olesen, J. M., J. Bascompte, H. Elberling, and P. Jordano. 2008. Temporal dynamics in a pollination network. Ecology 89:1573–1582.

Paine, R. T. 1988. Road maps of interactions or grist for theoretical development? Ecology 69:1648–1654.

Palmer, M. L., and F. J. Mazzotti. 2004. Structure of Everglades alligator holes. Wetlands 24:115–122.

Poisot, T., D. B. Stouffer, and D. Gravel. 2015. Beyond species: Why ecological interaction networks vary through space and time. Oikos 124:243–251.

Rooney, N., and K. S. McCann. 2012. Integrating food web diversity, structure and stability. Trends in Ecology and Evolution 27:40–46.

Sanders, D., C. G. Jones, E. Thébault, T. J. Bouma, T. van der Heide, J. van Belzen, and S. Barot. 2014. Integrating ecosystem engineering and food webs. Oikos 123:513–524.

Sanders, D., and F. J. F. van Veen. 2011. Ecosystem engineering and predation: The multi-trophic impact of two ant species. Journal of Animal Ecology 80:569–576.

Simberloff, D. 2011. How common are invasion-induced ecosystem impacts? Biological Invasions 13:1255–1268.

Simberloff, D., J. L. Martin, P. Genovesi, V. Maris, D. A. Wardle, J. Aronson, F. Courchamp, B. Galil, E. García-Berthou, M. Pascal, P. Pyšek, R. Sousa, E. Tabacchi, and M. Vilà. 2013, January. Impacts of biological invasions: What's what and the way forward.

Strassburg, B. B. N., A. Iribarrem, H. L. Beyer, C. L. Cordeiro, R. Crouzeilles, C. C. Jakovac, A. Braga Junqueira, E. Lacerda, A. E. Latawiec, A. Balmford, T. M. Brooks, S. H. M. Butchart, R. L. Chazdon, K. H. Erb, P. Brancalion, G. Buchanan, D. Cooper, S. Díaz, P. F. Donald, V. Kapos, D. Leclère, L. Miles, M.

Obersteiner, C. Plutzar, C. A. Carlos, F. R. Scarano, and P. Visconti. 2020. Global priority areas for ecosystem restoration. Nature 586:724–729.

Strickland, B. A. 2020. Beyond Predation: How Do Consumers Mediate Bottom-up Processes in Ecosystems? Florida International University, Miami, FL.

van Valen, L. 1965. Morphological Variation and Width of Ecological Niche. Source: The American Naturalist 99:377–390.

Wainright, C. A., C. C. Muhlfeld, J. J. Elser, S. L. Bourret, and S. P. Devlin. 2021. Species invasion progressively disrupts the trophic structure of native food webs. Proceedings of the National Academy of Sciences of the United States of America 118.

Wang, S., and U. Brose. 2018. Biodiversity and ecosystem functioning in food webs: the vertical diversity hypothesis. Ecology Letters 21:9–20.

Wetzel, W. C., R. M. Screen, I. Li, J. Mckenzie, K. A. Phillips, M. Cruz, W. Zhang, A. Greene, E. Lee, N. Singh, C. Tran, and L. H. Yang. 2016. Ecosystem engineering by a gall-forming wasp indirectly suppresses diversity and density of herbivores on oak trees. Page Ecology.

Winemiller, K. O. 1990. Spatial and temporal variation in tropical fish trophic networks. Ecological Monographs 60:331–367.

van der Zee, E. M., C. Angelini, L. L. Govers, M. J. A. Christianen, A. H. Altieri, K. J. van der Reijden, B. R. Silliman, J. van de Koppel, M. van der Geest, J. A. van Gils, H. W. van der Veer, T. Piersma, P. C. de Ruiter, H. Olff, and T. van der Heide. 2016. How habitat-modifying organisms structure the food web of two coastal ecosystems. Proceedings of the Royal Society B: Biological Sciences 283.

CHAPTER II

FISHES IN A SEASONALLY PULSED WETLAND SHOW SPATIOTEMPORAL SHIFTS IN DIET AND TROPHIC NICHE BUT NOT SHIFTS IN TROPHIC POSITION

Abstract

I examine temporal (seasonal) and spatial (habitat) effects on consumers' diet, trophic position, trophic niche, and food-web topology in a subtropical oligotrophic wetland to illustrate how consumers and food webs respond to hydrologic pulsing in a spatially complex ecosystem. I ask if the annual flood pulse causes fishes to undergo a trophic shift or if fishes maintain a constant diet, trophic position, and trophic niche all year and across habitats, as is often assumed. Furthermore, I ask if the flood pulse alters food-web topology in different habitats and if invasive fishes overlap in trophic niche with native fishes in this ecosystem. I found that trophic dispersion (shift in trophic-niche size) was common (66% and 71% of spatial and temporal comparisons respectively), trophic displacement (shift in trophic-niche location in niche space) was ubiquitous (spatial – 92%; temporal – 82%), and shifts in trophic position were relatively rare (spatial – 11%; temporal – 4%). Trophic dynamics were primarily driven by differing amounts of piscivory, detrital consumption, and diet plasticity across habitats and seasons. In the dry season, food-web topology indicated decreased complexity in all habitats (decreased number of links, link density, and connectance) and instability in ponds that may facilitate invasions. Both stomach contents and stable isotopes revealed trophic-niche overlap among native and invasive fishes, notably centrarchids and cichlids. Diverse, flexible trophic responses to seasonality across habitats may be pivotal to nutrient and energy cycling and in maintaining ecosystem stability and resilience, especially in regularly perturbed environments. Seasonal fluctuation typical of wetlands may require inter-habitat relocation, leading to the types of food-web changes I

document. I conclude that spatiotemporal trophic plasticity is probably common and deserves additional study given its ability to influence food-web structure and function.

Introduction

Fluctuations in abiotic conditions such as precipitation, temperature, nutrients, and light drive variation in resource availability across habitats and seasons. It is widely assumed that this variation drives temporal and spatial dynamics of species composition, consumer- resource interactions, and emergent food-web properties (Winemiller, 1990). Seasonality is probably the most common temporal driver of food-web structure, yet little is known about how food webs respond to seasonal variation (McMeans et al., 2015; Paine, 1988). Elton (1927) noted the importance of seasonal dynamics to animal communities almost a century ago, and contemporary work demonstrates that spatial variation in food webs is a stabilizing force (McCann et al., 2005; Winemiller, 1990).

Despite this recognition, most food webs have been treated as static (Poisot et al., 2015). While empirical evidence on the influence of seasonality on food webs is lacking, some examples demonstrate its importance to trophic position and network structure (Carnicer et al., 2009; McMeans et al., 2019; Olesen et al., 2008; Winemiller, 1990). Recently, both terrestrial and aquatic ecologists have recognized the need to incorporate seasonality in food-web studies (CaraDonna et al., 2017; Hampton et al., 2017; McMeans et al., 2019).

Habitat heterogeneity influences food-web structure by creating resource asynchronies across the landscape. In response, many consumers exhibit flexible foraging among habitats, increasing food-web stability (McCann et al., 2005). Moreover, in the absence of spatially adaptive foragers, increased food-web complexity destabilizes community composition (Kondoh, 2003). This theory – that spatial heterogeneity weakens interaction strength and increases food-web stability – has garnered empirical support in below-ground food webs, streams, and river floodplains (Bellmore et al., 2015; Moore and de Ruiter, 1991; Thompson and Townsend, 2005). Much of that work implies that food webs are static.

Recent studies have demonstrated inter-annual or seasonal changes in trophic position and omnivory, often associated with shifts in habitat use (Akin and Winemiller, 2006; Heng et al., 2018; McMeans et al., 2019; Ruiz-Cooley et al., 2017). These trophic dynamics among habitats have been recognized as important for food-web structure and function (Cross et al., 2013; McCann et al., 2005; Thompson et al., 2007). Many fishes are adaptive foragers that undergo seasonal diet changes both within and among habitats (Jepsen et al., 1997; Lowe-McConnell, 1969; McMeans et al., 2019; Winemiller, 1989; Winemiller and Jepsen, 1998). Temporal shifts in diet, in which consumers adaptively track different resources (Kratina et al., 2012; Křivan and Diehl, 2005; Takimoto et al., 2002), may have consequences for ecosystems as important as those recognized for spatial diet shifts.

Aquatic ecosystems with marked wet and dry seasons governed by the flood pulse (Junk et al., 1989) offer excellent opportunities to evaluate impacts of seasonal variation on food webs. Rising waters during the wet season increase available habitat that supports primary and secondary production (Arias et al., 2013; DeAngelis and White, 1994; Junk et al., 1989; Trexler et al., 2005). Alternatively, drying concentrates fish and invertebrate biomass, leading to increased foraging efficiency for higher-level consumers (McConnell and Lowe-McConnell, 1987; Winemiller, 1989). Such resource pulses have been appreciated as driving forces in ecosystems for several decades (Odum, 1969; Odum et al., 1995) and may reverse ecosystem declines caused by disturbance and climate change (Kominoski et al. 2018). More recently, pulses of detritus have been found to be similarly important (Moore et al., 2004; Rooney and McCann, 2012; Yang et al., 2008). As a result, it is unsurprising that in pulsed, tropical systems both inter- and intraspecific dietary niche overlap reveal spatiotemporal shifts with mixed trends among studies (Costa-Pereira et al., 2017; Lowe-McConnell, 1964; Moyle and Senanayake, 1984; Zaret and Rand, 1971). Moyle and Senanayake (1984) and Payne (1986) both found dietary niche overlap to be greatest when resources were most abundant. Similarly, Quirino et al. (2017) showed the dietary niche for a population of small characids was larger in the wet season due to abundant allochthonous resources. Conversely, other studies have demonstrated that dietary niche overlap was highest during the dry season while resources were limiting (Lowe-McConnell, 1969; Power, 1983; Prejs and Prejs, 1987). Correa and Winemiller (2014) found no consistent seasonal trend for dietary niche among Amazonian fishes. Seasonal shifts in dietary niche are likely specific to feeding behavior (Azevedo et al., 2021) and may explain mixed trends among studies. Further research in additional ecosystems with temporal and spatial hydrological pulsing is needed to advance our understanding of the impact of seasonality on dietary niche.

I evaluate the impact of flood-pulse seasonality on food webs in the Everglades, USA, a subtropical, seasonally pulsed wetland. Basal resources of the Everglades food web are dominated by algal and detrital energy routes that drive spatiotemporal food-web dynamics (Belicka et al., 2012; Trexler et al., 2015; Williams and Trexler, 2006). I quantified temporal (seasonal) and spatial (among habitats) food-web dynamics in this oligotrophic wetland using stomach contents and stable isotopes of carbon and nitrogen. I calculated diet, niche breadth, and food-web structure both within and among populations and communities across habitats and seasons. I predicted that diets and niche breadths would follow similar trends within trophic guilds because of similar feeding behaviors among species (Azevedo et al., 2021), that omnivory should be common in both seasons based on previous studies in flood-pulsed ecosystems (Heng et al., 2018, McMeans et al., 2019), and that niche area and overlap among species size classes would increase during the dry season as habitats contract and resources become limited (Lowe-McConnell, 1969, McHugh et al., 2015, Power, 1983). Furthermore, I predicted that food-web complexity, measured as connectance, would decrease during the dry season in shallow marsh habitats as species move to deep-water habitats in response to seasonal drying, and predicted the opposite for deep-water habitats where animals are concentrated in the dry season (donor control; Strong, 1992). I also investigated the relative importance of detrital energy routing across the same gradient and hypothesized that detritus will be consumed more in the dry season when alternate resources are limited, forcing species to use lower quality food. I report stable-isotope mixing models (SIMMs; Parnell et al., 2013) both with and without informative priors derived from stomach contents to test prey assimilation to complement prey-ingestion data from stomach contents. Finally, I discuss the implications of these results for ecosystem conservation and future work, particularly regarding species invasions that have occurred following the collection of our data.

Methods

2.1 Study Area

The study site was a one-square kilometer area in north-central Shark River

Slough, Everglades National Park (ENP), Florida, USA, at 25°38.090 N and 80°43.720 W, referred to in previous work as site Upper Slough (Loftus and Eklund, 1994), and site SRS 6 (Gatto et al., 2021; Gatto and Trexler, 2019; Ruetz et al., 2005; Trexler et al., 2002). The Everglades experiences wet (June through November; flood pulse) and dry (December through May; flood-pulse recession) seasonal rainfall resulting in water depths that are shallowest in late spring and deepest in late fall (Fig. A.1.1, Table A.1.1, Loftus and Kushlan, 1987). The study area included three major aquatic habitats (Gunderson and Loftus, 1993): a slough/prairie system (hereafter "spikerush marsh" or "spikerush") comprising about 35% of the wetland area, dominated by a relatively dense cover of *Eleocharis cellulosa*, *Panicum hemitomon*, and *Utricularia purpurea* and *U. foliosa*, a floating periphyton mat and some patches of white waterlily pads (*Nymphaea odorata*), with a peat substrate; sawgrass marshes, densely covered by *Cladium jamaicense* and comprising about 50% of the area (Davis et al., 1994); and alligator ponds, often surrounded by *Salix caroliniana*, C. *jamaicense, Pontedaria cordata,* and *Typha* spp., with open water over a peat or limestone bottom. The small areal coverage of ponds (< 0.1 % of the area) is contrasted by their ecological significance. In most years, alligator ponds hold water through the dry season, acting both as refuges and sinks for various species of fishes (Loftus and Kushlan, 1987).

2.2 Field Collections

Fishes and invertebrates analyzed for this project were collected between 1977 and 1994. Most specimens (> 90%) were collected between 1977 and 1981, but supplemental collections were taken from the same study area at the same time of year during the mid-1990s to increase sample sizes for less common species, mainly larger predators. Collections in the mid-1990s included non-native species that had colonized the Everglades (see Kline et al., 2014 for timeline), and all specimens for stable-isotope analysis. Wet-season samples were collected between January and early March, and dry-season samples in April. Most fishes were collected with 5% rotenone solution (Nox-Fish®), along with electrofishing, cast nets, dip nets, and angling. All fishes were preserved in 10% buffered formalin and transferred to 70% ethanol, except those captured for isotopic analysis, which were euthanized with MS-222 and frozen. In total, I collected 3,509 individuals from thirty-two species, of which 600 had empty stomachs (see Arrington et al., 2002). I analyzed stomach contents for the remaining 2,909 fishes, 83% of those collected. The target sample size was twenty-five individuals per species size class per habitat-season level and the average was sixteen (Table 1).

2.3 Laboratory Analyses

I separated species into body-size groups based on length, termed species size classes (Table A.1.2), to examine the role of ontogeny in trophic resource partitioning within and among species, because most fishes undergo a change in diet with growth (Wainwright and Richard, 1995). The incorporation of ontogenetic changes (i.e. changes that occur as an organism grows and develops) in diet studies helps alleviate biases associated with niche-overlap and diet-breadth indices (Piet et al., 1999). I recorded the length and wet mass of each specimen before removing the stomach. In species without defined stomachs, I removed the digestive tract from the esophagus to the first bend in the tract. Each stomach was flushed of contents which were examined with a Wild®5A dissecting microscope and identified to the lowest-level taxon possible. Fragmented remains of insects, fishes, and crustaceans were termed “unidentifiable.” Insect taxa are larval forms, unless otherwise indicated. The number of individuals per prey type were recorded for individual consumers (Hyslop, 1980).

For stable-isotope samples, I dissected muscle from small fishes and invertebrates for drying at 50 °C, then pulverized it (Fry, 2006). Samples of major primary producers were dried and powdered after being acid-washed in HCl before drying to remove carbonates. Three to five individuals were analyzed to produce a mean value per taxon. The samples were analyzed shortly thereafter in the mid-1990s at the University of Georgia Institute of Ecology isotope laboratory. The isotopic standards used for $\delta^{13}C$ and $\delta^{15}N$ were Pee Dee Belemnite (PDB) and air, respectively. Average isotopic error of replicate standards was $\delta^{15}N \leq 0.2$ ‰ and $\delta^{13}C \leq 0.1$ ‰.

2.4 Community Dynamics in Prey Consumption

I analyzed changes in the numeric stomach-content diet matrix across habitats and seasons. Spatiotemporal variation in prey consumption by the fish community was visualized with non-metric multidimensional scaling (NMDS, $k = 2$) using Morisita-Horn distances, which rely on relative abundance of taxa to avoid disproportionate influence of changes in absolute abundance when relative abundance remains the same (Jost et al., 2011). Visual differences among communities from different habitat-season levels were statistically assessed using permutational multivariate analysis of variance (PERMANOVA; 10,000 iterations). PERMANOVA can yield significant results for two reasons: differences in dispersion in multivariate space or differences in centroid location in multivariate space (Anderson and Walsh, 2013). I specifically tested for differences in multivariate dispersion (PERMDISP) by using the *betadisper* function and analyzing results using analysis of variance (ANOVA) followed by Tukey's HSD. Similarity percentages (SIMPER) were used to determine diet items contributing the most variance among habitat-season levels (Fig. 1). Analyses were done in R version 3.6.3 using the *vegan* package or base R (Oksanen et al., 2020; R Core Team, 2020). Additionally, I asked if the occurrence of detritus in diets of all consumers changed across habitats and seasons. To answer this question, I performed a one-way permuted ANOVA (10,000 iterations) to compare the amount of vascular detritus in stomachs among habitat-season levels followed by a pairwise permutation test (functionally like Tukey's HSD). These were conducted using *independence_test* and *pairwisePermutationTest* functions in *coin* and *rcompanion* packages respectively (Mangiafico, 2021; Zeileis et al., 2008). The relationship between these analyses and all subsequent analyses are summarized in Fig. 1.

2.5 Trophic Position

Trophic position was calculated using both stomach-content and stable-isotope data. To estimate trophic positions of species size classes based on stomach contents, I followed Adams et al. (1983), where trophic position of a predator is defined by the trophic levels of its prey (Appendix A.1.5, Eq. A.1, Table A.1.3). Trophic position distributions for each species size class at each habitat-season level were tested for normality using Shapiro-Wilk tests. Differences among size classes and among habitat-season levels were assessed using permuted ANOVA (10,000 iterations) via the *aovp* function from the *lmPerm* package in R (Wheeler and Torchiano, 2016). Species size class, habitat-season level, and the interaction of the two were included as fixed effects.

Trophic position per species was modeled using stable isotopes of carbon and nitrogen in the *tRophicPosition* package in R (Quezada-Romegialli et al., 2018). This Bayesian framework estimates posterior distributions for trophic position and a metric called alpha. Alpha is the proportion of $\delta^{15}N$ in a consumer's tissue derived from baseline one, which in this case was detritus. The Everglades aquatic food web has two baselines, green algae and detritus (Williams and Trexler, 2006). Therefore, alpha minus one is the proportion of $\delta^{15}N$ in a consumer's tissues derived from baseline two, green algae. I compared trophic position and alpha among species by calculating the probability one species had a trophic position or alpha less than or equal to the same metric for a different species for all possible pairwise comparisons (Fig. 1). I report the Gelman-Rubin diagnostic, values near one indicate convergence, for each model parameter (Gelman and Rubin, 1992).

2.6 Dietary Proportions

The percent contribution of diet items to a consumer was calculated for individuals using numeric stomach contents and for species using stable isotopes when data permitted. Dietary proportions were calculated using the 8-group stomach-content matrix (prey groups are the functional groups from Table A.1.3) for each individual. Then, the mean and standard deviation were calculated for each species size-class across habitats and seasons. Furthermore, I generated stable-isotope mixing models (SIMMs) using the *simmr* package in R to reconstruct the diet for each species as data permitted (Appendix A.1.6, Parnell and Inger, 2016; Parnell et al., 2010). Whenever possible, SIMMs were generated both with and without informative priors based on stomach contents (Fig.1, Appendix A.1.6, Table A.1.4). Trophic enrichment factors used for each source group follow those from McCutchen et al. (2003) based on the type of source and source tissue.

2.7 Ontogenetic Shifts in Diet

Many species of fishes undergo ontogenetic shifts in diet that are likely influenced by environmental conditions. Differences among species size classes, habitat-season levels, and the interaction of those factors were examined using PERMANOVA via the *adonis* function from the *vegan* package in R (Oksanen et al., 2020). Afterwards, PERMANOVAs were performed for all possible pairwise comparisons using the *pairwise.perm.manova* function in the *RVAideMemoire* package in R (Hervé, 2021). I summarized ontogenetic shifts by size class for each habitat-season level and shifts among habitat-season levels for each species size class (Fig. 1). Size classes are denoted in text in parentheses after the species (e.g., *Gambusia holbrooki* (1)).

2.8 Dietary Niche

Dietary-niche metrics were calculated for species size classes using stomach contents among habitat-season levels and for species using stable-isotope data. Trophic niches derived from stomach contents and stable isotopes may represent different aspects of the trophic niche, and it is important to denote which is being discussed (Petta et al., 2020). I use the term niche, for the sake of brevity, referring exclusively to trophic niche, and specify whether it is derived from stomach contents or stable isotopes. There is no spatiotemporal component to the isotope dataset (isotope samples were collected only from wet-season spikerush), so all such trends in niche are referring to trophic niche derived from stomach contents (Fig. 1).

Stomach-content niche metrics and stable-isotope niche metrics were calculated separately using the *SIBER* package in R that generates niche areas as ellipses (standard ellipse area – SEA_b) in a Bayesian framework (Jackson et al., 2011). The *SIBER* package was designed with stable isotopes of carbon and nitrogen in mind. However, it can accommodate any appropriate two-dimensional data set. For our stomach-content data, I used NMDS axes one and two. Stable isotopes of carbon and nitrogen act as a two-dimensional ordination of diet that is found in nature. Therefore, axes from our ordination of stomach contents are an appropriate analog (Appendix A.1.8). I extracted modes and credible intervals from posterior distributions for all groups (species size-classes) within each community (habitat-season) using stomach contents and for each species using stable isotopes (Fig. 1).

Trophic niches can change in two ways: trophic dispersion (changing diet variability – SEA_b area) and trophic displacement (diet switching – SEA_b location in niche space; Cucherousset et al., 2012, Vander Zanden et al., 1999, Wainwright et al., 2021). Differences in trophic dispersion among groups were determined by calculating the probability that group A's SEA_b posterior distribution was less than group B's for ontogenetic and spatiotemporal pairwise comparisons (Appendix A.1). Trophic displacement was calculated as the proportion of overlapping SEA_b between groups. Proportion of overlap was defined as the area of overlap between groups ($A \cap B$) divided by the total combined area of both groups ($A \cup B$).

I described dispersion in two-dimensional space using the Layman metrics via the Bayesian framework within *SIBER* (Jackson et al., 2011; Layman et al., 2007). These metrics were also developed using stable-isotope data; however, they are equally applicable to describing niche space derived from our two-dimensional NMDS. Layman metrics were calculated using *extractPosteriorMeans* and *bayesianLayman* functions in *SIBER* (Jackson et al., 2011). Comparisons of Layman metrics among populations within a habitat-season level and among communities across habitat-season levels were calculated as the probability that one distribution was less than another in the same manner as SEA_b.

2.9 Network Structure

Describing food-web topology can provide insight into food-web functioning (Kones et al., 2009). I sought to calculate topological attributes with well-understood biological meanings for food webs of each habitat-season level using the *foodweb* package in R (Perdomo et al., 2012). Diet information for taxa I did not directly study (e.g., invertebrates found in stomachs) were assembled using literature (Rader, 1994) and expert opinion (Loftus, Trexler), allowing us to create a symmetric presence/absence matrix for analysis. Some diet categories (e.g., Miscellaneous and Miscellaneous Insecta) were removed from the analysis because they are low-resolution trophic interactions already represented by more defined nodes. Afterwards, I filtered the dataset to isolate data from each habitat-season level into their own matrices. Then I used the *analyse.seq* function to calculate the following food-web network metrics: taxa richness, number of trophic links, link density, connectance, number of omnivores (defined as species feeding at multiple trophic levels), mean chain length, maximum chain length, number of basal taxa, number of intermediate taxa, number of top taxa, and prey to predator ratio (Perdomo et al., 2012).

Results

3.1 Community Dynamics in Prey Consumption

I sought to understand spatiotemporal fluctuation in resource use by quantifying community-wide changes in relative consumption of different prey among habitats and seasons. Spatiotemporal variation in prey communities was driven by increased consumption of larger prey (*Palaemonetes paludosus* and fishes), particularly in dry-season ponds. NMDS ($k = 2$, stress $= 0.17$) separated consumers by trophic position and prey size (Fig. 2). MDS 1 was a gradient from small diet items (relatively small invertebrates such as zooplankton had lower values) to large diet items (dragonfly larvae, shrimp, crayfish, and fishes had larger values). Meanwhile, MDS 2 represented a gradient of fish size from first-order consumers to top predators. As a result, variation among habitats and seasons along MDS 1 is driven by changes in relative abundance of consumed prey, while variation on MDS 2 is the result of changes in relative abundance of consumers. For instance, in all habitats in the dry season, fishes consumed a wider range of prey (wider range on MDS 1; Fig. 2). PERMANOVA demonstrated spatiotemporal differences in communities of consumed prey ($F_2 = 15.3$, $p < 0.001$; Table A.2.1a), while comparisons of multivariate dispersion (PERMDISP) showed differences in three habitat-season comparisons ($p < 0.05$, Table A.2.1b) contributing to PERMANOVA results. In spikerush and sawgrass SIMPER found similarities in consumed prey within and between seasons, while ponds differed from other habitats and between seasons (Fig. 3, Table A.2.2).

3.2 Diet Descriptions

Our analyses revealed that consumers filled several functional feeding groups with omnivory being the most common, vascular plants were rarely consumed, and detritus played an important role for a variety of consumers (Table A.2.3). Based on stomach contents, most fishes collected in ponds were omnivores, but dry-season water recession often resulted in increased carnivory and detrital consumption. Omnivorous invertebrates, excluding decapods, were most often the diet group that constituted the largest proportion of an individual's stomach contents (Table A.2.3). These omnivorous invertebrates were the most important diet item in each habitat-season level, except for dry-season ponds where decapods and detritus played more important roles. SIMMs confirmed the relative importance of different functional groups to consumer diets with some exceptions (Table A.2.4). Relative to stomach contents, SIMMs suggested elevated consumption of larger prey items (i.e., decapods and fishes) and decreased consumption of detritus (Table A.2.5). Higher quality prey items (decapods, fishes) seem to be assimilated more readily than lower quality prey items (detritus) relative to amounts ingested.

3.3 Basal Resource Use

I predicted that the food web would be detritally based, and that detrital consumption would increase in the dry season as resources become more limited. Our stable-isotope analyses confirmed that first prediction, and stomach contents demonstrated spatiotemporal variation in detrital consumption. Alpha, the proportion of $\delta^{15}N$ from detritus, ranged from 0.079 in mayfly larvae to 0.997 in *Procambarus fallax* (Fig. 4). Alpha values indicated that the ecosystem was detritally based, having twenty-nine taxa (88%) with an alpha greater than 0.5, including leafy bladderwort, *Utricularia foliosa*, a carnivorous vascular plant. Pond consumers consistently had more detritus in their gut contents than did consumers from spikerush and sawgrass habitats. One-way permuted ANOVA revealed a statistical difference in detrital consumption (maxT = 16.0, $p < 0.001$), and pairwise permutation tests revealed that 66% of species size classes shifted in detrital consumption among habitat-season levels (Table A.2.6). In the dry season, pond consumers ingested twice as much detritus as in the wet season. Spikerush and sawgrass marshes only differed in detrital consumption during the wet season when spikerush consumers had twice as much detritus in their guts on average.

3.4 Trophic Position

Trophic positions derived from stomach contents had a narrower range than those from stable isotopes. Trophic positions among size classes of Everglades fishes for groups where $n \geq 5$ derived from stomach contents ranged from 1.94 for *Heterandria formosa* (1) in dry-season ponds ($n = 5$) to 3.11 for the invasive *Mayaheros urophthalmus* (1) in wet-season spikerush ($n = 6$; Table A.2.7). Meanwhile, trophic positions modeled using stable isotopes ranged from 2.17 for larval *Caenis* spp. (Ephemeroptera) to 4.60 for *Micropterus salmoides* (Fig. 4). Unlike trophic positions calculated from stomach-content numeric data, several invertebrate taxa were included in the stable-isotope dataset. The range of trophic positions derived from stomach contents (1.17) was roughly half that of stable isotopes (2.43).

3.5 Niche Breadth

I predicted that changes among habitats and seasons in trophic-niche areas would be similar within trophic guilds, that sawgrass and spikerush would have a higher frequency of larger niche areas than ponds, and that niche areas would increase in the dry season. Niche breadth (SEA_b) derived from stomach contents demonstrated spatiotemporal variation in frequency of niche size (relative generalization to specialization). For stomach contents, SEA_b varied from 6.31×10^{-4} for *Lepomis marginatus* in wet-season ponds to 2.55 for *L. punctatus* (1) in wet-season spikerush (Table A.2.8). Overall, modal SEA_b was more variable among species size classes during the wet season, and all trophic guilds demonstrated a range of niche areas with no clear spatiotemporal trends based on trophic guild (Fig. 5: (all) x Dry and (all) x Wet). Ponds had the highest frequency of small niche areas in the wet season while sawgrass had the lowest. However, dry-season ponds and sawgrass had similar frequencies of small niche areas among consumers. The range of niche areas more than doubled in sawgrass during the dry season relative to the wet season (Fig. 5). Meanwhile, SEA_b based on stable isotopes ranged from 0.03 for mysid shrimp (*Taphromysis louisianae*) to 7.38 for *U. foliosa* and showed differences among functional groups of consumers (Table A.2.9 Fig. A.2.1, Fig. A.2.2).

Community-wide trophic-niche area (i.e., Layman metrics) for habitats and seasons revealed spatiotemporal changes in trophic diversity (CD) and trophic redundancy (NND). I expected these to be highest in ponds (deeper habitats) and decrease in the dry season in all habitats. Range along MDS 1 (gradient of diet item size) was highest in ponds during both seasons and lowest in spikerush during both seasons and dry-season sawgrass (Fig. 6). Conversely, all habitat-season levels had similar ranges along MDS 2 (gradient of consumer size). This suggests that among habitat-season levels differential prey consumption contributed more to trophic dynamics than differences in the relative abundance of consumers. Trophic diversity (TA) and mean trophic diversity (CD) were greatest in ponds with similar trends among habitat-season levels. Seasonal trends in spikerush were opposite those in ponds and sawgrass. In contrast, both trophic redundancy (NND) and evenness of trophic niches (SDNND) did not differ among habitat-season levels.

3.6. Non-Native vs Native Niche Overlap

I predicted that niche areas of non-native species would overlap with native species, especially non-native cichlids and native centrarchids. I found notable niche overlap between native and non-native species, particularly *M. urophthalmus* (Cichlidae). In total, I calculated the proportion of overlapping niche area for seventy-nine pairwise comparisons, forty-one from stomach contents and thirty-eight using stable isotopes (Table A.2.10). For stomach contents, proportional overlap ranged from 1.6% (*Clarias batrachus* vs. *A. natalis* (2), dry-season ponds) to 22.9% (*C. batrachus* vs. *L. punctatus* (3), dry-season ponds). In 10% of comparisons, proportional overlap exceeded 10% and most of these involved the native sunfish, *L. punctatus*. By contrast, stable isotopes revealed higher proportional overlap ranging from 0.34% (*P. mariae* vs. *L. platyrhincus*) to 64.0% (*M. urophthalmus* vs. *L. macrochirus*), with 21% of comparisons exceeding 10% overlap (Table A.2.10).

3.7 Ontogenetic Shifts in Trophic Dynamics

I predicted that most species would undergo ontogenetic diet shifts. There were thirty-eight instances of ontogenetic shifts in diet within a habitat-season level resulting from changes in carnivory (55% of observed ontogenetic shifts; often increased piscivory and decapod consumption with increasing size), diet plasticity (29%; similar mean contributions from prey categories between size classes, but different prey items within a category), herbivory (24%), and detritivory (21%). PERMANOVAs showed that eleven species (Table 2) underwent ontogenetic shifts (at least one size class's prey community differed, $p < 0.05$) in at least one habitat-season level (Table A.2.11). Ontogenetic shifts in diet were usually associated with trophic dispersion (90%) and trophic displacement was common (57%) (Fig. A.2.3, Table 2, A.2.12). A similar proportion of species underwent ontogenetic shifts in each habitat-season level.

3.8 Trophic Dynamics in Space and Time

3.8.1 Spatial Shifts in Trophic Dynamics

Spatial trophic shifts were ubiquitous and more common for trophic dispersion (66%) and displacement (92%) than trophic position (11%; Fig. 7, A.2.4, Table 3, A.2.13, and A.2.14). Trophic dispersion and displacement among habitats were more common in the wet season than the dry season. Changes in trophic dispersion and displacement were driven by differing amounts of diet plasticity (68% of comparisons), carnivory (33%), piscivory (33%), and to a lesser extent detritivory (18%) and herbivory (13%) (Table A.2.13, A.2.14). Diet plasticity was defined as a consumer eating similar proportions of prey items from different functional groups (Table A.8), while having a shift in trophic position and/or trophic niche. This indicated consumption of different prey items from the same functional group between habitats. Multiple drivers may be responsible for a given comparison, so percentages do not sum to one. There were only six species size classes that had statistically different trophic positions between habitats (Table 3). Carnivory (66%), piscivory (50%), and diet plasticity (50%) were the most important drivers of shifts in trophic position. Detritivory contributed to one spatial shift in trophic position while herbivory contributed to none.

3.8.2 Temporal Shifts in Trophic Dynamics

Temporal trophic shifts were commonplace and more likely for trophic dispersion (71%) and displacement (82%) than trophic position (4%; Fig. 7, A.2.5 Table 3, A.2.13, A.2.14). Seasonal trophic displacement was more common than trophic dispersion in spikerush (89% vs 63% of comparisons) and sawgrass (100% vs 83%), while trophic displacement (65%) and dispersion (71%) occurred at a similar frequency in ponds. Temporal shifts in trophic dispersion and displacement were driven by changes in relative amounts of piscivory (56%), diet plasticity (48%), carnivory (37%), detritivory (23%), and herbivory (13%). There were only two seasonal changes in trophic position driven by diet plasticity and piscivory respectively.

3.9 Network Structure

I predicted that during the dry season, food-web complexity would decrease in marshes and increase in ponds as aquatic species respond to water recession. Network topology differed among habitat-season levels with variable magnitudes and directions (Table 4). Taxon richness was highest in wet-season ponds and lowest in dry-season sawgrass. During the dry season, all habitats displayed a decrease in total number of links, link density, connectance, and proportion of omnivores and an increase in proportion of basal resources and prey-to-predator ratio. I observed mixed seasonal trends among habitats for proportion of herbivores that increased in ponds and sawgrass during the dry season yet decreased in spikerush. Similarly, proportion of cannibalism, intermediate consumers, and top consumers all showed mixed seasonal trends among habitats. Cannibalism (defined here as a consumer feeding within its own trophic guild, Perdomo et al., 2012) increased in sawgrass and decreased in ponds and spikerush during the dry season. In the dry season, intermediate consumers increased in ponds but decreased in spikerush and sawgrass. Proportion of top consumers roughly halved in ponds during the dry season, with spikerush and sawgrass showing no seasonal change.

Discussion

Spatial variation in resource profitability is important for ecosystem resilience and dynamic stability of fluctuating ecosystems (Leigh et al., 2010; Pettit et al., 2017). Our results that include many food-web metrics from a seasonally flood-pulsed ecosystem support the hypothesis that food webs are highly variable in space and time and that this variability is likely an underappreciated aspect of maintaining energy and material flows in dynamic ecosystems (McMeans et al. 2019). I found that spatiotemporal food-web dynamics of the Everglades aquatic biota were characterized by varying degrees of piscivory, detritivory, and diet plasticity. For a given species size class, changes in trophic position were uncommon. However, even when trophic position remained constant, spatiotemporal shifts in diet and trophic niche were common. Trophic dispersion decreased for many taxa in the dry season compared to the wet season, often accompanied by trophic displacement. Previous work in hydrologically pulsing systems found either a dry-season increase or no seasonal change in number of trophic links, link density, and connectance (Winemiller, 1990). Differences between this study and others in tropical, flood-pulsed ecosystems are likely the result of the temperate origin of native fauna that have had relatively little time to adapt (~ 6,000 years) to the hydrologic regime and radiate to take advantage of the available resource pools (Loftus and Kushlan, 1987; Turner et al., 1999). In this study, all habitats displayed a dry-season decrease in total number of trophic links, link density, connectance, and proportion of omnivores. In contrast, proportion of basal resources and prey-to-predator ratio increased during water recession. In addition to variability in food quantity (proportion of omnivores, prey-to-predator ratio) and quality (proportion of basal resources), consumer foraging behavior, habitat structure, and other ecological factors drive resource profitability (Almeida et al., 1997; Winemiller and Kelso-Winemiller, 1994).

Prey consumption varied across habitat-season levels. This was driven by differences in relative abundance of consumers collected in the field and could be driven by different relative abundances of prey items in time and space. For example, Everglades studies have documented spatiotemporal variation in periphyton, invertebrate, and fish communities that reveal dynamic resource pools (Gaiser et al., 2012; Gunderson and Loftus, 1993; Loftus and Kushlan, 1987; Rader, 1994; Trexler et al., 2005; Trexler and Loftus, 2016). Consumers may track specific prey or switch prey based on availability. As a result, increased consumption of a given prey in a certain habitat or season does not necessarily mean that prey is more abundant in that habitat or season. For instance, as drying progresses, fishes move from shallow wetlands to become concentrated in ponds, especially large-bodied predators like *L. platyrhincus*, *L. gulosus*, and *A. natalis* (Loftus and Kushlan, 1987; Parkos et al., 2011). During both seasons, omnivorous cyprinodontoids dominated spikerush and sawgrass habitats (Loftus and Kushlan, 1987). This dominance explains in part the shift towards larger prey (i.e., decapods and fishes) in pond-fish stomachs relative to marsh-fish stomachs, particularly in the dry season.

Primary producers were more common in diets in spikerush and sawgrass food webs than in ponds, although the food web was primarily detrital across all habitats and both seasons. This was consistent with previous findings (Belicka et al., 2012; Williams and Trexler, 2006). The largest heterotrophic pathway in this food web may be through *Utricularia* spp. Carnivorous bladders create a mutualism between *Utricularia* spp. and a community of algae and zooplankton that in the Everglades is dominated by rotifers (Richards, 2001). Based on our results, the extensive mats of *Utricularia* spp. are deriving their energy from detrital sources, likely assimilating microbes from decaying organisms in their bladders. Given their prevalence across the landscape (Davis et al., 1994), this may represent the greatest heterotrophic food-web pathway in terms of biomass.

McMeans et al. (2019) found that flexible omnivory permitted seasonal fluctuations in trophic position. While shifts in trophic position were uncommon in the Everglades habitats in this study, for the rare trophic position shifts that did occur, they were associated with three aspects of flexible omnivory – variable amounts of piscivory, detritivory, and diet plasticity. Differences in foraging tactics combined with spatial differences in invertebrate communities (food availability) may explain mixed direction (i.e., piscivory, detritivory, or diet plasticity) and magnitude of trophic shifts among taxa. For example, dry-season *G. holbrooki* demonstrated elevated piscivory. While stomach contents often could not identify prey to species, mixing models showed that most *G. holbrooki* piscivory was cannibalism. Cannibalism in *G. holbrooki* may be a mechanism to continually consume high-quality prey as other resource bases fluctuate in time and space. Meanwhile, trophic shifts in *F. chrysotus* were related to diet plasticity in the amount of non-aquatic invertebrates (e.g., Hymenoptera, Araneae) in their diets that could potentially increase (Araneae) or decrease (Hymenoptera) their trophic position. Additionally, ponds are phosphorus-enriched relative to adjacent marshes (Kushlan and Hunt, 1979). The impact of nutrient status on structuring freshwater invertebrate communities is well documented (Sterner and Elser, 2002; Trexler and Loftus, 2016). Species-specific foraging tactics that facilitate differential access to spatially variable prey (e.g., nutrient-enriched ponds versus more oligotrophic marshes) may contribute to shifts in trophic position. Species size classes that underwent spatiotemporal shifts in trophic position included some of the most abundant fishes in the freshwater Everglades, such as *G. holbrooki*, *H. formosa*, and *F. chrysotus* (Loftus and Eklund, 1994; Loftus and Kushlan, 1987; Trexler et al., 2005). The trophic flexibility of these species contributes to their success in a dynamic ecosystem.

The prevalence of shifts in trophic dispersion and displacement, while trophic position remained constant, indicates that most species forage on different yet trophically similar prey as resources fluctuate in space and time. Niche overlap among habitat-season levels for a given species size class never exceeded 12.5% and for most comparisons was less than 5%. Like shifts in trophic position, these changes were most often driven by changes in piscivory, detritivory, and diet plasticity, all of which are thought to increase food-web stability. For example, diet plasticity, such as facultative omnivory, where an organism consumes more prey from lower trophic levels as ideal prey become scarce, as documented in this study, is thought to increase food-web stability relative to fixed omnivory (Křivan and Diehl, 2005). Furthermore, spatial and temporal trophic shifts among spatiotemporally asynchronized prey items (e.g., dynamic invertebrate assemblages) also work to stabilize food webs by providing a consistent resource base (Takimoto et al., 2002). Detritivory also increases ecosystem stability and species persistence with positive effects on trophic structure and biodiversity (Moore et al., 2004; Rooney and McCann, 2012). In fact, the habitat in which fishes consumed the most detritus – ponds – had the most diverse communities in both seasons. Trophic flexibility within food webs, such as facultative omnivory and detritivory, which facilitates exploitation of asynchronized resource availability (e.g., detritus and prey communities), may be necessary for stability in nonequilibrium ecosystems (Kratina et al., 2012; McCann and Rooney, 2009).

Topology revealed that food-web complexity decreased in the dry season in all habitats. This finding aligned with my prediction for shallow-water habitats (spikerush and sawgrass) but was the opposite of my prediction for deep-water habitats (ponds). My prediction was based on the idea that, as consumers became concentrated in ponds during the dry season, additional trophic interactions would materialize and increase food-web complexity. However, I observed reduced food-web complexity represented by decreases in total number of links, connectance, and link density during the dry season in all habitats. Decreased food-web complexity corresponded with an increase in the proportion of empty stomachs. There was evidence of fishes concentrating in ponds during the dry season in the form of increased proportions of herbivores and intermediate consumers along with a decreased proportion of top consumers. Given the seasonal decrease in species richness in ponds and decreased complexity, the concentration of fishes in ponds may not have created additional trophic interactions during the dry season but instead strengthened links already present. If a preponderance of weak interactions leads to stability (McCann et al., 1998; Rooney and McCann, 2012), then fewer, stronger interactions in dry-season ponds may be relatively unstable. Interestingly, wet-season ponds were the only habitat-season level with a prey-predator ratio less than one. Practically, this means that there are more predatory taxa than prey taxa, which indicates food-web instability (Perdomo et al., 2012). Ponds are donor-controlled habitats (Strong, 1992) that can sustain a high predator-to-prey ratio for a relatively short period of time, until the next wet season when they are reconnected to the adjacent marsh. In the absence of re-flooding, pond food webs would probably collapse to a relatively small community. Solution hole communities, which provide temporary drought refuges in short-hydroperiod areas of the Everglades like alligator ponds in longer hydroperiod regions, have declining species richness and functional diversity after they become isolated from surface aquatic connections and water quality deteriorates as the dry season progresses (Kobza et al., 2004; Rehage et al., 2014).

Food webs play a pivotal role in understanding the consequences of biological invasions (David et al., 2017). Trophic impacts from invasive species at the top and bottom of food webs are well studied and can be dramatic (Capps et al., 2015; Sharpe et al., 2017; Simon and Townsend, 2003; Tait et al., 2015; Wahl et al., 2011; Zaret and Paine, 1973). Meanwhile, mid-level consumers (e.g., many of the cichlids in this ecosystem) are underrepresented in the literature, and their location in the food web facilitates multidirectional effects that may be difficult to interpret without taking a network approach (Flood et al., 2020). My study system, Everglades National Park, is home to seventeen non-native fish species (Kline et al., 2014). This study was conducted prior to the invasion of many of these, most notably the African Jewelfish (*Hemichromis* *letourneuxi*) and Asian Swamp Eel (*Monopterus albus/javanensis*). Therefore, the data reported here constitute a “vanishingly rare” description of a pre-invasion baseline (Strayer, 2012). This baseline food web accounts for spatial and temporal variation that could otherwise obscure effects of invasions. Furthermore, I can use results from analyses here to predict that native sunfishes, particularly *L. punctatus*, will decrease in abundance as a result of trophic overlap with invasive *M. urophthalmus*, and that trophic overlap between native taxa and another invasive cichlid, *H. letourneuxi*, will be similar.

Spatiotemporal shifts in trophic position, trophic niche, and diet are well documented in a variety of taxa and ecosystems, including stream macroinvertebrates (Hellmann et al., 2013), temperate forest birds and rodents (Nakano and Murakami, 2001; Stephens et al., 2019), tropical forest bats (Salinas-Ramos et al., 2015), and mammals from arid shrubland, desert, and temperate forests (Balestrieri et al., 2019; Dawson and Ellis, 1996; Soykan and Sabo, 2009). Conversely, these metrics, particularly trophic position, can remain static through space and time in some cases, as observed in stream fishes (Rybczynski et al., 2008), tropical floodplain fishes (Correa and Winemiller, 2014), black swamp snakes (Willson et al., 2010), and certain taxa in this study. Thus, spatiotemporally flexible omnivory seems variable in its direction and magnitude and more likely to result in a change in diet and niche breadth than trophic position. Questions remain about how spatiotemporal dynamics of flexible omnivory are influenced by individual specialization within a species (Bolnick et al., 2011; Matich et al., 2021) or species size class, and how spatiotemporal dynamics of omnivory change across ecosystems, particularly at higher latitudes (Hampton et al., 2017; McMeans et al., 2015).

5. Conclusions

Spatiotemporal variability in flexible omnivory is a critical food-web attribute that helps maintain energy and nutrient cycling, facilitates species coexistence, and influences ecosystem stability (McMeans et al., 2019; Post and Takimoto, 2007; Wootton, 2017). My results indicate that fishes in a subtropical, seasonally pulsed wetland show spatiotemporal shifts in trophic niche (i.e. trophic dispersion and trophic displacement) that are not associated with shifts in trophic position. Throughout the flood pulse, most species are tracking variable, yet trophically similar, prey. Trophic dynamics were driven by varying levels of piscivory, detrital consumption, and diet plasticity. Variability across time and space at both habitat and ecosystem scales likely facilitates divergent resource pools and foraging tactics. Spatiotemporal food-web dynamics are not confined to fishes in subtropical ecosystems, and further knowledge of how organisms adapt their foraging strategies across time and space is important for understanding potential impacts of climate change, hydrologic alterations, and invasive species. Spatiotemporal trophic plasticity seems to play an important role in maintaining ecosystem stability and resilience particularly in ecosystems with regular perturbations.

Acknowledgments

I thank W. Anderson, E. Gaiser, A. Harborne, D. Simberloff, two anonymous reviewers, and the associate editor for their helpful comments on an earlier version of this manuscript. I am grateful to Bill Loftus and Joel Trexler who originally conceived of this project and collected the data. This project was funded by U.S. NPS, USGS, and Florida Dept. of Environmental Protection under Florida Contracts SP-329 and SP-419. I am grateful to several technicians who assisted in project field and lab work, particularly J. Rehage and V. Foster. PJF and JCT were supported by Cooperative Agreement P18AC01074 between Everglades National Park and Florida International University while working on this project. PJF was supported by the Everglades Foundation as an FIU ForEverglades Scholar. This paper was developed in collaboration with Florida Coastal Everglades Long-Term Ecological Research program under National Science Foundation Grant No. DEB-1237517 and DEB-2025954. This material is based upon work supported by the National Science Foundation under Grant No. HRD-1547798 and HRD-2111661. This NSF Grant was awarded to Florida International University as part of the Centers of Research Excellence in Science and Technology (CREST) Program.

Literature Cited

Adams, S.M., Kimriel, B.L., Ploskey, G.R., 1983. Sources of Organic Matter for Reservoir Fish Production: A Trophic-Dynamics Analysis, Can. J. of Fish. and Aquat. Sci.

Akin, S., Winemiller, K.O., 2006. Seasonal variation in food web composition and structure in a temperate tidal estuary. Estuar. and Coasts 29, 552–567. https://doi.org/10.1007/BF02784282

Anderson, M.J., Walsh, D.C.I., 2013. PERMANOVA, ANOSIM, and the Mantel test in the face of heterogeneous dispersions: What null hypothesis are you testing?, Ecol. Monogr. https://doi.org/https://doi.org/10.1890/12-2010.1

Arias, M.E., Cochrane, T.A., Norton, D., Killeen, T.J., Khon, P., 2013. The flood pulse as the underlying driver of vegetation in the largest wetland and fishery of the Mekong basin. Ambio 42, 864–876. https://doi.org/10.1007/s13280-013-0424-4

Arrington, D.A., Winemiller, K.O., Loftus, W.F., Akin, S., 2002. How often do fishes “run on empty”? Ecology 83, 2145–2151. https://doi.org/10.1890/0012-9658(2002)083[2145:HODFRO]2.0.CO;2

Azevedo, L.S., Pestana, I.A., Almeida, M.G., Bastos, W.R., Souza, C.M.M., 2021. Do fish isotopic niches change in an Amazon floodplain lake over the hydrological regime? Ecol. Freshw. Fish. https://doi.org/10.1111/eff.12609

Balestrieri, A., Remonti, L., Saino, N., Raubenheimer, D., 2019. The 'omnivorous badger dilemma': towards an integration of nutrition with the dietary niche in wild mammals. Mamm. Rev. https://doi.org/10.1111/mam.12164

Belicka, L.L., Sokol, E.R., Hoch, J.M., Jaffé, R., Trexler, J.C., 2012. A molecular and stable isotopic approach to investigate algal and detrital energy pathways in a freshwater marsh. Wetlands 32, 531–542. https://doi.org/10.1007/s13157-012-0288-6

Bellmore, J.R., Baxter, C. V., Connolly, P.J., 2015. Spatial complexity reduces interaction strengths in the meta-food web of a river floodplain mosaic. Ecology 96, 274–283. https://doi.org/10.1890/14-0733.1

Bolnick, D.I., Amarasekare, P., Araújo, M.S., Bürger, R., Levine, J.M., Novak, M., Rudolf, V.H.W., Schreiber, S.J., Urban, M.C., Vasseur, D.A., 2011. Why intraspecific trait variation matters in community ecology. Trends Ecol. Evol. https://doi.org/10.1016/j.tree.2011.01.009

Bowen, S.H., 1983. Quantitative description of the diet, in: Nielsen, L.A., Johnson, D.L. (Eds.), Fisheries Techniques. American Fisheries Society, Bethesda, MD, pp. 325–336.

Capps, K.A., Atkinson, C.L., Rugenski, A.T., 2015. Implications of species addition and decline for nutrient dynamics in fresh waters. Freshw. Sci. 34, 485–496. https://doi.org/10.1086/681095

CaraDonna, P.J., Petry, W.K., Brennan, R.M., Cunningham, J.L., Bronstein, J.L., Waser, N.M., Sanders, N.J., 2017. Interaction rewiring and the rapid turnover of plant–pollinator networks. Ecol. Lett. https://doi.org/10.1111/ele.12740

Carnicer, J., Jordano, P., Melia´n, C.J., Melia´n, M., 2009. The temporal dynamics of resource use by frugivorous birds: a network approach, Ecology. https://doi.org/https://doi.org/10.1890/07-1939.1

Correa, S.B., Winemiller, K.O., 2014. Niche partitioning among frugivorous fishes in response to fluctuating resources in the Amazonian floodplain forest. Ecology 95, 210–224. https://doi.org/https://doi.org/10.1890/13-0393.1

Costa-Pereira, R., Tavares, L.E.R., de Camargo, P.B., Araújo, M.S., 2017. Seasonal population and individual niche dynamics in a tetra fish in the Pantanal wetlands. Biotropica 49, 531–538. https://doi.org/10.1111/btp.12434

Cross, W.F., Baxter, C. V., Rosi-Marshall, E.J., Hall, R.O., Kennedy, T.A., Donner, K.C., Kelly, H.A.W., Seegert, S.E.Z., Behn, K.E., Yard, M.D., 2013. Food-web dynamics in a large river discontinuum. Ecol. Monogr. 83, 311–337. https://doi.org/10.1890/12-1727.1

David, P., Thébault, E., Anneville, O., Duyck, P.F., Chapuis, E., Loeuille, N., 2017. Impacts of Invasive Species on Food Webs: A Review of Empirical Data. Adv. Ecol. Res. 56, 1–60. https://doi.org/10.1016/bs.aecr.2016.10.001

Davis, S.M., Gunderson, L.H., Park, W.A., Richardson, J.R., Mattson, J.E., 1994. Landscape dimension, composition, and function in a changing Everglades ecosystem, in: Davis, S.M., Ogden, J.C. (Eds.), Everglades: The Ecosystem and Its Restoration. Delray Beach, FL, pp. 419–444. https://doi.org/10.1201/9781466571754-25

Dawson, T.J., Ellis, B.A., 1996. Diets of mammalian herbivores in Australian arid, hilly shrublands: Seasonal effects on overlap between euros (hill kangaroos), sheep and feral goats, and on dietary niche breadths and electivities. J. Arid Environ. 34, 491–506. https://doi.org/10.1006/jare.1996.0127

de Almeida, V.L.L., Hahn, N.S., Vazzoler, A.E.A. de M., 1997. Feeding patterns in five predatory fishes of the high Paraná River floodplain (PR, Brazil). Ecol. Freshw. Fish 6, 123–133. https://doi.org/10.1111/j.1600-0633.1997.tb00154.x

DeAngelis, D.L., White, P.S., 1994. Ecosystems as products of spatially and temporally varying driving forces, ecological processes, and landscapes: a theoretical perspective, in: Davis, S., Ogden, J. (Eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, FL, pp. 9–27.

Elton, C., 1927. Animal ecology. Macmillan, New York, New York, USA.

Flood, P.J., Duran, A., Barton, M., Mercado-Molina, A.E., Trexler, J.C., 2020. Invasion impacts on functions and services of aquatic ecosystems. Hydrobiologia 847, 1571–1586. https://doi.org/10.1007/s10750-020-04211-3

Fry, B., 2006. Stable isotope ecology. Springer, New York, New York, USA.

Gaiser, E.E., Trexler, J.C., Wetzel, P.R., 2012. The Florida Everglades, in: Batzer, D.P., Baldwin, A.H. (Eds.), The Florida Everglades. University of California Press, Berkeley, CA, pp. 231–252. https://doi.org/10.5749/minnesota/9780816670260.003.0001

Gatto, J. V., Kline, J.L., Loftus, W.F., Trexler, J.C., 2021. Linking demographic transitions to population dynamics in a fluctuating environment. Can. J. Fish. Aquat. Sci. 78, 797–808. https://doi.org/10.1139/cjfas-2020-0101

Gatto, J. V., Trexler, J.C., 2019. Seasonality of fish recruitment in a pulsed floodplain ecosystem: Estimation and hydrological controls. Environ. Biol. Fishes 102, 595–613. https://doi.org/10.1007/s10641-019-00856-9

Gelman, A., Rubin, D.B., 1992. Inference from Iterative Simulation Using Multiple Sequences. Stat. Sci. 7, 45–52. https://doi.org/10.1214/ss/1177011136

Gunderson, L.H., Loftus, W.F., 1993. The Everglades, in: Martin, W.H., Boyce, S.G., Echternacht, E.G. (Eds.), Biodiversity of the Southeastern United States 2. John Wiley and Sons Inc., New York, New York, USA, pp. 199–255.

Hampton, S.E., Galloway, A.W.E., Powers, S.M., Ozersky, T., Woo, K.H., Batt, R.D., Labou, S.G., O'Reilly, C.M., Sharma, S., Lottig, N.R., Stanley, E.H., North, R.L., Stockwell, J.D., Adrian, R., Weyhenmeyer, G.A., Arvola, L., Baulch, H.M., Bertani, I., Bowman, L.L., Carey, C.C., Catalan, J., Colom-Montero, W., Domine, L.M., Felip, M., Granados, I., Gries, C., Grossart, H.P., Haberman, J., Haldna, M., Hayden, B., Higgins, S.N., Jolley, J.C., Kahilainen, K.K., Kaup, E., Kehoe, M.J., MacIntyre, S., Mackay, A.W., Mariash, H.L., McKay, R.M., Nixdorf, B., Nõges, P., Nõges, T., Palmer, M., Pierson, D.C., Post, D.M., Pruett, M.J., Rautio, M., Read, J.S., Roberts, S.L., Rücker, J., Sadro, S., Silow, E.A., Smith, D.E., Sterner, R.W., Swann, G.E.A., Timofeyev, M.A., Toro, M., Twiss, M.R., Vogt, R.J., Watson, S.B., Whiteford, E.J., Xenopoulos, M.A., 2017. Ecology under lake ice. Ecol. Lett. https://doi.org/10.1111/ele.12699

Hellmann, C., Wissel, B., Winkelmann, C., 2013. Omnivores as seasonally important predators in a stream food web 1. Freshw. Sci. 32, 548–562. https://doi.org/10.1899/12-020.1

Heng, K., Chevalier, M., Lek, S., Laffaille, P., 2018. Seasonal variations in diet composition, diet breadth and dietary overlap between three commercially important fish species within a flood-pulse system: The Tonle Sap Lake (Cambodia). PLoS One 13, 1-16. https://doi.org/10.1371/journal.pone.0198848

Hervé, M., 2021. RVAidMemoire: Testing and Plotting Procedures for Biostatistics.

Hyslop, E.J., 1980. Stomach contents analysis-a review of methods and their application, J. Fish Biol. https://doi.org/https://doi.org/10.1111/j.1095-8649.1980.tb02775.x

Jackson, A.L., Inger, R., Parnell, A.C., Bearhop, S., 2011. Comparing isotopic niche widths among and within communities: SIBER - Stable Isotope Bayesian Ellipses in R. J. Anim. Ecol. 80, 595–602. https://doi.org/10.1111/j.1365-2656.2011.01806.x

Jepsen, D.B., Winemiller, K.O., Taphorn, D.C., 1997. Temporal patterns of resource partitioning among Cichla species in a Venezuelan blackwater river. J. Fish Biol. 51, 1085–1108. https://doi.org/https://doi.org/10.1111/j.1095-8649.1997.tb01129.x

Jost, L., Chao, A., Chazdon, R.L., 2011. Compositional similarity and beta diversity, in: Magurran, A.E., McGill, B.J. (Eds.), Biological Diversity. Front. in Meas. and Assess. Oxford University Press (OUP), Oxford, UK, pp. 66–84.

Junk, W.J., Bayley, P.B., Sparks, R.E., 1989. The Flood Pulse Concept in River-Floodplain Systems. Can. Spec. Publ. Fish. Aquat. Sci. Honey Harbour, Ontario, Canada 7, 110–127. https://doi.org/10.1007/BF02564079

Kline, J.L., Loftus, W.F., Kotun, K., Trexler, J.C., Rehage, J.S., Lorenz, J.J., Robinson, M., 2014. Recent fish introductions into Everglades National Park: An unforeseen consequence of water management? Wetlands 34, S175–S187. https://doi.org/10.1007/s13157-012-0362-0

Kobza, R.M., Trexler, J.C., Loftus, W.F., Perry, S.A., 2004. Community structure of fishes inhabiting aquatic refuges in a threatened Karst wetland and its implications for ecosystem management. Biol. Conserv. 116, 153–165. https://doi.org/10.1016/S0006-3207(03)00186-1

Kominoski, J.S., Gaiser, E.E., Baer, S.G., 2018 Advancing theories of ecosystem development through long-term ecological research. BioScience. 68, 8, 554-562. https://doi.org.10.1093/biosci/biy070

Kondoh, M., 2003. Foraging Adaptation and the Relationship Between Food-Web Complexity and Stability. Science 299, 1388–1391. https://doi.org/10.1126/science.1079154

Kones, J.K., Soetaert, K., van Oevelen, D., Owino, J.O., 2009. Are network indices robust indicators of food web functioning? A Monte Carlo approach. Ecol. Modell. 220, 370–382. https://doi.org/10.1016/j.ecolmodel.2008.10.012

Kratina, P., LeCraw, R.M., Ingram, T., Anholt, B.R., 2012. Stability and persistence of food webs with omnivory: Is there a general pattern? Ecosphere 3, 1–18. https://doi.org/10.1890/es12-00121.1

Křivan, V., Diehl, S., 2005. Adaptive omnivory and species coexistence in tri-trophic food webs. Theor. Popul. Biol. 67, 85–99. https://doi.org/10.1016/j.tpb.2004.09.003

Kushlan, J.A., Hunt, B.P., 1979. Limnology of an alligator pond in South Florida. Florida Sci. 42, 65–84.

Layman, C.A., Arrington, D.A., Montaña, C.G., Post, D.M., 2007. Can stable isotope ratios provide for community-wide measures of trophic structure? Ecology 88, 42–48. https://doi.org/https://doi.org/10.1890/0012-9658(2007)88[42:CSIRPF]2.0.CO;2

Leigh, C., Burford, M.A., Sheldon, F., Bunn, S.E., 2010. Dynamic stability in dry season food webs within tropical floodplain rivers. Mar. Freshw. Res. 61, 357–368. https://doi.org/10.1071/MF09107

Loftus, W.F., Eklund, A.M., 1994. Long-term dynamics of an Everglades small-fish assemblage, in: Davis, S.M., Ogden, J.C. (Eds.), Everglades: The Ecosystem and Its Restoration. St. Lucie Press, Delray Beach, FL, pp. 461–483.

Loftus, W.F., Kushlan, J.A., 1987. Freshwater fishes of southern Florida. Bull. Florida State Museum, Biol. Sci. 31, 147–344.

Lowe-McConnell, R.H., 1969. The cichlid fishes of Guyana, South America, with notes on their ecology and breeding behaviour. Zool. J. Linn. Soc. 48, 255–302. https://doi.org/10.1111/j.1096-3642.1969.tb00714.x

Lowe-McConnell, R.H., 1964. The fishes of the Rupununi savanna district of British Guiana, South America: Part 1. Ecological groupings of fish species and effects of the seasonal cycle on the fish. J. Linn. Soc. London, Zool. 45, 103–144. https://doi.org/10.1111/j.1096-3642.1964.tb00490.x

Mangiafico, S., 2021. rcompanion: Functions to Support Extension Education Program Evaluation.

Matich, P., Bizzarro, J.J., Shipley, O.N., 2021. Are stable isotope ratios suitable for describing niche partitioning and individual specialization? Ecol. Appl. 31, 1–8. https://doi.org/10.1002/eap.2392

McCann, K.S., Hastings, A., Huxel, G.R., 1998. Weak trophic interactions and the balance of nature. Nature 395, 794–798. https://doi.org/10.1007/978-3-319-06257-0_9

McCann, K.S., Rasmussen, J.B., Umbanhowar, J., 2005. The dynamics of spatially coupled food webs. Ecol. Lett. 8, 513–523. https://doi.org/10.1111/j.1461-0248.2005.00742.x

McCann, K.S., Rooney, N., 2009. The more food webs change, the more they stay the same. Philos. Trans. R. Soc. B Biol. Sci. 364, 1789–1801. https://doi.org/10.1098/rstb.2008.0273

McConnell, R., Lowe-McConnell, R.H., 1987. Ecological studies in tropical fish communities. Cambridge University Press.

McCutchan, J.H., Lewis, W.M., Kendall, C., Mcgrath, C.C., 2003. Variation in trophic shift for stable isotope ratios of carbon, nitrogen, and sulfur. Oikos 102, 378–390. https://doi.org/https://doi.org/10.1034/j.1600-0706.2003.12098.x

McHugh, P.A., Thompson, R.M., Greig, H.S., Warburton, H.J., Mcintosh, A.R., 2015. Habitat size influences food web structure in drying streams. Ecography (Cop.). 38, 700–712. https://doi.org/10.1111/ecog.01193

McMeans, B.C., Kadoya, T., Pool, T.K., Holtgrieve, G.W., Lek, S., Kong, H., Winemiller, K., Elliott, V., Rooney, N., Laffaille, P., Mccann, K.S., 2019. Consumer trophic positions respond variably to seasonally fluctuating environments. Ecology 100, 1–10. https://doi.org/10.1002/ecy

McMeans, B.C., McCann, K.S., Humphries, M., Rooney, N., Fisk, A.T., 2015. Food web structure in temporally-forced ecosystems. Trends Ecol. Evol. 30, 662–672. https://doi.org/10.1016/j.tree.2015.09.001

Moore, J.C., Berlow, E.L., Coleman, D.C., De Suiter, P.C., Dong, Q., Hastings, A., Johnson, N.C., McCann, K.S., Melville, K., Morin, P.J., Nadelhoffer, K., Rosemond, A.D., Post, D.M., Sabo, J.L., Scow, K.M., Vanni, M.J., Wall, D.H., 2004. Detritus, trophic dynamics and biodiversity. Ecol. Lett. 7, 584–600. https://doi.org/10.1111/j.1461-0248.2004.00606.x

Moore, J.C., de Ruiter, P.C., 1991. Temporal and spatial heterogeneity of trophic interactions within below-ground food webs. Agric. Ecosyst. Environ. 34, 371–397. https://doi.org/10.1016/0167-8809(91)90122-E

Moyle, P.B., Senanayake, F.R., 1984. Resource partitioning among the fishes of rainforest streams in Sri Lanka. J. Zool. 202, 195–223. https://doi.org/10.1111/j.1469-7998.1984.tb05951.x

Nakano, S., Murakami, M., 2001. Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. Proc. Natl. Acad. Sci. U. S. A. 98, 166–170. https://doi.org/https://doi.org/10.1073/pnas.98.1.166

Odum, E.P., 1969. The strategy of ecosystem development. Science 164, 262–270. https://doi.org/10.1126/science.164.3877.262

Odum, W.E., Odum, E.P., Odum, H.Y., 1995. Nature’s Pulsing Paradigm. Estuaries 18, 547–555.

Oksanen, J., Gullaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O’Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.M., Szoecs, E., Wagner, H., 2020. vegan: Community Ecology Package.

Olesen, J.M., Bascompte, J., Elberling, H., Jordano, P., 2008. Temporal dynamics in a pollination network. Ecology 89, 1573–1582. https://doi.org/https://doi.org/10.1890/07-0451.1

Paine, R.T., 1988. Road maps of interactions or grist for theoretical development? Ecology 69, 1648–1654. https://doi.org/https://doi.org/10.2307/1941141

Parkos, J.J., Ruetz, C.R., Trexler, J.C., 2011. Disturbance regime and limits on benefits of refuge use for fishes in a fluctuating hydroscape. Oikos 120, 1519–1530. https://doi.org/10.1111/j.1600-0706.2011.19178.x

Parnell, A., Inger, R., 2016. simmr: A Stable Isotope Mixing Model.

Parnell, A.C., Inger, R., Bearhop, S., Jackson, A.L., 2010. Source partitioning using stable isotopes: Coping with too much variation. PLoS One 5, 1–5. https://doi.org/10.1371/journal.pone.0009672

Parnell, A.C., Phillips, D.L., Bearhop, S., Semmens, B.X., Ward, E.J., Moore, J.W., Jackson, A.L., Grey, J., Kelly, D.J., Inger, R., 2013. Bayesian stable isotope mixing models. Environmetrics 24, 387–399. https://doi.org/10.1002/env.2221

Payne, A.I., 1986. The ecology of tropical lakes and rivers. John Wiley and Sons Inc., New York, New York, USA.

Perdomo, G., Sunnucks, P., Thompson, R.M., 2012. foodweb: an open-source program for the visualisation and analysis of complex food webs.

Petta, J.C., Shipley, O.N., Wintner, S.P., Cliff, G., Dicken, M.L., Hussey, N.E., 2020. Are you really what you eat? Stomach content analysis and stable isotope ratios do not uniformly estimate dietary niche characteristics in three marine predators. Oecologia 192, 1111–1126. https://doi.org/10.1007/s00442-020-04628-6

Pettit, N.E., Naiman, R.J., Warfe, D.M., Jardine, T.D., Douglas, M.M., Bunn, S.E., Davies, P.M., 2017. Productivity and connectivity in tropical riverscapes of nothern australia: ecological insights for management. Ecosystems 20, 492–514. https://doi.org/10.1007/s10021-016-0037-4

Phillips, D.L., Inger, R., Bearhop, S., Jackson, A.L., Moore, J.W., Parnell, A.C., Semmens, B.X., Ward, E.J., 2014. Best practices for use of stable isotope mixing models in food-web studies. Can. J. Zool. 92, 823–835. https://doi.org/10.1139/cjz-2014-0127

Piet, G.J., Pet, J.S., Guruge, W.A.H.P., Vijverberg, J., Van Densen, W.L.T., 1999. Resource partitioning along three niche dimensions in a size-structured tropical fish assemblage. Can. J. Fish. Aquat. Sci. 56, 1241–1254. https://doi.org/10.1139/cjfas-56-7-1241

Poisot, T., Stouffer, D.B., Gravel, D., 2015. Beyond species: Why ecological interaction networks vary through space and time. Oikos 124, 243–251. https://doi.org/10.1111/oik.01719

Post, D.M., Takimoto, G., 2007. Proximate structural mechanisms for variation in food-chain length. Oikos 116, 775–782. https://doi.org/10.1111/j.2007.0030-1299.15552.x

Power, M.E., 1983. Grazing responses of tropical freshwater fishes to different scales of variation in their food. Environ. Biol. Fishes 9, 103–115. https://doi.org/10.1007/BF00690856

Prejs, A., Prejs, K., 1987. Feeding of tropical freshwater fishes: seasonality in resource availability and resource use. Oecologia 71, 397–404. https://doi.org/10.1007/BF00378713

Quezada-Romegialli, C., Jackson, A.L., Hayden, B., Kahilainen, K.K., Lopes, C., Harrod, C., 2018. tRophicPosition, an r package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods Ecol. Evol. 9, 1592–1599. https://doi.org/10.1111/2041-210X.13009

Quirino, B.A., Carniatto, N., Guglielmetti, R., Fugi, R., 2017. Changes in diet and niche breadth of a small fish species in response to the flood pulse in a Neotropical floodplain lake. Limnologica 62, 126–131. https://doi.org/10.1016/j.limno.2016.10.005

R Core Team, 2020. R: A Language and Enivronment for Statistical Computing.

Rader, R.B., 1994. Macroinvertebrates of the northern Everglades: Species composition and trophic structure. Florida Sci. 57, 22–33.

Rehage, J.S., Liston, S.E., Dunker, K.J., Loftus, W.F., 2014. Fish community responses to the combined effects of decreased hydroperiod and nonnative fish invasions in a Karst wetland: Are Everglades solution holes sinks for native fishes? Wetlands 34, S159–S173. https://doi.org/10.1007/s13157-012-0361-1

Richards, J.H., 2001. Bladder function in *Utricularia purpurea* (Lentibulariaceae): Is carnivory important? Am. J. Bot. 88, 170–176. https://doi.org/10.2307/2657137

Rooney, N., McCann, K.S., 2012. Integrating food web diversity, structure and stability. Trends Ecol. Evol. 27, 40–46. https://doi.org/10.1016/j.tree.2011.09.001

Ruetz, C.R., Trexler, J.C., Jordan, F., Loftus, W.F., Perry, S.A., 2005. Population dynamics of wetland fishes: Spatio-temporal patterns synchronized by hydrological disturbance? J. Anim. Ecol. 74, 322–332. https://doi.org/10.1111/j.1365-2656.2005.00926.x

Ruiz-Cooley, R.I., Gerrodette, T., Fiedler, P.C., Chivers, S.J., Danil, K., Ballance, L.T., 2017. Temporal variation in pelagic food chain length in response to environmental change. Sci. Adv. 3, 1–9. https://doi.org/10.1126/sciadv.1701140

Rybczynski, S.M., Walters, D.M., Fritz, K.M., Johnson, B.R., 2008. Comparing trophic position of stream fishes using stable isotope and gut contents analyses. Ecol. Freshw. Fish 17, 199–206. https://doi.org/10.1111/j.1600-0633.2007.00289.x

Salinas-Ramos, V.B., Herrera Montalvo, L.G., León-Regagnon, V., Arrizabalaga-Escudero, A., Clare, E.L., 2015. Dietary overlap and seasonality in three species of mormoopid bats from a tropical dry forest. Mol. Ecol. 24, 5296–5307. https://doi.org/10.1111/mec.13386

Sharpe, D.M.T., De León, L.F., González, R., Torchin, M.E., 2017. Tropical fish community does not recover 45 years after predator introduction. Ecology 98, 412–424. https://doi.org/10.1002/ecy.1648

Simon, K.S., Townsend, C.R., 2003. Impacts of freshwater invaders at different levels of ecological organisation, with emphasis on salmonids and ecosystem consequences. Freshw. Biol. 48, 982–994. https://doi.org/10.1046/j.1365-2427.2003.01069.x

Soykan, C.U., Sabo, J.L., 2009. Spatiotemporal food web dynamics along a desert riparian-upland transition. Ecography (Cop.). 32, 354–368. https://doi.org/10.1111/j.1600-0587.2008.05615.x

Stephens, R.B., Hobbie, E.A., Lee, T.D., Rowe, R.J., 2019. Pulsed resource availability changes dietary niche breadth and partitioning between generalist rodent consumers. Ecol. Evol. 9, 10681–10693. https://doi.org/10.1002/ece3.5587

Sterner, R.W., Elser, J.J., 2002. Ecological stoichiometry: the biology of elements from molecules to the biosphere. Princeton University Press, Princeton, NJ.

Strayer, D.L., 2012. Eight questions about invasions and ecosystem functioning. Ecol. Lett. 15, 1199–1210. https://doi.org/10.1111/j.1461-0248.2012.01817.x

Strong, D.R., 1992. Are trophic cascades all wet? Differentiation and donor-control in speciose ecosystems. Ecology 73, 747–754. https://doi.org/https://doi.org/10.2307/1940154

Tait, L.W., South, P.M., Lilley, S.A., Thomsen, M.S., Schiel, D.R., 2015. Assemblage and understory carbon production of native and invasive canopy-forming macroalgae. J. Exp. Mar. Bio. Ecol. 469, 10–17. https://doi.org/10.1016/j.jembe.2015.04.007

Takimoto, G., Iwata, T., Murakami, M., 2002. Seasonal subsidy stabilizes food web dynamics: Balance in a heterogeneous landscape. Ecol. Res. 17, 433–439. https://doi.org/10.1046/j.0912-3814.2002.00502.x

Thompson, R.M., Hemberg, M., Starzomski, B.M., Shurin, J.B., 2007. Trophic levels and trophic tangles: The prevalence of omnivory in real food webs. Ecology 88, 612–617. https://doi.org/10.1890/05-1454

Thompson, R.M., Townsend, C.R., 2005. Food-web topology varies with spatial scale in a patchy environment. Ecology 86, 1916–1925. https://doi.org/https://doi.org/10.1890/04-1352

Trexler, J.C., Gaiser, E.E., Kominoski, J.S., Sanchez, J.L., 2015. The role of periphyton mats in consumer community structure and function in calcareous wetlands: Lessons from the Everglades, in: Entry, J.A., Gottlieb, A.D., Javachandrahan, K., Ogram, A.

(Eds.), Microbiology of the Everglades Ecosystem. Science Publishers, CRC Press, pp. 155–179.

Trexler, J.C., Loftus, W.F., 2016. Invertebrates of the Florida Everglades, in: Batzer, D., Boix, D. (Eds.), Invertebrates in Freshwater Wetlands: An International Perspective on Their Ecology. Springer, New York, New York, USA, pp. 321–356.

Trexler, J.C., Loftus, W.F., Jordan, F., Chick, J.H., Kandl, K.L., McElroy, T.C., Bass, O.L., 2002. Ecological Scale and Its Implications for Freshwater Fishes in the Florida Everglades, in: Porter, J.W., Porter, K.G. (Eds.), The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook. CRC Press, Boca Raton, FL, pp. 153–184. https://doi.org/10.1201/9781420039412-8

Trexler, J.C., Loftus, W.F., Perry, S., 2005. Disturbance frequency and community structure in a twenty-five year intervention study. Oecologia 145, 140–152. https://doi.org/10.1007/s00442-005

Wahl, D.H., Wolfe, M.D., Santucci, V.J., Freedman, J.A., 2011. Invasive carp and prey community composition disrupt trophic cascades in eutrophic ponds. Hydrobiologia 678, 49–63. https://doi.org/10.1007/s10750-011-0820-3

Wainwright, P.C., Richard, B.A., 1995. Predicting patterns of prey use from morphology of fishes. Environ. Biol. Fishes 44, 97–113. https://doi.org/10.1007/BF00005909

Wheeler, B., Torchiano, M., 2016. lmPerm: Permutation Tests for Linear Models.

Williams, A.J., Trexler, J.C., 2006. A preliminary analysis of the correlation of food-web characteristics with hydrology and nutrient gradients in the southern Everglades. Hydrobiologia 569, 493–504. https://doi.org/10.1007/s10750-006-0151-y

Willson, J.D., Winne, C.T., Pilgrim, M.A., Romanek, C.S., Gibbons, J.W., 2010. Seasonal variation in terrestrial resource subsidies influences trophic niche width and overlap in two aquatic snake species: A stable isotope approach. Oikos 119, 1161–1171. https://doi.org/10.1111/j.1600-0706.2009.17939.x

Winemiller, K.O., 1990. Spatial and temporal variation in tropical fish trophic networks. Ecol. Monogr. 60, 331–367. https://doi.org/10.2307/1943061

Winemiller, K.O., 1989. Ontogenetic diet shifts and resource partitioning among piscivorous fishes in the Venezuelan ilanos. Environ. Biol. Fishes 26, 177–199. https://doi.org/10.1007/BF00004815

Winemiller, K.O., Jepsen, D.B., 1998. Effects of seasonality and fish movement on tropical river food webs. J. Fish Biol. 53, 267–296. https://doi.org/https://doi.org/10.1111/j.1095-8649.1998.tb01032.x

Winemiller, K.O., Kelso-Winemiller, L.C., 1994. Comparative ecology of the African pike, *Hepsetus odoe*, and tigerfish, *Hydrocynus forskahlii*, in the Zambezi River floodplain. J. Fish Biol. 45, 211–225. https://doi.org/10.1111/j.1095-8649.1994.tb01301.x

Wootton, K.L., 2017. Omnivory and stability in freshwater habitats: Does theory match reality? Freshw. Biol. 62, 821–832. https://doi.org/10.1111/fwb.12908

Yang, L.H., Bastow, J.L., Spence, K.O., Wright, A.N., 2008. What can we learn from resource pulses? Ecology 89, 621–634.

Zaret, T.M., Paine, R.T., 1973. Species introduction in a tropical lake. Science 182, 449–455. https://doi.org/10.1126/science.182.4111.449

Zaret, T.M., Rand, S.A., 1971. Competition in tropical stream fishes: Support for the competitive exclusion principle. Ecology 52, 336–342. https://doi.org/https://doi.org/10.2307/1934593

Zeileis, A., Hornik, K., Wiel, M.A., Hothorn, T., 2008. Implementing a class of permutation tests: The coin package. J. Stat. Softw. 28, 1–23.

Table 1. Common and scientific names, species abbreviations, number of size classes, and sample size for each species across habitat-season levels (P = pond, Sr = spikerush, Sg = sawgrass). Species are abbreviated in other tables and figures as first letter of the genus, underscore, followed by first three letters of the species.

				Wet Season			Dry Season		
Common Name	Species	Abbr.	Num. of Size Classes	P	Sr	Sg	P	Sr	Sg
Diamond Killifish	*Adinia xenica*	A_xen	1	0	0	0	0	2	0
Yellow Bullhead	*Ameiurus natalis*	A_nat	4	53	31	9	113	27	0
Bowfin	*Amia calva*	A_cal	1	0	1	0	18	0	0
Pike Killfish	*Belonesox belizanus*	B_bel	2	21	0	0	29	9	0
Black Acara	*Cichlasoma bimaculatum*	C_bim	1	2	1	0	1	0	0
Walking Catfish	*Clarias batrachus*	C_bat	1	17	0	0	14	0	0
Sheepshead Minnow	*Cyprinodon variegatus*	C_var	1	0	25	0	0	5	1
Everglades Pygmy Sunfish	*Elassoma evergladei*	E_eve	1	12	22	2	6	24	5
Bluespotted Sunfish	*Enneacanthus gloriosus*	E_glo	1	0	34	0	0	22	0
Lake Chubsucker	*Erimyzon sucetta*	E_suc	3	17	0	0	0	0	0
Swamp Darter	*Etheostoma fusiforme*	E_fus	1	5	2	0	0	0	0
Marsh Killifish	*Fundulus confluentus*	F_con	2	4	29	13	0	43	33
Golden Topminnow	*Fundulus chrysotus*	F_chr	2	21	72	62	8	68	76
Eastern Mosquitofish	*Gambusia holbrooki*	G_hol	2	50	51	51	49	48	43
Least Killifish	*Heterandria formosa*	H_for	2	14	50	36	14	48	44
Flagfish	*Jordanella floridae*	J_flo	2	17	50	20	0	31	31
Golden Silverside	*Labiesthes vanhyningi*	L_van	1	28	0	0	0	0	0
Florida Gar	*Lepisosteus platyrhincus*	L_pla	2	54	0	11	92	0	0
Warmouth	*Lepomis gulosus*	L_gul	4	84	4	4	40	6	1
Bluegill Sunfish	*Lepomis macrochirus*	L_mac	1	36	1	0	3	2	0
Dollar Sunfish	*Lepomis marginatus*	L_mar	1	22	42	14	7	57	1
Redear Sunfish	*Lepomis microlophus*	L_mic	2	17	11	0	2	1	0
Spotted Sunfish	*Lepomis punctatus*	L_pun	3	29	53	9	59	65	16

Bluefin Killifish	*Lucania goodei*	L_goo	2	43	86	60	0	79	70
Mayan Cichlid	*Mayaheros urophthalmus*	M_uro	2	16	0	0	0	0	0
Largemouth Bass	*Micropterus salmoides*	M_sal	1	24	0	0	0	0	0
Golden Shiner	*Notemigonus crysoleucas*	N_cry	1	4	0	0	0	0	0
Taillight Shiner	*Notropis maculatus*	N_mac	1	13	0	0	0	0	0
Coastal Shiner	*Notropis petersoni*	N_pet	1	26	0	0	0	0	0
Tadpole Madtom	*Noturus gyrinus*	N_gyr	1	3	20	4	6	8	1
Spotted Tilapia	*Pelmatolapia mariae*	P_mar	1	26	0	0	0	0	0
Sailfin Molly	*Poecilia latipinna*	P_lat	2	0	14	0	7	10	12

Table 2. Ontogenetic shifts in diet or trophic niche for species size classes within a habitat-season level. Pairwise PERMANOVA of stomach contents (p), probability of a change in niche breadth (p (A < B)), percent difference in SEA_b between groups A and B (SEA_b %Δ), percent niche overlap (Overlap), and driver for documented ontogenetic trophic dynamics. Driver abbreviations represent: H – herbivory, D – detritivory, DP – diet plasticity (a change in niche but not proportions of prey guilds), C – carnivory, and P – piscivory. Only species size classes with either a statistical difference in diet or SEA_b among size classes are include. See Table A.2.11 for full PERMANOVA results and Table A.2.12 for full ontogenetic SEA_b results.

Habitat-Season Level	Species	Size Class A	Size Class B	p	p (A < B)	SEA_b %Δ	Overlap	Driver
Pond - Wet	A_nat	1	2	0.018	0.810	31	7.34	C/P
		1	3	0.002	0.810	29	7.27	C/P
		1	4	0.002	0.300	-26	4.52	C/P
	E_suc	2	3	0.004	< 0.001	-99	0.03	DP
	F_chr	1	2	0.372	0.003	-85	1.13	H/D/P
	G_hol	1	2	0.002	1.000	656	12.90	H/D/P
	H_for	1	2	0.009	0.999	500	0.77	DP
	L_goo	1	2	0.004	0.945	67	0.99	H/DP
	L_gul	1	2	0.663	0.133	-29	3.25	H/D/DP
		1	3	0.002	0.418	-4	4.11	H/D/DP
		1	4	0.012	0.900	55	6.59	H/D/P
		2	3	0.012	0.848	35	4.23	DP
		2	4	0.067	0.991	118	6.70	P
	L_pun	1	2	0.029	0.988	160	9.72	P
		1	3	0.041	1.000	112	22.9	D
Pond - Dry	A_nat	1	2	0.020	< 0.001	-53	1.91	D/C/P
		1	3	0.230	0.844	26	5.33	P
		1	4	0.050	0.948	72	9.28	P
		2	3	0.130	1.000	168	5.36	P
		2	4	0.020	1.000	264	9.37	P

	G_hol	1	2	0.141	1.000	1013	6.36	H/D/P
	H_for	1	2	0.049	1.000	4233	2.76	D
	L_gul	1	3	1.000	0.006	-69	4.11	DP
		3	4	0.241	1.000	225	6.65	P
	L_pla	1	2	0.010	1.000	167	7.27	P
	L_pun	2	3	0.050	0.999	120	12.9	H/D/C/P
Spikerush - Wet	A_nat	1	2	0.004	0.226	-48	3.55	H/D/C
	F_chr	1	2	0.002	0.433	-7	2.95	P
	G_hol	1	2	0.002	1.00	4233	2.68	P
	L_goo	1	2	0.002	< 0.001	-67	0.42	DP
	L_pun	1	2	0.002	1.000	220	5.20	C/P
Spikerush - Dry	F_chr	1	2	0.002	1.000	159	7.36	C/P
	F_con	1	2	0.026	0.492	0	2.03	C/P
	G_hol	1	2	0.018	0.999	154	3.09	H/P
	H_for	1	2	0.010	0.161	-23	1.16	DP
	L_goo	1	2	0.009	0.990	100	1.22	DP
	L_pun	1	2	0.002	0.999	122	4.06	C/P
Sawgrass - Wet	F_chr	1	2	0.007	0.994	92	2.73	DP
	G_hol	1	2	0.004	< 0.001	-67	0.77	H/DP/P
	H_for	1	2	0.131	0.000	-92	0.17	H/DP
	L_goo	1	2	0.004	< 0.001	-60	0.38	DP
	L_pun	2	3	0.025	0.544	5	1.89	C/P
Sawgrass - Dry	F_chr	1	2	0.007	0.893	34	4.52	C/P
	F_con	1	2	0.085	1.000	325	3.45	C/P
	G_hol	1	2	0.013	0.671	13	2.47	H/P
	H_for	1	2	0.242	1.000	350	1.56	H/DP
	J_flo	1	2	0.037	< 0.001	-94	0.22	H/C
	L_goo	1	2	0.002	0.537	25	1.17	DP
	L_pun	1	2	0.002	1.000	589	6.68	C/P

Table 3. Permuted ANOVA for trophic positions among species size classes across habitat-season levels derived from stomach contents. Only comparisons with statistical differences are reported here, complete results are reported in Appendix A (Table A.2.7). Species names are abbreviated as first letter of the genus, underscore, first three letters of the species, followed by size class. Group A is from the first habitat-season level in the Habitat-Season Comparison column, and Group B is from the latter habitat-season level. Season 1 = Wet; season 2 = Dry. Driver abbreviations represent: H – herbivory, D – detritivory, DP – diet plasticity (a change in niche but not proportions of prey guilds), C – carnivory, and P – piscivory.

Ontogenetic Shifts

Group A	Group B	Habitat-Season Comparison	F	p-value	%Δ	Driver
A_nat1	A_nat4	Pond1 - Pond1	-2.48	0.01	25.5	C/P
L_gul2	L_gul3	Pond1 - Pond1	-2.40	0.02	17.1	DP
J_flo1	J_flo2	Spikerush1 - Pond1	-2.37	0.02	19.8	D/C
A_nat1	A_nat2	Pond1 - Pond1	-2.20	0.03	19.8	C/P
L_pun2	L_pun3	Sawgrass1 - Pond2	-2.20	0.03	14.1	H/D
A_nat1	A_nat4	Pond1 - Pond2	-2.15	0.03	27.4	C/P
F_chr1	F_chr2	Sawgrass2 - Pond1	-2.09	0.04	16.3	C/P
L_pun1	L_pun3	Sawgrass2 - Pond2	-2.06	0.04	11.3	H/D
A_nat1	A_nat4	Sawgrass1 - Pond1	-2.00	0.05	27.3	C/P
H_for1	H_for2	Pond2 - Spikerush1	-1.99	0.05	26.3	DP
H_for1	H_for2	Pond2 - Sawgrass1	-1.97	0.05	25.8	H/DP

Habitat-Season Shifts

Group A	Group B	Habitat-Season Comparison	F	p-value	%Δ	Driver
F_con1	F_con1	Sawgrass2 - Spikerush1	2.73	0.01	-13.1	C/P
A_nat1	A_nat1	Spikerush2 - Pond1	2.43	0.01	-15.9	D/C/P
A_nat1	A_nat1	Spikerush1 - Pond1	2.40	0.02	-14.9	D/C/P
L_gul3	L_gul3	Pond1 - Pond2	2.26	0.02	-18.9	DP
F_chr2	F_chr2	Sawgrass1 - Pond1	-2.19	0.03	16.8	C
H_for1	H_for1	Spikerush1 - Pond2	2.18	0.03	-23.6	DP
F_con1	F_con1	Sawgrass2 - Spikerush2	2.15	0.03	-9.9	DP/C
L_mar	L_mar	Spikerush1 - Pond2	2.10	0.04	-16.3	D/DP/C
L_pun2	L_pun2	Sawgrass1 - Sawgrass2	-2.05	0.04	28.1	P
F_chr2	F_chr2	Sawgrass2 - Pond1	-2.01	0.04	14.1	C/P
J_flo2	J_flo2	Sawgrass1 - Pond1	-1.99	0.05	17.4	DP

Table 4. Food-web network metrics for each habitat-season level (Prop. Omni. – proportion of omnivores, Prop. Cann. – proportion of cannibalism, Num. trophic positions – number of trophic positions, Prop. Basal – proportion of basal taxa, Prop. Intermediate – proportion of intermediate consumers, Prop. Top – proportion of top consumers, Prop. Herb. – proportion of herbivores, Prey:Predator – prey to predator ratio).

Network Metrics	Pond		Spikerush		Sawgrass	
	Wet	Dry	Wet	Dry	Wet	Dry
Species richness	40	37	34	35	33	32
Total # Links	298	225	209	197	188	169
Connectance	0.19	0.16	0.18	0.16	0.17	0.17
Link density	7.45	6.08	6.15	5.63	5.70	5.28
Prop. Omni	0.60	0.54	0.47	0.46	0.52	0.47
Prop. Cann.	0.40	0.38	0.29	0.26	0.24	0.28
Num. trophic positions	9	8	9	9	9	9
Prop. Basal	0.13	0.14	0.15	0.20	0.15	0.19
Prop. Intermediate	0.68	0.76	0.82	0.77	0.82	0.78
Prop. Top	0.20	0.11	0.03	0.03	0.03	0.03
Prop. Herb.	0.28	0.30	0.32	0.31	0.33	0.34
Prey:Predator	0.91	1.03	1.14	1.21	1.14	1.19

Figure 1. NMDS plots of consumers and diet items (Prey). Labels represent the mean for that group among habitat-season levels. For some groups, lines extend from the label to their location in NMDS space to minimize overlap. MDS 1 is a directly related to prey size and MDS 2 is inversely related to consumer size and trophic position. Ellipses represent habitat-season levels. Filled ellipses are 95% confidence intervals and open ellipses are 95% data ellipses (they contain 95% of the data). Changes in ellipses size and location show spatiotemporal expansion or contraction of community-wide prey consumption. Prey that group near a given consumer are likely eaten by that consumer. Output from this NMDS is the basis for stomach-content niche modeling.

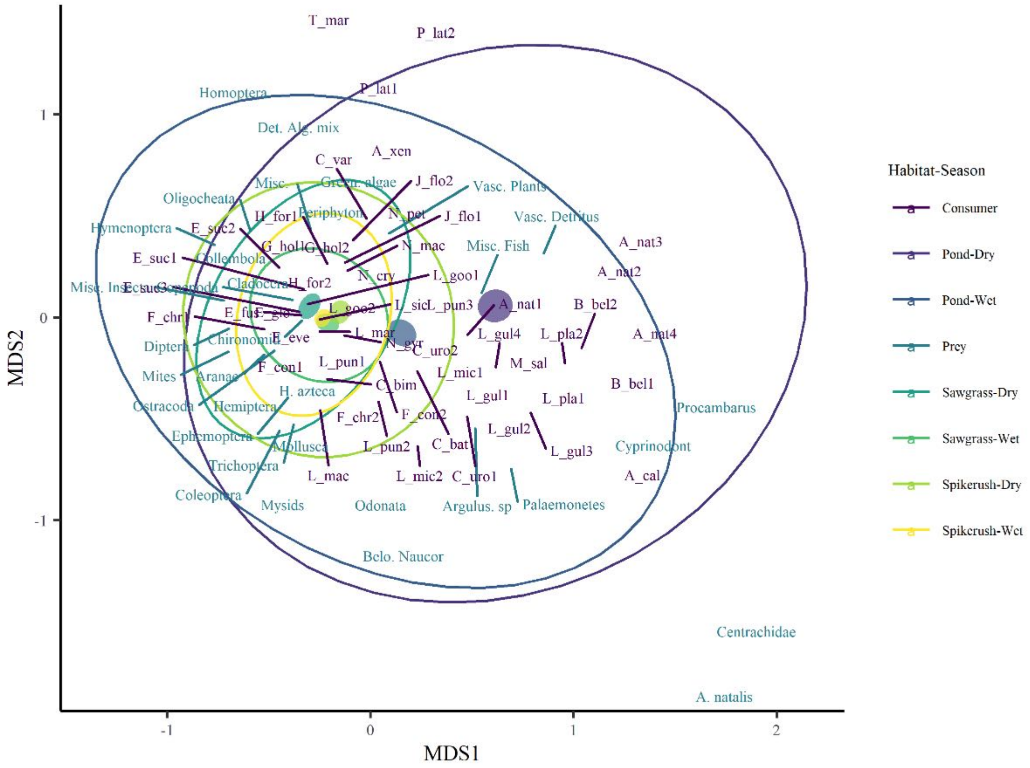


Figure 2. Dominance-diversity curves for all habitat-season levels (A – Pond-Wet, B – Spikerush-Wet, C – Sawgrass-Wet, D – Pond-Dry, E – Spikerush-Dry, F – Sawgrass-Dry). Midge larvae (CHIRON) are the most important diet item across habitat-season levels. Only ponds see a notable shift in fish-diet communities between seasons. Prey item codes: CHIRON – Chironomidae larvae, CLADOC – Cladocera, HYASPP – *Hyalella* spp., COPEPO – Copepoda, COLLEM – Collembola, OSTRAC – Ostracoda, MISC – Miscellaneous, PALPAL – *Palaemonetes paludosus*, TRICHO – Trichoptera, MOLLUS – Mollusca, ODONAT – Odonata, DIPTER – Diptera, MISFIS – Miscellaneous fish, CYPRIN – Cyprinodontiformes.

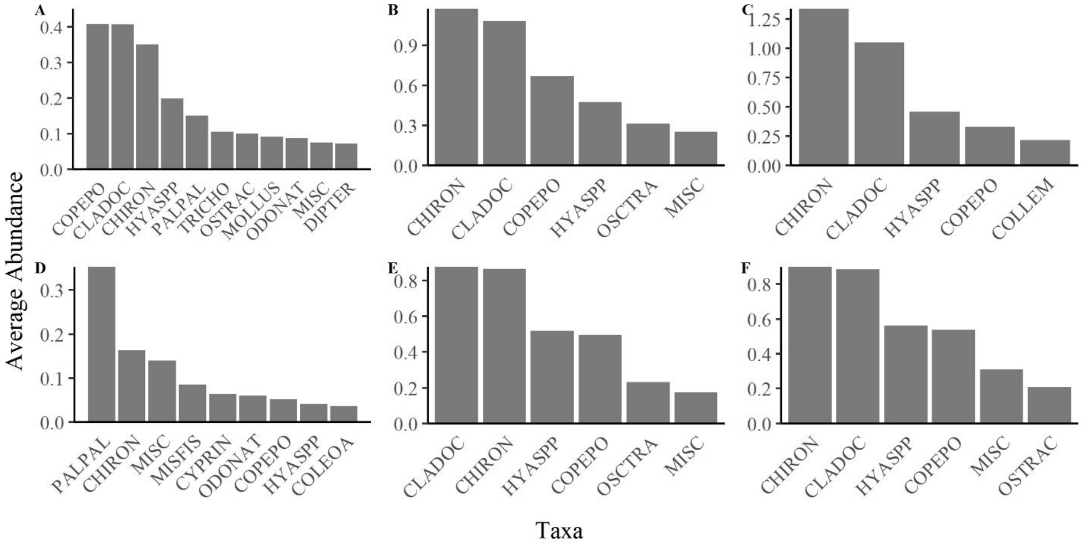


Figure 3. A) Trophic position of consumers modelled using stable isotopes. B) Alpha, the proportion of δ15N derived from detritus, of Everglades aquatic consumers. Colors and shapes correspond to trophic guilds from Table A.1, except for the carnivorous plant, Utricularia foliosa, which I placed in its own group “Carn. Plants”. Some consumers only have a lower bound of their 95% credibility interval on the plot denoted by a horizontal line because the upper limit of their trophic position extends off the plot.

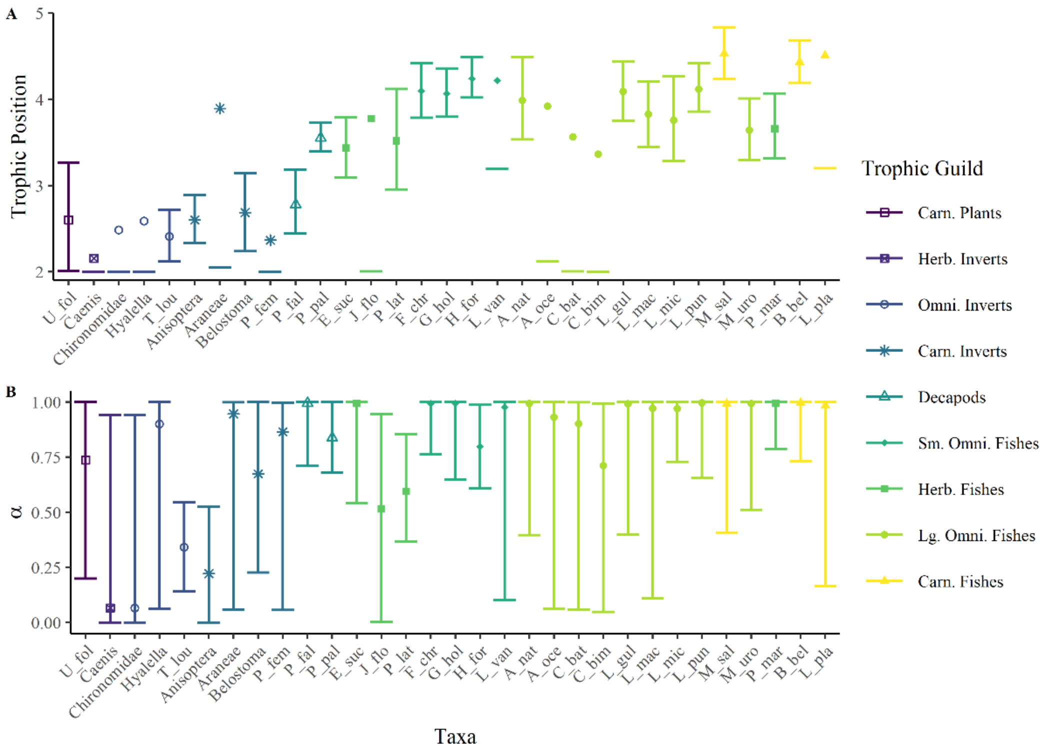


Figure 4. SEA$_b$ frequency distributions (colors correspond to trophic guild) for each habitat-season level, all wet-season consumers, all dry-season consumers, and all consumers. For all habitats, there was a higher frequency of larger niche sizes (increased generalization) in the dry season. Pond habitats had the highest frequency of smaller niche sizes (relative specialization) of any habitat for both seasons. Wet-season pond and dry-season pond both have a single outlier excluded from the plot.

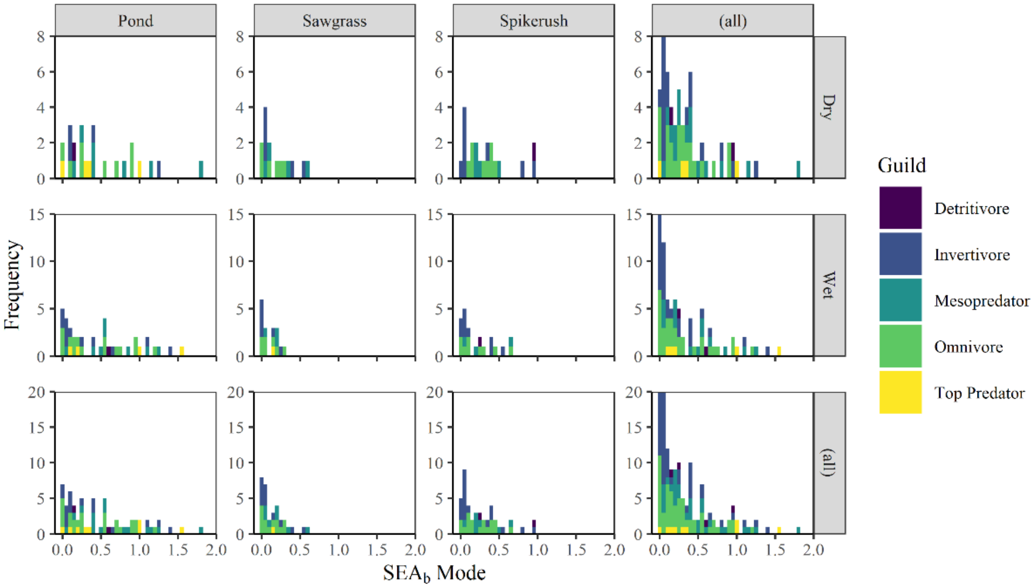


Figure 5. Layman metrics derived from stomach contents across habitat-season levels (1 – Wet, 2 – Dry). Letters denote statistical differences at a 95% threshold. Layman metrics are MDS 1 (x-axis) range, MDS 2 (y-axis) range, TA – total area, CD - centroid distance, NND – nearest neighbor distance, SDNND – standard deviation of NND.

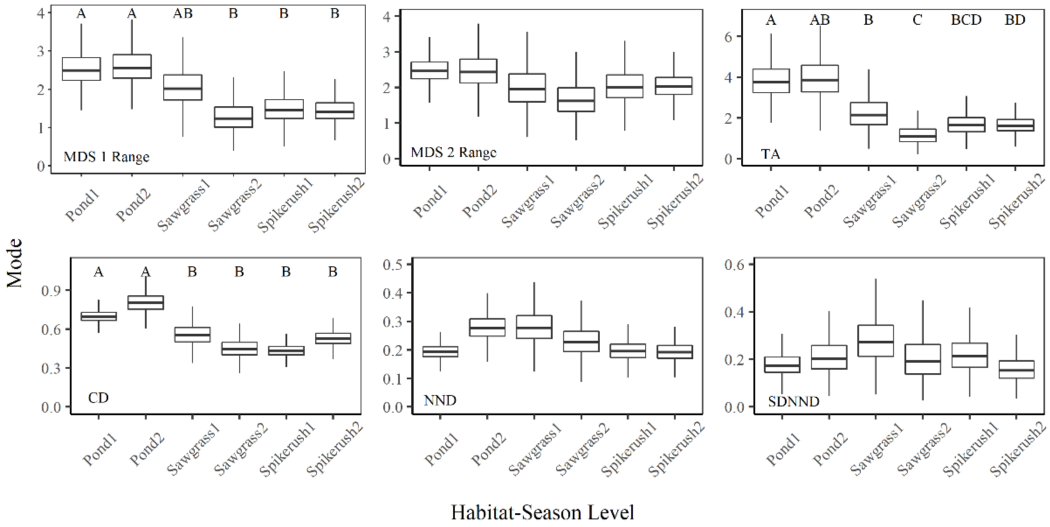
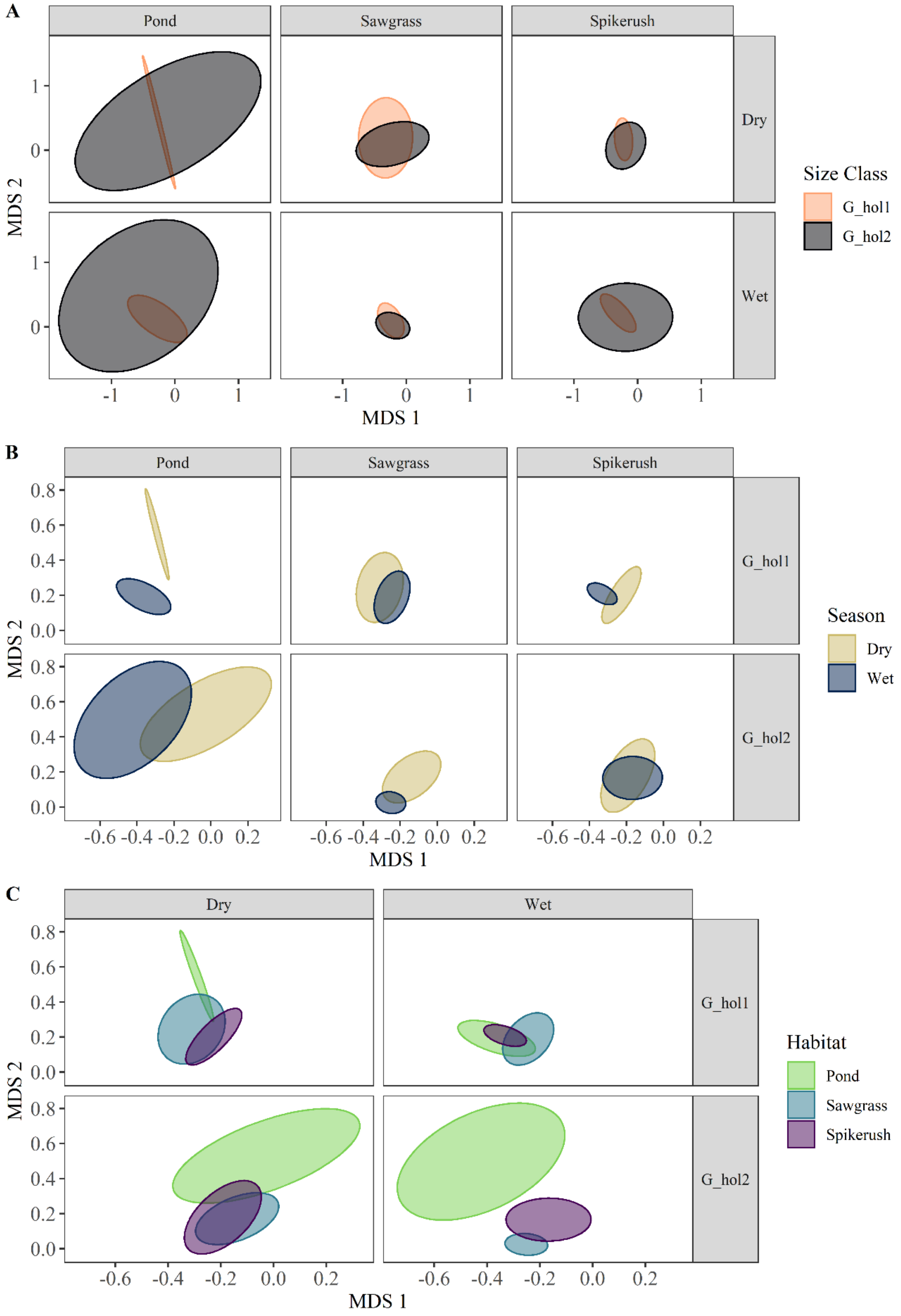


Figure 6. Trophic-niche ellipses (SEA_b) based on stomach contents in NMDS space for *G. holbrooki* across ontogeny (A), seasons (B), and habitats (C). Plots for other species are in Fig. A.2.4 and A.2.5. Note that the scale for each axis varies across plots to optimize each visualization. This results in the same ellipse looking slightly different among plots.



Appendix A.

A.1 Methods

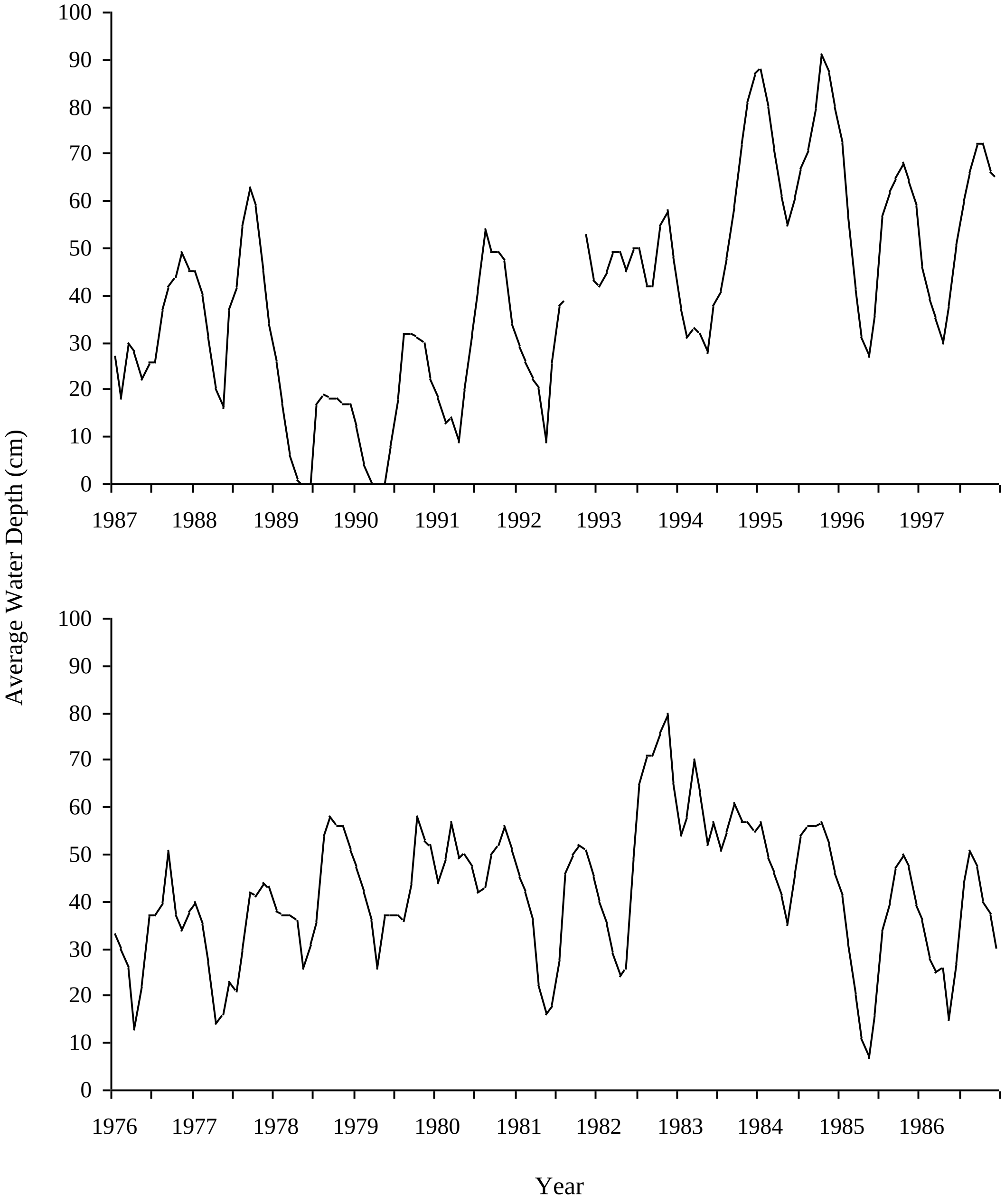
A.1.1 Study Area

Table A.1.1 Range of depths (cm) for each habitat sampled in both seasons along with the range of water temperatures (°C) among all habitats in both seasons.

		Habitat			
	Season	Pond	Spikerush	Sawgrass	Temp (°C)
Depth (cm)	Wet	100 - 120	25 - 40	25 - 40	14 - 20
	Dry	75 - 100	5 - 17	2 - 15	20 - 28

Our study area in north-central Shark River Slough has a relatively long hydroperiod; complete drying occurs only in severe droughts at a frequency of one to three times per decade (Figure A.1.1).

Figure A.1.1. Hydrograph of water depths from central Shark River Slough covering the time period when animals were collected for stomach contents (1977-1981) and stable isotopes (mid-1990s). The 1992 data gap resulted from Hurricane Andrew disruptions.



A.1.2 Field Collections

Specimens collected from 1977-1981 were used only for stomach analyses, representing over 90% of all samples. In the mid-1990s while collecting specimens for stable-isotope analysis, there were also supplemental collections used for stomach analyses to increase sample sizes for a few taxa. All stable-isotope samples were collected in the mid-1990s. All collections occurred at the same study site. Stable-isotope samples were only collected from wet-season spikerush. Accordingly, I did not use stable isotopes to examine spatiotemporal dynamics but to supply additional evidence for consumer diets, including invertebrates for which I did not perform stomach analyses, and to evaluate basal resource use (the alpha metric).

A.1.3 Laboratory Analyses

Laboratory analyses were performed when the samples were collected and completed by 1999, not more recently when the paper was written. Animals were identified to the lowest possible taxon (see https://trexlerlab.wordpress.com/id-guides-and-data-sets/). Individual lengths and masses are not reported here but are available in the raw data (see section A.3 Data Availability). Primary producers were acid washed with HCl to avoid damage to the mass spectrometer. Everglades periphyton is calcareous and will otherwise damage the mass spectrometer used for isotopic analyses. Consumer stable-isotope samples were not acid washed.

Table A.1.2 Size class ranges in millimeters for each species, and samples sizes of those size classes among habitat-season levels. Some species were not partitioned into size classes and do not have a size range in the Length column. All individuals of those taxa were treated as a single size class.

Species	Length (mm)	Size Class	Pond		Spikerush		Sawgrass	
			Wet	Dry	Wet	Dry	Wet	Dry
Ameiurus natalis	≤ 96	1	22	43	26	27	7	0
	97 < x ≤ 135	2	14	46	4	0	2	0
	135 < x ≤ 175	3	9	19	3	0	0	0
	> 175	4	8	5	0	0	0	0
Amia calva	-	1	0	18	1	0	0	0
Belonesox belizanus	≤ 64	1	2	7	0	3	0	0
	> 64	2	19	22	0	6	0	0
Cichlasoma bimaculatum	-	1	2	1	1	0	0	0
Clarias batrachus	-	1	17	14	0	0	0	0
Cyprinodon variegatus	-	1	0	0	25	5	0	1
Elassoma evergladei	-	1	12	6	22	24	2	5
Enneacanthus gloriosus	-	1	0	0	34	22	0	0
Erimyzon sucetta	≤ 100	1	2	0	0	0	0	0
	100 < x ≤ 150	2	7	0	0	0	0	0
	> 150	3	8	0	0	0	0	0
Etheostoma fusiforme	-	1	5	0	2	0	0	0
Fundulus confluentus	≤ 40	1	4	0	27	17	11	23
	> 40	2	0	0	2	26	2	10
Fundulus chrysotus	≤ 40	1	16	8	49	50	47	60
	> 40	2	5	0	23	18	15	16
Gambusia holbrooki	≤ 17	1	25	25	25	23	26	18
	> 17	2	25	23	26	25	25	25
Heterandria formosa	≤ 12	1	8	5	25	25	11	19
	> 12	2	6	9	25	23	25	25
Jordanella floridae	≤ 18	1	3	0	25	5	4	6
	> 18	2	14	0	25	26	16	25
Labidesthes vanhyningi	-	1	28	0	0	0	0	0
Lepisosteus platyrhincus	≤ 450	1	32	88	0	0	11	0
	> 450	2	22	4	0	0	0	0

Lepomis gulosus	≤ 71	1	18	6	0	1	2	1
	$71 < x \leq 95$	2	23	6	0	3	0	0
	$95 < x \leq 120$	3	29	12	0	2	1	0
	> 120	4	14	16	1	2	1	0
Lepomis macrochirus	-	1	36	3	1	2	0	0
Lepomis marginatus	-	1	22	7	42	57	14	1
Lepomis microlophus	≤ 100	1	1	2	10	1	0	0
	> 100	2	15	0	1	0	0	0
Lepomis punctatus	≤ 40	1	13	1	33	34	0	10
	$40 < x \leq 90$	2	10	42	18	31	5	5
	> 90	3	6	16	2	0	4	1
Lucania goodei	≤ 17.5	1	26	0	51	41	22	21
	> 17.5	2	17	0	35	38	38	49
Mayaheros urophthalmus	≤ 120	1	6	0	0	0	0	0
	> 120	2	10	0	0	0	0	0
Micropterus salmoides	-	1	22	0	0	0	0	0
Notemigonus crysoleucas	-	1	4	0	0	0	0	0
Noturus gyrinus	-	1	3	6	20	8	4	1
Notropis maculatus	-	1	13	0	0	0	0	0
Notropis petersoni	-	1	26	0	0	0	0	0
Pelmatolapia mariae	-	1	26	0	0	0	0	0
Poeilia latipinna	≤ 20	1	0	0	11	3	0	0
	> 20	2	2	5	0	10	2	10

A.1.4 Community Trends in Prey Consumption

I used Morisita-Horn distances, as opposed to commonly used Bray-Curtis, because of the arguments laid out in Jost et al. (2011). In brief, Morisita-Horn distances are density invariant which allows for separating effects of higher abundance overall from changes in relative abundances of species. When using Bray-Curtis distances, if some samples are from more productive sites with higher abundances but with similar relative abundances to other less productive sites, that higher total abundance will disproportionately influence dissimilarity. Despite relative abundances being similar, if one were to standardize their data then calculate Bray-Curtis distances the results would be functionally similar to Morisita-Horn distances.

A.1.5 Trophic Position

In laboratory analyses, stomach contents were quantified using both the volumetric approach (Bowen, 1983) and the numeric method reported in this manuscript (Hyslop, 1980). The small mass of most prey taxa required pooling for volumetric analysis of like taxa within a sample of each size-class. As a result, statistical analyses were performed on the count (numeric) dataset to allow for incorporation of inter-individual variation within a species size-class and I have presented those results. However, volumetric data are also available (see section A.3 Data Availability).

Table A.1.3. Aggregated fish-prey groups (34 categories, and 8 categories in bold) with trophic scores following Adams et al. (1983), Winemiller (1996), and this study. Trophic categories for invertebrates were based on literature data.

Primary Producers	Herbivorous Invertebrates	Omnivorous Invertebrates	Carnivorous Invertebrates	Decapods	Small Fishes	Large Fishes
Score = 0.0	Score = 1.0	Score = 1.5	Score = 2.0	Score = 2.0	Score = 2.37	Score = 2.5
Vascular Plants	Homoptera	*Hyalella* spp.	Hemiptera	*Procambarus*	Cyprinodontids	*Ameiurus natalis*
Green Algae	Collembola	Trichoptera	Odonata	*Palaemonetes paludosus*		Centrarchidae
Periphyton	Ephemeroptera	Chironomid	Aranae			Miscellaneous Fish
	Mollusca	Diptera	*Belostoma / Pelocoris*			
Detritus		Coleoptera	Mites			
Score = 0.2		Hymenoptera	*Argulus* spp.			
Vascular Detritus		Oligochaeta				
Detritus/Algal Mix		Cladocera				
		Copepoda				
		Ostracoda				
		Insecta				
		Mysids				
		Miscellaneous				

The raw stomach-content data contained over 300 diet categories that were condensed to 34 higher-level taxonomic groups. Those categories were further condensed to eight functional guilds to facilitate communicating results (Table A.1). To estimate trophic positions of fishes based on stomach contents, I followed Adams et al. (1983), where trophic position (τ) of a predator is defined by the trophic levels of its prey. Within the eight prey categories, I assigned trophic scores to primary producers and all invertebrates, except for decapods, based on literature (Table A.1) (Adams et al., 1983). For decapods and fishes, I calculated scores based on Everglades gut and stable-isotope data. The τ for a fish species was calculated by adding 1 to the weighted τ of its prey, assuming that a predator is one τ above its prey (Adams et al., 1983; Winemiller, 1990):

(EQ. A.1) $$\tau_i = 1.0 + \sum \tau_j \,(F_{ij}),$$

Where τ_i is τ of fish species *i*, τ_j is τ of food item *j* and, F_{ij} is the proportion of the food volume for species *i* comprised by item *j*. Equation (A.1) provided an independent, quantitative method for calculating trophic positions of species or size classes based on empirically derived gut-content data. Trophic position distributions for each species size class at each habitat-season level were tested for normality using Shapiro-Wilkes tests. Differences among size classes and among habitat-season levels were assessed using permuted ANOVA (10,000 iterations) via the *aovp* function from the *lmPerm* package in R (Wheeler and Torchiano, 2016). Species size class, habitat-season level, and the interaction of the two were included in the model as fixed effects.

For modeling trophic position and alpha using stable isotopes, flocculent matter (floc) and periphyton isotopes of both carbon and nitrogen were statistically indistinguishable. Therefore, I combined them into a single denoted as detritus. The bulk of Everglades periphyton mats are made up of detritus. Isotopic data for some basal resources, such as floc and periphyton, were not collected during field sampling for this study (floc), or there was not a useable $\delta^{15}N$ peak (periphyton). I used floc and periphyton isotopic data from a later study in a similar area (Williams and Trexler, 2006) to fill this gap.

I used the *multiSpeciesTP* function from *tRophicPosition* to estimate trophic position and alpha for each species followed by the *pairwiseComparison* function to compare trophic position and alpha among species (Quezada-Romegialli et al., 2018).

This function calculates the pairwise probability that a given metric (i.e., trophic position or alpha) is less than or equal between two groups (species in this case) for all possible comparisons.

A.1.6 Dietary Proportions

Source groups for fishes were determined based on the 34-group stomach-content data. For invertebrate consumers, source groups were determined from the literature (Rader, 1994) and expert opinion (Loftus, Trexler). SIMMs do not perform well as the number of sources increases and cannot differentiate isotopically similar sources (Phillips et al., 2014). To account for these limitations, only diet items that constituted at least 5% of a consumer's diet based on stomach contents were included (Table A.1.4). This convention prevents noise from rare diet items in the SIMM's output, but it may negatively affect the mixing polygon. One assumption of SIMMs is that all consumers fall within the mixing polygon – a convex polygon whose vertices are the outermost diet items after one applies a trophic enrichment factor (TEF) (Phillips et al., 2014). For some consumers, I lacked data to generate a valid mixing polygon. This happens when there is a missing source(s). In these instances, I did not run a SIMM. Furthermore, there is a mismatch in taxonomic resolution between the stable-isotope data (species level) and the thirty-four-group stomach-content data (often family or higher). As a result, assumptions had to be made about which member(s) of the taxonomic group should be included in SIMMs. Occasionally, sources were grouped (i.e., two confamilial fishes) to align better with stomach-content data and reduce the number of sources in the model.

For each SIMM, I calculated DIC and ΔDIC among SIMMs for each consumer. SIMMs with a $\Delta \text{DIC} \leq 2$ were considered equally informative for describing variance in the consumer’s isotopic signature. I calculated the Gelman-Rubin diagnostic used to assess model convergence for each SIMM (Gelman and Rubin, 1992). All SIMMs were generated using 500,000 iterations, 50,000 burn-in, thin of 100, and 4 chains.

Informative priors were generated using the *simmr_elicit* function in *simmr* (Parnell et al., 2010; Parnell and Inger 2016). In this case, the sum of prior proportions from stomach contents must sum to one. That was not always the case because of the exclusion of rare diet items (< 5% contribution to the diet) and missing sources (stable-isotope data were not collected for all diet items). In those situations, the difference between the sum of the diet item’s percent contribution to the diet was subtracted from one and the remainder proportionally added to the diet items in the SIMM based on the diet item’s proportional contribution to the consumer’s diet (Table A.1.5).

Table A.1.4. Example workflow for creation of informative priors for SIMMs based on stomach-content data. Species typically had many diet categories contributing very little to the overall diet, but several that constituted most of the diet. As a result, diet items contributing less than 5% based on their proportion in stomach contents were excluded in source groups for SIMMs. This caused the prior proportions of source groups to not sum to 1. To remedy this, I took the remainder (1 – sum of prior proportions for included source groups) and proportionally added it to source groups based on the proportion of each item to the diet, excluding those contributing < 5%. For example, source A below is 25% of the diet when considering all sources but 27% of the diet when considering only sources contributing greater than 5% to the diet. Therefore, source A gets 27% of the remainder (0.08 times 0.27 = 0.022) added to it and the final proportion becomes 0.272 (0.25 + 0.022 = 0.272).

Source Group	Proportion of Stomach Contents	Source Included in SIMM	Proportions w/ only groups > 5%	Added from Remainder	Final Prior Proportion
A	0.25	Yes	0.25	0.022	0.272
B	0.4	Yes	0.4	0.035	0.435
C	0.11	Yes	0.11	0.010	0.120
D	0.16	Yes	0.16	0.014	0.174
E	0.04	No	-	-	-
F	0.04	No	-	-	-
Total	1		0.92	0.080	1.000

A.1.7 Ontogenetic Shifts in Diet

This study included fifty-two size classes across thirty-one species. Size structuring of populations is often overlooked. The results of this, particularly for trophic studies, are well documented (Piet et al., 1999; Wainwright and Richard, 1995). Accordingly, I used size classes as the unit of observation for stomach-content samples to account for this. Stable-isotope samples are grouped at the species level because I did not have large samples sizes for each species to subdivide into size classes.

A.1.8 Dietary Niche

SIBER models require four input columns: “iso1”, “iso2”, “group”, and “community”. Our model used NMDS axes MDS1 and MDS2 as iso1 and iso2, respectively, species size-classes as group, and habitat-season level as community. SIBER models were run using 2,000,000 iterations, 10,000 burn-in, thin of 100, 2 chains, and uninformative priors recommended by package authors (Jackson et al., 2011). For SIBER models using stable-isotope data I used species as group and trophic guild as community. Trophic dispersion and displacement were calculated the same way for stomach content and stable isotope SIBER model output.

Layman metrics can be calculated for populations or communities. The Layman metrics are nitrogen range (spread along the y-axis; MDS2 for stomach contents), carbon range (spread along the x-axis; MDS1 for stomach contents), total area (TA – measure of overall trophic diversity), mean distance to centroid (CD – measure of average trophic diversity), mean nearest neighbor distance (NND – measure of individual or species packing), and standard deviation of nearest neighbor distance (SDNND – measure of evenness of individual or species packing). I modeled Layman metrics for habitat-season level communities using stomach contents and for the entire community partitioned into trophic guilds using stable isotopes. As a result, NND and SDNND are a measure of species packing and evenness of species packing respectively.

A.1.9 Network Structure

Here I define the following food-web network metrics from that I calculated and their biological significance: taxa richness (S) – number of biological taxa (nodes) in the food web, number of trophic links (L) – number of links (edges) between taxa, link density (L/S) – number of links per taxa that measures dietary specialization across the food web, connectance ($C = L/S^2$) – proportion of potential trophic links that do occur as an indicator of degree of inter-connectivity in a food-web (typically 0.05 – 0.15), number of omnivores – number of taxa that feed on more than one trophic level, mean chain length – average number of links found in a food chain across the food web (food chains length with increased energy supply), maximum chain length – maximum number of links found in any food chain in the food web, number of basal taxa (b) – number of taxa which do not consume any other taxa, number of intermediate taxa (i) – number of taxa which are both consumed by and consume other taxa, number of top taxa (t) – number of taxa which are not consumed by any other taxa, predator:prey ratio (b + i / t + i) – measure of food-web shape where high values are more triangular, low values are more square, and when < 1 food web has an inverted structure that may indicate instability (Perdomo et al., 2012).

A.2 Results

A.2.1 Community Trends in Prey Consumption

Table A.2.1. PERMANOVA output for prey consumption among communities from different habitat-season levels (top) and Tukey's HSD analyzing multivariate dispersion among communities from different habitat-season levels (bottom). Lower and upper bounds of multivariate dispersion are for 95% intervals.

A. PERMANOVA

	DF	SS	Mean SS	F	R^2	p - value
Habitat	2	84.37	42.2	130.4	0.08	0.001
Season	1	14.70	14.7	45.4	0.01	0.001
Habitat*Season	2	9.87	4.9	15.3	0.01	0.001
Residuals	2901	938.27	0.3		0.90	
Total	2906	1047.21			1.00	

B. Tukey's HSD on Multivariate Dispersion

Habitat-Season Comparison	Difference	Lower Bound	Upper Bound	p - value
Cladium Dry-Cladium Wet	0.10	0.05	0.14	< 0.001
Eleocharis Wet-Cladium Wet	0.06	0.02	0.10	< 0.001

Eleocharis Dry-Cladium Wet	0.13	0.09	0.16	0.00
Pond Wet-Cladium Wet	0.27	0.23	0.31	0.00
Pond Dry-Cladium Wet	0.26	0.22	0.30	0.00
Eleocharis Wet-Cladium Dry	-0.04	-0.07	0.00	0.07
Eleocharis Dry-Cladium Dry	0.03	-0.01	0.07	0.19
Pond Wet-Cladium Dry	0.17	0.14	0.21	0.00
Pond Dry-Cladium Dry	0.16	0.12	0.20	0.00
Eleocharis Dry-Eleocharis Wet	0.07	0.03	0.10	< 0.001
Pond Wet-Eleocharis Wet	0.21	0.18	0.24	0.00
Pond Dry-Eleocharis Wet	0.20	0.16	0.23	0.00
Pond Wet-Eleocharis Dry	0.14	0.11	0.17	0.00
Pond Dry-Eleocharis Dry	0.13	0.10	0.16	0.00
Pond Dry-Pond Wet	-0.01	-0.04	0.02	0.90

Table A.2.2. SIMPER output for the top 10 species contributing to variance between habitat-season levels for each pairwise comparison.

Comparison	Diet Category	Average	SD	Ratio	Average A	Average B	Cum. Sum.
Pond-Wet vs Pond-Dry	Palaemonetes	0.14	0.24	0.61	0.15	0.35	0.15
	Chironomid	0.10	0.18	0.55	0.35	0.16	0.26
	Cladocera	0.08	0.16	0.49	0.41	0.04	0.34
	Copepoda	0.07	0.15	0.47	0.41	0.05	0.42
	Misc.	0.05	0.14	0.35	0.08	0.14	0.47
	Misc..Fish	0.05	0.15	0.33	0.06	0.09	0.52
	Cyprinodont	0.05	0.15	0.32	0.05	0.06	0.57
	H..spp.	0.04	0.11	0.39	0.20	0.04	0.62
	Odonata	0.04	0.12	0.33	0.09	0.06	0.66
	Procambarus	0.03	0.12	0.25	0.05	0.01	0.69
Pond-Wet vs Spikerush-Wet	Cladocera	0.15	0.15	1.05	0.41	1.08	0.18
	Chironomid	0.15	0.15	1.04	0.35	1.17	0.35
	Copepoda	0.10	0.12	0.82	0.41	0.67	0.46
	H..spp.	0.07	0.10	0.70	0.20	0.47	0.55
	Misc.	0.05	0.10	0.46	0.08	0.25	0.60
	Ostracoda	0.04	0.07	0.56	0.10	0.31	0.64
	Mollusca	0.04	0.09	0.40	0.09	0.18	0.68
	Palaemonetes	0.03	0.08	0.35	0.15	0.03	0.72
	Diptera	0.03	0.06	0.43	0.07	0.16	0.75
	Misc..Fish	0.03	0.09	0.30	0.06	0.10	0.78
Pond-Wet vs Spikerush-Dry	Cladocera	0.14	0.15	0.96	0.41	0.87	0.16
	Chironomid	0.13	0.15	0.89	0.35	0.86	0.31
	Copepoda	0.09	0.12	0.72	0.41	0.50	0.41
	H..spp.	0.08	0.12	0.70	0.20	0.52	0.51
	Palaemonetes	0.06	0.13	0.45	0.15	0.15	0.57
	Misc.	0.04	0.11	0.38	0.08	0.18	0.62
	Ostracoda	0.03	0.07	0.49	0.10	0.23	0.66
	Mollusca	0.03	0.09	0.35	0.09	0.12	0.69
	Diptera	0.03	0.07	0.35	0.07	0.11	0.72
	Cyprinodont	0.02	0.09	0.26	0.05	0.04	0.75
Pond-Wet vs Sawgrass-Wet	Chironomid	0.19	0.16	1.19	0.35	1.34	0.22
	Cladocera	0.16	0.15	1.08	0.41	1.05	0.41
	Copepoda	0.08	0.11	0.69	0.41	0.33	0.50
	H..spp.	0.08	0.11	0.71	0.20	0.46	0.59
	Collembola	0.04	0.10	0.43	0.05	0.22	0.63
	Palaemonetes	0.03	0.09	0.38	0.15	0.04	0.67

	Misc.	0.03	0.08	0.38	0.08	0.14	0.71
	Diptera	0.03	0.06	0.41	0.07	0.13	0.74
	Mollusca	0.02	0.08	0.32	0.09	0.08	0.76
	Trichoptera	0.02	0.06	0.39	0.11	0.10	0.79
Pond-Wet vs Sawgrass-Dry	Cladocera	0.14	0.13	1.01	0.41	0.88	0.15
	Chironomid	0.13	0.14	0.93	0.35	0.90	0.30
	Copepoda	0.10	0.12	0.82	0.41	0.54	0.41
	H..spp.	0.08	0.11	0.76	0.20	0.56	0.50
	Misc.	0.07	0.15	0.48	0.08	0.31	0.58
	Ostracoda	0.03	0.07	0.46	0.10	0.21	0.62
	Palaemonetes	0.03	0.09	0.37	0.15	0.04	0.66
	Collembola	0.03	0.08	0.40	0.05	0.18	0.69
	Mollusca	0.03	0.07	0.40	0.09	0.13	0.73
	Diptera	0.03	0.06	0.42	0.07	0.14	0.76
Pond-Dry vs Spikerush-Dry	Cladocera	0.15	0.16	0.93	0.04	0.87	0.16
	Chironomid	0.14	0.16	0.88	0.16	0.86	0.31
	Palaemonetes	0.11	0.20	0.56	0.35	0.15	0.43
	H..spp.	0.08	0.13	0.65	0.04	0.52	0.52
	Copepoda	0.07	0.11	0.65	0.05	0.50	0.59
	Misc.	0.07	0.16	0.43	0.14	0.18	0.66
	Cyprinodont	0.04	0.12	0.29	0.06	0.04	0.70
	Ostracoda	0.03	0.07	0.47	0.03	0.23	0.74
	Misc..Fish	0.02	0.10	0.25	0.09	0.02	0.76
	Diptera	0.02	0.07	0.30	0.01	0.11	0.78
Pond-Dry vs Sawgrass-Dry	Cladocera	0.14	0.14	1.00	0.04	0.88	0.15
	Chironomid	0.14	0.15	0.93	0.16	0.90	0.30
	Misc.	0.09	0.18	0.52	0.14	0.31	0.40
	H..spp.	0.09	0.12	0.71	0.04	0.56	0.49
	Copepoda	0.09	0.11	0.75	0.05	0.54	0.58
	Palaemonetes	0.08	0.16	0.50	0.35	0.04	0.67
	Ostracoda	0.03	0.08	0.42	0.03	0.21	0.70
	Collembola	0.03	0.08	0.37	0.00	0.18	0.73
	Cyprinodont	0.03	0.09	0.29	0.06	0.04	0.76
	Misc..Fish	0.02	0.09	0.27	0.09	0.05	0.79
Spikerush-Wet vs Pond-Dry	Chironomid	0.17	0.16	1.05	1.17	0.16	0.18
	Cladocera	0.17	0.16	1.04	1.08	0.04	0.36
	Copepoda	0.09	0.13	0.75	0.67	0.05	0.46
	H..spp.	0.07	0.11	0.65	0.47	0.04	0.53
	Palaemonetes	0.07	0.15	0.49	0.03	0.35	0.61
	Misc.	0.06	0.13	0.50	0.25	0.14	0.68

	Ostracoda	0.04	0.08	0.54	0.31	0.03	0.72
	Mollusca	0.03	0.09	0.36	0.18	0.01	0.75
	Misc..Fish	0.03	0.10	0.30	0.10	0.09	0.79
	Diptera	0.02	0.06	0.40	0.16	0.01	0.81
Spikerush-Wet vs Spikerush-Dry	Chironomid	0.14	0.13	1.09	1.17	0.86	0.18
	Cladocera	0.13	0.12	1.05	1.08	0.87	0.36
	Copepoda	0.09	0.10	0.89	0.67	0.50	0.47
	H..spp.	0.08	0.10	0.82	0.47	0.52	0.58
	Misc.	0.05	0.10	0.51	0.25	0.18	0.64
	Ostracoda	0.04	0.06	0.67	0.31	0.23	0.70
	Mollusca	0.03	0.07	0.45	0.18	0.12	0.74
	Diptera	0.03	0.05	0.47	0.16	0.11	0.77
	Palaemonetes	0.03	0.07	0.35	0.03	0.15	0.81
	Hemiptera	0.02	0.04	0.37	0.06	0.09	0.83
Spikerush-Wet vs Sawgrass-Dry	Chironomid	0.13	0.12	1.10	1.17	0.90	0.18
	Cladocera	0.12	0.11	1.07	1.08	0.88	0.34
	Copepoda	0.09	0.09	0.94	0.67	0.54	0.45
	H..spp.	0.08	0.09	0.87	0.47	0.56	0.56
	Misc.	0.06	0.11	0.58	0.25	0.31	0.64
	Ostracoda	0.04	0.06	0.65	0.31	0.21	0.70
	Mollusca	0.03	0.06	0.49	0.18	0.13	0.74
	Collembola	0.03	0.06	0.45	0.10	0.18	0.78
	Diptera	0.03	0.05	0.54	0.16	0.14	0.81
	Misc..Fish	0.02	0.06	0.27	0.10	0.05	0.84
Spikerush-Wet vs Sawgrass-Wet	Chironomid	0.15	0.13	1.17	1.17	1.34	0.21
	Cladocera	0.13	0.11	1.13	1.08	1.05	0.39
	Copepoda	0.08	0.09	0.87	0.67	0.33	0.50
	H..spp.	0.07	0.08	0.85	0.47	0.46	0.60
	Misc.	0.04	0.08	0.53	0.25	0.14	0.66
	Collembola	0.03	0.07	0.47	0.10	0.22	0.70
	Ostracoda	0.03	0.06	0.59	0.31	0.08	0.75
	Mollusca	0.03	0.06	0.43	0.18	0.08	0.79
	Diptera	0.03	0.05	0.52	0.16	0.13	0.83
	Trichoptera	0.01	0.04	0.39	0.06	0.10	0.85

Spikerush-Dry vs Sawgrass-Dry	Chironomid	0.13	0.12	1.04	0.86	0.90	0.17
	Cladocera	0.12	0.12	1.03	0.87	0.88	0.33
	H..spp.	0.09	0.10	0.86	0.52	0.56	0.44
	Copepoda	0.08	0.09	0.90	0.50	0.54	0.55
	Misc.	0.07	0.13	0.52	0.18	0.31	0.64
	Ostracoda	0.04	0.07	0.59	0.23	0.21	0.69
	Palaemonetes	0.03	0.08	0.36	0.15	0.04	0.73
	Diptera	0.03	0.05	0.47	0.11	0.14	0.76
	Mollusca	0.02	0.05	0.49	0.12	0.13	0.79
	Collembola	0.02	0.06	0.36	0.01	0.18	0.82
Sawgrass-Wet vs Sawgrass-Dry	Chironomid	0.15	0.13	1.14	1.34	0.90	0.20
	Cladocera	0.13	0.12	1.09	1.05	0.88	0.38
	H..spp.	0.08	0.09	0.89	0.46	0.56	0.49
	Copepoda	0.07	0.08	0.87	0.33	0.54	0.58
	Misc.	0.05	0.10	0.54	0.14	0.31	0.66
	Collembola	0.04	0.08	0.52	0.22	0.18	0.72
	Ostracoda	0.03	0.05	0.50	0.08	0.21	0.75
	Diptera	0.03	0.05	0.52	0.13	0.14	0.79
	Mollusca	0.02	0.05	0.44	0.08	0.13	0.82
	Hemiptera	0.02	0.05	0.39	0.08	0.10	0.84
Sawgrass-Wet vs Spikerush-Dry	Chironomid	0.16	0.14	1.13	1.34	0.86	0.22
	Cladocera	0.14	0.13	1.07	1.05	0.87	0.40
	H..spp.	0.08	0.10	0.84	0.46	0.52	0.51
	Copepoda	0.07	0.09	0.80	0.33	0.50	0.60
	Misc.	0.04	0.09	0.46	0.14	0.18	0.66
	Collembola	0.03	0.08	0.39	0.22	0.01	0.70
	Palaemonetes	0.03	0.08	0.38	0.04	0.15	0.73
	Ostracoda	0.03	0.05	0.54	0.08	0.23	0.77
	Diptera	0.03	0.06	0.46	0.13	0.11	0.81
	Mollusca	0.02	0.05	0.39	0.08	0.12	0.83

Sawgrass-Wet vs Pond-Dry	Chironomid	0.23	0.18	1.26	1.34	0.16	0.24
	Cladocera	0.18	0.17	1.08	1.05	0.04	0.43
	Palaemonetes	0.08	0.15	0.53	0.04	0.35	0.52
	H..spp.	0.08	0.12	0.66	0.46	0.04	0.60
	Copepoda	0.06	0.11	0.58	0.33	0.05	0.66
	Misc.	0.05	0.11	0.43	0.14	0.14	0.71
	Collembola	0.04	0.10	0.40	0.22	0.00	0.76
	Cyprinodont	0.03	0.10	0.27	0.03	0.06	0.79
	Diptera	0.02	0.06	0.37	0.13	0.01	0.81
	Misc..Fish	0.02	0.08	0.24	0.02	0.09	0.83

Midge larvae (Chironomidae) were the most abundant prey item in sawgrass and spikerush, expect for dry-season spikerush where they were second (Fig. 2, Table A.2.2). Midge larvae were still an important prey item in pond prey communities (3rd and 2nd most abundant in the wet and dry seasons, respectively), but copepods and cladocerans were the two most abundant dry-season pond diet items. *Palaemonetes paludosus* was the most abundant wet-season pond diet item. Larger prey items such as fishes and dragonfly larvae were also more abundant in guts of dry-season pond consumers than consumers in other habitat-season levels.

A.2.2 Diet Descriptions

The relative importance of different feeding guilds varied among habitat-season levels. In ponds during the wet season, fishes belonged to all trophic guilds present in the Everglades. The majority were omnivores and carnivores, with few herbivores and detritivores. Only *Poecilia latipinna*, *Pelmatolapia mariae*, and larger *Gambusia holbrooki* were limited to feeding on algae and detritus. Two cyprinids collected only in ponds during the wet season, *Notropis maculatus* and *N. petersoni*, were omnivorous consumers that fed on algae, detritus, small invertebrates, and fish larvae (Table A.2.3).

The atherinid, *Labidesthes vanhyningi*, was also collected only in ponds during the wet season and was a strict carnivore. Dry-season pond fishes were generally either omnivores or carnivores. Higher-order consumers mostly fed on *Palaemonetes paludosus* and fishes, while detritus was consumed by many species, including *A. natalis*, *G. holbrooki*, and *Heterandria formosa*, that were primarily carnivorous or algivorous during the wet season.

In spikerush marshes during both seasons, most fishes collected were omnivores with the remaining species evenly representing other trophic guilds. Small-bodied poeciliids and fundulids that consumed invertebrates, algae, and detritus were most abundant. Higher-order consumers were mostly juveniles of large species. *Adinia xenica* and *Belonesox belizanus* were collected only during the dry season. Coupled with increased carnivory in *Fundulus confluentus* and *Lepomis punctatus*, I noted an increase in the number of species acting as omnivores (*A. xenica*) and displaying piscivory (*B. belizanus, F. confluentus,* and *L. punctatus*) during the dry season. In the wet season, few species consumed much algae, detritus, or fishes, while centrarchids fed heavily on decapods. Conversely, during the dry season, more small species consumed algae, while higher-order consumers ate fishes and decapods (predominantly *P. paludosus*).

Sawgrass food webs were similar between seasons because most consumers collected were small species or juveniles of large species. In the wet season, most were omnivores, herbivores, or detritivores. Higher-order consumers were juvenile *L. platyrhincus* and centrarchids during the wet season and *Fundulus chrysotus* during the dry season. In sawgrass marshes, decapods and fishes were less important diet items in both seasons relative to other habitats. Primary producers (algae) were mostly important in the diets of two taxa, *Jordanella floridae* and *P. latipinna*, while detritus was also consumed less in sawgrass marshes than in other habitats.

Table A.2.3. Mean dietary proportion for each of the 8 prey categories, trophic position, and trophic niche (SEA$_b$) per species size class present in each habitat-season level based on numeric stomach contents.

Species	Size Class	Habitat	Season	N	Producers	Detritus	Herb. Inverts	Omni. Inverts	Carn. Inverts	Decapods	Sm. Fish	Lg. Fish	Trophic Position	SEA$_b$
A_nat	1	Pond	1	22	0.00	0.05	0.00	0.04	0.07	0.02	0.00	0.00	2.12	0.44
A_nat	2	Pond	1	14	0.00	0.08	0.01	0.02	0.03	0.16	0.00	0.02	2.54	0.03
A_nat	3	Pond	1	9	0.02	0.17	0.02	0.00	0.02	0.16	0.00	0.02	2.46	1.20
A_nat	4	Pond	1	8	0.00	0.04	0.01	0.00	0.00	0.29	0.03	0.08	2.66	0.13
B_bel	1	Pond	1	2	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	3.50	-
B_bel	2	Pond	1	19	0.00	0.00	0.00	0.00	0.00	0.03	0.47	0.16	2.38	1.17
C_bat	1	Pond	1	17	0.00	0.03	0.01	0.05	0.04	0.02	0.00	0.00	2.43	0.81
C_bim	1	Pond	1	2	0.01	0.00	0.13	0.04	0.00	0.00	0.00	0.00	2.73	-
M_uro	1	Pond	1	6	0.00	0.09	0.11	0.00	0.01	0.12	0.03	0.00	3.11	0.11
M_uro	2	Pond	1	10	0.01	0.02	0.07	0.02	0.00	0.00	0.09	0.10	2.75	0.29
E_eve	1	Pond	1	12	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.21	0.68
E_fus	1	Pond	1	5	0.00	0.00	0.01	0.07	0.01	0.00	0.00	0.00	2.36	0.40
E_suc	1	Pond	1	2	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.49	-
E_suc	2	Pond	1	7	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.42	0.95
E_suc	3	Pond	1	8	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.30	0.93
F_chr	1	Pond	1	16	0.00	0.00	0.03	0.06	0.02	0.00	0.00	0.00	2.43	0.33
F_chr	2	Pond	1	5	0.00	0.00	0.03	0.07	0.00	0.00	0.00	0.00	2.92	-
F_con	1	Pond	1	4	0.00	0.00	0.03	0.06	0.01	0.00	0.00	0.00	2.51	0.22
G_hol	1	Pond	1	25	0.00	0.00	0.06	0.05	0.01	0.00	0.00	0.00	2.42	0.18
G_hol	2	Pond	1	25	0.04	0.01	0.09	0.04	0.01	0.00	0.00	0.00	2.45	0.01
H_for	1	Pond	1	8	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.54	0.26
H_for	2	Pond	1	6	0.00	0.01	0.00	0.08	0.00	0.00	0.00	0.00	2.49	0.01
J_flo	1	Pond	1	3	0.00	0.01	0.00	0.08	0.00	0.00	0.00	0.00	2.37	-
J_flo	2	Pond	1	14	0.02	0.03	0.00	0.07	0.00	0.00	0.00	0.00	2.84	0.56
L_goo	1	Pond	1	26	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.65	0.67
L_goo	2	Pond	1	17	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.40	0.64
L_gul	1	Pond	1	18	0.02	0.03	0.01	0.02	0.02	0.23	0.01	0.01	2.56	1.36
L_gul	2	Pond	1	23	0.00	0.02	0.03	0.01	0.04	0.24	0.01	0.00	2.40	0.20
L_gul	3	Pond	1	29	0.00	0.02	0.00	0.01	0.04	0.26	0.00	0.01	2.81	0.56
L_gul	4	Pond	1	14	0.00	0.04	0.00	0.01	0.04	0.30	0.05	0.00	2.50	0.54
L_mac	1	Pond	1	36	0.00	0.00	0.00	0.06	0.04	0.01	0.01	0.01	2.64	1.11
L_mar	1	Pond	1	22	0.00	0.03	0.01	0.07	0.00	0.00	0.00	0.00	2.36	0.00
L_mic	1	Pond	1	2	0.00	0.00	0.10	0.04	0.00	0.05	0.00	0.00	2.38	-
L_mic	2	Pond	1	15	0.00	0.03	0.20	0.00	0.01	0.00	0.00	0.01	2.39	0.48

L_pla	1	Pond	1	32	0.00	0.02	0.00	0.00	0.01	0.15	0.11	0.15	2.39	0.01
L_pla	2	Pond	1	22	0.00	0.07	0.00	0.01	0.01	0.09	0.05	0.18	2.47	0.66
L_pun	1	Pond	1	13	0.00	0.00	0.02	0.06	0.01	0.02	0.01	0.00	2.43	1.23
L_pun	2	Pond	1	10	0.00	0.00	0.03	0.05	0.01	0.06	0.11	0.00	2.69	0.03
L_pun	3	Pond	1	6	0.03	0.13	0.04	0.04	0.00	0.00	0.00	0.00	2.41	0.08
L_var	1	Pond	1	28	0.00	0.00	0.01	0.07	0.01	0.00	0.01	0.00	2.54	0.96
M_sal	1	Pond	1	24	0.00	0.00	0.00	0.00	0.00	0.25	0.25	0.05	2.48	0.10
N_cry	1	Pond	1	4	0.00	0.00	0.01	0.06	0.01	0.00	0.00	0.03	2.57	0.37
N_gyr	1	Pond	1	3	0.00	0.01	0.02	0.04	0.07	0.00	0.00	0.00	2.92	-
N_mac	1	Pond	1	13	0.03	0.01	0.00	0.07	0.00	0.00	0.00	0.00	2.56	0.73
N_pet	1	Pond	1	26	0.07	0.02	0.00	0.04	0.02	0.00	0.00	0.05	2.41	0.05
P_mar	1	Pond	1	26	0.05	0.38	0.00	0.01	0.00	0.00	0.00	0.00	2.38	0.10
A_cal	1	Pond	2	18	0.00	0.00	0.00	0.00	0.01	0.10	0.00	0.24	2.73	0.13
A_nat	1	Pond	2	43	0.00	0.32	0.00	0.01	0.00	0.11	0.05	0.00	2.39	0.51
A_nat	2	Pond	2	46	0.00	0.41	0.00	0.00	0.01	0.04	0.00	0.01	2.42	0.16
A_nat	3	Pond	2	19	0.00	0.33	0.00	0.00	0.02	0.05	0.00	0.02	2.46	0.06
A_nat	4	Pond	2	5	0.00	0.20	0.00	0.00	0.00	0.10	0.00	0.13	2.70	0.00
B_bel	1	Pond	2	7	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	2.71	-
B_bel	2	Pond	2	22	0.00	0.00	0.00	0.00	0.00	0.00	0.91	0.03	2.44	0.02
C_bat	1	Pond	2	14	0.00	0.17	0.00	0.03	0.02	0.04	0.00	0.04	2.42	1.80
C_bim	1	Pond	2	1	0.00	0.00	0.10	0.05	0.00	0.00	0.00	0.00	1.84	-
E_eve	1	Pond	2	6	0.01	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.45	0.36
F_chr	1	Pond	2	8	0.01	0.00	0.01	0.04	0.03	0.00	0.10	0.06	2.74	0.22
G_hol	1	Pond	2	25	0.00	0.11	0.00	0.06	0.00	0.00	0.00	0.00	2.50	0.02
G_hol	2	Pond	2	24	0.03	0.06	0.00	0.05	0.01	0.00	0.00	0.04	2.37	0.13
H_for	1	Pond	2	5	0.00	0.01	0.00	0.08	0.00	0.00	0.00	0.00	1.94	0.05
H_for	2	Pond	2	9	0.00	0.23	0.00	0.04	0.00	0.00	0.00	0.00	2.54	0.01
L_gul	1	Pond	2	6	0.00	0.01	0.00	0.01	0.00	0.24	0.00	0.11	2.90	0.24
L_gul	2	Pond	2	6	0.00	0.00	0.00	0.01	0.03	0.24	0.00	0.06	2.71	0.00
L_gul	3	Pond	2	12	0.00	0.01	0.01	0.01	0.04	0.25	0.03	0.00	2.28	0.40
L_gul	4	Pond	2	16	0.00	0.01	0.00	0.01	0.04	0.11	0.00	0.12	2.62	0.88
L_mac	1	Pond	2	3	0.00	0.00	0.00	0.05	0.01	0.00	0.00	0.11	2.48	-
L_mar	1	Pond	2	7	0.00	0.01	0.00	0.06	0.00	0.07	0.00	0.00	2.15	0.54
L_mic	1	Pond	2	2	0.00	0.01	0.00	0.00	0.00	0.49	0.00	0.00	2.26	-
L_pla	1	Pond	2	88	0.00	0.03	0.00	0.00	0.00	0.39	0.06	0.03	2.52	0.01
L_pla	2	Pond	2	4	0.08	0.00	0.00	0.00	0.00	0.19	0.00	0.13	2.65	0.09
L_pun	1	Pond	2	1	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.50	-
L_pun	2	Pond	2	42	0.01	0.02	0.01	0.03	0.01	0.16	0.01	0.03	2.43	0.03
L_pun	3	Pond	2	16	0.10	0.03	0.01	0.03	0.02	0.02	0.01	0.01	2.75	0.76
N_gyr	1	Pond	2	6	0.06	0.00	0.06	0.05	0.00	0.00	0.00	0.00	2.43	1.53
P_lat	1	Pond	2	2	0.00	0.50	0.00	0.00	0.00	0.00	0.00	0.00	1.75	-
P_lat	2	Pond	2	5	0.00	0.41	0.00	0.01	0.00	0.00	0.00	0.00	2.40	0.01
A_cal	1	Spikerush	1	1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.33	2.00	-
A_nat	1	Spikerush	1	26	0.01	0.02	0.03	0.05	0.01	0.01	0.00	0.01	2.49	0.79
A_nat	2	Spikerush	1	4	0.00	0.00	0.01	0.02	0.09	0.13	0.00	0.00	2.36	0.05
A_nat	3	Spikerush	1	1	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	3.00	-
C_bim	1	Spikerush	1	1	0.00	0.00	0.17	0.03	0.00	0.00	0.00	0.00	2.08	-
C_var	1	Spikerush	1	25	0.08	0.00	0.00	0.05	0.00	0.00	0.00	0.03	2.44	0.22

E_eve	1	Spikerush	1	22	0.00	0.00	0.02	0.07	0.00	0.00	0.00	0.00	2.51	0.88
E_fus	1	Spikerush	1	2	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.41	-
E_glo	1	Spikerush	1	34	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.46	0.38
F_chr	1	Spikerush	1	49	0.00	0.00	0.03	0.06	0.02	0.00	0.01	0.01	2.50	0.92
F_chr	2	Spikerush	1	23	0.00	0.00	0.09	0.03	0.01	0.00	0.01	0.04	2.53	0.13
F_con	1	Spikerush	1	27	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.33	0.26
F_con	2	Spikerush	1	2	0.00	0.00	0.03	0.04	0.00	0.00	0.00	0.11	2.75	-
G_hol	1	Spikerush	1	25	0.00	0.00	0.05	0.06	0.00	0.00	0.00	0.00	2.62	0.06
G_hol	2	Spikerush	1	26	0.00	0.00	0.03	0.04	0.01	0.00	0.00	0.08	2.51	0.01
H_for	1	Spikerush	1	25	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.54	0.24
H_for	2	Spikerush	1	25	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.45	0.01
J_flo	1	Spikerush	1	25	0.00	0.03	0.00	0.07	0.00	0.00	0.00	0.00	2.37	0.09
J_flo	2	Spikerush	1	25	0.02	0.02	0.00	0.07	0.00	0.00	0.00	0.00	2.59	0.09
L_goo	1	Spikerush	1	51	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.43	0.33
L_goo	2	Spikerush	1	35	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.49	0.01
L_gul	3	Spikerush	1	3	0.00	0.00	0.00	0.00	0.03	0.39	0.06	0.00	2.65	-
L_gul	4	Spikerush	1	1	0.00	0.05	0.00	0.00	0.00	0.46	0.00	0.00	2.38	-
L_mac	1	Spikerush	1	1	0.00	0.00	0.00	0.06	0.05	0.00	0.00	0.00	2.11	-
L_mar	1	Spikerush	1	42	0.00	0.00	0.00	0.07	0.00	0.01	0.00	0.00	2.57	0.26
L_mic	1	Spikerush	1	10	0.00	0.00	0.03	0.02	0.00	0.28	0.00	0.00	2.46	0.38
L_mic	2	Spikerush	1	1	0.00	0.00	0.25	0.00	0.00	0.00	0.00	0.00	2.50	-
L_pun	1	Spikerush	1	33	0.00	0.00	0.01	0.07	0.01	0.00	0.00	0.00	2.49	2.55
L_pun	2	Spikerush	1	18	0.00	0.00	0.03	0.05	0.02	0.04	0.04	0.00	2.41	1.14
L_pun	3	Spikerush	1	2	0.00	0.00	0.00	0.04	0.04	0.00	0.25	0.00	3.06	-
N_gyr	1	Spikerush	1	20	0.00	0.00	0.01	0.07	0.01	0.00	0.00	0.00	2.35	0.52
P_lat	1	Spikerush	1	11	0.22	0.00	0.00	0.03	0.00	0.00	0.00	0.00	2.20	0.14
P_lat	2	Spikerush	1	3	0.23	0.00	0.00	0.02	0.00	0.00	0.00	0.00	3.17	-
A_nat	1	Spikerush	2	27	0.00	0.02	0.01	0.01	0.02	0.25	0.03	0.03	2.52	0.03
A_xen	1	Spikerush	2	2	0.18	0.00	0.00	0.04	0.00	0.00	0.00	0.00	2.75	-
B_bel	1	Spikerush	2	3	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	2.96	-
B_bel	2	Spikerush	2	6	0.00	0.00	0.00	0.00	0.00	0.00	1.00	0.00	2.50	-
C_var	1	Spikerush	2	5	0.21	0.00	0.00	0.03	0.00	0.00	0.00	0.00	2.48	0.02
E_eve	1	Spikerush	2	24	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.60	0.93
E_glo	1	Spikerush	2	22	0.00	0.00	0.00	0.07	0.01	0.00	0.00	0.00	2.37	0.26
F_chr	1	Spikerush	2	50	0.00	0.00	0.01	0.06	0.03	0.02	0.00	0.00	2.57	0.22
F_chr	2	Spikerush	2	18	0.00	0.01	0.04	0.04	0.02	0.07	0.06	0.00	2.50	0.02
F_con	1	Spikerush	2	17	0.00	0.00	0.01	0.06	0.02	0.00	0.00	0.02	2.36	0.34
F_con	2	Spikerush	2	26	0.00	0.01	0.01	0.05	0.00	0.11	0.11	0.00	2.58	0.02
G_hol	1	Spikerush	2	23	0.02	0.00	0.00	0.07	0.01	0.00	0.00	0.00	2.46	0.05
G_hol	2	Spikerush	2	25	0.04	0.00	0.00	0.06	0.01	0.01	0.00	0.01	2.43	0.25
H_for	1	Spikerush	2	25	0.06	0.00	0.00	0.06	0.00	0.00	0.00	0.00	2.49	0.08
H_for	2	Spikerush	2	23	0.04	0.00	0.00	0.07	0.00	0.00	0.00	0.00	2.44	0.41
J_flo	1	Spikerush	2	5	0.09	0.00	0.00	0.06	0.00	0.00	0.00	0.00	2.44	0.05
J_flo	2	Spikerush	2	26	0.18	0.00	0.00	0.04	0.00	0.00	0.00	0.00	2.43	0.06
L_goo	1	Spikerush	2	41	0.00	0.00	0.01	0.07	0.01	0.00	0.00	0.00	1.15	0.52
L_goo	2	Spikerush	2	38	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.50	0.03
L_gul	1	Spikerush	2	1	0.00	0.00	0.00	0.00	0.00	0.33	0.33	0.00	3.00	-
L_gul	2	Spikerush	2	3	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	1.89	-

L_gul	4	Spikerush	2	2	0.00	0.00	0.00	0.00	0.01	0.46	0.00	0.00	2.46	-
L_mac	1	Spikerush	2	2	0.00	0.00	0.03	0.04	0.04	0.08	0.00	0.00	3.00	-
L_mar	1	Spikerush	2	57	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.53	0.25
L_mic	1	Spikerush	2	1	0.00	0.00	0.25	0.00	0.00	0.00	0.00	0.00	2.51	-
L_pun	1	Spikerush	2	34	0.00	0.00	0.01	0.06	0.02	0.04	0.01	0.00	2.41	0.28
L_pun	2	Spikerush	2	31	0.01	0.00	0.01	0.04	0.03	0.09	0.05	0.01	2.39	0.56
N_gyr	1	Spikerush	2	8	0.13	0.00	0.03	0.04	0.00	0.00	0.00	0.00	2.48	0.18
P_lat	2	Spikerush	2	10	0.27	0.00	0.00	0.01	0.00	0.00	0.00	0.00	2.42	0.21
A_nat	1	Sawgrass	1	7	0.00	0.01	0.00	0.05	0.04	0.05	0.00	0.00	2.09	0.22
A_nat	2	Sawgrass	1	2	0.00	0.00	0.14	0.01	0.03	0.06	0.00	0.00	2.94	-
E_eve	1	Sawgrass	1	2	0.00	0.00	0.00	0.07	0.00	0.00	0.00	0.03	2.85	-
F_chr	1	Sawgrass	1	47	0.00	0.00	0.04	0.06	0.01	0.00	0.00	0.00	2.51	0.44
F_chr	2	Sawgrass	1	15	0.00	0.00	0.05	0.05	0.02	0.01	0.00	0.00	2.50	0.10
F_con	1	Sawgrass	1	11	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.43	0.08
F_con	2	Sawgrass	1	2	0.00	0.00	0.01	0.08	0.00	0.00	0.00	0.00	2.04	-
G_hol	1	Sawgrass	1	26	0.01	0.00	0.04	0.06	0.01	0.00	0.00	0.00	2.34	0.04
G_hol	2	Sawgrass	1	25	0.00	0.00	0.04	0.06	0.00	0.00	0.00	0.01	2.44	0.00
H_for	1	Sawgrass	1	11	0.03	0.00	0.00	0.07	0.00	0.00	0.00	0.00	2.59	0.44
H_for	2	Sawgrass	1	25	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.44	0.03
J_flo	1	Sawgrass	1	4	0.01	0.00	0.00	0.07	0.00	0.00	0.00	0.00	2.52	0.05
J_flo	2	Sawgrass	1	16	0.01	0.04	0.00	0.07	0.00	0.00	0.00	0.00	2.42	0.64
L_goo	1	Sawgrass	1	22	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.49	0.13
L_goo	2	Sawgrass	1	38	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.19	0.05
L_gul	1	Sawgrass	1	2	0.00	0.02	0.00	0.00	0.00	0.25	0.46	0.00	2.50	-
L_gul	3	Sawgrass	1	1	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	1.20	-
L_gul	4	Sawgrass	1	1	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	2.43	-
L_mar	1	Sawgrass	1	14	0.00	0.00	0.00	0.08	0.00	0.00	0.01	0.00	2.21	1.19
L_pla	1	Sawgrass	1	11	0.00	0.00	0.00	0.00	0.00	0.14	0.64	0.03	2.54	0.01
L_pun	2	Sawgrass	1	5	0.00	0.00	0.02	0.06	0.02	0.02	0.00	0.00	2.40	0.57
L_pun	3	Sawgrass	1	4	0.00	0.00	0.04	0.02	0.04	0.12	0.06	0.00	2.51	0.88
N_gyr	1	Sawgrass	1	4	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.39	0.99
C_var	1	Sawgrass	2	1	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.00	-
E_eve	1	Sawgrass	2	5	0.00	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.26	0.42
F_chr	1	Sawgrass	2	60	0.00	0.00	0.04	0.05	0.02	0.01	0.03	0.00	2.48	0.02
F_chr	2	Sawgrass	2	16	0.00	0.00	0.03	0.04	0.03	0.08	0.08	0.00	2.56	0.39
F_con	1	Sawgrass	2	23	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.68	0.32
F_con	2	Sawgrass	2	10	0.00	0.00	0.02	0.06	0.00	0.04	0.05	0.00	2.66	0.02
G_hol	1	Sawgrass	2	18	0.00	0.00	0.05	0.05	0.02	0.00	0.00	0.00	2.51	0.13
G_hol	2	Sawgrass	2	25	0.02	0.00	0.01	0.06	0.01	0.00	0.01	0.04	2.54	0.27
H_for	1	Sawgrass	2	19	0.01	0.00	0.00	0.08	0.00	0.00	0.00	0.00	2.37	0.08
H_for	2	Sawgrass	2	25	0.02	0.00	0.00	0.07	0.00	0.00	0.00	0.00	2.50	0.37
J_flo	1	Sawgrass	2	6	0.12	0.00	0.00	0.05	0.00	0.00	0.00	0.00	2.51	0.41
J_flo	2	Sawgrass	2	25	0.02	0.00	0.00	0.07	0.00	0.00	0.00	0.00	2.57	0.06
L_goo	1	Sawgrass	2	21	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.00	0.20
L_goo	2	Sawgrass	2	49	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.43	0.13
L_gul	1	Sawgrass	2	1	0.00	0.00	0.00	0.00	0.00	0.50	0.00	0.00	2.34	-
L_mar	1	Sawgrass	2	1	0.00	0.00	0.01	0.07	0.00	0.00	0.00	0.00	2.83	-
L_pun	1	Sawgrass	2	10	0.00	0.00	0.01	0.07	0.01	0.02	0.01	0.00	2.47	0.03

L_pun	2	Sawgrass	2	5	0.00	0.00	0.01	0.04	0.03	0.00	0.22	0.00	2.42	0.11
L_pun	3	Sawgrass	2	1	0.01	0.00	0.06	0.02	0.00	0.00	0.49	0.00	2.51	-
N_gyr	1	Sawgrass	2	1	0.00	0.00	0.04	0.01	0.11	0.00	0.00	0.00	2.65	-
P_lat	1	Sawgrass	2	2	0.18	0.00	0.00	0.00	0.08	0.00	0.00	0.00	2.25	-
P_lat	2	Sawgrass	2	10	0.33	0.00	0.00	0.00	0.00	0.00	0.00	0.00	2.61	-

SIMMs revealed that consumers filled several functional feeding groups, that vascular plants were rarely consumed, and that detritus played an important role for some consumers. Based on our models, I found that invertebrates such as anisopteran naiads, *Belostoma*, and *Pelocoris* were carnivores (Table A.2.5). Meanwhile, both *P. paludosus* and *P. fallax* were omnivores that mostly consumed primary producers and detritus, while *Hyalella* spp. made the single largest percent contribution of any diet item (Table A.2.5). Notably, *P. fallax* is the only consumer that SIMMs indicate is directly consuming emergent vascular plants (10% of its diet). For fishes considered herbivorous in the Everglades, I either did not collect enough individuals for stable-isotope analysis (*J. floridae*) or were unable to generate a SIMM with a valid mixing polygon (*P. latipinna*, Table A.2.4). Furthermore, while omnivory was common in stomach contents, for most consumers, detritus and primary producers contributed less than 5% of the diet – less than the threshold for inclusion in SIMMs. As a result, SIMMs underestimated the prevalence of omnivory, with only two fishes, *A. natalis* and invasive *Mayaheros urophthalmus* displaying omnivory in SIMMs, where both consume detritus. Nevertheless, detritus contributed importantly to the diet of both species, 15% and 25% respectively, and most consumers fed across trophic levels.

SIMMs were occasionally improved by using informative priors based on stomach contents and often demonstrated increased piscivory relative to stomach contents. Four of ten fishes (*G. holbrooki*, *L. platyrhincus*, *L. gulosus*, and *A. natalis*) had their best-fit models with informative prior distributions derived from stomach-content data (Table A.2). *Fundulus chrysotus* and *G. holbrooki* were the only two smaller fish species with valid SIMMs. *Poecilia latipinna* and *H. formosa* fell outside the polygon of their potential prey items (Phillips et al., 2014), and I lacked data for *Lucania goodei*. For both *G. holbrooki* and *F. chrysotus*, SIMMs suggested fishes, specifically *G. holbrooki* and *H. formosa*, played a more important role in their diets than was reflected by stomach-content data. Interestingly, the *G. holbrooki* SIMM suggested cannibalism was an important part of their diet (24%) and, without including cannibalism, there was not a valid mixing polygon. Similarly, SIMMs suggested elevated diet importance of fishes and larger prey items for most consumers. Furthermore, *E. sucetta* was an important prey item for several predators, including invasive *B. belizanus* – a strict carnivore – for which they constituted 43% of the diet. Fish predation may be an episodic event missed by the snapshot nature of stomach analysis.

Table A.2.4. List of all SIMMs created per consumer with pD (number of model parameters), DIC, and ΔDIC among models. Invertebrate trophic guilds follow Rader (1994). Trophic guilds for fishes are more detailed than in Table A.1.3 (facilitated by higher taxonomic resolution of stable-isotope data relative to stomach contents) and are as follows: detritivore – largely consumes detritus, also includes algae (they are often inseparable in this system); herbivore – primarily consume vascular plants; omnivore – consumes a combination of detritus, algae, and/or vascular plants along with animal prey; invertivore – consumes zooplankton and zoobenthos along with macroinvertebrates; mesopredator – consumers invertebrates and small fishes; top predator – primarily consumes fishes, may also include larger invertebrates such as decapods and anisopteran naiads.

Consumer	Trophic Guild	Sources	SIMM	Prior	Valid Mixing Polygon	pD	DIC	ΔDIC
Anisoptera naiad	Predator	Chironomids, *Caenis, Pelocoris,* Cladocera	1	n	y*	8.4	33.8	4.1
		Chironomids, *Caenis, Pelocoris*	2	n	y*	6.2	29.7	0
Belostoma	Predator	Cladocera, *Caenis,* Anisoptera, *Pelocoris, Planorbella duryi, Palaemonetes paludosus,* Periphyton	1	n	y	5.1	39.6	0.1
		Cladocera, *Caenis,* Anisoptera, *Pelocoris, Planorbella duryi, Palaemonetes paludosus*	2	n	y	5.8	40.6	1.1
		Cladocera, *Caenis,* Anisoptera, *Pelocoris, Planorbella duryi*	3	n	y	5.1	39.5	0
		Cladocera, *Caenis,* Anisoptera, *Pelocoris*	4	n	y	5.7	40.5	1
Pelocoris	Predator	*Caenis*, Chironomids, *Hyalella* spp., Cladocera, Periphyton	1	n	y	14.3	12.8	0.5
		Caenis, Chironomids, *Hyalella* spp., Cladocera	2	n	y	10	12.8	0.5
		Chironomids, *Hyalella* spp., Cladocera	3	n	y	9.1	12.3	0
		Caenis, Hyalella spp., Cladocera	4	n	y	10	13	0.7

Palaemonetes paludosus	G, C/G, H	*Utricularia foliosa*, Green Algae, Periphyton, Flocculent Matter, Chironomids, *Hyalella* spp.	1	n	y^{p}	14.6	14.5	0
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, *Hyalella* spp.	2	n	y^{p}	15.4	27.1	12.6
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Chironomids	3	n	y^{p}*	10.1	33.7	19.2
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter	4	n	y^{p}*	8.9	35.9	21.4
Procambarus fallax	G, C/G, H	*Utricularia foliosa*, Green Algae, Periphyton, Flocculent Matter, *Caenis*, Anisoptera, *Planorbella duryi*, *Belostoma*, *Pelocoris*, *Hyalella* spp., Chironomids	0	n	n	-	-	-
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, *Caenis*, Anisoptera, *Planorbella duryi*, *Belostoma*, *Pelocoris*, *Hyalella* spp., Chironomids, Vascular Plants, Cladocera	1	n	y	14.3	37.1	9.2
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, *Caenis*, Anisoptera, *Planorbella duryi*, *Belostoma*, *Pelocoris*, *Hyalella* spp., Chironomids, Vascular Plants	2	n	y	11.6	36	8.1
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, *Caenis*, Anisoptera, *Planorbella duryi*, *Belostoma*, *Pelocoris*, *Hyalella* spp., Chironomids, Cladocera	3	n	y	19.1	43.6	15.7
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Vascular Plants	7	n	n	6.4	34.1	6.2
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Vascular Plants, *Hyalella* spp.	4	n	y	8	29.5	1.6

		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Vascular Plants, *Hyalella* spp., *Planorbella duryi*	5	n	y	8.7	30.9	3
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Vascular Plants, *Hyalella* spp., *Pelocoris*	6	n	y	7.5	27.9	0
		Utricularia foliosa, Green Algae, Periphyton, Flocculent Matter, Vascular Plants, *Hyalella* spp., *Pelocoris*, Chironomids	8	n	y	8.7	29.9	2
Funudulus chrysotus	Invertivore	*Palaemonetes paludosus*, *Hyalella* spp., *Planorbella duryi*, Cladocera, Chironomids	-	-	n	-	-	-
		Palaemonetes paludosus, *Hyalella* spp., *Planorbella duryi*, Chironomid, *Gambusia holbrooki*, *Heterandria formosa*	1	n	y	8.7	27.7	0
		Palaemonetes paludosus, *Hyalella* spp., *Planorbella duryi*, Chironomid, *Gambusia holbrooki*, *Heterandria formosa*	1.1	y	y	11.8	32.5	4.8
		Palaemonetes paludosus, *Hyalella* spp., *Planorbella duryi*, Cladocera, Chironomid, *Gambusia holbrooki, Heterandria formosa*	2	n	y	11	32.3	4.6
		Palaemonetes paludosus, *Hyalella* spp., *Planorbella duryi*, Cladocera, Chironomid, *Gambusia holbrooki, Heterandria formosa*	2.1	y	y	14.4	37.7	10
		Palaemonetes paludosus, Hyalella spp., Planorbella duryi, Chironomid, Cyprinodontiform	3	n	y*	10.5	30.7	3
		Palaemonetes paludosus, Hyalella spp., Planorbella duryi, Chironomid, Cyprinodontiform	3.1	y	y*	9.1	29.1	1.4
Gambusia holbrooki	Omnivore	Chironomids, Cladocera, *Heterandria formosa*, *Gambusia holbrooki*	1	n	y	5.6	51	1.7
		Chironomids, Cladocera, *Heterandria formosa,*	1.1	y	y	8.1	55	5.7

		Gambusia holbrooki						
		Chironomids, Cladocera, *Hyalella* spp., *Heterandria formosa, Gambusia holbrooki*	2	n	y	6.4	49.8	0.5
		Chironomids, Cladocera, *Hyalella* spp., *Heterandria formosa, Gambusia holbrooki*	2.1	y	y	6.9	49.3	0
Heterandria formosa	Omnivore	Chironomids, Cladocera, Periphyton	0	-	n	-	-	-
Lepisosteus platyrhincus	Top Predator	*Gambusia holbrooki, Heterandria formosa, Lepomis punctatus, Lepomis microlophus, Lepomis macrochirus, Ameiurus natalis, Procambarus*	1	n	y	9.3	25.9	0.8
		Gambusia holbrooki, Heterandria formosa, Lepomis punctatus, Lepomis microlophus, Lepomis macrochirus, Ameiurus natalis, Procambarus	1.1	y	y	9.2	25.4	0.3
		Cyprinodontiform, Lepomis punctatus, Lepomis microlophus, Lepomis macrochirus, Ameiurus natalis, Procambarus	2	n	y	8.9	25.3	0.2
		Cyprinodontiform, Lepomis punctatus, Lepomis microlophus, Lepomis macrochirus, Ameiurus natalis, Procambarus	2.1	y	y	10.1	27.9	2.8
		Gambusia holbrooki, Heterandria formosa, Lepomis macrochirus, Lepomis, Ameiurus natalis, Procambarus	3	n	y	9.1	25.8	0.7
		Gambusia holbrooki, Heterandria formosa, Lepomis macrochirus, Lepomis, Ameiurus natalis, Procambarus	3.1	y	y	9.1	25.1	0
Lepomis gulosus	Mesopredator	Herbivorous Fishes, *Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus*	1	n	n	8.9	36.8	5.6

Herbivorous Fishes, *Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus*	1.1	y	n	16.4	46.1	14.9
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus	2	n	n	8.5	34.6	3.4
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus	2.1	y	n	10.6	37.8	6.6
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta	3	n	n	7.8	33.7	2.5
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta	3.1	y	n	10.8	38.1	6.9
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	4	n	y*	7.5	31.2	0
Gambusia holbrooki, Heterandria formosa, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	4.1	y	y*	10.1	36.4	7.7
Gambusia holbrooki, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	5	n	y*	6.7	29.6	0.9
Gambusia holbrooki, Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	5.1	y	y*	6.9	28.7	0
Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	6	n	y*	7.2	30.7	2
Belostoma, Palaemonetes paludosus, Procambarus, Erimyzon sucetta, Lepomis macrochirus	6.1	y	y*	7.8	30.2	1.5

Lepomis macrochirus	Invertivore	*Gambusia holbrooki, Heterandria formosa, Hyalella* spp., Chironomids	-	-	n	-	-	-
		Gambusia holbrooki, Heterandria formosa, Hyalella spp., Chironomids	-	-	n	-	-	-
Lepomis microlophus	Invertivore	Anisoptera naiad, *Planorbella duryi*, Flocculent Matter, *Ameiurus natalis*	1	n	y	7.5	33.1	0.5
		Anisoptera naiad, *Planorbella duryi*, Flocculent Matter, *Ameiurus natalis*	1.1	y	y	5.9	33.1	0.5
		Procambarus, Palaemonetes paludosus, Hyalella spp., *Planorbella duryi*, Chironomids	2	n	n	8.8	32.6	0
		Procambarus, Palaemonetes paludosus, Hyalella spp., *Planorbella duryi*, Chironomids	2.1	y	n	12.4	38.5	5.9
Lepomis punctatus	Invertivore	Palaemonetes *paludosus*, Odonata, *Hyalella* spp., Chironomids	-	n	n	-	-	-
		Palaemonetes paludosus, Odonata, *Hyalella* spp., Chironomids, *Gambusia holbrooki*, *Heterandria formosa*	1	n	y	13.6	30	0
		Palaemonetes paludosus, Odonata, *Hyalella* spp., Chironomids, *Gambusia holbrooki*, *Heterandria formosa*	1.1	y	y	21.7	43.2	13.2
		Palaemonetes paludosus, Odonata, *Hyalella* spp., Chironomids, Cyprinodont	-	n	n	-	-	-
Micropterus salmoides	Top Predator	*Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki*	1	n	n	7.4	28.9	2.1
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki	1.1	y	n	8	29.4	2.6

Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Labidesthes vanhyningi, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	2	n	y	6.6	27.4	0.6
Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Labidesthes vanhyningi, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	2.1	y	y	7.5	28.8	2
Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	3	n	y	6.2	26.8	0
Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	3.1	y	y	7.7	29	2.2
Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	4	n	y	6.8	27.7	0.9
Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Lepomis microlophus, Lepomis punctatus, Lepomis macrochirus	4.1	y	y	7.9	29.4	2.6

		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Labidesthes vanhyningi, Lepomis macrochirus	5	n	y	6.9	27.9	1.1
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Labidesthes vanhyningi, Lepomis macrochirus	5.1	y	y	7.7	29.1	2.3
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Lepomis macrochirus	6	n	y	6.8	27.8	1
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Erimyzon sucetta, Lepomis macrochirus	6.1	y	y	8.1	29.6	2.8
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Lepomis macrochirus	7	n	y	7.4	28.8	2
		Procambarus, Palaemonetes paludosus, Heterandria formosa, Gambusia holbrooki, Lepomis macrochirus	7.1	y	y	7.9	29.5	2.7
Poecilia latipinna	Detritivore	Periphyton, Green Algae, Cladocerans	-	n	n	-	-	-

Ameiurus natalis	Omnivore	Procambarus, Palaemonetes paludosus, Flocculent Matter, Gambusia holbrooki, Heterandria formosa, Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus	1	n	y	7.5	35.5	3.2
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Gambusia holbrooki, Heterandria formosa, Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus*	1.1	y	y	9.2	36.6	4.3
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Gambusia holbrooki, Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus*	2	n	y	7.1	33.6	1.3
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Gambusia holbrooki, Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus*	2.1	y	y	6.7	32.3	0
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus*	3	n	y	6.9	33.9	1.6
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Lepomis microlophus, Lepomis macrochirus, Lepomis punctatus*	3.1	y	y	7.5	34.3	2
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Gambusia holbrooki, Lepomis microlophus*	4	n	y	7.2	33.3	1
		Procambarus, Palaemonetes paludosus, Flocculent Matter, *Gambusia holbrooki, Lepomis microlophus*	4.1	y	y	7.2	32.7	0.4

Belonesox belizanus	Top Predator	*Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa, Poecilia latipinna, Erimyzon sucetta*	1	n	y	11.1	16.1	3.1
		Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa, Poecilia latipinna, Erimyzon sucetta	1.1	y	y	9.2	15.3	2.3
		Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa, Erimyzon sucetta	2	n	y	8.8	13	0
		Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa, Erimyzon sucetta	2.1	y	y	5	17.7	4.7
		Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa	3	n	n	10.4	22.3	9.3
		Palaemonetes paludosus, Gambusia holbrooki, Heterandria formosa	3.1	y	n	11.8	46.5	33.5
Mayaheros urophthalmus	Omnivore	Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Heterandria formosa, Erimyzon sucetta*	1	n	y*	8.9	38	4.4
		Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Heterandria formosa, Erimyzon sucetta*	1.1	y	y*	6.3	37.4	3.8
		Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Erimyzon sucetta*	2	n	y*	7.6	35.4	1.8
		Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Erimyzon sucetta*	2.1	y	y*	6.4	36.8	3.2
		Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Erimyzon sucetta, Hyalella* spp.	3	n	y*	6.8	33.6	0
		Flocculent Matter, *Planorbella duryi*, Chironomids, *Gambusia holbrooki, Erimyzon sucetta, Hyalella* spp.	3.1	y	y*	9.5	38.6	5

Table A.2.5. Output from the best fit SIMM for each consumer including the mode (50% quantile), 95% credible interval (2.5% to 97.5% quantile range), mean, and standard deviation (SD) for each diet item in all models. Gelman-Rubin values near one indicate additional iterations of the model would not improve convergence.

Consumer	Diet Items	Gelman-Rubin	Quantile			Mean	SD
			2.5%	50%	97.5%		
Anisoptera Naiad	*Caenis*	1.001	0.13	0.52	0.81	0.51	0.17
	Chironomidae	1.001	0.04	0.28	0.72	0.30	0.18
	Pelocoris spp	1.001	0.03	0.17	0.46	0.19	0.11
Belostoma spp	*Caenis*	1.001	0.02	0.15	0.51	0.18	0.13
	Anisoptera naiad	1.001	0.02	0.18	0.61	0.22	0.16
	Pelocoris spp	1.001	0.02	0.18	0.61	0.22	0.16
	Cladocerans	1.001	0.02	0.11	0.24	0.12	0.06
	Planorbella duryi	1.001	0.02	0.23	0.64	0.26	0.18
Pelocoris spp	Chironomidae	1.001	0.33	0.47	0.59	0.46	0.06
	Hyalella spp.	1.001	0.16	0.29	0.42	0.29	0.06
	Cladocerans	1.002	0.22	0.24	0.27	0.25	0.01
Palaemonetes paludosus	Green Algae	1.001	0.04	0.22	0.37	0.21	0.09
	Utricularia foliosa	1.001	0.02	0.17	0.50	0.19	0.13
	Periphyton	1.001	0.01	0.05	0.19	0.06	0.05
	Flocculent Matter	1.001	0.01	0.05	0.19	0.06	0.05
	Hyalella spp.	1.001	0.18	0.49	0.70	0.47	0.12
Procambarus fallax	Green Algae	1.001	0.01	0.07	0.24	0.08	0.06
	Utricularia foliosa	1.001	0.01	0.08	0.31	0.10	0.08
	Periphyton	1.001	0.01	0.11	0.41	0.14	0.11
	Flocculent Matter	1.001	0.01	0.10	0.37	0.13	0.10
	Vascular Plants	1.001	0.02	0.10	0.21	0.10	0.05
	Hyalella spp.	1.001	0.02	0.21	0.54	0.23	0.14
	Pelocoris spp	1.001	0.02	0.17	0.64	0.22	0.17
Fundulus chrysotus	Chironomidae	1.001	0.02	0.13	0.34	0.14	0.09
	Hyalella spp.	1.001	0.01	0.08	0.28	0.10	0.07
	Palaemonetes paludosus	1.001	0.04	0.35	0.77	0.36	0.21
	Planorbella duryi	1.001	0.01	0.09	0.28	0.10	0.07
	Gambusia holbrooki	1.001	0.02	0.12	0.33	0.13	0.08
	Heterandria formosa	1.001	0.02	0.15	0.35	0.16	0.09
Gambusia holbrooki[p]	Chironomidae	1.001	0.04	0.19	0.39	0.19	0.09
	Hyalella spp.	1.001	0.12	0.32	0.53	0.33	0.11
	Cladocerans	1.001	0.01	0.03	0.08	0.04	0.02
	Gambusia holbrooki	1.001	0.06	0.24	0.45	0.24	0.10

Lepisosteus platyrhincus^{p}	*Heterandria formosa*	1.001	0.04	0.20	0.39	0.20	0.10
	Procambarus fallax	1.001	0.06	0.23	0.58	0.25	0.13
	Gambusia holbrooki	1.001	0.00	0.03	0.23	0.05	0.06
	Heterandria formosa	1.001	0.00	0.03	0.17	0.04	0.05
	Ameiurus natalis	1.001	0.05	0.37	0.74	0.38	0.19
	Lepomis microlophus	1.001	0.01	0.11	0.46	0.14	0.12
	Lepomis spp	1.001	0.01	0.11	0.43	0.14	0.11
	Belostoma spp	1.001	0.01	0.05	0.23	0.07	0.06
Lepomis gulosus^{p}	*Palaemonetes paludosus*	1.001	0.01	0.06	0.34	0.09	0.09
	Procambarus fallax	1.001	0.14	0.36	0.54	0.36	0.10
	Gambusia holbrooki	1.001	0.09	0.33	0.51	0.32	0.11
	Lepomis macrochirus	1.001	0.01	0.08	0.24	0.09	0.06
	Erimyzon sucetta	1.001	0.01	0.05	0.34	0.08	0.09
Lepomis microlophus	Chironomidae	1.001	0.01	0.08	0.24	0.09	0.06
	Hyalella spp.	1.001	0.02	0.15	0.44	0.17	0.11
	Palaemonetes paludosus	1.001	0.06	0.44	0.77	0.43	0.19
	Procambarus fallax	1.001	0.02	0.15	0.52	0.18	0.14
	Planorbella duryi	1.001	0.02	0.10	0.35	0.12	0.09
Lepomis punctatus	Chironomidae	1.001	0.01	0.06	0.22	0.08	0.06
	Hyalella spp.	1.001	0.05	0.29	0.45	0.27	0.10
	Anisoptera naiad	1.001	0.01	0.06	0.23	0.08	0.06
	Palaemonetes paludosus	1.001	0.01	0.09	0.42	0.12	0.11
	Gambusia holbrooki	1.001	0.06	0.34	0.50	0.32	0.12
	Heterandria formosa	1.001	0.01	0.10	0.36	0.12	0.09
Micropterus salmoides	*Palaemonetes paludosus*	1.001	0.01	0.12	0.39	0.15	0.11
	Procambarus fallax	1.001	0.02	0.12	0.26	0.12	0.07
	Gambusia holbrooki	1.001	0.01	0.07	0.32	0.10	0.08
	Heterandria formosa	1.001	0.01	0.07	0.27	0.09	0.07
	Labidesthes vanhyningi	1.001	0.01	0.07	0.26	0.09	0.07
	Lepomis microlophus	1.001	0.01	0.09	0.43	0.12	0.11
	Lepomis macrochirus	1.001	0.01	0.08	0.22	0.09	0.06
	Lepomis	1.001	0.01	0.07	0.32	0.10	0.08

	punctatus						
	Erimyzon sucetta	1.001	0.01	0.10	0.50	0.14	0.13
Ameiurus natalis^{p}	Flocculent Matter	1.001	0.02	0.15	0.35	0.16	0.10
	Palaemonetes paludosus	1.001	0.02	0.10	0.30	0.12	0.07
	Procambarus fallax	1.001	0.01	0.27	0.64	0.28	0.19
	Gambusia holbrooki	1.001	0.10	0.32	0.55	0.32	0.12
	Lepomis macrochirus	1.001	0.02	0.11	0.28	0.12	0.07
Belonesox belizanus	*Palaemonetes paludosus*	1.001	0.19	0.35	0.55	0.36	0.09
	Gambusia holbrooki	1.001	0.02	0.12	0.32	0.14	0.08
	Heterandria formosa	1.001	0.02	0.08	0.19	0.09	0.05
Mayaheros urophthalmus	*Erimyzon sucetta*	1.001	0.11	0.43	0.69	0.42	0.15
	Flocculent Matter	1.001	0.04	0.25	0.46	0.25	0.12
	Planorbella duryi	1.001	0.02	0.13	0.45	0.16	0.12
	Chironomidae	1.001	0.02	0.09	0.37	0.12	0.09
	Gambusia holbrooki	1.001	0.02	0.19	0.43	0.20	0.12
	Erimyzon sucetta	1.001	0.03	0.27	0.56	0.27	0.15

A.2.3 Basal Resource Use

Table A.2.6. Pairwise comparisons of detritus consumption among habitat-season levels (1 – wet season, 2 – dry season). Permuted ANOVA showed statistical differences among habitat-season levels (resample = 10,000, maxT = 16.0, p < 0.001).

Habitat-Season Comparison	Stat	p-value
Sawgrass1 - Sawgrass2	-1.611	0.107
Sawgrass1 - Spikerush1	-2.219	0.027
Sawgrass1 - Spikerush2	-1.444	0.149
Sawgrass1 - Pond1	-7.035	< 0.001
Sawgrass1 - Pond2	-10.690	< 0.001
Sawgrass2 - Spikerush1	-0.576	0.565
Sawgrass2 - Spikerush2	0.325	0.746
Sawgrass2 - Pond1	-6.247	< 0.001
Sawgrass2 - Pond2	-10.290	< 0.001
Spikerush1 - Spikerush2	1.050	0.294
Spikerush1 - Pond1	-7.349	< 0.001
Spikerush1 - Pond2	-12.450	< 0.001
Spikerush2 - Pond1	-7.993	< 0.001
Spikerush2 - Pond2	-12.840	< 0.001
Pond1 - Pond2	-6.046	< 0.001

A.2.4 Trophic Position

Permuted ANOVA revealed statistical differences in trophic position among species size classes (F_{48} = 1.37, p = 0.047), while habitat-season (F_5 = 0.46, p = 0.81) and the interaction of species size class and habitat-season (F_{81} = 0.92, p = 0.67) both showed no trends. Pairwise comparisons among species size classes resulted in fourteen species size classes having different trophic positions at different habitat-season levels. However, only two of these – *Lepomis gulosus* (3) in ponds and *Lepomis punctatus* (2) in sawgrass – changed trophic position between seasons within the same habitat. Similarly, I found eleven species size classes to have ontogenetic changes in trophic position, but only three of these were within the same habitat-season level (Table 3).

Table A.2.7. Trophic positions for all species sizes classes in each habitat-season level calculated with stomach contents and stable isotopes when possible. Stable-isotope trophic positions are modes of posterior distributions.

Abbrv	Species	Size Class	Wet Season									Dry Season									Species Mean S.C.	Species Mode S.I.
			Pond			Spikerush			Sawgrass			Pond			Eleocharis			Sawgrass				
			N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD	N	Mean	SD		
A_cal	*Amia calva*	1	-	-	-	1	2	-	-	-	-	18	2.73	0.56	-	-	-	-	-	-	2.70	-
A_nat	*Amiuruas natalis*	1	22	2.12	0.52	26	2.49	0.49	7	2.09	0.61	43	2.39	0.6	27	2.52	0.56	-	-	-	2.43	4.02
A_nat		2	14	2.54	0.54	4	2.36	0.3	2	2.94	0.09	46	2.42	0.65	-	-	-	-	-	-		
A_nat		3	9	2.46	0.83	1	3	-	-	-	-	19	2.46	0.64	-	-	-	-	-	-		
A_nat		4	8	2.66	0.34	-	-	-	-	-	-	5	2.7	0.45	-	-	-	-	-	-		
A_xen	*Adinia xenica*	1	-	-	-	-	-	-	-	-	-	-	-	-	2	2.75	0.35	-	-	-	2.75	-
B_bel	*Belonesox belizanus*	1	2	3.5	0	-	-	-	-	-	-	7	2.71	0.81	3	2.96	0.44	-	-	-	2.52	4.46
B_bel		2	19	2.38	0.56	-	-	-	-	-	-	22	2.44	0.79	6	2.5	0.45	-	-	-		
C_bim	*Cichlasoma bimaculatum*	1	2	2.73	0.24	1	2.08	-	-	-	-	1	1.84	-	-	-	-	-	-	-	2.35	3.37
C_bat	*Clarias batrachus*	1	17	2.43	0.62	-	-	-	-	-	-	14	2.42	0.68	-	-	-	-	-	-	2.42	3.58
C_var	*Cyprinodon variegatus*	1	-	-	-	25	2.44	0.76	-	-	-	-	-	-	5	2.48	0.36	1	2	-	2.43	-
E_eve	*Elassoma evergladei*	1	12	2.21	0.54	22	2.51	0.36	2	2.85	0.21	6	2.45	0.25	24	2.6	0.59	5	2.26	0.38	2.48	-
E_glo	*Enneacanthus gloriosus*	1	-	-	-	34	2.46	0.47	-	-	-	-	-	-	22	2.37	0.49	-	-	-	2.42	-
E_suc	*Erimyzon sucetta*	1	2	2.49	0.9	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.37	3.47
E_suc		2	7	2.42	0.52	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		
E_suc		3	8	2.3	0.47	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		
E_fus	*Etheostoma fusiforme*	1	5	2.36	0.25	2	2.41	0.45	-	-	-	-	-	-	-	-	-	-	-	-	2.37	-
F_chr	*Fundulus chrysotus*	1	16	2.43	0.68	49	2.5	0.53	47	2.51	0.47	8	2.74	0.29	50	2.57	0.47	60	2.48	0.45	2.52	4.10
F_chr		2	5	2.92	0.28	23	2.53	0.58	15	2.5	0.35	-	-	-	18	2.5	0.5	16	2.56	0.34		
F_con	*Fundulus confluentus*	1	4	2.51	0.55	27	2.33	0.45	11	2.43	0.57	-	-	-	17	2.36	0.53	23	2.68	0.38	2.50	-
F_con		2	-	-	-	2	2.75	0.69	2	2.04	1.01	-	-	-	26	2.58	0.61	10	2.66	0.53		
G_hol	*Gambusia*	1	25	2.42	0.6	25	2.62	0.49	26	2.34	0.51	25	2.5	0.59	23	2.46	0.6	18	2.51	0.38	2.46	4.08

G_hol	*holbrooki*	2	25	2.45	0.49	26	2.51	0.51	25	2.44	0.56	24	2.37	0.49	25	2.43	0.71	25	2.54	0.35		
H_for	*Heterandria formosa*	1	8	2.54	0.61	25	2.54	0.47	11	2.59	0.44	5	1.94	0.77	25	2.49	0.63	19	2.37	0.48	2.47	4.21
H_for		2	6	2.49	0.33	25	2.45	0.43	25	2.44	0.43	9	2.54	0.56	23	2.44	0.63	25	2.5	0.52		
J_flo	*Jordanella floridae*	1	3	2.37	1.2	25	2.37	0.54	4	2.52	1.03	-	-	-	5	2.44	0.83	6	2.51	0.44	2.51	3.70
J_flo		2	14	2.84	0.59	25	2.59	0.61	16	2.42	0.52	-	-	-	26	2.43	0.62	25	2.57	0.66		
L_sic	*Labidesthes vanhyningi*	1	28	2.54	0.48	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.54	2.43
L_pla	*Lepisosteus platyrhincus*	1	32	2.39	0.71	-	-	-	11	2.49	0.63	88	2.52	0.66	-	-	-	-	-	-	2.48	4.53
L_pla		2	22	2.47	0.59	-	-	-	-	-	-	4	2.65	0.24	-	-	-	-	-	-		
L_gul	*Lepomis gulosus*	1	18	2.56	0.54	-	-	-	2	2.19	1.14	6	2.9	0.37	1	1.15	-	1	2	-	2.56	4.12
L_gul		2	23	2.4	0.59	-	-	-	-	-	-	6	2.71	0.25	3	2.5	0.5	-	-	-		
L_gul		3	29	2.81	0.57	3	2.65	0.22	1	2.5	-	12	2.28	0.81	-	-	-	-	-	-		
L_gul		4	14	2.5	0.87	1	2.38	-	1	1.2	-	16	2.62	0.58	2	3	0.71	-	-	-		
L_mac	*Lepomis macrochirus*	1	36	2.64	0.4	1	2.11	-	-	-	-	3	2.48	0.53	2	1.89	1.04	-	-	-	2.58	3.84
L_mar	*Lepomis marginatus*	1	22	2.36	0.48	42	2.57	0.45	14	2.43	0.55	7	2.15	0.56	57	2.46	0.44	1	2.43	-	2.46	-
L_mic	*Lepomis microlophus*	1	2	2.38	0.16	10	2.46	0.66	-	-	-	2	2.26	0.34	1	3	-	-	-	-	2.42	3.74
L_mic		2	15	2.39	0.63	1	2.5	-	-	-	-	-	-	-	-	-	-	-	-	-		
L_pun	*Lepomis punctatus*	1	13	2.43	0.76	33	2.49	0.44	-	-	-	1	2.5	-	34	2.53	0.42	10	2.34	0.53	2.50	4.15
L_pun		2	10	2.69	0.34	18	2.41	0.7	5	2.21	0.49	42	2.43	0.59	31	2.51	0.66	5	2.83	0.2		
L_pun		3	6	2.41	0.66	2	3.06	0.2	4	2.54	0.28	16	2.75	0.41	-	-	-	1	2.47	-		
L_goo	*Lucania goodei*	1	26	2.65	0.55	51	2.43	0.4	22	2.4	0.45	-	-	-	41	2.41	0.48	21	2.42	0.52	2.46	-
L_goo		2	17	2.4	0.41	35	2.49	0.47	38	2.51	0.51	-	-	-	38	2.39	0.52	49	2.51	0.46		
C_uro	*Mayaheros urophthalmus*	1	6	3.11	0.35	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.88	3.66
C_uro		2	10	2.75	0.6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-		
M_sal	*Micropterus salmoides*	1	24	2.48	0.65	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.47	4.55
N_cry	*Notemigonus crysoleucas*	1	4	2.57	0.31	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.57	-
N_mac	*Notropis maculatus*	1	13	2.56	0.36	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.56	-
N_pet	*Notropis petersoni*	1	26	2.41	0.52	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.41	-
N_gyr	*Noturus*	1	3	2.92	0.1	20	2.35	0.52	4	2.39	0.31	6	2.43	0.93	8	2.48	0.69	1	2.65	-	2.44	-

	gyrinus																					
P_lat	*Poecilia latipinna*	1	-	-	-	11	2.2	0.61	-	-	-	2	1.75	1.06	-	-	-	2	2.25	0.35	2.42	3.45
P_lat		2	-	-	-	3	3.17	0.29	-	-	-	5	2.4	0.42	10	2.42	0.62	10	2.61	0.75		
T_mar	*Pelmatolapia mariae*	1	26	2.38	0.75	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2.38	3.68

A.2.5 Niche Breadth

Our analyses revealed a range of niche breadths among producers and consumers in this ecosystem. For the thirty taxa with stable-isotope data, SEA_b ranged from 0.03 for mysid shrimp (*Taphromysis louisianae*) to 7.38 for *U. foliosa* (Table A.15). These taxa represented eight trophic guilds: primary producers, omnivorous invertebrates, carnivorous invertebrates, decapods, herbivorous fishes, omnivorous fishes, mesopredators, and top predators. Of the nine primary producers, *U. foliosa* and *Bacopa caroliana* had the widest niche breadth while *Paspalidium geminatum* had the narrowest (Fig. A.2.1). There were two herbivorous invertebrates: *T. louisianae* had the smallest trophic niche measured by stable isotopes (0.032), and Cladocera with the fifth smallest trophic niche overall (0.127). Both carnivorous invertebrates (*Belostoma* spp. and anisopteran naiads) had larger niche breadths than either of the omnivorous decapods, for which *P. fallax* had a larger niche breadth than *P. palaemonetes*. Herbivorous fishes (*P. latipinna*, and *P. mariae*) had similar niche breadths. Conversely, omnivorous fishes had statistically different niche breadths. *G. holbrooki* had the broadest niche in this guild, followed by *F. chrysostus*, then *H. formosa*. *Lepomis macrochirus* had the largest SEA_b of any mesopredator, while *L. punctatus* had the narrowest. All remaining mesopredators were intermediate and statistically indistinct from one another. Among top predators, *L. platyrhincus* and *M. salmoides* had similar niche breadths and both were larger than that of *B. belizanus*.

Layman metrics revealed differences in trophic structure among feeding guilds (Fig. A.2.2). The largest $\delta^{15}N$ ranges belonged to primary producers and omnivorous invertebrates followed by decapods and mesopredators, while all other trophic guilds had smaller, indistinguishable $\delta^{15}N$. Similarly, the largest $\delta^{13}C$ ranges were measured in primary producers and omnivorous invertebrates. Unlike the $\delta^{15}N$ range, primary producers had a larger $\delta^{13}C$ range than omnivorous invertebrates. Mesopredators had the next largest $\delta^{13}C$ range while top predators had the smallest $\delta^{13}C$ range. Trophic diversity (CD) was greatest in primary producers and omnivorous invertebrates. Decapods and mesopredators displayed the next most trophic diversity, top predators had the least, and the remaining trophic guilds overlapped with all other guilds. Trophic redundancy (NND) was highest among omnivorous invertebrates, intermediate among decapods and primary producers, lowest among herbivorous fishes, omnivorous fishes, mesopredators and top predators, while carnivorous invertebrates overlapped with all guilds besides omnivorous invertebrates. Finally, the highest SDNND (evenness of trophic niches) was in primary producers, while other trophic guilds had lower, indistinguishable SDNNDs or had too few taxa to calculate.

Table A.2.8. *SIBER* package output for all species sizes classes across each habitat-season level including multiple measures of trophic niche (TA – convex polygon containing all individuals, SEA – 95% confidence ellipse of all consumers, SEA_c, SEA corrected for small sample sizes, and SEA_b – SEA modeled in a Bayesian framework). TA increases as sample size increases, SEA can be misleading when sample sizes are small, hence the creation of SEA_c, while SEA_b (really the MCMC algorithm) is robust against small sample sizes and allows for direct statistical comparisons between posterior distributions.

				Pond - SEA_b					Spikerush - SEA_b						Sawgrass - SEA_b					
Species	Size Class	Season	N	Mode	50% C.I.		95% C.I.		N	Mode	50% C.I.		95% C.I.		N	Mode	50% C.I.		95% C.I.	
Ameiurus natalis	1	1	22	0.44	0.38	0.50	0.30	0.65	26	0.79	0.64	1.07	0.41	1.89	7	0.22	0.20	0.26	0.17	0.33
	1	2	43	0.51	0.45	0.58	0.36	0.74	27	0.03	0.03	0.03	0.02	0.04	0	-	-	-	-	-
	2	1	14	0.03	0.03	0.04	0.02	0.06	4	0.05	0.04	0.06	0.03	0.08	2	-	-	-	-	-
	2	2	46	0.16	0.14	0.18	0.11	0.24	0	-	-	-	-	-	0	-	-	-	-	-
	3	1	9	1.20	1.05	1.38	0.82	1.84	3	-	-	-	-	-	0	-	-	-	-	-
	3	2	19	0.06	0.04	0.09	0.02	0.22	0	-	-	-	-	-	0	-	-	-	-	-
	4	1	8	0.13	0.11	0.16	0.08	0.24	0	-	-	-	-	-	0	-	-	-	-	-
	4	2	5	0.00	0.00	0.00	0.00	0.00	0	-	-	-	-	-	0	-	-	-	-	-
Amia calva	1	1	0	-	-	-	-	-	1	-	-	-	-	-	0	-	-	-	-	-
	1	2	18	0.13	0.10	0.16	0.07	0.24	0	-	-	-	-	-	0	-	-	-	-	-
Belonesox belizanus	1	1	2	-	-	-	-	-	0	-	-	-	-	-	0	-	-	-	-	-
	1	2	7	-	-	-	-	-	3	-	-	-	-	-	0	-	-	-	-	-
	2	1	19	1.17	0.97	1.40	0.68	2.07	0	-	-	-	-	-	0	-	-	-	-	-
	2	2	22	0.02	0.01	0.02	0.01	0.03	6	-	-	-	-	-	0	-	-	-	-	-
Cichlasoma bimaculatum	1	1	2	-	-	-	-	-	1	-	-	-	-	-	0	-	-	-	-	-
	1	2	1	-	-	-	-	-	0	-	-	-	-	-	0	-	-	-	-	-
Clarias batrachus	1	1	17	0.81	0.73	0.90	0.60	1.11	0	-	-	-	-	-	0	-	-	-	-	-
	1	2	14	1.79	1.52	2.14	1.10	3.07	0	-	-	-	-	-	0	-	-	-	-	-
Cyprinodon variegatus	1	1	0	-	-	-	-	-	25	0.22	0.15	0.32	0.08	0.79	0	-	-	-	-	-
	1	2	0	-	-	-	-	-	5	0.02	0.01	0.02	0.01	0.03	1	-	-	-	-	-
Elassoma evergladei	1	1	12	0.68	0.58	0.80	0.44	1.12	22	0.88	0.64	1.25	0.37	2.71	2	-	-	-	-	-
	1	2	6	0.36	0.34	0.39	0.30	0.45	24	0.93	0.67	1.42	0.36	3.45	5	0.42	0.31	0.56	0.18	1.09
Enneacanthus gloriosus	1	1	0	-	-	-	-	-	34	0.38	0.29	0.53	0.18	1.04	0	-	-	-	-	-
	1	2	0	-	-	-	-	-	22	0.26	0.19	0.34	0.10	0.67	0	-	-	-	-	-
Erimyzon	1	1	2	-	-	-	-	-	0	-	-	-	-	-	0	-	-	-	-	-

sucetta	2	1	7	0.95	0.79	1.14	0.57	1.71	0	-	-	-	-	-	0	-	-	-	-	-
	3	1	8	0.93	0.74	1.21	0.50	2.04	0	-	-	-	-	-	0	-	-	-	-	-
Etheostoma fusiforme	1	1	5	0.40	0.34	0.47	0.24	0.69	2	-	-	-	-	-	0	-	-	-	-	-
Fundulus confluentus	1	1	4	0.22	0.19	0.26	0.14	0.37	27	0.26	0.23	0.30	0.18	0.40	11	0.08	0.07	0.09	0.05	0.12
	1	2	0	-	-	-	-	-	17	0.34	0.27	0.42	0.18	0.68	23	0.32	0.24	0.43	0.14	0.83
	2	1	0	-	-	-	-	-	2	-	-	-	-	-	2	-	-	-	-	-
	2	2	0	-	-	-	-	-	26	0.02	0.02	0.03	0.02	0.04	10	0.02	0.01	0.02	0.01	0.03
Fundulus chrysotus	1	1	16	0.33	0.29	0.38	0.22	0.50	49	0.92	0.74	1.17	0.50	1.90	47	0.44	0.31	0.60	0.16	1.26
	1	2	8	0.22	0.19	0.26	0.14	0.37	50	0.22	0.20	0.26	0.15	0.34	60	0.39	0.28	0.55	0.16	1.18
	2	1	5	0.31	0.28	0.36	0.22	0.48	23	0.13	0.12	0.15	0.09	0.20	15	0.10	0.09	0.11	0.07	0.15
	2	2	0	-	-	-	-	-	18	0.02	0.02	0.02	0.02	0.03	16	0.04	0.03	0.04	0.03	0.05
Gambusia holbrooki	1	1	25	0.17	0.16	0.20	0.12	0.27	25	0.06	0.06	0.07	0.04	0.10	26	0.04	0.03	0.05	0.02	0.08
	1	2	25	0.02	0.01	0.02	0.01	0.06	23	0.05	0.04	0.06	0.03	0.08	18	0.13	0.10	0.16	0.07	0.25
	2	1	25	0.01	0.01	0.01	0.01	0.02	26	0.01	0.01	0.01	0.01	0.02	25	0.00	0.00	0.00	0.00	0.01
	2	2	23	0.13	0.12	0.14	0.10	0.18	25	0.25	0.21	0.31	0.15	0.45	25	0.27	0.21	0.36	0.13	0.66
Heterandria formosa	1	1	8	0.26	0.23	0.30	0.18	0.39	25	0.24	0.19	0.30	0.13	0.48	11	0.44	0.38	0.50	0.30	0.67
	1	2	5	0.05	0.04	0.05	0.03	0.07	25	0.08	0.07	0.09	0.06	0.13	19	0.08	0.07	0.10	0.06	0.13
	2	1	6	0.01	0.01	0.01	0.01	0.02	25	0.01	0.01	0.01	0.01	0.01	25	0.03	0.03	0.03	0.02	0.04
	2	2	9	0.01	0.01	0.01	0.01	0.02	23	0.41	0.37	0.45	0.31	0.54	25	0.37	0.33	0.44	0.25	0.58
Jordanella floridae	1	1	3	-	-	-	-	-	25	0.09	0.08	0.10	0.06	0.14	4	0.05	0.04	0.05	0.03	0.07
	1	2	0	-	-	-	-	-	5	0.05	0.04	0.05	0.04	0.06	6	0.41	0.38	0.45	0.32	0.54
	2	1	14	0.56	0.46	0.66	0.34	0.94	25	0.09	0.07	0.11	0.05	0.18	16	0.64	0.46	0.89	0.25	1.91
	2	2	0	-	-	-	-	-	26	0.06	0.04	0.08	0.02	0.18	25	0.05	0.05	0.06	0.04	0.08
Labidesthes vanhyningi	1	1	28	0.96	0.82	1.13	0.61	1.59	0	-	-	-	-	-	0	-	-	-	-	-
Lepisosteus platyrhincus	1	1	32	0.01	0.00	0.01	0.00	0.02	0	-	-	-	-	-	11	0.01	0.01	0.01	0.00	0.01
	1	2	88	0.01	0.01	0.01	0.01	0.02	0	-	-	-	-	-	0	-	-	-	-	-
	2	1	22	0.66	0.55	0.78	0.40	1.12	0	-	-	-	-	-	0	-	-	-	-	-
	2	2	4	0.09	0.07	0.14	0.04	0.29	0	-	-	-	-	-	0	-	-	-	-	-
Lepomis gulosus	1	1	18	1.35	1.21	1.58	0.94	2.08	0	-	-	-	-	-	2	-	-	-	-	-
	1	2	6	0.24	0.19	0.28	0.14	0.43	1	-	-	-	-	-	1	-	-	-	-	-
	2	1	23	0.20	0.15	0.26	0.09	0.47	0	-	-	-	-	-	0	-	-	-	-	-
	2	2	6	0.00	0.00	0.00	0.00	0.00	3	-	-	-	-	-	0	-	-	-	-	-

	3	1	29	0.56	0.47	0.66	0.35	0.92	0	-	-	-	-	-	1	-	-	-	-	-
	3	2	12	0.40	0.35	0.46	0.27	0.62	2	-	-	-	-	-	0	-	-	-	-	-
	4	1	14	0.54	0.48	0.61	0.38	0.80	1	-	-	-	-	-	1	-	-	-	-	-
	4	2	16	0.88	0.73	1.05	0.51	1.55	2	-	-	-	-	-	0	-	-	-	-	-
Lepomis macrochirus	1	1	36	1.11	1.00	1.25	0.81	1.58	1	-	-	-	-	-	0	-	-	-	-	-
	1	2	3	-	-	-	-	-	2	-	-	-	-	-	0	-	-	-	-	-
Lepomis marginatus	1	1	22	0.00	0.00	0.00	0.00	0.00	42	0.26	0.20	0.32	0.13	0.55	14	1.19	0.96	1.60	0.62	2.80
	1	2	7	0.54	0.49	0.60	0.40	0.74	57	0.25	0.22	0.27	0.19	0.34	1	-	-	-	-	-
Lepomis microlophus	1	1	1	-	-	-	-	-	10	0.38	0.32	0.46	0.22	0.66	0	-	-	-	-	-
	1	2	2	-	-	-	-	-	1	-	-	-	-	-	0	-	-	-	-	-
	2	1	15	0.48	0.41	0.60	0.29	0.91	1	-	-	-	-	-	0	-	-	-	-	-
Lepomis punctatus	1	1	13	1.23	0.99	1.55	0.67	2.53	33	2.55	1.94	3.52	1.16	6.98	0	-	-	-	-	-
	1	2	1	-	-	-	-	-	34	0.28	0.24	0.33	0.19	0.44	10	0.03	0.03	0.04	0.02	0.06
	2	1	10	0.03	0.02	0.05	0.01	0.10	18	1.14	0.97	1.35	0.72	1.93	5	0.57	0.50	0.66	0.40	0.88
	2	2	42	0.76	0.56	1.01	0.33	1.97	31	0.56	0.45	0.70	0.29	1.13	5	0.11	0.10	0.13	0.07	0.19
	3	1	6	0.08	0.07	0.09	0.05	0.12	2	-	-	-	-	-	4	0.88	0.76	1.02	0.58	1.37
	3	2	16	0.12	0.09	0.18	0.05	0.38	0	-	-	-	-	-	1	-	-	-	-	-
Lucania goodei	1	1	26	0.67	0.58	0.76	0.44	1.00	51	0.33	0.23	0.49	0.12	1.20	22	0.12	0.11	0.15	0.08	0.20
	1	2	0	-	-	-	-	-	41	0.52	0.42	0.66	0.28	1.07	21	0.20	0.18	0.22	0.14	0.28
	2	1	17	0.64	0.54	0.75	0.39	1.05	35	0.01	0.01	0.02	0.01	0.02	38	0.05	0.04	0.06	0.03	0.08
	2	2	0	-	-	-	-	-	38	0.03	0.02	0.03	0.02	0.04	49	0.13	0.12	0.15	0.09	0.21
Mayaheros urophthalmus	1	1	6	0.11	0.09	0.16	0.05	0.31	0	-	-	-	-	-	0	-	-	-	-	-
	2	1	10	0.29	0.24	0.34	0.18	0.47	0	-	-	-	-	-	0	-	-	-	-	-
Micropterus salmoides	1	1	22	0.10	0.07	0.14	0.04	0.35	0	-	-	-	-	-	0	-	-	-	-	-
Notemigonus crysoleucas	1	1	4	0.37	0.34	0.41	0.28	0.50	0	-	-	-	-	-	0	-	-	-	-	-
Noturus gyrinus	1	1	3	-	-	-	-	-	20	0.52	0.41	0.68	0.26	1.19	4	0.99	0.88	1.12	0.70	1.43
	1	2	6	1.55	1.34	1.80	1.02	2.45	8	0.18	0.16	0.21	0.12	0.29	1	-	-	-	-	-
Notropis maculatus	1	1	13	0.73	0.63	0.85	0.48	1.14	0	-	-	-	-	-	0	-	-	-	-	-
Notropis petersoni	1	1	26	0.05	0.05	0.06	0.04	0.08	0	-	-	-	-	-	0	-	-	-	-	-

Pelmatolapia mariae	1	1	26	0.09	0.07	0.12	0.05	0.23	0	-	-	-	-	-	0	-	-	-	-	-
Poeilia latipinna	1	1	0	-	-	-	-	-	11	0.14	0.12	0.19	0.08	0.29	0	-	-	-	-	-
	2	1	2	-	-	-	-	-	0	-	-	-	-	-	2	-	-	-	-	-
	1	2	0	-	-	-	-	-	3	-	-	-	-	-	0	-	-	-	-	-
	2	2	5	0.01	0.01	0.01	0.00	0.03	10	0.21	0.15	0.30	0.09	0.64	10	-	-	-	-	-

Figure A.2.1. SEA$_b$ per taxon derived from stable-isotope data by trophic guild (A – primary producers, B – herbivorous invertebrates, C – carnivorous invertebrates, D – decapods, E – herbivorous fishes, F – invertivorous fishes, G – mesopredators, H – top predators).

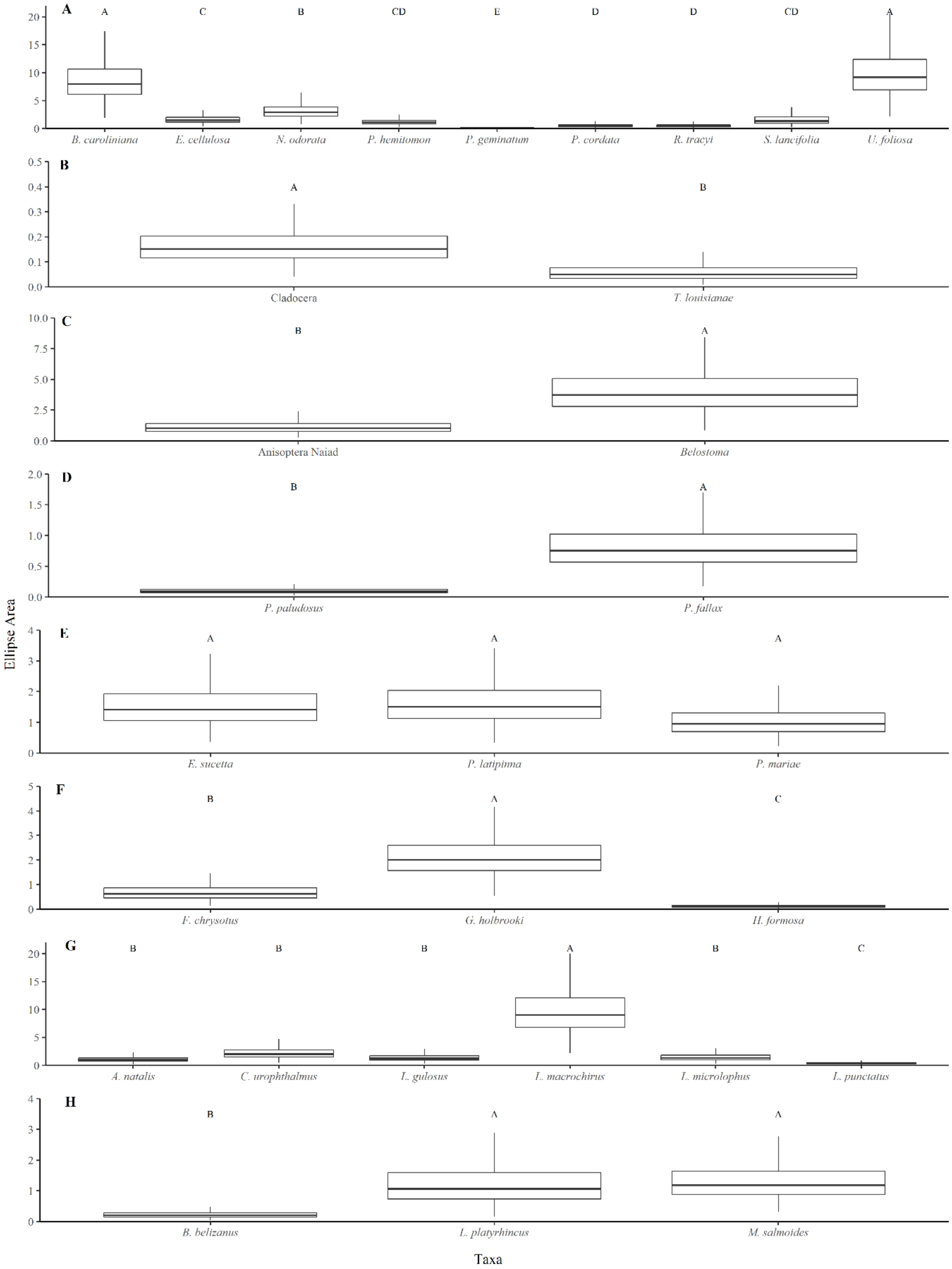


Figure A.2.2. Layman metrics per trophic guild derived from stable isotopes. Missing values had too few taxa per trophic guild to be calculated.

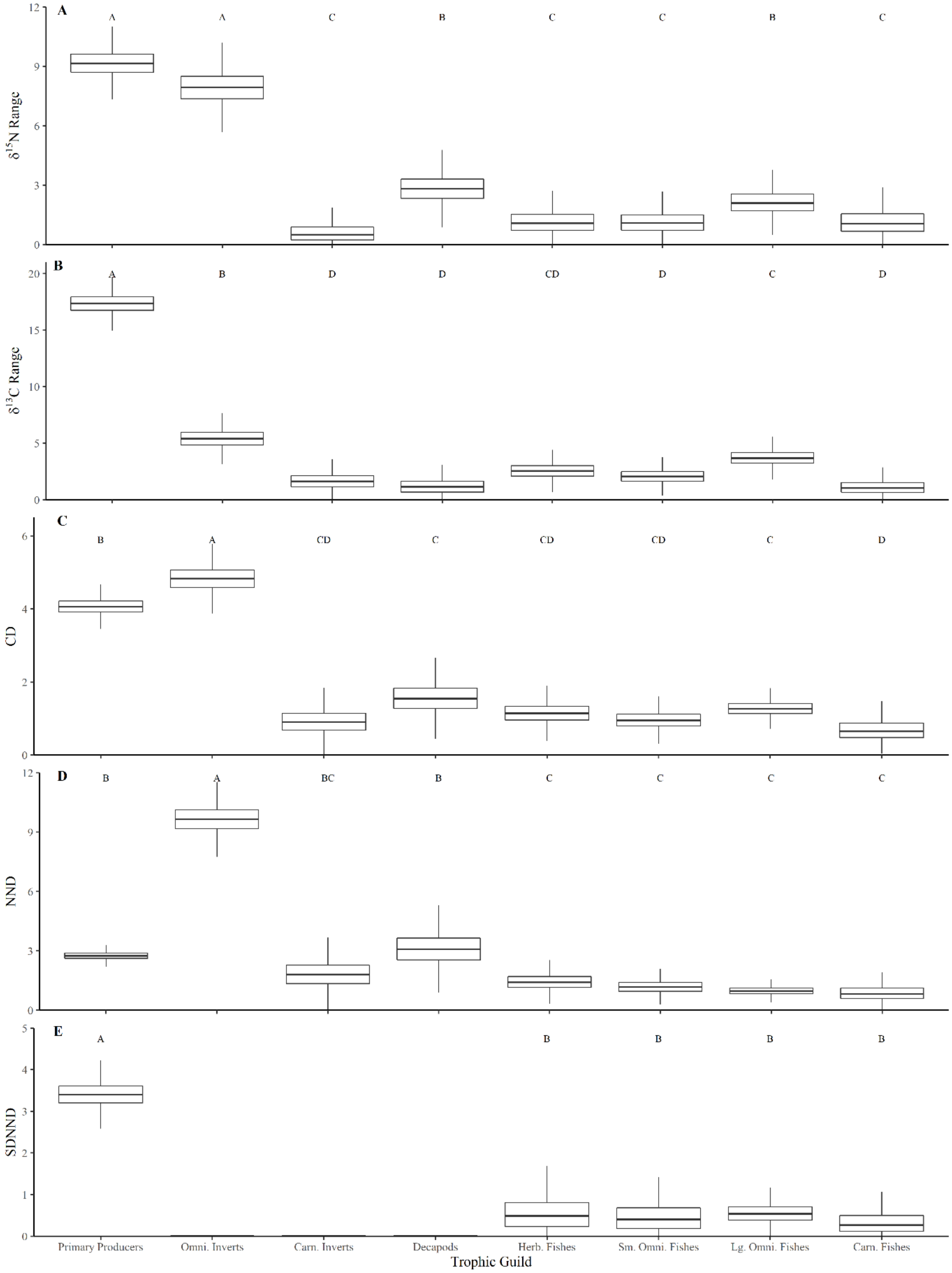


Table A.2.9. SEA_b mode and 95% credibility interval (CI) for each taxon in the isotope data set with sufficient sample size ($n \geq 3$).

Taxa	SEA_b mode	95% CI
Bacopa caroliniana	6.616	2.89 - 17.6
Eleocharis cellulosa	1.252	0.55 - 3.27
Nymphaea odorata	2.399	1.05 - 6.34
Panicum hemitomon	0.919	0.42 - 2.47
Paspalidium geminatum	0.047	0.01 - 0.21
Pontedaria cordata	0.312	0.09 - 1.53
Rhyncospora tracyii	0.343	0.16 - 1.44
Sagittaria lancifolia	0.939	0.27 - 4.32
Utricularia foliosa	7.381	3.06 - 22.6
Cladocera	0.127	0.06 - 0.33
Taphromysis louisianae	0.032	0.01 - 0.16
Anisoptera naiads	0.774	0.34 - 2.43
Belostoma spp	3.026	1.21 - 9.07
Palaemonetes paludosus	0.069	0.03 - 0.21
Procambarus fallax	0.605	0.25 - 1.93
Fundulus chrysotus	0.480	0.20 - 1.50
Gambusia holbrooki	1.713	0.81 - 4.19
Heterandria formosa	0.090	0.04 - 0.28
Ameiurus natalis	0.786	0.31 - 2.31
Mayaheros urophthalmus	1.525	0.66 - 4.72
Lepomis gulosus	0.965	0.42 - 2.97
Lepomis macrochirus	7.143	3.00 - 22.1
Lepomis microlophus	1.069	0.42 - 3.15
Lepomic punctatus	0.274	0.12 - 0.86
Erimyzon sucetta	1.143	0.46 - 3.46
Poecilia latipinna	1.215	0.50 - 3.69
Pelmatolapia mariae	0.698	0.32 - 2.25
Belonesox belizanus	0.163	0.01 - 0.49
Lepisosteus platyrhincus	0.714	0.23 - 3.61
Micropterus salmoides	0.959	0.39 - 2.86

A.2.6 Non-Native vs Native Niche Overlap

There are seventeen non-native species and thirty-four native species of freshwater fishes in Everglades National Park (Kline et al., 2013). Unsurprisingly, invasive fishes have recently become of elevated interest to managers. Ecosystem restoration has not always taken invasive fishes into account and is at least partially responsible for their spread (Kline et al., 2013). Our data were collected prior to the invasion of most of these fishes. Five non-native species (*Belonesox belizanus, Clarias batrachus, Cichlasoma bimaculatum, Mayaherous urophthalmus*, and *Pelmatolapia mariae*) were present by the mid-1990s and are represented in these data. The trophic overlap between these taxa and native species will be used to form hypotheses about species abundance through time and to serve as a baseline condition for more recent invasions. African Jewelfish (*Hemichromis letourneuxi*) and Asian Swamp Eels (*Monopterus albus/javanensis*) have invaded Everglades National Park in recent decades with negative impacts on the densities of native fishes (Trexler unpublished data).

Table A.2.10. Percent overlap in SEA$_{b}$ between native and non-native species using A) stomach contents and B) stable isotopes.

Stomach Contents						
Group A		Group B		Habitat	Season	Overlap
Species	Size Class	Species	Size Class			
Clarias batrachus	-	*Lepomis punctatus*	3	Pond	Dry	22.9%
Mayaheros urophthalmus	2	*Lepomis punctatus*	3	Pond	Wet	22.9%
Mayaheros urophthalmus	1	*Lepomis punctatus*	3	Pond	Wet	22.9%
Belonesox belizanus	2	*Lepisosteus platyrhincus*	2	Pond	Wet	10.9%
Clarias batrachus	-	*Lepomis punctatus*	2	Pond	Wet	9.6%
Mayaheros urophthalmus	2	*Lepomis punctatus*	2	Pond	Wet	9.5%

Mayaheros urophthalmus	1	*Lepomis punctatus*	2	Pond	Wet	9.3%
Clarias batrachus	-	*Ameiurus natalis*	4	Pond	Dry	9.0%
Mayaheros urophthalmus	2	*Lepomic macrochirus*	-	Pond	Wet	7.9%
Mayaheros urophthalmus	1	*Lepomis macrochirus*	-	Pond	Wet	7.7%
Belonesox belizanus	2	*Ameiurus natalis*	3	Pond	Wet	7.6%
Belonesox belizanus	2	*Ameiurus natalis*	2	Pond	Wet	7.4%
Belonesox belizanus	2	*Lepisosteus platyrhincus*	1	Pond	Wet	7.2%
Clarias batrachus	-	*Ameiurus natalis*	3	Pond	Wet	7.0%
Clarias batrachus	-	*Ameiurus natalis*	2	Pond	Wet	7.0%
Mayaheros urophthalmus	2	*Lepomis gulosus*	4	Pond	Wet	6.7%
Mayaheros urophthalmus	1	*Lepomis gulosus*	4	Pond	Wet	6.4%
Mayaheros urophthalmus	2	*Ameiurus natalis*	1	Pond	Wet	5.5%
Mayaheros urophthalmus	1	*Ameiurus natalis*	1	Pond	Wet	5.4%
Clarias batrachus	-	*Mayaheros urophthalmus*	1	Pond	Wet	5.2%
Clarias batrachus	-	*Ameiurus natalis*	1	Pond	Wet	5.2%
Belonesox belizanus	2	*Ameiurus natalis*	4	Pond	Wet	5.0%

Clarias batrachus	-	*Ameiurus natalis*	3	Pond	Wet	4.9%
Clarias batrachus	-	*Mayaheros urophthalmus*	2	Pond	Wet	4.5%
Clarias batrachus	-	*Ameiurus natalis*	4	Pond	Wet	4.4%
Mayaheros urophthalmus	1	*Lepomis gulosus*	1	Pond	Wet	4.3%
Mayaheros urophthalmus	2	*Lepomis gulosus*	1	Pond	Wet	4.2%
Mayaheros urophthalmus	2	*Lepomis punctatus*	1	Pond	Wet	4.1%
Mayaheros urophthalmus	1	*Lepomis gulosus*	3	Pond	Wet	3.9%
Mayaheros urophthalmus	2	*Lepomis gulosus*	3	Pond	Wet	3.8%
Mayaheros urophthalmus	1	*Lepomis punctatus*	1	Pond	Wet	3.8%
Clarias batrachus	-	*Ameiurus natalis*	1	Pond	Dry	3.6%
Mayaheros urophthalmus	1	*Lepomis gulosus*	2	Pond	Wet	3.1%
Mayaheros urophthalmus	2	*Lepomis gulosus*	2	Pond	Wet	3.1%
Mayaheros urophthalmus	1	*Lepomis microlophus*	2	Pond	Wet	3.0%
Mayaheros urophthalmus	2	*Lepomis microlophus*	2	Pond	Wet	3.0%
Clarias batrachus	-	*Lepomis marginatus*	-	Pond	Wet	2.2%

Mayaheros urophthalmus	1	*Lepomis marginatus*	-	Pond	Wet	1.9%
Belonesox belizanus	2	*Micropterus salmoides*	-	Pond	Wet	1.9%
Clarias batrachus	-	*Ameiurus natalis*	2	Pond	Dry	1.6%

Stable Isotopes

Group A	Group B	Overlap
Mayaheros urophthalmus	*Lepomis macrochirus*	64.0%
Ameiurus natalis	*Mayaheros urophthalmus*	15.4%
Gambusia holbrooki	*Mayaheros urophthalmus*	15.2%
Poecilia latipinna	*Mayaheros urophthalmus*	15.0%
Heterandria formosa	*Mayaheros urophthalmus*	14.7%
Mayaheros urophthalmus	*Erimyzon sucetta*	11.2%
Mayaheros urophthalmus	*Lepomis microlophus*	10.3%
Mayaheros urophthalmus	*Lepomis gulosus*	9.5%
Mayaheros urophthalmus	*Micropterus salmoides*	8.9%
Lepomis microlophus	*Pelmatolapia mariae*	7.7%
Erimyzon sucetta	*Pelmatolapia mariae*	7.6%
Mayaheros urophthalmus	*Pelmatolapia mariae*	7.5%
Lepomis punctatus	*Pelmatolapia mariae*	7.5%
Ameiurus natalis	*Pelmatolapia mariae*	7.4%
Lepomis gulosus	*Pelmatolapia mariae*	7.3%
Gambusia holbrooki	*Pelmatolapia mariae*	7.3%
Lepomis macrochirus	*Pelmatolapia mariae*	7.1%
Poecilia latipinna	*Pelmatolapia mariae*	7.1%
Heterandria formosa	*Pelmatolapia mariae*	7.1%
Micropterus salmoides	*Pelmatolapia mariae*	7.1%
Micropterus salmoides	*Belonesox belizanus*	1.7%
Micropterus salmoides	*Belonesox belizanus*	1.7%
Lepomis gulosus	*Belonesox belizanus*	1.6%

Gambusia holbrooki	*Belonesox belizanus*	1.6%
Lepomis punctatus	*Belonesox belizanus*	1.6%
Lepomis microlophus	*Belonesox belizanus*	1.5%
Lepomis microlophus	*Belonesox belizanus*	1.5%
Lepomis macrochirus	*Belonesox belizanus*	1.5%
Ameiurus natalis	*Belonesox belizanus*	1.5%
Mayaheros urophthalmus	*Belonesox belizanus*	1.5%
Erimyzon sucetta	*Belonesox belizanus*	1.5%
Heterandria formosa	*Belonesox belizanus*	1.5%
Poecilia latipinna	*Belonesox belizanus*	1.5%
Pelmatolapia mariae	*Belonesox belizanus*	1.5%
Mayaheros urophthalmus	*Lepomis punctatus*	1.2%
Belonesox belizanus	*Lepisosteus platyrhincus*	0.4%
Mayaheros urophthalmus	*Lepisosteus platyrhincus*	0.3%
Pelmatolapia mariae	*Lepisosteus platyrhincus*	0.3%

A.2.7 Ontogenetic Shifts in Trophic Dynamics

Within the eleven species that underwent ontogenetic shifts in diet, nine intraspecific size class comparisons were differentiable statistically in at least one habitat-season level but were not differentiable in at least one other habitat-season level. For example, *G. holbrooki* underwent an ontogenetic shift in all habitat-season levels, except for dry-season ponds (Fig. A.2.3). In some instances, direction and magnitude of ontogenetic shifts differed among habitat-season levels. Larger *G. holbrooki* in wet-season ponds demonstrated increased detritivory and herbivory compared to smaller individuals. However, in all other habitat-season levels where *G. holbrooki* displayed an ontogenetic shift, it resulted, at least in part, from increased piscivory.

Table A.2.11. Pairwise PERMANOVA output among species size classes within a habitat-season level (A – wet-season ponds, B – dry-season ponds, C – wet-season spikerush, D – dry-season spikerush, E – wet-season sawgrass, F – dry-season sawgrass). Each matrix is symmetric, lower triangles are p-values, upper triangles are R^2 values, and diagonal has been left empty. Bolded values indicate intraspecific ontogenetic diet comparisons.

A.	A_nat1	A_nat2	A_nat3	A_nat4	B_bel1	B_bel2	C_bat	C_bim	C_uro1	C_uro2	E_eve	E_fus	E_suc1	E_suc2	E_suc3	F_chr1	F_chr2	F_con1	G_hol1	G_hol2	H_for1	H_for2	J_flo1	J_flo2	L_goo1	L_goo2
A_nat1		0.1	0.2	0.2	0.2	0.3	0.1	0.2	0.2	0.2	0.3	0.2	0.1	0.3	0.4	0.2	0.1	0.2	0.3	0.3	0.4	0.4	0.3	0.4	0.4	0.5
A_nat2	**0.0**		0.0	0.1	0.2	0.2	0.1	0.1	0.1	0.1	0.3	0.2	0.1	0.2	0.3	0.1	0.1	0.2	0.2	0.2	0.4	0.3	0.2	0.3	0.3	0.4
A_nat3	**0.0**	**0.6**		0.1	0.3	0.2	0.2	0.2	0.2	0.1	0.4	0.3	0.2	0.4	0.5	0.2	0.3	0.3	0.2	0.2	0.5	0.4	0.3	0.4	0.4	0.5
A_nat4	**0.0**	**0.3**	**0.3**		0.3	0.2	0.2	0.3	0.2	0.2	0.4	0.4	0.3	0.4	0.6	0.3	0.4	0.4	0.3	0.2	0.6	0.5	0.4	0.5	0.4	0.6
B_bel1	0.0	0.0	0.0	0.0		0.1	0.2	1.0	0.4	0.2	0.5	0.7	0.8	0.6	1.0	0.2	0.6	0.8	0.2	0.1	1.0	0.9	1.0	0.6	0.3	0.8
B_bel2	0.0	0.0	0.0	0.0	**0.5**		0.3	0.2	0.2	0.1	0.4	0.3	0.2	0.3	0.4	0.3	0.3	0.3	0.3	0.3	0.4	0.4	0.3	0.4	0.4	0.5
C_bat	0.0	0.0	0.0	0.0	0.0	0.0		0.2	0.2	0.2	0.3	0.2	0.1	0.3	0.3	0.2	0.1	0.2	0.3	0.2	0.4	0.4	0.3	0.4	0.4	0.5
C_bim	0.0	0.0	0.0	0.0	0.4	0.0	0.0		0.2	0.2	0.5	0.7	0.5	0.6	1.0	0.2	0.5	0.8	0.2	0.1	0.9	0.8	1.0	0.6	0.3	0.8
C_uro1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.3		0.1	0.5	0.5	0.3	0.5	0.6	0.2	0.4	0.5	0.2	0.2	0.7	0.6	0.5	0.5	0.4	0.6
C_uro2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	**0.1**		0.3	0.3	0.1	0.3	0.4	0.2	0.2	0.2	0.2	0.2	0.4	0.3	0.3	0.4	0.3	0.4
E_eve	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1	0.1	0.0	0.3	0.3	0.1	0.2	0.2	0.5	0.1	0.1	0.1	0.0	0.1
E_fus	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.9		0.1	0.1	0.0	0.2	0.5	0.1	0.1	0.2	0.6	0.2	0.1	0.1	0.0	0.2
E_suc1	0.0	0.1	0.0	0.0	0.4	0.0	0.0	0.4	0.0	0.3	0.5	0.7		0.2	0.3	0.1	0.4	0.2	0.1	0.1	0.3	0.0	0.3	0.1	0.0	0.3
E_suc2	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.2	0.4	**0.3**		0.2	0.2	0.5	0.3	0.2	0.2	0.6	0.3	0.3	0.3	0.1	0.2
E_suc3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.9	**0.2**	**0.0**		0.3	0.6	0.4	0.1	0.3	1.0	0.5	0.9	0.3	0.1	0.4
F_chr1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.1	0.2	0.1	0.2	0.4	0.3	0.3	0.4	0.3	0.4
F_chr2	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	**0.4**		0.4	0.2	0.2	0.8	0.6	0.6	0.5	0.3	0.6
F_con1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.2	0.7	0.4	0.0	0.0	0.0	0.0		0.1	0.1	0.8	0.3	0.1	0.1	0.0	0.4
G_hol1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3	0.0	0.0	0.0	0.0	0.2		0.1	0.3	0.1	0.1	0.2	0.2	0.3
G_hol2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.3	0.2	0.1	0.3	0.3	0.4
H_for1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.6	1.0	0.6	0.4	0.8
H_for2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.7	0.0	0.0	0.0	0.0	0.1	0.0	0.0	**0.0**		0.2	0.1	0.0	0.4
J_flo1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.5	0.4	0.3	0.0	0.0	0.0	0.0	0.6	0.1	0.0	0.0	0.3		0.0	0.0	0.3
J_flo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.1	**1.0**		0.1	0.4
L_goo1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.6	0.4	0.0	0.1	0.0	0.0	0.4	0.0	0.0	0.0	0.3	0.9	0.2		0.2
L_goo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**	
L_gul1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_gul2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_gul3	0.0	0.1	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_gul4	0.0	0.4	0.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_mac	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

L_mic1	0.0	0.3	0.2	0.2	0.4	0.0	0.1	0.4	0.6	0.4	0.0	0.1	0.4	0.0	0.0	0.1	0.1	0.1	0.0	0.2	0.0	0.0	0.1	0.0	0.0	0.0
L_mic2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_pla1	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_pla2	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_pun1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
L_pun2	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_pun3	0.0	0.3	0.4	0.0	0.1	0.0	0.0	0.1	0.2	0.3	0.0	0.0	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_sic	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
M_sal	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
N_cry	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.1	0.0	0.1	0.2	0.0	0.0	0.3	0.1	0.2	0.6	0.0	0.0	0.0	0.1	0.0	0.0	0.0
N_gyr	0.7	0.1	0.0	0.0	0.1	0.0	0.2	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
N_mac	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.5	0.6	0.2	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.2	0.4	0.0	0.0	0.2
N_pet	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
T_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

A. cont.	L_gul1	L_gul2	L_gul3	L_gul4	L_mac	L_mar	L_mic1	L_mic2	L_pla1	L_pla2	L_pun1	L_pun2	L_pun3	L_sic	M_sal	N_cry	N_gyr	N_mac	N_pet	T_mar
A_nat1	0.3	0.2	0.2	0.2	0.1	0.2	0.1	0.4	0.2	0.2	0.2	0.1	0.1	0.5	0.3	0.2	0.0	0.3	0.1	0.4
A_nat2	0.1	0.1	0.0	0.0	0.1	0.2	0.1	0.3	0.1	0.1	0.1	0.1	0.1	0.3	0.1	0.1	0.1	0.2	0.1	0.3
A_nat3	0.2	0.1	0.1	0.1	0.1	0.2	0.1	0.3	0.1	0.1	0.2	0.2	0.1	0.4	0.2	0.3	0.3	0.2	0.1	0.3
A_nat4	0.2	0.2	0.0	0.1	0.1	0.3	0.2	0.4	0.0	0.1	0.2	0.2	0.2	0.5	0.1	0.3	0.4	0.3	0.1	0.3
B_bel1	0.2	0.2	0.1	0.2	0.1	0.2	0.6	0.4	0.1	0.1	0.2	0.2	0.3	0.4	0.1	0.7	0.8	0.3	0.1	0.2
B_bel2	0.3	0.3	0.2	0.2	0.2	0.3	0.1	0.4	0.1	0.1	0.2	0.2	0.2	0.5	0.1	0.2	0.2	0.3	0.1	0.4
C_bat	0.2	0.2	0.1	0.2	0.1	0.2	0.1	0.4	0.2	0.2	0.2	0.1	0.1	0.4	0.3	0.2	0.1	0.3	0.1	0.4
C_bim	0.2	0.2	0.1	0.2	0.1	0.2	0.4	0.1	0.1	0.1	0.2	0.2	0.2	0.4	0.2	0.7	0.8	0.2	0.1	0.2
C_uro1	0.1	0.1	0.1	0.2	0.1	0.2	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.5	0.1	0.4	0.4	0.3	0.1	0.3
C_uro2	0.2	0.2	0.1	0.2	0.1	0.2	0.1	0.2	0.1	0.1	0.2	0.1	0.1	0.3	0.2	0.1	0.3	0.2	0.0	0.3
E_eve	0.4	0.4	0.3	0.3	0.2	0.1	0.3	0.6	0.2	0.3	0.2	0.3	0.3	0.5	0.4	0.3	0.4	0.1	0.2	0.5
E_fus	0.3	0.2	0.2	0.3	0.1	0.1	0.5	0.5	0.1	0.2	0.1	0.3	0.3	0.5	0.3	0.4	0.6	0.1	0.1	0.4
E_suc1	0.2	0.1	0.1	0.2	0.0	0.1	0.4	0.3	0.1	0.1	0.1	0.1	0.2	0.3	0.2	0.4	0.6	0.1	0.0	0.2
E_suc2	0.3	0.3	0.2	0.3	0.1	0.1	0.4	0.6	0.2	0.2	0.1	0.3	0.3	0.5	0.3	0.4	0.6	0.1	0.1	0.4
E_suc3	0.4	0.4	0.3	0.4	0.1	0.1	0.7	0.7	0.2	0.3	0.2	0.4	0.4	0.5	0.4	0.5	0.9	0.1	0.2	0.5
F_chr1	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.4	0.2	0.2	0.1	0.1	0.1	0.1	0.3	0.1	0.2	0.2	0.1	0.3
F_chr2	0.3	0.2	0.2	0.2	0.0	0.1	0.4	0.4	0.1	0.2	0.2	0.1	0.2	0.2	0.3	0.3	0.4	0.2	0.1	0.3
F_con1	0.3	0.2	0.2	0.2	0.1	0.1	0.5	0.5	0.1	0.2	0.2	0.3	0.3	0.4	0.3	0.3	0.7	0.1	0.1	0.3
G_hol1	0.3	0.3	0.3	0.3	0.2	0.2	0.1	0.4	0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.0	0.2	0.1	0.1	0.4
G_hol2	0.3	0.3	0.2	0.2	0.2	0.2	0.1	0.4	0.2	0.2	0.2	0.2	0.1	0.4	0.3	0.1	0.2	0.2	0.1	0.3
H_for1	0.5	0.4	0.3	0.4	0.2	0.3	0.7	0.7	0.2	0.3	0.3	0.5	0.5	0.7	0.4	0.7	0.9	0.3	0.2	0.5
H_for2	0.4	0.3	0.2	0.3	0.2	0.1	0.6	0.6	0.2	0.2	0.2	0.4	0.4	0.6	0.4	0.5	0.8	0.1	0.1	0.4
J_flo1	0.3	0.2	0.2	0.2	0.1	0.1	0.6	0.5	0.1	0.2	0.2	0.3	0.4	0.5	0.3	0.5	0.9	0.1	0.1	0.3
J_flo2	0.4	0.4	0.3	0.4	0.3	0.3	0.4	0.6	0.3	0.3	0.3	0.4	0.4	0.7	0.4	0.3	0.6	0.2	0.2	0.5
L_goo1	0.4	0.4	0.4	0.4	0.3	0.2	0.2	0.6	0.3	0.3	0.2	0.3	0.3	0.6	0.4	0.2	0.3	0.1	0.2	0.5

L_goo2	0.5	0.5	0.4	0.5	0.2	0.1	0.6	0.7	0.3	0.4	0.2	0.4	0.5	0.7	0.5	0.6	0.7	0.1	0.3	0.6
L_gul1		0.0	0.1	0.1	0.1	0.3	0.1	0.4	0.1	0.2	0.2	0.1	0.2	0.5	0.1	0.2	0.2	0.3	0.2	0.4
L_gul2	**0.7**		0.1	0.1	0.2	0.3	0.1	0.4	0.1	0.2	0.2	0.1	0.1	0.5	0.1	0.2	0.2	0.3	0.2	0.4
L_gul3	**0.0**	**0.0**		0.0	0.1	0.2	0.1	0.3	0.1	0.1	0.1	0.1	0.1	0.4	0.1	0.1	0.1	0.2	0.1	0.3
L_gul4	**0.0**	**0.1**	**0.6**		0.1	0.2	0.1	0.3	0.1	0.1	0.2	0.1	0.2	0.4	0.1	0.2	0.2	0.2	0.1	0.3
L_mac	0.0	0.0	0.0	0.0		0.1	0.0	0.3	0.1	0.1	0.1	0.0	0.1	0.2	0.2	0.1	0.1	0.1	0.1	0.3
L_mar	0.0	0.0	0.0	0.0	0.0		0.1	0.4	0.2	0.2	0.0	0.1	0.1	0.3	0.3	0.1	0.2	0.1	0.1	0.4
L_mic1	0.1	0.2	0.2	0.3	0.1	0.0		0.1	0.0	0.1	0.1	0.1	0.1	0.3	0.1	0.4	0.5	0.2	0.0	0.2
L_mic2	0.0	0.0	0.0	0.0	0.0	0.0	**0.1**		0.3	0.3	0.4	0.3	0.2	0.6	0.4	0.5	0.4	0.4	0.2	0.5
L_pla1	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0		0.0	0.1	0.1	0.1	0.3	0.1	0.1	0.1	0.2	0.1	0.3
L_pla2	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	**0.2**		0.1	0.1	0.1	0.4	0.1	0.1	0.1	0.2	0.1	0.2
L_pun1	0.0	0.0	0.0	0.0	0.0	0.4	0.1	0.0	0.0	0.0		0.1	0.1	0.3	0.2	0.2	0.2	0.1	0.1	0.3
L_pun2	0.0	0.0	0.0	0.0	0.1	0.0	0.5	0.0	0.0	0.0	**0.0**		0.1	0.4	0.2	0.2	0.2	0.2	0.1	0.3
L_pun3	0.0	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0	**0.0**	**0.2**		0.3	0.2	0.2	0.3	0.2	0.1	0.3
L_sic	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.5	0.1	0.5	0.4	0.3	0.6
M_sal	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.2	0.2	0.3	0.2	0.4
N_cry	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0		0.6	0.2	0.1	0.3
N_gyr	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.3	0.1	0.3
N_mac	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.4
N_pet	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0		0.2
T_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

B.	A_cal	A_nat1	A_nat2	A_nat3	A_nat4	B_bel1	B_bel2	C_bat	C_bim	E_eve	F_chr1	G_hol1	G_hol2	H_for1	H_for2	L_gul1	L_gul2	L_gul3	L_gul4	L_mac	L_mar	L_mic1	L_pla1	L_pla2
A_cal		0.3	0.4	0.3	0.2	0.4	0.5	0.2	0.1	0.3	0.2	0.3	0.3	0.4	0.3	0.1	0.1	0.2	0.1	0.2	0.3	0.1	0.2	0.2
A_nat1	0.0		0.1	0.0	0.1	0.3	0.4	0.1	0.1	0.2	0.2	0.4	0.3	0.3	0.2	0.1	0.1	0.2	0.2	0.1	0.2	0.1	0.3	0.1
A_nat2	0.0	**0.0**		0.0	0.1	0.5	0.6	0.2	0.1	0.4	0.3	0.5	0.4	0.4	0.4	0.2	0.3	0.3	0.3	0.2	0.4	0.2	0.5	0.2
A_nat3	0.0	**0.2**	**0.1**		0.1	0.5	0.5	0.1	0.1	0.3	0.2	0.3	0.3	0.4	0.3	0.2	0.2	0.2	0.2	0.2	0.3	0.2	0.2	0.2
A_nat4	0.0	**0.0**	**0.0**	**0.2**		0.5	0.4	0.1	0.2	0.3	0.2	0.2	0.2	0.5	0.2	0.2	0.2	0.2	0.1	0.2	0.4	0.3	0.1	0.1
B_bel1	0.0	0.0	0.0	0.0	0.0		0.0	0.4	1.0	0.7	0.4	0.4	0.4	1.0	0.5	0.5	0.5	0.4	0.3	0.7	0.7	1.0	0.2	0.6
B_bel2	0.0	0.0	0.0	0.0	0.0	**1.0**		0.4	0.3	0.6	0.4	0.5	0.5	0.7	0.5	0.4	0.4	0.4	0.4	0.4	0.6	0.5	0.4	0.4
C_bat	0.0	0.0	0.0	0.0	0.3	0.0	0.0		0.1	0.3	0.1	0.2	0.2	0.3	0.2	0.1	0.1	0.1	0.1	0.1	0.2	0.2	0.2	0.1
C_bim	0.1	0.0	0.1	0.1	0.4	0.2	0.1	0.6		0.4	0.2	0.1	0.1	0.9	0.2	0.2	0.2	0.1	0.1	0.3	0.1	1.0	0.0	0.3
E_eve	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2		0.2	0.3	0.2	0.6	0.3	0.3	0.3	0.3	0.2	0.1	0.5	0.6	0.2	0.4
F_chr1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0		0.2	0.1	0.3	0.2	0.2	0.2	0.2	0.1	0.1	0.3	0.3	0.2	0.2
G_hol1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.0	0.1	0.1	0.2	0.2	0.3	0.2	0.2	0.3	0.2	0.3	0.2
G_hol2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	**0.1**		0.1	0.1	0.2	0.1	0.2	0.2	0.1	0.3	0.2	0.3	0.1
H_for1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.1		0.2	0.5	0.4	0.4	0.3	0.6	0.7	0.9	0.2	0.5
H_for2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.2	0.2	**0.0**		0.2	0.2	0.2	0.2	0.2	0.3	0.3	0.2	0.2
L_gul1	0.2	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.1	0.0	0.2	0.3	0.1	0.0	0.2
L_gul2	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	**1.0**		0.0	0.0	0.2	0.3	0.1	0.0	0.2
L_gul3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	**0.3**	**0.6**		0.1	0.2	0.2	0.1	0.1	0.2

L_gul4	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0	**0.4**	**0.7**	**0.2**		0.1	0.2	0.1	0.1	0.1
L_mac	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.5	0.4	0.8	0.0	0.0	0.0	0.1	0.1	0.2	0.1	0.4		0.3	0.6	0.1	0.2
L_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1		0.5	0.2	0.4
L_mic1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.4	0.1	0.0	0.0	0.0	0.1	0.0	0.8	0.4	0.5	0.2	0.1	0.1		0.0	0.4
L_pla1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.9		0.1
L_pla2	0.0	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.6	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.1	0.0	0.1	0.3	0.0	0.1	**0.0**	
L_pun1	0.1	0.0	0.1	0.1	0.4	0.2	0.1	0.4		0.2	0.3	0.0	0.1	0.2	0.1	0.6	0.5	0.2	0.1	0.5	0.3	0.4	0.1	0.6
L_pun2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.1	0.1	0.1	0.1	0.1	0.2	0.0	0.0
L_pun3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
N_gyr	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.1	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.2
P_lat1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	1.0	0.0	0.0	0.1	0.0	0.1	0.5	0.1	0.1	0.0	0.0	0.1	0.0	0.4	0.0	0.1
P_lat2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

B. cont.	L_pun1	L_pun2	L_pun3	N_gyr	P_lat1	P_lat2
A_cal	0.1	0.1	0.2	0.2	0.2	0.3
A_nat1	0.1	0.2	0.2	0.2	0.1	0.2
A_nat2	0.1	0.3	0.3	0.3	0.2	0.4
A_nat3	0.1	0.2	0.2	0.2	0.2	0.3
A_nat4	0.2	0.1	0.1	0.2	0.4	0.4
B_bel1	1.0	0.2	0.4	0.5	1.0	0.8
B_bel2	0.3	0.3	0.4	0.4	0.5	0.6
C_bat	0.1	0.1	0.1	0.1	0.2	0.3
C_bim	1.0	0.0	0.1	0.2	1.0	0.6
E_eve	0.3	0.1	0.3	0.2	0.6	0.6
F_chr1	0.2	0.1	0.1	0.1	0.3	0.3
G_hol1	0.1	0.2	0.3	0.2	0.1	0.1
G_hol2	0.1	0.2	0.2	0.2	0.1	0.1
H_for1	0.9	0.2	0.3	0.4	0.9	0.7
H_for2	0.2	0.1	0.2	0.2	0.1	0.1
L_gul1	0.2	0.0	0.1	0.2	0.3	0.4
L_gul2	0.2	0.0	0.1	0.2	0.3	0.4
L_gul3	0.1	0.0	0.1	0.2	0.2	0.3
L_gul4	0.1	0.0	0.1	0.1	0.2	0.2
L_mac	0.3	0.0	0.1	0.2	0.6	0.5
L_mar	0.2	0.1	0.1	0.2	0.5	0.5
L_mic1	1.0	0.0	0.2	0.3	1.0	0.7
L_pla1	0.0	0.1	0.2	0.1	0.1	0.2
L_pla2	0.3	0.1	0.1	0.2	0.4	0.4
L_pun1		0.0	0.1	0.2	1.0	0.6
L_pun2	**0.2**		0.0	0.0	0.1	0.1
L_pun3	**0.2**	**0.1**		0.1	0.2	0.3

N_gyr	0.9	0.1	0.1		0.3	0.3
P_lat1	1.0	0.0	0.0	0.0		0.1
P_lat2	0.2	0.0	0.0	0.0	**1.0**	

C.	A_cal	A_nat1	A_nat2	A_nat3	C_bim	C_var	E_eve	E_fus	E_glo	F_chr1	F_chr2	F_con1	F_con2	G_hol1	G_hol2	H_for1	H_for2	J_flo1	J_flo2	L_goo1	L_goo2	L_gul3	L_gul4
A_cal		0.1	0.5	1.0	1.0	0.1	0.3	0.9	0.3	0.1	0.1	0.3	0.7	0.3	0.1	0.3	0.3	0.2	0.2	0.2	0.5	0.8	1.0
A_nat1	0.1		0.2	0.1	0.1	0.2	0.1	0.1	0.2	0.1	0.2	0.1	0.0	0.3	0.2	0.3	0.3	0.3	0.2	0.2	0.3	0.2	0.1
A_nat2	0.2	**0.0**		0.5	0.5	0.2	0.4	0.6	0.5	0.1	0.2	0.4	0.5	0.5	0.3	0.5	0.5	0.4	0.4	0.4	0.7	0.5	0.4
A_nat3	-	**0.1**	**0.2**		1.0	0.1	0.3	0.9	0.3	0.1	0.1	0.3	0.8	0.3	0.2	0.3	0.3	0.2	0.2	0.2	0.5	0.8	1.0
C_bim	-	0.2	0.2	-		0.1	0.2	0.9	0.3	0.1	0.0	0.3	0.7	0.3	0.2	0.3	0.3	0.2	0.2	0.2	0.5	0.8	1.0
C_var	0.1	0.0	0.0	0.1	0.1		0.2	0.1	0.2	0.1	0.3	0.3	0.1	0.2	0.2	0.1	0.1	0.1	0.1	0.2	0.3	0.2	0.1
E_eve	0.1	0.0	0.0	0.1	0.1	0.0		0.1	0.1	0.1	0.4	0.3	0.2	0.3	0.3	0.3	0.2	0.3	0.3	0.2	0.3	0.5	0.3
E_fus	0.4	0.1	0.1	0.4	0.4	0.2	0.3		0.1	0.0	0.2	0.2	0.7	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.4	0.8	0.9
E_glo	0.1	0.0	0.0	0.0	0.1	0.0	0.1	0.1		0.1	0.5	0.4	0.3	0.4	0.3	0.3	0.1	0.2	0.2	0.2	0.4	0.5	0.3
F_chr1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0		0.2	0.2	0.0	0.1	0.1	0.2	0.1	0.2	0.1	0.1	0.1	0.1	0.1
F_chr2	0.1	0.0	0.0	0.1	0.5	0.0	0.0	0.0	0.0	**0.0**		0.3	0.0	0.5	0.3	0.5	0.5	0.5	0.5	0.5	0.5	0.3	0.1
F_con1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.5	0.3	0.6	0.5	0.5	0.4	0.4	0.4	0.5	0.3
F_con2	0.7	0.3	0.2	0.4	0.4	0.0	0.0	0.4	0.0	0.1	0.4	**0.3**		0.3	0.1	0.4	0.3	0.2	0.2	0.2	0.4	0.8	0.8
G_hol1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.2	0.2	0.2	0.2	0.2	0.4	0.5	0.3
G_hol2	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.3	0.3	0.3	0.2	0.3	0.3	0.3	0.2
H_for1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.1	0.1	0.2	0.6	0.5	0.3
H_for2	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.1**		0.1	0.0	0.0	0.4	0.5	0.3
J_flo1	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1		0.0	0.1	0.4	0.4	0.2
J_flo2	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	**0.2**		0.1	0.3	0.4	0.2
L_goo1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0		0.3	0.4	0.2
L_goo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.7	0.5
L_gul3	0.3	0.0	0.1	0.3	0.3	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1
L_gul4	-	0.1	0.2	-	-	0.1	0.1	0.4	0.0	0.0	0.1	0.0	0.4	0.0	0.0	0.1	0.0	0.1	0.1	0.0	0.1	**0.8**	
L_mac	-	0.3	0.2	-	-	0.2	0.3	0.4	0.1	0.2	0.1	0.0	0.4	0.1	0.1	0.1	0.1	0.1	0.1	0.0	0.0	0.3	
L_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_mic1	0.1	0.0	0.1	0.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3
L_mic2	-	0.2	0.2	-	-	0.1	0.1	0.4	0.1	0.1	0.4	0.1	0.4	0.1	0.1	0.1	0.0	0.1	0.1	0.0	0.0	0.3	-
L_pun1	0.0	0.0	0.0	0.0	0.1	0.0	0.3	0.2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_pun2	0.1	0.0	0.0	0.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2
L_pun3	1.0	0.1	0.2	1.0	1.0	0.1	0.0	0.4	0.0	0.0	0.1	0.0	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	1.0
N_gyr	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.3	0.4	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
P_lat1	0.1	0.0	0.0	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
P_lat2	0.3	0.0	0.0	0.3	0.3	0.2	0.0	0.2	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.3

C.	L_mac	L_mar	L_mic1	L_mic2	L_pun1	L_pun2	L_pun3	N_gyr	P_lat1	P_lat2
A_cal	1.0	0.5	0.2	1.0	0.2	0.1	0.5	0.2	0.5	0.6
A_nat1	0.1	0.3	0.2	0.1	0.1	0.1	0.1	0.1	0.4	0.2
A_nat2	0.5	0.6	0.2	0.5	0.3	0.1	0.4	0.3	0.6	0.5
A_nat3	1.0	0.5	0.2	1.0	0.2	0.1	0.5	0.2	0.5	0.6
C_bim	1.0	0.5	0.1	1.0	0.2	0.1	0.5	0.2	0.5	0.6
C_var	0.1	0.4	0.2	0.1	0.2	0.2	0.1	0.2	0.2	0.1
E_eve	0.1	0.4	0.4	0.2	0.0	0.2	0.2	0.2	0.6	0.3
E_fus	0.7	0.5	0.3	0.9	0.0	0.1	0.5	0.1	0.6	0.5
E_glo	0.1	0.6	0.5	0.3	0.0	0.4	0.3	0.3	0.7	0.3
F_chr1	0.0	0.2	0.2	0.1	0.1	0.1	0.1	0.1	0.3	0.1
F_chr2	0.1	0.5	0.2	0.0	0.4	0.1	0.1	0.3	0.5	0.2
F_con1	0.2	0.2	0.4	0.3	0.3	0.1	0.2	0.0	0.7	0.4
F_con2	0.7	0.2	0.2	0.7	0.2	0.0	0.4	0.0	0.6	0.6
G_hol1	0.2	0.5	0.4	0.3	0.3	0.3	0.3	0.3	0.6	0.3
G_hol2	0.1	0.3	0.3	0.2	0.3	0.2	0.2	0.2	0.5	0.2
H_for1	0.2	0.7	0.5	0.3	0.2	0.4	0.3	0.4	0.6	0.3
H_for2	0.2	0.6	0.5	0.3	0.1	0.4	0.3	0.3	0.6	0.3
J_flo1	0.2	0.6	0.4	0.2	0.2	0.3	0.2	0.3	0.5	0.2
J_flo2	0.2	0.5	0.4	0.2	0.2	0.3	0.2	0.3	0.5	0.2
L_goo1	0.1	0.5	0.4	0.2	0.1	0.3	0.2	0.3	0.6	0.3
L_goo2	0.4	0.2	0.5	0.5	0.3	0.3	0.5	0.3	0.8	0.6
L_gul3	0.8	0.7	0.2	0.8	0.4	0.2	0.5	0.4	0.6	0.6
L_gul4	1.0	0.5	0.1	1.0	0.2	0.1	0.5	0.2	0.5	0.6
L_mac		0.3	0.2	1.0	0.0	0.1	0.4	0.1	0.5	0.6
L_mar	0.0		0.5	0.5	0.4	0.2	0.4	0.2	0.8	0.6
L_mic1	0.1	0.0		0.1	0.4	0.1	0.2	0.3	0.5	0.3
L_mic2	-	0.0	**0.3**		0.2	0.1	0.5	0.2	0.5	0.6
L_pun1	0.2	0.0	0.0	0.0		0.3	0.2	0.2	0.6	0.2
L_pun2	0.3	0.0	0.0	0.2	**0.0**		0.1	0.1	0.4	0.2
L_pun3	1.0	0.0	0.1	1.0	**0.0**	**0.5**		0.1	0.4	0.4
N_gyr	0.1	0.0	0.0	0.1	0.0	0.0	0.1		0.5	0.3
P_lat1	0.1	0.0	0.0	0.1	0.0	0.0	0.0	0.0		0.2
P_lat2	0.3	0.0	0.0	0.3	0.0	0.0	0.1	0.0	**0.2**	

D.	A_nat1	A_xen	B_bel1	B_bel2	C_var	E_eve	E_glo	F_chr1	F_chr2	F_con1	F_con2	G_hol1	G_hol2	H_for1	H_for2	J_flo1	J_flo2	L_goo1	L_goo2	L_gul1	L_gul2	L_gul4
A_nat1		0.1	0.2	0.3	0.2	0.4	0.4	0.3	0.1	0.3	0.1	0.3	0.3	0.3	0.4	0.2	0.3	0.5	0.5	0.0	0.0	0.0
A_xen	0.0		0.7	0.8	0.1	0.2	0.3	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.3	0.2	0.6	0.7	0.4

B_bel1	0.0	0.1		-	0.6	0.5	0.6	0.2	0.2	0.3	0.2	0.3	0.3	0.3	0.4	0.5	0.2	0.6	0.6	1.0	1.0	0.6
B_bel2	0.0	0.1	-		0.7	0.6	0.7	0.3	0.3	0.5	0.3	0.4	0.4	0.4	0.5	0.6	0.4	0.8	0.7	1.0	1.0	0.7
C_var	0.0	0.6	0.0	0.0		0.3	0.4	0.1	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.2	0.0	0.4	0.4	0.4	0.6	0.3
E_eve	0.0	0.0	0.0	0.0	0.0		0.0	0.1	0.2	0.1	0.2	0.3	0.2	0.4	0.2	0.4	0.4	0.1	0.2	0.3	0.5	0.3
E_glo	0.0	0.0	0.0	0.0	0.0	0.3		0.2	0.3	0.2	0.3	0.3	0.2	0.4	0.3	0.5	0.4	0.2	0.2	0.4	0.6	0.4
F_chr1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.1	0.1	0.2	0.1	0.3	0.2	0.1	0.3	0.2	0.2	0.1	0.2	0.1
F_chr2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	**0.0**		0.1	0.0	0.2	0.2	0.3	0.2	0.1	0.2	0.3	0.3	0.1	0.2	0.1
F_con1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.2	0.1	0.3	0.2	0.3	0.3	0.2	0.1	0.2	0.3	0.2
F_con2	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.2	**0.0**		0.2	0.1	0.3	0.3	0.1	0.3	0.3	0.3	0.0	0.1	0.1
G_hol1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.1	0.1	0.1	0.2	0.2	0.2	0.1	0.3	0.1
G_hol2	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.2	0.2	0.2	0.2	0.2	0.1	0.1	0.3	0.1
H_for1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0		0.1	0.1	0.1	0.4	0.4	0.1	0.3	0.2
H_for2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.2	0.2	0.2	0.3	0.2	0.4	0.2
J_flo1	0.0	0.4	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0		0.1	0.5	0.4	0.3	0.5	0.3
J_flo2	0.0	0.3	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.1**		0.5	0.5	0.1	0.2	0.1
L_goo1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.4	0.6	0.4
L_goo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.3	0.6	0.3
L_gul1	0.9	0.4	1.0	1.0	0.2	0.1	0.1	0.1	0.4	0.1	0.5	0.1	0.0	0.1	0.0	0.2	0.0	0.0	0.0		1.0	0.3
L_gul2	0.3	0.1	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**1.0**		0.4
L_gul4	0.4	0.4	0.1	0.0	0.1	0.0	0.0	0.0	0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	**1.0**	**0.1**	
L_mac	0.2	0.4	0.1	0.1	0.1	0.0	0.0	0.1	0.9	0.3	0.8	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.4	0.1	0.4
L_mar	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
L_mic1	0.2	0.4	1.0	1.0	0.2	0.1	0.1	0.1	0.4	0.2	0.1	0.0	0.1	0.0	0.1	0.2	0.0	0.0	0.0	-	0.3	1.0
L_pun1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
L_pun2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.1
N_gyr	0.0	0.8	0.0	0.0	0.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.3	0.0	0.0	0.4	0.0	0.2
P_lat2	0.0	0.5	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0

D. cont.	L_mac	L_mar	L_mic1	L_pun1	L_pun2	N_gyr	P_lat2
A_nat1	0.1	0.5	0.1	0.3	0.2	0.2	0.3
A_xen	0.5	0.2	0.6	0.2	0.1	0.1	0.1
B_bel1	0.9	0.5	1.0	0.3	0.2	0.3	0.6
B_bel2	0.9	0.7	1.0	0.5	0.3	0.4	0.7
C_var	0.5	0.4	0.4	0.2	0.2	0.1	0.1
E_eve	0.2	0.2	0.2	0.1	0.4	0.3	0.6
E_glo	0.3	0.2	0.4	0.1	0.4	0.3	0.7
F_chr1	0.0	0.3	0.1	0.1	0.2	0.1	0.3
F_chr2	0.0	0.3	0.1	0.2	0.1	0.1	0.3
F_con1	0.1	0.1	0.1	0.1	0.2	0.1	0.4
F_con2	0.0	0.2	0.1	0.2	0.1	0.1	0.3
G_hol1	0.2	0.3	0.1	0.3	0.4	0.1	0.4

G_hol2	0.1	0.2	0.1	0.3	0.3	0.1	0.3
H_for1	0.2	0.5	0.1	0.3	0.4	0.1	0.3
H_for2	0.2	0.4	0.2	0.2	0.4	0.2	0.4
J_flo1	0.4	0.4	0.3	0.3	0.2	0.1	0.3
J_flo2	0.2	0.5	0.1	0.3	0.3	0.0	0.1
L_goo1	0.3	0.2	0.4	0.2	0.5	0.3	0.7
L_goo2	0.2	0.1	0.3	0.3	0.5	0.3	0.7
L_gul1	0.7	0.2	1.0	0.1	0.0	0.2	0.4
L_gul2	0.8	0.5	1.0	0.3	0.1	0.3	0.6
L_gul4	0.4	0.3	0.5	0.2	0.1	0.2	0.4
L_mac		0.1	0.8	0.1	0.0	0.2	0.5
L_mar	0.0		0.2	0.2	0.4	0.3	0.6
L_mic1	0.4	0.0		0.1	0.1	0.2	0.4
L_pun1	0.0	0.0	0.1		0.2	0.2	0.5
L_pun2	0.5	0.0	0.2	**0.0**		0.1	0.4
N_gyr	0.2	0.0	0.4	0.0	0.0		0.1
P_lat2	0.0	0.0	0.1	0.0	0.0	0.0	

E.	A_nat1	A_nat2	E_eve	F_chr1	F_chr2	F_con1	F_con2	G_hol1	G_hol2	H_for1	H_for2	J_flo1	J_flo2	L_goo1	L_goo2	L_gul1	L_gul3	L_gul4	L_mar	L_pla1	L_pun2	L_pun3	N_gyr
A_nat1		0.1	0.1	0.2	0.1	0.1	0.1	0.3	0.3	0.3	0.4	0.3	0.2	0.4	0.4	0.2	0.2	0.2	0.2	0.3	0.2	0.2	0.2
A_nat2	**0.5**		0.6	0.1	0.1	0.4	0.5	0.2	0.3	0.4	0.4	0.6	0.2	0.5	0.6	0.3	0.6	0.5	0.5	0.2	0.4	0.2	0.6
E_eve	0.6	0.4		0.0	0.0	0.0	0.1	0.1	0.1	0.3	0.1	0.5	0.1	0.3	0.2	0.6	0.9	0.9	0.0	0.3	0.4	0.5	0.2
F_chr1	0.0	0.0	0.3		0.1	0.2	0.0	0.1	0.0	0.2	0.2	0.1	0.1	0.2	0.1	0.1	0.1	0.1	0.1	0.4	0.2	0.2	0.0
F_chr2	0.1	0.1	0.7	**0.0**		0.1	0.0	0.2	0.2	0.3	0.3	0.2	0.2	0.3	0.3	0.2	0.2	0.2	0.1	0.3	0.1	0.2	0.1
F_con1	0.1	0.0	0.7	0.0	0.2		0.0	0.3	0.3	0.5	0.5	0.5	0.3	0.6	0.5	0.5	0.5	0.5	0.1	0.5	0.4	0.5	0.2
F_con2	0.7	0.4	1.0	0.2	0.9	**0.9**		0.1	0.1	0.3	0.2	0.6	0.1	0.3	0.2	0.6	1.0	1.0	0.0	0.3	0.4	0.5	0.3
G_hol1	0.0	0.0	0.1	0.0	0.0	0.0	0.1		0.1	0.1	0.2	0.1	0.1	0.1	0.2	0.2	0.2	0.2	0.2	0.4	0.3	0.4	0.1
G_hol2	0.0	0.0	0.2	0.0	0.0	0.0	0.2	**0.0**		0.4	0.4	0.3	0.2	0.4	0.2	0.3	0.3	0.3	0.2	0.5	0.4	0.5	0.1
H_for1	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0		0.1	0.1	0.0	0.2	0.4	0.3	0.3	0.3	0.5	0.4	0.4	0.5	0.3
H_for2	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	**0.1**		0.1	0.0	0.2	0.3	0.3	0.3	0.3	0.4	0.5	0.4	0.5	0.2
J_flo1	0.0	0.1	0.1	0.0	0.0	0.0	0.1	0.1	0.0	0.5	0.1		0.0	0.2	0.3	0.6	0.8	0.8	0.4	0.4	0.5	0.6	0.4
J_flo2	0.0	0.0	0.4	0.0	0.0	0.0	0.1	0.0	0.0	1.0	0.4	**0.6**		0.1	0.2	0.1	0.1	0.1	0.2	0.3	0.2	0.2	0.1
L_goo1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.5	0.5	0.5	0.4	0.6	0.5	0.6	0.2
L_goo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.6	0.7	0.7	0.3	0.7	0.6	0.7	0.1
L_gul1	0.2	0.7	0.4	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0		0.3	0.3	0.5	0.0	0.5	0.2	0.6
L_gul3	0.3	0.4	0.4	0.0	0.1	0.1	0.4	0.1	0.1	0.1	0.0	0.2	0.1	0.1	0.0	**1.0**		1.0	0.6	0.2	0.7	0.4	0.8
L_gul4	0.3	0.7	0.4	0.0	0.1	0.1	0.4	0.1	0.1	0.1	0.0	0.2	0.1	0.1	0.0	**1.0**	-		0.6	0.1	0.7	0.3	0.8
L_mar	0.0	0.0	0.7	0.0	0.0	0.0	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1		0.6	0.4	0.6	0.0
L_pla1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.2	0.4	0.0		0.4	0.2	0.4
L_pun2	0.1	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2	0.2	0.0	0.0		0.3	0.4

L_pun3	0.1	0.7	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.6	0.2	0.4	0.0	0.0	**0.0**		0.6
N_gyr	0.2	0.1	0.5	0.1	0.3	0.1	0.4	0.0	0.0	0.0	0.0	0.2	0.2	0.0	0.0	0.1	0.2	0.2	0.7	0.0	0.0	0.0	

F.	C_var	E_eve	F_chr1	F_chr2	F_con1	F_con2	G_hol1	G_hol2	H_for1	H_for2	J_flo1	J_flo2	L_goo1	L_goo2	L_gul1	L_mar	L_pun1	L_pun2	L_pun3	N_gyr	P_lat1	P_lat2
C_var		0.4	0.1	0.1	0.3	0.2	0.1	0.1	0.2	0.1	0.1	0.0	0.3	0.2	1.0	1.0	0.5	0.5	1.0	1.0	0.6	1.0
E_eve	0.2		0.0	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.4	0.6	0.1	0.1	0.4	0.0	0.1	0.3	0.4	0.4	0.4	0.8
F_chr1	0.0	0.0		0.0	0.1	0.0	0.1	0.1	0.2	0.2	0.2	0.5	0.2	0.2	0.1	0.0	0.1	0.1	0.1	0.1	0.1	0.4
F_chr2	0.1	0.1	**0.0**		0.2	0.1	0.2	0.2	0.3	0.3	0.3	0.5	0.3	0.3	0.1	0.0	0.2	0.1	0.1	0.1	0.2	0.5
F_con1	0.1	0.0	0.0	0.0		0.1	0.3	0.2	0.3	0.3	0.5	0.7	0.2	0.2	0.3	0.0	0.2	0.3	0.2	0.2	0.3	0.7
F_con2	0.2	0.4	0.1	0.3	**0.1**		0.1	0.1	0.3	0.2	0.4	0.6	0.2	0.2	0.2	0.0	0.2	0.2	0.2	0.2	0.2	0.6
G_hol1	0.2	0.0	0.0	0.0	0.0	0.0		0.1	0.1	0.1	0.3	0.5	0.2	0.2	0.2	0.1	0.2	0.3	0.2	0.1	0.2	0.6
G_hol2	0.1	0.0	0.0	0.0	0.0	0.0	**0.0**		0.2	0.1	0.3	0.5	0.1	0.1	0.1	0.0	0.1	0.2	0.1	0.1	0.2	0.5
H_for1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.3	0.6	0.1	0.2	0.3	0.1	0.2	0.4	0.3	0.2	0.3	0.7
H_for2	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.2**		0.3	0.5	0.1	0.2	0.2	0.1	0.1	0.3	0.2	0.2	0.2	0.6
J_flo1	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.2	0.5	0.4	0.4	0.4	0.6	0.5	0.4	0.4	0.3	0.6
J_flo2	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.7	0.7	0.5	0.4	0.7	0.6	0.5	0.4	0.5	0.8
L_goo1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.1	0.4	0.0	0.1	0.5	0.4	0.3	0.4	0.8
L_goo2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	**0.0**		0.2	0.0	0.2	0.3	0.2	0.1	0.3	0.7
L_gul1	-	0.2	0.1	0.3	0.1	0.2	0.1	0.0	0.1	0.1	0.2	0.1	0.1	0.0		1.0	0.4	0.5	1.0	1.0	0.6	1.0
L_mar	-	0.8	0.7	0.6	0.4	0.6	0.5	0.5	0.4	0.2	0.2	0.1	0.6	0.6	-		0.1	0.2	1.0	1.0	0.6	1.0
L_pun1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.4		0.5	0.5	0.4	0.5	0.8
L_pun2	0.2	0.1	0.0	0.1	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.5	**0.0**		0.3	0.3	0.5	0.8
L_pun3	-	0.2	0.1	0.3	0.1	0.2	0.1	0.2	0.1	0.1	0.2	0.1	0.1	0.0	-	-	**0.1**	**0.4**		1.0	0.6	1.0
N_gyr	-	0.2	0.1	0.3	0.1	0.2	0.2	0.2	0.1	0.1	0.2	0.0	0.1	0.0	-	-	0.1	0.5	-		0.6	1.0
P_lat1	0.4	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0	0.0	0.4	0.4	0.0	0.1	0.4	0.4		0.5
P_lat2	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0	1.0	0.0	0.0	1.0	1.0	**0.2**	

Table A.2.12. Summary of all ontogenetic comparisons within a habitat-season level. As p (A < B) – the probability that the ellipse of group A is less than that of group B – nears one there is a high probability group A has a smaller trophic niche area than group B. The opposite is true as p (A < B) approaches zero. The percent difference in modal SEA_b between groups is expressed with the sign to indicate whether there was an increase or decrease in trophic niche area (SEA_b %Δ).

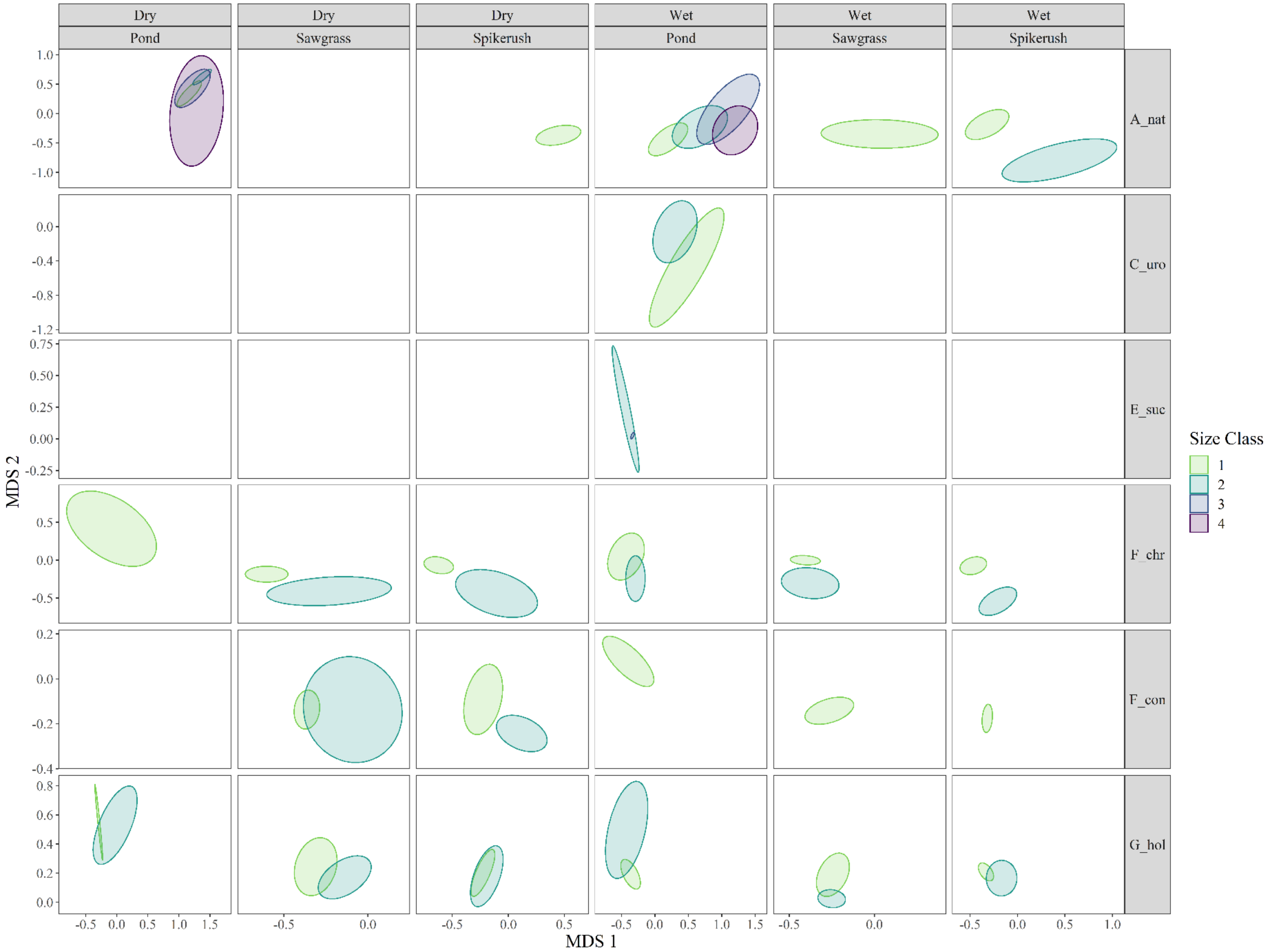
Habitat-Season Level	Species	Size Class		p (A < B)	SEA_b %Δ
		A	B		
Pond-Wet	*Ameiurus natalis*	1	2	0.000	-92
		1	3	1.000	175
		1	4	0.001	-69
		2	3	1.000	3424
		2	4	1.000	272
		3	4	0.000	-89
	Erimyzon sucetta	2	3	0.548	-2
	Fundulus chrysotus	1	2	0.447	-5
	Gambusia holbrooki	1	2	0.000	-94
	Heterandria formosa	1	2	0.000	-95
	Lucania goodei	1	2	0.466	-4
	Lepomis gulosus	1	2	0.000	-85
		1	3	0.003	-59
		1	4	0.000	-60
		2	3	0.974	182
		2	4	0.971	172
		3	4	0.428	-3
	Lepisosteus platyrhincus	1	2	1.000	10833
	Lepomis punctatus	1	2	0.000	-97
		1	3	0.000	-94
		2	3	0.892	130
	Mayaheros urophthalmus	1	2	0.928	152

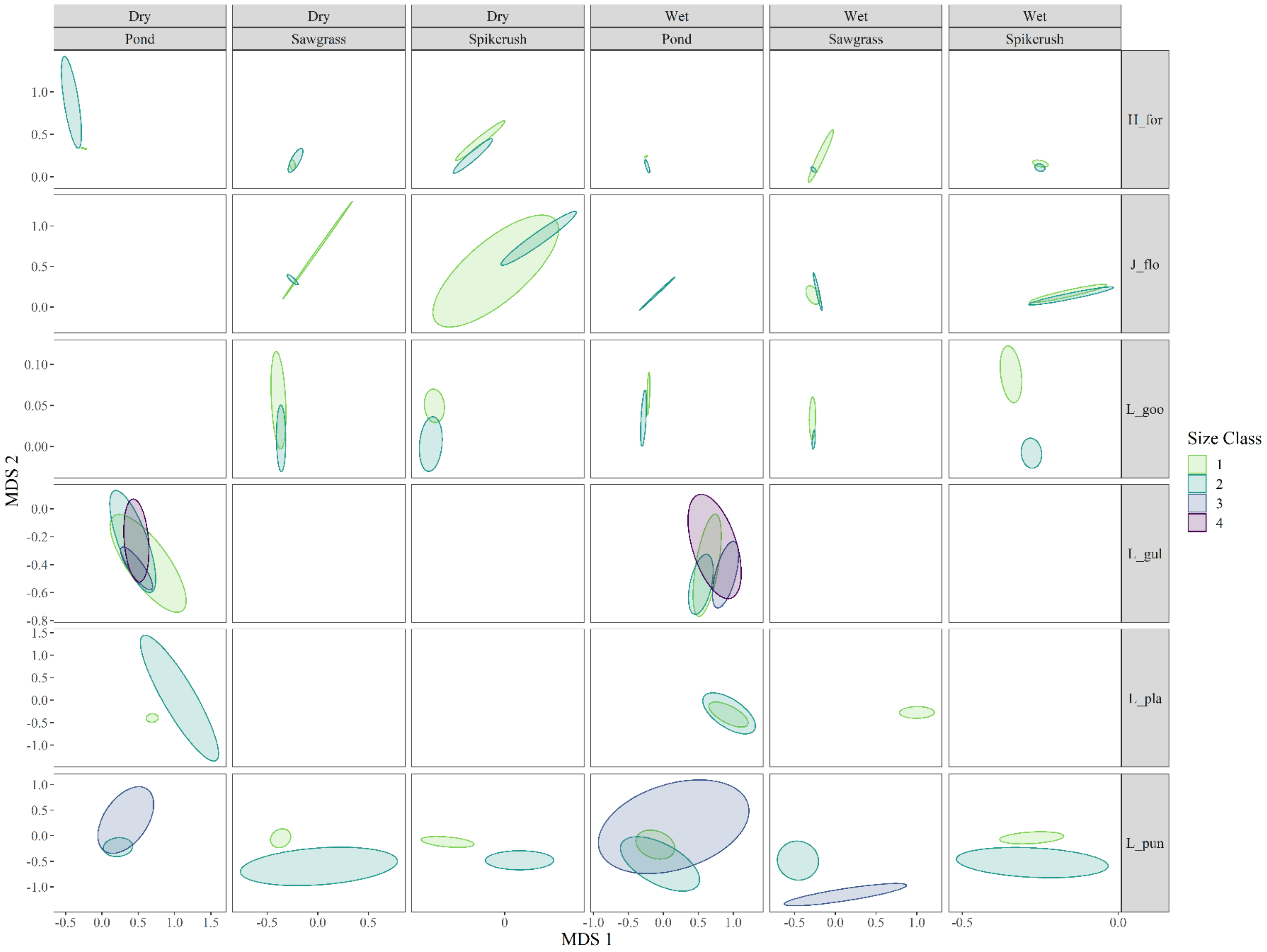
Pond-Dry	*Ameiurus natalis*	1	2	0.000	-70
		1	3	0.003	-88
		1	4	0.000	-100
		2	3	0.126	-62
		2	4	0.000	-99
		3	4	0.000	-98
	Gambusia holbrooki	1	2	0.997	719
	Heterandria formosa	1	2	0.000	-77
	Lepomis gulosus	1	2	0.000	-99
		1	3	0.916	69
		1	4	1.000	274
		2	3	1.000	19700
		2	4	1.000	43800
		3	4	0.992	122
	Lepisosteus platyrhincus	1	2	1.000	755
	Lepomis punctatus	2	3	0.005	-84
Spikerush - Wet	*Ameiurus natalis*	1	2	0.000	-94
	Fundulus chrysotus	1	2	0.000	-86
	Gambusia holbrooki	1	2	0.000	-83
	Heterandria formosa	1	2	0.000	-96
	Jordanella floridae	1	2	0.536	-3
	Lucania goodei	1	2	0.000	-96
	Lepomis punctatus	1	2	0.018	-55

Spikerush - Dry	*Fundulus chrysotus*	1	2	0.000	-91
	Fundulus confluentus		2	0.000	-93
	Gambusia holbrooki	1	2	1.000	436
	Heterandria formosa	1	2	1.000	399
	Jordanella floridae	1	2	0.817	23
	Lucania goodei	1	2	0.000	-95
	Lepomis punctatus	1	2	0.978	99
Sawgrass - Wet	*Fundulus chrysotus*	1	2	0.000	-77
	Gambusia holbrooki	1	2	0.000	-90
	Heterandria formosa	1	2	0.000	-93
	Jordanella floridae	1	2	1.000	1329
	Lucania goodei	1	2	0.002	-58
	Lepomis punctatus	2	3	0.930	53

Sawgrass - Dry	*Fundulus chrysotus*	1	2	0.000	-95
	Fundulus confluentus		2	0.000	-90
	Gambusia holbrooki	1	2	0.954	114
	Heterandria formosa	1	2	1.000	351
	Jordanella floridae	1	2	0.000	-87
	Lucania goodei	1	2	0.091	-33
	Lepomis punctatus	1	2	0.999	277

Figure A.2.3. Ontogenetic shifts in trophic niche ellipses (SEAb) in different habitat-season levels (columns) for each species (rows) that had multiple size classes within the same habitat-season level.




Dry
Pond
Dry
Sawgrass
Dry
Spikerush
Wet
Pond
Wet
Sawgrass
Wet
Spikerush
H_for
J_flo
L_goo
L_gul
L_pla
L_pun
MDS 2
MDS 1
1.0
0.5
0.0
0.10
0.05
0.00
-0.2
-0.4
-0.6
-0.8
1.5
-0.5
-1.0
0.0
0.5
1.0
1.5
-0.5
0
-1.0
Size Class
1
2
3
4

A.2.8 Trophic Dynamic Shifts in Space and Time

I found seventy-four instances where a species size class had a spatiotemporal change in SEA_b (defined as the probability of SEA_{b1} being > or < SEA_{b2} was ≥ 0.95), and 234 species size classes with a spatiotemporal shift in niche location (proportion of overlap ≤ 5%) (Table A.2.13, A.2.14). In all habitats, from the wet to dry season, SEA_b was more likely to decrease than increase, trophic niche area became smaller (Table A.2.13). Several species had recurring seasonal trends across habitats. For example, during the dry season *G. holbrooki* (2) and *L.goodei* (2) increased in SEA_b, while *H. formosa* (1) SEA_b decreased. However, in some cases the direction of seasonal effects varied among habitats. *G. holbrooki* (1) increased in SEA_b in sawgrass during the dry season, yet SEA_b decreased or had a marginal decrease (probability SEA_{b1} > or < SEA_{b2} ≥ 0.75) in spikerush and sawgrass.

Spatial shifts in SEA_b among habitats were relatively common across taxa, more likely to occur in the wet season, and almost always had overlap below 5% (92% of comparisons). I found thirty-nine species size classes that differed in SEA_b among habitats, twenty-eight in the wet season, and eleven in the dry season (Table A.2.14). All spatial changes in SEA_b during the wet season had overlap below 5% (Table A.2.13, A.2.14). There were fifteen species size classes that occurred in all habitats within the same season (ten – wet, five – dry). In the wet season, half of these had their largest niche area in ponds, while half had their smallest niche area in sawgrass. Meanwhile, during the dry season, four of the five had their largest niche in sawgrass (the other was spikerush) and all had their smallest niche area in ponds.

Table A.2.13. Changes in SEA$_b$ (niche breadth) for different species size classes. A) Spatial changes among habitats. B) Seasonal changes within a habitat. Magnitudes are either significant (95% threshold) or marginal (75% threshold).

A. Spatial			SEA$_b$						
Species Size Class	Season	Habitat Comparison	p (A < B)	%Δ	Magnitude	Direction	Overlap	Shift(s)	Driver(s)
A_nat1	Wet	Pond - Spikerush	0.97	80	Significant	Increase	4.78%	Ds/Dp	D/C/P
A_nat1	Wet	Pond - Sawgrass	0.01	-50	Significant	Decrease	2.63%	Ds/Dp	D
A_nat1	Wet	Spikerush - Sawgrass	0.00	-72	Significant	Decrease	2.61%	Ds/Dp	H/C
F_chr1	Wet	Pond - Spikerush	1.00	181	Significant	Increase	3.10%	Ds/Dp	C/P
F_chr1	Wet	Pond - Sawgrass	0.82	33	Marginal	Increase	1.00%	Dp	DP/P
F_chr1	Wet	Spikerush - Sawgrass	0.12	-53	Marginal	Decrease	1.14%	Dp	DP/C
F_chr2	Wet	Pond - Spikerush	0.00	-57	Significant	Decrease	3.25%	Ds/Dp	C/P
F_chr2	Wet	Pond - Sawgrass	0.00	-68	Significant	Decrease	2.72%	Ds/Dp	C
F_chr2	Wet	Spikerush - Sawgrass	0.16	-25	Marginal	Decrease	2.28%	Dp	C/P
F_con1	Wet	Pond - Spikerush	0.67	18	-	Increase	0.52%	Dp	C
F_con1	Wet	Pond - Sawgrass	0.00	-65	Significant	Decrease	0.56%	Ds/Dp	DP
F_con1	Wet	Spikerush - Sawgrass	0.00	-71	Significant	Decrease	0.97%	Ds/Dp	DP
G_hol1	Wet	Pond - Spikerush	0.00	-69	Significant	Decrease	0.71%	Ds/Dp	DP/C
G_hol1	Wet	Pond - Sawgrass	0.00	-77	Significant	Decrease	1.77%	Ds/Dp	DP/C
G_hol1	Wet	Spikerush - Sawgrass	0.18	-25	Marginal	Decrease	2.08%	Dp	DP/P
G_hol2	Wet	Pond - Spikerush	0.55	10	-	Increase	1.90%	Dp	H/C/P
G_hol2	Wet	Pond - Sawgrass	0.00	-60	Significant	Decrease	0.46%	Ds/Dp	C/P
G_hol2	Wet	Spikerush - Sawgrass	0.00	-64	Significant	Decrease	0.65%	Ds/Dp	DP/C
H_for1	Wet	Pond - Spikerush	0.48	-8	-	Decrease	1.08%	Dp	DP
H_for1	Wet	Pond - Sawgrass	0.97	71	Significant	Increase	1.57%	Ds/Dp	P/DP
H_for1	Wet	Spikerush - Sawgrass	0.92	85	Marginal	Increase	1.48%	Dp	DP
H_for2	Wet	Pond - Spikerush	0.14	-25	Marginal	Decrease	0.86%	Dp	DP

H_for2	Wet	Pond - Sawgrass	1.00	150	Significant	Increase	0.79%	Ds/Dp	DP
H_for2	Wet	Spikerush - Sawgrass	1.00	233	Significant	Increase	0.74%	Ds/Dp	DP
J_flo2	Wet	Pond - Spikerush	0.00	-84	Significant	Decrease	1.23%	Ds/Dp	P/DP
J_flo2	Wet	Pond - Sawgrass	0.72	16	-	Increase	0.30%	Dp	H/DP
J_flo2	Wet	Spikerush - Sawgrass	1.00	631	Significant	Increase	0.33%	Ds/Dp	P/DP
L_mar	Wet	Pond - Spikerush	1.00	25700	Significant	Increase	0.13%	Ds/Dp	D/DP
L_mar	Wet	Pond - Sawgrass	1.00	119200	Significant	Increase	0.17%	Ds/Dp	D/DP/P
L_mar	Wet	Spikerush - Sawgrass	1.00	362	Significant	Increase	0.92%	Ds/Dp	DP/P
L_pun2	Wet	Pond - Spikerush	1.00	3352	Significant	Increase	4.70%	Ds/Dp	DP/P
L_pun2	Wet	Pond - Sawgrass	1.00	1639	Significant	Increase	2.25%	Ds/Dp	DP/P
L_pun2	Wet	Spikerush - Sawgrass	0.01	-50	Significant	Decrease	2.50%	Ds/Dp	DP/P
L_goo1	Wet	Pond - Spikerush	0.22	-51	Marginal	Decrease	1.15%	Dp	DP
L_goo1	Wet	Pond - Sawgrass	0.00	-81	Significant	Decrease	0.90%	Ds/Dp	DP
L_goo1	Wet	Spikerush - Sawgrass	0.01	-62	Significant	Decrease	0.46%	Ds/Dp	DP
L_goo2	Wet	Pond - Spikerush	0.00	-98	Significant	Decrease	0.77%	Ds/Dp	DP
L_goo2	Wet	Pond - Sawgrass	0.00	-92	Significant	Decrease	0.39%	Ds/Dp	DP
L_goo2	Wet	Spikerush - Sawgrass	1.00	300	Significant	Increase	0.40%	Ds/Dp	DP
E_eve	Dry	Pond - Spikerush	1.00	158	Significant	Increase	1.97%	Ds/Dp	DP
E_eve	Dry	Pond - Sawgrass	0.77	15	Marginal	Increase	1.00%	Dp	DP
E_eve	Dry	Spikerush - Sawgrass	0.07	-55	Marginal	Decrease	1.51%	Dp	DP
F_chr1	Dry	Pond - Spikerush	0.50	1	-	Decrease	10.25%	-	-
F_chr1	Dry	Pond - Sawgrass	0.93	74	Marginal	Increase	2.80%	Dp	DP/C
F_chr1	Dry	Spikerush - Sawgrass	0.95	65	Marginal	Increase	3.50%	Dp	DP/C/P
G_hol1	Dry	Pond - Spikerush	0.91	194	Marginal	Increase	0.49%	Dp	DP/C
G_hol1	Dry	Pond - Sawgrass	1.00	700	Significant	Increase	2.14%	Ds/Dp	D/P
G_hol1	Dry	Spikerush - Sawgrass	1.00	172	Significant	Increase	2.43%	Ds/Dp	H/DP
G_hol2	Dry	Pond - Spikerush	0.99	92	Significant	Increase	6.52%	-	H/D/C
G_hol2	Dry	Pond - Sawgrass	0.99	109	Significant	Increase	1.97%	Ds/Dp	H/D/C
G_hol2	Dry	Spikerush - Sawgrass	0.63	9	-	Increase	2.41%	Dp	H/D/C

H_for1	Dry	Pond - Spikerush	0.98	74	Significant	Increase	0.01%	Ds/Dp	DP
H_for1	Dry	Pond - Sawgrass	0.98	77	Significant	Increase	0.97%	Ds/Dp	DP
H_for1	Dry	Spikerush - Sawgrass	0.52	1	-	Increase	0.25%	Dp	DP
H_for2	Dry	Pond - Spikerush	1.00	3618	Significant	Increase	2.27%	Ds/Dp	D/DP
H_for2	Dry	Pond - Sawgrass	1.00	3300	Significant	Increase	0.78%	Ds/Dp	D/C
H_for2	Dry	Spikerush - Sawgrass	0.43	-9	-	Decrease	1.34%	Dp	DP
L_pun2	Dry	Pond - Spikerush	0.24	-26	Marginal	Decrease	6.05%	-	-
L_pun2	Dry	Pond - Sawgrass	0.00	-85	Significant	Decrease	6.63%	-	P
L_pun2	Dry	Spikerush - Sawgrass	0.00	-80	Significant	Decrease	6.80%	-	H/D/P

B. Temporal							SEA$_b$		
Species Size Class	Habitat	Seasonal Comparison	p (A < B)	%Δ	Magnitude	Direction	Overlap	Shifts(s)	Driver(s)
A_nat1	Pond	Wet - Dry	0.72	17	-	Increase	3.63%	Dp	DP/P
A_nat2	Pond	Wet - Dry	1.00	356	Significant	Increase	1.64%	Ds/Dp	D/C/P
A_nat3	Pond	Wet - Dry	0.00	-95	Significant	Decrease	5.02%	Ds	D
A_nat4	Pond	Wet - Dry	0.00	-99	Significant	Decrease	9.38%	Ds	D
B_bel2	Pond	Wet - Dry	0.00	-99	Significant	Decrease	-	Ds	C/P
C_bat	Pond	Wet - Dry	1.00	121	Significant	Increase	8.47%	Ds	D/P
E_eve	Pond	Wet - Dry	0.00	-47	Significant	Decrease	1.99%	Ds/Dp	DP
F_chr1	Pond	Wet - Dry	0.12	-32	Marginal	Decrease	10.14%	Ds	C/P
G_hol1	Pond	Wet - Dry	0.00	-91	Significant	Decrease	0.48%	Ds/Dp	D
G_hol2	Pond	Wet - Dry	1.00	1210	Significant	Increase	6.21%	Ds	H/P
H_for1	Pond	Wet - Dry	0.00	-82	Significant	Decrease	0.01%	Ds/Dp	DP
H_for2	Pond	Wet - Dry	0.31	-8	-	Decrease	2.76%	Dp	D/C/P
L_pla1	Pond	Wet - Dry	0.82	83	Marginal	Increase	2.57%	Dp	C/P
L_pla2	Pond	Wet - Dry	0.00	-86	Significant	Decrease	7.19%	Ds	C
L_gul1	Pond	Wet - Dry	0.00	-83	Significant	Decrease	3.18%	Ds/Dp	P
L_gul2	Pond	Wet - Dry	0.00	-97	Significant	Decrease	2.31%	Ds/Dp	D/P
L_gul3	Pond	Wet - Dry	0.13	-29	Marginal	Decrease	0.93%	Dp	DP/P
L_gul4	Pond	Wet - Dry	0.95	63	Marginal	Increase	3.14%	Dp	D/P
L_mar	Pond	Wet - Dry	1.00	53900	Significant	Increase	0.95%	Ds/Dp	C
L_pun2	Pond	Wet - Dry	1.00	2215	Significant	Increase	5.56%	Ds	D/C/P
L_pun3	Pond	Wet - Dry	0.92	59	Marginal	Increase	12.44%	-	H/D/C/P
A_nat1	Spikerush	Wet - Dry	0.00	-96	Significant	Decrease	3.14%	Ds/Dp	C/P
C_var	Spikerush	Wet - Dry	0.00	-93	Significant	Decrease	-	Ds	H/C
E_eve	Spikerush	Wet - Dry	0.57	6	-	Increase	0.91%	Dp	DP/C

E_glo	Spikerush	Wet - Dry	0.23	-32	Marginal	Decrease	1.22%	Dp	DP
F_chr1	Spikerush	Wet - Dry	0.00	-76	Significant	Decrease	3.10%	Ds/Dp	C/P
F_chr2	Spikerush	Wet - Dry	0.00	-84	Significant	Decrease	7.33%	Ds/Dp	C/P
F_con1	Spikerush	Wet - Dry	0.80	31	Marginal	Increase	2.47%	Dp	DP/P
G_hol1	Spikerush	Wet - Dry	0.19	-27	Marginal	Decrease	1.55%	Dp	DP
G_hol2	Spikerush	Wet - Dry	1.00	129	Significant	Increase	2.89%	Ds/Dp	DP
H_for1	Spikerush	Wet - Dry	0.00	-66	Significant	Decrease	1.40%	Ds/Dp	H/DP
H_for2	Spikerush	Wet - Dry	1.00	4444	Significant	Increase	1.47%	Ds/Dp	DP
J_flo1	Spikerush	Wet - Dry	0.00	-47	Significant	Decrease	1.40%	Ds/Dp	H/D/DP
J_flo2	Spikerush	Wet - Dry	0.30	-33	-	Decrease	2.14%	Dp	H/D/C/P
L_mar	Spikerush	Wet - Dry	0.35	-4	-	Decrease	1.02%	Dp	DP
L_pun1	Spikerush	Wet - Dry	0.00	-89	Significant	Decrease	2.24%	Ds/Dp	DP/P
L_pun2	Spikerush	Wet - Dry	0.05	-51	Marginal	Decrease	3.89%	Dp	H/DP/P
L_goo1	Spikerush	Wet - Dry	0.67	60	-	Increase	0.72%	Dp	DP
L_goo2	Spikerush	Wet - Dry	1.00	108	Significant	Increase	1.23%	Ds/Dp	DP
N_gyr	Spikerush	Wet - Dry	0.00	-65	Significant	Decrease	6.65%	Ds/Dp	C/P
F_chr1	Sawgrass	Wet - Dry	0.45	-12	-	Decrease	3.57%	Dp	C/P
F_chr2	Sawgrass	Wet - Dry	0.00	-63	Significant	Decrease	4.65%	Ds/Dp	C/P
F_con1	Sawgrass	Wet - Dry	1.00	321	Significant	Increase	1.31%	Ds/Dp	DP/P
G_hol1	Sawgrass	Wet - Dry	0.99	212	Significant	Increase	3.96%	Ds/Dp	DP
G_hol2	Sawgrass	Wet - Dry	1.00	6750	Significant	Increase	2.67%	Ds/Dp	DP/P
H_for1	Sawgrass	Wet - Dry	0.00	-64	Significant	Decrease	0.27%	Ds/Dp	DP
H_for2	Sawgrass	Wet - Dry	1.00	1147	Significant	Increase	1.56%	Ds/Dp	DP
J_flo1	Sawgrass	Wet - Dry	1.00	820	Significant	Increase	0.27%	Ds/Dp	P/C
J_flo2	Sawgrass	Wet - Dry	0.00	-91	Significant	Decrease	0.41%	Ds/Dp	P/DP
L_pun2	Sawgrass	Wet - Dry	0.00	-80	Significant	Decrease	-	Ds	C/P
L_goo1	Sawgrass	Wet - Dry	0.93	58	Marginal	Increase	1.29%	Dp	DP
L_goo2	Sawgrass	Wet - Dry	1.00	156	Significant	Increase	1.29%	Ds/Dp	DP

Table A.2.14. Percent overlap in SEA_b between species size class pairs across A) space and time and B) ontogeny; P – pond, Sr – spikerush, Sg – sawgrass, W – wet season, D – dry season.

A. Spatiotemporal				
Species	Size Class	Group A	Group B	Overlap
Ameiurus natalis	1	P-W	P-D	3.63%
Ameiurus natalis	1	P-W	Sr-W	4.78%
Ameiurus natalis	1	P-W	Sr-D	3.37%
Ameiurus natalis	1	P-W	Sg-W	2.63%
Ameiurus natalis	1	P-D	Sg-W	2.52%
Ameiurus natalis	1	P-D	Sg-D	2.52%
Ameiurus natalis	1	Sr-W	P-D	3.38%
Ameiurus natalis	1	Sr-W	Sr-D	3.14%
Ameiurus natalis	1	Sr-W	Sg-W	2.61%
Ameiurus natalis	1	Sr-D	P-D	3.78%
Ameiurus natalis	1	Sr-D	Sg-W	2.80%
Ameiurus natalis	2	P-W	P-D	1.64%
Ameiurus natalis	2	P-W	Sr-W	3.77%
Ameiurus natalis	2	Sr-W	P-D	1.50%
Ameiurus natalis	3	P-W	P-D	5.02%
Ameiurus natalis	4	P-W	P-D	9.38%
Clarias batrachus	-	P-W	P-D	8.47%
Elassoma evergladei	1	P-W	P-D	1.99%
Elassoma evergladei	1	P-W	Sr-W	1.28%
Elassoma evergladei	1	P-W	Sr-D	1.11%
Elassoma evergladei	1	P-W	Sg-D	1.50%
Elassoma evergladei	1	P-D	Sg-D	1.00%
Elassoma evergladei	1	Sr-W	P-D	1.91%
Elassoma evergladei	1	Sr-W	Sr-D	0.91%
Elassoma evergladei	1	Sr-W	Sg-D	1.43%
Elassoma evergladei	1	Sr-D	P-D	1.97%
Elassoma evergladei	1	Sr-D	Sg-D	1.51%
Enneacanthus gloriosus	-	Sr-W	Sr-D	1.22%
Fundulus chrysotus	1	P-W	P-D	10.14%
Fundulus chrysotus	1	P-W	Sr-W	3.10%
Fundulus chrysotus	1	P-W	Sr-D	2.83%
Fundulus chrysotus	1	P-W	Sg-W	1.00%
Fundulus chrysotus	1	P-W	Sg-D	3.16%
Fundulus chrysotus	1	P-D	Sg-W	0.90%

Fundulus chrysotus	1	P-D	Sg-D	2.80%
Fundulus chrysotus	1	Sr-W	P-D	10.20%
Fundulus chrysotus	1	Sr-W	Sr-D	3.10%
Fundulus chrysotus	1	Sr-W	Sg-W	1.14%
Fundulus chrysotus	1	Sr-W	Sg-D	3.44%
Fundulus chrysotus	1	Sr-D	P-D	10.25%
Fundulus chrysotus	1	Sr-D	Sg-W	1.16%
Fundulus chrysotus	1	Sr-D	Sg-D	3.50%
Fundulus chrysotus	1	Sg-W	Sg-D	3.57%
Fundulus chrysotus	2	P-W	Sr-W	3.25%
Fundulus chrysotus	2	P-W	Sr-D	7.36%
Fundulus chrysotus	2	P-W	Sg-W	2.72%
Fundulus chrysotus	2	P-W	Sg-D	4.65%
Fundulus chrysotus	2	Sr-W	Sr-D	7.33%
Fundulus chrysotus	2	Sr-W	Sg-W	2.28%
Fundulus chrysotus	2	Sr-W	Sg-D	4.62%
Fundulus chrysotus	2	Sr-D	Sg-W	2.01%
Fundulus chrysotus	2	Sr-D	Sg-D	4.26%
Fundulus chrysotus	2	Sg-W	Sg-D	4.65%
Fundulus confluentus	1	P-W	Sr-W	0.52%
Fundulus confluentus	1	P-W	Sr-D	2.20%
Fundulus confluentus	1	P-W	Sg-W	0.56%
Fundulus confluentus	1	P-W	Sg-D	0.89%
Fundulus confluentus	1	Sr-W	Sr-D	2.47%
Fundulus confluentus	1	Sr-W	Sg-W	0.97%
Fundulus confluentus	1	Sr-W	Sg-D	1.50%
Fundulus confluentus	1	Sr-D	Sg-W	0.50%
Fundulus confluentus	1	Sr-D	Sg-D	0.84%
Fundulus confluentus	1	Sg-W	Sg-D	1.31%
Fundulus confluentus	2	Sr-D	Sg-D	3.31%
Gambusia holbrooki	1	P-W	P-D	0.48%
Gambusia holbrooki	1	P-W	Sr-W	0.71%
Gambusia holbrooki	1	P-W	Sr-D	1.24%
Gambusia holbrooki	1	P-W	Sg-W	1.77%
Gambusia holbrooki	1	P-W	Sg-D	2.25%
Gambusia holbrooki	1	P-D	Sg-W	1.68%
Gambusia holbrooki	1	P-D	Sg-D	2.14%
Gambusia holbrooki	1	Sr-W	P-D	0.59%
Gambusia holbrooki	1	Sr-W	Sr-D	1.55%
Gambusia holbrooki	1	Sr-W	Sg-W	2.08%

Gambusia holbrooki	1	Sr-W	Sg-D	2.49%
Gambusia holbrooki	1	Sr-D	P-D	0.49%
Gambusia holbrooki	1	Sr-D	Sg-W	2.06%
Gambusia holbrooki	1	Sr-D	Sg-D	2.43%
Gambusia holbrooki	1	Sg-W	Sg-D	3.96%
Gambusia holbrooki	2	P-W	P-D	6.21%
Gambusia holbrooki	2	P-W	Sr-W	1.90%
Gambusia holbrooki	2	P-W	Sr-D	2.36%
Gambusia holbrooki	2	P-W	Sg-W	0.46%
Gambusia holbrooki	2	P-W	Sg-D	1.89%
Gambusia holbrooki	2	P-D	Sg-W	0.48%
Gambusia holbrooki	2	P-D	Sg-D	1.97%
Gambusia holbrooki	2	Sr-W	P-D	6.57%
Gambusia holbrooki	2	Sr-W	Sr-D	2.89%
Gambusia holbrooki	2	Sr-W	Sg-W	0.65%
Gambusia holbrooki	2	Sr-W	Sg-D	2.50%
Gambusia holbrooki	2	Sr-D	P-D	6.52%
Gambusia holbrooki	2	Sr-D	Sg-W	0.61%
Gambusia holbrooki	2	Sr-D	Sg-D	2.41%
Gambusia holbrooki	2	Sg-W	Sg-D	2.67%
Heterandria formosa	1	P-W	P-D	0.01%
Heterandria formosa	1	P-W	Sr-W	1.08%
Heterandria formosa	1	P-W	Sr-D	1.56%
Heterandria formosa	1	P-W	Sg-W	1.57%
Heterandria formosa	1	P-W	Sg-D	1.10%
Heterandria formosa	1	P-D	Sg-W	1.35%
Heterandria formosa	1	P-D	Sg-D	0.97%
Heterandria formosa	1	Sr-W	P-D	0.04%
Heterandria formosa	1	Sr-W	Sr-D	1.40%
Heterandria formosa	1	Sr-W	Sg-W	1.48%
Heterandria formosa	1	Sr-W	Sg-D	1.08%
Heterandria formosa	1	Sr-D	P-D	0.01%
Heterandria formosa	1	Sr-D	Sg-W	1.37%
Heterandria formosa	1	Sr-D	Sg-D	0.25%
Heterandria formosa	1	Sg-W	Sg-D	0.27%
Heterandria formosa	2	P-W	P-D	2.76%
Heterandria formosa	2	P-W	Sr-W	0.86%
Heterandria formosa	2	P-W	Sr-D	1.40%
Heterandria formosa	2	P-W	Sg-W	0.79%
Heterandria formosa	2	P-W	Sg-D	1.56%

Heterandria formosa	2	P-D	Sg-W	0.11%
Heterandria formosa	2	P-D	Sg-D	0.78%
Heterandria formosa	2	Sr-W	P-D	2.76%
Heterandria formosa	2	Sr-W	Sr-D	1.47%
Heterandria formosa	2	Sr-W	Sg-W	0.74%
Heterandria formosa	2	Sr-W	Sg-D	1.56%
Heterandria formosa	2	Sr-D	P-D	2.27%
Heterandria formosa	2	Sr-D	Sg-W	0.18%
Heterandria formosa	2	Sr-D	Sg-D	1.34%
Heterandria formosa	2	Sg-W	Sg-D	1.56%
Jordanella floridae	1	Sr-W	Sr-D	1.40%
Jordanella floridae	1	Sr-W	Sg-W	1.48%
Jordanella floridae	1	Sr-W	Sg-D	1.08%
Jordanella floridae	1	Sr-D	Sg-W	1.37%
Jordanella floridae	1	Sr-D	Sg-W	1.37%
Jordanella floridae	1	Sg-W	Sg-D	0.27%
Jordanella floridae	2	P-W	Sr-W	1.23%
Jordanella floridae	2	P-W	Sr-D	1.97%
Jordanella floridae	2	P-W	Sg-W	0.30%
Jordanella floridae	2	P-W	Sg-D	0.24%
Jordanella floridae	2	Sr-W	Sr-D	2.14%
Jordanella floridae	2	Sr-W	Sg-W	0.33%
Jordanella floridae	2	Sr-W	Sg-D	0.28%
Jordanella floridae	2	Sr-D	Sg-W	0.28%
Jordanella floridae	2	Sr-D	Sg-D	0.21%
Jordanella floridae	2	Sg-W	Sg-D	0.41%
Lepisosteous platyrhincus	1	P-W	P-D	2.57%
Lepisosteous platyrhincus	1	P-W	Sg-W	1.25%
Lepisosteous platyrhincus	1	P-D	Sg-W	1.25%
Lepisosteous platyrhincus	2	P-W	P-D	7.19%
Lepomis gulosus	1	P-W	P-D	3.18%
Lepomis gulosus	2	P-W	P-D	2.31%
Lepomis gulosus	3	P-W	P-D	0.93%
Lepomis gulosus	4	P-W	P-D	3.14%
Lepomis marginatus	-	P-W	P-D	0.95%
Lepomis marginatus	-	P-W	Sr-W	0.13%
Lepomis marginatus	-	P-W	Sr-D	0.30%
Lepomis marginatus	-	P-W	Sg-W	0.17%
Lepomis marginatus	-	P-D	Sg-W	0.28%
Lepomis marginatus	-	Sr-W	P-D	1.66%

Lepomis marginatus	-	Sr-W	Sr-D	1.02%
Lepomis marginatus	-	Sr-W	Sg-W	0.92%
Lepomis marginatus	-	Sr-D	P-D	1.39%
Lepomis marginatus	-	Sr-D	Sg-W	0.52%
Lepomis punctatus	1	P-W	Sr-W	1.62%
Lepomis punctatus	1	P-W	Sr-D	1.87%
Lepomis punctatus	1	P-W	Sg-D	0.83%
Lepomis punctatus	1	Sr-W	Sr-D	2.24%
Lepomis punctatus	1	Sr-W	Sg-D	1.16%
Lepomis punctatus	1	Sr-D	Sg-D	1.06%
Lepomis punctatus	2	P-W	P-D	5.56%
Lepomis punctatus	2	P-W	Sr-W	4.70%
Lepomis punctatus	2	P-W	Sr-D	3.61%
Lepomis punctatus	2	P-W	Sg-W	2.25%
Lepomis punctatus	2	P-W	Sg-D	6.40%
Lepomis punctatus	2	P-D	Sg-W	2.39%
Lepomis punctatus	2	P-D	Sg-D	6.63%
Lepomis punctatus	2	Sr-W	P-D	5.83%
Lepomis punctatus	2	Sr-W	Sr-D	3.89%
Lepomis punctatus	2	Sr-W	Sg-W	2.50%
Lepomis punctatus	2	Sr-W	Sg-D	6.65%
Lepomis punctatus	2	Sr-D	P-D	6.05%
Lepomis punctatus	2	Sr-D	Sg-W	2.50%
Lepomis punctatus	2	Sr-D	Sg-D	6.80%
Lepomis punctatus	2	Sg-W	Sg-D	4.85%
Lepomis punctatus	3	P-W	P-D	12.44%
Lepomis punctatus	3	P-W	Sg-W	1.79%
Lepomis punctatus	3	P-D	Sg-W	1.82%
Lucania goodei	1	P-W	Sr-W	1.15%
Lucania goodei	1	P-W	Sr-D	1.02%
Lucania goodei	1	P-W	Sg-W	0.90%
Lucania goodei	1	P-W	Sg-D	1.27%
Lucania goodei	1	Sr-W	Sr-D	0.72%
Lucania goodei	1	Sr-W	Sg-W	0.46%
Lucania goodei	1	Sr-W	Sg-D	1.27%
Lucania goodei	1	Sr-D	Sg-W	0.60%
Lucania goodei	1	Sr-D	Sg-D	1.27%
Lucania goodei	1	Sg-W	Sg-D	1.29%
Lucania goodei	2	P-W	Sr-W	0.77%
Lucania goodei	2	P-W	Sr-D	1.23%

Lucania goodei	2	P-W	Sg-W	0.39%
Lucania goodei	2	P-W	Sg-D	1.29%
Lucania goodei	2	Sr-W	Sr-D	1.23%
Lucania goodei	2	Sr-W	Sg-W	0.40%
Lucania goodei	2	Sr-W	Sg-D	1.29%
Lucania goodei	2	Sr-D	Sg-W	0.14%
Lucania goodei	2	Sr-D	Sg-D	1.23%
Lucania goodei	2	Sg-W	Sg-D	1.29%
Noturus gyrinus	-	P-D	Sg-W	0.12%
Noturus gyrinus	-	Sr-W	P-D	3.12%
Noturus gyrinus	-	Sr-W	Sr-D	6.65%
Noturus gyrinus	-	Sr-W	Sg-W	0.20%
Noturus gyrinus	-	Sr-D	P-D	2.91%
Noturus gyrinus	-	Sr-D	Sg-W	0.11%

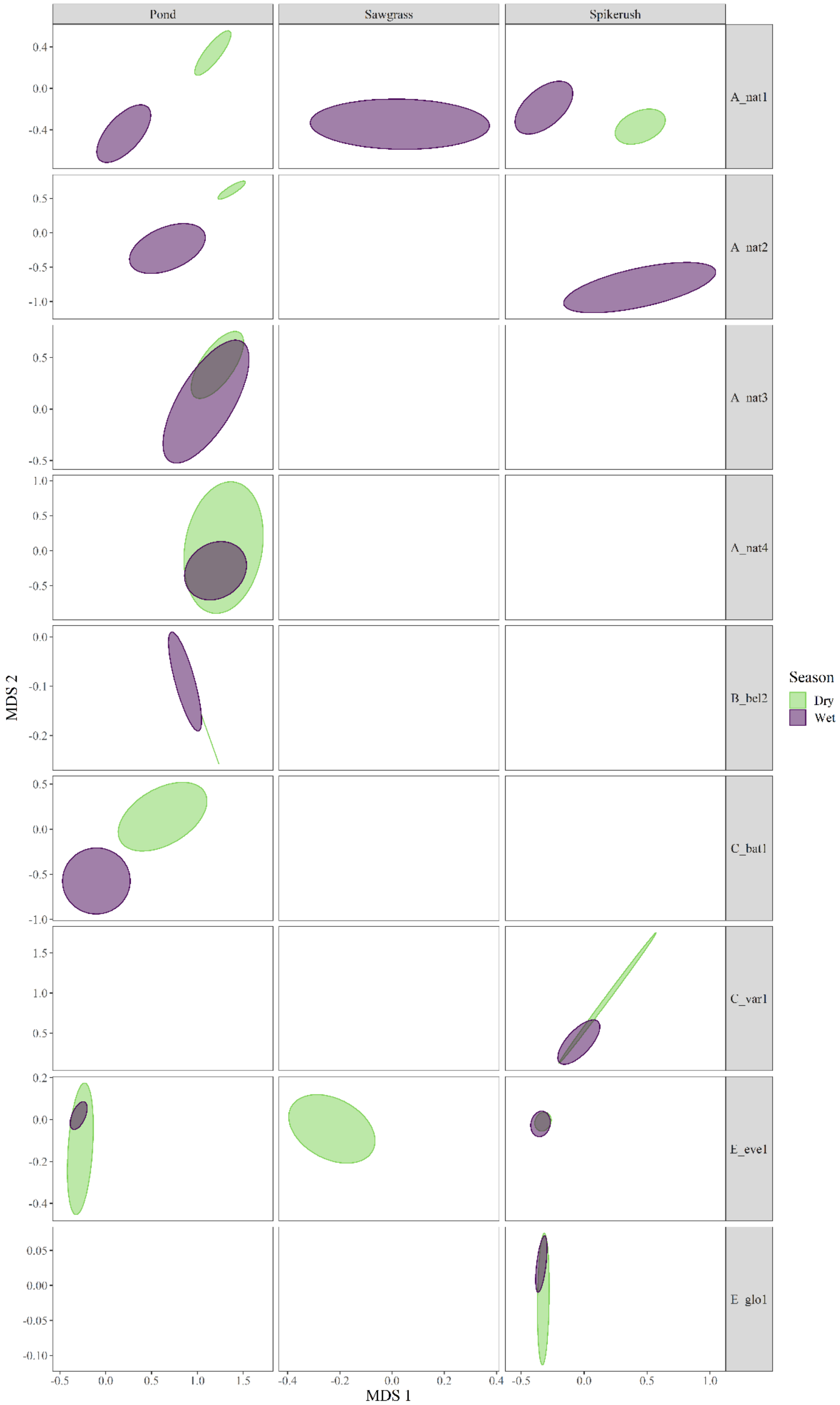
B. Ontogenetic

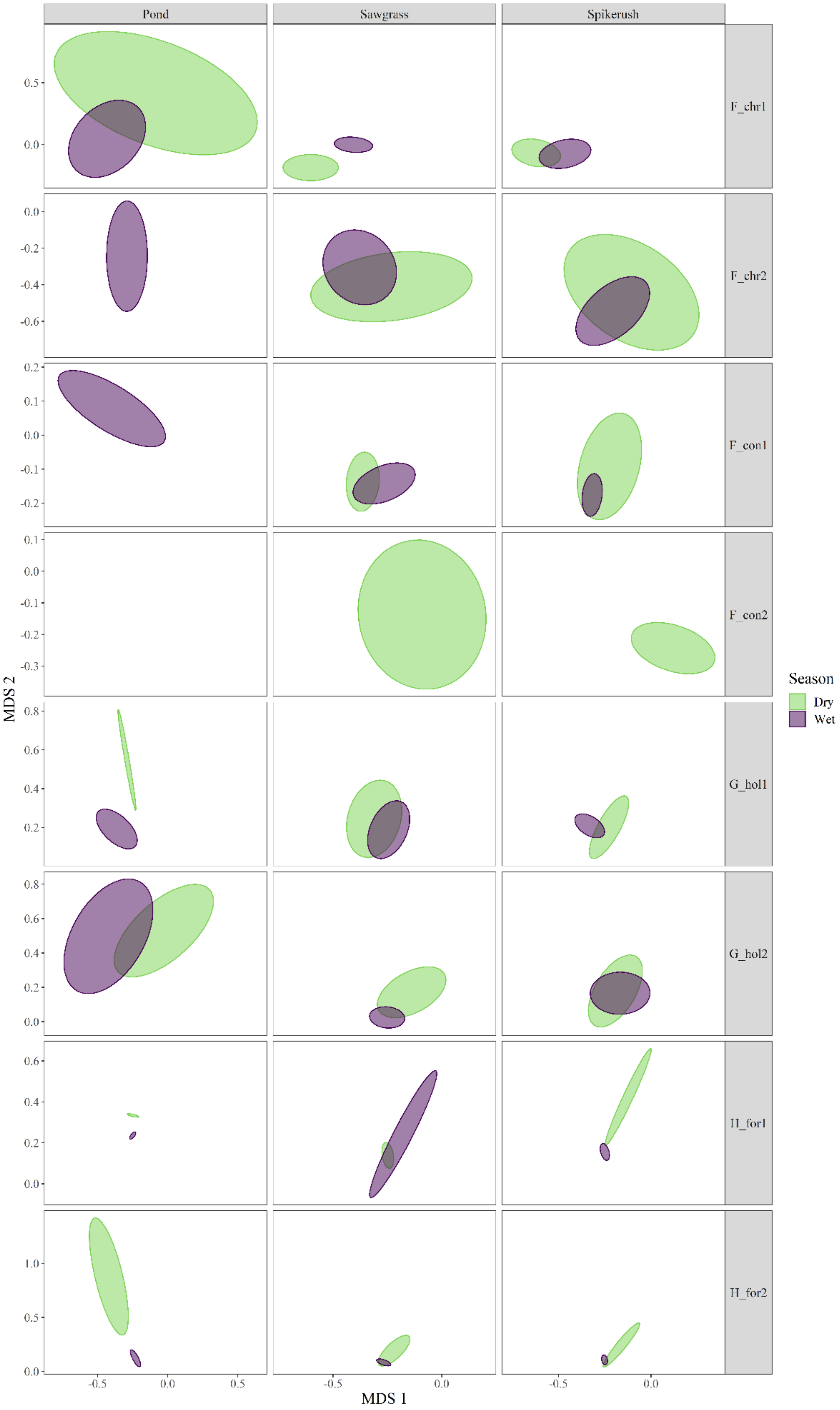
Species	Group A	Group B	Habitat-Season	Overlap
Ameiurus natalis	1	2	P-D	1.91%
Ameiurus natalis	1	3	P-D	5.33%
Ameiurus natalis	1	4	P-D	9.28%
Ameiurus natalis	2	3	P-D	5.36%
Ameiurus natalis	2	4	P-D	9.37%
Ameiurus natalis	3	4	P-D	9.30%
Ameiurus natalis	1	2	P-W	7.34%
Ameiurus natalis	1	3	P-W	7.27%
Ameiurus natalis	1	4	P-W	4.52%
Ameiurus natalis	2	3	P-W	7.37%
Ameiurus natalis	2	4	P-W	4.63%
Ameiurus natalis	3	4	P-W	4.60%
Ameiurus natalis	1	2	Sr-W	3.55%
Erimyzon sucetta	2	3	P-W	0.03%
Fundulus chrysotus	1	2	P-W	1.13%
Fundulus chrysotus	1	2	Sg-D	4.52%
Fundulus chrysotus	1	2	Sg-W	2.73%
Fundulus chrysotus	1	2	Sr-D	7.36%
Fundulus chrysotus	1	2	Sr-W	2.95%
Fundulus confluentus	1	2	Sg-D	3.45%
Fundulus confluentus	1	2	Sr-D	2.03%
Gambusia holbrooki	1	2	P-D	6.36%

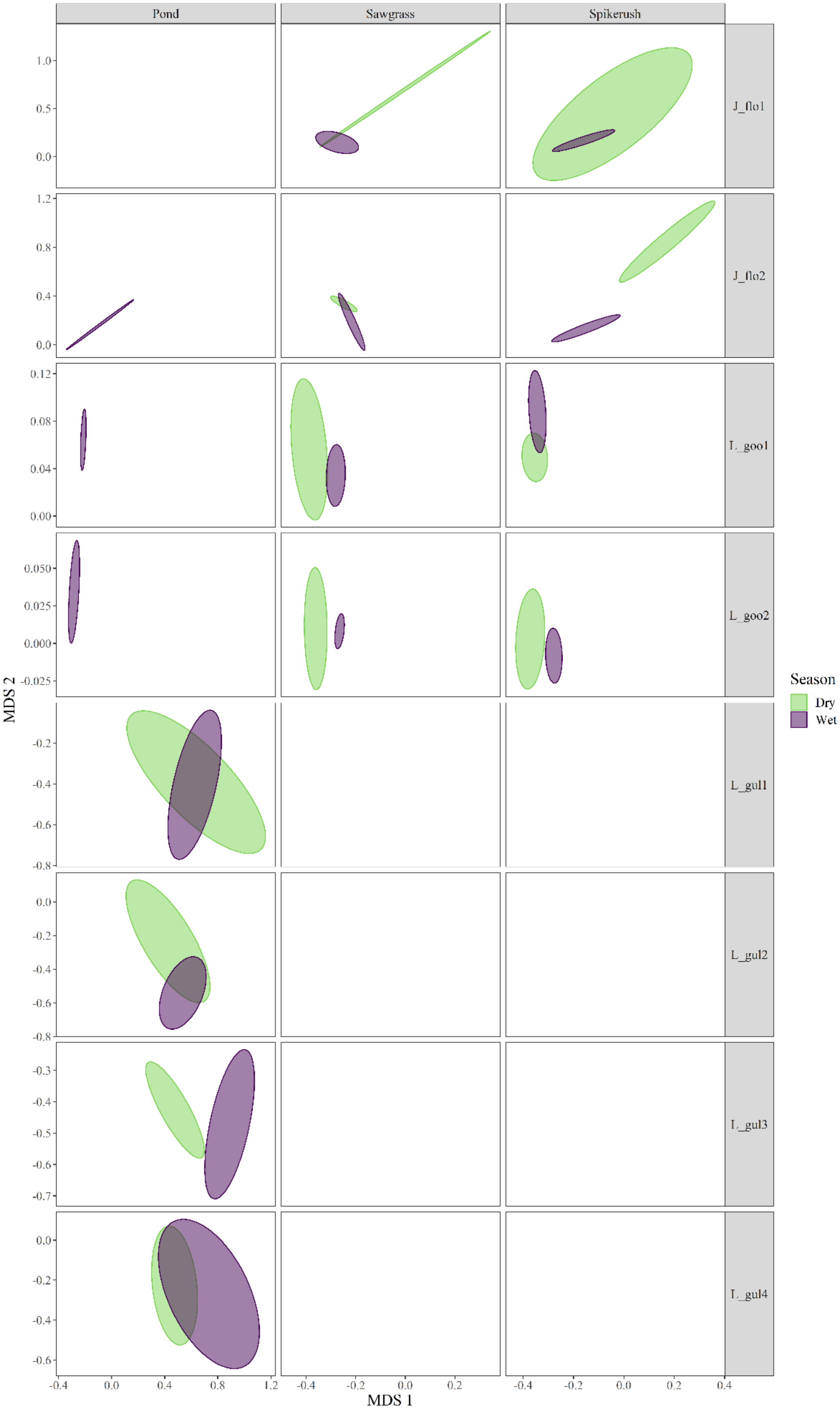
Gambusia holbrooki	1	2	P-W	12.89%
Gambusia holbrooki	1	2	Sg-D	2.47%
Gambusia holbrooki	1	2	Sg-W	0.77%
Gambusia holbrooki	1	2	Sr-D	3.09%
Gambusia holbrooki	1	2	Sr-W	2.68%
Heterandria formosa	1	2	P-D	2.76%
Heterandria formosa	1	2	P-W	0.77%
Heterandria formosa	1	2	Sg-D	1.56%
Heterandria formosa	1	2	Sg-W	0.17%
Heterandria formosa	1	2	Sr-D	1.16%
Heterandria formosa	1	2	Sr-W	0.69%
Jordanella floridae	1	2	Sg-D	0.22%
Jordanella floridae	1	2	Sg-W	0.65%
Jordanella floridae	1	2	Sr-D	1.87%
Jordanella floridae	1	2	Sr-W	1.29%
Lepisosteous platyrhincus	1	2	P-D	7.27%
Lepisosteous platyrhincus	1	2	P-W	10.88%
Lepomis gulosus	1	2	P-D	2.43%
Lepomis gulosus	1	3	P-D	1.04%
Lepomis gulosus	1	4	P-D	3.27%
Lepomis gulosus	2	3	P-D	1.18%
Lepomis gulosus	3	4	P-D	3.63%
Lepomis gulosus	1	2	P-W	3.25%
Lepomis gulosus	1	3	P-W	4.11%
Lepomis gulosus	1	4	P-W	6.59%
Lepomis gulosus	2	3	P-W	4.23%
Lepomis gulosus	2	4	P-W	6.70%
Lepomis gulosus	3	4	P-W	6.65%
Lepomis punctatus	2	3	P-D	12.91%
Lepomis punctatus	1	2	P-W	9.72%
Lepomis punctatus	1	3	P-W	22.90%
Lepomis punctatus	2	3	P-W	22.82%
Lepomis punctatus	1	2	Sg-D	6.68%
Lepomis punctatus	2	3	Sg-W	1.89%
Lepomis punctatus	1	2	Sr-D	4.06%
Lepomis punctatus	1	2	Sr-W	5.20%
Lucania goodei	1	2	P-W	0.99%
Lucania goodei	1	2	Sg-D	1.17%
Lucania goodei	1	2	Sg-W	0.38%
Lucania goodei	1	2	Sr-D	1.22%

Lucania goodei	1	2	Sr-W	0.42%
Mayaheros urophthalmus	1	2	P-W	4.56%

Figure A.2.4. Seasonal comparisons of SEA$_b$ for species size classes (rows) in each habitat (columns) when that species size class was present within a habitat in both seasons. See Table A.2.13 and A.2.14 for statistical comparisons of ellipses illustrated in this figure, Table 1 for species size class abbreviations, and Table A.1.2 for sample sizes for each species size class in each habitat-season level.




Pond
Sawgrass
Spikerush
F_chr1
F_chr2
F_con1
F_con2
G_hol1
G_hol2
H_for1
H_for2
MDS 2
MDS 1
-0.5
0.0
0.5
Season
Dry
Wet


Pond
Sawgrass
Spikerush
J_flo1
J_flo2
L_goo1
L_goo2
L_gul1
L_gul2
L_gul3
L_gul4
1.0
0.5
0.0
1.2
0.8
0.4
0.0
0.12
0.08
0.04
0.00
0.050
0.025
0.000
-0.025
-0.2
-0.4
-0.6
-0.8
0.0
-0.2
-0.4
-0.6
-0.8
-0.3
-0.4
-0.5
-0.6
-0.7
0.0
-0.2
-0.4
-0.6
-0.4
0.0
0.4
0.8
1.2
-0.4
-0.2
0.0
0.2
-0.4
-0.2
0.0
0.2
0.4
MDS 1
MDS 2
Season
Dry
Wet

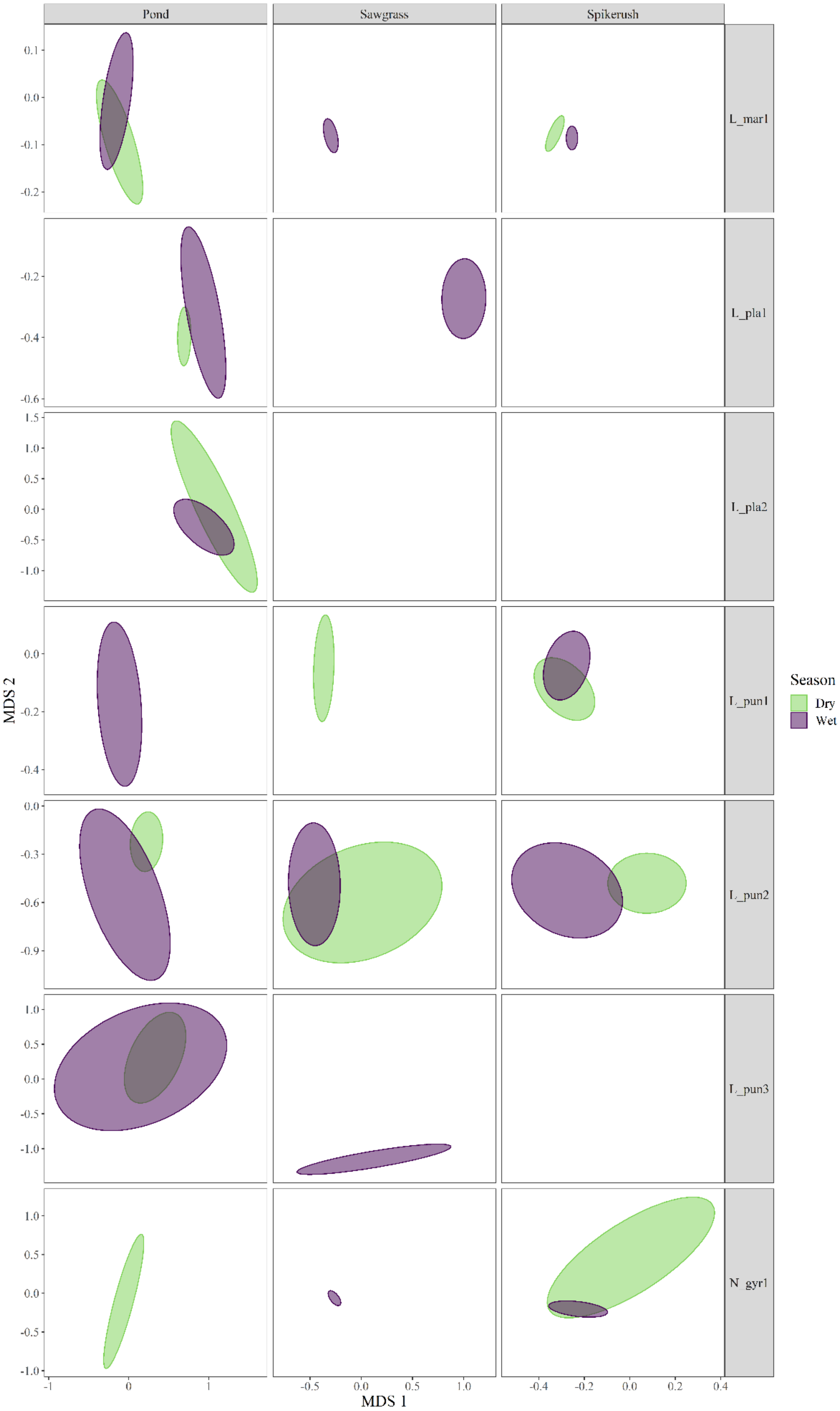
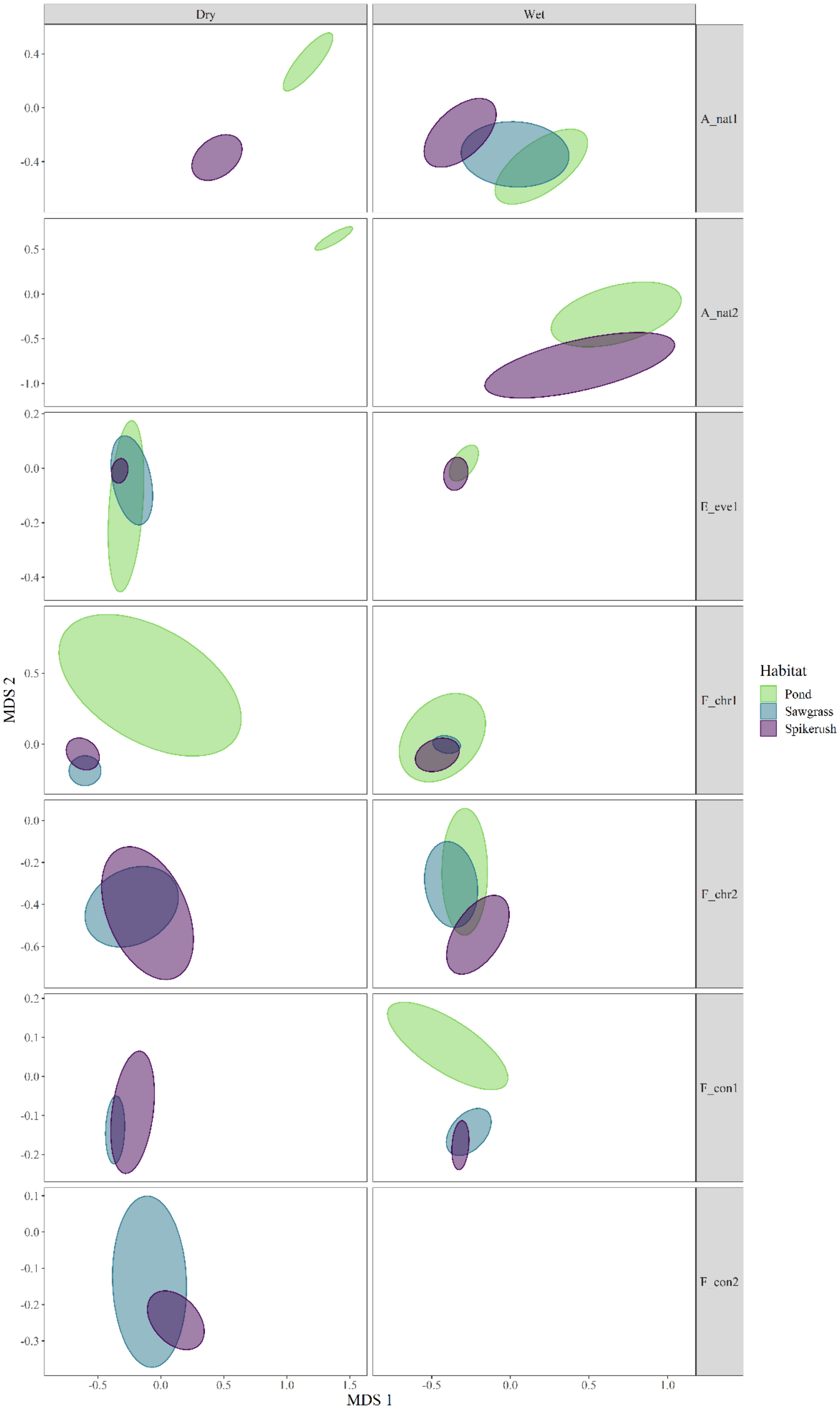
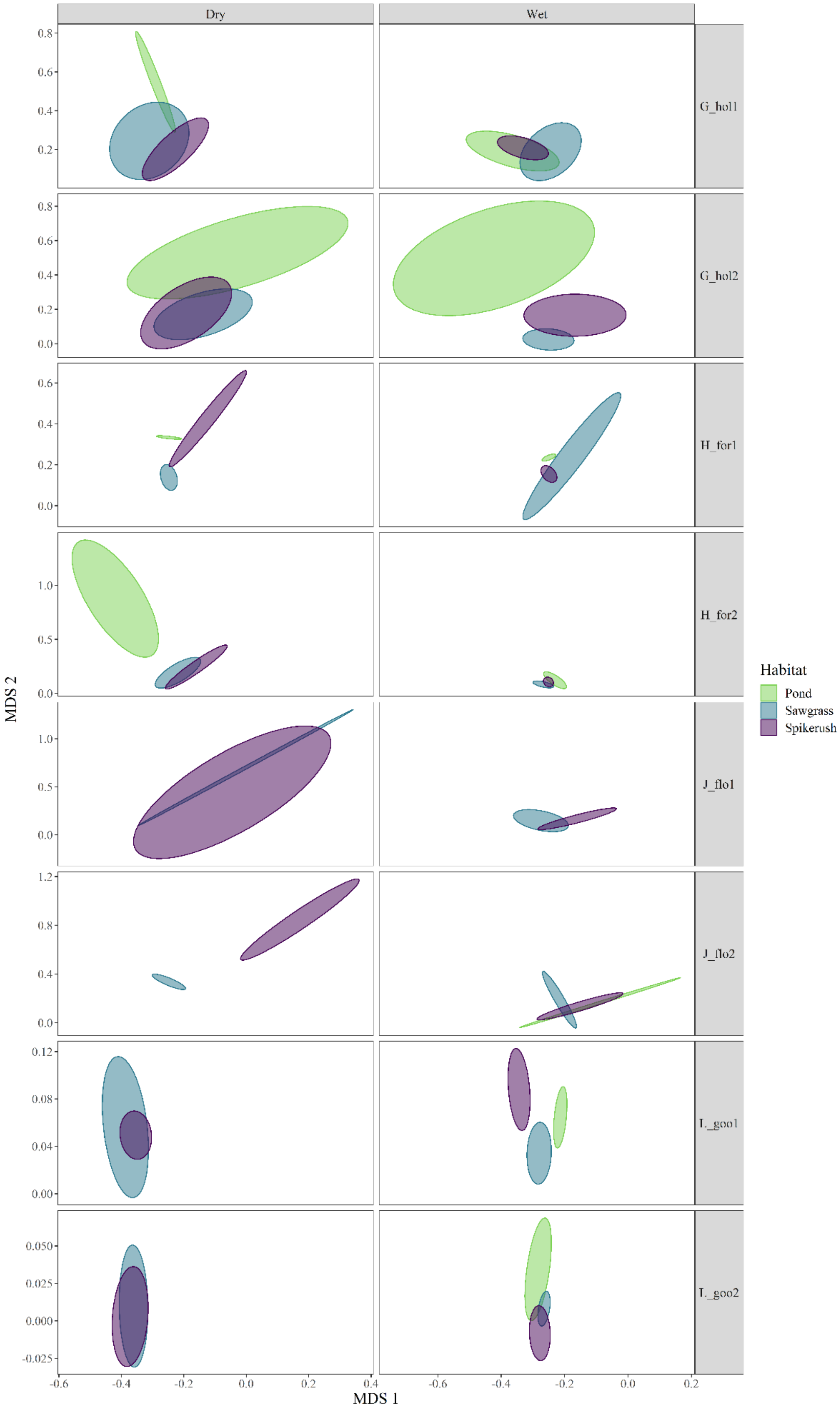
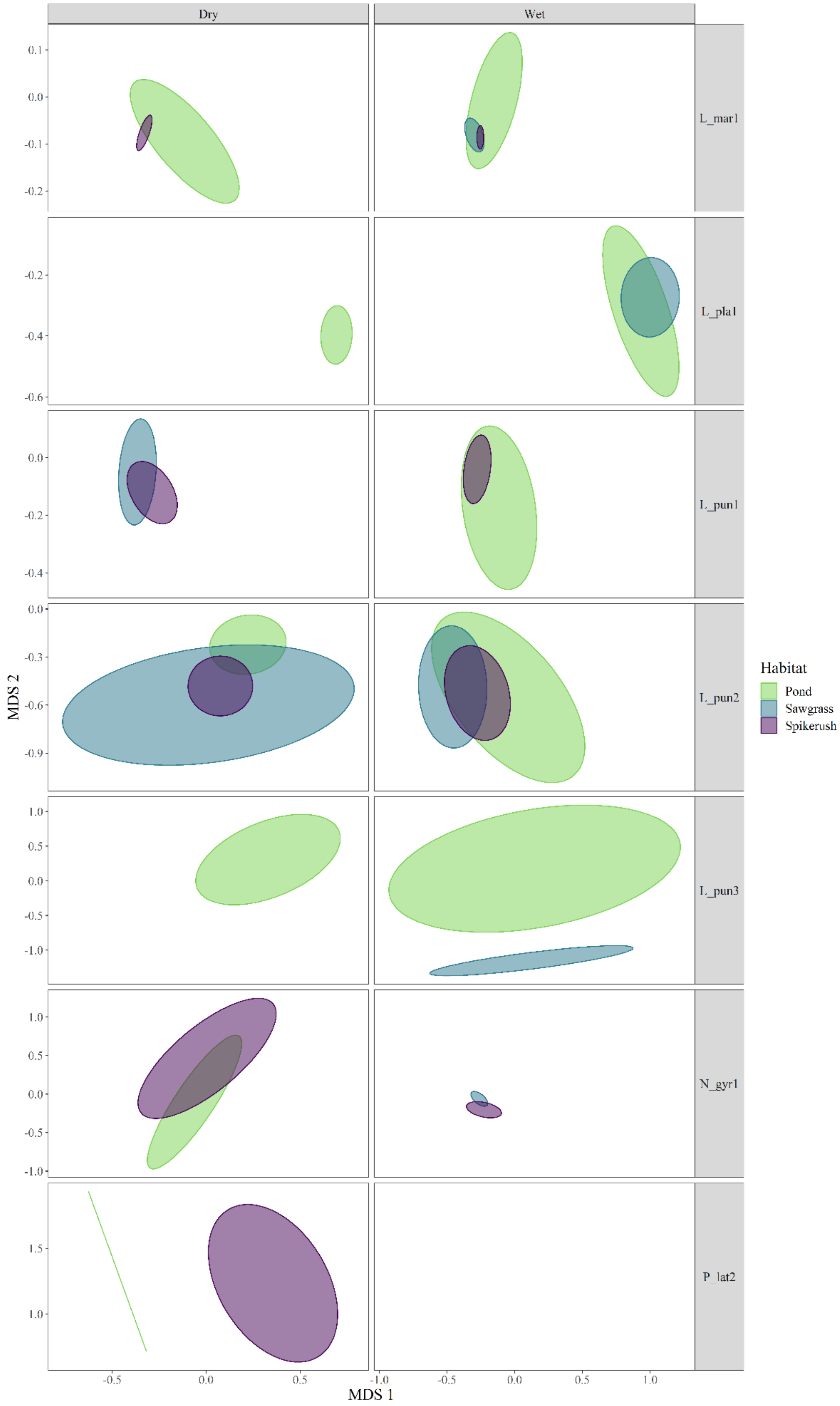

Pond
Sawgrass
Spikerush
L_mar1
L_pla1
L_pla2
L_pun1
L_pun2
L_pun3
N_gyr1
0.1
0.0
-0.1
-0.2
-0.2
-0.4
-0.6
1.5
1.0
0.5
0.0
-0.5
-1.0
0.0
-0.2
-0.4
0.0
-0.3
-0.6
-0.9
1.0
0.5
0.0
-0.5
-1.0
1.0
0.5
0.0
-0.5
-1.0
-1
0
1
-0.5
0.0
0.5
1.0
-0.4
-0.2
0.0
0.2
0.4
MDS 1
MDS 2
Season
Dry
Wet

Figure A.2.5. Habitat comparisons of SEA_b for species size classes (rows) that occurred in multiple habitats within the same season (columns). See Table A.2.13 and A.2.14 for statistical comparisons of ellipses illustrated in this figure, Table 1 for species size class abbreviations, and Table A.1.2 for sample sizes for each species size class in each habitat-season level.





Dry
Wet
G_hol1
G_hol2
H_for1
H_for2
J_flo1
J_flo2
L_goo1
L_goo2
MDS 2
MDS 1
Habitat
Pond
Sawgrass
Spikerush
0.8
0.6
0.4
0.2
0.0
1.0
0.5
1.2
0.12
0.08
0.04
0.00
0.050
0.025
0.000
-0.025
-0.6
-0.4
-0.2
0.0
0.2
0.4



Dry
Wet
L_mar1
L_pla1
L_pun1
L_pun2
L_pun3
N_gyr1
P_lat2
MDS 2
MDS 1
0.1
0.0
-0.1
-0.2
-0.2
-0.4
-0.6
0.0
-0.2
-0.4
0.0
-0.3
-0.6
-0.9
1.0
0.5
0.0
-0.5
-1.0
1.0
0.5
0.0
-0.5
-1.0
1.5
1.0
-0.5
0.0
0.5
-1.0
-0.5
0.0
0.5
1.0
Habitat
Pond
Sawgrass
Spikerush

Appendix A.3 Data Availability

All of the raw data (stomach-content counts and volume, and stable-isotope data) and R code used for this manuscript will be made available through the Florida Coastal Everglades Long-Term Ecological Research website (https://fce-lter.fiu.edu/data/core/#results) within one year of publication.

CHAPTER III

HABITAT ENGINEERING BY AN APEX PREDATOR GENERATES SPATIAL TROPHIC DYNAMICS ACROSS A TEMPORAL ENVIRONMENTAL STRESS GRADIENT

Abstract

The Stress Gradient Hypothesis (SGH) predicts that the strength of facilitative and competitive species interactions will be inversely related with facilitative interactions increasing under stressful conditions. Ecosystem engineering is one such facilitative interaction and generates bottom-up effects on species abundance, species richness, and food-web structure and function. The American Alligator (*Alligator mississippiensis*) is an abundant, large-bodied predator that physically modifies the environment by creation and maintenance of "alligator ponds". I used complementary datasets of stomach contents and stable isotopes of carbon and nitrogen to test the SGH and effects of ecosystem engineering on trophic dynamics by quantifying diets, trophic niche sizes, trophic positions, and basal resource use (autotrophic vs heterotrophic) among habitats in both wet and dry seasons creating an environmental stress gradient (stress increasing during the dry season). I found that many consumers underwent dietary shifts in engineered habitats, primarily as a result of differential consumption of omnivorous invertebrates and detritus, that trophic niches increased in ponds during the dry season based on stomach contents, yet decreased based on stable isotopes, that when trophic position did shift it usually increased in deeper habitats, and that consumers primarily relied on detrital energy. My findings demonstrate the bottom-up effects of an ecosystem engineer that is also a top predator on trophic dynamics. Trophic niches derived from stomach contents indicated decreased competition in the dry season consistent with the SGH, while trophic niches derived from stable isotopes suggested more complex dynamics (e.g., temporal interaction of consumption and competition) may be governing species interactions. Alligator engineering ameliorates environmental stress by creating deep-water habitats

that provide dry-season refuge for taxa that would other lack suitable habitat, inherently increasing individual fitness.

Introduction

The dual nature of ecosystem engineers as food-web members that modify the physical environment, thus affecting other species through both feeding and habitat alteration, is understudied and may be one of the most important connections between trophic and non-trophic interactions (Sanders et al. 2014). Facilitation by ecosystem engineers extends across trophic levels altering species densities, diversity, and food-web structure (Sanders and van Veen 2011, van der Zee et al. 2016). Furthermore, trait-mediated indirect effects that result from these habitat modifications can intensify bottom-up effects and diminish top-down effects (Zhong et al. 2017). The portrait that materializes from this work demonstrates that habitat engineering, through direct and trait-mediated indirect effects, interacts with environmental variability to influence heterogeneity in trophic dynamics (Wetzel et al. 2016). Despite the profound influences on food webs found in a handful of studies, little is known about the effects of engineering on food-web structure and function, while even less is known about the potentially compounding effects of animals that are both predators and engineers (Sanders and van Veen 2011, Sanders et al. 2014).

Ecosystem engineers may ameliorate the effects of environmental harshness and stress across spatial and temporal gradients by reducing competitive interactions (Lowney and Thomson 2021, 2022). The Stress Gradient Hypothesis (SGH) states that the intensity of facilitation and competition will be inversely related along environmental stress gradients, with facilitative interactions increasing under stressful conditions (Bertness and Callaway 1994, Malkinson and Tielbörger 2010). While this hypothesis has predominantly been tested in plant communities, recently there have been examples using birds and arthropods (Dangles et al. 2018, Lowney and Thomson 2021, 2022, García-Navas et al. 2021). The Niche Variation Hypothesis (NVH) is related to the SGH and predicts that competition strength and niche area are inversely related (van Valen 1965, Bolnick et al. 2007). I hypothesize a link between ecosystem engineering, SGH, and NVH, and I predict that effects of ecosystem engineering (facilitation) intensify under environmental stress that reduces competition (SGH) and increases niche areas (NVH) (see Bolnick et al. 2010).

In the Everglades, USA, American Alligators (*Alligator mississippiensis*) are an abundant predator that modify the environment by creating and maintaining "alligator ponds" through removal of vegetation and redistribution of sediment (Kushlan 1974, Campbell and Mazzotti 2004, Palmer and Mazzotti 2004, Rosenblatt et al. 2015). This results in altered plant, invertebrate, and fish communities in addition to a gradient of nutrient enrichment with alligator ponds having elevated nutrient levels (Strickland 2020). The Everglades experiences seasonal rainfall leading to dry seasons with a dropping water table and concentration of mobile aquatic animals (Loftus and Kushlan 1987, Gaiser et al. 2012). The functional importance of alligator engineering is likely to change across resulting hydrological stress gradients. Ameliorating drying-induced stress may be essential for maintaining ecosystem function in short-hydroperiod environments prone to drying and by creating competitor- or predator-free habitat in more benign environments (Crain and Bertness 2006).

In this study I use stomach contents and stable isotopes of carbon and nitrogen to assess the impacts of alligator-engineering on diets, trophic niches, trophic positions, and basal resource use (autotrophic versus heterotrophic) of aquatic consumers across environmental gradients in space and time. Additionally, I test the SGH across this gradient mediated by the NVH using trophic niches. I predicted that consumers will undergo dietary shifts in engineered habitats as a result of altered plant and consumer communities (Strickland 2020), that trophic niches will increase as environmental stress increases (i.e., in the dry season) and competition decreases (Bolnick et al. 2010, Malkinson and Tielbörger 2010), and that trophic positions will increase in deeper habitats (McHugh et al. 2015). Furthermore, I predicted that elevated nutrient levels in engineered habitats will generate bottom-up effects that lead to increased autotrophic energy use (Power 1992, Polis et al. 1997, Polis 1999).

Methods

Study Design

Everglades National Park, Florida, USA has two major drainages, Shark River Slough (SRS) and Taylor Slough (TS), that carry freshwater through the Everglades to Florida Bay. TS is smaller and dries faster than SRS, however the two sloughs have many similarities (Kotun and Renshaw 2014). These sloughs are predominantly wet prairie (emergent rush marsh, *Eleocharis* spp.) with large stocks of periphyton (Gunderson and Loftus 1993, Turner et al. 1999). Throughout this landscape alligators dig and maintain alligator ponds or alligator holes (Campbell and Mazzotti 2004, Brandt et al. 2010). This alligator-engineering results in two distinct habitats: 1) a pool or semi-open water habitat that is typically 1 m deeper than the surrounding marsh (hereafter "pond") and 2) a dense ring of thick vegetation, including woody vegetation, immediately surrounding the pond (hereafter "near-pond") (Campbell and Mazzotti 2004, Palmer and Mazzotti 2004). Using a halo design, I sampled these two habitats and the adjacent marsh (three habitats total) at ten ponds, five in SRS and five in TS (Fig. 1), once to represent the wet season (October – December 2018) and once to represent the dry season (March-April 2019) unless prevented by water depth (e.g., near-pond habitats in TS had dried by the time of our dry-season sampling, Table S1). This creates environmental gradients of depth (near-ponds were the shallowest habitat and ponds the deepest) and of environmental harshness (dry-season water recession increases stress).

Field Collections

I collected basal resources and aquatic consumers for analysis of stable isotopes of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) and of stomach contents for vertebrates (Appendix S1). Basal resources included flocculent organic benthic matter (hereafter "floc"), vascular plants, *Utricularia* spp., *Nostoc* spp. (cyanobacteria colony and associated community that forms a gelatinous ball), and periphyton (Table S2). I used a variety of methods to collect aquatic consumers including benthic meiofauna, macroinvertebrates, fishes, and amphibians (Appendix S2). Vertebrates were sorted into size classes based on length (Table S3) to account for ontogenetic shift in feeding (Wainwright and Richard 1995, Piet et al. 1999). Our target sample size for each species size class in each habitat during each season was twenty individuals, however that was not always the case (Table S2).

Statistical Analyses

To explore how consumer diets shifted in alligator-engineered habitats, I quantified diets based on counts of stomach contents for both the entire consumer community and individual species size classes (populations) in all habitats and seasons. I used non-metric multidimensional scaling (NMDS, k = 2) with Morisita-Horn distances (Jost et al. 2011) to visualize community-wide dynamics in prey consumption. Permutational multivariate analysis of variance (PERMANOVA; 10,000 iterations) follow by similarity percentages (SIMPER) was used to determine differences in the prey community of all consumers among habitats and seasons. Furthermore, spatiotemporal changes in diets of species size classes were examined using pairwise PERMANOVAs. These and all analyses were conducted using the *vegan* package, *RVAideMemoire* package (*pariwise.perm.manova* function), and base R (Hervé 2021, Oksanen et al. 2022, R Core Team 2022).

To examine the effects of alligator engineering on trophic niches of aquatic consumers, I modeled trophic niches for species size classes using both stomach contents and $\delta^{15}N$ and $\delta^{13}C$. Trophic niches were modeled with the *SIBER* package in R (Jackson et al. 2011). For niches derived from stomach contents, I used axes one and two from NMDS in place of $\delta^{13}C$ and $\delta^{15}N$ (Chapter 2). Differences in trophic niche area (trophic dispersion) for species size classes in different habitats and seasons were determined using the *pairwiseComparisons* function from *tRophicPosition* package (Quezada-Romegialli et al. 2018).

To quantify the impact of alligator engineering on trophic position, trophic positions were modeled separately from stomach contents and stable isotopes and compared across habitats and seasons. Trophic positions based on stomach contents were a function of prey trophic position following Adams et al. (1983). Permuted analysis of variance (permuted ANOVA) and subsequent pairwise permutation tests were used for habitat and season comparisons. These were conducted in R using *aovp* and *pairwisePermutationTest* function in *lmPerm* and *rcompanion* packages respectively (Wheeler and Torchiano 2016, Mangiafico 2021). Meanwhile, I used the *tRophicPosition* package to model trophic positions and a metric 'alpha' based on stable isotopes (Quezada-Romegialli et al. 2018). Alpha is the proportion of $\delta^{15}N$ in a consumer's tissues derived from detritus (floc) and one minus alpha is the proportion derived from green algae. As a result, alpha represent a spectrum of brown (< 0.5) to green (> 0.5) energy use. Trophic position and alpha were modeled using *multiSpeciesTP* function and compared across habitats and season with the *pairwiseComparisons* function.

Results

Dietary Shifts

Our analyses revealed that aquatic consumer communities and populations relied on different prey in alligator-engineered habitats compared to adjacent marshes. PERMANOVA revealed that prey ingestion was structured by an interaction among consumer size classes, slough, habitat, and season ($F = 1.75$, $p < 0.005$, Table S3) and NMDS ($k = 2$, stress $= 0.23$) separated prey by size on MDS 1 (Fig. S1). Meiofauna were the most abundant prey in consumer guts across habitats and seasons, however, the relative abundances of taxa shifted (Fig 1). For example, in SRS during the wet season, for marsh consumers, ostracods and aquatic mites (Hydrachnidia) were the second and third most abundant prey items, while in ponds cladocerans were the second most abundant prey approximately tripling in average count per individual stomach from marsh consumers to pond consumers (Fig 1). Aquatic snails (Mollusca) were the most abundant prey item in guts of consumers from SRS marshes and ponds during the wet season, but were rare or absent in the guts of consumers from other slough-habitat-season levels. Dietary shifts (pairwise PERMANOVA $p \leq 0.05$) for species size classes occurred in 21% of seasonal comparisons, 11% of habitat comparisons, and 8% of slough comparisons (Table S4). Habitat dietary shifts were more common in near-pond-pond comparisons (19%) than marsh-pond comparisons (13%) and more frequent during the dry season (73% of shifts). Dietary shifts between habitats were driven (≥ 10% change in percent contribution to diet per prey category) by shifts in consumption of omnivorous invertebrates (100% of shifts), detritus (73%), carnivorous invertebrates (64%), herbivorous invertebrates (36%), and producers (9%).

Trophic Dispersion

For trophic niches based on stomach contents, I documented trophic dispersion between habitats more frequently when comparing niches between ponds and near-ponds (the two alligator-engineered habitats). Trophic niche sizes increased under elevated environmental stress (i.e., during the dry season) in ponds. For comparisons within a species size class, trophic dispersion occurred in 47% of seasonal comparisons, 40% of slough comparisons, and 37% of habitat comparisons (Table 1, S5, Fig 3). For the species size classes that underwent trophic dispersion among habitats, trophic dispersion was more likely in near-pond-pond comparisons (47%) than either marsh-pond (32%) or marsh-near-pond (33%) comparisons (Table 1, S5). Magnitude (absolute value of percent change) of spatial shifts was greatest in near-pond-pond comparisons (1,809%), half as much in marsh-pond comparisons (907%), and substantially lower in marsh-near-pond comparisons (108%) (Table 1). Trophic niches increased in ponds compared to marshes (70% of shifts) and near-ponds (100% of shifts) (Fig. 3, Table S5). Seasonal shifts occurred in 75% of comparisons in SRS near-ponds (TS near-ponds dried prior to dry season sampling), while seasonal shifts were more common in marsh comparisons (SRS marsh = 40%; TS marsh = 60%) than pond comparisons (SRS ponds = 38%; TS ponds = 50%) (Table S5). Magnitude of seasonal change was greatest in SRS near-ponds (3330%), similar in SRS marshes (167%) and ponds (197%), but more than twice as large in TS marshes (746%) compared to TS ponds (354%). In ponds, trophic niche sizes usually increased during the dry season (63% of shifts), yet typically decreased in all other habitats.

Comparisons of trophic niches modeled using stable isotopes had similar trends revealing trophic dispersion among slough-habitat-season levels with trophic dispersion being more frequent in near-pond-pond comparisons. I documented trophic dispersion in 43% of seasonal comparisons, 42% of habitat comparisons, and 41% of slough comparisons (Table 2, S6, Fig 4). Similar to stomach contents, trophic dispersion based on stable isotopes occurred more frequently in near-pond-pond comparisons (52%) than either marsh-pond (41%) or marsh-near-pond comparisons (37%). However, with isotopes I saw the greatest magnitude of spatial change in marsh-near-pond comparisons (1074%) and approximately half that in marsh-pond (499%) and near-pond-pond (482%) comparisons (Table S6). Trophic niches typically increased in ponds relative to near-ponds (71% of shifts), while increases and decreases were equally likely in marshes relative to ponds (50% of shifts). I observed seasonal shifts at similar rates among habitats in SRS (marsh = 44% of comparisons; near-pond = 44%; pond = 42%), while seasonal shifts were more common in TS marshes (50%) than near-ponds (33%) or ponds (33%). In both sloughs, magnitude of seasonal shifts was greatest in ponds (SRS = 343%; TS = 1762%) followed by marshes (SRS = 101%, TS = 414%) with the smallest changes found in near-ponds (SRS = 71%; TS = 67%). In ponds, most species size classes displayed decreased trophic niche area in the dry season, while in other habits most species size classes had increased trophic niche areas.

Trophic Position

I found that shifts in trophic position were uncommon and that when changes in trophic position did occur they were usually higher in alligator-engineered habitats. For trophic positions based on stomach contents there were only shifts in trophic position for three species size classes in different habitats that on average shifted half a trophic level (Table 3, S7). Based on stable isotopes, trophic positions shifted more between sloughs (35%) than between seasons (19%) or among habitats (17%) (Table 4). For comparisons across both sloughs and seasons, trophic position was typically higher in alligator-engineered habitats (59% of comparisons) than in adjacent marshes (Table S8). The direct of change in trophic position between habitats was typically directly related to depth. Additionally, most seasonal shifts were decreases in trophic position, except for consumers in ponds where 66% of shifts in trophic position were increases of approximately half of a trophic level.

Basal Energy Use

Shifts in basal energy use (alpha) were rare, but those that did occur were usually towards more autotrophic (green) energy used in near-pond habitats. I documented shifts in alpha in 14% of slough comparisons, 8% of seasonal comparisons, and 4% of habitat comparisons (Fig. 6, Table S9). For habitat shifts that did occur, alpha decreased in near-ponds relative to marshes and ponds in all cases (Table S9). Alpha was higher in the wet season relative to the dry season in all habitats of both sloughs, except for SRS ponds (Table S9).

Discussion

The effects of ecosystem engineers on food webs, especially ecosystem engineers that are also predators, are not well understood (Sanders and van Veen 2011, Sanders et al. 2014). In this study I demonstrated that a large-bodied predator and ecosystem engineer, through trait-mediated indirect effects, drove dietary shifts in aquatic consumers, differential shifts in trophic niche dynamics across an environmental stress gradient (i.e., increased environmental stress during the dry season), and increased trophic positions in deeper habitats. I found that there were rarely changes in basal resource use, despite an alligator-engineered phosphorus (P) gradient (Strickland 2020). Meanwhile, support for the classical interpretation of the SGH depended on diet tracing method (i.e. stomach content vs stable isotopes), yet overall alligator-engineered ponds ameliorated dry-season environmental stress by providing suitable habitat and facilitating increased individual fitness.

Dietary shifts among habitats were more common in the dry season when aquatic animal densities are high, and when comparing ponds to other habitats. All shifts in diet resulted at least in part from differential consumption of omnivorous invertebrates, a group that contains meiofauna (e.g., copepods and cladocerans), among habitats. In freshwater lakes, the relative abundance of different meiofauna changed based on the limiting nutrient (Sterner and Elser 2002). This trend holds in our study system, an oligotrophic (P-limited) wetland, across this alligator-engineered habitat and P gradient (Strickland 2020). Therefore, it is not surprising that these bottom-up effects of nutrient enrichment that drive spatial variation in prey availability extend across trophic levels to spatial variation in prey consumption. While facilitation by ecosystem engineers has been shown to have similar effects on trophic dynamics in other ecosystems (Sanders and van Veen 2011, van der Zee et al. 2016), to our knowledge this is the first time such effects have been demonstrated for an ecosystem engineer that is also a large-bodied predator.

Alligator-engineering results in habitats with different depths (Kushlan 1974, Strickland 2020) and I found that when species did undergo shifts in trophic position, it was usually higher in habitats that were deeper. Seasonal shifts in trophic position in response to dry-season water recession were decreases for all habitats, except for alligator-engineered ponds. Aside from ponds, this is consistent with other studies that have shown habitat size and availability are directly related to food chain length (Takimoto and Post 2013, McHugh et al. 2015, Brauns et al. 2022). In our system, I suspect that as animals become concentrated in ponds during the dry season (Loftus and Kushlan 1987, Parkos et al. 2011) this leads to increased consumption of invertebrates by smaller fishes and invertebrate predators, and increased piscivory by larger fishes, leading to increased dry-season trophic positions in ponds. This pattern may reverse as the dry season continues and prey become scarce, and consumers switch to lower quality prey such as detritus.

Consumers mostly relied on detrital energy and shifts in basal resource use were rare across sloughs, seasons, and habitats. The lack of shifts defied our expectations based on increased P-levels in alligator-engineered habitats (Strickland 2020) and was in contrast to previous work that found habitat modification by ecosystem engineers increased the strength of bottom-up effects (Zhong et al. 2017). I predicted that increased availability of the limiting nutrient (P) would lead to increased producer biomass and growth resulting in a bottom-up shift towards autotrophic energy throughout the food web (Power 1992, Polis et al. 1997, Polis 1999, Sterner and Elser 2002). I suspect that this was in fact the case, however, increased primary production in alligator-engineered habitats entered the food web through the microbial loop via positive algal priming of decomposition (Kuehn et al. 2014, Halvorson et al. 2019). Further research is needed to measure changes in primary production across the alligator-engineered habitat gradient and test for algal priming.

I found direct support for the SGH based on trophic niches derived from stomach contents, but not those derived from stable isotopes. Relative to the wet season, trophic niche size based on stomach contents usually increased in ponds and decreased in both marshes and near-ponds. Under the NVH, this suggests that competition decreased in dry-season ponds, which is exactly what is predicted by the SGH (van Valen 1965, Bertness and Callaway 1994). According to these results from stomach contents, alligator-engineering of ponds ameliorated environmental stress caused by dry-season water recession and reduced competition as demonstrated by increased trophic niche areas. However, I found the opposite trend based on trophic niches derived from stable isotopes. I suspect that these conflicting results are the product of the relative importance of competition and consumption in structuring freshwater communities and animal movement among habitats in response to those pressures (Jackson et al. 2001, Alofs and

Jackson 2014, Werner et al. 1983) combined with the varying time periods reflected by stomach contents (24 – 48 hr) and stable isotopes (several weeks) (Nielsen et al. 2018). Perhaps, over short periods, such as those reflected by stomach contents, foraging behavior, and therein trophic niche size, are more influenced by competition and reflect the hypothesized trends. In contrast, over longer periods there is likely more variability in space use over this alligator-engineered habitat gradient in response to predation risk (Werner et al. 1983, Heithaus and Dill 2002).

Meta-analysis has demonstrated a trade-off between the effects of competition and consumption where competition has a greater effect on growth and consumption has a greater effect on survival (Gurevitch et al. 2000). Aquatic consumers may differentially use available habitats to optimize this trade-off, particularly in the dry season when fishes, especially larger predators, become concentrated and then restricted to ponds, increasing predation pressure (Loftus and Kushlan 1987, Parkos et al. 2011). As a result, consumers, prior to being restricted to ponds towards the end of the dry season, may spend less time in any single habitat and forage more across habitats that have distinct floral and faunal communities and nutrient profiles (Strickland 2020), which would result in larger trophic niche areas like I observed. Malikson and Tielborger (2010) note that when testing the SGH the net outcome of facilitative (alligator-engineering) and competitive interactions is not the additive effects of these processes, but the change in individual fitness. Regardless of changes in competition, or predation pressure, alligator-engineered ponds provide the only available habitat for many species in the dry season which facilitates an increase in individual fitness when the alternative is desiccation.

I demonstrated that alligators, which act as both top predators and ecosystem engineers, drove spatial trophic dynamics across a temporal environmental stress gradient in the form of shifts in diets, trophic niches, and higher trophic positions in deeper habitats. Increased nutrient levels in alligator-engineered habitats did not result in bottom-up changes to autotrophic versus heterotrophic energy use; however, there may be bottom-up effects increasing primary productivity and more autotrophic energy entering the food web in alligator-modified habitats through algal priming where increased algal production stimulates growth of saprotrophs increasing the decomposition rate (Kuehn et al. 2014, Halvorson et al. 2019). I found support for the SGH as defined by Bertness and Callaway (1994) using trophic niches derived from stomach contents that indicated decreased competition in the dry season. Trophic niches based on stable isotopes that integrate diets over a longer time period suggested more complex dynamics governing species interactions. However, the existence of alligator-engineered ponds ameliorates dry-season environmental stress by providing the only habitat for many species as water recedes inherently increasing fitness and supporting the SGH (Malkinson and Tielbörger 2010). This alligator-engineered habitat gradient and seasonal dynamic creates an opportunity for testing a variety of fundamental ecological questions about the effects of environmental stress, nutrients, competition, consumption, and animal movement on structuring communities and food webs. Future research should examine bottom-up effects mediated by algal priming, top-down effects (consumption and creating a landscape of fear) from alligators and other predators that occupy ponds, differences among ponds based on their physical characteristics (i.e., depth, area, canopy cover, etc.), and additional hypotheses such as biotic resistance and refuge partitioning among others.

Acknowledgements

I would like to thank Bradley Strickland for his help designing and conducting this research and the technicians and volunteers who assisted with field and laboratory work Sarah Sisco, Maria Sabando, Ariana Jonas, and Venus Garcia. This project was funded by the U. S. National Park Service Critical Ecosystem Studies Initiative (CESI) under Cooperative Agreement P18AC01074 with Florida International University. This material is based upon work supported by the National Science Foundation through the Florida Coastal Everglades Long-Term Ecological Research program under Cooperative Agreement No. DEB-1832229 and DEB-2025954 through the FIU CREST Center for Aquatic Chemistry and Environment under Grant No. HRD-1547798 and HRD-2111661. Any opinions, findings, conclusions, or recommendations expressed in the material are those of the authors and do not necessarily reflect the views of the National Science Foundation. PJF and BAS were supported by the Everglades Foundation as FIU ForEverglades Scholars. Research and animal procedures were conducted under the auspices of protocol #IACUC-18-067-CR01 from the Institutional Animal Care and Use Committee of the University of Florida and in accordance with scientific research permit #EVER-2017-SCI-0060 authorized by Everglades National Park and scientific collector's permits #S-18-06 and #S-19-05 granted by Florida Fish and Wildlife Conservation Commission.

Literature Cited

Adams, S. M., B. L. Kimmel, and G. R. Ploskey. 1983. Sources of organic matter for reservoir fish production: a trophic-dynamics analysis. Canadian Journal of Fisheries and Aquatic Sciences 40:1480–1495.

Bertness, M. D., and R. Callaway. 1994. Positive interactions in communities. Trends in Ecology and Evolution 9:191–193.

Bolnick, D. I., T. Ingram, W. E. Stutz, L. K. Snowberg, O. L. Lau, and J. S. Pauli. 2010. Ecological release from interspecific competition leads to decoupled changes in population and individual niche width. Pages 1789–1797 Proceedings of the Royal Society B: Biological Sciences. Royal Society.

Bolnick, D. I., R. Svanbä, S. Araú Jo §, and L. Persson. 2007. Comparative support for the niche variation hypothesis that more generalized populations also are more heterogeneous.

Brandt, L. A., M. R. Campbell, and F. J. Mazzotti. 2010. Spatial Distribution of Alligator Holes in the Central Everglades. Southeastern Naturalist 9:487–496.

Brauns, M., D. Kneis, M. Brabender, and M. Weitere. 2022. Habitat availability determines food chain length and interaction strength in food webs of a large lowland river. River Research and Applications 38:323–333.

Campbell, M. R., and F. J. Mazzotti. 2004. Characterization of natural and artificial alligator holes. Southeastern Naturalist 3:583–594.

Crain, C. M., and M. D. Bertness. 2006. Ecosystem engineering across environmental gradients: implications for conservation and management. BioScience 56:211–218.

Dangles, O., M. Herrera, C. Carpio, and C. J. Lortie. 2018. Facilitation costs and benefits function simultaneously on stress gradients for animals. Proceedings of the Royal Society B: Biological Sciences 285.

Gaiser, E.E., Trexler, J.C., Wetzel, P.R., 2012. The Florida Everglades, in: Batzer, D.P., Baldwin, A.H. (Eds.), The Florida Everglades. University of California Press, Berkeley, CA. 231–252.

García-Navas, V., T. Sattler, H. Schmid, and A. Ozgul. 2021. Bird species co-occurrence patterns in an alpine environment supports the stress-gradient hypothesis. Oikos 130:1905–1918.

Gunderson, L. H., and W. F. Loftus. 1993. The Everglades. Pages 199–255 *in* W. H. Martin, S. G. Boyce, and E. G. Echternacht, editors. Biodiversity of the Southeastern United States 2. John Wiley and Sons Inc., New York, New York, USA.

Gurevitch, J., J. A. Morrison, and L. v Hedges. 2000. The Interaction between Competition and Predation: A Meta-analysis of Field Experiments. Page Am. Nat.

Halvorson, H. M., S. N. Francoeur, R. H. Findlay, and K. A. Kuehn. 2019, February 26. Algal-mediated priming effects on the ecological stoichiometry of leaf litter decomposition: A meta-analysis. Frontiers Media S.A.

Hervé, M. 2021. RVAidMemoire: Testing and Plotting Procedures for Biostatistics.

Jackson, A. L., R. Inger, A. C. Parnell, and S. Bearhop. 2011. Comparing isotopic niche widths among and within communities: SIBER - Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology 80:595–602.

Jost, L., A. Chao, and R. L. Chazdon. 2011. Compositional similarity and beta diversity. Pages 66–84 *in* A. E. Magurran and B. J. McGill, editors. Biological Diversity. Frontiers in Measurement and Assessment. Oxford University Press (OUP), Oxford, UK.

Kotun, K., and A. Renshaw. 2014. Taylor slough hydrology fifty years of water management 1961 - 2010. Wetlands 34.

Kuehn, K. A., S. N. Francoeur, R. H. Findlay, and R. K. Neely. 2014. Priming in the microbial landscape: periphytic algal stimulation of litter-associated microbial decomposers. Page Ecology.

Kushlan, J. A. 1974. Observations on the role of the American Alligator in the Southern Florida Wetlands. Copeia 4:993–996.

Loftus, W. F., and J. A. Kushlan. 1987. Freshwater fishes of southern Florida. Bulletin of the Florida State Museum, Biological Sciences 31:147–344.

Lowney, A. M., and R. L. Thomson. 2021. Ecological engineering across a temporal gradient: Sociable weaver colonies create year-round animal biodiversity hotspots. Journal of Animal Ecology 90:2362–2376.

Lowney, A. M., and R. L. Thomson. 2022. Ecological engineering across a spatial gradient: Sociable weaver colonies facilitate animal associations with increasing environmental harshness. Journal of Animal Ecology 91:1385–1399.

Malkinson, D., and K. Tielbörger. 2010. What does the stress-gradient hypothesis predict? Resolving the discrepancies. Oikos 119:1546–1552.

Mangiafico, S. 2021. rcompanion: Functions to Support Extension Education Program Evaluation.

McHugh, P. A., R. M. Thompson, H. S. Greig, H. J. Warburton, and A. R. Mcintosh. 2015. Habitat size influences food web structure in drying streams. Ecography 38:700–712.

Nielsen, J. M., E. L. Clare, B. Hayden, M. T. Brett, and P. Kratina. 2018, February 1. Diet tracing in ecology: Method comparison and selection. British Ecological Society.

Oksanen, J., F. Gullaume Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. M. Stevens, E. Szoecs, and H. Wagner. 2022. vegan: Community Ecology Package.

Palmer, M. L., and F. J. Mazzotti. 2004. Structure of Everglades alligator holes. Wetlands 24:115–122.

Parkos, J. J., C. R. Ruetz, and J. C. Trexler. 2011. Disturbance regime and limits on benefits of refuge use for fishes in a fluctuating hydroscape. Oikos 120:1519–1530.

Piet, G. J., J. S. Pet, W. A. H. P. Guruge, J. Vijverberg, and W. L. T. van Densen. 1999. Resource partitioning along three niche dimensions in a size-structured tropical fish assemblage. Canadian Journal of Fisheries and Aquatic Sciences 56:1241–1254.

Polis, G. A. 1999. Why Are Parts of the World Green? Multiple Factors Control Productivity and the Distribution of Biomass. Oikos 86:3–15.

Polis, G. A., W. B. Anderson, and R. D. Holt. 1997. Toward an Integration of Landscape and Food Web Ecology: The Dynamics of Spatially Subsidized Food Webs. Annual Review of Ecology and Systematics 28:289–316.

Power, M. E. 1992. Top-Down and Bottom-Up Forces in Food Webs: Do Plants Have Primacy. Ecology 73:733–746.

Quezada-Romegialli, C., A. L. Jackson, B. Hayden, K. K. Kahilainen, C. Lopes, and C. Harrod. 2018. tRophicPosition, an r package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods in Ecology and Evolution 9:1592–1599.

R Core Team. 2022. R: A Language and Enivronment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rosenblatt, A. E., J. C. Nifong, M. R. Heithaus, F. J. Mazzotti, M. S. Cherkiss, B. M. Jeffery, R. M. Elsey, R. A. Decker, B. R. Silliman, L. J. Guillette, R. H. Lowers, and J. C. Larson. 2015. Factors affecting individual foraging specialization and temporal diet stability across the range of a large "generalist" apex predator. Oecologia 178:5–16.

Sanders, D., C. G. Jones, E. Thébault, T. J. Bouma, T. van der Heide, J. van Belzen, and S. Barot. 2014. Integrating ecosystem engineering and food webs. Oikos 123:513–524.

Sanders, D., and F. J. F. van Veen. 2011. Ecosystem engineering and predation: The multi-trophic impact of two ant species. Journal of Animal Ecology 80:569–576.

Sterner, R. W., and J. J. Elser. 2002. Ecological stoichiometry: the biology of elements from molecules to the biosphere. Page (R. W. Sterner and J. J. Elser, Eds.). Princeton University Press, Princeton, NJ.

Strickland, B. A. 2020. Beyond Predation: How Do Consumers Mediate Bottom-up Processes in Ecosystems? Florida International University, Miami, FL.

Takimoto, G., and D. M. Post. 2013. Environmental determinants of food-chain length: A meta-analysis. Ecological Research 28:675–681.

Turner, A. M., J. C. Trexler, Frank Jordan C, S. J. Slack, P. Geddes, J. H. Chick, and W. F. Loftus. 1999. Targeting Ecosystem Features for Conservation: Standing Crops in the Florida Everglades. Conservation Biology 13:898–911.

van Valen, L. 1965. Morphological Variation and Width of Ecological Niche. Source: The American Naturalist 99:377–390.

Wainwright, P. C., and B. A. Richard. 1995. Predicting patterns of prey use from morphology of fishes. Environmental Biology of Fishes 44:97–113.

Wetzel, W. C., R. M. Screen, I. Li, J. Mckenzie, K. A. Phillips, M. Cruz, W. Zhang, A. Greene, E. Lee, N. Singh, C. Tran, and L. H. Yang. 2016. Ecosystem engineering by a gall-forming wasp indirectly suppresses diversity and density of herbivores on oak trees. Page Ecology.

Wheeler, B., and M. Torchiano. 2016. lmPerm: Permutation Tests for Linear Models.

van der Zee, E. M., C. Angelini, L. L. Govers, M. J. A. Christianen, A. H. Altieri, K. J. van der Reijden, B. R. Silliman, J. van de Koppel, M. van der Geest, J. A. van Gils, H. W. van der Veer, T. Piersma, P. C. de Ruiter, H. Olff, and T. van der Heide. 2016. How habitat-modifying organisms structure the food web of two coastal ecosystems. Proceedings of the Royal Society B: Biological Sciences 283.

Zhong, Z., X. Li, D. Pearson, D. Wang, D. Sanders, Y. Zhu, and L. Wang. 2017. Ecosystem engineering strengthens bottom-up and weakens top-down effects via trait-mediated indirect interactions. Proceedings of the Royal Society B: Biological Sciences 284.

Table 1. Frequency (percent of comparisons that underwent a shift) and magnitude (percent change) of trophic dispersion based on stomach contents and stable isotopes for different spatiotemporal comparisons. Mean percent change was calculated based on the absolute value for each comparison.

	Stomach Contents		Stable Isotopes	
Comparison	Frequency	Magnitude	Frequency	Magnitude
Habitat	37%	1039% ± 2086%	42%	836% ± 2099%
Marsh - Near-Pond	33%	108% ± 67%	37%	1074% ± 2722%
Marsh - Pond	32%	888% ± 2035%	41%	894% ± 2222%
Near-Pond - Pond	47%	1809% ± 2651%	52%	482% ± 739%
Season	47%	813% ± 1387%	43%	389% ± 966%
Marsh	47%	415% ± 532%	47%	267% ± 568%
Near-Pond	75%	3230% ± 2139%	42%	70% ± 14%
Pond	42%	256% ± 325%	39%	671% ± 1436%
Slough	40%	144% ± 118%	43%	1033% ± 2474%

Table 2. Permuted ANOVA of trophic position based on stomach contents for spatiotemporal comparisons per species size class that underwent shifts in trophic position. I included two “marginal” shifts where p = 0.055.

	Group A			Group B				
Comparison	Size Class	Habitat	Season	Size Class	Habitat	Season	Stat	p
Habitat	G.hol1	Marsh	Dry	G.hol1	Pond	Dry	-2.28	0.02
	H.for1	NP	Wet	H.for1	Pond	Wet	-1.97	0.05
	G.hol1	Marsh	Wet	G.hol1	Pond	Wet	1.92	0.06
Season	G.hol1	Marsh	Dry	G.hol1	Marsh	Wet	-2.51	0.01
	L.goo1	Pond	Dry	L.goo1	Pond	Wet	-1.92	0.06

Table 3. Frequency (percent of comparisons that underwent a shift) and magnitude (percent change) of trophic position shifts based on stomach contents and stable isotopes for different spatiotemporal comparisons. Mean percent change was calculated based on the absolute value for each comparison.

	Stable Isotopes	
Comparison	Frequency	Magnitude
Habitat	18%	0.55 ± 0.21
Marsh - Near-Pond	23%	0.62 ± 0.22
Marsh - Pond	15%	0.52 ± 0.20
Near-Pond - Pond	16%	0.42 ± 0.15
Season	21%	0.59 ± 0.62
Marsh	11%	1.13 ± 1.46
Near-Pond	36%	0.38 ± 0.12
Pond	26%	0.47 ± 0.15
Slough	28%	0.67 ± 0.62

Figure 1. Map of sites in Everglades National Park, FL, USA – five in each major drainage in the ecosystem: Shark River Slough and Taylor Slough.

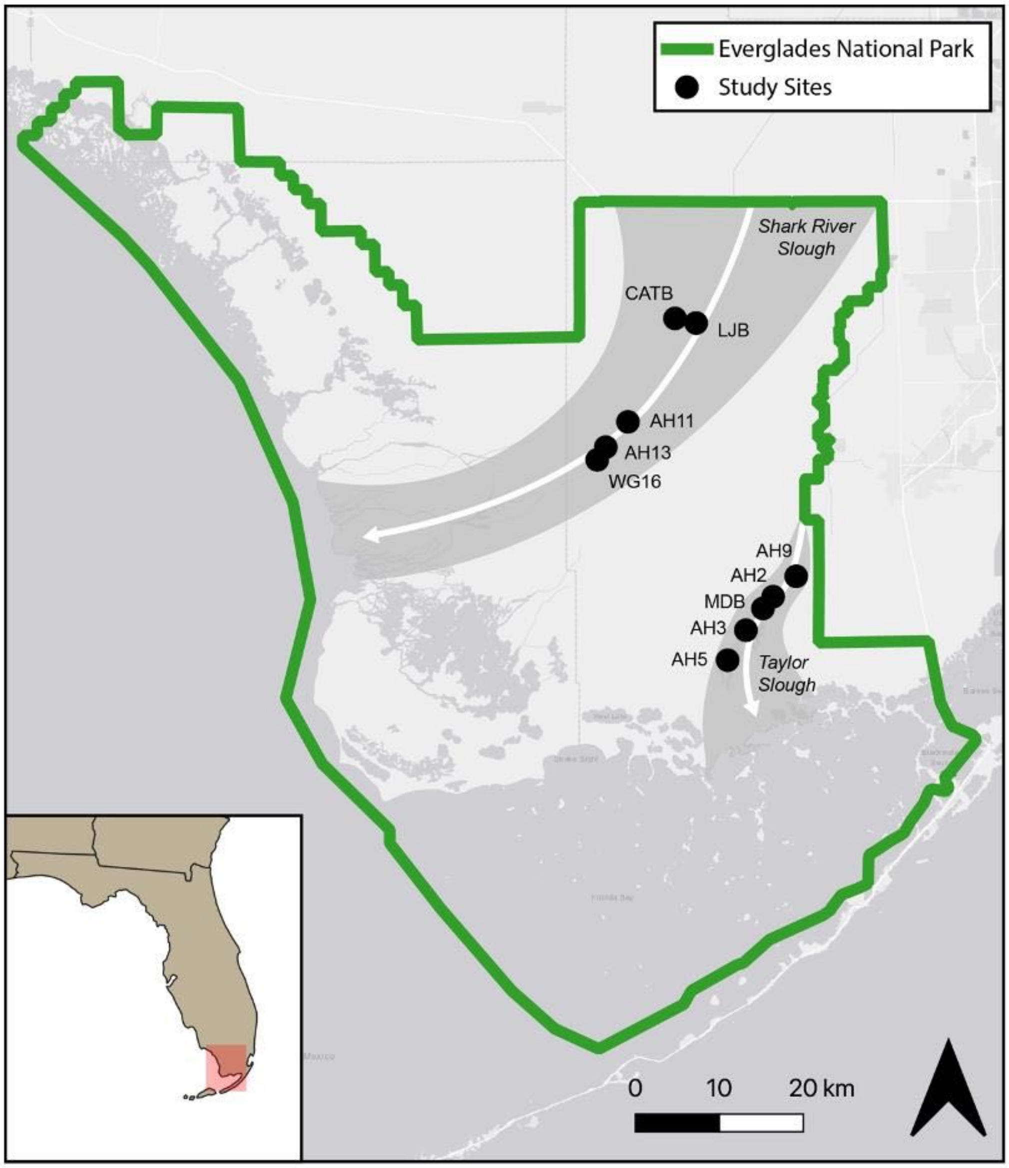


Figure 2. The average number prey per category found in consumer stomachs from Shark River Slough (SRS) and Taylor Slough (TSL) in both seasons across the alligator-engineered habitat gradient.

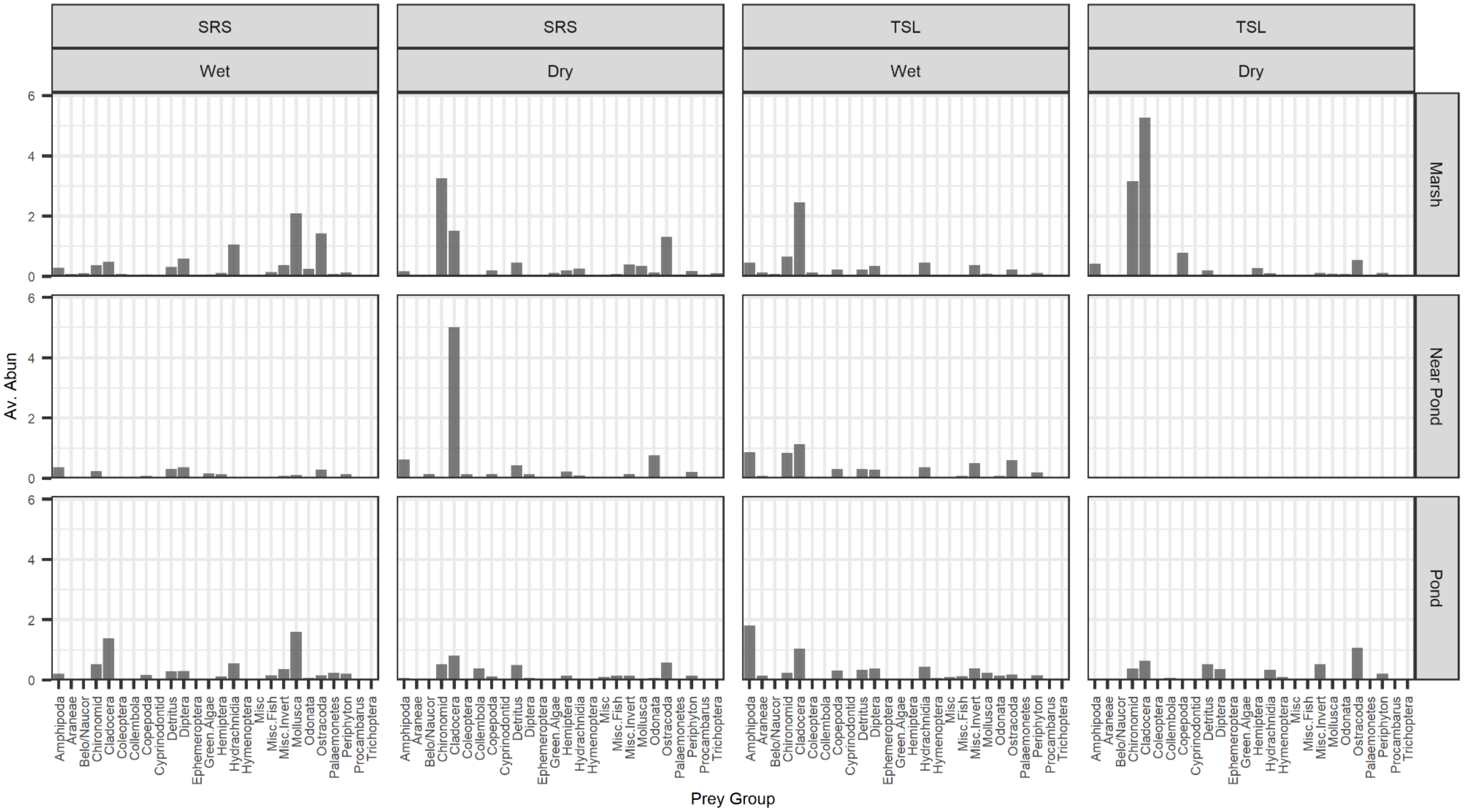


Figure 3. Trophic dispersion (Bayesian standard ellipse areas – SEA_b) modeled using counts of stomach contents across sloughs, seasons, and the alligator-engineered habitat gradient.

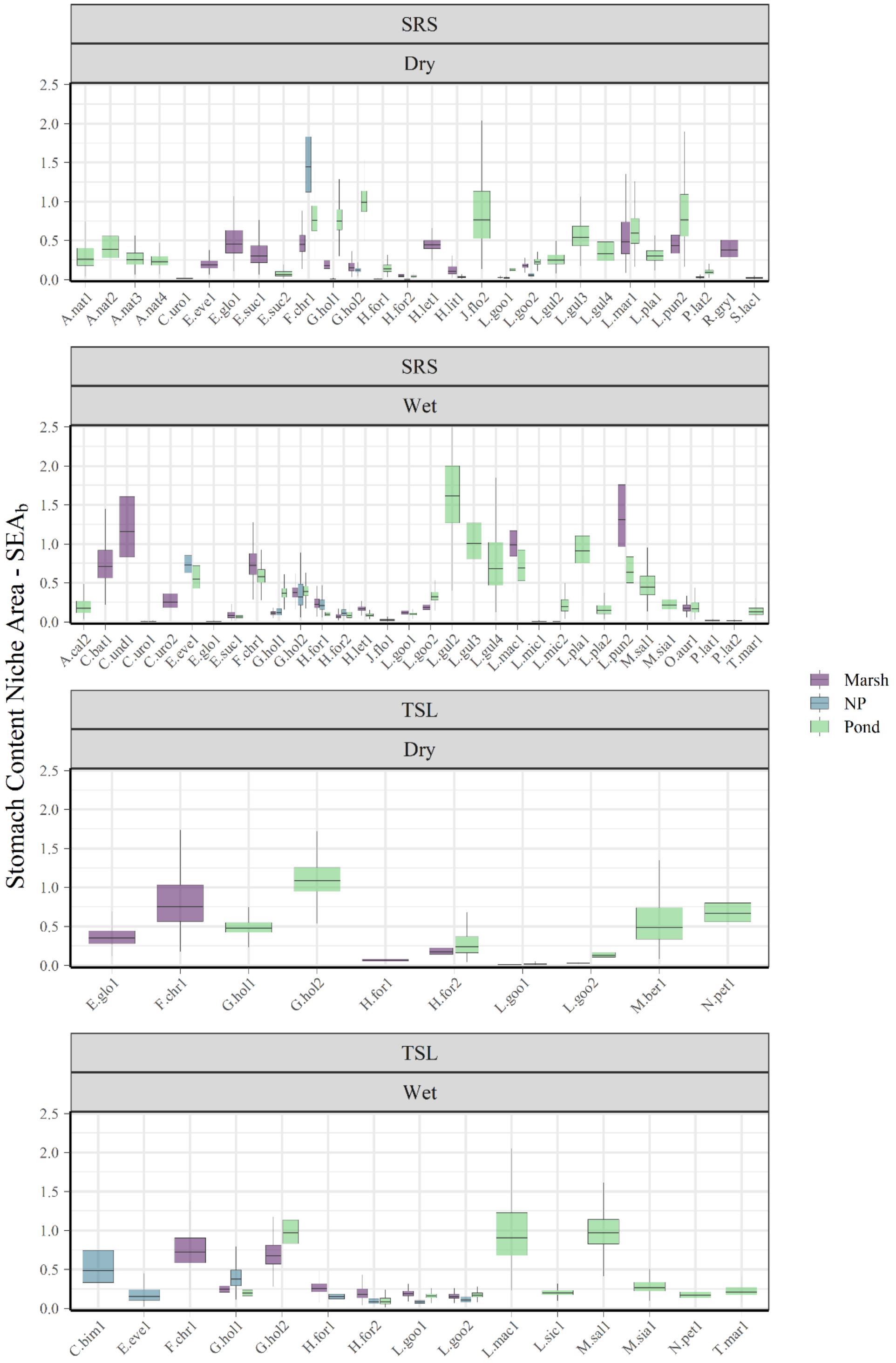
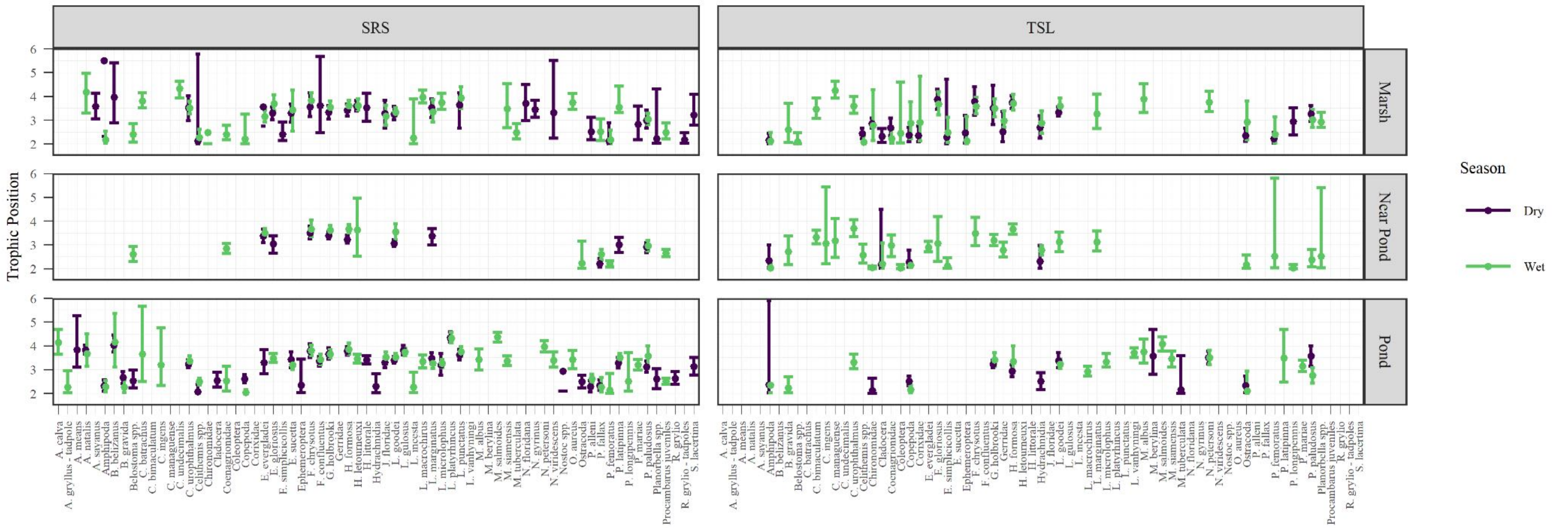


Figure 4. Trophic dispersion (Bayesian standard ellipse areas – SEA_b) modeled using stable isotopes of carbon and nitrogen across sloughs, seasons, and the alligator-engineered habitat gradient.

Figure 5. Trophic positions modeled using stable isotopes of carbon and nitrogen among sloughs, seasons, and the alligator-engineered habitat gradient.



Appendix

Methods

Field Collections

Basal resources in the form of vascular plants, *Utricularia* spp., *Nostoc* spp., periphyton, and flocculent organic benthic matter (hereafter “floc”) were collected when present at each habitat, in each season, and at each site. Floc samples were taken from three different locations within each habitat and aggregated into a single sample. For vascular plants, samples were taken from leaves and mid-stems of multiple individuals of dominant taxa to create an aggregate sample per habitat, per season, per site. Similarly, *Utricularia* spp., *Nostoc* spp., and periphyton (both mats and epiphyton), when present, were taken from multiple locations within a habitat to create an aggregate sample.

Consumers were collected through a variety of methods to maximize number of taxa and individuals per taxon. Brakke’s (1976) modified Whiteside-Williams (1975) samplers (hereafter “funnel traps”) were deployed across the study design (when permitted by water levels) to collect small, benthic invertebrates (e.g., copepods, ostracods, cladocerans, etc.). Each funnel trap consisted of a 4 x 4 array of funnels for a total of sixteen per trap that resulted in 1 L of water being collected per funnel trap. Funnel traps were placed on top of the sediment for 24-hours to capture invertebrates during diel migration. These samples were stored on ice for transport and then refrigerated until processing. Organisms were sorted under a dissecting microscope and identified to taxonomic level (i.e., order, class, family).

Three, one-m^2 throw-trap samples were performed per habitat (pond and marsh) at each site (six total throws per site). When water depth was < 5 cm or > 1 m, we did not sample because throw traps are ineffective at these depths (Jordan et al. 1997). This led to twenty-two throw-trap samples in the wet season and twenty-seven in the dry season for forty-nine total (Table S1). Additionally, we used minnow traps, dip nets, drift fences, and electrofishing to collect consumers. Consumers were euthanized via a lethal dose of MS-222 and frozen. In the laboratory, consumers were identified to species, counted, measured, weighed, and sexed (Table S2). Vertebrates were sorted into size classes based on length (Table S3).

Laboratory Analyses

We processed samples for both stable isotopes and stomach contents. For vertebrates, gastrointestinal tracts were removed and stored in 95% ethanol for further analysis. The remainder for vertebrates and the entire sample for invertebrates and basal resources was freeze-dried, crushed, and weighed in tin capsules for isotopic analysis. For larger consumers (e.g., Florida Gar – *Lepisosteus platyrhincus*) we physically removed bone and larger pieces after grinding in a ball mill. Periphyton samples were acid-washed in HCl before drying to remove carbonates. Basal resources and small invertebrate taxa that lacked enough mass to be processed as individuals were made into aggregate samples. All other isotope samples represent individual organisms. Most samples were analyzed at Florida International University Stable Isotope Lab, while samples from small invertebrates were sent to Duke University Environmental Stable Isotope Lab (DEVIL). DEVIL used international isotopic standards USGS 26 and USGS 40 in addition to internal standards of Costech acetanilide and Duke sucrose. External precision relative to reference materials was approximately ± 0.1 ‰ at one standard deviation. FIU isotopic standards for δ^{13}C and δ^{15}N were Pee Dee Belemnite and atmospheric air respectively. Average isotopic error of replicate standards was ± 0.4 for $\delta^{13}C$ and ± 0.3 for $\delta^{15}N$ at FIU.

Stomach contents of vertebrates were analyzed by flushing contents, identifying them under a microscope to the lowest possible taxonomic level (Table S2, S3). For species without a defined stomach, we removed contents from the esophagus to the first bend in the gastrointestinal tract. Fragmented and unidentifiable remains were termed "Miscellaneous Fish", "Miscellaneous Invertebrate", or "Miscellaneous" depending on quality and type of remains. Stomach contents were quantified using both volumetric and numeric approaches (Hyslop 1980, Bowen 1983). Statistical analyses were performed on numeric data to match analyses performed on pre-invasion data (Chapter 2).

Statistical Analyses – Bayesian Model Specifications

SIBER models of trophic niches were run using 2,000,000 iterations, burn-in of 10,000, thin of 100, and 2 chains (Jackson et al. 2011). Meanwhile, *tRophicPosition* models of trophic positions were run using 10,000 iterations, burn-in of 1,000, thin of 10, and 2 chains (Quezada-Romegialli et al. 2018). Both sets of models used null (uninformative) priors recommended by package authors.

Appendix Literature Cited

Bowen, S. H. 1983. Quantitative description of the diet. Pages 325–336 *in* L. A. Nielsen and D. L. Johnson, editors. Fisheries Techniques. American Fisheries Society, Bethesda, MD.

Brakke, D. F. 1976. Modification of the Whiteside–Williams Pattern Sampler. Journal of the Fisheries Research Board of Canada 33:2861–2863.

Hyslop, E. J. 1980. Stomach contents analysis-a review of methods and their application. Page J. Fish Biol.

Jackson, A. L., R. Inger, A. C. Parnell, and S. Bearhop. 2011. Comparing isotopic niche widths among and within communities: SIBER - Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology 80:595–602.

Jordan, F., S. Coyne, and J. C. Trexler. 1997. Sampling Fishes in Vegetated Habitats: Effects of Habitat Structure on Sampling Characteristics of the 1-m 2 Throw Trap.Transactions of the American Fisheries Society 126:1012–1020.

Quezada-Romegialli, C., A. L. Jackson, B. Hayden, K. K. Kahilainen, C. Lopes, and C. Harrod. 2018. tRophicPosition, an r package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods in Ecology and Evolution 9:1592–1599.

Whiteside, M. C., and J. B. Williams. 1975. A new sampling technique for aquatic ecologists. SIL Proceedings, 1922-2010 19:1534–1539.

Table S1. Throw trapping effort at each site in both seasons.

Slough	Site	Habitat	Wet	Dry	Total
SRS	LJB	P	2	3	5
		NP	2	0	2
		M	2	3	5
	CPB	P	0	0	0
		NP	3	3	6
		M	3	3	6
	AH11	P	0	3	3
		NP	0	0	0
		M	3	3	6
	AH13	P	3	3	6
		NP	3	3	6
		M	3	3	6
	WG16	P	3	3	6
		NP	3	0	3
		M	3	3	6
TSL	AH9	P	3	3	6
		NP	3	0	3
		M	3	0	3
	AH2	P	3	3	6
		NP	3	0	3
		M	3	3	6
	AHMDB	P	3	0	3
		NP	3	0	3
		M	3	0	3
	AH3	P	3	0	3
		NP	3	0	3
		M	3	0	3
	AH5	P	3	3	6
		NP	3	0	3
		M	3	3	6
		Total	78	48	126

Table S2. Sample sizes for each species size class in each habitat (M – marsh, N – near-pond, P – pond) in both sloughs and both seasons for A) stables isotopes and B) stomach contents.

A. Stable Isotopes			Shark River Slough						Taylor Slough					
Species	Abbr.	Size Class	Wet			Dry			Wet			Dry		
			M	N	P	M	N	P	M	N	P	M	N	P
Amia calva	A.cal	2	1	-	4	-	-	-	-	-	-	-	-	-
Acris gryllus	A.gry	1	-	-	4	-	1	1	-	-	-	-	-	-
Amphiuma means	A.mea	1	-	-	-	1	-	3	-	-	-	-	-	-
Ameiurus natalis	A.nat	1	-	-	-	2	-	5	-	-	-	-	-	-
		2	1	-	-	-	-	5	-	-	-	-	-	-
		3	-	-	3	-	-	6	-	-	-	-	-	-
		4	3	-	-	-	-	10	-	-	-	-	-	-
Anguilla rostrata	A.ros	1	-	-	-	-	-	-	-	-	1	-	-	-
Aphredoderus sayanus	A.say	1	1	1	-	9	-	-	-	-	-	-	-	-
Nostoc spp.	-	-	1	1	1	-	-	3	2	2	1	1	-	1
Amphipoda	-	-	20	2	13	8	-	12	73	24	3	24	8	3
Belonesox belizanus	B.bel	1	-	-	3	3	1	4	-	2	1	1	-	-
		2	-	-	-	-	1	-	1	-	-	-	-	-
Belostoma spp.	-	-	6	5	2	1	2	3	6	-	-	-	-	-
Hydrachnidia	-	-	2	2	2	-	-	13	8	12	2	10	9	18
Brachymesia gravida	-	-	2	-	10	-	-	11	5	4	4	1	-	1

Clarias batrachus	C.bat	1	12	-	3	-	-	1	1	-	1	-	-	-
Cichlasoma bimaculatum	C.bim	1	-	-	-	-	-	-	9	12	-	-	-	-
Parachromis managuensis	P.man	1	-	-	-	-	-	1	3	4	1	-	-	-
Cichla ocellatus	C.oce	1	2	-	-	-	-	-	-	-	2	-	-	-
Centropomis undecimalis	C.und	1	9	-	-	-	-	1	-	-	-	-	-	-
Mayaheros urophthalmus	M.uro	1	9	-	8	4	1	17	12	6	3	-	-	-
		2	7	-	3	-	-	2	1	-	10	-	-	-
Cyprinodon variegatus	C.var	1	1	-	-	-	-	-	-	-	2	-	-	-
Celithemis spp.	-	-	5	2	8	6	-	6	6	4	-	6	-	-
Chironomidae	-	-	3	1	1	-	-	2	4	4	-	7	-	3
Cladocera	-	-	1	-	1	-	-	6	-	5	-	16	4	2
Cladium jamaicense	-	-	-	-	3	-	-	2	2	-	-	-	-	-
Coenagrionidae	-	-	7	5	5	-	-	-	4	3	-	5	-	-
Coleoptera	-	-	1	-	-	-	-	-	4	6	-	-	-	-
Copepoda	-	-	4	2	17	-	-	23	5	10	5	8	3	8
Coryphaeschna ingens	-	-	-	1	3	-	-	-	-	3	-	-	-	-
Corixidae	-	-	1	1	1	-	-	-	3	1	-	4	-	-
Cybster spp.	-	-	-	2	-	-	-	1	-	-	-	-	-	-
Esox a. americanus	E.ame	1	-	-	-	1	-	-	-	-	-	-	-	-

Elassoma evergladei	E.eve	1	6	23	-	9	8	5	2	4	-	1	-	2
Enneacanthus gloriosus	E.glo	1	5	1	15	13	7	1	7	4	-	10	-	-
Erimyzon sucetta	E.suc	1	4	-	6	6	-	2	-	-	-	-	-	-
		2	-	-	7	-	-	4	-	-	-	-	-	-
Eleocharis cellulosa	-	-	-	-	3	1	1	1	1	1	2	-	-	-
Ephemeroptera	-	-	-	2	-	-	-	3	4	-	-	4	-	-
Epitheca stella	-	-	-	-	2	-	-	-	2	-	-	2	-	-
Erythemis simplicicollis	-	-	1	-	1	6	-	-	6	6	-	3	-	-
Fundulus chrysotus	F.chr	1	17	3	29	12	14	21	14	3	-	5	-	-
		2	11	2	8	10	3	2	-	1	-	-	-	-
Fundulus confluentus	F.con	1	1	-	9	3	1	17	-	-	-	-	-	-
Platyhelminthes	-	-	-	-	-	-	-	1	-	-	-	-	-	-
Flocculent Matter	-	-	4	4	12	3	4	11	5	7	5	6	-	5
Gambusia holbrooki	G.hol	1	21	5	57	10	10	22	24	7	14	1	-	35
		2	39	30	121	29	28	28	25	8	72	3	-	29
Gerridae	-	-	-	-	-	-	-	-	4	4	-	4	-	-
Heterandria formosa	H.for	1	9	8	10	1	4	11	14	12	-	13	-	-
		2	5	7	4	9	6	7	7	7	4	11	-	4
Hemichromis letourneuxi	H.let	1	43	3	23	47	3	2	-	-	-	-	-	-

Hoplisternum littorale	H.lit	1	1	-	-	5	-	13	-	-	-	-	-	-
Jordanella floridae	J.flo	1	6	-	2	-	-	3	-	-	-	-	-	-
		2	5	1	4	4	2	9	-	-	-	-	-	-
Lucania goodei	L.goo	1	24	-	57	19	9	27	24	9	27	24	-	4
		2	50	6	84	31	21	30	31	6	41	27	-	10
Lepomis gulosus	L.gul	1	-	-	1	-	-	3	-	-	-	-	-	-
		2	-	-	8	1	-	11	-	-	-	-	-	-
		3	1	-	21	-	-	14	-	-	1	-	-	-
		4	-	-	9	-	-	9	-	-	-	-	-	-
Lepomis macrochirus	L.mac	1	19	-	8	-	-	2	-	-	6	-	-	-
Lepomis marginatus	L.mar	1	6	-	12	7	6	13	4	5	1	-	-	-
Lepomis microlophus	L.mic	1	3	1	8	2	-	4	-	-	2	-	-	-
		2	5	-	7	-	-	-	-	-	1	-	-	-
Lepisosteus platyrhincus	L.pla	1	-	-	19	-	-	25	-	-	2	-	-	-
		2	-	-	3	-	-	-	-	-	-	-	-	-
Lepomis punctatus	L.pun	1	-	-	-	2	5	-	-	-	-	-	-	-
		2	5	-	8	9	-	6	-	-	-	-	-	-
		3	-	-	2	2	-	3	-	-	-	-	-	-
Labidesthes vanhyningi	L.van	1	-	-	-	-	-	-	-	-	28	-	-	-
Libellula incesta	-	-	3	-	4	1	-	1	-	-	-	-	-	-
Litroridinops monroensis	-	-	-	-	-	-	-	1	-	-	-	1	-	-
Monopterus albus	M.alb	1	-	-	-	-	-	-	2	-	-	-	-	-
		2	-	-	1	-	-	-	1	-	3	1	-	-

		3	1	-	2	-	-	-	2	-	1	-	-	-
		4	-	-	1	-	-	-	1	-	1	-	-	-
Menidia berylina	M.ber	1	-	-	-	-	-	-	-	-	-	-	-	4
Micropterus salmoides	M.sal	1	1	-	18	-	-	2	-	-	33	-	-	-
Macrognathus siamensis	M.sia	1	3	-	29	-	-	-	-	-	58	-	-	-
Melanoides tuberculata	-	-	4	-	-	-	-	-	-	1	-	-	-	3
Nerodia floridana	N.flo	1	-	-	-	4	-	1	-	-	-	-	-	-
Noturus gyrinus	N.gyr	1	-	-	-	3	-	-	-	-	-	-	-	-
Notropis petersoni	N.pet	1	2	1	5	-	-	-	5	-	16	-	-	23
Notophthalmus viridescens	N.vir	1	1	-	4	3	1	1	-	-	-	-	-	-
Oreochromis aureus	O.aur	1	21	-	6	2	-	-	-	-	2	-	-	-
Ostracoda	-	-	2	5	1	-	-	31	9	7	4	12	2	20
Pseudobranchus a. belli	P.bel	1	1	-	-	1	-	-	-	-	-	-	-	-
Poecilia latipinna	P.lat	1	1	-	6	-	1	-	-	-	-	-	-	-
		2	8	2	10	2	6	11	-	-	3	-	-	-
Pachydiplax longipennis	-	-	-	-	3	-	-	1	-	7	-	6	-	-
Palaemonetes paludosus	-	-	11	6	3	6	6	6	7	6	8	13	-	6

Panicum hemitomon	-	-	2	1	1	1	1	1	1	1	1	-	-	-
Pelocoris femoratus	-	-	6	6	5	17	-	-	6	3	-	17	-	-
Periphyton - Epiphytic	-	-	2	-	5	-	-	-	6	-	-	2	-	-
Periphyton - Mat	-	-	3	-	1	4	-	-	2	1	-	2	-	-
Planorbella	-	-	-	-	-	3	-	9	5	3	-	2	-	-
Pomacea maculata	-	-	-	-	-	-	-	1	-	-	-	-	-	-
Pomacea paludosa	-	-	-	-	1	-	1	2	-	-	-	-	-	-
Pontedaria cordata	-	-	-	2	1	-	-	-	-	3	-	-	-	-
Procambarus alleni	-	-	2	1	17	9	1	8	-	-	-	-	-	-
Procambarus fallax	-	-	5	6	4	-	8	17	-	-	-	-	-	-
Procambarus juveniles	-	-	4	6	3	1	-	1	-	-	-	-	-	-
Pseudosuccinea columella	-	-	-	2	-	-	-	-	-	2	-	-	-	-
Rana grylio	R.gry	1	-	-	2	9	2	2	-	-	-	-	-	-
		2	-	-	-	2	-	6	-	-	-	-	-	-
Oligochaeta	-	-	-	-	1	-	-	-	-	-	-	-	-	-
Sirens lacertina	S.lac	1	2	-	2	8	-	7	-	-	-	-	-	-
Sagitarria lancifolia	-	-	-	3	1	-	2	2	1	1	1	-	-	-
Sphaeridae	-	-	-	-	-	-	-	-	-	-	-	1	-	-

Pelmatolapia mariae	P.mar	1	-	-	5	5	-	-	1	1	14	-	-	-
Utricularia foliosa	-	-	1	-	2	1	-	2	3	1	-	-	-	-
Utricularia purpurea	-	-	2	-	1	1	-	2	2	-	1	-	-	-

B. Stomach Contents			Shark River Slough						Taylor Slough				
Species	Abbr.	Size Class	Wet			Dry			Wet			Dry	
			M	N	P	M	N	P	M	N	P	M	P
Amia calva	A.cal	2	1	-	4	-	-	-	-	-	-	-	-
Acris gryllus	A.gry	1	-	-	2	-	-	-	-	-	-	-	-
Ameiurus natalis	A.nat	1	-	-	-	2	-	3	-	-	-	-	-
		2	1	-	-	-	-	4	-	-	-	-	-
		3	-	-	3	-	-	6	-	-	-	-	-
		4	2	-	-	-	-	8	-	-	-	-	-
Anguilla rostrata	A.ros	1	-	-	-	-	-	-	-	-	1	-	-
Aphredoderus sayanus	A.say	1	-	-	-	1	-	-	-	-	-	-	-
Belonesox belizanus	B.bel	1	-	-	3	1	-	2	-	1	1	1	-
Clarias batrachus	C.bat	1	10	-	3	-	-	1	1	-	1	-	-
Cichlasoma bimaculatum	C.bim	1	-	-	-	-	-	-	-	3	-	-	-
Parachromis manguensis	P.man	1	-	-	-	-	-	1	-	1	1	-	-
Cichla ocellaris	C.oce	1	1	-	-	-	-	-	-	-	2	-	-

Centropomis undecimalis	C.und	1	3	-	-	-	-	1	-	-	-	-	-
Mayaheros urophthalmus	M.uro	1	16	-	6	4	-	-	1	2	2	-	-
		2	6	-	3	-	-	1	1	-	6	-	-
Cyprinodon variegatus	C.var	1	1	-	-	-	-	-	-	-	-	-	-
Esox a. americanus	E.ame	1	-	-	-	1	-	-	-	-	-	-	-
Elassoma evergladei	E.eve	1	4	24	13	10	-	1	1	3	-	1	2
Enneacanthus gloriosus	E.glo	1	1	-	3	5	-	-	-	2	-	9	-
Erimyzon sucetta	E.suc	1	3	-	5	4	-	2	-	-	-	-	-
		2	-	-	6	-	-	4	-	-	-	-	-
Fundulus chrysotus	F.chr	1	14	1	23	10	5	13	11	1	-	5	-
		2	-	-	1	1	-	2	-	1	-	-	-
Fundulus confluentus	F.con	1	-	-	-	2	-	1	-	-	-	-	-
Gambusia holbrooki	G.hol	1	19	4	18	7	9	16	17	7	13	1	27
		2	18	3	21	5	11	29	16	1	19	2	25
Heterandria formosa	H.for	1	8	8	9	1	4	8	14	11	-	13	-
		2	5	5	3	5	5	5	5	4	3	10	3
Hemichromis letourneuxi	H.let	1	25	-	15	35	-	-	-	-	-	-	-
Hoplosternun littorale	H.lit	1	1	-	-	5	-	11	-	-	-	-	-
Jordanella	J.flo	1	3	-	2	-	-	1	-	-	-	-	-

floridae		2	1	-	1	1	-	3	-	-	-	-	-
Lucania goodei	L.goo	1	19	-	20	18	7	25	18	9	20	25	4
		2	16	1	19	25	7	25	17	5	20	27	9
Lepomis gulosus	L.gul	1	-	-	1	-	-	3	-	-	-	-	-
		2	-	-	7	-	-	11	-	-	-	-	-
		3	1	-	19	-	-	11	-	-	1	-	-
		4	-	-	7	-	-	7	-	-	-	-	-
Lepomis macrochirus	L.mac	1	17	-	7	-	-	2	-	-	5	-	-
Lepomis marginatus	L.mar	1	-	-	3	3	-	9	-	-	1	-	-
Lepomis microlophus	L.mic	1	3	-	2	1	-	-	-	-	2	-	-
		2	4	-	5	-	-	-	-	-	1	-	-
Lepisosteus platyrhincus	L.pla	1	-	-	16	-	-	21	-	-	1	-	-
		2	-	-	4	-	-	-	-	-	-	-	-
Lepomis punctatus	L.pun	2	3	-	7	7	-	4	-	-	-	-	-
		3	-	-	2	1	-	3	-	-	-	-	-
Labidesthes vanhyningi	L.var	1	-	-	-	-	-	-	-	-	25	-	-
Monopterus albus	M.alb	1	-	-	-	-	-	-	1	-	-	-	-
		2	-	-	-	-	-	-	1	-	3	1	-
		3	1	-	1	-	-	-	2	-	1	-	-
		4	-	-	1	-	-	-	-	-	1	-	-
Menidia berylina	M.ber	1	-	-	-	-	-	-	-	-	-	-	3
Micropterus salmoides	M.sal	1	1	-	14	-	-	1	-	-	25	-	-
Macrognathus siamensis	M.sia	1	5	-	25	-	-	-	-	-	50	-	-

Nerodia floridana	N.flo	1	-	-	-	-	-	1	-	-	-	-	-
Noturus gyrinus	N.gyr	1	-	-	-	1	-	-	-	-	-	-	-
Notropis petersoni	N.pet	1	-	-	3	-	-	-	4	-	15	-	16
Notophthalmus viridescens	N.vir	1	-	-	-	1	-	-	-	-	-	-	-
Oreochromis aureus	O.aur	1	18	-	5	1	-	-	-	-	1	-	-
Pseudobranchus a. belli	P.bel	1	1	-	-	2	-	-	-	-	-	-	-
Poecilia latipinna	P.lat	1	-	-	5	-	1	-	-	-	-	-	-
		2	2	-	8	2	5	6	-	-	2	-	-
Rana grylio	R.gry	1	-	-	-	6	-	-	-	-	-	-	-
		2	-	-	-	-	-	1	-	-	-	-	-
Sirens lacertina	S.lac	1	1	-	1	3	-	2	-	-	-	-	-
Pelmatolapia mariae	P.mar	1	-	-	5	5	-	-	-	-	12	-	-

Table S3. Size class delimitations based on standard length (mm) for vertebrate taxa.

Species	Length (mm)	Size Class
Ameiurus natalis	≤ 96	1
	97 < x ≤ 135	2
	135 < x ≤ 175	3
	> 175	4
Amia calva	≤ 70	1
	> 70	2
Amphiuma means	-	1
Anguilla rostrata	-	1
Aphredoderus sayanus	-	1
Astronotus ocellatus	-	1
Belonesox belizanus	≤ 64	1
	> 64	2
Cichla ocellaris	-	1
Cichlasoma bimaculatum	-	1
Clarias batrachus	-	1
Cyprinodon variegatus	-	1
Elassoma evergladei	-	1
Enneacanthus gloriosus	-	1
Erimyzon sucetta	≤ 100	1
	100 < x ≤ 150	2
	> 150	3

Esox a. americanus	-	1
Fundulus confluentus	≤ 40	1
	> 40	2
Fundulus chrysotus	≤ 40	1
	> 40	2
Gambusia holbrooki	≤ 17	1
	> 17	2
Hemichromis letourneuxi	-	1
Heterandria formosa	≤ 12	1
	> 12	2
Hoplisternum littorale	-	1
Jordanella floridae	≤ 18	1
	> 18	2
Labidesthes vanhyningi	-	1
Lepisosteus platyrhincus	≤ 450	1
	> 450	2
Lepomis gulosus	≤ 71	1
	71 < x ≤ 95	2
	95 < x ≤ 120	3
	> 120	4
Lepomis macrochirus	-	1
Lepomis marginatus	-	1
Lepomis microlophus	≤ 100	1
	> 100	2
Lepomis punctatus	≤ 40	1
	40 < x ≤ 90	2
	> 90	3
Lucania goodei	≤ 17.5	1
	> 17.5	2
Macgronathus siamemis	-	1

Mayaheros urophthalmus	≤ 120	1
	> 120	2
Monopterus albus	≤ 275	1
	275 < x ≤ 450	2
	450 < x ≤ 625	3
	> 625	4
Micropterus salmoides	-	1
Notemigonus crysoleucas	-	1
Noturus gyrinus	-	1
Notropis maculatus	-	1
Notropis petersoni	-	1
Parachromis managuense	-	1
Pelmatolapia mariae	-	1
Poeilia latipinna	≤ 20	1
	> 20	2

Table S4. PERMANOVA comparing prey in consumer’s guts among size classes, seasons, sloughs, habitats, and the subsequent interactions. There was a significant interaction of size class, season, slough, and habitat which suggests responses to spatiotemporal variation differ by species size class.

Variables	Df	SS	Mean SS	F	R^2	p
Size Class	64	175.25	2.74	10.62	0.30	0.001
Season	1	5.55	5.55	21.54	0.01	0.001
Slough	1	2.41	2.41	9.34	0.00	0.001
Habitat	2	4.26	2.13	8.27	0.01	0.001
Size Class*Season	41	27.55	0.67	2.61	0.05	0.001
Size Class*Slough	23	15.58	0.68	2.63	0.03	0.001
Season*Slough	1	0.51	0.51	1.99	0.00	0.050
Size Class*Habitat	43	18.99	0.44	1.71	0.03	0.001
Season*Habitat	2	5.72	2.86	11.10	0.01	0.001
Slough*Habitat	2	1.30	0.65	2.53	0.00	0.001
Size Class*Season*Slough	8	2.90	0.36	1.41	0.00	0.015
Size Class*Season*Habitat	18	6.95	0.39	1.50	0.01	0.001
Size Class*Slough*Habitat	13	4.42	0.34	1.32	0.01	0.013
Season*Slough*Habitat	1	0.42	0.42	1.61	0.00	0.105
Size Class*Season*Slough*Habitat	4	1.80	0.45	1.75	0.00	0.005
Residuals	1209	311.65	0.26		0.53	
Total	1433	585.27			1.00	

Table S5. Trophic dispersion based on trophic niches derived from stomach contents compared across A) habitats, B) seasons, and C) sloughs. Percent Change is calculated as the percent change from the first group in the comparison relative to the second group (e.g., for Marsh < NP the marsh value is considered the initial value). Probabilities in the column comparing the two group (e.g., Marsh < NP) are the probability that the posterior distribution of the first group is less than that of the second group. Therefore, values ≥ 0.95 we considered an increase in trophic dispersion, while values ≤ 0.05 represent the opposite one-tailed hypothesis (i.e., p(A > B) ≥ 0.95) and we considered a decrease in trophic dispersion.

A. Habitat Comparisons

Slough	Season	Size Class	Marsh	NP	Marsh < NP	Percent Change
SRS	Wet	G.hol1	0.1	0.08	0.55	-18
SRS	Wet	G.hol2	0.35	0.21	0.4	-40
SRS	Wet	H.for1	0.19	0.17	0.44	-14
SRS	Wet	H.for2	0.05	0.09	0.76	60
SRS	Dry	F.chr1	0.39	1.26	0.99	225
SRS	Dry	G.hol1	0.14	0.01	0	-94
SRS	Dry	G.hol2	0.12	0.1	0.32	-9
SRS	Dry	H.for2	0.04	0	0	-96
SRS	Dry	L.goo1	0.03	0.02	0.22	-33
SRS	Dry	L.goo2	0.17	0.05	0.01	-69
TS	Wet	G.hol1	0.23	0.32	0.83	39
TS	Wet	H.for1	0.22	0.13	0.1	-39
TS	Wet	H.for2	0.14	0.07	0.14	-54
TS	Wet	L.goo1	0.17	0.07	0.03	-58
TS	Wet	L.goo2	0.14	0.09	0.26	-39

Slough	Season	Size Class	Marsh	Pond	Marsh < Pond	Percent Change
SRS	Wet	C.uro1	0.01	0.01	0.46	-16
SRS	Wet	E.suc1	0.05	0.05	0.36	-13
SRS	Wet	F.chr1	0.66	0.55	0.26	-16
SRS	Wet	G.hol1	0.1	0.34	1	230
SRS	Wet	G.hol2	0.35	0.37	0.56	6
SRS	Wet	H.for1	0.19	0.09	0.04	-55
SRS	Wet	H.for2	0.05	0.05	0.56	-10
SRS	Wet	H.let1	0.16	0.08	0.03	-52
SRS	Wet	L.goo1	0.11	0.09	0.29	-17
SRS	Wet	L.goo2	0.17	0.3	0.95	77

SRS	Wet	L.mac1	0.91	0.56	0.24	-39
SRS	Wet	L.mic2	0	0.15	1	6650
SRS	Wet	L.pun2	1.06	0.55	0.1	-49
SRS	Wet	O.aur1	0.15	0.12	0.49	-20
SRS	Dry	F.chr1	0.39	0.69	0.87	78
SRS	Dry	G.hol1	0.14	0.7	1	389
SRS	Dry	G.hol2	0.12	0.95	1	720
SRS	Dry	H.for2	0.04	0.03	0.37	-22
SRS	Dry	H.lit1	0.08	0.03	0.04	-65
SRS	Dry	L.goo1	0.03	0.12	1	359
SRS	Dry	L.goo2	0.17	0.21	0.8	27
SRS	Dry	L.mar1	0.3	0.5	0.61	63
SRS	Dry	L.pun2	0.36	0.55	0.82	53
TS	Wet	G.hol1	0.23	0.18	0.27	-23
TS	Wet	G.hol2	0.62	0.89	0.85	43
TS	Wet	H.for2	0.14	0.06	0.17	-58
TS	Wet	L.goo1	0.17	0.15	0.31	-15
TS	Wet	L.goo2	0.14	0.16	0.64	15
TS	Dry	H.for2	0.15	0.16	0.67	3
TS	Dry	L.goo1	0.01	0.01	0.86	40
TS	Dry	L.goo2	0.03	0.11	1	289

Slough	Season	Size Class	NP	Pond	NP < Pond	Percent Change
SRS	Wet	E.eve1	0.69	0.46	0.27	-34
SRS	Wet	G.hol1	0.08	0.34	0.96	303
SRS	Wet	G.hol2	0.21	0.37	0.63	75
SRS	Wet	H.for1	0.17	0.09	0.07	-48
SRS	Wet	H.for2	0.09	0.05	0.33	-44
SRS	Dry	F.chr1	1.26	0.69	0.08	-45
SRS	Dry	G.hol1	0.01	0.7	1	8084
SRS	Dry	G.hol2	0.1	0.95	1	805
SRS	Dry	H.for1	0	0.11	1	2412
SRS	Dry	H.for2	0	0.03	1	1714
SRS	Dry	L.goo1	0.02	0.12	1	589
SRS	Dry	L.goo2	0.05	0.21	1	310
SRS	Dry	P.lat2	0.02	0.07	0.97	255
TS	Wet	G.hol1	0.32	0.18	0.08	-45
TS	Wet	H.for2	0.07	0.06	0.49	-10
TS	Wet	L.goo1	0.07	0.15	0.94	103

TS	Wet	L.goo2	0.09	0.16	0.8	88

B. Season Comparisons

Slough	Habitat	Size Class	Wet	Dry	Wet < Dry	Percent Change
SRS	Marsh	C.uro1	0.01	0.01	0.92	0
SRS	Marsh	E.suc1	0.05	0.23	0.95	360
SRS	Marsh	F.chr1	0.66	0.39	0.14	-41
SRS	Marsh	G.hol1	0.1	0.14	0.86	40
SRS	Marsh	G.hol2	0.35	0.12	0.06	-66
SRS	Marsh	H.for2	0.05	0.04	0.29	-20
SRS	Marsh	H.let1	0.16	0.43	1	169
SRS	Marsh	L.goo1	0.11	0.03	0	-73
SRS	Marsh	L.goo2	0.17	0.17	0.45	0
SRS	Marsh	L.pun2	1.06	0.36	0.03	-66
SRS	NP	G.hol1	0.08	0.01	0	-88
SRS	NP	G.hol2	0.21	0.1	0.06	-52
SRS	NP	H.for1	0.17	0	0	-100
SRS	NP	H.for2	0.09	0	0	-100
SRS	Pond	F.chr1	0.55	0.69	0.78	25
SRS	Pond	G.hol1	0.34	0.7	0.98	106
SRS	Pond	G.hol2	0.37	0.95	1	157
SRS	Pond	H.for1	0.09	0.11	0.73	22
SRS	Pond	H.for2	0.05	0.03	0.17	-40
SRS	Pond	L.goo1	0.09	0.12	0.76	33
SRS	Pond	L.goo2	0.3	0.21	0.12	-30
SRS	Pond	L.gul2	1.52	0.22	0	-86
SRS	Pond	L.gul3	0.89	0.47	0.1	-47
SRS	Pond	L.gul4	0.46	0.25	0.17	-46
SRS	Pond	L.pla1	0.81	0.27	0	-67
SRS	Pond	L.pun2	0.55	0.55	0.62	0
SRS	Pond	P.lat2	0.01	0.07	1	600
TS	Marsh	F.chr1	0.63	0.6	0.53	-5
TS	Marsh	H.for1	0.22	0.06	0	-73
TS	Marsh	H.for2	0.14	0.15	0.48	7
TS	Marsh	L.goo1	0.17	0.01	0	-94
TS	Marsh	L.goo2	0.14	0.03	0	-79
TS	Pond	G.hol1	0.18	0.46	0.99	156
TS	Pond	G.hol2	0.89	1.04	0.65	17
TS	Pond	H.for2	0.06	0.16	0.88	167

TS	Pond	L.goo1	0.15	0.01	0	-93
TS	Pond	L.goo2	0.16	0.11	0.26	-31
TS	Pond	N.pet1	0.15	0.6	1	300

C. Slough Comparisons

Season	Habitat	Size Class	SRS	TS	SRS < TS	Percent Change
Dry	Marsh	E.glo1	0.35	0.3	0.32	-13
Dry	Marsh	F.chr1	0.39	0.6	0.83	54
Dry	Marsh	H.for2	0.04	0.15	0.98	288
Dry	Marsh	L.goo1	0.03	0.01	0	-60
Dry	Marsh	L.goo2	0.17	0.03	0	-83
Dry	Pond	G.hol1	0.7	0.46	0.08	-34
Dry	Pond	G.hol2	0.95	1.04	0.62	10
Dry	Pond	H.for2	0.03	0.16	0.99	414
Dry	Pond	L.goo1	0.12	0.01	0	-88
Dry	Pond	L.goo2	0.21	0.11	0.1	-48
Wet	Marsh	F.chr1	0.66	0.63	0.5	-5
Wet	Marsh	G.hol1	0.1	0.23	0.99	122
Wet	Marsh	G.hol2	0.35	0.62	0.95	78
Wet	Marsh	H.for1	0.19	0.22	0.6	13
Wet	Marsh	H.for2	0.05	0.14	0.93	161
Wet	Marsh	L.goo1	0.11	0.17	0.9	55
Wet	Marsh	L.goo2	0.17	0.14	0.3	-17
Wet	NP	E.eve1	0.69	0.11	0.02	-84
Wet	NP	G.hol1	0.08	0.32	0.95	277
Wet	NP	H.for1	0.17	0.13	0.25	-20
Wet	NP	H.for2	0.09	0.07	0.35	-25
Wet	Pond	G.hol1	0.34	0.18	0.05	-48
Wet	Pond	G.hol2	0.37	0.89	1	140
Wet	Pond	H.for2	0.05	0.06	0.53	22
Wet	Pond	L.goo1	0.09	0.15	0.93	58
Wet	Pond	L.goo2	0.3	0.16	0.02	-46
Wet	Pond	L.mac1	0.56	0.74	0.68	33
Wet	Pond	M.sal1	0.38	0.91	0.94	142
Wet	Pond	M.sia1	0.16	0.24	0.67	47
Wet	Pond	T.mar1	0.1	0.19	0.82	90

Table S6. Trophic dispersion based on trophic niches derived from stable isotopes of carbon and nitrogen compared across A) habitats, B) seasons, and C) sloughs. Percent Change is calculated as the percent change from the first group in the comparison relative to the second group (e.g., for Marsh < NP the marsh value is considered the initial value). Probabilities in the column comparing the two group (e.g., Marsh < NP) are the probability that the posterior distribution of the first group is less than that of the second group. Therefore, values ≥ 0.95 we considered an increase in trophic dispersion, while values ≤ 0.05 represent the opposite one-tailed hypothesis (i.e., $p(A > B) \geq 0.95$) and we considered a decrease in trophic dispersion.

A. Habitat Comparisons

Size Class	Slough	Season	Marsh	NP	Marsh < NP	Percent Change
Belostoma spp.	SRS	Wet	2.01	1.72	0.42	-14
Coenagrionidae			0.66	0.56	0.42	-14
E.eve1			0.62	1.28	0.88	107
F.chr1			1.73	0.44	0.07	-75
G.hol1			1.3	0.64	0.17	-51
G.hol2			0.88	0.68	0.17	-22
H.for1			0.68	2.44	0.99	260
H.for2			1.94	0.53	0.01	-73
H.let1			2.49	1.49	0.39	-40
L.goo2			5.22	1.37	0.01	-74
P. paludosus			45.16	0.44	0	-99
P. femoratus			3.42	0.3	0	-91
P. fallax			1.16	0.65	0.15	-44
Procambarus spp.			0.62	0.36	0.17	-42
E.eve1		Dry	1.94	3.1	0.83	60
E.glo1			2.74	1.78	0.25	-35
F.chr1			4.37	4.77	0.58	9
F.chr2			3.22	0.42	0.02	-87
G.hol1			0.8	1.02	0.7	28
G.hol2			2	2.8	0.89	40
H.for2			2.31	1.27	0.18	-45
L.goo1			2.54	1.41	0.11	-44
L.goo2			2.65	2.74	0.56	3
L.mar1			2.05	2.34	0.61	14
P. paludosus			20.76	0.57	0	-97
Hydrachnidia	TSL	Wet	102.14	54.42	0.07	-47
B. gravida			1.38	1.15	0.44	-17

C.bim1			5.18	4.55	0.35	-12
C.man1			0.31	4.48	1	1344
C.uro1			3.08	1.56	0.13	-50
Celithimus spp.			0.06	0.3	0.99	396
Chironomidae			83.42	11.21	0	-87
Coenagrionidae			0.34	0.35	0.52	2
Coleoptera			9.41	0.13	0	-99
Copepoda			106.69	18.76	0	-82
E.glo1			2.84	1.92	0.38	-32
E. simplicicollis			4.93	3.5	0.3	-29
F.chr1			1.48	0.3	0.05	-80
G.hol1			3.44	0.73	0	-79
G.hol2			2.68	1.24	0.07	-54
Gerridae			0.16	0.54	0.94	230
H.for1			3.26	1.51	0.03	-54
H.for2			1.81	2.33	0.67	29
L.goo1			5.69	2.27	0.02	-60
L.goo2			3.39	2.3	0.32	-32
L.mar1			1.04	3.4	0.94	228
P. paludosus			0.72	52.15	1	7171
P. femoratus			16.18	7.26	0.21	-55
Planorbella spp.			0.96	97.8	1	10087
Hydrachnidia		Dry	112.99	168.24	0.81	49
Copepoda			67.45	14.52	0.05	-78

Size Class	Slough	Season	Marsh	Pond	Marsh < Pond	Percent Change
C.bat1	SRS	Wet	3.64	0.65	0.03	-82
C.uro1			3.74	1.1	0.01	-71
C.uro2			2.27	2.36	0.64	4
Celithimus spp.			0.08	0.89	1	1013
Coenagrionidae			0.66	3.31	1	403
E.glo1			1.5	1.08	0.19	-28
E.suc1			2.22	1.51	0.23	-32
F.chr1			1.73	4.95	1	186
F.chr2			0.9	2.24	0.98	150
H.for1			0.68	6.99	1	933
H.for2			1.94	0.86	0.14	-56
H.let1			2.49	2.65	0.63	6
J.flo2			3.96	1.01	0.03	-74
L.goo1			3.88	4.15	0.57	7

L.goo2			5.22	4.56	0.22	-13
L.mac1			1.99	1.75	0.46	-12
L.mar1			3.3	3.67	0.5	11
L.mic1			2.88	1.45	0.08	-50
L.mic2			3.45	1.02	0.01	-71
L.pun2			2.49	1.29	0.11	-48
L. incesta			0.01	1.35	1	10785
M.sia1			0.76	2.53	0.88	235
O.aur1			4.05	4.52	0.68	12
P.lat2			0.46	0.97	0.92	111
P. paludosus			45.16	13.17	0.09	-71
P. femoratus			3.42	4.97	0.73	46
P. fallax			1.16	1.56	0.69	35
Procambarus spp.			0.62	0.02	0	-96
B.bel1		Dry	0.46	0.44	0.4	-5
C.uro1			1.33	1.93	0.62	45
Celithimus spp.			0.78	0.74	0.44	-6
E.eve1			1.94	2.04	0.62	5
F.chr1			4.37	5.29	0.66	21
F.con1			3.52	0.81	0	-77
G.hol1			0.8	4.67	1	485
G.hol2			2	4	1	101
H.for2			2.31	0.33	0	-86
H.lit1			2.49	1.81	0.19	-27
J.flo2			2.29	1.27	0.11	-44
L.goo1			2.54	1.86	0.14	-27
L.goo2			2.65	3.6	0.89	36
L.mar1			2.05	1.63	0.28	-20
L.pun2			1.32	1.65	0.7	25
P. paludosus			20.76	0.67	0	-97
Planorbella spp.			5.98	77.76	1	1201
P. alleni			1.88	1.12	0.16	-40
S.lac1			2.22	4.24	0.89	91
B. gravida	TSL	Wet	1.38	2.42	0.82	75
C.uro1			3.08	1.06	0.12	-66
Copepoda			106.69	0.42	0	-100
G.hol1			3.44	2.05	0.08	-41
G.hol2			2.68	3.54	0.85	32
H.for2			1.81	0.31	0.01	-83
L.goo1			5.69	7.09	0.77	25

L.goo2			3.39	6.12	0.99	81
N.pet1			2.1	2.84	0.63	35
P. paludosus			0.72	31.46	1	4286
Chironomidae		Dry	34.73	0.49	0	-99
Copepoda			67.45	43.34	0.19	-36
G.hol2			0.11	2.61	1	2374
H.for2			3.16	0.17	0	-95
L.goo1			1.28	2.1	0.92	63
L.goo2			1.99	4.81	1	142
P. paludosus			3.75	0.6	0	-84

Size Class	Slough	Season	NP	Pond	NP < Pond	Percent Change
Coenagrionidae	SRS	Wet	0.56	3.31	1	488
F.chr1			0.44	4.95	1	1031
H.for1			2.44	6.99	0.98	187
H.for2			0.53	0.86	0.84	63
H.let1			1.49	2.65	0.66	77
L.goo2			1.37	4.56	0.98	234
P. paludosus			0.44	13.17	1	2915
P. femoratus			0.3	4.97	1	1562
P. fallax			0.65	1.56	0.93	140
Procambarus spp.			0.36	0.02	0	-94
E.eve1		Dry	3.1	2.04	0.31	-34
F.chr1			4.77	5.29	0.59	11
G.hol1			1.02	4.67	1	358
G.hol2			2.8	4	0.91	43
H.for1			1	3.39	0.95	237
H.for2			1.27	0.33	0.01	-74
L.goo1			1.41	1.86	0.69	31
L.goo2			2.74	3.6	0.83	31
L.mar1			2.34	1.63	0.19	-30
P.lat2			2.09	2.68	0.62	28
P. paludosus			0.57	0.67	0.63	17
P. fallax			0.8	1.35	0.83	67
B. gravida	TSL	Wet	1.15	2.42	0.84	110
C.uro1			1.56	1.06	0.37	-32
Copepoda			18.76	0.42	0	-98
G.hol1			0.73	2.05	0.97	179
G.hol2			1.24	3.54	0.98	184

H.for2			2.33	0.31	0	-87
L.goo1			2.27	7.09	0.99	212
L.goo2			2.3	6.12	0.95	166
P. paludosus			52.15	31.46	0.15	-40
Amphipoda		Dry	147.3	6.75	0	-95
Copepoda			14.52	43.34	0.87	199

B. Seasonal Comparisons

Size Class	Slough	Habitat	Wet	Dry	Wet < Dry	Percent Change
C.uro1	SRS	Marsh	3.74	1.33	0.09	181
Celithimus spp.			0.08	0.78	1	-90
E.eve1			0.62	1.94	0.97	-68
E.glo1			1.5	2.74	0.81	-45
E.suc1			2.22	0.8	0.04	178
F.chr1			1.73	4.37	1	-60
F.chr2			0.9	3.22	1	-72
G.hol1			1.3	0.8	0.14	63
G.hol2			0.88	2	1	-56
H.for2			1.94	2.31	0.53	-16
H.let1			2.49	1.92	0.1	30
J.flo2			3.96	2.29	0.23	73
L.goo1			3.88	2.54	0.09	53
L.goo2			5.22	2.65	0	97
L.mar1			3.3	2.05	0.18	61
L.pun2			2.49	1.32	0.1	89
P. paludosus			45.16	20.76	0.1	117
P. femoratus			3.42	1.18	0	189
E.eve1		NP	1.28	3.1	0.99	-59
F.chr1			0.44	4.77	1	-91
G.hol1			0.64	1.02	0.71	-38
G.hol2			0.68	2.8	1	-76
H.for1			2.44	1	0.12	143
H.for2			0.53	1.27	0.95	-59
L.goo2			1.37	2.74	0.86	-50
P. paludosus			0.44	0.57	0.66	-24
P. fallax			0.65	0.8	0.62	-19
A.nat3		Pond	0.06	1.09	1	-95
B.bel1			0.47	0.44	0.39	9
B. gravida			2.92	2.29	0.28	27

C.uro1			1.1	1.93	0.85	-43
Celithimus spp.			0.89	0.74	0.36	21
E.suc2			1.21	0.14	0	744
F.chr1			4.95	5.29	0.62	-7
F.con1			0.65	0.81	0.64	-20
H.for1			6.99	3.39	0.04	106
H.for2			0.86	0.33	0.04	159
J.flo2			1.01	1.27	0.57	-20
L.goo1			4.15	1.86	0	123
L.goo2			4.56	3.6	0.16	27
L.gul2			0.85	0.43	0.05	97
L.gul3			0.89	0.74	0.34	20
L.gul4			0.84	1.86	0.95	-55
L.mar1			3.67	1.63	0.03	125
L.mic1			1.45	1.25	0.48	16
L.pla1			3.14	2.3	0.15	36
L.pun2			1.29	1.65	0.69	-22
P.lat2			0.97	2.68	0.99	-64
P. paludosus			13.17	0.67	0	1867
P. alleni			0.64	1.12	0.93	-43
P. fallax			1.56	1.35	0.25	16
Hydrachnidia	TSL	Marsh	102.14	112.99	0.56	-10
Celithimus spp.			0.06	0.71	1	-92
Chironomidae			83.42	34.73	0.06	140
Coenagrionidae			0.34	2.38	0.99	-86
Copepoda			106.69	67.45	0.17	58
Corixidae			2.62	0.73	0.04	259
E.glo1			2.84	1.74	0.15	63
Ephemeroptera			1.84	1.6	0.45	15
E. simplicicollis			4.93	4.68	0.57	5
F.chr1			1.48	2.28	0.86	-35
G.hol2			2.68	0.11	0	2447
Gerridae			0.16	57.62	1	-100
H.for1			3.26	2.86	0.36	14
H.for2			1.81	3.16	0.84	-43
L.goo1			5.69	1.28	0	343
L.goo2			3.39	1.99	0.02	71
P. paludosus			0.72	3.75	1	-81
P. femoratus			16.18	4.7	0	245
Hydrachnidia		NP	54.42	168.24	0.99	-68

Cladocera			93.19	71.71	0.4	30
Copepoda			18.76	14.52	0.5	29
Amphipoda		Pond	5.21	6.75	0.61	-23
Copepoda			0.42	43.34	1	-99
G.hol1			2.05	3.68	0.95	-44
G.hol2			3.54	2.61	0.1	36
H.for2			0.31	0.17	0.21	85
L.goo1			7.09	2.1	0.06	238
L.goo2			6.12	4.81	0.33	27
N.pet1			2.84	3.04	0.54	-6
P. paludosus			31.46	0.6	0	5143

C. Slough Comparisons

Size Class	Season	Habitat	TSL	SRS	TSL < SRS	Percent Change
Celithimus spp.	Dry	Marsh	0.71	0.78	0.43	-9
E.glo1			1.74	2.74	0.17	-37
E. simplicicollis			4.68	0.56	1	737
F.chr1			2.28	4.37	0.18	-48
G.hol2			0.11	2	0	-95
H.for2			3.16	2.31	0.74	36
L.goo1			1.28	2.54	0.01	-49
L.goo2			1.99	2.65	0.13	-25
P. paludosus			3.75	20.76	0	-82
P. femoratus			4.7	1.18	1	297
Copepoda		Pond	43.34	72.42	0.16	-40
G.hol1			3.68	4.67	0.16	-21
G.hol2			2.61	4	0.06	-35
H.for2			0.17	0.33	0.2	-50
L.goo1			2.1	1.86	0.75	13
L.goo2			4.81	3.6	0.84	34
P. paludosus			0.6	0.67	0.41	-10
Belostoma spp.	Wet	Marsh	0.42	2.01	0.01	-79
C.uro1			3.08	3.74	0.33	-18
Celithimus spp.			0.06	0.08	0.32	-25
Chironomidae			83.42	4.87	1	1612
Coenagrionidae			0.34	0.66	0.22	-48
Copepoda			106.69	2.25	1	4650
E.glo1			2.84	1.5	0.83	89

F.chr1		1.48	1.73	0.37	-15
G.hol1		3.44	1.3	1	165
G.hol2		2.68	0.88	1	206
H.for1		3.26	0.68	1	382
H.for2		1.81	1.94	0.39	-7
L.goo1		5.69	3.88	0.9	47
L.goo2		3.39	5.22	0.04	-35
L.mar1		1.04	3.3	0.07	-69
P. paludosus		0.72	45.16	0	-98
P. femoratus		16.18	3.42	0.99	374
Coenagrionidae	NP	0.35	0.56	0.32	-38
E.eve1		0.42	1.28	0.09	-67
F.chr1		0.3	0.44	0.35	-31
G.hol1		0.73	0.64	0.53	15
G.hol2		1.24	0.68	0.97	82
H.for1		1.51	2.44	0.13	-38
H.for2		2.33	0.53	1	343
L.goo2		2.3	1.37	0.82	69
P. paludosus		52.15	0.44	1	11840
P. femoratus		7.26	0.3	1	2324
Amphipoda	Pond	5.21	60.5	0.01	-91
B. gravida		2.42	2.92	0.49	-17
C.uro1		1.06	1.1	0.57	-3
C.uro2		1.07	2.36	0.04	-55
H.for2		0.31	0.86	0.08	-64
L.goo1		7.09	4.15	0.99	71
L.goo2		6.12	4.56	0.94	34
L.mac1		0.39	1.75	0.01	-78
M.sal1		3.01	1.23	1	145
M.sia1		2.04	2.53	0.15	-20
N.pet1		2.84	0.6	0.99	371
P.lat2		1.35	0.97	0.82	39
P. paludosus		31.46	13.17	0.81	139
T.mar1		8.49	0.51	1	1563

Table S7. Trophic position by species size class based on counts of stomach contents.

Size Class	Slough	Season	Habitat	N	Mean	SD
A.cal2	SRS	Wet	Marsh	1	1.75	-
A.cal2	SRS	Wet	Pond	4	2.27	0.52
A.gry1	SRS	Wet	Pond	2	2.69	0.44
A.nat1	SRS	Dry	Marsh	2	2.63	0.18
A.nat1	SRS	Dry	Pond	3	2.42	1.28
A.nat2	SRS	Dry	Pond	4	3.00	0.41
A.nat2	SRS	Wet	Marsh	1	2.69	-
A.nat3	SRS	Dry	Pond	6	2.23	0.60
A.nat3	SRS	Wet	Pond	1	2.96	-
A.nat4	SRS	Dry	Pond	8	2.56	0.48
A.nat4	SRS	Wet	Marsh	1	3.37	-
A.ros1	TS	Wet	Pond	1	2.67	-
B.bel1	SRS	Dry	Marsh	1	2.50	-
B.bel1	SRS	Dry	Pond	2	2.50	0.00
B.bel1	SRS	Wet	Pond	1	3.17	-
B.bel1	TS	Dry	Marsh	3	2.50	-
B.bel1	TS	Wet	NP	1	3.00	-
B.bel1	TS	Wet	Pond	1	2.50	-
C.bat1	SRS	Dry	Pond	1	1.10	-
C.bat1	SRS	Wet	Marsh	8	2.74	0.54
C.bat1	SRS	Wet	Pond	1	2.50	-
C.bat1	TS	Wet	Marsh	1	2.05	-
C.bat1	TS	Wet	Pond	1	2.99	-
C.bim1	TS	Wet	NP	3	2.89	0.19
C.man1	SRS	Dry	Pond	1	3.37	-
C.man1	TS	Wet	NP	1	2.20	-
C.oce1	SRS	Wet	Marsh	1	2.50	-
C.oce1	TS	Wet	Pond	1	2.50	-
C.und1	SRS	Wet	Marsh	3	2.33	0.29
C.uro1	SRS	Dry	Marsh	4	2.86	0.33
C.uro1	SRS	Wet	Marsh	15	2.62	0.54
C.uro1	SRS	Wet	Pond	6	2.01	0.83
C.uro1	TS	Wet	Marsh	1	2.33	-
C.uro1	TS	Wet	NP	2	2.33	0.94
C.uro1	TS	Wet	Pond	2	2.51	0.02
C.uro2	SRS	Dry	Pond	1	3.50	-
C.uro2	SRS	Wet	Marsh	4	2.08	0.44

C.uro2	SRS	Wet	Pond	2	1.58	0.82
C.uro2	TS	Wet	Marsh	1	3.00	-
C.uro2	TS	Wet	Pond	2	3.17	0.28
C.var1	SRS	Wet	Marsh	1	3.00	-
E.ame1	SRS	Dry	Marsh	1	2.50	-
E.eve1	SRS	Dry	Marsh	8	2.50	0.70
E.eve1	SRS	Dry	Pond	1	1.00	-
E.eve1	SRS	Wet	Marsh	1	2.00	-
E.eve1	SRS	Wet	NP	2	2.60	0.78
E.eve1	SRS	Wet	Pond	19	2.78	0.52
E.eve1	TS	Dry	Marsh	7	1.25	-
E.eve1	TS	Wet	Marsh	1	2.44	-
E.eve1	TS	Wet	NP	3	2.44	0.14
E.glo1	SRS	Dry	Marsh	5	2.51	0.44
E.glo1	SRS	Wet	Marsh	9	2.50	-
E.glo1	SRS	Wet	Pond	1	2.46	1.00
E.glo1	TS	Dry	Marsh	3	2.32	0.51
E.glo1	TS	Wet	NP	2	2.63	0.18
E.suc1	SRS	Dry	Marsh	4	2.43	0.29
E.suc1	SRS	Dry	Pond	2	2.41	0.40
E.suc1	SRS	Wet	Marsh	3	2.41	0.11
E.suc1	SRS	Wet	Pond	5	2.53	0.63
E.suc2	SRS	Dry	Pond	3	2.83	0.58
E.suc2	SRS	Wet	Pond	2	2.69	0.37
F.chr1	SRS	Dry	Marsh	9	2.34	0.71
F.chr1	SRS	Dry	NP	5	2.73	0.43
F.chr1	SRS	Dry	Pond	11	2.27	0.50
F.chr1	SRS	Wet	Marsh	5	2.47	0.54
F.chr1	SRS	Wet	NP	14	2.38	-
F.chr1	SRS	Wet	Pond	1	2.29	0.56
F.chr1	TS	Dry	Marsh	23	2.26	0.65
F.chr1	TS	Wet	Marsh	11	2.58	0.51
F.chr1	TS	Wet	NP	1	1.83	-
F.chr2	SRS	Dry	Marsh	1	1.00	-
F.chr2	SRS	Dry	Pond	1	3.00	-
F.chr2	SRS	Wet	Pond	1	2.90	-
F.con1	SRS	Dry	Marsh	2	2.20	0.42
F.con1	SRS	Dry	Pond	1	2.50	-
G.hol1	SRS	Dry	Marsh	7	1.92	0.89
G.hol1	SRS	Dry	NP	9	2.40	0.54

G.hol1	SRS	Dry	Pond	16	2.55	0.32
G.hol1	SRS	Wet	Marsh	1	2.67	0.44
G.hol1	SRS	Wet	NP	26	2.79	0.25
G.hol1	SRS	Wet	Pond	18	2.30	0.65
G.hol1	TS	Dry	Marsh	4	1.75	-
G.hol1	TS	Dry	Pond	18	2.23	0.68
G.hol1	TS	Wet	Marsh	17	2.40	0.48
G.hol1	TS	Wet	NP	7	2.39	0.78
G.hol1	TS	Wet	Pond	13	2.39	0.65
G.hol2	SRS	Dry	Marsh	5	2.58	0.71
G.hol2	SRS	Dry	NP	11	2.49	0.46
G.hol2	SRS	Dry	Pond	27	2.47	0.50
G.hol2	SRS	Wet	Marsh	2	2.37	0.51
G.hol2	SRS	Wet	NP	23	2.61	0.13
G.hol2	SRS	Wet	Pond	18	2.45	0.42
G.hol2	TS	Dry	Marsh	3	2.50	0.00
G.hol2	TS	Dry	Pond	21	2.13	0.72
G.hol2	TS	Wet	Marsh	15	2.72	0.38
G.hol2	TS	Wet	NP	1	2.30	-
G.hol2	TS	Wet	Pond	19	2.65	0.40
H.for1	SRS	Dry	Marsh	1	1.25	-
H.for1	SRS	Dry	NP	4	2.22	0.75
H.for1	SRS	Dry	Pond	5	2.33	0.55
H.for1	SRS	Wet	Marsh	13	2.48	0.61
H.for1	SRS	Wet	NP	8	2.07	0.56
H.for1	SRS	Wet	Pond	6	2.78	0.64
H.for1	TS	Dry	Marsh	9	2.39	0.48
H.for1	TS	Wet	Marsh	11	2.81	0.71
H.for1	TS	Wet	NP	11	2.73	0.42
H.for2	SRS	Dry	Marsh	5	2.48	0.47
H.for2	SRS	Dry	NP	5	2.29	0.80
H.for2	SRS	Dry	Pond	5	2.35	0.49
H.for2	SRS	Wet	Marsh	10	2.54	0.35
H.for2	SRS	Wet	NP	3	2.48	0.51
H.for2	SRS	Wet	Pond	5	2.54	0.31
H.for2	TS	Dry	Marsh	5	2.55	0.53
H.for2	TS	Dry	Pond	3	2.39	0.64
H.for2	TS	Wet	Marsh	5	2.78	0.46
H.for2	TS	Wet	NP	4	2.30	0.57
H.for2	TS	Wet	Pond	3	2.43	0.28

H.let1	SRS	Dry	Marsh	33	2.45	0.50
H.let1	SRS	Wet	Marsh	24	2.54	0.53
H.let1	SRS	Wet	Pond	14	2.62	0.47
H.lit1	SRS	Dry	Marsh	3	2.81	0.27
H.lit1	SRS	Dry	Pond	8	2.66	0.44
J.flo1	SRS	Dry	Pond	1	2.30	-
J.flo1	SRS	Wet	Marsh	3	2.25	0.43
J.flo1	SRS	Wet	Pond	2	1.88	1.24
J.flo2	SRS	Dry	Marsh	1	3.12	-
J.flo2	SRS	Dry	Pond	3	2.67	0.14
J.flo2	SRS	Wet	Marsh	1	1.85	-
J.flo2	SRS	Wet	Pond	1	2.67	-
L.goo1	SRS	Dry	Marsh	18	2.42	0.43
L.goo1	SRS	Dry	NP	7	2.43	0.56
L.goo1	SRS	Dry	Pond	22	2.34	0.55
L.goo1	SRS	Wet	Marsh	24	2.37	0.47
L.goo1	SRS	Wet	Pond	4	2.63	0.36
L.goo1	TS	Dry	Marsh	16	2.51	0.46
L.goo1	TS	Dry	Pond	20	2.23	0.26
L.goo1	TS	Wet	Marsh	18	2.48	0.42
L.goo1	TS	Wet	NP	9	2.51	0.55
L.goo1	TS	Wet	Pond	20	2.33	0.54
L.goo2	SRS	Dry	Marsh	25	2.58	0.54
L.goo2	SRS	Dry	NP	7	2.55	0.39
L.goo2	SRS	Dry	Pond	23	2.51	0.54
L.goo2	SRS	Wet	Marsh	27	2.54	0.40
L.goo2	SRS	Wet	NP	9	2.07	-
L.goo2	SRS	Wet	Pond	16	2.21	0.63
L.goo2	TS	Dry	Marsh	1	2.54	0.48
L.goo2	TS	Dry	Pond	19	2.57	0.28
L.goo2	TS	Wet	Marsh	17	2.54	0.40
L.goo2	TS	Wet	NP	5	2.86	0.32
L.goo2	TS	Wet	Pond	20	2.51	0.45
L.gul1	SRS	Dry	Pond	1	3.00	-
L.gul1	SRS	Wet	Pond	1	1.73	-
L.gul2	SRS	Dry	Pond	9	2.37	0.57
L.gul2	SRS	Wet	Pond	4	2.21	0.83
L.gul3	SRS	Dry	Pond	9	2.37	0.60
L.gul3	SRS	Wet	Pond	9	2.44	0.62
L.gul3	TS	Wet	Pond	1	3.33	-

L.gul4	SRS	Dry	Pond	4	2.42	0.55
L.gul4	SRS	Wet	Pond	3	2.51	0.56
L.mac1	SRS	Dry	Pond	2	2.69	0.80
L.mac1	SRS	Wet	Marsh	17	2.67	0.47
L.mac1	SRS	Wet	Pond	6	2.47	0.74
L.mac1	TS	Wet	Pond	5	2.25	0.83
L.mar1	SRS	Dry	Marsh	3	2.56	0.26
L.mar1	SRS	Dry	Pond	7	2.40	0.63
L.mar1	SRS	Wet	Pond	3	3.18	0.19
L.mar1	TS	Wet	Pond	1	2.50	-
L.mic1	SRS	Dry	Marsh	1	3.50	-
L.mic1	SRS	Wet	Marsh	3	1.97	0.84
L.mic1	SRS	Wet	Pond	2	2.25	0.35
L.mic1	TS	Wet	Pond	2	2.56	0.32
L.mic2	SRS	Wet	Marsh	3	2.83	0.52
L.mic2	SRS	Wet	Pond	4	2.63	0.25
L.mic2	TS	Wet	Pond	1	2.50	-
L.pla1	SRS	Dry	Pond	11	2.46	0.59
L.pla1	SRS	Wet	Pond	13	2.63	0.53
L.pla2	SRS	Wet	Pond	4	2.73	0.43
L.pun2	SRS	Dry	Marsh	7	2.41	0.29
L.pun2	SRS	Dry	Pond	4	2.64	0.16
L.pun2	SRS	Wet	Marsh	3	2.58	0.52
L.pun2	SRS	Wet	Pond	7	2.63	0.27
L.pun3	SRS	Dry	Marsh	1	1.00	-
L.pun3	SRS	Dry	Pond	2	1.50	0.71
L.pun3	SRS	Wet	Pond	1	2.50	-
L.sic1	TS	Wet	Pond	23	2.60	0.49
M.alb1	TS	Wet	Marsh	1	2.50	-
M.alb2	TS	Dry	Marsh	1	3.00	-
M.alb2	TS	Wet	Marsh	1	2.50	-
M.alb3	TS	Wet	Marsh	2	2.63	0.18
M.alb4	SRS	Wet	Pond	1	2.10	-
M.ber1	TS	Dry	Pond	3	2.72	0.25
M.sal1	SRS	Dry	Pond	1	3.00	-
M.sal1	SRS	Wet	Pond	7	2.68	0.37
M.sal1	TS	Wet	Pond	18	2.63	0.56
M.sia1	SRS	Wet	Marsh	1	3.00	-
M.sia1	SRS	Wet	Pond	6	2.68	0.84
M.sia1	TS	Wet	Pond	11	2.47	0.63

N.flo1	SRS	Dry	Pond	1	3.00	-
N.gyr1	SRS	Dry	Marsh	1	3.00	-
N.pet1	SRS	Wet	Pond	15	2.75	0.35
N.pet1	TS	Dry	Pond	2	2.48	0.52
N.pet1	TS	Wet	Marsh	1	3.00	-
N.pet1	TS	Wet	Pond	10	2.79	0.40
N.vir1	SRS	Dry	Marsh	1	2.67	-
O.aur1	SRS	Dry	Marsh	1	2.00	-
O.aur1	SRS	Wet	Marsh	10	2.51	0.84
O.aur1	SRS	Wet	Pond	4	2.63	0.25
P.axa1	SRS	Dry	Marsh	2	3.08	0.12
P.axa1	SRS	Wet	Marsh	1	2.50	-
P.lat1	SRS	Dry	NP	1	3.00	-
P.lat1	SRS	Wet	Pond	5	2.24	0.88
P.lat2	SRS	Dry	Marsh	2	2.75	0.35
P.lat2	SRS	Dry	NP	5	2.35	0.80
P.lat2	SRS	Dry	Pond	6	2.88	0.56
P.lat2	SRS	Wet	Marsh	2	2.75	0.35
P.lat2	SRS	Wet	Pond	8	2.38	0.64
P.lat2	TS	Wet	Pond	1	1.20	-
R.gry1	SRS	Dry	Marsh	6	2.49	0.36
R.gry2	SRS	Dry	Pond	1	2.50	-
S.lac1	SRS	Dry	Marsh	3	1.80	0.78
S.lac1	SRS	Dry	Pond	2	2.08	0.12
S.lac1	SRS	Wet	Marsh	1	2.00	-
S.lac1	SRS	Wet	Pond	1	2.50	-
T.mar1	SRS	Dry	Marsh	4	2.38	0.95
T.mar1	SRS	Wet	Pond	5	2.30	0.27
T.mar1	TS	Wet	Pond	10	2.38	0.55

Table S8. Comparisons of trophic positions modeled with stable isotopes of carbon and nitrogen across A) habitats, B) season, and C) sloughs. Statistical comparisons are in the form of Table S5, S6. We also calculated the change in trophic position (ΔTP) from the first group to the second (e.g., in the first row the change in trophic position from marsh to near-pond).

A. Habitat Comparisons

Species	Slough	Season	Marsh	NP	Marsh < NP	Percent Change	ΔTP
Belostoma spp.	SRS	Wet	2.44	2.6	0.74	6.4	-0.16
Coenagrionidae			2.41	2.82	0.99	17.0	-0.41
E. evergladei			3.18	3.49	0.98	9.8	-0.31
F. chrysotus			3.82	3.68	0.25	-3.7	0.14
G. holbrooki			3.56	3.63	0.72	1.9	-0.07
H. letourneuxi			3.63	3.63	0.5	0.1	0
H. formosa			3.64	3.66	0.56	0.5	-0.02
L. goodei			3.35	3.57	0.89	6.8	-0.22
P. paludosus			3.17	2.98	0.18	-6.0	0.19
P. femoratus			2.19	2.2	0.54	0.7	-0.01
P. fallax			2.51	2.61	0.68	3.9	-0.1
Procambarus spp.			2.5	2.67	0.86	6.8	-0.17
E. evergladei		Dry	3.52	3.37	0.22	-4.4	0.15
E. gloriosus			3.31	3.05	0.12	-7.9	0.26
F. chrysotus			3.55	3.5	0.37	-1.7	0.05
G. holbrooki			3.28	3.39	0.82	3.3	-0.11
H. formosa			3.39	3.22	0.12	-5.0	0.17
L. marginatus			3.48	3.36	0.3	-3.5	0.12
L. goodei			3.2	3.06	0.14	-4.2	0.14
P. paludosus			2.9	2.91	0.52	0.2	-0.01
Amphipoda	TSL	Wet	2.14	2.04	0.21	-4.7	0.1
Hydrachnidia			2.95	2.74	0.25	-7.1	0.21
B. gravida			2.58	2.66	0.57	2.9	-0.08
Celithemus spp.			2.08	2.53	0.96	21.8	-0.45
Chironomidae			2.66	2.06	0.06	-22.5	0.6
C. bimaculatum			3.43	3.31	0.33	-3.5	0.12
C. managuense			4.26	3.13	0.02	-26.5	1.13
C. urophthalmus			3.58	3.67	0.64	2.3	-0.09
Coenagrionidae			2.19	2.94	0.99	34.5	-0.75
Coleoptera			2.42	2.03	0.06	-16.0	0.39
Copepoda			2.88	2.11	0.02	-26.5	0.77
E. gloriosus			3.63	3.02	0.08	-16.9	0.61

E. simplicicollis			2.46	2.11	0.11	-14.3	0.35
F. chrysotus			3.57	3.47	0.36	-2.8	0.1
G. holbrooki			3.47	3.17	0.06	-8.4	0.3
Gerridae			2.92	2.73	0.15	-6.4	0.19
H. formosa			3.67	3.62	0.36	-1.5	0.05
L. marginatus			3.23	3.1	0.31	-4.3	0.13
L. goodei			3.58	3.07	0.02	-14.2	0.51
Ostracoda			2.69	2.1	0.05	-22.1	0.59
P. paludosus			2.98	2.32	0.01	-22.2	0.66
P. femoratus			2.39	2.51	0.59	4.7	-0.12
Planorbella spp.			2.92	2.54	0.29	-13.0	0.38
Amphipoda		Dry	2.14	2.37	0.8	10.6	-0.23
Hydrachnidia			2.67	2.33	0.2	-12.6	0.34
Cladocera			2.31	2.18	0.38	-5.4	0.13
Copepoda			2.38	2.32	0.41	-2.8	0.06

Species	Slough	Season	Marsh	Pond	Marsh < Pond	Percent Change	ΔTP
A. natalis	SRS	Wet	4.2	3.67	0.13	-12.48	0.53
Amphipoda			2.12	2.2	0.67	3.73	-0.08
Celithemus spp.			2.3	2.48	0.88	7.97	-0.18
C. urophthalmus			3.52	3.36	0.14	-4.66	0.16
C. batrachus			3.82	3.64	0.35	-4.84	0.18
Coenagrionidae			2.41	2.52	0.65	4.36	-0.11
Copepoda			2.29	2.04	0.11	-10.96	0.25
E. gloriosus			3.71	3.47	0.08	-6.45	0.24
E. sucetta			3.46	3.2	0.22	-7.49	0.26
F. chrysotus			3.82	3.79	0.44	-0.6	0.03
G. holbrooki			3.56	3.63	0.74	2.05	-0.07
H. letourneuxi			3.63	3.44	0.08	-5.05	0.19
H. formosa			3.64	3.84	0.91	5.35	-0.2
J. floridae			3.21	3.52	0.9	9.57	-0.31
L. macrochirus			3.97	3.34	0	-15.88	0.63
L. marginatus			3.35	3.25	0.32	-2.92	0.1
L. microlophus			3.77	3.25	0	-13.8	0.52
L. punctatus			3.96	3.74	0.16	-5.5	0.22
L. incesta			2.21	2.26	0.53	1.94	-0.05
L. goodei			3.35	3.53	0.93	5.41	-0.18
M. siamensis			3.48	3.36	0.33	-3.59	0.12
O. aureus			3.73	3.42	0.1	-8.36	0.31
P. paludosus			3.17	3.54	0.88	11.77	-0.37

P. femoratus			2.19	2.14	0.43	-2.15	0.05
P. latipinna			3.56	3.49	0.33	-1.99	0.07
P. fallax			2.51	2.26	0.16	-9.87	0.25
Procambarus spp.			2.5	2.51	0.52	0.28	-0.01
Amphipoda		Dry	4.94	2.26	0.06	-54.35	2.68
B. belizanus			3.94	4.06	0.64	3.1	-0.12
Celithemus spp.			2.1	2.07	0.4	-1.38	0.03
C. urophthalmus			3.5	3.28	0.13	-6.37	0.22
E. evergladei			3.52	3.34	0.23	-5.28	0.18
E. sucetta			3.29	3.47	0.77	5.41	-0.18
F. chrysotus			3.55	3.79	0.84	6.72	-0.24
F. confluentus			3.56	3.37	0.34	-5.31	0.19
G. holbrooki			3.28	3.68	0.96	11.97	-0.4
H. formosa			3.39	3.83	1	13.16	-0.44
H. littorale			3.52	3.46	0.41	-1.79	0.06
J. floridae			3.25	3.33	0.65	2.65	-0.08
L. marginatus			3.48	3.52	0.58	1.06	-0.04
L. punctatus			3.6	3.65	0.62	1.42	-0.05
L. goodei			3.2	3.44	0.97	7.66	-0.24
P. paludosus			2.9	3.16	0.9	8.99	-0.26
Planorbella spp.			2.21	2.65	0.84	20.01	-0.44
P. alleni			2.48	2.28	0.16	-8.18	0.2
S. lacertina			3.16	3.17	0.53	0.32	-0.01
Amphipoda	TSL	Wet	2.14	2.48	0.77	15.77	-0.34
B. gravida			2.58	2.22	0.18	-14.06	0.36
C. urophthalmus			3.58	3.32	0.12	-7.45	0.26
Copepoda			2.88	2.09	0.02	-27.17	0.79
G. holbrooki			3.47	3.53	0.62	1.79	-0.06
H. formosa			3.67	3.34	0.09	-9.01	0.33
L. goodei			3.58	3.21	0.03	-10.25	0.37
M. albus			3.86	3.74	0.36	-3.21	0.12
N. petersoni			3.72	3.5	0.18	-5.86	0.22
Ostracoda			2.69	2.12	0.1	-21.26	0.57
P. paludosus			2.98	2.62	0.05	-12.17	0.36
Amphipoda		Dry	2.14	2.46	0.77	14.91	-0.32
Hydrachnidia			2.67	2.47	0.25	-7.57	0.2
Chironomidae			2.87	2.1	0.08	-26.59	0.77
Copepoda			2.38	2.47	0.64	3.57	-0.09
G. holbrooki			3.5	3.21	0.16	-8.18	0.29
H. formosa			3.7	2.92	0	-21	0.78

L. goodei			3.3	3.36	0.63	1.85	-0.06
Ostracoda			2.25	2.29	0.57	1.87	-0.04
P. paludosus			3.25	3.55	0.89	9.4	-0.3

Species	Slough	Season	NP	Pond	NP < Pond	Percent Change	ΔTP
Coenagrionidae	SRS	Wet	2.82	2.52	0.11	-10.78	0.3
F. chrysotus			3.68	3.79	0.72	3.21	-0.11
G. holbrooki			3.63	3.63	0.54	0.17	0
H. letourneuxi			3.63	3.44	0.28	-5.15	0.19
H. formosa			3.66	3.84	0.88	4.84	-0.18
L. goodei			3.57	3.53	0.4	-1.32	0.04
P. paludosus			2.98	3.54	0.96	18.86	-0.56
P. femoratus			2.2	2.14	0.37	-2.82	0.06
P. fallax			2.61	2.26	0.06	-13.29	0.35
Procambarus spp.			2.67	2.51	0.06	-6.14	0.16
E. evergladei		Dry	3.37	3.34	0.44	-0.95	0.03
F. chrysotus			3.5	3.79	0.94	8.53	-0.29
G. holbrooki			3.39	3.68	0.99	8.4	-0.29
H. formosa			3.22	3.83	1	19.11	-0.61
L. marginatus			3.36	3.52	0.8	4.71	-0.16
L. goodei			3.06	3.44	1	12.4	-0.38
P. paludosus			2.91	3.16	0.96	8.77	-0.25
P. latipinna			3.01	3.35	0.96	11.24	-0.34
P. fallax			2.21	2.33	0.8	5.33	-0.12
Amphipoda	TSL	Wet	2.04	2.48	0.9	21.5	-0.44
B. gravida			2.66	2.22	0.08	-16.48	0.44
C. urophthalmus			3.67	3.32	0.06	-9.52	0.35
Copepoda			2.11	2.09	0.45	-0.85	0.02
G. holbrooki			3.17	3.53	0.93	11.15	-0.36
H. formosa			3.62	3.34	0.11	-7.65	0.28
L. goodei			3.07	3.21	0.73	4.56	-0.14
Ostracoda			2.1	2.12	0.56	1.1	-0.02
P. paludosus			2.32	2.62	0.9	12.84	-0.3
Amphipoda		Dry	2.37	2.46	0.59	3.93	-0.09
Hydrachnidia			2.33	2.47	0.63	5.79	-0.14
Copepoda			2.32	2.47	0.78	6.56	-0.15

B. Seasonal Comparisons

Species	Slough	Habitat	Wet	Dry	Wet < Dry	Percent Change	ΔTP
Amphipoda	SRS	Marsh	2.12	4.94	0.97	133.1	-2.82
Celithemus spp.			2.3	2.1	0.21	-8.5	0.2
C. urophthalmus			3.52	3.5	0.45	-0.6	0.02
E. evergladei			3.18	3.52	0.95	10.8	-0.34
E. gloriosus			3.71	3.31	0.11	-10.8	0.4
E. sucetta			3.46	3.29	0.33	-4.8	0.17
F. chrysotus			3.82	3.55	0.14	-6.9	0.27
G. holbrooki			3.56	3.28	0.05	-7.8	0.28
H. letourneuxi			3.63	3.57	0.36	-1.5	0.06
H. formosa			3.64	3.39	0.05	-7.0	0.25
J. floridae			3.21	3.25	0.55	1.3	-0.04
L. marginatus			3.35	3.48	0.68	3.8	-0.13
L. punctatus			3.96	3.6	0.07	-9.3	0.36
L. goodei			3.35	3.2	0.12	-4.4	0.15
P. paludosus			3.17	2.9	0.14	-8.4	0.27
P. femoratus			2.19	2.09	0.33	-4.3	0.1
Periphyton (Mat)			2.44	2.4	0.48	-1.6	0.04
E. evergladei		NP	3.49	3.37	0.21	-3.5	0.12
F. chrysotus			3.68	3.5	0.2	-4.9	0.18
G. holbrooki			3.63	3.39	0.02	-6.6	0.24
H. formosa			3.66	3.22	0	-12.1	0.44
L. goodei			3.57	3.06	0.01	-14.2	0.51
P. paludosus			2.98	2.91	0.3	-2.4	0.07
P. fallax			2.61	2.21	0.01	-15.2	0.4
A. natalis		Pond	3.67	3.87	0.82	5.3	-0.2
Amphipoda			2.2	2.26	0.64	2.6	-0.06
B. belizanus			4.16	4.06	0.36	-2.2	0.1
B. gravida			2.25	2.71	0.99	20.4	-0.46
Celithemus spp.			2.48	2.07	0.01	-16.5	0.41
C. urophthalmus			3.36	3.28	0.23	-2.4	0.08
Copepoda			2.04	2.64	1	29.6	-0.6
E. sucetta			3.2	3.47	0.93	8.5	-0.27
F. chrysotus			3.79	3.79	0.49	0.0	0
F. confluentus			3.42	3.37	0.39	-1.2	0.05
G. holbrooki			3.63	3.68	0.61	1.1	-0.05
H. formosa			3.84	3.83	0.49	-0.1	0.01
J. floridae			3.52	3.33	0.11	-5.2	0.19
L. gulosus			3.73	3.84	0.81	3.0	-0.11

L. marginatus			3.25	3.52	0.94	8.1	-0.27
L. microlophus			3.25	3.21	0.41	-1.4	0.04
L. platyrhincus			4.33	4.38	0.6	1.0	-0.05
L. punctatus			3.74	3.65	0.24	-2.6	0.09
L. goodei			3.53	3.44	0.23	-2.3	0.09
P. paludosus			3.54	3.16	0.07	-10.7	0.38
P. latipinna			3.49	3.35	0.16	-4.1	0.14
P. alleni			2.6	2.28	0.03	-12.5	0.32
P. fallax			2.26	2.33	0.64	3.1	-0.07
Amphipoda	TSL	Marsh	2.14	2.14	0.49	-0.2	0
Hydrachnidia			2.95	2.67	0.23	-9.7	0.28
Celithemus spp.			2.08	2.4	0.94	15.6	-0.32
Chironomidae			2.66	2.87	0.69	7.6	-0.21
Coenagrionidae			2.19	2.66	0.95	21.5	-0.47
Copepoda			2.88	2.38	0.14	-17.1	0.5
Corixidae			2.92	2.2	0.06	-24.4	0.72
E. gloriosus			3.63	3.83	0.72	5.5	-0.2
Ephemeroptera			2.12	2.43	0.83	14.8	-0.31
E. simplicicollis			2.46	2.27	0.39	-7.7	0.19
F. chrysotus			3.57	3.76	0.72	5.3	-0.19
G. holbrooki			3.47	3.5	0.53	0.9	-0.03
Gerridae			2.92	2.47	0.1	-15.4	0.45
H. formosa			3.67	3.7	0.54	0.6	-0.03
L. goodei			3.58	3.3	0.06	-7.7	0.28
Ostracoda			2.69	2.25	0.12	-16.4	0.44
P. paludosus			2.98	3.25	0.85	8.9	-0.27
P. femoratus			2.39	2.2	0.25	-8.0	0.19
Amphipoda		NP	2.04	2.37	0.92	15.8	-0.33
Hydrachnidia			2.74	2.33	0.11	-15.0	0.41
Cladocera			2.17	2.18	0.51	0.6	-0.01
Copepoda			2.11	2.32	0.91	9.7	-0.21
Amphipoda		Pond	2.48	2.46	0.5	-0.9	0.02
Copepoda			2.09	2.47	0.96	17.9	-0.38
G. holbrooki			3.53	3.21	0.07	-9.0	0.32
H. formosa			3.34	2.92	0.05	-12.6	0.42
L. goodei			3.21	3.36	0.78	4.7	-0.15
N. petersoni			3.5	3.5	0.48	-0.2	0
Ostracoda			2.12	2.29	0.72	8.2	-0.17
P. paludosus			2.62	3.55	0.99	35.6	-0.93

C. Slough Comparisons							
Species	Season	Habitat	SRS	TSL	SRS < TSL	Percent Change	ΔTP
Amphipoda	Wet	Marsh	2.12	2.14	0.55	1.1	-0.02
Belostoma spp.			2.44	2.11	0.09	-13.6	0.33
Celithemus spp.			2.3	2.08	0.09	-9.7	0.22
Chironomidae			2.65	2.66	0.49	0.5	-0.01
C. urophthalmus			3.52	3.58	0.61	1.8	-0.06
Coenagrionidae			2.41	2.19	0.16	-9.3	0.22
Copepoda			2.29	2.88	0.84	25.6	-0.59
E. gloriosus			3.71	3.63	0.4	-2.0	0.08
F. chrysotus			3.82	3.57	0.14	-6.5	0.25
G. holbrooki			3.56	3.47	0.29	-2.7	0.09
H. formosa			3.64	3.67	0.56	0.9	-0.03
L. marginatus			3.35	3.23	0.36	-3.5	0.12
L. goodei			3.35	3.58	0.89	7.0	-0.23
P. paludosus			3.17	2.98	0.24	-5.9	0.19
P. femoratus			2.19	2.39	0.75	9.5	-0.2
Coenagrionidae		NP	2.82	2.94	0.77	4.3	-0.12
E. evergladei			3.49	2.87	0	-17.7	0.62
F. chrysotus			3.68	3.47	0.25	-5.7	0.21
G. holbrooki			3.63	3.17	0	-12.5	0.46
H. formosa			3.66	3.62	0.36	-1.1	0.04
L. goodei			3.57	3.07	0.03	-14.0	0.5
Ostracoda			2.21	2.1	0.3	-5.2	0.11
P. paludosus			2.98	2.32	0.01	-22.1	0.66
P. femoratus			2.2	2.51	0.78	13.9	-0.31
Amphipoda		Pond	2.2	2.48	0.72	12.9	-0.28
B. gravida			2.25	2.22	0.44	-1.6	0.03
C. urophthalmus			3.36	3.32	0.4	-1.2	0.04
Copepoda			2.04	2.09	0.72	2.7	-0.05
G. holbrooki			3.63	3.53	0.47	-2.9	0.1
H. formosa			3.84	3.34	0.03	-12.9	0.5
L. macrochirus			3.34	2.9	0.01	-13.2	0.44
L. microlophus			3.25	3.32	0.7	2.1	-0.07
L. goodei			3.53	3.21	0.01	-8.9	0.32
M. siamensis			3.36	3.45	0.69	2.8	-0.09
M. salmoides			4.36	4.06	0.04	-6.8	0.3
M. albus			3.42	3.74	0.87	9.3	-0.32
N. petersoni			3.96	3.5	0.01	-11.6	0.46

P. paludosus			3.54	2.62	0.02	-26.1	0.92
P. latipinna			3.49	3.46	0.45	-0.9	0.03
T. mariae			3.19	3.13	0.32	-2.1	0.06
Amphipoda	Dry	Marsh	4.94	2.14	0.03	-56.7	2.8
Celithemus spp.			2.1	2.4	0.83	14.1	-0.3
E. gloriosus			3.31	3.83	0.9	15.8	-0.52
E. simplicicollis			2.41	2.27	0.38	-5.8	0.14
F. chrysotus			3.55	3.76	0.72	5.7	-0.21
G. holbrooki			3.28	3.5	0.76	6.5	-0.22
H. formosa			3.39	3.7	0.95	9.1	-0.31
L. goodei			3.2	3.3	0.79	3.2	-0.1
P. paludosus			2.9	3.25	0.92	11.8	-0.35
P. femoratus			2.09	2.2	0.72	5.2	-0.11
Amphipoda		Pond	2.26	2.46	0.67	9.0	-0.2
Hydrachnidia			2.37	2.47	0.62	4.2	-0.1
Copepoda			2.64	2.47	0.15	-6.6	0.17
G. holbrooki			3.68	3.21	0	-12.7	0.47
H. formosa			3.83	2.92	0	-23.8	0.91
L. goodei			3.44	3.36	0.33	-2.4	0.08
Ostracoda			2.39	2.29	0.33	-4.3	0.1
P. paludosus			3.16	3.55	0.95	12.2	-0.39

Figure S1. NMDS plot of consumers’ diets with species size classes in black and prey in red. Axis 1 and 2 (MDS 1 and MDS 2) were the basis of trophic niches we modeled using stomach contents.

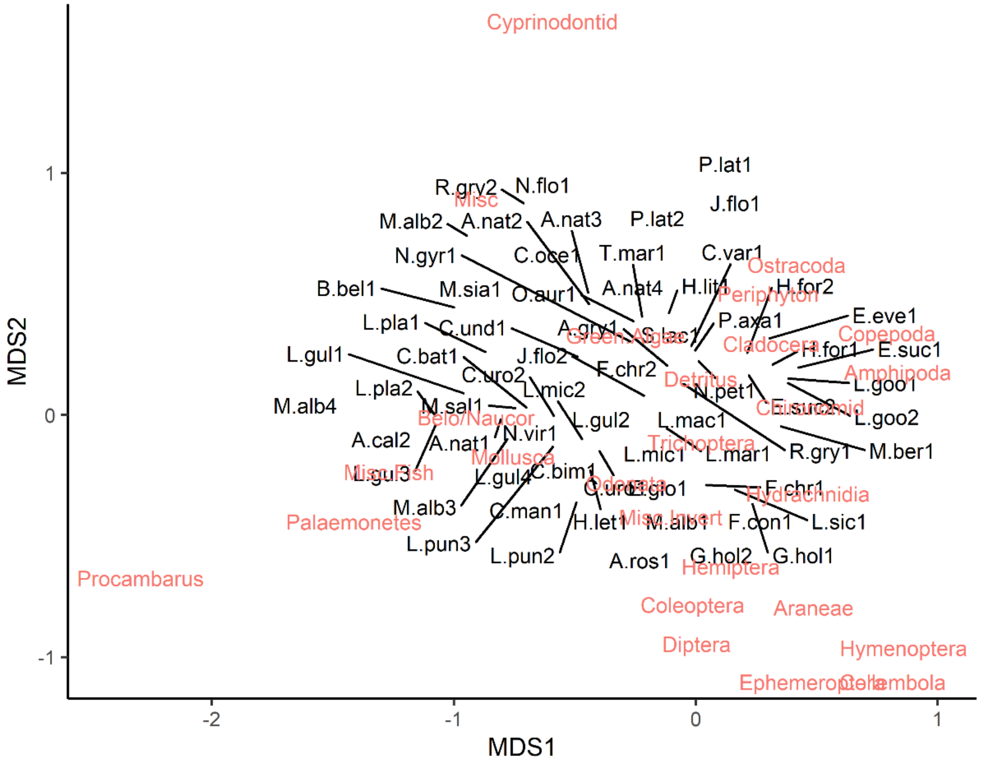
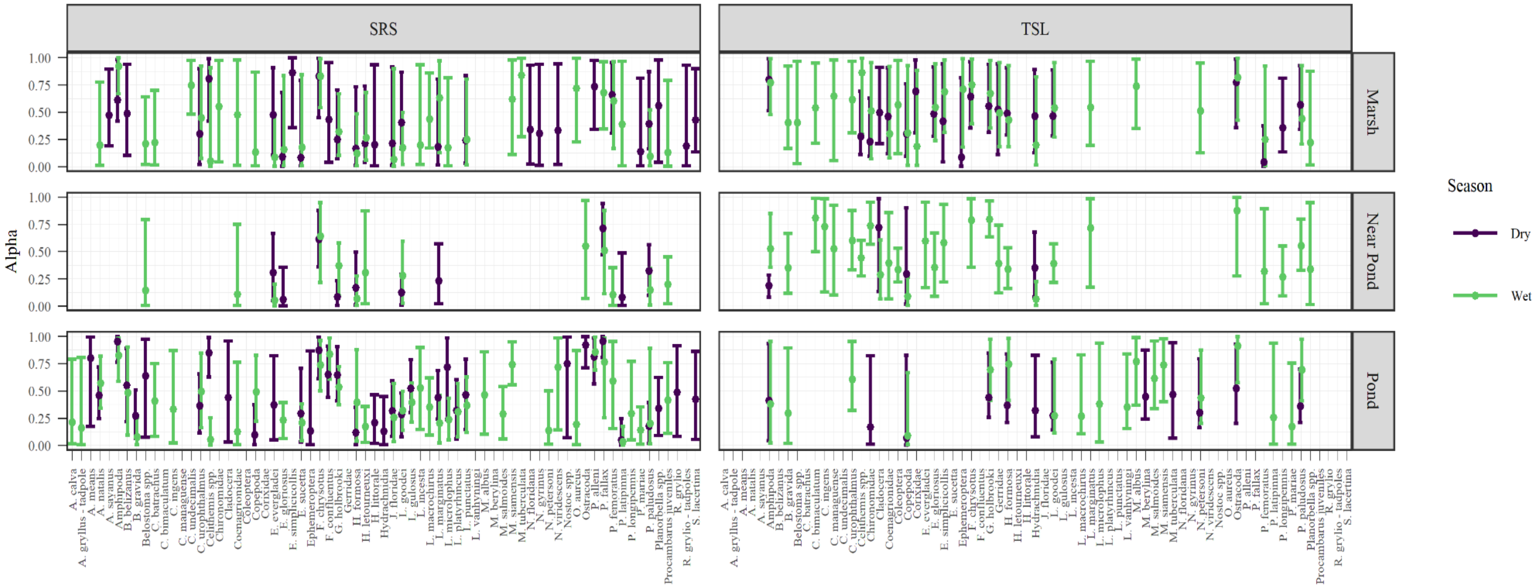


Figure S2. Alpha per taxa across an alligator-engineered habitat gradient between seasons.



CHAPTER IV

TROPHIC DISRUPTION BY AN INVASIVE SPECIES ALTERS SPATIOTEMPORAL FOOD-WEB DYNAMICS AND ENERGY FLUXES

Abstract

The Trophic Disruption Hypothesis (TDH) predicts that invasive species cause native species to undergo trophic dispersion (change in trophic niche size) and trophic displacement (diet switching) of native species in invaded communities, altering food-web structure and biodiversity. African Jewelfish (*Hemichromis letourneuxi*) underwent recent, drastic increases in density associated with decreased density of native taxa in Everglades National Park, USA. I used complementary datasets of stomach contents and nitrogen ($\delta^{15}N$) and carbon ($\delta^{13}C$) stable isotopes to quantify pre- and post-invasion consumer diets, trophic positions, trophic niches, and basal energy use (autotrophic vs heterotrophic) to test the TDH. Additionally, I calculated pre- and post-invasion energy fluxes among trophic guilds. My findings support the TDH. For native consumers, I observed trophic displacement in nearly half of comparisons pre- vs post-invasion, trophic dispersion for most comparisons, decreased trophic positions, and greater reliance on autotrophic energy. Post-invasion trophic shifts among habitats and between seasons occurred at a similar frequency but greater magnitude than pre-invasion. Post-invasion food-web structure and function revealed increased relative abundance of mesopredators (including African Jewelfish) and reduced biomass and energy fluxes into and out of small fishes (e.g., Cyprinodontiformes). In the Everglades, African Jewelfish invasion altered spatiotemporal trophic dynamics and energy fluxes through declines in native fishes and invertebrates, which have indirectly impacted trophic relationships at the regional scale. As a result, I suggest extending the TDH to explicitly include the potential for invasive species to alter basal energy use, spatiotemporal trophic dynamics, energy fluxes, and therein, food-web structure and function.

Introduction

Recent efforts have sought to assess ecosystem vulnerability and prioritize management strategies through our ability to understand how biological invasions alter trophic dynamics (Catford et al. 2012, McDonald-Madden et al. 2016, Strassburg et al. 2020). However, quantifying and forecasting trophic responses of ecosystems under invasion remains difficult because of compounding ecological variability through time (Theoharides and Dukes 2007, Polis and Winemiller 2013, David et al. 2017). While food-web ecologists have begun to appreciate the importance of trophic dynamics among habitats and seasons (McCann et al. 2005, McMeans et al. 2019, Chapter 2), spatiotemporal dynamics have largely been overlooked when studying the trophic effects of species invasions.

The recently formalized Trophic Disruption Hypothesis (THD; Wainright et al. 2021) provides a framework for testing the role of invasive species in trophic dynamics. TDH posits that invasive species disrupt food webs by causing native species to undergo trophic displacement (diet switching) and/or trophic dispersion (change in trophic niche size) (Vander Zanden et al. 1999, Cucherousset et al. 2012) without specific predictions about the direction of these effects. For example, consumers may become more generalist in response to invasive competitors or to avoid invasive predators (McMeans et al. 2016). Conversely, the Niche Variation Hypothesis states that interspecific competition, like from an invasive competitor, will decrease population niche areas (van Valen 1965, Bolnick et al. 2007). Similar trophic dynamics occur among habitats and seasons within an ecosystem (McCann et al. 2005, McMeans et al. 2019). Such heterogeneity generally results in increased food-web stability (Moore and de Ruiter 1991, Thompson and

Townsend 2005, Bellmore et al. 2015). It remains to be seen what is different about the frequency and magnitude of destabilizing trophic disruption by invasive species compared to similar, natural spatiotemporal trophic dynamics that are considered stabilizing, and if invasive species can disrupt spatiotemporal trophic dynamics.

Over the past several decades approximately one hundred species of fishes have been introduced into the freshwaters of Florida, eighteen of which have been found in our study system, Everglades National Park (ENP) (Kline et al. 2014). Recently, African Jewelfish (*Hemichromis letourneuxi*, hereafter "Jewelfish") have undergone drastic increases in density, coinciding with sharp declines in native fishes and invertebrates (Pintar, Dorn, Trexler pers. com.). Jewelfish have also demonstrated deleterious effects on native fauna in mesocosm experiments and ephemeral seasonal refuges (Rehage et al. 2014, Schofield et al. 2014). However, Jewelfish effects on Everglades freshwater food webs remain unknown.

In this study, we test the TDH premise that native consumers will undergo trophic dispersion and displacement and extend its predictions to include disruptions in trophic position, basal energy use, and energy flow. We compare frequency and magnitude of trophic disruption to trophic dynamics among habitats and between seasons (wet and dry) in ENP using both stable isotopes of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) and stomach contents. We take advantage of a "vanishingly rare" opportunity to compare post-invasion data to a robust pre-invasion data set (Strayer 2012, Flood et al. 2022). In the post-invasion food web, we predict that 1) native taxa will display trophic displacement by taking fewer prey that are also consumed by Jewelfish (Hickley and Bailey 1987, Rehage et al. 2014), 2) native taxa will undergo trophic dispersion quantified as increased trophic niche size to accommodate increased pressure on the same resource pool, 3) native consumers will rely primarily on detrital energy pre- and post-invasion (Williams and Trexler 2004, Belicka et al. 2012), 4) that post-invasion spatiotemporal trophic dynamics will occur at a similar frequency and magnitude as pre-invasion, and 5) there will be altered energy flow through the food web given the above-predicted changes in trophic structure and function. Coincident with Jewelfish invasion, Everglades restoration has altered water delivery to ENP since pre-invasion data were collected. Increased hydroperiod as a result of restoration is an alternative mechanism for changes in consumer communities and therein food webs. As a result, we quantify changes in hydrology and community composition to address potential confounding effects.

Methods

I compared pre- and post-invasion wet-season marsh food webs using stable isotopes, and wet- and dry-season food webs in marshes and ponds using gut contents. For post-invasion collection of stable isotope and stomach-content data, I selected five alligator ponds and adjacent marsh habitats in Shark River Slough (SRS) as sites to represent habitat heterogeneity in the Everglades. At each site, I sampled a pool or semi-open water habitat (hereafter "pond") and adjacent spikerush-dominated (*Eleocharis* spp.) marsh (hereafter "marsh"). I sampled each habitat at each site once to reflect the wet season (October–December 2018) and once during the dry season (March–April 2019), unless a site dried (Table S1). Pre-invasion stomach-content samples were collected in ponds and marshes from the same location (SRS) during both seasons between 1977 and 1981, while pre-invasion stable-isotope samples were collected from wet-season marshes in 1994. As a result, comparisons of frequency and magnitude of spatiotemporal changes pre- vs post-invasion are only made with counts of stomach contents to match pre-invasion data (Flood et al. 2022, Appendix S1). Post-invasion stable-isotope data were collected from all habitats in both seasons; those data were analyzed to compare pre- and post-invasion wet-season marshes (Appendix S1). Some vertebrate taxa (Table S2, Table S3) were divided into body-size groups (Table S4) to account for ontogeny in trophic dynamics (Wainwright and Richard 1995, Piet et al. 1999). Pre-invasion stable-isotope data were not size structured so comparisons are by species. To evaluate potential confounding effects between the pre- and post-invasion datasets, we examine Jewelfish density and biomass through time, hydroperiod (number of days per year with water depth ≥ 5 cm), and community composition of fishes and invertebrates (Appendix S1).

To explore trophic displacement, I quantified pre- and post-invasion spatiotemporal patterns in stomach contents of the entire consumer community. Non-metric multidimensional scaling (NMDS, $k = 2$) using Morisita-Horn distances (Jost et al. 2011) was used to visualize differences in community-wide prey consumption. We applied permutational multivariate analysis of variance (PERMANOVA; 10,000 iterations) followed by similarity percentages (SIMPER) to examine differences in the prey community for all consumers among habitat-season levels. Additionally, I contrasted diets for each species size class pre- vs post-invasion, and among habitat-season levels post-invasion with pairwise PERMANOVAs to examine frequency and magnitude of spatiotemporal diet shifts relative to pre-invasion. These analyses were conducted using the *vegan* package, *RVAideMemoire* package (*pairwise.perm.manova* function), and base R (Hervé 2021; Oksanen et al. 2022, R Core Team 2022).

To quantify trophic dispersion, trophic niches were modeled separately from counts of stomach contents and stable isotopes of carbon and nitrogen. Trophic niches were modeled using the *SIBER* package in R (Jackson et al. 2011). For stomach-content niches, in place of stable isotopes of carbon and nitrogen that are typically used with *SIBER*, I used axes one and two from NMDS (Flood et al. 2022). Differences among groups were determined using *pairwiseComparisons* function from *tRophicPosition* package (Quezada-Romegialli et al. 2018). Trophic niches compared in this manner pre- vs post-invasion for both isotopes and stomach contents, and among spatiotemporal variables for stomach contents.

To explore vertical trophic disruption, trophic positions were compared across habitats and seasons using stomach contents, as well as pre- and post-invasion using both stomach contents and stable isotopes. Trophic positions based on stomach contents were calculated according to Adams et al. (1983), where trophic position of a consumer is defined by trophic levels of its prey. Permuted analysis of variance (permuted ANOVA) followed by pairwise permutation tests were used to determine differences by size classes across habitats, seasons, and pre- vs post-invasion. These were performed in R using *aovp* and *pairwisePermutationTest* functions in *lmPerm* and *rcompanion* packages respectively (Wheeler and Torchiano 2016, Mangiafico 2021). For modeling trophic positions from stable isotopes, we used *tRophicPosition* package in R to calculate trophic positions derived from both δ^{13}C and δ^{15}N and alpha (Quezada-Romegialli et al. 2018). In our study, alpha is the proportion of δ^{15}N in a consumer's tissues derived from detritus (found in the flocculent layer). One minus alpha is the proportion derived from green algae. This creates a spectrum of detrital “brown” (alpha > 0.5) to algal “green” (alpha < 0.5) energy use by consumers. Trophic position and alpha were modeled using *multiSpeciesTP* function followed by the *pairwiseComparisons* function to explore variation pre- vs post-invasion.

To explore trophic disruption of food-web structure and function, we modeled energy flow in marshes pre- and post-invasion using diet relationships from this study and biomass from long-term monitoring data that starts in 1996 and continues to present (Appendix S1) (Trexler et al. 2003, 2010, Gatto et al. 2021). These long-term datasets include 1-m^2 throw-trap sampling of macroinvertebrates and fishes < 8 cm standard length (SL) (following methods from Jordan et al. 1997) and electrofishing of fishes ≥ 8 cm SL (Chick et al. 1999). Biomass was estimated from those data using length-weight relationships and converted to ash-free dry mass (AFDM) using known conversion factors (Kushlan et al. 1986, Benke et al. 1999, Brey 2001, Brey et al. 2010, Klassen et al. 2014). I built a graphical energy flux food web using *fluxweb* in R with node connections determined and quantified by biomass (in AFDM), metabolic losses, assimilation efficiency, and diet relationships (see Appendix S1 for details, Table S5) (Gauzens et al. 2019).

Results

Jewelfish Abundance, Hydrology, and Community Composition

Evaluation of potential confounding effects suggested a direct effect of Jewelfish invasion on trophic dynamics. At long-term monitoring sites in SRS, Jewelfish demonstrated boom-bust dynamics. Jewelfish density and biomass rapidly increased starting in 2012 (boom) and had largely declined (bust) prior to our sampling in 2018 and 2019 (Fig S1). Despite this decline in Jewelfish, multivariate analyses of community composition (Appendix S1) revealed that both fish and invertebrate communities were similar during the Jewelfish boom and our sampling (post-boom), while both boom and post-boom communities were dissimilar from pre-invasion communities (Fig. S2). Therefore, the fish and invertebrate communities when I sampled were indicative communities during the Jewelfish boom. Additionally, Permutated ANOVA demonstrated that annual hydroperiod did not differ between the Jewelfish boom, the period of my sampling, or pre-invasion for the sites we sampled in SRS (Table S5). The lack of change in hydroperiod suggests that community structure pre-invasion, during the boom, and during our sampling were driven by other factors, such as Jewelfish.

Trophic Displacement

Jewelfish invasion changed community-wide prey consumption (Table S6) with 42% of species size classes demonstrating trophic displacement (Table S7). Across all consumers, I saw reduced total consumption of some of the most abundant pre-invasion prey items including amphipods and copepods. An overall reduction in stomach fullness was also noted post-invasion. No post-invasion consumer averaged more than two individuals of any prey group per stomach, while prior to invasion at least one prey group did so in each habitat-season (Fig. S3). Post-invasion consumers were eating both fewer small prey items (e.g., copepods, amphipods, cladocerans) and fewer large prey items (e.g., fishes and decapods) (Fig. S3, S4). Dietary shifts post-invasion relative to pre-invasion were primarily driven by differential consumption of omnivorous invertebrates

(82% of pre- vs post-invasion comparisons), carnivorous invertebrates (45%), detritus (40%), and herbivorous invertebrates (32%).

Trophic Dispersion

Post-invasion trophic dispersion usually was revealed by decreased isotopic niche size but increased stomach content niche size. Based on stable isotopes, 46% of species underwent post-invasion trophic dispersion – 75% of which were decreases with a mean absolute value of percent change of 357 ± 499% (Table 1, Fig. S5). Meanwhile for niches derived from stomach contents, 54% of species size classes displayed post-invasion trophic dispersion – 73% of these shifts were increases with a mean percent change of 278 ± 524% (Table S8, Fig. 1).

Niches modeled with stomach contents showed a similar frequency but greater magnitude of spatiotemporal shifts post-invasion. Compared to pre-invasion, post-invasion changes between habitats were less frequent (pre = 70% of consumers had between habitat differences, post = 39%), but had a larger average magnitude of change (pre = 174 ± 354%, post = 210 ± 390%;) (Table S9, Fig S6). Conversely, post-invasion temporal shifts occurred at a similar frequency (pre = 51%, post = 54%), and had a lower average magnitude of change (pre = 118 ± 285%, post = 31 ± 196%). Overall, trophic dispersion that resulted from trophic disruption by Jewelfish occurred at the same frequency (54% of comparisons), but had more than double the average magnitude (131% vs 278%) than spatiotemporal trophic dispersion (Appendix S1). Post-invasion trends in trophic dispersion were either opposite or inconsistent with post-invasion seasonal trends – our proxy for restoration effects (Appendix S1).

Trophic Position

Post-invasion shifts in trophic position were relatively uncommon for both diet-tracing methods. I observed no apparent change in either frequency or magnitude of post-invasion spatiotemporal shifts in trophic position based on stomach contents. For the few post-invasion shifts that did occur, all but one were decreases of one-half to three-quarters of a trophic level (Table S11, S12, Fig. 2).

Basal Energy Use

Post-invasion, all consumer relied more on autotrophic energy (lower alpha) than pre-invasion. Six of fourteen species (43%) had statistical increases in their autotrophic (green) energy use by 78 ± 7% on average (Table S12, Fig. 2). This was opposite of the post-invasion trend from dry season to wet season, our proxy for restoration effects (Appendix S1). Post-invasion consumers increased their reliance on heterotrophic (brown) energy during the wet season relative to the dry season.

Trophic Disruption of Energy Flow

Food-web structure and function were disrupted after the Jewelfish invasion. Post-invasion biomass of small fishes decreased by 30% in the wet season and 58% in the dry season (Fig. 3, Appendix S1). Relative to small-fish biomass, mesopredator biomass saw a post-invasion increase during the wet season (pre = 70% of small-fish biomass; post = 576%) and a decrease during the dry season (pre = 631% of small-fish biomass; post = 135%). However, post-invasion during both seasons there was a higher proportion of total energy flux that went through the mesopredator node and a reduction through the small-fishes node (Fig 3, S9).

Discussion

Assessments of the impacts of invasive species on food webs often overlook the potential for invasion to lead to dietary shifts in native species that occur concurrently with changes in ontogeny and spatiotemporal variation in diet. In this study, I observed widespread post-invasion trophic displacement and dispersion in nearly half of native species size classes based on stomach contents. Trophic dispersion indicated by stable isotopes decreased in most species. Trophic disruption also was demonstrated by decreased trophic positions, increased reliance on autotrophic energy, increased magnitude of spatiotemporal trophic shifts, decreased biomass of small fishes with a corresponding increase in Jewelfish biomass, and altered energy fluxes. Analysis of potential confounding effects supported Jewelfish as a driver of the impacts I documented. Overall, my results demonstrate support for the TDH and extend trophic disruption to influencing spatiotemporal trophic dynamics and energy fluxes.

Trophic disruption by Jewelfish caused trophic dispersion at a similar frequency as spatiotemporal trophic dynamics, but to a greater magnitude. While spatiotemporal variation is stabilizing in food webs (McCann et al. 2005), stronger effects such as the trophic disruption I documented may be destabilizing. Spatiotemporal trophic disruption will likely be magnified by climate change subverting seasonal patterns that may alter spatiotemporal food-web dynamics from stabilizing to disruptive (Perkins et al. 2010). Post-invasion increased magnitude of spatiotemporal trophic dynamics, combined with reduced overall prey consumption (Fig. S1), suggests decreased prey availability, such as often-overlooked species at the bottom of the food web (Paine 1988). This may have yielded a reduction in the number of weak, stabilizing trophic interactions (McCann et al.

1998), as evidenced by decreased post-invasion isotopic niche areas, thus, destabilizing the food web.

Trophic dispersion and displacement occurred while trophic position remained constant. Previous researchers have stated that trophic dispersion implicitly includes variability in trophic position (vander Zanden et al. 1999, Wainright et al. 2021). In this study, trophic dispersion and displacement were both common, while altered trophic position was uncommon, with consequences for food-web structure and function. This decoupling of changes in trophic dispersion and displacement with trophic position highlights the need to explore multiple dimensions of an invader's impacts (Flood et al. 2020). For instance, if I had only examined trophic position (e.g., Cucherousset et al. 2012), I would have missed most of the trophic disruption in this study. Assessments of an invasive species' trophic impacts should include multiple metrics (e.g., diet, niche, and trophic position).

The magnitude of trophic dispersion was greater in invertivorous and omnivorous fishes relative to meso- and top predators, suggesting different mechanisms behind changes in trophic niche. Many cyprinodontiform fishes are simultaneously competing with Jewelfish for invertebrate prey (exploitative competition) while also being eaten by Jewelfish, particularly at earlier life stages. Centrarchids are likely experiencing exploitative competition and interference competition from Jewelfish, and other invasive cichlids, as a result of evolutionary convergence in ecomorphology, trophic ecology, and species in both families build and defend nests (Montana and Winemiller 2013). Exploitative competition is expected to result in the persistence of generalists at the expense of specialists (Holt et al. 1999). However, I observed large declines in abundance and biomass of abundant, generalist cyprinodontiform fishes that drove shifts in community composition post-invasion. This suggests that consumptive effects of invasive Jewelfish played a larger role in restructuring the aquatic community than either exploitative or interference competition.

I found that invasion of an omnivorous mesopredator, Jewelfish, altered energy flow in two major ways. First, native consumers relied more on autotrophic energy routes after invasion, and second, invasion rerouted energy that previously went through small fishes to moving through mesopredators. The mesopredator group contains multiple other invasive species besides Jewelfish. Based on changes in biomass and energy flux, mesopredators, including invasive cichlids, seem to be assuming the trophic role previously held by small fishes in the freshwater Everglades ecosystem similar to effects of invasive fishes found in other ecosystems (Wainright et al. 2021). It is likely that changes to ecosystem function, in combination with altered spatiotemporal food-web dynamics, have resulted in a new ecological regime driven by invasive species. This is perhaps unsurprising as invasive species (e.g., cichlids) in ENP are of tropical origin and evolved in systems with wet-dry season dynamics, unlike native taxa of temperate origin (Junk et al. 1989, Kocher 2004).

Ultimately, I found ubiquitous trophic disruption and increased magnitude of spatiotemporal trophic dynamics caused by an invasive fish. In this study, I provide evidence that Jewelfish have altered trophic dynamics and energy flow, which likely explains declines in native fishes and invertebrates seen in long-term data. Therefore, I suggest extending the TDH to state that invasive species can cause native species to undergo trophic dispersion, trophic displacement, shifts in basal resource use, altered spatiotemporal trophic dynamics, and disruption of food-web structure and function that given time will likely affect biodiversity. By incorporating this framework into future studies, ecologists can better understand and predict the overall impacts of biological invasions on ecosystem structure and function.

Acknowledgements

I would like to thank the technicians and volunteers who assisted with field and laboratory work, especially Bradley Strickland, Sarah Sisco, Maria Sabando, Ariana Jonas, and Venus Garcia. This project was funded by the U.S. National Park Service Critical Ecosystem Studies Initiative (CESI) under Cooperative Agreement P18AC01074 with Florida International University. This material is based upon work supported by the National Science Foundation through the Florida Coastal Everglades Long-Term Ecological Research program under Cooperative Agreement No. DEB-1832229 and DEB-2025954 through the FIU CREST Center for Aquatic Chemistry and Environment under Grant No. HRD-1547798 and HRD-2111661. Any opinions, findings, conclusions, or recommendations expressed in the material are my own and do not necessarily reflect the views of the National Science Foundation. I was supported by the Everglades Foundation as FIU ForEverglades Scholars. Research and animal procedures were conducted under the auspices of protocol #IACUC-18-067-CR01 from the Institutional Animal Care and Use Committee of the University of Florida and in accordance with scientific research permit #EVER-2017-SCI-0060 authorized by Everglades National Park and scientific collector’s permits #S-18-06 and #S-19-05 granted by Florida Fish and Wildlife Conservation Commission.

Literature Cited

Adams, S. M., B. L. Kimmel, and G. R. Ploskey. 1983. Sources of organic matter for reservoir fish production: a trophic-dynamics analysis. Canadian Journal of Fisheries and Aquatic Sciences 40:1480–1495.

Bellmore, J. R., C. v. Baxter, and P. J. Connolly. 2015. Spatial complexity reduces interaction strengths in the meta-food web of a river floodplain mosaic. Ecology 96:274–283.

Benke, A. C., A. D. Huryn, L. A. Smock, and B. J. Wallace. 1999. Length-mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. Page Am Benthol. Soc.

Bolnick, D. I., R. Svanbä, S. Araú Jo §, and L. Persson. 2007. Comparative support for the niche variation hypothesis that more generalized populations also are more heterogeneous.

Brey, T. 2001. Population dynamics in benthic invertebrates. A virtual handbook.

Brey, T., C. Müller-Wiegmann, Z. M. C. Zittier, and W. Hagen. 2010. Body composition in aquatic organisms - A global data bank of relationships between mass, elemental composition and energy content. Journal of Sea Research 64:334–340.

Catford, J. A., P. A. Vesk, D. M. Richardson, and P. Pyšek. 2012, January. Quantifying levels of biological invasion: Towards the objective classification of invaded and invasible ecosystems.

Chick, J. H., S. Coyne, and J. C. Trexler. 1999. Effectiveness of Airboat Electrofishing for Sampling Fishes in Shallow, Vegetated Habitats. North American Journal of Fisheries Management 19:957–967.

Cucherousset, J., S. Blanchet, and J. D. Olden. 2012. Non-native species promote trophic dispersion of food webs. Frontiers in Ecology and the Environment 10:406–408.

David, P., E. Thébault, O. Anneville, P. F. Duyck, E. Chapuis, and N. Loeuille. 2017. Impacts of Invasive Species on Food Webs: A Review of Empirical Data. Advances in Ecological Research 56:1–60.

Flood, P. J., A. Duran, M. Barton, A. E. Mercado-Molina, and J. C. Trexler. 2020. Invasion impacts on functions and services of aquatic ecosystems. Hydrobiologia 847:1571–1586.

Gatto, J. v., J. L. Kline, W. F. Loftus, and J. C. Trexler. 2021. Linking demographic transitions to population dynamics in a fluctuating environment. Canadian Journal of Fisheries and Aquatic Sciences 78:797–808.

Gauzens, B., A. Barnes, D. P. Giling, J. Hines, M. Jochum, J. S. Lefcheck, B. Rosenbaum, S. Wang, and U. Brose. 2019. fluxweb: An R package to easily estimate energy fluxes in food webs. Methods in Ecology and Evolution 10:270–279.

Hervé, M. 2021. RVAidMemoire: Testing and Plotting Procedures for Biostatistics.

Hickley, P., and R. G. Bailey. 1987. Food and feeding relationships of fish in the Sudd swamps (River Nile, southern Sudan). J. Fish Biol 30:147–159.

Jackson, A. L., R. Inger, A. C. Parnell, and S. Bearhop. 2011. Comparing isotopic niche widths among and within communities: SIBER - Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology 80:595–602.

Jordan, F., S. Coyne, and J. C. Trexler. 1997. Sampling Fishes in Vegetated Habitats: Effects of Habitat Structure on Sampling Characteristics of the 1-m 2 Throw Trap. Transactions of the American Fisheries Society 126:1012–1020.

Jost, L., A. Chao, and R. L. Chazdon. 2011. Compositional similarity and beta diversity. Pages 66–84 *in* A. E. Magurran and B. J. McGill, editors. Biological Diversity. Frontiers in Measurement and Assessment. Oxford University Press (OUP), Oxford, UK.

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The Flood Pulse Concept in River-Floodplain Systems. Canadian Special Publication of Fisheries and Aquatic Sciences Honey Harbour, Ontario, Canada 7:110–127.

Klassen, J. A., D. E. Gawlik, and B. A. Botson. 2014. Length-weight and length-length relationships for common fish and crayfish species in the Everglades, Florida, USA. Journal of Applied Ichthyology 30:564–566.

Kline, J. L., W. F. Loftus, K. Kotun, J. C. Trexler, J. S. Rehage, J. J. Lorenz, and M. Robinson. 2013. Recent fish introductions into Everglades National Park: An unforeseen consequence of water management? Wetlands 34:S175–S187.

Kobza, R. M., J. C. Trexler, W. F. Loftus, and S. A. Perry. 2004. Community structure of fishes inhabiting aquatic refuges in a threatened Karst wetland and its implications for ecosystem management. Biological Conservation 116:153–165.

Kocher, T. D. 2004, April. Adaptive evolution and explosive speciation: The cichlid fish model.

Kushlan, J. A., S. A. Voorhees, W. F. Loftus, and P. C. Frohring. 1986. Length, mass, and calorific relationships of Everglades animals. Florida Scientist 49:65–79.

Mangiafico, S. 2021. rcompanion: Functions to Support Extension Education Program Evaluation.

McCann, K., A. Hastings, and G. R. Huxel. 1998. Weak trophic interactions and the balance of nature. Nature 395:794–798.

McCann, K. S., J. B. Rasmussen, and J. Umbanhowar. 2005. The dynamics of spatially coupled food webs. Ecology Letters 8:513–523.

McDonald-Madden, E., R. Sabbadin, E. T. Game, P. W. J. Baxter, I. Chadès, and H. P. Possingham. 2016. Using food-web theory to conserve ecosystems. Nature Communications 7.

McMeans, B. C., T. Kadoya, T. K. Pool, G. W. Holtgrieve, S. Lek, H. Kong, K. Winemiller, V. Elliott, N. Rooney, P. Laffaille, and K. S. Mccann. 2019. Consumer trophic positions respond variably to seasonally fluctuating environments. Ecology 100:1–10.

McMeans, B. C., K. S. McCann, T. D. Tunney, A. T. Fisk, A. M. Muir, N. Lester, B. Shutter, and N. Rooney. 2016. The adaptive capacity of lake food webs: from individuals to ecosystems. Ecological Monographs 86:4–19.

Moore, J. C., and P. C. de Ruiter. 1991. Temporal and spatial heterogeneity of trophic interactions within below-ground food webs. Agriculture, Ecosystems and Environment 34:371–397.

Oksanen, J., F. Gullaume Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P. R. Minchin, R. B. O'Hara, G. L. Simpson, P. Solymos, M. H. M. Stevens, E. Szoecs, and H. Wagner. 2022. vegan: Community Ecology Package.

Paine, R. T. 1988. Road maps of interactions or grist for theoretical development? Ecology 69:1648–1654.

Perkins, D. M., J. Reiss, G. Yvon-Durocher, and G. Woodward. 2010. Global change and food webs in running waters. Hydrobiologia 657:181–198.

Piet, G. J., J. S. Pet, W. A. H. P. Guruge, J. Vijverberg, and W. L. T. van Densen. 1999. Resource partitioning along three niche dimensions in a size-structured tropical fish assemblage. Canadian Journal of Fisheries and Aquatic Sciences 56:1241–1254.

Polis, G. A., and K. O. Winemiller. 2013. Food webs: integration of patterns & dynamics. Springer Science & Business Media.

Quezada-Romegialli, C., A. L. Jackson, B. Hayden, K. K. Kahilainen, C. Lopes, and C. Harrod. 2018. tRophicPosition, an r package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods in Ecology and Evolution 9:1592–1599.

Rehage, J. S., S. E. Liston, K. J. Dunker, and W. F. Loftus. 2014. Fish community responses to the combined effects of decreased hydroperiod and nonnative fish invasions in a Karst wetland: Are Everglades solution holes sinks for native fishes? Wetlands 34.

Ruehl, C. B., M. R. Pintar, and J. C. Trexler. 2022. Predator-permanence hypothesis in time: Community dynamics in a seasonally flooded wetland. Freshwater Science 41.

Sarker, S. K., J. S. Kominoski, E. E. Gaiser, L. J. Scinto, and D. T. Rudnick. 2020. Quantifying effects of increased hydroperiod on wetland nutrient concentrations during early phases of freshwater restoration of the Florida Everglades. Restoration Ecology 28:1561–1573.

Schofield, P. J., D. H. Slone, D. R. Gregoire, and W. F. Loftus. 2014. Effects of a non-native cichlid fish (African jewelfish, Hemichromis letourneuxi Sauvage 1880) on a simulated Everglades aquatic community. Hydrobiologia 722:171–182.

Spear, M. J., J. R. Walsh, A. Ricciardi, and M. J. vander Zanden. 2021. The Invasion Ecology of Sleeper Populations: Prevalence, Persistence, and Abrupt Shifts. BioScience 71:357–369.

Strassburg, B. B. N., A. Iribarrem, H. L. Beyer, C. L. Cordeiro, R. Crouzeilles, C. C. Jakovac, A. Braga Junqueira, E. Lacerda, A. E. Latawiec, A. Balmford, T. M. Brooks, S. H. M. Butchart, R. L. Chazdon, K. H. Erb, P. Brancalion, G. Buchanan, D. Cooper, S. Díaz, P. F. Donald, V. Kapos, D. Leclère, L. Miles, M. Obersteiner, C. Plutzar, C. A. Carlos, F. R. Scarano, and P. Visconti. 2020. Global priority areas for ecosystem restoration. Nature 586:724–729.

Strayer, D. L. 2012. Eight questions about invasions and ecosystem functioning. Ecology Letters 15:1199–1210.

Theoharides, K. A., and J. S. Dukes. 2007, October. Plant invasion across space and time: Factors affecting nonindigenous species success during four stages of invasion.

Thompson, R. M., and C. R. Townsend. 2005. Food-web topology varies with spatial scale in a patchy environment. Ecology 86:1916–1925.

Trexler, J. C., W. F. Loftus, and J. Chick. 2003. Setting and monitoring restoration goals in the absence of historical data: the case of fishes in the Florida Everglades. Pages 351–376 *in* D. Busch and J. C. Trexler, editors. Monitoring Ecoregional Initiatives: Interdisciplinary Approaches for Determining Status and Trends of Ecosystem. Island Press, Washington, D.C.

Trexler, J. C., W. F. Loftus, F. Jordan, J. H. Chick, K. L. Kandl, T. C. McElroy, and O. L. Bass Jr. 2010. Ecological scale and its implications for freshwater fishes in the

Florida Everglades. Pages 151–176 The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: An Ecosystem Sourcebook. CRC Press, Boca Raton, FL.

van Valen, L. 1965. Morphological Variation and Width of Ecological Niche. Source: The American Naturalist 99:377–390.

Wainright, C. A., C. C. Muhlfeld, J. J. Elser, S. L. Bourret, and S. P. Devlin. 2021. Species invasion progressively disrupts the trophic structure of native food webs. Proceedings of the National Academy of Sciences of the United States of America 118.

Wainwright, P. C., and B. A. Richard. 1995. Predicting patterns of prey use from morphology of fishes. Environmental Biology of Fishes 44:97–113.

Wheeler, B., and M. Torchiano. 2016. lmPerm: Permutation Tests for Linear Models.

Vander Zanden, M. J., J. M. Casselman, and J. B. Rasmussen. 1999. Stable isotope evidence for the food web consequences of species invasions in lakes. Nature 401:464–467.

Table 1. Trophic dispersion based on shifts in trophic niche area modeled using stable isotopes of carbon and nitrogen. Pre-Invasion and Post-Invasion are the trophic niche areas for those time periods respectively. Pre < Post is the probability that the pre-invasion trophic niche area is less than the post-invasion trophic niche area. Direction indicates whether trophic niche area increased, decreased, or did not change and %Δ is the percent change from pre-invasion to post-invasion. Statistical differences in Pre < Post are in bold. Non-native fishes are denoted with *.

Species	Pre-Invasion	Post-Invasion	Pre < Post	Direction	%Δ
A. natalis	1.14	0.81	0.36	No Change	-28.9
Anisoptera	2.01	1.63	0.34	No Change	-19.0
Belostoma spp.	2.03	3.12	0.74	No Change	53.5
*C. urophthalmus**	0.08	1.14	**1**	Increase	1270.9
E. sucetta	4.29	0.49	**0**	Decrease	-88.5
F. chrysotus	2.28	0.1	**0**	Decrease	-95.8
G. holbrooki	0.62	7.13	**1**	Increase	1047.0
H. formosa	2.25	1.01	0.14	No Change	-55.0
L. macrochirus	1.08	0.29	**0.02**	Decrease	-73.6
L. microlophus	1.46	0.07	**0**	Decrease	-95.3
L. punctatus	1.92	1.2	0.27	No Change	-37.6
P. paludosus	4.43	0.61	**0**	Decrease	-86.2
P. latipinna	2.43	4.76	0.81	No Change	95.8
P. fallax	45.05	1.45	**0**	Decrease	-96.8

Figure 1. Frequency of effect sizes of trophic dispersion based on stomach contents for different habitats and seasons. Taxa that did not display trophic dispersion are not included. For taxa that underwent trophic dispersion, absolute value of mean percent change was 325% ± 491% and ranged from 39% to 2,141%.

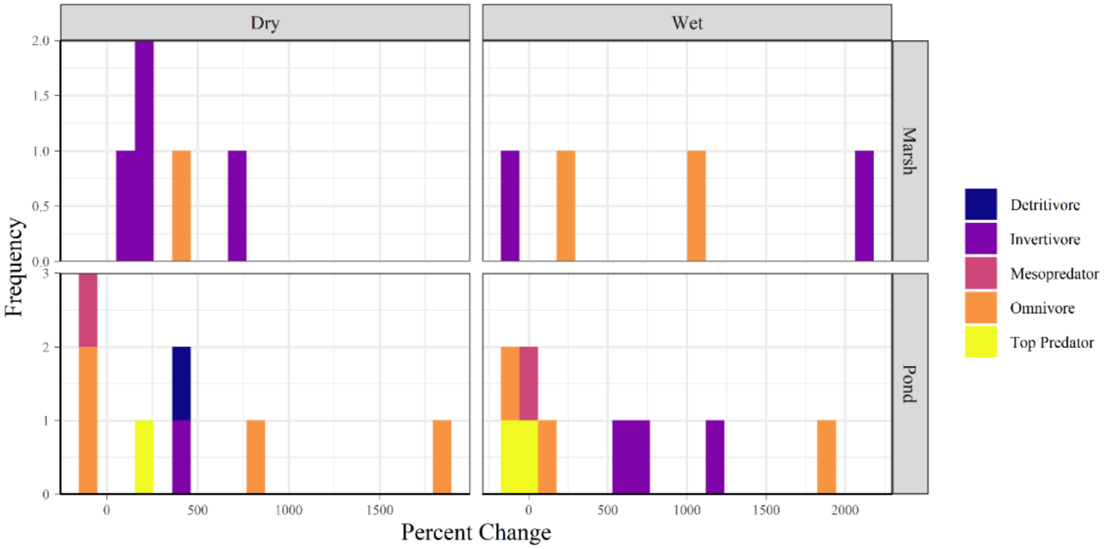


Figure 2. Pre- versus post-invasion A) trophic position and B) alpha derived from stable isotopes.

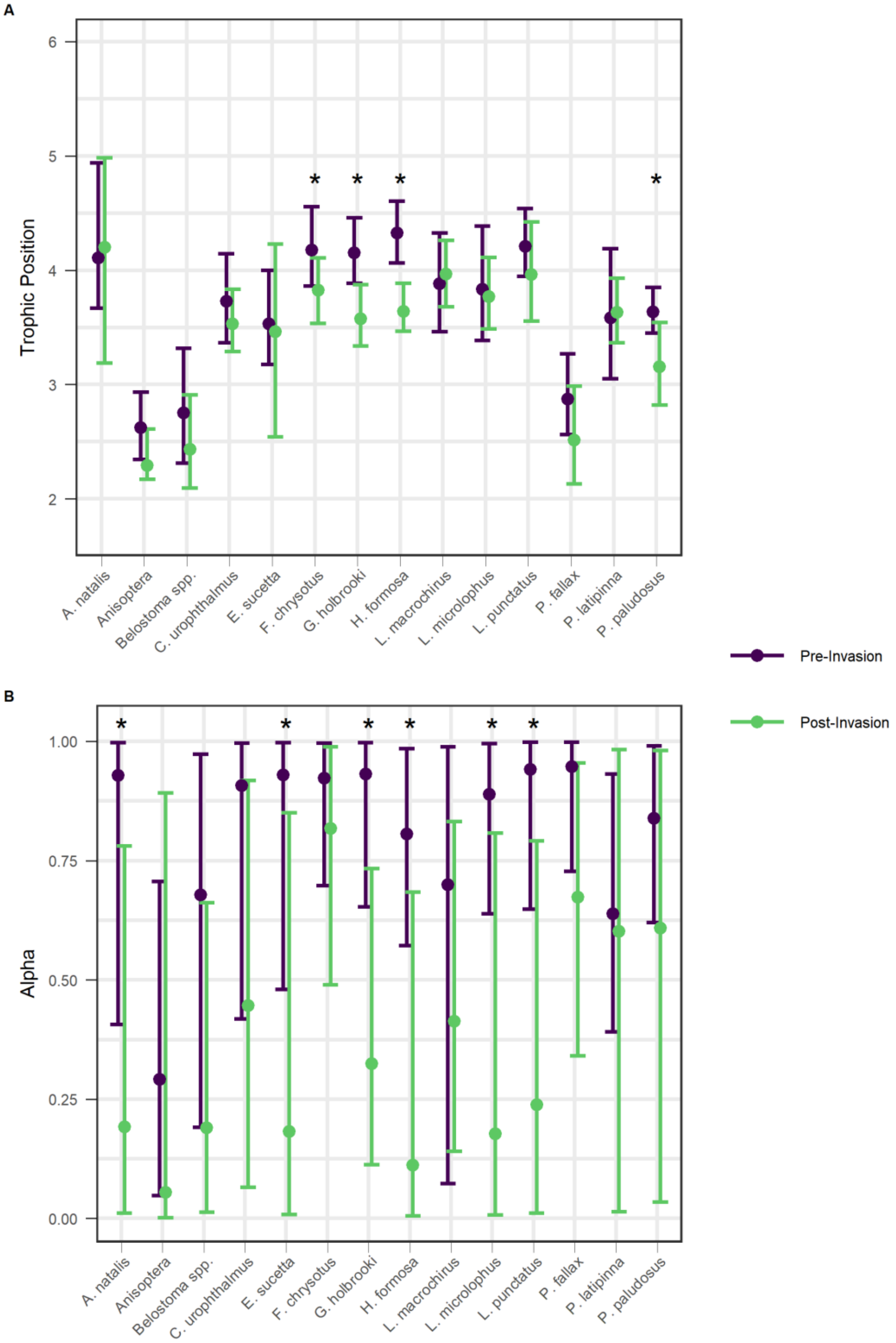
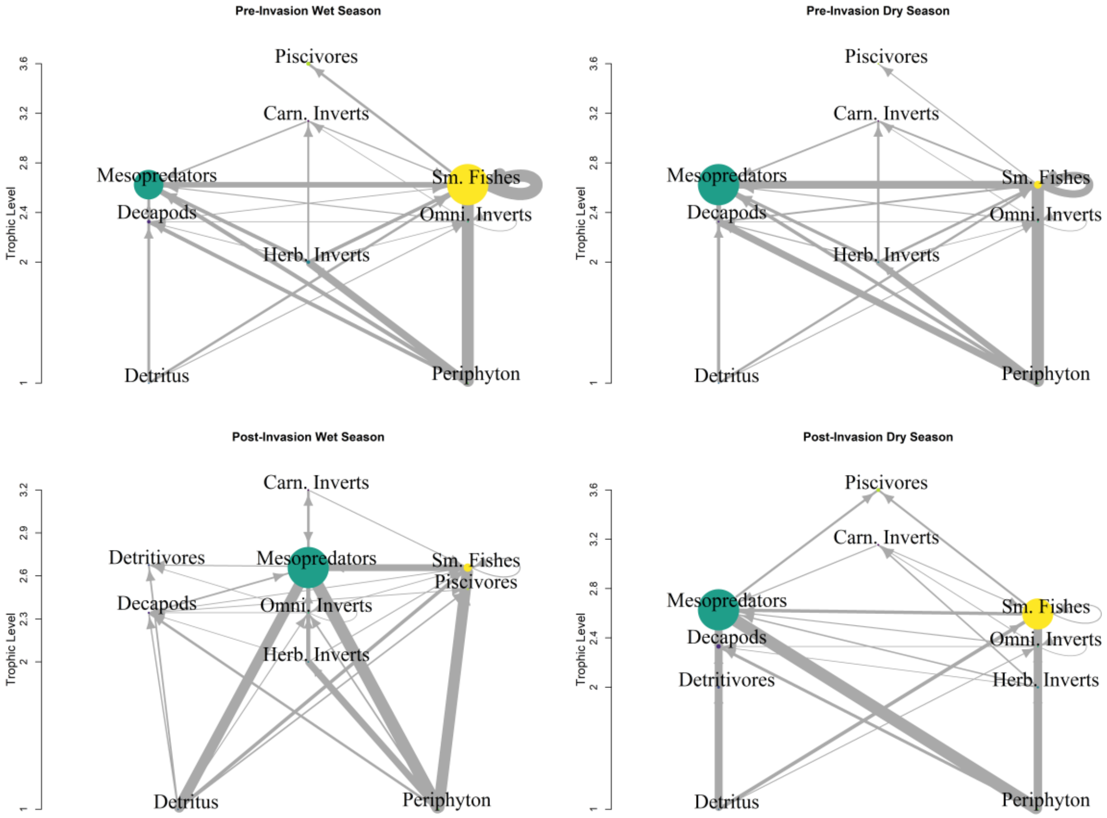


Figure 3. Energy flow among guilds (nodes) pre- vs post-invasion for mashes in both seasons. Nodes are g AFDM · m-^{2} and fluxes (arrows) are J · s^{-1} · m^{-2}.



Pre-Invasion Wet Season

Pre-Invasion Dry Season

Post-Invasion Wet Season

Post-Invasion Dry Season

Appendix

Methods

Field Collections

Basal resources in the form of vascular plants, *Utricularia* spp., *Nostoc* spp., periphyton, and flocculent organic benthic matter (hereafter "floc") were collected when present at each habitat, in each season, and at each site. Floc samples were taken from three different locations within each habitat and aggregated into a single sample. For vascular plants, samples were taken from leaves and mid-stems of multiple individuals of dominant taxa to create an aggregate sample per habitat, per season, per site. Similarly, *Utricularia* spp., *Nostoc* spp., and periphyton (both mats and epiphyton), when present, were taken from multiple locations within a habitat to create an aggregate sample.

Consumers were collected through a variety of methods to maximize number of taxa and individuals per taxon. Brakke's (1976) modified Whiteside-Williams (1975) samplers (hereafter "funnel traps") were deployed across the study design (when permitted by water levels) to collect small, benthic invertebrates (e.g., copepods, ostracods, cladocerans, etc.). Each funnel trap consisted of a 4 x 4 array of funnels for a total of sixteen per trap that resulted in 1 L of water being collected per funnel trap. Funnel traps were placed on top of the sediment for 24-hours to capture invertebrates during diel migration. These samples were stored on ice for transport and then refrigerated until processing within 48 hr. Organisms were sorted under a dissecting microscope and identified to taxonomic level (i.e., order, class, family).

Three, one-m^2 throw-trap samples were performed per habitat (pond and marsh) at each site (six total throws per site). When water depth was < 5 cm or > 1 m, we did not sample because throw traps are ineffective at these depths (Jordan et al. 1997). This led to twenty-two throw-trap samples in the wet season and twenty-seven in the dry season for forty-nine total (Table S2). Additionally, I used minnow traps, dip nets, drift fences, and electrofishing to collect consumers. Consumers were euthanized via a lethal dose of MS-222 and frozen. In the laboratory, consumers were identified to species, counted, measured, weighed, and sexed (Table S3). Vertebrates were sorted into size classes based on length (Table S4) and size classes will be denoted in parentheses after the common name in the text with one being the smallest size class. For example, juvenile Eastern Mosquitofish (*Gambusia holbrooki*) will be denoted as Eastern Mosquitofish (1).

Laboratory Analyses

Samples were processed for both stable isotopes and stomach contents. For vertebrates, gastrointestinal tracts were removed and stored in 95% ethanol for further analysis. The remainder for vertebrates and the entire individual for invertebrates and basal resources was freeze-dried, crushed, and weighed in tin capsules for isotopic analysis. For larger consumers (e.g., Florida Gar – *Lepisosteus platyrhincus*) we physically removed bone and larger pieces after grinding in a ball mill. Periphyton samples were acid-washed in HCl before drying to remove carbonates. Basal resources and small invertebrate taxa that lacked enough mass to be processed as individuals were made into aggregate samples. All other isotope samples represent individual organisms. Most samples were analyzed at Florida International University Stable Isotope Lab, while samples from small invertebrates were sent to Duke University Environmental Stable Isotope Lab (DEVIL). DEVIL used international isotopic standards USGS 26 and USGS

40 in addition to internal standards of Costech acetanilide and Duke sucrose. External precision relative to reference materials was approximately ± 0.1 ‰ at one standard deviation. FIU isotopic standards for $\delta^{13}C$ and $\delta^{15}N$ were Pee Dee Belemnite and atmospheric air respectively. Average isotopic error of replicate standards was ± 0.4 for $\delta^{13}C$ and ± 0.3 for $\delta^{15}N$ at FIU.

Stomach contents of vertebrates were analyzed by flushing contents, identifying individuals under a microscope to the lowest possible taxonomic level or category, for example detritus (Table S3, S4). For species without a defined stomach, contents from the esophagus to the first bend in the gastrointestinal tract were analyzed. Fragmented and unidentifiable remains were termed “Miscellaneous Fish”, “Miscellaneous Invertebrate”, or “Miscellaneous” depending on quality and type of remains. Stomach contents were quantified using both volumetric and numeric approaches (Hyslop 1980, Bowen 1983). Statistical analyses were performed on numeric data to match analyses performed on pre-invasion data (Flood et al. 2022).

Statistical Analyses – Bayesian Model Specifications

SIBER models of trophic niches were run using 2,000,000 iterations, burn-in of 10,000, thin of 100, and 2 chains (Jackson et al. 2011). Meanwhile, *tRophicPosition* models of trophic positions were run using 10,000 iterations, burn-in of 1,000, thin of 10, and 2 chains (Quezada-Romegialli et al. 2018). Both sets of models used null (uninformative) priors recommended by package authors.

Statistical Analyses – Hydrology, Jewelfish Abundance, and Community Composition

To evaluate potential confounding effects when comparing pre- and post-invasion food webs, statistical comparisons of hydrology and community composition were performed among different time periods. Time periods refer to when different data were collected and the monitoring regime at the time. Time periods are as follows: “Pre-Stomachs” – 1978-1981 (when pre-invasion stomach content data were collected), “Pre-MDW” – 1982-1989 (indicates sampling prior to the current iteration of the long-term monitoring project), “Pre-Isotopes” – 1990-1995 (when the pre-invasion stable isotopes samples were collected, also prior to the current iteration of the monitoring project), “Pre-Invasion” – 1996-1999 (years immediately preceding Jewelfish invasion, current monitoring regime began in 1996), “Pre-Boom” – 2000-2011 (period of time from Jewelfish invasion in 2000 up until the increase in their population), “Boom” – 2012-2017 (years of elevated Jewelfish relative biomass and abundance), “Sampling” – 2018-2019 (when sampling for this project occurred), “Post-Boom” – 2020-2021 (years after sampling for this project while Jewelfish were at low levels of relative abundance and biomass).

To compare hydrology through time, permuted ANOVA was used to determine if there were changes in the mean annual hydroperiod (days per year with water depth ≥ 5 cm) among time periods. Permuted ANOVA was performed using *aovp* function following by a pairwise permutation test (functionally similar to Tukey’s HSD) via *pairwisePermutationTest* function from the *rcompanion* package. To account for changes in community composition through time, communities were visualized with NMDS and statistically compared via PERMANOVA (permutational multivariate analysis of variance). Multivariate analyses were conducted using the *vegan* package in R.

Statistical Analyses – Post-Invasion Spatiotemporal Dynamics from Stable Isotopes

To address my hypothesis that restoration effects would be similar to (in the same direction as) effects going from dry season to wet season post-invasion, we made comparisons for trophic niches and trophic positions within a habitat between season (i.e. pond-dry vs pond-wet and marsh-dry vs marsh-wet) using the post-invasion data. Pre-invasion stable-isotope data are only from wet-season marshes so these comparisons were not possible using the pre-invasion stable-isotope dataset. Differences among groups were determined using the *pairwiseComparisons* function from the *tRophipcPosition* package (Quezada-Romegialli et al. 2018). This function determines if the probably of one posterior distribution (A) is greater than or less than another (B). If $p(A \geq B) \geq 0.95$, that was determined to be a statistical increase for that metric (either trophic position or trophic niche). Similarly, if $p(A \geq B) \leq 0.05$ that was determined to be a statistical decrease for that metric because the opposite one-tailed test (i.e. $p(A \leq B)$) had a probability $\geq 95\%$.

Statistical Analyses – Energy Fluxes

Energy fluxes among functional groups were derived from counts of animals and volume of periphyton from long-term monitoring data (Trexler et al. 2002, 2003, Gatto et al. 2021). Abundance estimates for small fishes (< 8 cm standard length) and macroinvertebrates (all retained on a 2-mm sieve) were obtained by sampling with a 1-m^2 throw trap following methods in Jordan et al. (1997). Abundance of large fishes (> 8 cm standard length) were obtained via airboat-mounted electrofisher (Chick et al. 1999). We extracted pre-invasion data from 1998 during the months of July, October, and December (wet season) and 1999 during the months of February and April (dry season). This is twenty years prior to the post-invasion diet data collections and the final water year (wet-dry cycle) prior to the Jewelfish invasion. Post-invasion data were extracted from the same months of 2018 and 2019 respectively to coincide with contemporary diet data collection. Rare taxa with counts of less than 25 individuals summed across all extracted data (i.e., pre- and post-invasion counts) were dropped from analyses.

I estimated ash-free dry mass (AFDM) of periphyton, detritus, and animals. For periphyton, the relationship between the volume collected per m^2 (i.e., per 1-m^2 throw trap sample) to AFDM per m^2 was converted using an equation derived from periphyton samples from Shark River Slough that had been processed to AFDM as part of the Comprehensive Everglades Restoration Plan (CERP) Monitoring Assessment Plan (MAP) (Evelyn Gaiser pers comm). Detrital volumes were not collected as part of these monitoring projects. AFDM of detritus was set to 10 g/m^2 as a conservative (under) estimate to saturate the amount of detritus (prevent detritus from being limiting) for calculating energy fluxes. For comparison, the lowest periphyton AFDM was 64 g/m^2.

For animals, in some cases counts were converted directly to biomass (wet weight) based on pre-existing, Everglades-specific conversion factors (Kushlan et al. 1986, Klassen et al. 2014). For the remaining animals, average lengths were calculated based on field collections (Kline unpublished) and conversion factors were extracted from the literature (Benke et al. 1999, Edwards et al. 2009, Atkinson et al. 2020) (Table S5). Based on these, counts were converted into wet weights. All wet weights were then converted to AFDM based on known conversion factors in the literature (Brey 2001, Brey et al. 2010) (Table S6). Counts of small fishes (length ≤ 8 cm) and invertebrates from throw traps are inherently per m^2 (the size of the trap). For electrofishing data, transect areas were not recorded. Average transect area was estimated to be 1,000 m^2 (conservatively an overestimate) based on measured transect lengths (mean = 221.05 m) and the sampleable width to either side of the airboat (2.5 m) from a different project using the same protocol (Janelle Goeke pers. comm.).

Metabolic losses and assimilation efficiencies were estimated based on consumer or prey type respectively. Metabolic rates were calculated per taxa (usually species) based on allometric body-mass metabolic relationships (Brown et al. 2004). Metabolic losses were then averaged per functional group. Assimilation efficiencies were based on prey type (animal prey = 0.906; detritus and periphyton = 0.4) (Gergs and Rothhaupt 2008, Gauzens et al. 2019).

This analysis relies on a lot of assumptions. That is why we grouped taxa by functional groups and interpret the results as relative change, not as absolute representations of the exact energy fluxes. Additional information about calculation of energy fluxes using the *fluxweb* package can be found at https://rfrelat.github.io/BalticFoodWeb.html (Kortsch et al. 2021).

Results

Trophic displacement

Trophic displacement was demonstrated in 42% of species size classes, including most size classes of the most abundant fish taxa, such as Eastern Mosquitofish, Bluefin Killifish, Golden Topminnows, and Least Killifish across habitat-season levels. Trophic displacement was more common in smaller size classes of Cyprinodontiform fishes (55% of comparisons). In contrast, trophic displacement was only observed in larger size classes of larger fishes (e.g., Centrarchids and *Ameiurus natalis* – Yellow Bullhead). The most abundant diet items in consumer's guts across all habitat-season levels both pre- and post-invasion were mainly meiofauna such as amphipods, cladocerans, chironomid larvae, copepods, and ostracods (Fig. S3, S4). Post-invasion, SIMPER revealed that mollusks (mostly aquatic snails) were the most abundant prey item in guts of wet-season consumers in both ponds and marshes – over a threefold increase from pre-invasion (Fig. S4) and mollusks were an important diet item for Jewelfish. Pre-invasion, the most abundant prey taxa were chironomid larvae in marshes during both seasons, copepods in wet-season ponds, and grass shrimp (*Palaemonetes paludosus*) in dry-season ponds. Both SIMPER and NMDS demonstrated a post-invasion reduction in consumption of meiofauna and larger prey items (i.e., fishes, crayfish, and grass shrimp).

Trophic Dispersion

For trophic niches based on stomach contents, pre-invasion to post-invasion shifts in trophic niche size were usually increases (73% of shifts), while wet season to dry season shifts were usually decreases (54%). Trophic dispersion post-invasion was somewhat more common in the dry season (50%) than the wet season (43%) and similar between habitats within a season (dry-season pond = 50%; dry-season marsh = 50%; wet-season pond = 45%; wet-season marsh = 40%). Mean percent change in trophic niche size pre- vs post-invasion was 325% ± 491% ranging from 2% to 2,141% (Table S8).

Percent changes in trophic niche size exceeding 500% were found in Eastern Mosquitofish (1), Least Killifish (1, 2), Bluefin Killifish (1), and Dollar Sunfish (1).

Trophic dispersion modeled with stable isotopes was more likely to occur post-invasion compared to pre-invasion than among spatiotemporal factors post-invasion (i.e., habitat and season). There were shifts in trophic niche size in approximately 43% of comparisons for each spatiotemporal factor (Table S10). Meanwhile, pre- versus post-invasion comparisons resulted in a shift in trophic niche size for 57% of comparisons (Table 1). Eastern Mosquitofish and Mayan Cichlids had post-invasion trophic niches over ten times larger than pre-invasion. Increases in trophic niche size were more common, occurring in six of fourteen species for which we had pre- and post-invasion stable isotope data. Bluegill Sunfish, *Lepomis macrochirus*, and Redear Sunfish decreased in trophic niche size post-invasion. Redear Sunfish, well known molluscivores, had the largest decrease (95%) in trophic niche size. Post-invasion there were an equal number of seasonal increases and decreases in trophic niche size.

Trophic Position

Based on stomach contents, shifts in trophic position post-invasion compared to pre-invasion and spatiotemporal shifts both pre- and post-invasion were uncommon. Permuted analysis of variance revealed post-invasion trophic position shifts within a habitat-season level for three species size classes (7%). These were Mayan Cichlids (*Mayaheros urophthalmus*,1), Golden Topminnows (*Fundulus chrysotus,* 1), and Everglades Pygmy Sunfish (*Elassoma evergladei*, 1). Mayan Cichlids (1) decreased by an entire trophic level, Golden Topminnows (1) decreased by half a trophic level, and

Everglades Pygmy Sunfish (1) increased by half a trophic level. Post-invasion there were three species size classes with spatial shifts in trophic position and three species size classes with temporal shifts in trophic position (Table S14).

Trophic positions derived from stable isotopes shifted more frequently as a result of trophic disruption (29% of comparisons) than as a result of spatiotemporal dynamics (habitat – 15%; season – 21%). Pre- versus post-invasion trophic positions either decreased or had no statistical difference (Fig. 2A, Table S15). There were post-invasion statistical decreases in trophic position for four species: Golden Topminnows, Eastern Mosquitofish, Least Killifish, and grass shrimp. These decreases were on the order of one-half to three-quarters of a trophic level. In the post-invasion data, we documented dry-season decreases in trophic position relative to the wet season (although not necessarily a statistical difference) for each of these taxa.

Basal Energy Use

Post-invasion consumers relied more on autotrophic energy (lower alpha) than pre-invasion consumers. In the post-invasion data, we observed elevated use of heterotrophic energy (higher alpha) in the wet season relative to the dry season. Pre- versus post-invasion shifts in alpha occurred in 43% of comparisons, roughly five times as often temporal shifts (season – 8%) and over twenty times as often as spatial shifts (2%) (Fig. S8, Table 1, S13, S16). Alpha decreased or had no statistical difference for all taxa post-invasion (Fig. 3B, Table S13). Yellow Bullhead Catfish (*Ameiurus natalis*), Lake Chubsuckers (*Erimyzon sucetta*), Eastern Mosquitofish, Least Killifish, Redear Sunfish (*Lepomis microlophus*), and Spotted Sunfish all used more autotrophic energy (lower alpha) post-invasion. The magnitude of these changes ranged from 66% (Eastern Mosquitofish) to 87% (Least Killifish) (Table S13). These same species all decreased in alpha (although not always a statistical difference) in the wet season relative to the dry season in the post-invasion data (Fig. S8, Table S11). This was the opposite of the trend we observed when comparing pre- and post-invasion alpha.

Trophic Disruption of Energy Flow

Post-invasion during both seasons there were reductions in biomass of small fishes and the proportion of energy fluxes that went through the small-fishes node. Post-invasion biomass reductions of small fishes were -30% during the wet season and -58% during the dry season (Fig. 3, Table S17). Post-invasion mesopredator biomass increased by 470% during the wet season, yet decreased by 91% during the dry season. Meanwhile, pre-invasion outflux of energy from the small-fishes node was 41% of the total energy flux during the wet season and 27% during the dry season as opposed to 8% and 3% during the wet and dry seasons respectively post-invasion (Fig. 3, S7A). Changes in energy influx to the small-fishes and mesopredator nodes were even more pronounced. During the wet season, influx to the small-fishes nodes went from 70% pre-invasion to 27% post-invasion, while during the dry season that influx went from 52% to 25% (Fig. S9B). Conversely, during the wet season influx to the mesopredator node went from 14% pre-invasion to 62% post-invasion, while during the dry season that influx went from 25% to 70% from pre-invasion to post-invasion.

Appendix References

Atkinson, C. L., T. B. Parr, B. C. van Ee, D. D. Knapp, M. Winebarger, K. J. Madoni, and W. R. Haag. 2020. Length-mass equations for freshwater unionid mussel assemblages: Implications for estimating ecosystem function. Freshwater Science 39:377–390.

Benke, A. C., A. D. Huryn, L. A. Smock, and B. J. Wallace. 1999. Length-mass relationships for freshwater macroinvertebrates in North America with particular reference to the southeastern United States. Page Am Benthol. Soc.

Bowen, S. H. 1983. Quantitative description of the diet. Pages 325–336 *in* L. A. Nielsen and D. L. Johnson, editors. Fisheries Techniques. American Fisheries Society, Bethesda, MD.

Brakke, D. F. 1976. Modification of the Whiteside–Williams Pattern Sampler. Journal of the Fisheries Research Board of Canada 33:2861–2863.

Brey, T. 2001. Population dynamics in benthic invertebrates. A virtual handbook.

Brey, T., C. Müller-Wiegmann, Z. M. C. Zittier, and W. Hagen. 2010. Body composition in aquatic organisms - A global data bank of relationships between mass, elemental composition and energy content. Journal of Sea Research 64:334–340.

Brown, J. H., J. F. Gillooly, A. P. Allen, V. M. Savage, and G. B. West. 2004. Toward a metabolic theory of ecology. Ecology 85:1771–1789.

Chick, J. H., S. Coyne, and J. C. Trexler. 1999. Effectiveness of Airboat Electrofishing for Sampling Fishes in Shallow, Vegetated Habitats. North American Journal of Fisheries Management 19:957–967.

Edwards, F. K., R. B. Lauridsen, L. Armand, H. M. Vincent, and J. I. Jones. 2009. The relationship between length, mass and preservation time for three species of freshwater leeches (Hirudinea). Fundamental and Applied Limnology 173:321–327.

Flood, P. J., W. F. Loftus, and J. C. Trexler. 2022. Fishes in a seasonally pulsed wetland show spatiotemporal shifts in diet and trophic niche not associated with shifts in trophic position. Food Webs In Review.

Gatto, J. v., J. L. Kline, W. F. Loftus, and J. C. Trexler. 2021. Linking demographic transitions to population dynamics in a fluctuating environment. Canadian Journal of Fisheries and Aquatic Sciences 78:797–808.

Gauzens, B., A. Barnes, D. P. Giling, J. Hines, M. Jochum, J. S. Lefcheck, B. Rosenbaum, S. Wang, and U. Brose. 2019. fluxweb: An R package to easily estimate energy fluxes in food webs. Methods in Ecology and Evolution 10:270–279.

Gergs, R., and K. O. Rothhaupt. 2008. Feeding rates, assimilation efficiencies and growth of two amphipod species on biodeposited material from zebra mussels. Freshwater Biology 53:2494–2503.

Hyslop, E. J. 1980. Stomach contents analysis-a review of methods and their application. Page J. Fish Biol.

Jackson, A. L., R. Inger, A. C. Parnell, and S. Bearhop. 2011. Comparing isotopic niche widths among and within communities: SIBER - Stable Isotope Bayesian Ellipses in R. Journal of Animal Ecology 80:595–602.

Jordan, F., S. Coyne, and J. C. Trexler. 1997. Sampling Fishes in Vegetated Habitats: Effects of Habitat Structure on Sampling Characteristics of the 1-m 2 Throw Trap. Transactions of the American Fisheries Society 126:1012–1020.

Klassen, J. A., D. E. Gawlik, and B. A. Botson. 2014. Length-weight and length-length relationships for common fish and crayfish species in the Everglades, Florida, USA. Journal of Applied Ichthyology 30:564–566.

Kortsch, S., R. Frelat, L. Pecuchet, P. Olivier, I. Putnis, E. Bonsdorff, H. Ojaveer, I. Jurgensone, S. Strāķe, G. Rubene, Ē. Krūze, and M. C. Nordström. 2021. Disentangling temporal food web dynamics facilitates understanding of ecosystem functioning. Journal of Animal Ecology 90:1205–1216.

Kushlan, J. A., S. A. Voorhees, W. F. Loftus, and P. C. Frohring. 1986. Length, mass, and calorific relationships of Everglades animals. Florida Scientist 49:65–79.

Quezada-Romegialli, C., A. L. Jackson, B. Hayden, K. K. Kahilainen, C. Lopes, and C. Harrod. 2018. tRophicPosition, an r package for the Bayesian estimation of trophic position from consumer stable isotope ratios. Methods in Ecology and Evolution 9:1592–1599.

Trexler, J. C., W. F. Loftus, and J. Chick. 2003. Setting and monitoring restoration goals in the absence of historical data: the case of fishes in the Florida Everglades. Pages 351–376 *in* D. Busch and J. C. Trexler, editors. Monitoring Ecoregional Initiatives: Interdisciplinary Approaches for Determining Status and Trends of Ecosystem. Island Press, Washington, D.C.

Trexler, J. C., W. F. Loftus, F. Jordan, J. H. Chick, K. L. Kandl, T. C. McElroy, and O. L. Bass. 2002. Ecological Scale and Its Implications for Freshwater Fishes in the Florida Everglades. Pages 153–184 *in* J. W. Porter and K. G. Porter, editors. The Everglades, Florida Bay, and Coral Reefs of the Florida Keys: an ecosystem sourcebook. CRC Press, Boca Raton, FL.

Whiteside, M. C., and J. B. Williams. 1975. A new sampling technique for aquatic ecologists. SIL Proceedings, 1922-2010 19:1534–1539.

Tables

Table S1. Site locations in Shark River Slough (SRS), Everglades, Florida, United States of America where we sampled each habitat (P – pond, M – marsh) in the wet and dry seasons. Wet, Dry, and Total are the number of throw traps performed at a site in each habitat during each season and total.

Slough	Northing	Easting	Site	Habitat	Wet	Dry	Total
SRS	2818459	516433	LJB	P	2	3	5
				M	2	3	5
	2822986	520095	CPB	P	0	0	0
				M	3	3	6
	2819974	517453	AH11	P	0	3	3
				M	3	3	6
	2835306	525698	AH13	P	3	3	6
				M	3	3	6
	2834758	528209	WG16	P	3	3	6
				M	3	3	6
				Total	22	27	49

Table S2. This is a list of Latin and common names for taxa in this study. In some tables and figures taxa are abbreviated (Abbr.) as shown here.

Latin Name	Common Name	Abbr.
Acris gryllus	Southern Cricket Frog	A.gry
Ameiurus natalis	Yellow Bullhead Catfish	A.nat
Amia calva	Bowfin	A.cal
Amphipoda	Amphipods	-
Amphiuma means	Two-Toed Amphiuma	A.mea
Anguilla rostrata	American Eel	A.ros
Aphredoderus sayanus	Pirate Perch	A.say
Belonesox belizanus	Pike Killifish	B.bel
Belostoma spp.	Giant Water Bugs	-
Brachymesia gravida	Four-Spotted Pennant	-
Celithemis spp.	Pennants	-
Centropomis undecimalis	Common Snook	C.und
Chironomidae	Midge Larvae	-
Cichla ocellatus	Peacock Bass	C.oce
Cichlasoma bimaculatum	Black Acara	C.bim
Cladium jamaicense	Sawgrass	-
Cladocera	Cladocerans	-
Clarias batrachus	Walking Catfish	C.bat
Coenagrionidae	Damselflies	-
Coleoptera	Beetles	-
Copepoda	Copepods	-
Corixidae	Water Boatmen	-
Coryphaeschna ingens	Regal Darner	-
Cybister spp.	Giant Diving Beetles	-
Cyprinodon variegatus	Sheepshead Minnow	C.var
Elassoma evergladei	Everglades Pygmy Sunfish	E.eve
Eleocharis cellulosa	Spikerush	-
Enneacanthus gloriosus	Bluespotted Sunfish	E.glo
Ephemeroptera	Mayflies	-
Epitheca stella	Florida Baskettail	-
Erimyzon sucetta	Lake Chubsucker	E.suc
Erythemis simplicicollis	Eastern Pondhawk	-
Esox a. americanus	Redfin Pickerel	E.ame
Flocculent Matter	Detritus ("floc")	-
Fundulus chrysotus	Golden Topminnow	F.chr
Fundulus confluentus	Marsh Killifish	F.con

Gambusia holbrooki	Eastern Mosquitofish	G.hol
Gerridae	Water Striders	-
Hemichromis letourneuxi	African Jewelfish	H.let
Heterandria formosa	Least Killifish	H.for
Hoplisternum littorale	Armored Catfish	H.lit
Hydrachnidia	Aquatic Mites	-
Jordanella floridae	Flagfish	J.flo
Labidesthes vanhyningi	Golden Silverside	L.van
Lepisosteus platyrhincus	Florida Gar	L.pla
Lepomis gulosus	Warmouth	L.gul
Lepomis macrochirus	Bluegill	L.mac
Lepomis marginatus	Dollar Sunfish	L.mar
Lepomis microlophus	Redear Sunfish	L.mic
Lepomis punctatus	Spotted Sunfish	L.pun
Libellula incesta	Slaty Skimmer	-
Littoridinops monroensis	Aquatic Snail (Hydrobiidae)	-
Lucania goodei	Bluefin Killifish	L.goo
Macrognathus siamensis	Peacock Eel	M.sia
Mayaheros urophthalmus	Mayan Cichlid	M.uro
Melanoides tuberculata	Red-Rimmed Melania	-
Menidia berylina	Inland Silverside	M.ber
Micropterus salmoides	Largemouth Bass	M.sal
Monopterus albus	Asian Swamp Eel	M.alb
Nerodia floridana	Florida Green Watersnake	N.flo
Nostoc spp.	Cyanobacteria	-
Notophthalmus viridescens	Peninsula Newt	N.vir
Notropis petersoni	Coastal Shiner	N.pet
Noturus gyrinus	Tadpole Madtom	N.gyr
Oligochaeta	earthworms	-
Oreochromis aureus	Blue Tilapia	O.aur
Ostracoda	Ostracods	-
Pachydiplax longipennis	Blue Dasher	-
Palaemonetes paludosus	Grass Shrimp	-
Panicum hemitomon	Maidencane	-
Parachromis managuensis	Jaguar Cichlid	P.man
Pelmatolapia mariae	Spotted Tilapia	P.mar
Pelocoris femoratus	Creeping Water Bug	-
Periphyton - Epiphytic	epiphytic periphyton	-
Periphyton - Mat	floating periphyton	-
Planorbella	Ram's Horn Snails	-

Platyhelminthes	flatworms	-
Poecilia latipinna	Sailfin Molly	P.lat
Pomacea maculata	Giant Applesnail	-
Pomacea paludosa	Florida Applesnail	-
Pontedaria cordata	Pickerelweed	-
Procambarus alleni	Everglades Crayfish	P.all
Procambarus fallax	Slough Crayfish	P.fal
Procambarus juveniles	juvenile crayfish	P.spp
Pseudobranchus a. belli	Everglades Pygmy Siren	P.bel
Pseudosuccinea columella	American Ribbed Fluke Snail	-
Rana grylio	Pig Frog	R.gry
Sagitarria lancifolia	Lanceleaf Arrowhead	-
Siren lacertina	Greater Siren	S.lac
Sphaeriidae	Fingernail Clams	-
Utricularia foliosa	Leafy Bladderwort	-
Utricularia purpurea	Eastern Purple Bladderwort	-

Table S3. Most fishes were analyzed by species size classes based on length in millimeters to account for dietary changes based on ontogeny.

Species	Length (mm)	Size Class
Ameiurus natalis	≤ 96	1
	97 < x ≤ 135	2
	135 < x ≤ 175	3
	> 175	4
Amia calva	≤ 70	1
	> 70	2
Amphiuma means	-	1
Anguilla rostrata	-	1
Aphredoderus sayanus	-	1
Astronotus ocellatus	-	1
Belonesox belizanus	≤ 64	1
	> 64	2
Cichla ocellaris	-	1
Cichlasoma bimaculatum	-	1
Clarias batrachus	-	1
Cyprinodon variegatus	-	1
Elassoma evergladei	-	1
Enneacanthus gloriosus	-	1
Erimyzon sucetta	≤ 100	1
	100 < x ≤ 150	2
	> 150	3
Esox a. americanus	-	1
Fundulus confluentus	≤ 40	1
	> 40	2
Fundulus chrysotus	≤ 40	1
	> 40	2
Gambusia holbrooki	≤ 17	1
	> 17	2
Hemichromis	-	1

letourneuxi		
Heterandria formosa	≤ 12	1
	> 12	2
Hoplisternum littorale	-	1
Jordanella floridae	≤ 18	1
	> 18	2
Labidesthes vanhyningi	-	1
Lepisosteus platyrhincus	≤ 450	1
	> 450	2
Lepomis gulosus	≤ 71	1
	71 < x ≤ 95	2
	95 < x ≤ 120	3
	> 120	4
Lepomis macrochirus	-	1
Lepomis marginatus	-	1
Lepomis microlophus	≤ 100	1
	> 100	2
Lepomis punctatus	≤ 40	1
	40 < x ≤ 90	2
	> 90	3
Lucania goodei	≤ 17.5	1
	> 17.5	2
Macgronathus siamemis	-	1
Mayaheros urophthalmus	≤ 120	1
	> 120	2
Monopterus albus	≤ 275	1
	275 < x ≤ 450	2
	450 < x ≤ 625	3
	> 625	4
Micropterus salmoides	-	1
Notemigonus crysoleucas	-	1
Noturus gyrinus	-	1
Notropis maculatus	-	1
Notropis petersoni	-	1
Parachromis managuense	-	1

Pelmatolapia mariae	-	1
Poeilia latipinna	≤ 20	1
	> 20	2

Table S4. Samples sizes stratified by size class, habitat (marsh, pond) and season (wet, dry) for A) stable isotopes of carbon and nitrogen and B) stomach contents. In total, we analyzed 2200 stable isotope samples and stomach contents of 997 individuals.

A. Stable Isotopes		Shark River Slough				Size Class Total
Species	Size Class	Wet		Dry		
		Marsh	Pond	Marsh	Pond	
Amia calva	2	1	4	-	-	5
Acris gryllus	1	-	4	-	1	5
Amphiuma means	1	-	-	1	3	4
Ameiurus natalis	1	-	-	2	5	7
	2	1	-	-	5	6
	3	-	3	-	6	9
	4	3	-	-	10	13
Anguilla rostrata	1	-	-	-	-	0
Aphredoderus sayanus	1	1	-	9	-	10
Nostoc spp.	-	1	1	-	3	5
Amphipoda	-	20	13	8	12	53
Belonesox belizanus	1	-	3	3	4	10
	2	-	-	-	-	0
Belostoma spp.	-	6	2	1	3	12
Hydrachnidia	-	2	2	-	13	17
Brachymesia gravida	-	2	10	-	11	23
Clarias batrachus	1	12	3	-	1	16
Cichlasoma bimaculatum	1	-	-	-	-	0
Parachromis managuensis	1	-	-	-	1	1
Cichla ocellatus	1	2	-	-	-	2
Centropomis undecimalis	1	9	-	-	1	10
Mayaheros urophthalmus	1	9	8	4	17	38
	2	7	3	-	2	12
Cyprinodon variegatus	1	1	-	-	-	1
Celithemis spp.	-	5	8	6	6	25
Chironomidae	-	3	1	-	2	6
Cladocera	-	1	1	-	6	8
Cladium jamaicense	-	-	3	-	2	5
Coenagrionidae	-	7	5	-	-	12
Coleoptera	-	1	-	-	-	1

Copepoda	-	4	17	-	23	44
Coryphaeschna ingens	-	-	3	-	-	3
Corixidae	-	1	1	-	-	2
Cybster spp.	-	-	-	-	1	1
Esox a. americanus	1	-	-	1	-	1
Elassoma evergladei	1	6	-	9	5	20
Enneacanthus gloriosus	1	5	15	13	1	34
Erimyzon sucetta	1	4	6	6	2	18
	2	-	7	-	4	11
Eleocharis cellulosa	-	-	3	1	1	5
Ephemeroptera	-	-	-	-	3	3
Epitheca stella	-	-	2	-	-	2
Erythemis simplicicollis	-	1	1	6	-	8
Fundulus chrysotus	1	17	29	12	21	79
	2	11	8	10	2	31
Fundulus confluentus	1	1	9	3	17	30
Platyhelminthes	-	-	-	-	1	1
Flocculent Matter	-	4	12	3	11	30
Gambusia holbrooki	1	21	57	10	22	110
	2	39	121	29	28	217
Gerridae	-	-	-	-	-	0
Heterandria formosa	1	9	10	1	11	31
	2	5	4	9	7	25
Hemichromis letourneuxi	1	43	23	47	2	115
Hoplisternum littorale	1	1	-	5	13	19
Jordanella floridae	1	6	2	-	3	11
	2	5	4	4	9	22
Lucania goodei	1	24	57	19	27	127
	2	50	84	31	30	195
Lepomis gulosus	1	-	1	-	3	4
	2	-	8	1	11	20
	3	1	21	-	14	36
	4	-	9	-	9	18
Lepomis macrochirus	1	19	8	-	2	29
Lepomis marginatus	1	6	12	7	13	38
Lepomis microlophus	1	3	8	2	4	17
	2	5	7	-	-	12
Lepisosteus platyrhincus	1	-	19	-	25	44
	2	-	3	-	-	3

Lepomis punctatus	1	-	-	2	-	2
	2	5	8	9	6	28
	3	-	2	2	3	7
Labidesthes vanhyningi	1	-	-	-	-	0
Libellula incesta	-	3	4	1	1	9
Litroridinops monroensis	-	-	-	-	1	1
Monopterus albus	1	-	-	-	-	0
Monopterus albus	2	-	1	-	-	1
Monopterus albus	3	1	2	-	-	3
Monopterus albus	4	-	1	-	-	1
Menidia berylina	1	-	-	-	-	0
Micropterus salmoides	1	1	18	-	2	21
Macrognathus siamensis	1	3	29	-	-	32
Melanoides tuberculata	-	4	-	-	-	4
Nerodia floridana	1	-	-	4	1	5
Noturus gyrinus	1	-	-	3	-	3
Notropis petersoni	1	2	5	-	-	7
Notophthalmus viridescens	1	1	4	3	1	9
Oreochromis aureus	1	21	6	2	-	29
Ostracoda	-	2	1	-	31	34
Pseudobranchus a. belli	1	1	-	1	-	2
Poecilia latipinna	1	1	6	-	-	7
	2	8	10	2	11	31
Pachydiplax longipennis	-	-	3	-	1	4
Palaemonetes paludosus	-	11	3	6	6	26
Panicum hemitomon	-	2	1	1	1	5
Pelocoris femoratus	-	6	5	17	-	28
Periphyton - Epiphytic	-	2	5	-	-	7
Periphyton - Mat	-	3	1	4	-	8
Planorbella	-	-	-	3	9	12
Pomacea maculata	-	-	-	-	1	1
Pomacea paludosa	-	-	1	-	2	3
Pontedaria cordata	-	-	1	-	-	1
Procambarus alleni	-	2	17	9	8	36
Procambarus fallax	-	5	4	-	17	26
Procambarus juveniles	-	4	3	1	1	9
Pseudosuccinea columella	-	-	-	-	-	0
Rana grylio	1	-	2	9	2	13

	2	-	-	2	6	8
Oligochaeta	-	-	1	-	-	1
Sirens lacertina	1	2	2	8	7	19
Sagitarria lancifolia	-	-	1	-	2	3
Sphaeridae	-	-	-	-	-	0
Pelmatolapia mariae	1	-	5	5	-	10
Utricularia foliosa	-	1	2	1	2	6
Utricularia purpurea	-	2	1	1	2	6
	Total Number of Stable Isotope Samples					2200

B. Stomach Contents		Shark River Slough				Size Class Total
		Wet		Dry		
Species	Size Class	Marsh	Pond	Marsh	Pond	
Amia calva	2	1	4	-	-	5
Acris gryllus	1	-	2	-	-	2
Ameiurus natalis	1	-	-	2	3	5
	2	1	-	-	4	5
	3	-	3	-	6	9
	4	2	-	-	8	10
Anguilla rostrata	1	-	-	-	-	0
Aphredoderus sayanus	1	-	-	1	-	1
Belonesox belizanus	1	-	3	1	2	6
Clarias batrachus	1	10	3	-	1	14
Cichlasoma bimaculatum	1	-	-	-	-	0
Parachromis manguensis	1	-	-	-	1	1
Cichla ocellaris	1	1	-	-	-	1
Centropomis undecimalis	1	3	-	-	1	4
Mayaheros urophthalmus	1	16	6	4	-	26
	2	6	3	-	1	10
Cyprinodon variegatus	1	1	-	-	-	1
Esox a. americanus	1	-	-	1	-	1
Elassoma evergladei	1	4	13	10	1	28
Enneacanthus gloriosus	1	1	3	5	-	9
Erimyzon sucetta	1	3	5	4	2	14
	2	-	6	-	4	10
Fundulus chrysotus	1	14	23	10	13	60
	2	-	1	1	2	4
Fundulus confluentus	1	-	-	2	1	3

Gambusia holbrooki	1	19	18	7	16	60
	2	18	21	5	29	73
Heterandria formosa	1	8	9	1	8	26
	2	5	3	5	5	18
Hemichromis letourneuxi	1	25	15	35	-	75
Hoplosternun littorale	1	1	-	5	11	17
Jordanella floridae	1	3	2	-	1	6
	2	1	1	1	3	6
Lucania goodei	1	19	20	18	25	82
	2	16	19	25	25	85
Lepomis gulosus	1	-	1	-	3	4
	2	-	7	-	11	18
	3	1	19	-	11	31
	4	-	7	-	7	14
Lepomis macrochirus	1	17	7	-	2	26
Lepomis marginatus	1	-	3	3	9	15
Lepomis microlophus	1	3	2	1	-	6
	2	4	5	-	-	9
Lepisosteus platyrhincus	1	-	16	-	21	37
	2	-	4	-	-	4
Lepomis punctatus	2	3	7	7	4	21
	3	-	2	1	3	6
Labidesthes vanhyningi	1	-	-	-	-	0
Monopterus albus	1	-	-	-	-	0
	2	-	-	-	-	0
	3	1	1	-	-	2
	4	-	1	-	-	1
Menidia berylina	1	-	-	-	-	0
Micropterus salmoides	1	1	14	-	1	16
Macrognathus siamensis	1	5	25	-	-	30
Nerodia floridana	1	-	-	-	1	1
Noturus gyrinus	1	-	-	1	-	1
Notropis petersoni	1	-	3	-	-	3
Notophthalmus viridescens	1	-	-	1	-	1
Oreochromis aureus	1	18	5	1	-	24
Pseudobranchus a. belli	1	1	-	2	-	3
Poecilia latipinna	1	-	5	-	-	5
	2	2	8	2	6	18
Rana grylio	1	-	-	6	-	6

	2	-	-	-	1	1
Sirens lacertina	1	1	1	3	2	7
Pelmatolapia mariae	1	-	5	5	-	10
	Total Number of Stomachs Analyzed					997

Table S5. Permuted ANOVA results comparing annual hydroperiod (number of days with water depth ≥ 5 cm) among time periods. Time periods are: “Pre-Stomachs” – 1978-1981 (when pre-invasion stomach content data were collected), “Pre-MDW” – 1982-1989 (indicates sampling prior to the current iteration of the long-term monitoring project), “Pre-Isotopes” – 1990-1995 (when the pre-invasion stable isotopes samples were collected, also prior to the current iteration of the monitoring project), “Pre-Invasion” – 1996-1999 (years immediately preceding Jewelfish invasion, current monitoring regime began in 1996), “Pre-Boom” – 2000-2011 (period of time from Jewelfish invasion in 2000 up until the increase in their population), “Boom” – 2012-2017 (years of elevated Jewelfish relative biomass and abundance), “Sampling” – 2018-2019 (when sampling for this project occurred), “Post-Boom” – 2020-2021 (years after sampling for this project while Jewelfish were at low levels of relative abundance and biomass).

Comparison	Psuedo-F	p
Boom - Post-Boom	0.2019	0.8711
Boom - Pre-Boom	0.7457	0.6719
Boom - Pre-Invasion	-1.367	0.5689
Boom - Pre-Isotopes	1.504	0.5689
Boom - Pre-MDW	0.953	0.6146
Boom - Pre-Stomachs	0.6033	0.7553
Boom - Sampling	-0.8485	0.6163
Post-Boom - Pre-Boom	0.3279	0.8322
Post-Boom - Pre-Invasion	-1.867	0.5689
Post-Boom - Pre-Isotopes	0.8507	0.6163
Post-Boom - Pre-MDW	0.4906	0.7553
Post-Boom - Pre-Stomachs	0.2834	0.8367
Post-Boom - Sampling	-1.217	0.5689
Pre-Boom - Pre-Invasion	-1.493	0.5689
Pre-Boom - Pre-Isotopes	1.224	0.5689
Pre-Boom - Pre-MDW	0.5134	0.7553
Pre-Boom - Pre-Stomachs	-0.1347	0.8928
Pre-Boom - Sampling	-0.9883	0.6146
Pre-Invasion - Pre-Isotopes	1.77	0.5689
Pre-Invasion - Pre-MDW	1.378	0.5689
Pre-Invasion - Pre-Stomachs	1.714	0.5689
Pre-Invasion - Sampling	1.572	0.5689
Pre-Isotopes - Pre-MDW	-0.5685	0.7553
Pre-Isotopes - Pre-Stomachs	-0.9612	0.6146
Pre-Isotopes - Sampling	-1.238	0.5689
Pre-MDW - Pre-Stomachs	-0.4574	0.7553

Pre-MDW - Sampling	-0.9322	0.6146
Pre-Stomachs - Sampling	-1.143	0.5903

Table S6. Conversion factors for invertebrate taxa included in biomass and energy flux modeling. We calculated biomass at a higher taxonomic resolution (Biomass Group) and then summed that biomass by guild to calculate energy fluxes among guilds. For some taxa, there are known, Everglades-specific count-biomass conversion factors (N – no, Y – yes). For the remaining taxa, biomass was estimated based on average length for that group and length-weight (L-W) relationships derived from the literature: 1 – Benke et al. (1999), 2 – Edwards et al. (2009), 3 – Klassen et al. (2014), and 4 – Atkinson et al. (2020) and constants a and b. We then converted wet weight (WM) into ash-free dry mass (AFDM) using known conversion factors (Brey 2001, Brey et al. 2010).

Scientific name	**Common Name**	**Biomass Group**	**Guild**	**Count-Biomass**	**L-W**	**Length (mm)**	**b**	**a**	**DM/ WM**	**AFDM/ DM**	**AFDM/ WM**
Suborder Anisoptera	unidentified Dragonfly Larva	Odonata	Carn. Inverts	N	1	12.2	2.792	0.0078	0.205	0.943	0.19332
Family Chironomidae	Midge Larvae	Diptera	Omni. Inverts	N	1	3	2.692	0.0025	0.163	0.917	0.14947
Family Coenagrionidae	Damselfly Larvae	Odonata	Carn. Inverts	N	1	11.8	2.792	0.0078	0.277	0.925	0.25623
Family Corixidae	Water Boatmen	Hemiptera	Carn. Inverts	N	1	3.75	2.734	0.0108	0.26	0.947	0.24622
Cybister fimbriolatus	Predaceous Diving Beetle	Coleoptera	Omni. Inverts	N	1	29.6	2.91	0.0077	0.307	0.921	0.28275
Cybister spp.	Predaceous Diving Beetle	Coleoptera	Carn. Inverts	N	1	34.6	2.91	0.0077	0.307	0.921	0.28275
Family Gerridae	water striders	Hemiptera	Carn. Inverts	N	1	4.9	2.734	0.0108	0.26	0.947	0.24622
Suborder Heteroptera	aquatic bugs	Hemiptera	Carn. Inverts	N	1	5	2.734	0.0108	0.26	0.947	0.24622
Class Hirudinea	Leeches	Hirudinea	Carn. Inverts	N	2	17.5	2.73	0.104	0.204	0.943	0.19237
Pachydiplax longipennis	blue dasher	Odonata	Carn. Inverts	N	1	13.8	2.792	0.0078	0.234	0.932	0.21809
Procambarus alleni	Everglades crayfish	PROSPP	Decapods	N	3	13.2	2.85	0.217	0.249	0.837	0.20841
Procambarus fallax	Slough crayfish	PROSPP	Decapods	N	3	13.9	3.03	0.192	0.249	0.837	0.20841
Family Sphaeriidae	fingernail clams	Mollusca	Herb. Inverts	N	4	6	2.87	0.000352	0.19	0.788	0.14972

Taphromysis louisianae	mysid shrimp	Decapoda	Decapods	N	1	2.9	3.626	0.0147	0.224	0.685	0.15344
Belostoma spp.	Giant Water Bug	BELSPP	Carn. Inverts	Y					0.26	0.947	0.24622
Brachymesia gravida	Four-Spotted Pennant	BRASPP	Carn. Inverts	Y					0.234	0.932	0.21809
Brachymesia spp.	Pennant species	BRASPP	Carn. Inverts	Y					0.234	0.932	0.21809
Celithemis bertha	Red-Veined Pennant	CELSPP	Carn. Inverts	Y					0.234	0.932	0.21809
Celithemis eponina	Halloween Pennant	CELSPP	Carn. Inverts	Y					0.234	0.932	0.21809
Celithemis ornata	Faded Pennant	CELSPP	Carn. Inverts	Y					0.234	0.932	0.21809
Celithemis species	Pennant species	CELSPP	Carn. Inverts	Y					0.234	0.932	0.21809
Order Coleoptera	Aquatic Beetle Adults	COLEOP	Omni. Inverts	Y					0.307	0.921	0.28275
Order Coleoptera	Aquatic Beetle Larvae	COLEOP	Omni. Inverts	Y					0.307	0.921	0.28275
Order Coleoptera	Aquatic Beetles	COLEOP	Omni. Inverts	Y					0.307	0.921	0.28275
Coryphaeschna ingens	Regal Darner	CORING	Carn. Inverts	Y					0.234	0.932	0.21809
Order Diptera	unidentified fly larva	DIPSPP	Omni. Inverts	Y					0.185	0.915	0.16928
Order Diptera	unidentified fly larva	DIPSPP	Omni. Inverts	Y					0.185	0.915	0.16928
Order Ephemeroptera	mayfly larvae	EPHEME	Carn. Inverts	Y					0.227	0.933	0.21179
Epicordulia princeps regina	dragonfly	EPISPP	Carn. Inverts	Y					0.234	0.932	0.21809
Epicordulia princeps regima	dragonfly	EPISPP	Carn. Inverts	Y					0.234	0.932	0.21809
Epitheca stella/sepia	Florida baskettail	EPISPP	Carn. Inverts	Y					0.234	0.932	0.21809
Erythemis plebeja	Pin-tailed Pondhawk	ERYSPP	Carn. Inverts	Y					0.234	0.932	0.21809

Erythemis simplicicollis	Eastern Pondhawk	ERYSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Erythemis spp.	Pondhawk genus	ERYSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Erythemis vesiculosa	Great pondhawk	ERYSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Haitia spp.	unidentified physid snail	HAISPP	Herb. Inverts	Y			0.275	0.9	0.2475
Libellula auripennis	golden-winged skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula axilena	Bar-winged Skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula deplanata	Blue Corporal	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula incesta	Slaty skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula needhami	Needham's skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula semifasciata	Painted skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula species	Skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Libellula vibrans	great blue skimmer	LIBSPP	Carn. Inverts	Y			0.234	0.932	0.21809
Palaemonetes paludosus	grass shrimp	Decapoda	Decapods	Y	-5.597	3.321	0.253	0.837	0.21176
Pelocoris femoratus	creeping water bug	PELFEM	Carn. Inverts	Y			0.26	0.947	0.24622
Planorbella spp.	unidentified planorbid snail	PLASPP	Herb. Inverts	Y			0.275	0.9	0.2475
Pomacea paludosa	apple snail	POMPAL	Herb. Inverts	Y			0.275	0.9	0.2475

Table S7. PERMANOVA results for pre- vs post-invasion diets from Shark River Slough ponds and marshes from both seasons. Nestedness prevented testing all possible interactions.

Variables	DF	SS	Mean SS	F	R^2	p
Invasion Status	1	24.1	24.1	111.7	0.02	0.001
Season	1	12.4	12.4	57.5	0.01	0.001
Habitat	2	75.0	37.5	173.9	0.06	0.001
Size Class	110	398.1	3.6	16.8	0.31	0.001
Invasion Status:Season	1	3.8	3.8	17.5	0.00	0.001
Season:Habitat	2	8.7	4.3	20.1	0.01	0.001
Season:Size Class	75	65.8	0.9	4.1	0.05	0.001
Habitat:Size Class	62	44.3	0.7	3.3	0.04	0.001
Season:Habitat:Size Class	25	18.7	0.7	3.5	0.01	0.001
Residuals	2847	613.9	0.2		0.49	
Total	3126	1264.7			1.00	

Table S8. Pairwise PERMANOVA results comparing pre- and post-invasion diets of a species size class within the same habitat and season. Drivers are the prey functional groups that have changed post-invasion (P – producers, Dt – Detritus, H – herbivorous invertebrates, O – omnivorous invertebrates, C – carnivorous invertebrates, Dc – decapods, S – small fishes, L – large fishes). Size classes are abbreviated as first letter of the genus, period, first three letters of the species, and the number denotes the size class based on standard length.

Size Class	Habitat	Season	p	Driver
A.nat2	Marsh	Wet	0.25	-
C.var1			0.23	-
E.eve1			0.11	-
E.glo1			0.05	H
F.chr1			**0**	O/C
G.hol1			**0**	H/C
G.hol2			**0**	C
H.for1			**0**	Dt/O/C
H.for2			**0.01**	Dt/O/C
J.flo1			**0**	P/O
J.flo2			0.08	P/O
L.goo1			**0**	Dt/O
L.goo2			**0**	H/O
L.mac1			0.29	-
L.mic1			**0.01**	H/O/Dc
L.mic2			0.78	-
L.pun2			0.34	-
P.lat2			0.73	-
A.nat1		Dry	0.15	-
B.bel1			1	-
E.eve1			**0.01**	DP
E.glo1			**0**	O/C
F.chr1			**0**	O/C
F.chr2			0.2	-
F.con1			0.05	H/O/C
G.hol1			**0.02**	Dt/C
G.hol2			0.24	-
H.for1			0.15	-

H.for2			**0.02**	DP
J.flo2			0.06	P/D
L.goo1			0.27	-
L.goo2			**0.02**	O
L.mar1			**0.04**	Dt/O
L.pun2			**0**	H/O/C
N.gyr1			0.38	-
P.lat2			0.21	-
A.nat1	Pond	Dry	0.06	Dt/Dc/L
A.nat2			0.22	-
A.nat3			**0**	Dt/O
A.nat4			0.07	Dt/Dc
B.bel1			0.27	-
C.bat1			0.2	-
E.eve1			0.19	-
F.chr1			0.11	-
G.hol1			**0**	O/C
G.hol2			**0**	P/C
H.for1			**0.02**	Dt/O
H.for2			0.21	-
L.gul1			0.6	-
L.gul2			**0**	Dt/C/Dc/L
L.gul3			**0**	O/C/Dc/L
L.gul4			0.49	-
L.mac1			0.35	-
L.mar1			**0.03**	O/L
L.pla1			**0**	O/Dc/L
L.pun2			0.07	C/Dc
L.pun3			**0.03**	P/Dt/H/O/C
P.lat2			**0.01**	P/O
A.nat3		Wet	0.25	-
B.bel1			0.14	-
C.bat1			0.43	-
C.uro1			0.05	Dt/H/Dc
C.uro2			0.58	-
E.eve1			**0**	H/O
E.suc1			0.15	-
E.suc2			0.42	-
F.chr1			**0**	DP

F.chr2	0.21	-
G.hol1	**0**	H/C
G.hol2	**0**	P/H/O
H.for1	**0**	Dt/O
H.for2	**0.02**	Dt/O/C
J.flo1	0.14	-
J.flo2	0.1	-
L.goo1	0.06	Dt/O
L.goo2	**0**	O
L.gul1	0.82	-
L.gul2	0.54	-
L.gul3	0.3	-
L.gul4	0.49	-
L.mac1	**0.01**	P/O
L.mar1	**0.01**	H/O
L.mic1	0.37	-
L.mic2	0.38	-
L.pla1	**0.03**	Dt/O/Dc/L
L.pla2	0.1	-
L.pun2	0.16	-
L.pun3	0.36	-
M.sal1	**0.01**	Dc/S/L
N.pet1	0.23	-
T.mar1	0.56	-

Table S9. Trophic dispersion based on changes in trophic niche size modeled using stomach contents pre- vs post-invasion. Pre-invasion and post-invasion columns are modes of trophic niche area. Pre < Post is the probability that the pre-invasion niche area is smaller than the post-invasion niche area. Values > 0.95 indicated a statistical increase in niche size post-invasion, while values < 0.05 indicated a statistical decrease in niche size post-invasion. This corresponds to the direction of change and %Δ is the percent change in niche area pre- to post-invasion.

Season	Habitat	Species	Size Class	Pre-Invasion	Post-Invasion	Pre < Post	Direction	Percent Change
Dry	Marsh	*E. evergladei*	1	0.06	0.18	**1.000**	Increase	230
		E. gloriosus	1	0.04	0.13	**0.998**	Increase	214
		F. chrysotus	1	0.4	0.24	0.147	No Change	-41
		G. holbrooki	1	0.09	0.45	**1.000**	Increase	402
			2	0.22	0.13	0.282	No Change	-40
		H. formosa	2	0.07	0.08	0.770	No Change	18
		L. goodei	1	0.03	0.03	0.375	No Change	-10
			2	0.05	0.11	**1.000**	Increase	136
		L. marginatus	1	0.03	0.24	**1.000**	Increase	730
		L. punctatus	2	0.59	0.28	0.109	No Change	-52
	Pond	*A. natalis*	1	0.18	0.01	**0.001**	Decrease	-94
			2	0.16	0.18	0.789	No Change	13
			3	0.19	0.09	0.090	No Change	-53
			4	0.24	0.09	**0.033**	Decrease	-63
		F. chrysotus	1	1.7	0.59	0.147	No Change	-65
		G. holbrooki	1	0.1	0.89	**1.000**	Increase	786
			2	0.25	1.35	0.282	No Change	433
		H. formosa	1	0.01	0.16	**1.000**	Increase	1853
			2	0.13	0.02	0.770	No Change	-81
		L. gulosus	2	0.27	0.21	0.307	No Change	-23
			3	0.32	0.43	0.822	No Change	35
			4	0.41	0.07	**0.008**	Decrease	-84
		L. marginatus	1	0.13	0.61	**1.000**	Increase	371
		L. platyrhincus	1	0.11	0.33	**1.000**	Increase	211
		L. punctatus	2	1	0.67	0.109	No Change	-33
		P. latipinna	2	0.03	0.14	**0.994**	Increase	410
Wet	Marsh	*F. chrysotus*	1	0.45	1.05	0.147	No Change	135
		G. holbrooki	1	0.13	0.37	**1.000**	Increase	188
			2	0.4	1.08	0.282	No Change	167

	H. formosa	1	0.03	0.32	**1.000**	Increase	1022
		2	0.04	0.15	0.770	No Change	298
	J. floridae	1	0.08	0.02	**0.056**	No Change	-76
	L. goodei	1	0.04	0.14	0.375	No Change	289
		2	0.02	0.35	**1.000**	Increase	2141
	L. microlophus	1	0.23	0.01	**0.000**	Decrease	-98
	L. punctatus	2	0.38	0.27	0.109	No Change	-29
Pond	*E. evergladei*	1	0.04	0.26	**1.000**	Increase	598
	F. chrysotus	1	0.78	0.56	0.147	No Change	-28
	G. holbrooki	1	0.27	0.59	**1.000**	Increase	121
		2	1.57	0.92	0.282	No Change	-41
	H. formosa	1	0.01	0.11	**1.000**	Increase	1931
		2	0.01	0.12	0.770	No Change	1105
	L. goodei	1	0.01	0.1	0.375	No Change	841
		2	0.02	0.3	**1.000**	Increase	1177
	L. gulosus	2	0.28	0.51	0.307	No Change	79
		3	0.25	0.64	0.822	No Change	156
		4	0.38	0.31	**0.008**	Decrease	-19
	L. macrochirus	1	1.23	0.66	0.168	No Change	-46
	L. marginatus	1	0.23	2.02	**1.000**	Increase	763
	L. microlophus	2	0.06	0.02	0.086	No Change	-69
	L. platyrhincus	1	0.61	0.37	**1.000**	Increase	-39
		2	0.86	0.03	**0.000**	Decrease	-97
	L. punctatus	2	0.74	0.41	0.109	No Change	-44
	M. salmoides	1	0.2	0.2	0.582	No Change	-2
	M. urophthalmus	1	0.18	0.01	**0.000**	Decrease	-95
	P. mariae	1	0.15	0.19	0.822	No Change	30

Table S10. Post-invasion trophic niches modeled using stomach contents and compared across habitats (A) and seasons (B). Statistical differences are in bold.

A. Habitat Comparisons

Size Class	Slough	Season	Marsh	Pond	Marsh < Pond	Percent Change
C.bat1	SRS	Wet	3.64	0.65	**0.03**	-82.02
C.uro1			3.74	1.1	**0.01**	-70.68
C.uro2			2.27	2.36	0.64	3.62
Celithimus spp.			0.08	0.89	**1**	1013.3
Coenagrionidae			0.66	3.31	**1**	403.06
E.glo1			1.5	1.08	0.19	-27.81
E.suc1			2.22	1.51	0.23	-32.06
F.chr1			1.73	4.95	**1**	185.89
F.chr2			0.9	2.24	**0.98**	150.2
H.for1			0.68	6.99	**1**	933.22
H.for2			1.94	0.86	0.14	-55.94
H.let1			2.49	2.65	0.63	6.29
J.flo2			3.96	1.01	**0.03**	-74.44
L.goo1			3.88	4.15	0.57	6.95
L.goo2			5.22	4.56	0.22	-12.57
L.mac1			1.99	1.75	0.46	-12.11
L.mar1			3.3	3.67	0.5	11.1
L.mic1			2.88	1.45	0.08	-49.59
L.mic2			3.45	1.02	**0.01**	-70.51
L.pun2			2.49	1.29	0.11	-48.3
L. incesta			0.01	1.35	**1**	10785
M.sia1			0.76	2.53	0.88	235.1
O.aur1			4.05	4.52	0.68	11.59
P.lat2			0.46	0.97	0.92	111.18
P. paludosus			45.16	13.17	0.09	-70.84
P. femoratus			3.42	4.97	0.73	45.62
P. fallax			1.16	1.56	0.69	34.7
Procambarus spp.			0.62	0.02	**0**	-96.32
B.bel1		Dry	0.46	0.44	0.4	-5.39
C.uro1			1.33	1.93	0.62	44.65
Celithimus spp.			0.78	0.74	0.44	-5.85
E.eve1			1.94	2.04	0.62	5.37

F.chr1			4.37	5.29	0.66	21.16
F.con1			3.52	0.81	**0**	-76.87
G.hol1			0.8	4.67	**1**	485.36
G.hol2			2	4	**1**	100.59
H.for2			2.31	0.33	**0**	-85.7
H.lit1			2.49	1.81	0.19	-27.07
J.flo2			2.29	1.27	0.11	-44.4
L.goo1			2.54	1.86	0.14	-26.79
L.goo2			2.65	3.6	0.89	35.74
L.mar1			2.05	1.63	0.28	-20.39
L.pun2			1.32	1.65	0.7	25.39
P. paludosus			20.76	0.67	**0**	-96.78
Planorbella spp.			5.98	77.76	**1**	1200.8
P. alleni			1.88	1.12	0.16	-40.44
S.lac1			2.22	4.24	0.89	91.11

B. Seasonal Comparisons

Size Class	Slough	Habitat	Wet	Dry	Wet < Dry	Percent Change
C.uro1	SRS	Marsh	3.74	1.33	0.09	180.8
Celithimus spp.			0.08	0.78	**1**	-89.8
E.eve1			0.62	1.94	**0.97**	-68.1
E.glo1			1.5	2.74	0.81	-45.1
E.suc1			2.22	0.8	**0.04**	177.7
F.chr1			1.73	4.37	**1**	-60.4
F.chr2			0.9	3.22	**1**	-72.2
G.hol1			1.3	0.8	0.14	63.0
G.hol2			0.88	2	**1**	-56.1
H.for2			1.94	2.31	0.53	-16.1
H.let1			2.49	1.92	0.1	29.9
J.flo2			3.96	2.29	0.23	73.1
L.goo1			3.88	2.54	0.09	53.0
L.goo2			5.22	2.65	**0**	96.9
L.mar1			3.3	2.05	0.18	61.2
L.pun2			2.49	1.32	0.1	89.2
P. paludosus			45.16	20.76	0.1	117.5
P. femoratus			3.42	1.18	**0**	188.7
A.nat3		Pond	0.06	1.09	**1**	-94.5
B.bel1			0.47	0.44	0.39	8.9
B. gravida			2.92	2.29	0.28	27.2

C.uro1	1.1	1.93	0.85	-43.1
Celithimus spp.	0.89	0.74	0.36	21.0
E.suc2	1.21	0.14	**0**	743.7
F.chr1	4.95	5.29	0.62	-6.5
F.con1	0.65	0.81	0.64	-19.7
H.for1	6.99	3.39	**0.04**	106.5
H.for2	0.86	0.33	**0.04**	158.6
J.flo2	1.01	1.27	0.57	-20.4
L.goo1	4.15	1.86	**0**	123.5
L.goo2	4.56	3.6	0.16	26.8
L.gul2	0.85	0.43	**0.05**	97.3
L.gul3	0.89	0.74	0.34	20.0
L.gul4	0.84	1.86	**0.95**	-54.9
L.mar1	3.67	1.63	**0.03**	124.9
L.mic1	1.45	1.25	0.48	16.1
L.pla1	3.14	2.3	0.15	36.2
L.pun2	1.29	1.65	0.69	-22.0
P.lat2	0.97	2.68	**0.99**	-63.8
P. paludosus	13.17	0.67	**0**	1866.5
P. alleni	0.64	1.12	0.93	-42.9
P. fallax	1.56	1.35	0.25	16.1

Table S12. Statistical shifts in trophic position derived from stomach contents for a species size class for different types of comparisons: pre- vs post-invasion within the same habitat-season, post-invasion size class shifts within a habitat-season, post-invasion habitat shifts within the same habitat-season, and post-invasion seasonal shift within the same habitat. Comparisons that were not statistically different are not presented.

	Group A				Group B					
Comparison	Size Class	Habitat	Season	Invasion Status	Size Class	Habitat	Season	Invasion Status	Stat	p
Invasion	C.uro1	Pond	Wet	Post-Invasion	C.uro1	Pond	Wet	Pre-Invasion	-2.28	0.02
	F.chr1	Pond	Dry	Post-Invasion	F.chr1	Pond	Dry	Pre-Invasion	-2.13	0.03
	E.eve1	Pond	Wet	Post-Invasion	E.eve1	Pond	Wet	Pre-Invasion	2.04	0.04
Size Class	L.goo1	Pond	Wet	Post-Invasion	L.goo2	Pond	Wet	Post-Invasion	2.43	0.01
Habitat	G.hol1	Marsh	Dry	Post-Invasion	G.hol1	Pond	Dry	Post-Invasion	-2.28	0.02
	G.hol1	Marsh	Wet	Post-Invasion	G.hol1	Pond	Wet	Post-Invasion	1.92	0.06
Season	G.hol1	Marsh	Dry	Post-Invasion	G.hol1	Marsh	Wet	Post-Invasion	-2.51	0.01
	L.goo1	Pond	Dry	Post-Invasion	L.goo1	Pond	Wet	Post-Invasion	-1.92	0.06

Table S13. Trophic position (A) and alpha (B) modeled using stable isotopes compared pre- to post-invasion. For both pre- and post-invasion, we report 95% credibility intervals and modes for trophic position and alpha respectively. Pre < Post and %Δ are the same as in previous tables with statistical differences in bold.

A. Trophic Position

Species	Pre-Invasion		Post-Invasion		Comparison	
	95% C.I.	Mode	95% C.I.	Mode	Pre < Post	Percent Change
A. nebulosus	3.65 - 6.15	4.14	3.24 - 4.95	4.21	0.55	1.69
Anisoptera	2.35 - 2.97	2.63	2.18 - 2.66	2.31	0.07	-12.17
Belostoma spp.	2.33 - 3.27	2.8	2.08 - 2.88	2.43	0.11	-13.21
C. urophthalmus	3.37 - 4.29	3.78	3.3 - 3.81	3.52	0.13	-6.88
E. sucetta	3.2 - 4.02	3.56	2.54 - 4.25	3.45	0.38	-3.09
F. chrysotus	3.93 - 4.58	4.24	3.52 - 4.13	3.82	**0.03**	-9.91
G. holbrooki	3.9 - 4.5	4.2	3.39 - 3.79	3.56	**0.00**	-15.24
H. formosa	4.11 - 4.66	4.38	3.48 - 3.91	3.64	**0.00**	-16.89
L. macrochirus	3.47 - 4.37	3.9	3.74 - 4.26	3.97	0.61	1.79
L. microlophus	3.42 - 4.74	3.91	3.49 - 4.14	3.77	0.31	-3.58
L. punctatus	3.92 - 4.58	4.25	3.53 - 4.47	3.97	0.14	-6.59
P. fallax	2.55 - 3.35	2.92	2.14 - 3.06	2.53	0.09	-13.36
P. latipinna	3.06 - 4.21	3.63	3.3 - 4.03	3.6	0.47	-0.83
P. paludosus	3.48 - 3.9	3.68	2.8 - 3.59	3.15	**0.02**	-14.40

B. Alpha

Species	Pre-Invasion		Post-Invasion		Comparison	
	95% C.I.	Mode	95% C.I.	Mode	Pre < Post	Percent Change
A. nebulosus	0.28 - 1	0.91	0.01 -	0.2	**0.04**	-78.02

			0.79			
Anisoptera	0.03 - 0.67	0.29	0 - 0.94	0.08	0.34	-72.41
Belostoma spp.	0.2 - 0.98	0.7	0.01 - 0.68	0.19	0.06	-72.86
C. urophthalmus	0.37 - 1	0.91	0.05 - 0.91	0.44	0.08	-51.65
E. sucetta	0.38 - 1	0.92	0.01 - 0.86	0.17	**0.04**	-81.52
F. chrysotus	0.73 - 1	0.94	0.52 - 0.99	0.82	0.21	-12.77
G. holbrooki	0.52 - 1	0.93	0.08 - 0.69	0.32	**0.01**	-65.59
H. formosa	0.63 - 0.99	0.85	0 - 0.69	0.11	**0.01**	-87.06
L. macrochirus	0.06 - 0.99	0.68	0.16 - 0.8	0.42	0.29	-38.24
L. microlophus	0.68 - 1	0.91	0.01 - 0.8	0.17	**0.01**	-81.32
L. punctatus	0.48 - 1	0.95	0.01 - 0.83	0.24	**0.03**	-74.74
P. fallax	0.53 - 1	0.95	0.32 - 0.96	0.67	0.10	-29.47
P. latipinna	0.41 - 0.93	0.66	0.01 - 0.98	0.55	0.37	-16.67
P. paludosus	0.66 - 0.99	0.86	0.04 - 0.98	0.57	0.18	-33.72

Table S14. Post-invasion trophic positions calculated from stomach contents for species size classes among habitat-season levels.

Size Class	Slough	Season	Habitat	N	Mean	SD
A.cal2	SRS	Wet	Marsh	1	1.75	-
A.nat2	SRS	Wet	Marsh	1	2.69	-
A.nat4	SRS	Wet	Marsh	1	3.37	-
C.bat1	SRS	Wet	Marsh	8	2.74	0.54
C.oce1	SRS	Wet	Marsh	1	2.50	-
C.und1	SRS	Wet	Marsh	3	2.33	0.29
C.uro1	SRS	Wet	Marsh	15	2.62	0.54
C.uro2	SRS	Wet	Marsh	4	2.08	0.44
C.var1	SRS	Wet	Marsh	1	3.00	-
E.eve1	SRS	Wet	Marsh	1	2.00	-
E.glo1	SRS	Wet	Marsh	9	2.50	-
E.suc1	SRS	Wet	Marsh	3	2.41	0.11
F.chr1	SRS	Wet	Marsh	5	2.47	0.54
G.hol1	SRS	Wet	Marsh	1	2.67	0.44
G.hol2	SRS	Wet	Marsh	2	2.37	0.51
H.for1	SRS	Wet	Marsh	13	2.48	0.61
H.for2	SRS	Wet	Marsh	10	2.54	0.35
H.let1	SRS	Wet	Marsh	24	2.54	0.53
J.flo1	SRS	Wet	Marsh	3	2.25	0.43
J.flo2	SRS	Wet	Marsh	1	1.85	-
L.goo1	SRS	Wet	Marsh	24	2.37	0.47
L.goo2	SRS	Wet	Marsh	27	2.54	0.40
L.mac1	SRS	Wet	Marsh	17	2.67	0.47
L.mic1	SRS	Wet	Marsh	3	1.97	0.84
L.mic2	SRS	Wet	Marsh	3	2.83	0.52
L.pun2	SRS	Wet	Marsh	3	2.58	0.52
M.sia1	SRS	Wet	Marsh	1	3.00	-
O.aur1	SRS	Wet	Marsh	10	2.51	0.84
P.axa1	SRS	Wet	Marsh	1	2.50	-
P.lat2	SRS	Wet	Marsh	2	2.75	0.35
S.lac1	SRS	Wet	Marsh	1	2.00	-
A.cal2	SRS	Wet	Pond	4	2.27	0.52
A.gry1	SRS	Wet	Pond	2	2.69	0.44
A.nat3	SRS	Wet	Pond	1	2.96	-
B.bel1	SRS	Wet	Pond	1	3.17	-
C.bat1	SRS	Wet	Pond	1	2.50	-

C.uro1	SRS	Wet	Pond	6	2.01	0.83
C.uro2	SRS	Wet	Pond	2	1.58	0.82
E.eve1	SRS	Wet	Pond	19	2.78	0.52
E.glo1	SRS	Wet	Pond	1	2.46	1.00
E.suc1	SRS	Wet	Pond	5	2.53	0.63
E.suc2	SRS	Wet	Pond	2	2.69	0.37
F.chr1	SRS	Wet	Pond	1	2.29	0.56
F.chr2	SRS	Wet	Pond	1	2.90	-
G.hol1	SRS	Wet	Pond	18	2.30	0.65
G.hol2	SRS	Wet	Pond	18	2.45	0.42
H.for1	SRS	Wet	Pond	6	2.78	0.64
H.for2	SRS	Wet	Pond	5	2.54	0.31
H.let1	SRS	Wet	Pond	14	2.62	0.47
J.flo1	SRS	Wet	Pond	2	1.88	1.24
J.flo2	SRS	Wet	Pond	1	2.67	-
L.goo1	SRS	Wet	Pond	4	2.63	0.36
L.goo2	SRS	Wet	Pond	16	2.21	0.63
L.gul1	SRS	Wet	Pond	1	1.73	-
L.gul2	SRS	Wet	Pond	4	2.21	0.83
L.gul3	SRS	Wet	Pond	9	2.44	0.62
L.gul4	SRS	Wet	Pond	3	2.51	0.56
L.mac1	SRS	Wet	Pond	6	2.47	0.74
L.mar1	SRS	Wet	Pond	3	3.18	0.19
L.mic1	SRS	Wet	Pond	2	2.25	0.35
L.mic2	SRS	Wet	Pond	4	2.63	0.25
L.pla1	SRS	Wet	Pond	13	2.63	0.53
L.pla2	SRS	Wet	Pond	4	2.73	0.43
L.pun2	SRS	Wet	Pond	7	2.63	0.27
L.pun3	SRS	Wet	Pond	1	2.50	-
M.alb4	SRS	Wet	Pond	1	2.10	-
M.sal1	SRS	Wet	Pond	7	2.68	0.37
M.sia1	SRS	Wet	Pond	6	2.68	0.84
N.pet1	SRS	Wet	Pond	15	2.75	0.35
O.aur1	SRS	Wet	Pond	4	2.63	0.25
P.lat1	SRS	Wet	Pond	5	2.24	0.88
P.lat2	SRS	Wet	Pond	8	2.38	0.64
S.lac1	SRS	Wet	Pond	1	2.50	-
T.mar1	SRS	Wet	Pond	5	2.30	0.27
A.nat1	SRS	Dry	Marsh	2	2.63	0.18
B.bel1	SRS	Dry	Marsh	1	2.50	-

C.uro1	SRS	Dry	Marsh	4	2.86	0.33
E.ame1	SRS	Dry	Marsh	1	2.50	-
E.eve1	SRS	Dry	Marsh	8	2.50	0.70
E.glo1	SRS	Dry	Marsh	5	2.51	0.44
E.suc1	SRS	Dry	Marsh	4	2.43	0.29
F.chr1	SRS	Dry	Marsh	9	2.34	0.71
F.chr2	SRS	Dry	Marsh	1	1.00	-
F.con1	SRS	Dry	Marsh	2	2.20	0.42
G.hol1	SRS	Dry	Marsh	7	1.92	0.89
G.hol2	SRS	Dry	Marsh	5	2.58	0.71
H.for1	SRS	Dry	Marsh	1	1.25	-
H.for2	SRS	Dry	Marsh	5	2.48	0.47
H.let1	SRS	Dry	Marsh	33	2.45	0.50
H.lit1	SRS	Dry	Marsh	3	2.81	0.27
J.flo2	SRS	Dry	Marsh	1	3.12	-
L.goo1	SRS	Dry	Marsh	18	2.42	0.43
L.goo2	SRS	Dry	Marsh	25	2.58	0.54
L.mar1	SRS	Dry	Marsh	3	2.56	0.26
L.mic1	SRS	Dry	Marsh	1	3.50	-
L.pun2	SRS	Dry	Marsh	7	2.41	0.29
L.pun3	SRS	Dry	Marsh	1	1.00	-
N.gyr1	SRS	Dry	Marsh	1	3.00	-
N.vir1	SRS	Dry	Marsh	1	2.67	-
O.aur1	SRS	Dry	Marsh	1	2.00	-
P.axa1	SRS	Dry	Marsh	2	3.08	0.12
P.lat2	SRS	Dry	Marsh	2	2.75	0.35
R.gry1	SRS	Dry	Marsh	6	2.49	0.36
S.lac1	SRS	Dry	Marsh	3	1.80	0.78
T.mar1	SRS	Dry	Marsh	4	2.38	0.95
A.nat1	SRS	Dry	Pond	3	2.42	1.28
A.nat2	SRS	Dry	Pond	4	3.00	0.41
A.nat3	SRS	Dry	Pond	6	2.23	0.60
A.nat4	SRS	Dry	Pond	8	2.56	0.48
B.bel1	SRS	Dry	Pond	2	2.50	0.00
C.bat1	SRS	Dry	Pond	1	1.10	-
C.man1	SRS	Dry	Pond	1	3.37	-
C.uro2	SRS	Dry	Pond	1	3.50	-
E.eve1	SRS	Dry	Pond	1	1.00	-
E.suc1	SRS	Dry	Pond	2	2.41	0.40
E.suc2	SRS	Dry	Pond	3	2.83	0.58

F.chr1	SRS	Dry	Pond	11	2.27	0.50
F.chr2	SRS	Dry	Pond	1	3.00	-
F.con1	SRS	Dry	Pond	1	2.50	-
G.hol1	SRS	Dry	Pond	16	2.55	0.32
G.hol2	SRS	Dry	Pond	27	2.47	0.50
H.for1	SRS	Dry	Pond	5	2.33	0.55
H.for2	SRS	Dry	Pond	5	2.35	0.49
H.lit1	SRS	Dry	Pond	8	2.66	0.44
J.flo1	SRS	Dry	Pond	1	2.30	-
J.flo2	SRS	Dry	Pond	3	2.67	0.14
L.goo1	SRS	Dry	Pond	22	2.34	0.55
L.goo2	SRS	Dry	Pond	23	2.51	0.54
L.gul1	SRS	Dry	Pond	1	3.00	-
L.gul2	SRS	Dry	Pond	9	2.37	0.57
L.gul3	SRS	Dry	Pond	9	2.37	0.60
L.gul4	SRS	Dry	Pond	4	2.42	0.55
L.mac1	SRS	Dry	Pond	2	2.69	0.80
L.mar1	SRS	Dry	Pond	7	2.40	0.63
L.pla1	SRS	Dry	Pond	11	2.46	0.59
L.pun2	SRS	Dry	Pond	4	2.64	0.16
L.pun3	SRS	Dry	Pond	2	1.50	0.71
M.sal1	SRS	Dry	Pond	1	3.00	-
N.flo1	SRS	Dry	Pond	1	3.00	-
P.lat2	SRS	Dry	Pond	6	2.88	0.56
R.gry2	SRS	Dry	Pond	1	2.50	-
S.lac1	SRS	Dry	Pond	2	2.08	0.12

Table S15. Statistical comparisons of trophic positions by species size classes modeled from stable isotopes of carbon and nitrogen across habitats (A) and seasons (B).

A. Habitat Comparisons

Species	Slough	Season	Marsh	Pond	Marsh < Pond	Percent Change
A. natalis	SRS	Wet	4.2	3.67	0.13	-12.48
Amphipoda			2.12	2.2	0.67	3.73
Celithemus spp.			2.3	2.48	0.88	7.97
C. urophthalmus			3.52	3.36	0.14	-4.66
C. batrachus			3.82	3.64	0.35	-4.84
Coenagrionidae			2.41	2.52	0.65	4.36
Copepoda			2.29	2.04	0.11	-10.96
E. gloriosus			3.71	3.47	0.08	-6.45
E. sucetta			3.46	3.2	0.22	-7.49
F. chrysotus			3.82	3.79	0.44	-0.6
G. holbrooki			3.56	3.63	0.74	2.05
H. letourneuxi			3.63	3.44	0.08	-5.05
H. formosa			3.64	3.84	0.91	5.35
J. floridae			3.21	3.52	0.9	9.57
L. macrochirus			3.97	3.34	0	-15.88
L. marginatus			3.35	3.25	0.32	-2.92
L. microlophus			3.77	3.25	0	-13.8
L. punctatus			3.96	3.74	0.16	-5.5
L. incesta			2.21	2.26	0.53	1.94
L. goodei			3.35	3.53	0.93	5.41
M. siamensis			3.48	3.36	0.33	-3.59
O. aureus			3.73	3.42	0.1	-8.36
P. paludosus			3.17	3.54	0.88	11.77
P. femoratus			2.19	2.14	0.43	-2.15
P. latipinna			3.56	3.49	0.33	-1.99
P. fallax			2.51	2.26	0.16	-9.87
Procambarus spp.			2.5	2.51	0.52	0.28
Amphipoda		Dry	4.94	2.26	0.06	-54.35
B. belizanus			3.94	4.06	0.64	3.1
Celithemus spp.			2.1	2.07	0.4	-1.38
C. urophthalmus			3.5	3.28	0.13	-6.37
E. evergladei			3.52	3.34	0.23	-5.28

E. sucetta			3.29	3.47	0.77	5.41
F. chrysotus			3.55	3.79	0.84	6.72
F. confluentus			3.56	3.37	0.34	-5.31
G. holbrooki			3.28	3.68	0.96	11.97
H. formosa			3.39	3.83	1	13.16
H. littorale			3.52	3.46	0.41	-1.79
J. floridae			3.25	3.33	0.65	2.65
L. marginatus			3.48	3.52	0.58	1.06
L. punctatus			3.6	3.65	0.62	1.42
L. goodei			3.2	3.44	0.97	7.66
P. paludosus			2.9	3.16	0.9	8.99
Planorbella spp.			2.21	2.65	0.84	20.01
P. alleni			2.48	2.28	0.16	-8.18
S. lacertina			3.16	3.17	0.53	0.32

B. Seasonal Comparisons

Species	Slough	Habitat	Wet	Dry	Wet < Dry	Percent Change
Amphipoda	SRS	Marsh	2.12	4.94	**0.97**	133.1
Celithemus spp.			2.3	2.1	0.21	-8.5
C. urophthalmus			3.52	3.5	0.45	-0.6
E. evergladei			3.18	3.52	**0.95**	10.8
E. gloriosus			3.71	3.31	0.11	-10.8
E. sucetta			3.46	3.29	0.33	-4.8
F. chrysotus			3.82	3.55	0.14	-6.9
G. holbrooki			3.56	3.28	**0.05**	-7.8
H. letourneuxi			3.63	3.57	0.36	-1.5
H. formosa			3.64	3.39	**0.05**	-7.0
J. floridae			3.21	3.25	0.55	1.3
L. marginatus			3.35	3.48	0.68	3.8
L. punctatus			3.96	3.6	0.07	-9.3
L. goodei			3.35	3.2	0.12	-4.4
P. paludosus			3.17	2.9	0.14	-8.4
P. femoratus			2.19	2.09	0.33	-4.3
Periphyton (Mat)			2.44	2.4	0.48	-1.6
A. natalis		Pond	3.67	3.87	0.82	5.3
Amphipoda			2.2	2.26	0.64	2.6
B. belizanus			4.16	4.06	0.36	-2.2
B. gravida			2.25	2.71	**0.99**	20.4
Celithemus spp.			2.48	2.07	**0.01**	-16.5

C. urophthalmus	3.36	3.28	0.23	-2.4
Copepoda	2.04	2.64	**1**	29.6
E. sucetta	3.2	3.47	0.93	8.5
F. chrysotus	3.79	3.79	0.49	0.0
F. confluentus	3.42	3.37	0.39	-1.2
G. holbrooki	3.63	3.68	0.61	1.1
H. formosa	3.84	3.83	0.49	-0.1
J. floridae	3.52	3.33	0.11	-5.2
L. gulosus	3.73	3.84	0.81	3.0
L. marginatus	3.25	3.52	0.94	8.1
L. microlophus	3.25	3.21	0.41	-1.4
L. platyrhincus	4.33	4.38	0.6	1.0
L. punctatus	3.74	3.65	0.24	-2.6
L. goodei	3.53	3.44	0.23	-2.3
P. paludosus	3.54	3.16	0.07	-10.7
P. latipinna	3.49	3.35	0.16	-4.1
P. alleni	2.6	2.28	0.03	-12.5
P. fallax	2.26	2.33	0.64	3.1

Table S16. Alpha, the proportion of δ^{15}N, derived from detritus (floc), and one minus alpha is the proportion of δ^{15}N derived from autotrophic energy (green algae). Post-invasion, these are statistically compared across habitats (A) and seasons (B). Statistical differences are in bold. Trends from dry season to wet season are a proxy for restoration effects.

A. Habitat Comparisons

Species	Slough	Season	Marsh	Pond	Marsh < Pond	Percent Change
A. natalis	SRS	Wet	0.2	0.57	0.9	190.8
Amphipoda			0.81	0.65	0.33	-18.9
Celithemus spp.			0.06	0.05	0.4	-19.4
C. urophthalmus			0.44	0.49	0.56	10.2
C. batrachus			0.22	0.41	0.77	87.3
Coenagrionidae			0.46	0.13	0.25	-71.8
Copepoda			0.82	0.91	0.68	10.6
E. gloriosus			0.15	0.22	0.61	44.7
E. sucetta			0.19	0.2	0.49	3.2
F. chrysotus			0.82	0.74	0.36	-10.0
G. holbrooki			0.31	0.53	0.91	71.1
H. letourneuxi			0.24	0.17	0.32	-28.3
H. formosa			0.12	0.38	0.84	231.9
J. floridae			0.09	0.25	0.7	192.9
L. macrochirus			0.42	0.34	0.35	-18.4
L. marginatus			0.61	0.2	0.08	-67.6
L. microlophus			0.18	0.23	0.59	30.3
L. punctatus			0.24	0.36	0.66	46.7
L. incesta			0.19	0.52	0.84	174.1
L. goodei			0.16	0.32	0.83	92.7
M. siamensis			0.61	0.74	0.72	22.5
O. aureus			0.72	0.2	0.11	-72.6
P. paludosus			0.6	0.52	0.45	-12.2
P. femoratus			0.59	0.6	0.51	2.6
P. latipinna			0.4	0.03	0.11	-91.6
P. fallax			0.67	0.77	0.63	14.6
Procambarus spp.			0.13	0.41	0.83	217.7
Amphipoda		Dry	0.5	0.66	0.59	32.5
B. belizanus			0.38	0.52	0.66	37.3
Celithemus spp.			0.78	0.83	0.6	6.1
C. urophthalmus			0.24	0.29	0.54	20.5
E. evergladei			0.36	0.31	0.43	-14.2
E. sucetta			0.27	0.23	0.46	-13.1
F. chrysotus			0.79	0.85	0.61	8.1

F. confluentus			0.37	0.61	0.75	66.2
G. holbrooki			0.12	0.6	0.92	383.9
H. formosa			0.15	0.08	0.32	-50.3
H. littorale			0.36	0.15	0.26	-58.3
J. floridae			0.28	0.23	0.42	-16.0
L. marginatus			0.14	0.38	0.79	171.9
L. punctatus			0.15	0.41	0.79	175.5
L. goodei			0.26	0.21	0.39	-19.8
P. paludosus			0.6	0.12	0.1	-80.5
Planorbella spp.			0.52	0.58	0.55	12.4
P. alleni			0.67	0.79	0.71	19.2
S. lacertina			0.29	0.35	0.58	22.2

B. Seasonal Comparisons

Species	Slough	Habitat	Wet	Dry	Wet < Dry	Percent Change
Amphipoda	SRS	Marsh	0.81	0.5	0.22	-37.7
Celithemus spp.			0.06	0.78	0.94	1164.5
C. urophthalmus			0.44	0.24	0.32	-44.7
E. evergladei			0.09	0.36	0.85	313.8
E. gloriosus			0.15	0.26	0.61	74.3
E. sucetta			0.19	0.27	0.57	41.1
F. chrysotus			0.82	0.79	0.46	-3.3
G. holbrooki			0.31	0.12	0.22	-59.7
H. letourneuxi			0.24	0.11	0.29	-55.3
H. formosa			0.12	0.15	0.58	31.9
J. floridae			0.09	0.28	0.68	223.5
L. marginatus			0.61	0.14	0.14	-77.1
L. punctatus			0.24	0.15	0.38	-39.8
L. goodei			0.16	0.26	0.69	56.4
P. paludosus			0.6	0.6	0.49	-0.3
P. femoratus			0.59	0.58	0.5	-0.7
Periphyton (Mat)			0.74	0.81	0.58	8.3
A. natalis		Pond	0.57	0.4	0.15	-30.0
Amphipoda			0.65	0.66	0.51	1.7
B. belizanus			0.47	0.52	0.56	9.3
B. gravida			0.07	0.2	0.84	194.0
Celithemus spp.			0.05	0.83	**1**	1564.0
C. urophthalmus			0.49	0.29	0.19	-39.5
Copepoda			0.91	0.69	0.21	-24.0
E. sucetta			0.2	0.23	0.59	18.9
F. chrysotus			0.74	0.85	0.72	16.2
F. confluentus			0.83	0.61	0.11	-26.9
G. holbrooki			0.53	0.6	0.67	13.9

H. formosa	0.38	0.08	0.09	-80.3
J. floridae	0.25	0.23	0.45	-7.2
L. gulosus	0.39	0.47	0.68	19.1
L. marginatus	0.2	0.38	0.85	91.9
L. microlophus	0.23	0.7	**0.96**	200.0
L. platyrhincus	0.3	0.26	0.4	-15.6
L. punctatus	0.36	0.41	0.58	13.1
L. goodei	0.32	0.21	0.21	-34.9
P. paludosus	0.52	0.12	0.13	-77.9
P. latipinna	0.03	0.05	0.62	52.9
P. alleni	0.86	0.79	0.33	-7.6
P. fallax	0.77	0.95	0.88	23.8

Table S17. Change in ash-free dry mass density (g AFDM · m^{-2}) per guild pre- vs post-invasion in both seasons and the percent change (%Δ).

	Pre-Invasion		Post-Invasion		%Δ	
Guild	Dry	Wet	Dry	Wet	Dry	Wet
Carn. Inverts	0.07	0.05	0.05	0.04	-27.08	-14.49
Decapods	0.53	0.30	0.41	0.28	-23.55	-6.81
Detritivores	-	-	0.15	0.13	-	-
Detritus	10.00	10.00	10.00	10.00	0.00	0.00
Herb. Inverts	0.34	0.31	0.09	0.10	-74.08	-69.33
Mesopredators	58.72	4.26	5.27	24.28	-91.03	470.10
Omni. Inverts	0.00	0.00	0.00	0.00	1320.07	1111.74
Periphyton	64.29	90.66	89.11	99.97	38.61	10.27
Piscivores	0.05	0.41	0.29	0.24	437.96	-41.53
Sm. Fishes	9.29	6.03	3.91	4.22	-57.87	-30.06

Table S18. Changes in energy fluxes ($J \cdot s^{-1} \cdot m^{-2}$) among nodes (guilds) pre- vs post-invasion in both seasons.

Pre-Wet

	Carn. Inverts	Decapods	Detritus	Herb. Inverts	Mesopredators	Omni. Inverts	Periphyton	Piscivores	Sm. Fishes
Carn. Inverts	0	0	0	0	1.8279E-05	0	0	0	9.7721E-06
Decapods	0	0	0	0	7.704E-05	0	0	0	4.8842E-07
Detritus	0	2.3139E-05	0	0	9.3155E-05	1.58471E-06	0	0	7.6611E-05
Herb. Inverts	3.477E-05	7.2422E-07	0	0	0.00018222	4.95984E-08	0	0	0.00015815
Mesopredators	0	0	0	0	0	0	0	0	0
Omni. Inverts	2.5492E-08	5.3097E-10	0	0	3.9578E-06	3.63638E-11	0	0	2.4233E-06
Periphyton	0	0.00020978	0	0.00098616	0.00025336	1.43666E-05	0	0	0.00270428
Piscivores	0	0	0	0	0	0	0	0	0
Sm. Fishes	0	0	0	0	0.00047759	0	0	7.3692E-05	0.00278237

Pre-Dry

	Carn. Inverts	Decapods	Detritus	Herb. Inverts	Mesopredators	Omni. Inverts	Periphyton	Piscivores	Sm. Fishes
Carn. Inverts	0	0	0	0	0.00014379	0	0	0	0.00016257
Decapods	0	0	0	0	0.00225811	0	0	0	0.00018697
Detritus	0	0.00082327	0	0	0.0007566	1.28319E-05	0	0	0.00036369
Herb. Inverts	0.0003437	2.8335E-05	0	0	0.00081377	4.41647E-07	0	0	0.00044506
Mesopredators	0	0	0	0	0	0	0	0	0
Omni. Inverts	2.8226E-07	2.327E-08	0	0	2.4633E-05	3.62696E-10	0	0	1.3569E-05
Periphyton	0	0.00529258	0	0.00412925	0.00101333	8.24927E-05	0	0	0.01749206
Piscivores	0	0	0	0	0	0	0	0	0
Sm. Fishes	0	0	0	0	0.00673588	0	0	9.3599E-06	0.00596888

Post-Wet

	Carn.	Decapods	Detritivores	Detritus	Herb. Inverts	Mesopredators	Omni.	Periphyton	Piscivores	Sm. Fishes

	Inverts						Inverts			
Carn. Inverts	0	0	0	0	0	8.7542E-05	0	0	0	4.3388E-06
Decapods	0	0	0	0	0	3.05796E-05	0	0	1.2677E-06	2.9262E-07
Detritivores	0	0	0	0	0	0	0	0	0	0
Detritus	0	1.0568E-05	3.20634E-05	0	0	0.002887055	3.23694E-06	0	1.5132E-05	0.00023054
Herb. Inverts	0.00010174	1.0145E-07	0	0	0	0.000359332	3.10731E-08	0	0	6.2368E-06
Mesopredators	0	0	0	0	0	0	0	0	0	0
Omni. Inverts	2.9471E-06	2.9385E-09	5.3492E-09	0	0	9.93755E-06	9.00041E-10	0	0	1.1509E-06
Periphyton	0	0.00010565	0	0	0.00118283	0.003980837	3.23588E-05	0	7.5637E-05	0.00345696
Piscivores	0	0	0	0	0	0	0	0	0	0
Sm. Fishes	0	0	4.05774E-06	0	0	0.001133898	0	0	3.1918E-06	4.4205E-06

	Post-Dry									
	Carn. Inverts	Decapods	Detritivores	Detritus	Herb. Inverts	Mesopredators	Omni. Inverts	Periphyton	Piscivores	Sm. Fishes
Carn. Inverts	0	0	0	0	0	3.47658E-06	0	0	0	1.4986E-06
Decapods	0	0	0	0	0	1.12112E-05	0	0	0	0
Detritivores	0	0	0	0	0	0	0	0	0	0
Detritus	0	8.198E-06	1.15697E-05	0	0	0.000805237	9.9175E-07	0	0	0.00012171
Herb. Inverts	9.3255E-06	7.3149E-08	0	0	0	9.2057E-06	8.84915E-09	0	0	6.1089E-07
Mesopredators	0	0	0	0	0	0	0	0	4.0343E-05	0
Omni. Inverts	4.195E-07	3.2906E-09	0	0	0	1.83826E-06	3.98074E-10	0	0	1.3572E-06
Periphyton	0	7.305E-05	0	0	6.1277E-05	0.002018012	8.8371E-06	0	0	0.00091508
Piscivores	0	0	0	0	0	0	0	0	0	0
Sm. Fishes	0	0	0	0	0	0.000108328	0	0	2.9974E-05	1.4885E-06

Supplementary Figure Captions

Figure S1. Relative biomass (top panel) and abundance (bottom panel) of Jewelfish compared to all fishes at sites in Shark River Slough, Everglades, FL, USA.

Figure S2. Fish community structure at sites 6 and 23 in Shark River Slough from 1979 through 2021. Ellipses represent 95% confidence intervals around time-periods centroids. Time periods are: “Pre-Stomachs” – 1978-1981 (when pre-invasion stomach content data were collected), “Pre-MDW” – 1982-1989 (indicates sampling prior to the current iteration of the long-term monitoring project), “Pre-Isotopes” – 1990-1995 (when the pre-invasion stable isotopes samples were collected, also prior to the current iteration of the monitoring project), “Pre-Invasion” – 1996-1999 (years immediately preceding Jewelfish invasion, current monitoring regime began in 1996), “Pre-Boom” – 2000-2011 (period of time from Jewelfish invasion in 2000 up until the increase in their population), “Boom” – 2012-2017 (years of elevated Jewelfish relative biomass and abundance), “Sampling” – 2018-2019 (when sampling for this project occurred), “Post-Boom” – 2020-2021 (years after sampling for this project while Jewelfish were at low levels of relative abundance and biomass).

Figure S3. Average abundance of prey items in consumer stomachs for shared habitats and seasons between pre- and post-invasion.

Figure S4. NMDS of diets using counts of stomach contents from shared slough-habitat-season levels pre- and post-invasion

Figure S5. Trophic niche area derived from stable isotopes for taxa pre- vs post-invasion. *P. fallax* is omitted from the plot because of the large size of its pre-invasion niche (Table 1).

Figure S6. Effect sizes of A) habitat and B) season shifts post-invasion compared to pre-invasion.

Figure S7. Post-invasion trophic positions from stable isotopes among slough-habitat-season levels.

Figure S8. Post-invasion alpha among slough-habitat-season levels.

Figure S9. Proportional energy fluxes in and out of different nodes (guilds).

Figure S1.

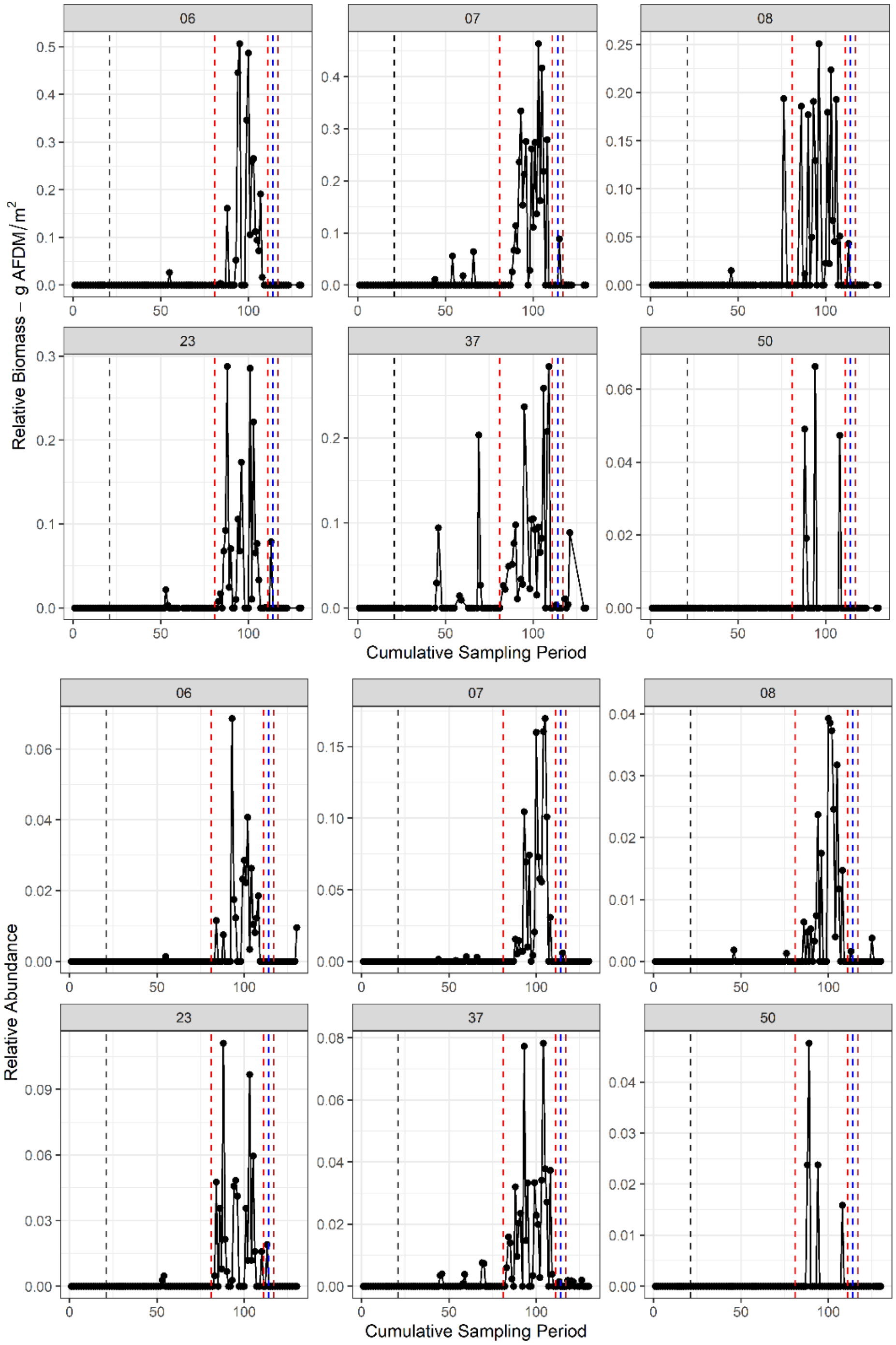
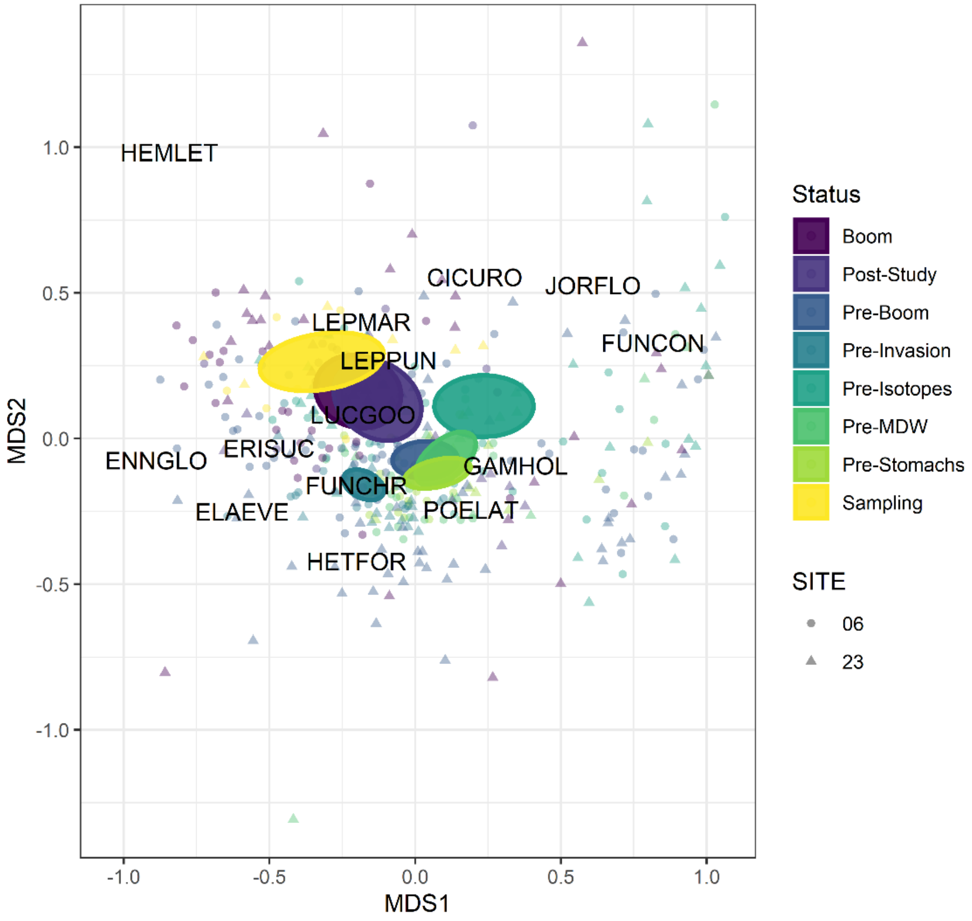
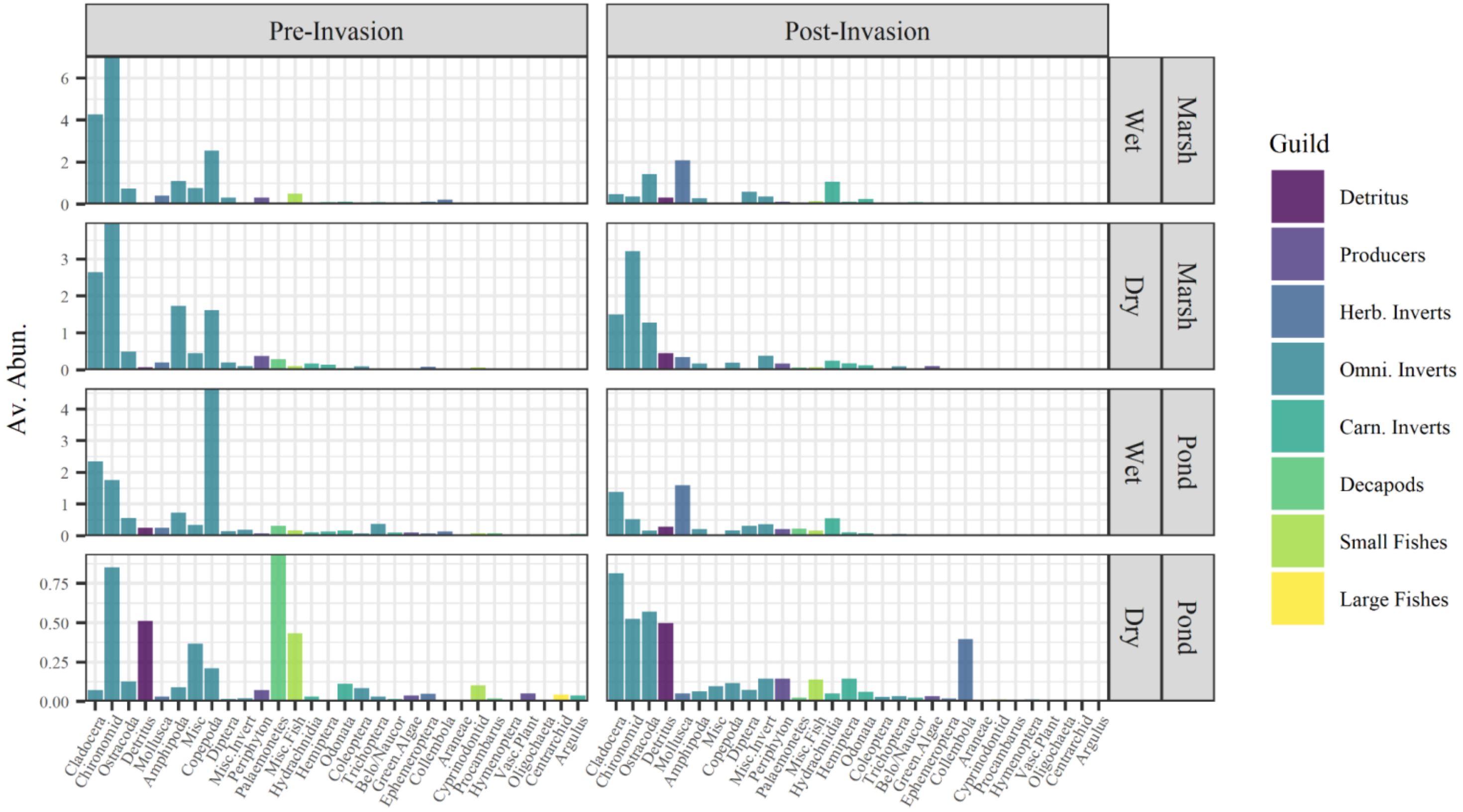

06
07
08
23
37
50
Relative Biomass – g AFDM/m^2
0.0
0.1
0.2
0.3
0.4
0.5
0.00
0.05
0.10
0.15
0.20
0.25
0.02
0.04
0.06
0
50
100
Cumulative Sampling Period
06
07
08
23
37
50
Relative Abundance
0.00
0.02
0.04
0.06
0.05
0.10
0.15
0.01
0.03
0.09
0.08
0
50
100
Cumulative Sampling Period

Figure S2.



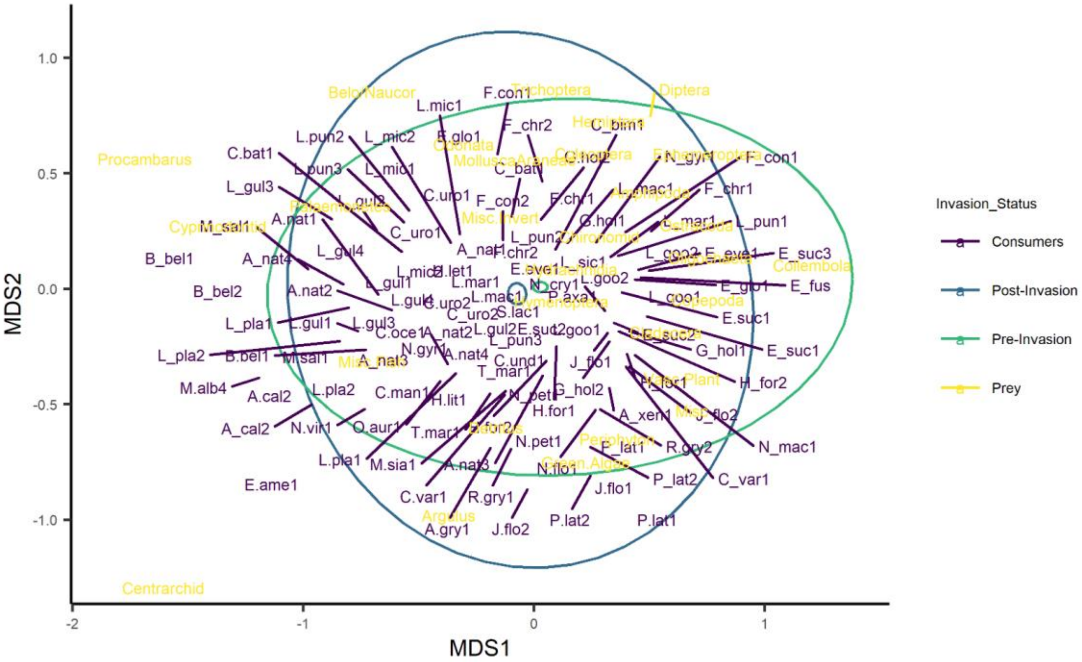
HEMLET
CICURO
JORFLO
LEPMAR
FUNCON
LEPPUN
LUCGOO
ENNGLO
ERISUC
GAMHOL
FUNCHR
ELAEVE
POELAT
HETFOR
1.0
0.5
0.0
-0.5
-1.0
MDS2
-1.0
-0.5
0.0
0.5
1.0
MDS1
Status
Boom
Post-Study
Pre-Boom
Pre-Invasion
Pre-Isotopes
Pre-MDW
Pre-Stomachs
Sampling
SITE
06
23

Figure S3.



Pre-Invasion
Post-Invasion
Wet
Marsh
Dry
Marsh
Wet
Pond
Dry
Pond
Av. Abun.
0
2
4
6
0
1
2
3
0
1
2
3
4
0.00
0.25
0.50
0.75
Cladocera
Chironomid
Ostracoda
Detritus
Mollusca
Amphipoda
Misc
Copepoda
Diptera
Misc.Invert
Periphyton
Palaemonetes
Misc.Fish
Hydrachnidia
Hemiptera
Odonata
Coleoptera
Trichoptera
Belo/Naucor
Green.Algae
Ephemeroptera
Collembola
Araneae
Cyprinodontid
Procambarus
Hymenoptera
Vasc.Plant
Oligochaeta
Centrarchid
Argulus
Guild
Detritus
Producers
Herb. Inverts
Omni. Inverts
Carn. Inverts
Decapods
Small Fishes
Large Fishes

Figure S4.



MDS2
MDS1
1.0
0.5
0.0
-0.5
-1.0
-2
-1
0
1
Invasion_Status
Consumers
Post-Invasion
Pre-Invasion
Prey
Belo/Naucor
Trichoptera
Diptera
Hemiptera
Odonata
Mollusca
Araneae
Coleoptera
Ephemeroptera
Procambarus
Palaemonetes
Cyprinodontid
Misc.Invert
Amphipoda
Ostracoda
Chironomid
Oligochaeta
Collembola
Hydrachnidia
Hymenoptera
Copepoda
Cladocera
Vasc.Plant
Misc.Fish
Misc
Detritus
Periphyton
Green.Algae
Argulus
Centrarchid
F.con1
L.mic1
L.pun2
L_mic2
E.glo1
F_chr2
C_bim1
C.bat1
L.pun3
L_mic1
C_bat1
G.hol2
N_gyr1
F_con1
L_gul3
L_gul2
C.uro1
F_con2
F.chr1
L_mac1
F_chr1
M_sal1
A.nat1
C_uro1
Misc.Invert
G.hol1
L_mar1
L_pun1
B_bel1
A_nat4
L_gul4
A_nat1
F.chr2
L_pun2
L_sic1
L_goo2
E_eve1
E_suc3
L.mic2
H.let1
E.eve1
L.mar1
N_cry1
L.goo2
E_glo1
E_fus
B_bel2
A.nat2
L_gul1
L.gul4
C.uro2
L.mac1
P_axa1
L_goo1
E.suc1
L_pla1
L.gul1
L.gul3
C_uro2
S.lac1
C.oce1
A_nat2
L.gul2
E.suc2
L.goo1
E_suc2
G_hol1
E_suc1
L_pla2
B.bel1
M.sal1
A_nat3
N.gyr1
A.nat4
L_pun3
C.und1
T_mar1
J_flo1
H_for1
H_for2
M.alb4
A.cal2
L.pla2
C.man1
H.lit1
N_pet1
G_hol2
H.for1
A_xen1
J_flo2
A_cal2
N.vir1
O.aur1
T.mar1
E.for2
N.pet1
E_lat1
R.gry2
N_mac1
L.pla1
M.sia1
A.nat3
N.flo1
P_lat2
C_var1
J.flo1
E.ame1
C.var1
R.gry1
A.gry1
J.flo2
P.lat2
P.lat1

Figure S5.

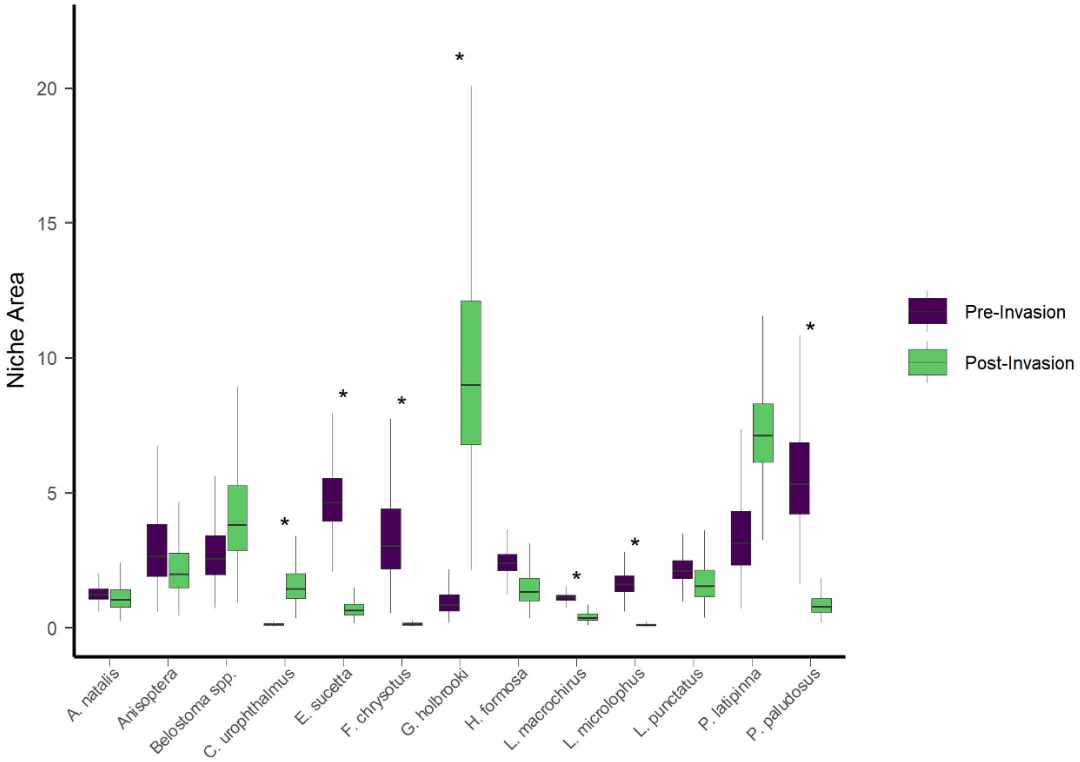

Niche Area
0
5
10
15
20
*
*
*
*
*
*
*
A. natalis
Anisoptera
Belostoma spp.
C. urophthalmus
E. sucetta
F. chrysotus
G. holbrooki
H. formosa
L. macrochirus
L. microlophus
L. punctatus
P. latipinna
P. paludosus
Pre-Invasion
Post-Invasion

Figure S6.

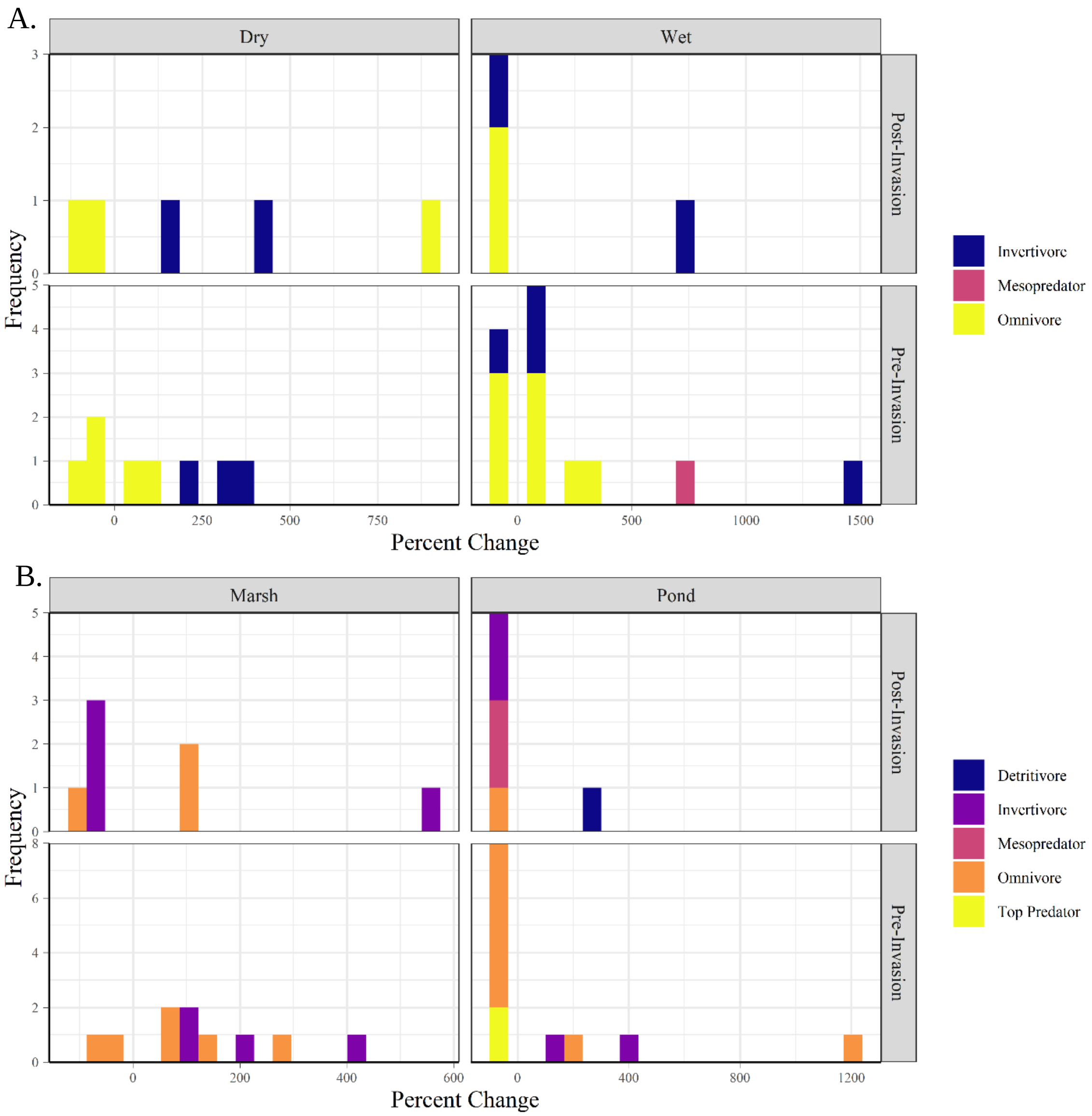

A.
Dry
Wet
Post-Invasion
Pre-Invasion
Frequency
0
1
2
3
4
5
0
250
500
750
0
500
1000
1500
Percent Change
Invertivore
Mesopredator
Omnivore
B.
Marsh
Pond
Post-Invasion
Pre-Invasion
Frequency
0
1
2
3
4
5
0
2
4
6
8
0
200
400
600
0
400
800
1200
Percent Change
Detritivore
Invertivore
Mesopredator
Omnivore
Top Predator

Figure S7.

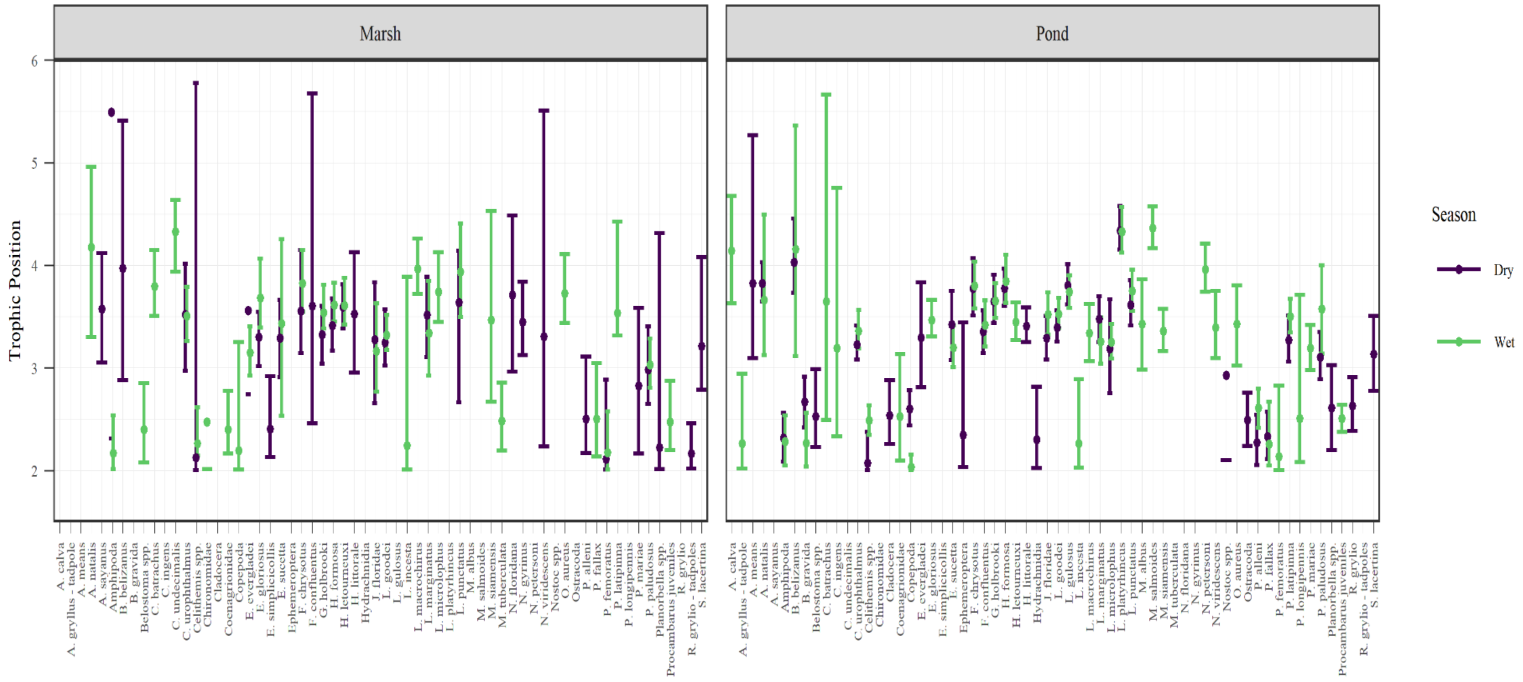

Marsh
Pond
Trophic Position
6
5
4
3
2
Season
Dry
Wet
A. calva
A. gryllus - tadpole
A. means
A. natalis
A. sayanus
Amphipoda
B. belizanus
B. gravida
Belostoma spp.
C. batrachus
C. ingens
C. undecimalis
C. urophthalmus
Celithemis spp.
Chironomidae
Cladocera
Coenagrionidae
Copepoda
E. evergladei
E. gloriosus
E. simplicicollis
E. sucetta
Ephemeroptera
F. chrysotus
F. confluentus
G. holbrooki
H. formosa
H. letourneuxi
H. littorale
Hydrachnidia
J. floridae
L. goodei
L. gulosus
L. incesta
L. macrochirus
L. marginatus
L. microlophus
L. platyrhincus
L. punctatus
M. albus
M. salmoides
M. siamensis
M. tuberculata
N. floridana
N. gyrinus
N. petersoni
N. viridescens
Nostoc spp.
O. aureus
Ostracoda
P. alleni
P. fallax
P. femoratus
P. latipinna
P. longipennis
P. mariae
P. paludosus
Planorbella spp.
Procambarus juveniles
R. grylio
R. grylio - tadpoles
S. lacertina

Figure S8.

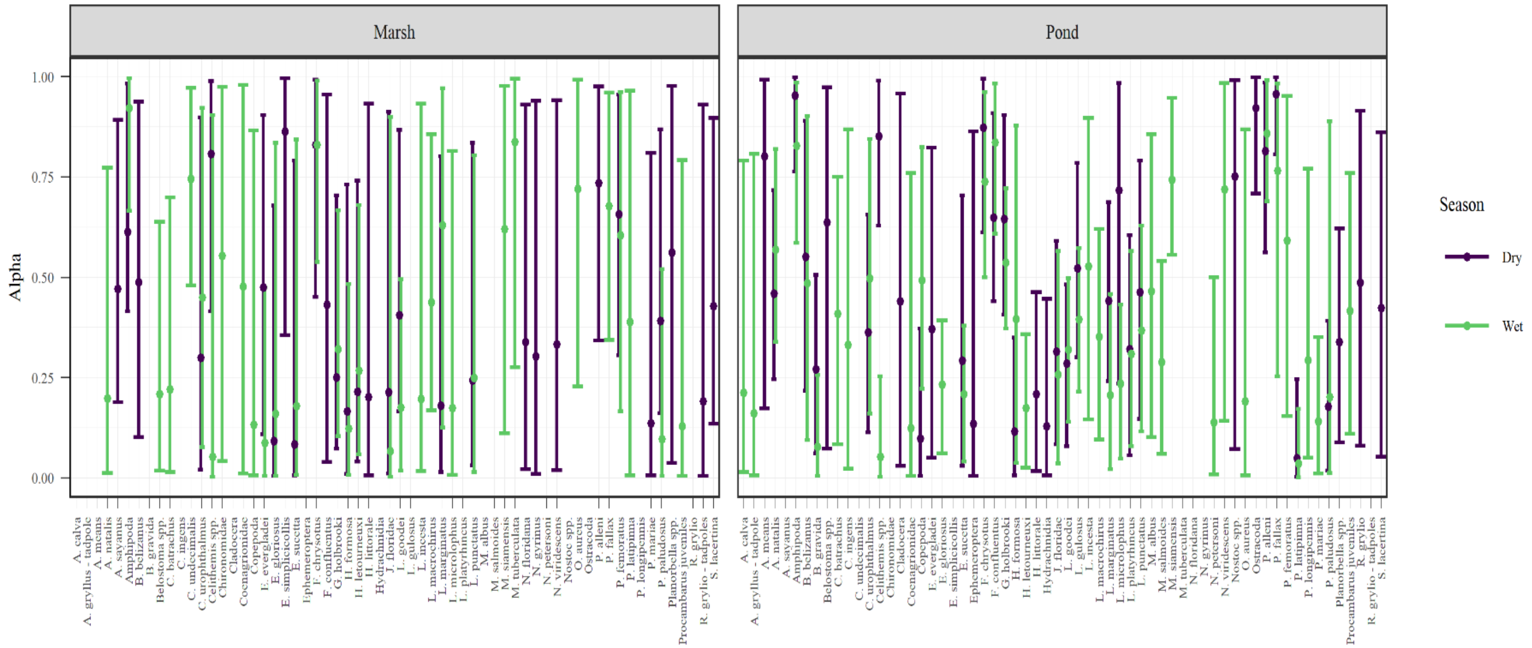
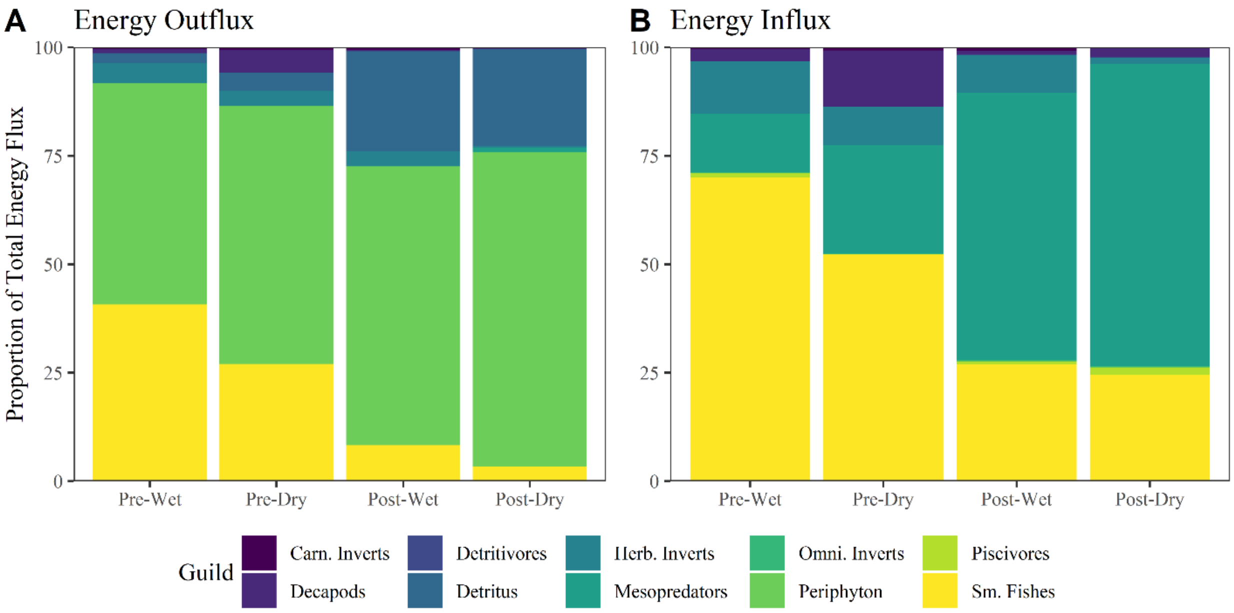

Marsh
Pond
Alpha
1.00
0.75
0.50
0.25
0.00
A. calva
A. gryllus - tadpole
A. means
A. natalis
A. sayanus
Amphipoda
B. belizanus
B. gravida
Belostoma spp.
C. batrachus
C. ingens
C. undecimalis
C. urophthalmus
Celithemis spp.
Chironomidae
Cladocera
Coenagrionidae
Copepoda
E. evergladei
E. gloriosus
E. simplicicollis
E. sucetta
Ephemeroptera
F. chrysotus
F. confluentus
G. holbrooki
H. formosa
H. letourneuxi
H. littorale
Hydrachnidia
J. floridae
L. goodei
L. gulosus
L. incesta
L. macrochirus
L. marginatus
L. microlophus
L. platyrhincus
L. punctatus
M. albus
M. salmoides
M. siamensis
M. tuberculata
N. floridana
N. gyrinus
N. petersoni
N. viridescens
Nostoc spp.
O. aureus
Ostracoda
P. alleni
P. fallax
P. femoratus
P. latipinna
P. longipennis
P. mariae
P. paludosus
Planorbella spp.
Procambarus juveniles
R. grylio
R. grylio - tadpoles
S. lacertina
Season
Dry
Wet

Figure S9.


A
Energy Outflux
B
Energy Influx
Proportion of Total Energy Flux
100
75
50
25
0
Pre-Wet
Pre-Dry
Post-Wet
Post-Dry
Guild
Carn. Inverts
Decapods
Detritivores
Detritus
Herb. Inverts
Mesopredators
Omni. Inverts
Periphyton
Piscivores
Sm. Fishes

CHAPTER V

GENERAL CONCLUSIONS

Global biodiversity loss has detrimental consequences for human wellbeing (Pimm et al. 2014). Governments worldwide have entered into international agreements to address and mitigate these losses (CBD 2010, UN 2015). Invasive species are one aspect of anthropogenic-induced biodiversity loss that are responsible for ecological and economic damage across the globe (Vitousek et al. 1996, Mooney and Cleland 2001). This includes critical, often unexpected, changes at all levels of ecological organization from individuals to ecosystems and these impacts are accelerating (Simberloff et al. 2013, Flood et al. 2020, Pyšek et al. 2020). For example, invasive species are responsible for hundreds of billions of dollars in damages each year in the United States (Pimentel et al. 2005) and for declines in approximately half of taxa listed by the International Union for Conservation of Nature Red List and US Endangered Species Act (Clavero and García-Berthou 2005). If management efforts are to be successful, food-web theory, and the factors that alter food webs must be part of the strategic response (Catford et al. 2012, McDonald-Madden et al. 2016, Strassburg et al. 2020). Therefore, advancing food-web theory by increasing our understanding of the impacts of spatiotemporal variation, ecosystem engineering, and invasive species on food webs is more valuable now than ever.

Everglades restoration is the largest restoration project in human history, a multi-decade, multi-billion-dollar effort that recognized the importance of food-web theory by developing the Everglades Trophic Hypothesis – hydrological restoration is predicted to restore dynamics of small-fish and invertebrate communities permitting recovery of higher trophic level taxa (e.g., wading birds) (Trexler and Goss 2009, Paudel et al. 2020). However, until this dissertation there were no direct comparisons using field data for how spatiotemporal variation, ecosystem engineering by alligators, or invasive fishes impacted trophic dynamics and therein the Everglades Trophic Hypothesis. Invasive fishes have been largely overlooked by Everglades restoration planning and lack of data has sometimes been correlated with lack of effects (Schofield and Loftus 2015). In fact, restoration activities have facilitated their introductions to ENP (Kline et al. 2013). Here, I quantified the effects of these three distinct but related drivers (spatiotemporal variation, ecosystem engineering, and invasive species) on food-web structure and function (Chapter 2, 3) and ecosystem function (Chapter 4).

In Chapter 2, by modeling food-web metrics from diets to network structure between habitats and among season I was able to define a pre-Jewelfish invasion baseline condition that included spatiotemporal trophic dynamics. I found that fishes in an oligotrophic, seasonally pulsed wetland showed spatiotemporal shifts in trophic niches and diets that were not associated with shifts in trophic position. Throughout the flood, most species were tracking variable yet trophically similar prey. Varying levels of piscivory, detrital consumption, and diet plasticity drove spatiotemporal trophic dynamics. This spatiotemporal variability in flexible omnivory is a critical food-web attribute that helps maintain energy and nutrient cycling, facilitates species coexistence, and influences ecosystem stability (Post and Takimoto 2007, Wootton 2017, McMeans et al. 2019). These dynamics are not confined to fishes or aquatic ecosystems, and are critical to our understanding of anthropogenic stressors such as climate change, hydrologic alteration, and invasive species. Trophic plasticity in space and time seems to be an important factor in ecosystem stability and resiliency.

Little is known about the impacts of ecosystem engineers on food webs (Sanders et al. 2014). Chapter 3 revealed the effects of alligator-engineering on Everglades trophic dynamics, found support for the Stress Gradient Hypothesis (SGH) (Bertness and Callaway 1994) based on stomach content niches and the creation deep-water refuges, and suggested more complex dynamics than predicted by the SGH may be structuring communities over longer time periods. I found that many taxa underwent shifts in diet and trophic niche in alligator-engineered habitats compared to the adjacent marsh. Additionally, the direction and magnitude of seasonal shifts in trophic metrics was typically opposite between ponds and the other two habitats (marshes and near-ponds). From wet season to dry season, stomach contents niches increased in ponds, which according to the Niche Variation Hypothesis (NVH) (van Valen 1965, Bolnick et al. 2007) means that competition decreased. This supported the SGH. However, I found the opposite trend based on stable isotope niches and this may indicate that consumptive interactions play a more important role than competitive interactions in structuring these communities, and therein trophic dynamics, over longer time scales (Werner et al. 1983, Jackson et al. 2001, Alofs and Jackson 2014). Overall, alligator-engineered ponds ameliorate environmental stress by providing the only available habitat for many aquatic taxa during the dry season facilitating increased individual fitness, which supports the SGH (Malkinson and Tielbörger 2010).

Chapter 4 documented that invasion of an omnivorous mesopredator, African Jewelfish, resulted in trophic disruption across levels of ecological organization from populations to ecosystem function while accounting for ontogeny and spatiotemporal variation. I observed post-invasion trophic dispersion and displacement in nearly half of species size classes analyzed based on stomach contents, while trophic dispersion based on stable isotopes decreased for the majority of consumers sampled. My analysis revealed that post-invasion spatiotemporal trophic dynamics occurred at a similar frequency as pre-invasion, but had increased magnitudes. Spatiotemporal variation is usually a stabilizing force (McCann et al. 2005), however the increased intensity of this variation may be destabilizing. Furthermore, I found that food-web function, and thus ecosystem function, was altered as a result of consumers relying more on autotrophic energy channels and a post-invasion rerouting of energy fluxes through the food web. These changes to ecosystem function in tandem with altered spatiotemporal trophic dynamics likely represent a new ecological regime driven by invasive species. Ultimately, we found ubiquitous support for the Trophic Disruption Hypothesis (Wainright et al. 2021) and extend this hypothesis to include not only altered diets and trophic niches, but also basal energy use and food-web energy fluxes.

Overall, we still know relatively little about the effects of spatiotemporal variation, ecosystem engineers, and invasive species on food webs and how this should influence ecosystem management. Future work on food webs should take care to incorporate spatial and temporal variation, particularly seasonality (Hampton et al. 2017, CaraDonna et al. 2017). Further research on alligator engineering in the Everglades should investigate bottom-up effects mediated by algal priming (Kuehn et al. 2014, Halvorson et al. 2019). This alligator-engineered habitat and seasonal dynamic create an opportunity for testing a variety of fundamental ecological hypotheses (e.g., biotic resistance, refuge partitioning, etc.). Meanwhile, successfully managing ecosystems plagued by invasive species may require management actions outside of designated management zones, particularly in regularly perturbed ecosystems, to mitigate the potential effects of sleeper populations (Spear et al. 2021). My findings highlight the detrimental effects of invasive species on food webs and ecosystem function and provides a framework for understanding and making hypotheses about the effects of spatiotemporal variation, ecosystem engineers, and invasive species on food webs. Future trophic ecologists should examine these drivers, if present in their ecosystem, to increase their holistic understanding of trophic dynamics and their variability.

Literature Cited

Alofs, K. M., and D. A. Jackson. 2014. Meta-analysis suggests biotic resistance in freshwater environments is driven by consumption rather than competition. Ecology 95:3259–3270.

Bertness, M. D., and R. Callaway. 1994. Positive interactions in communities. Trends in Ecology and Evolution 9:191–193.

Bolnick, D. I., R. Svanbä, S. Araú Jo §, and L. Persson. 2007. Comparative support for the niche variation hypothesis that more generalized populations also are more heterogeneous.

CaraDonna, P. J., W. K. Petry, R. M. Brennan, J. L. Cunningham, J. L. Bronstein, N. M. Waser, and N. J. Sanders. 2017, March 1. Interaction rewiring and the rapid turnover of plant–pollinator networks. Blackwell Publishing Ltd.

Catford, J. A., P. A. Vesk, D. M. Richardson, and P. Pyšek. 2012, January. Quantifying levels of biological invasion: Towards the objective classification of invaded and invasible ecosystems.

CBD. 2010. Conference of the Parties Decision X/2: Strategic plan for biodiversity 2011-2020.

Clavero, M., and E. García-Berthou. 2005. Invasive species are a leading cause of animal extinctions. Elsevier Ltd.

Flood, P. J., A. Duran, M. Barton, A. E. Mercado-Molina, and J. C. Trexler. 2020. Invasion impacts on functions and services of aquatic ecosystems. Hydrobiologia 847:1571–1586.

Halvorson, H. M., S. N. Francoeur, R. H. Findlay, and K. A. Kuehn. 2019, February 26. Algal-mediated priming effects on the ecological stoichiometry of leaf litter decomposition: A meta-analysis. Frontiers Media S.A.

Hampton, S. E., A. W. E. Galloway, S. M. Powers, T. Ozersky, K. H. Woo, R. D. Batt, S. G. Labou, C. M. O'Reilly, S. Sharma, N. R. Lottig, E. H. Stanley, R. L. North, J. D. Stockwell, R. Adrian, G. A. Weyhenmeyer, L. Arvola, H. M. Baulch, I. Bertani, L. L. Bowman, C. C. Carey, J. Catalan, W. Colom-Montero, L. M. Domine, M. Felip, I. Granados, C. Gries, H. P. Grossart, J. Haberman, M. Haldna, B. Hayden, S. N. Higgins, J. C. Jolley, K. K. Kahilainen, E. Kaup, M. J. Kehoe, S. MacIntyre, A. W. Mackay, H. L. Mariash, R. M. McKay, B. Nixdorf, P. Nõges, T. Nõges, M. Palmer, D. C. Pierson, D. M. Post, M. J. Pruett, M. Rautio, J. S. Read, S. L. Roberts, J. Rücker, S. Sadro, E. A. Silow, D. E. Smith, R. W. Sterner, G. E. A. Swann, M. A. Timofeyev, M. Toro, M. R. Twiss, R. J. Vogt, S. B. Watson, E. J. Whiteford, and M. A. Xenopoulos. 2017, January 1. Ecology under lake ice. Blackwell Publishing Ltd.

Jackson, D. A., P. R. Peres-Neto, and J. D. Olden. 2001. What controls who is where in freshwater fish communities-the roles of biotic, abiotic, and spatial factors 1. Canadian Journal of Fisheries and Aquatic Science 58:157–170.

Kline, J. L., W. F. Loftus, K. Kotun, J. C. Trexler, J. S. Rehage, J. J. Lorenz, and M. Robinson. 2013. Recent fish introductions into Everglades National Park: An unforeseen consequence of water management? Wetlands 34:S175–S187.

Kuehn, K. A., S. N. Francoeur, R. H. Findlay, and R. K. Neely. 2014. Priming in the microbial landscape: periphytic algal stimulation of litter-associated microbial decomposers. Page Ecology.

Malkinson, D., and K. Tielbörger. 2010. What does the stress-gradient hypothesis predict? Resolving the discrepancies. Oikos 119:1546–1552.

McCann, K. S., J. B. Rasmussen, and J. Umbanhowar. 2005. The dynamics of spatially coupled food webs. Ecology Letters 8:513–523.

McDonald-Madden, E., R. Sabbadin, E. T. Game, P. W. J. Baxter, I. Chadès, and H. P. Possingham. 2016. Using food-web theory to conserve ecosystems. Nature Communications 7.

McMeans, B. C., T. Kadoya, T. K. Pool, G. W. Holtgrieve, S. Lek, H. Kong, K. Winemiller, V. Elliott, N. Rooney, P. Laffaille, and K. S. Mccann. 2019. Consumer trophic positions respond variably to seasonally fluctuating environments. Ecology 100:1–10.

Mooney, H. A., and E. E. Cleland. 2001. The evolutionary impact of invasive species. Proceedings of the National Academy of Sciences 98:5446–5451.

Paudel, R., T. van Lent, G. M. Naja, Y. Khare, R. Wiederholt, and S. E. Davis III. 2020. Assessing the Hydrologic Response of Key Restoration Components to Everglades Ecosystem. Journal of Water Resources Planning and Management 146.

Pimentel, D., R. Zuniga, and D. Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273–288.

Pimm, S. L., C. N. Jenkins, R. Abell, † T M Brooks, J. L. Gittleman, L. N. Joppa, P. H. Raven, C. M. Roberts, and J. O. Sexton. 2014. The biodiversity of species and their rates of extinction, distribution, and protection. Science 344.

Post, D. M., and G. Takimoto. 2007. Proximate structural mechanisms for variation in food-chain length. Oikos 116:775–782.

Pyšek, P., P. E. Hulme, D. Simberloff, S. Bacher, T. M. Blackburn, J. T. Carlton, W. Dawson, F. Essl, L. C. Foxcroft, P. Genovesi, J. M. Jeschke, I. Kühn, A. M. Liebhold, N. E. Mandrak, L. A. Meyerson, A. Pauchard, J. Pergl, H. E. Roy, H. Seebens, M. van Kleunen, M. Vilà, M. J. Wingfield, and D. M. Richardson. 2020. Scientists' warning on invasive alien species. Biological Reviews 95:1511–1534.

Sanders, D., C. G. Jones, E. Thébault, T. J. Bouma, T. van der Heide, J. van Belzen, and S. Barot. 2014. Integrating ecosystem engineering and food webs. Oikos 123:513–524.

Schofield, P. J., and W. F. Loftus. 2015, March 1. Non-native fishes in Florida freshwaters: a literature review and synthesis. Kluwer Academic Publishers.

Simberloff, D., J. L. Martin, P. Genovesi, V. Maris, D. A. Wardle, J. Aronson, F. Courchamp, B. Galil, E. García-Berthou, M. Pascal, P. Pyšek, R. Sousa, E. Tabacchi, and M. Vilà. 2013, January. Impacts of biological invasions: What's what and the way forward.

Spear, M. J., J. R. Walsh, A. Ricciardi, and M. J. vander Zanden. 2021. The Invasion Ecology of Sleeper Populations: Prevalence, Persistence, and Abrupt Shifts. BioScience 71:357–369.

Strassburg, B. B. N., A. Iribarrem, H. L. Beyer, C. L. Cordeiro, R. Crouzeilles, C. C. Jakovac, A. Braga Junqueira, E. Lacerda, A. E. Latawiec, A. Balmford, T. M. Brooks, S. H. M. Butchart, R. L. Chazdon, K. H. Erb, P. Brancalion, G. Buchanan, D. Cooper, S. Díaz, P. F. Donald, V. Kapos, D. Leclère, L. Miles, M. Obersteiner, C. Plutzar, C. A. Carlos, F. R. Scarano, and P. Visconti. 2020. Global priority areas for ecosystem restoration. Nature 586:724–729.

Trexler, J. C., and C. W. Goss. 2009. Aquatic fauna as indicators for Everglades restoration: Applying dynamic targets in assessments. Ecological Indicators 9.

UN. 2015. Transforming our world: the 2030 agenda for sustainable development.

van Valen, L. 1965. Morphological Variation and Width of Ecological Niche. Source: The American Naturalist 99:377–390.

Vitousek, P. M., C. M. D'Antonio, L. L. Loope, and R. Westbrooks. 1996. Biological invasions as global environmental change. American Scientist 84.

Wainright, C. A., C. C. Muhlfeld, J. J. Elser, S. L. Bourret, and S. P. Devlin. 2021. Species invasion progressively disrupts the trophic structure of native food webs. Proceedings of the National Academy of Sciences of the United States of America 118.

Werner, E. E., J. F. Gilliam, D. J. Hall, and G. G. Mittelbach. 1983. An experimental test of the effects of predation risk on habitat use in fish. Ecology 64:1540–1548.

Wootton, K. L. 2017. Omnivory and stability in freshwater habitats: Does theory match reality? Freshwater Biology 62:821–832.

VITA

PETER JOSHUA FLOOD

2011-2015	B.S. Marine Sciences – Biology University of Tampa Tampa, Florida
2016	M.S. Biology – No Degree Earned University of Southern Mississippi Hattiesburg, Mississippi
2017-2022	M.S. Biology Florida International University Miami, Florida

PUBLICATIONS

Flood, P. J., Loftus, W. L., & Trexler, J. C. (2022) Fishes in a seasonally pulsed wetland show spatiotemporal shifts in diet and trophic niche not associated with shifts in trophic position. Food Webs. *Revision in Review*.

Strickland, B.A., Flood, P. J., Kline, J. L., Mazzotti, F., Heithaus, M. R., & Trexler, J. C. (2022) An apex predator engineers wetland food-web heterogeneity through nutrient enrichment and habitat. Journal of Animal Ecology. *In Review.*

Flood, P. J., Duran, A., Barton, M., Mercado-Molina, A., & Trexler, J. C. (2020) Invasion Impacts on Functions and Services of Aquatic Ecosystems. Hydrobiologia 847: 1571-1586

Hubbell, J., Schaefer, J. F., Flood, P. J., Warren, M. L., & Sterling, K. (2020). Fragmentation Alters Ecological Gradients and Headwater Fish Assemblage Composition Relative to Land Use in a Dendritic River System. Canadian Journal of Fisheries and Aquatic Sciences 77(8) doi.org/10.1139/cjfas-2019-0080.