DAMS, DAM REMOVALS, AND FRESHWATER MUSSEL CONSERVATION

by

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DEDICATION

To my father, without whom none of this would be possible or nearly as difficult.

May he rest in peace.

Richard G. Dascher
(1955 - 2014)

He was never there.
I never looked for him.
He was never far.

ACKNOWLEDGEMENTS

Thanks all.

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ABSTRACT

This study uses variety of analytical and geospatial techniques to analyze the connections between fragmentation and freshwater mussel distribution and community composition in Texas and the Guadalupe San Antonio River System (GSARS). Additionally, dam removal is assessed and promoted as a strategy for freshwater mussel conservation. The distribution of dams is related to Texas' climate gradient and the location of population centers. Models of connectivity reveal the increasing amount of fragmentation dams have created in the GSARS through time and the substantial number of undocumented sources of fragmentation in this river system. Patterns of freshwater mussel distribution and community composition are related to the distribution of host fish, climate gradients, hydrologic regimes, and land use. Two dam removal prioritization models are created for the GSARS that incorporate metrics associated with freshwater mussel conservation and individual dam attributes. These models act as broad scale decision support tools for freshwater mussel conservation that can be built upon and further refined with additional data sources.

1. INTRODUCTION

Dam removal restores many processes to a river, including flow, sediment, and thermal regimes (Bednarek 2001), yet, removing a dam can act as disturbance to already altered systems (Tullos et al. 2014; Riggsbee et al. 2007; Stanley & Doyle 2003). Dam removals involve a tradeoff between environmental costs and benefits that result in a complex array of integrated biotic and abiotic responses (Tullos et al. 2014; Riggsbee et al. 2007; Stanley & Doyle 2003; Bednarek 2001). Dam removals are increasingly utilized to restore migratory fish populations and habitats, but the value of dam removal for the restoration and conservation of freshwater mussel species requires more attention.

The positive response of fish to dam removal and the importance of host fish to mussel distribution suggests that dam removal may result in an increase of native mussel species (Gottgens 2009). However, dam removal may lead to increased mussel mortality as opposed to proliferation. During rapid dam removal events, sediments flushed from the reservoir may bury mussels, impeding their lateral migration leading to their mortality (Cooper 2011; Sethi et al. 2004). Mussels previously located in the impoundment of the dam may also experience high rates of mortality due to stranding, desiccation, and predation due to rapid dewatering (Sethi et al. 2004). High flows or floods post-dam removal may have a positive effect, creating suitable habitat for host fish spawning, or a negative effect, washing out mussels and/or burying them in sediment (Hauer 2015).

Texas has over 7,000 dams (USACE 2016), resulting in the second largest surface area of lakes and volume of inland water in the nation (Sansom 2008). Texas has more dams than any other state in the nation (USACE 2016), yet this number does not account for the large number of small and medium sized dams (Chin et al. 2008). In addition to the large number of impoundments in Texas, there are also approximately 50 freshwater mussel species (Appendix 1.1), 15 of which are currently listed as threatened at the state level (TPWD 2016). Six of these 15 state threatened mussel species are also candidates to be listed as federally endangered (TPWD, 2016). The majority of these threatened freshwater mussel species are considered either regionally endemic or endemic to the state of Texas (Burlakova et al., 2011), and these endemic species are a critical component of the diversity and uniqueness of unionid communities in the state (Burlakova et al., 2011a).

1.1. RESEARCH OBJECTIVES AND CHAPTER ORGANIZATION

The overarching question of this research is: How are the spatial and temporal patterns of dams and fragmentation related to the environmental and anthropogenic controls on the distribution of freshwater mussels, and what are the implications of these for dam removal science? To answer this question three main objectives were achieved. First, I analyzed the spatial and temporal trends of dams in Texas, and the patterns of river fragmentation created by dams in the Guadalupe – San Antonio River System (GSARS). Second, I examined the distribution of freshwater mussels and their host fish at the state and basin scale, and investigated the environmental and anthropogenic controls of their community compositions in the GSARS. The third objective involved two main parts: I assessed the spatiotemporal patterns of dam removal in Texas, and created a dam removal prioritization model for the GSARS based on the findings from objectives 1 and 2, existing literature, and expert opinion. And, finally, I provided a set of recommendations for freshwater mussel conservation and dam removal prioritization in Texas. The specific questions for objectives 1-3 are listed below:

1. Objective 1: What are the spatial patterns of dam occurrence in Texas?
    a. How do these patterns change over time?
    b. What spatial and temporal patterns of fragmentation do dams create in the Guadalupe – San Antonio River System?
    c. How many functional river networks (FRNs) exist in the Guadalupe – San Antonio River System, and how do these FRNs relate to dam size and age?
2. Objective 2: What are the current spatial patterns of freshwater mussels in Texas?
    a. What are the broad scale spatial patterns of freshwater mussels and their host fish distribution in Texas?
    b. What are the broad scale spatial patterns of freshwater mussel distribution and community composition in the Guadalupe – San Antonio River System?

c. What are the environmental and anthropogenic controls on freshwater mussel community composition in the Guadalupe – San Antonio River System?

3. Objective 3: What are the spatial and temporal patterns of dam removal in Texas?
    a. How can these patterns inform mussel conservation?
    b. What are the criteria for prioritizing dam removals in the Guadalupe – San Antonio River System, and how should they be weighted?
    c. What dams have the highest and lowest priority for removal in the Guadalupe – San Antonio River System based on a dam removal prioritization model that prioritizes freshwater mussel conservation?

This dissertation is organized in four chapters. Chapter 1 is a brief introduction to the research area, specific objectives of the research, and the study area. Chapters 2, 3, and 4 individually address the research objectives 1,2 and 3 respectively. Chapter 4 concludes by incorporating the major research findings of this dissertation and providing management recommendations for future freshwater mussel conservation efforts. The tables and figures for each chapter follow the body of the text for each chapter.

1.2. STUDY AREA

Texas is the second largest state in the Unites States, comprising an area of 695,619 square kilometers. Texas has an east to west climate gradient, with portions of East Texas receiving approximately 1525 mm of precipitation a year, while parts of West Texas experience less than 205 mm of annual average rainfall (PRISM 2013). Potential evapotranspiration (PET) can be as low as 1124 mm per year on average in northeast portions of the state, and increase to as much as 1879 mm per year in areas of western and southwestern Texas (CGIAR-CSI 2008). There are 26 major Texas river basins (Figure 1.1) that belong to two major drainages, the Mississippi River and the Gulf of Mexico Drainage Basin, and all rivers flowing through Texas eventually drain into the Gulf Coast. As water availability decreases from east to west in Texas, many central and western river basins are prone to periods of no-flow. The National Weather Service identifies Central Texas as the most flash flood prone area in the United States, with all basins subject to intense flooding. The large number of dams in Texas results from the state's size and the need for flood control and water supply to protect and provide for the state's population.

The Guadalupe – San Antonio River System (GSARS) (Figure 1.1, 1.2), located in Central Texas, includes the Guadalupe and San Antonio river basins. The Guadalupe (15,480.1 km^2) and San Antonio river basins (10.866.4 km^2) collectively have a total area of 26,346.5 km^2. Both basins drain south-east from the Texas Hill Country, across the Balcones Escarpment, and eventually meet the Gulf of Mexico. The area has a mean precipitation range of 27.9 to 40.4 inches annually (709 – 1026 mm/yr) (PRISM 2013) and a mean PET range of 54.4 to 62.6 inches annually (1382 – 1590 mm/yr) (CGIAR-CSI 2008). There are nineteen major population centers within the GSRS, including the entirety of San Antonio, TX with a population of 1,327,407 and the Southeast portion of Austin, TX that has a total population of 790,390 (U.S. Census 2010) (Figure 1.2).

There are 375 dams total in the GSRS, 216 dams located in the Guadalupe Basin, and 160 in the San Antonio Basin creating $1.94^\wedge10^9$ and $0.87^\wedge10^9$ m^3 of water storage, respectively (TCEQ 2014). There are at least 13 different species of freshwater mussels that inhabit the GSARS, including two endemic species *Quadrula aurea* found only in the Guadalupe and San Antonio river basins and *Quadrula petrina* found only in the Colorado and Guadalupe river basins.

The GSARS was chosen to examine patterns of fragmentation and mussel community composition based on multiple factors: 1) the availability of data for a large number of impoundments; 2) the availability of mussel and fish data; and 3) practicality of using the Barrier Assessment Tool (BAT) on account of the required data preparation. The BAT was originally developed as an ArcGIS plugin for the Northeast Aquatic Connectivity (NAC) project to prioritize dam removal (Martin & Apse 2011). The BAT calculates several river network measurements and is freely available by contacting the authors of the NAC report (Martin & Apse 2011).

1.3. TABLES AND FIGURES

Figure 1.1. The 26 major river basins in Texas.

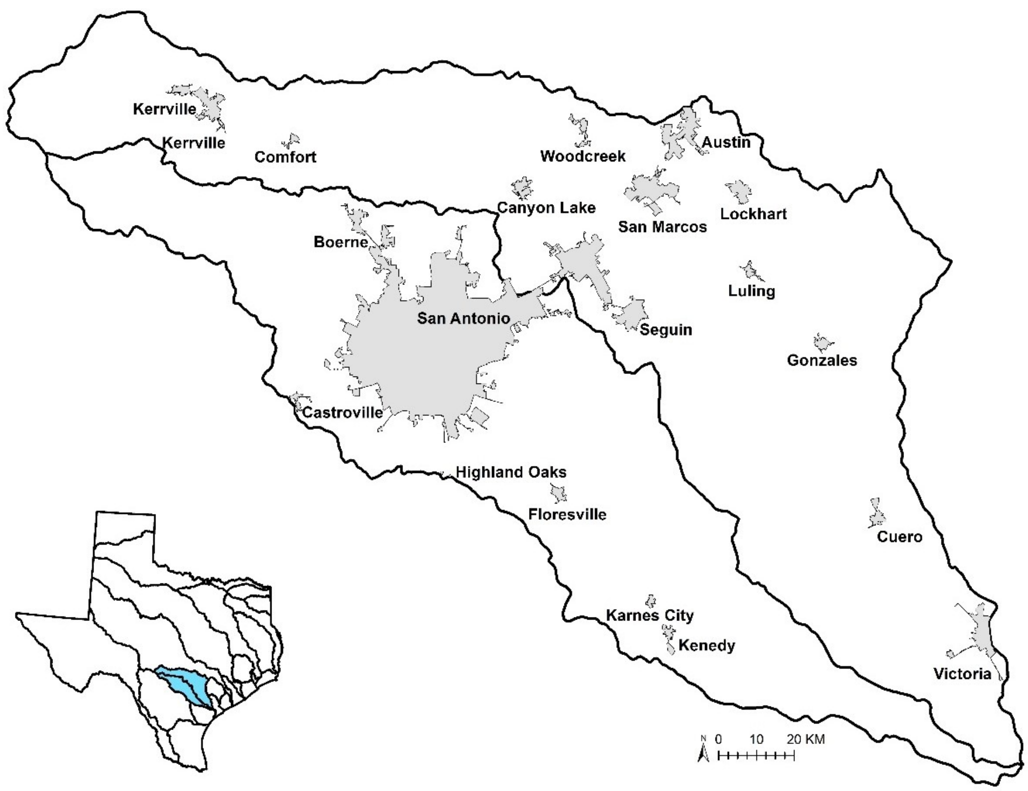

Figure 1.2. The Guadalupe – San Antonio River System with major population centers labeled.

2. SPATIAL AND TEMPORAL TRENDS IN TEXAS DAMS

2.1. INTRODUCTION

The rate of dam building has slowed in the United States over the last few decades, but in Texas the creation of reservoirs is cited as a necessary part of the state's future water supply to help mitigate impacts of future droughts (TWDB 2017). In 2011, Texas experienced the worst single-year drought in recorded history (Folger et al. 2012). During October of 2011, 88% of the state experienced exceptional drought, and much of the state continued to experience extreme to exceptional drought conditions through January 2012 (Folger et al. 2012). The winter of 2012 brought relief through increased precipitation to the eastern portion of Texas, but much of the state remained in drought conditions ranging from moderate to exceptional (Folger et al. 2012; U.S. Drought Monitor 2012).

Partially in response to the 2011 drought, Proposition 6 passed in 2013 to fund water projects outlined in the State Water Plan (Henry 2012). Proposition 6 was a constitutional amendment that transferred two billion dollars from Texas's "Rainy Day Fund" to create the State Water Implementation Fund of Texas (SWIFT) (Henry 2012). As part of numerous water projects outlined, the 2012 Texas State Water Plan recommended 26 new major reservoir sites in parts of North, East, and Central Texas to be built by 2060 (TWDB 2012). As of 2017, three of these reservoirs received the necessary permits and funding to begin construction (TWDB 2017). These three new major reservoirs are the Lake Ralph Hall Reservoir planned for the Upper Trinity River Basin, the Turkey Peak

Reservoir in the Upper Brazos River Basin, and the Lower Basin Reservoir in the lower Colorado River Basin (TWDB 2017).

These reservoirs would be in addition to the 7,395 dams already registered in Texas, according to the National Inventory of Dams (NID) dataset (USACE 2016), 188 of which are major reservoirs (5,000 af of more storage capacity) (TWDB 2012). There are also a large number of unregistered small and medium sized dams not accounted for in the NID dataset (Chin et al. 2008). Collectively, these existing and proposed dams fragment Texas rivers. A review of the available data on dams and the fragmentation they create is important to understanding how they impact river systems.

The Barrier Assessment Tool (BAT), developed by the Nature Conservancy (TNC) (Martian and Apse 2011), provides a way to model longitudinal connectivity in river systems, and allows for broad scale analysis of dam-related fragmentation. In previous applications, including the Northeast Aquatic Connectivity project (Martin and Apse 2011) and North Carolina's Freshwater Resilience project (Benner et al. 2014), BAT analyses have informed dam removal prioritization. In this analysis, I used the BAT to model temporal changes in longitudinal connectivity, to reveal the extent and patterns of fragmentation in the Guadalupe - San Antonio River System (GSARS) from registered dams.

The purpose of this study is to explore the spatial and temporal trends in the available data on Texas dams and to assess the patterns of fragmentation they create in the GSARS. To an extent, this study builds on previous research containing data on Texas dams presented by Chin et al. (2008), by incorporating new data and an additional scale of analysis. First, I address the following specific research questions: (1) What are the spatial patterns of dam occurrence in Texas? and (2) How do these patterns change over time? Next, focusing on longitudinal fragmentation, this study uses the BAT to model the fragmentation created by registered dams in GSARS for multiple time periods. The questions answered by this secondary analysis are: 3) What is the length and number of functionally connected river networks in the GSARS? 4) How many functional river networks (FRNs) exist in the Guadalupe – San Antonio River System, and how do these FRNs relate to dam size and age? By using the BAT in a GIS environment, temporal and spatial patterns of river connectivity and fragmentation can be modeled and analyzed at the basin-scale.

2.1.1. A Brief History of Dams

The earliest dams were constructed 5000 years ago (Petts & Gurnell 2005). They were small impoundments, likely built as earthen structures to store water for use during drier periods (ICOLD 2007). As civilizations grew, dam use began to diversify to include water supply, irrigation, flood control, navigation, water quality purposes, sediment control, energy generation, and recreation; and today most dams are multipurpose (ICOLD 2007). The Romans built a large and complex system of dams for water supply, many of which are still in use today (ICOLD 2007). During the 16th century, Spain began to build large dams for irrigation, and in the 1800s dams began to be built for navigation and hydropower (ICOLD 2007). The construction of mega dams was begun by European engineers in the 19th century (ICOLD 2007), but by the 20th century the United States led the world in dam construction (Clark 2009).

Large dams became symbols of technological and social advancement (Duchiem 2009; Petts & Grunell 2005). This was especially true of hydropower projects that were viewed as important for both the prosperity of the nation and national defense (Reinhardt 2011). While the Hoover Dam ushered in the modern era of dam building in the United States (Solomon 2010; Petts & Grunell 2005; Reisner 1986), the number of large dams commissioned did not drastically increase until after WWII (Petts & Grunell 2005).

In the US, the Bureau of Reclamation alone constructed forty hydropower dams between 1945 and 1955 (Reinhardt 2011), and during the 1960s the number of dams continued to increase at a rate of nearly two dams a day worldwide (Petts & Grunell 2005). A total of 35,236 dams were completed in the United States between 1940 and 1970, and during this time the 1960s were recognized as the "dam building" decade (USACE 2016; Graf 2005). Once started this rapid pace of dam construction would not slow until the 1980s (Solomon 2010; Petts & Gurnell 2005; WCD 2000).

As dams increased in number and size across the landscape, so did the understanding of their impacts on river systems. Studies on downstream effects of dams began in the 1920s, yet as early as 1784 efforts were made to prevent dam construction, due to the already apparent impact on migratory fishes along East Coast Rivers in the United States (Graf 2005). Despite the growing scientific understanding of environmental impacts created by dams, the dam building era would not slow until the 1970s, when American attitudes toward the environment shifted (Solomon 2010). By this time, ideal sites to build new large dams had already become scarce (Reisner 1986), and today every major river in the U.S. is, in part, controlled and impacted by dams and their reservoirs (Graf 2006).

2.1.2. Dam Impacts

Riverine systems are connected across four dimensions of connectivity: longitudinal, lateral, vertical, and temporal (Ward 1989). Individual dams disrupt all four dimensions of lotic ecosystems, and multiple dams within a river network segment once connected systems into separate fragments. The sheer number of dams, large and small, have fragmented entire watersheds (Chin et al. 2008; Graf 2006), and this fragmentation modifies the river system's form, function, and ecology.

The Serial Discontinuity Concept (SDC), devised by Ward & Stanford (1983) stipulates that most riparian ecosystems are no longer free-flowing but rather a series of lentic and lotic ecosystems. The SDC was a response to the River Continuum Concept (RCC) that highlights the importance of a river's longitudinal connectivity asserting that a river is a connected system from its headwaters to its mouth (Vannote et al. 1980). The physical parameters of a free-flowing river operate along a continuous gradient (Vannote et al. 1980), but dams act as reset points disrupting the natural longitudinal connectivity of a river system (Ward & Stanford 1983, Stanford & Ward 2001).

Dams significantly alter the downstream magnitude, timing, duration, and frequency of stream flows (Graf 2006; Magilligan & Nislow 2001, 2005; Hirsch et al. 1990; Williams and Wolman 1984). These hydrologic alterations can impact the geomorphology (Curtis et al. 2010; Graf 2006; Grant et al. 2003; Galay 1983) and ecology of the river system (Bunn and Arthington 2002; Graf 2006; Katz et al. 2005; Merrit & Wohl 2006; Poff et al. 1997). A dam's reservoir turns once lotic ecosystems into lentic systems, and can trap large volumes of sediment. Stream channels downstream of dams can erode as sediment starved flow released from the structure scours the channel bed and banks (Graf 2006; Grant et al. 2003; Petts & Gurnell 2005). This process can simplify channel and floodplain geomorphology downstream of dams resulting in homogeneous ecosystems (Dynesius and Nilsson 1994; Graf 2006; Poff et al. 2007).

Dams also disrupt lateral connectivity along a river, separating the main river channel from its adjoining floodplains (Junek et al. 1989). Dams are often built as flood control devices eliminating floods from the flow regime of a river (Graf 2006; Junk et al. 1989). This disconnection results in less active floodplain surfaces, and confined lateral migration (Graf 2006). The floodplains upstream of dams lose their function and become incorporated with the inundated reservoir (Junk et al. 1989). This change, in combination with the loss of connectivity, alters the cycling of floodplain nutrients upstream and downstream of dams (Junk et al. 1989).

Dams similarly alter the vertical connectivity of rivers (i.e. the interaction of ground and surface waters) (Sawyer at el. 2009; Lautz et al. 2006). Hyporheic exchanges my increase due to slower water velocities and increased flow complexity (Lautz et al. 2006); and where groundwater would normally flow towards a river this relationship may be reversed, with surface water flowing towards the underground aquifer (Sawyer et al. 2009).

Most studies on the impacts of dams involve large, singular structures, and those that look at small dams report correspondingly smaller impacts on river hydrology, geomorphology, and ecology (Csiki & Rhoads 2013, Wyrick et al. 2009, Ambers 2007). However, when small dams are studied as a network, opposed to singularly, the high density of dams can have a significant impact on river systems (de Araújo & Medeiros 2013, Nathan & Lowe 2012, Mantel Hughes, & Muller 2010, Mantel, Muller, & Hughes 2010a). The cumulative effects of smaller dams include changes to sediment regimes (Berg et al. 2015; de Araújo & Medeiros 2013; Smith et al. 2002), water quality deterioration (de Araújo & Medeiros 2013; Mantel et al. 2010, 2010a), and changes in flow regimes and water availability (Mantel et al. 2010, 2010a). Chin et al. (2008) emphasizes the prevalence of small and medium sized dams across Texas and their fragmentation of river systems.

Rivers are not isolated segments, but are complex systems composed of hierarchical branching networks (Benda et al. 2004). In these complex and systems, longitudinal connectivity includes connection between upper and lower reaches, and the connection of tributaries and their confluences which are important sources of habitat and biodiversity (Benda et al. 2004). As hierarchical networks, rivers are strongly influenced by longitudinal connectivity (McCluney et al. 2014), and the spatial arrangement, along with the type, extent, and intensity, of anthropogenic impacts on these networks, influence their ecology (McCluney et al. 2014; Benda et al. 2004).

2.2 MATERIALS AND METHODS

2.2.1. Data

The analyses in this study used a variety of data sources. The first analysis included available data for registered dams and subsets of national precipitation and potential evaporation (PET) datasets to analyze the temporal and spatial patterns of dams in Texas. The secondary analysis used a subset of the registered dam data in conjunction with the National Hydrography Dataset High Resolution (NHD-HR) to model connectivity in the GSARS.

Two dam datasets exist for the state of Texas, one is managed by the National Inventory of Dams (NID) and includes 7,395 registered dams (USACE 2016), and the other is managed by the Texas Commission on Environmental Quality (TCEQ) and includes 7,280 dams (TCEQ, 2014). On account of federal governmental limitations on data use, I used the state-level TCEQ dataset for this research. Through a memorandum of user agreement, TCEQ provided a geodatabase that included information on the location and attributes of dams in Texas. Dam attributes used for these analyses included: year complete, purpose, and maximum storage capacity.

Of the registered dams, 7,161 included a year of completion (98.4%) and 6,567 had at least one purpose identified (90.3%). Multiple purposes were listed for dams in order of importance reflecting current use. All dams had a maximum storage value (defined as the total storage space in a reservoir below the maximum attainable water surface elevation, including any surcharge storage), however 37 (0.005%) contained 0 af indicating a lack of data.

A table of ownership information, including the organization type of the owner, was provided as a separate file. There were over 10,000 entries in the ownership table, the result of multiple owners for individual dams. Entries in the ownership file that matched a corresponding ID in the dam shapefile were joined to the attribute table of the dam shapefile. Dams that did not have an owner listed or that did not have a matching owner ID were less than 0.02% (N = 101).

Precipitation data was obtained from PRISM, a national 4 km resolution raster file of the 30-year annual average (1981 - 2010) precipitation produced by the PRISM Climate Group of Oregon State University (PRISM 2013). Potential evapotranspiration (PET) data was acquired from the CGIAR-CSI Global Aridity and Global-PET Database (CGIAR-CSI). The Global-PET Database is a global 1 km resolution raster of the fifty-year annual average (1950 – 2000) (Zomer et al. 2007, Zomer et al. 2008). The PRISM and PET datasets are freely available online, and were used to account for climatic trends across Texas. The United States Census Bureau (2013) provided shapefiles of Texas, and TWDB (n.d.) provided shapefiles of the 23 major Texas river basins.

The NHD-HR is a digital database of line features (1:24,000 or better scale) that is part of *The National Map* maintained by the United States Geological Society (USGS) (2017). The NHD-HR represents the nation's drainage networks and related features, and is the most current and detailed hydrography dataset for the United States (USGS 2017). The data is maintained and updated through stewardship partnerships with states and other collaborative entities (USGS 2017). Flowlines representing irrigation and/or drainage canals were removed from the NHD-HR dataset before use in this study.

2.2.2. Analyses of Temporal and Spatial Patterns in Texas Dams

I used ArcGIS 10.2.1 (ESRI 2014) to organize and analyze the available data on registered dams and climate in Texas. Twelve of the 7,280 dams in the TCEQ geodatabase had inaccurate or problematic latitude and longitude coordinates. Of these 12, six were relocated and six were deleted as their true coordinates could not be determined. This resulted in a final dataset of 7,274 dams. I subdivide this state-wide dataset by river basin generating twenty-three sub-datasets, for a total of 24 dam datasets. Analysis of the Global-PET and national PRISM datasets determined the average, minimum, and maximum precipitation and PET values for each river basin using spatial analysis. Basin areas were also calculated in ArcGIS.

Next, I used IBM SPSS Statistics 22 (2014) to analyze the dam and climate data. I calculated the total reservoir storage and percentage of total storage for each of the 23 sub datasets, at the state-wide scale, and for ten major river basins: The Trinity, Brazos, Colorado, Red, and Nueces, Sabine, Rio Grande, Neches, Guadalupe, and San Antonio river basins (Figure 2.1). These ten basins represented nearly 90% of all the dams, 85% of the storage, and over 80% of the surface area in Texas. I created the variables of size, time period (of completion), primary purpose, and ownership from the dam attributes of maximum storage, year complete, purpose, and organization type (of owner) respectively.

There were 13 separate organization types in the TCEQ database for ownership. I aggregated these 13 organizations into six ownership classifications: federal, state, and local governments, private entities, other, and not listed. I transformed the attribute of dam purpose into primary purpose, and only included the first purpose for dams with multiple purposes.

There were multiple size classifications available for dams; the TCEQ used a size classification based on dam height and reservoir storage. For the purposes of this analysis the size, I used the size classification developed in 2002 by the Heinz Center, and later modified by Graf (2005) (Table 2.1). I then sorted dams by size, and used descriptive statistics to analyze the variables of time period, primary purpose, and ownership for each size class.

I completed a similar analysis for time periods. I sorted dams by five time periods, and used descriptive statistics to analyze each time period by size, purpose, and ownership. I classified time periods using existing literature and logical breaks within the data. I grouped dams completed between 1800 and 1899 together, as there were many dams completed in 1800 and very few between 1800 and 1899. This instigated suspicion of the legitimacy of these completion dates. It seemed probable that many older dams, known to have been built sometime in the 1800s, had 1800 as year of completion, when the exact year of completion was actually unknown.

I grouped dams completed between 1900 and 1939 together, as this represents an early age in dam building before World War II. A WWII/post-WWII period between 1940 and 1959 designated the time when dam building began to progressively develop. The heyday of dam building was between 1960 and 1980. Thus, I grouped dams completed between 1960 to 1979, and 1980 to 2014 as the last two time periods classes.

2.2.3. Modeling Connectivity

I used the Barrier Assessment Tool (BAT) to create several models of connectivity in the GSARS. The BAT modeled connectivity by measuring the length of connected river segments between barriers and created an output of these connected segments in a river system. These functional river networks (FRNs), represented discreet river segments of available longitudinal movement. Each dam was associated with an upstream and downstream FRN that represented the connected stream length between that dam and either another dam or the river mouth.

For modelling connectivity using the BAT, the NHD-HR provided a data set of single-flowline dendritic networks representing the hydrology of the river system. The BAT program identified any bifurcations during preparation of the stream network. I manually edited bifurcations by moving one of the nodes of the two downstream river segments upstream of the other.

I used a subset of TCEQ shapefile of registered dams to represent hard barriers in the GSARS. For the BAT to accept the dataset of hard barriers, each dam I snapped to the river network, and to ensure dams were snapped to the right location, I manually snapped a large portion of them. I removed dams listed were not located on the NHD-HR stream network in the GSARS from the dataset for this portion of the analysis. This resulted in 359 dams in the GSARS snapped to the river network, 202 in the Guadalupe River Basin and 157 in the San Antonio River Basin.

Of these 202 dams, two in the Guadalupe River Basin did not have a date of completion, and were only used in the fifth and sixth iterations of the model.

Dams removed from the dataset were mainly associated with drainage canals for housing projects and a small number of dams that presumably had an incorrect latitude and longitude associated that could not be successfully relocated. Once I successfully snapped all dams in a watershed to the river network, I used the BAT to create a barrier output table and exported the functional networks.

To analyze the temporal aspects of fragmentation in the GSARS, I complete the BAT analysis for five consecutive time periods, defined above, based on dam completion using a simplified river network. I edited the simplified river network so that only flowlines that eventually connected to the main stem of a river remained, and I deleted all other segments.

The initial analysis utilized dams completed from 1800-1899, and each successive iteration utilized dams from the previous period of completion in addition to dams from the next period. The fifth iteration incorporated all listed dams with a known date of completion in the GSARS (1800 – 2014). For each time period analysis, I manually manipulated the output of the BAT so that the connectivity model reflected only FRNs created by registered dams. FRNs unassociated with registered dams were potentially the result of sharp changes in the angle of flowlines, gaps created by the removal of flowlines representing irrigation and/or drainage canals, and/or other irregularities.

A vast number of small and medium sized dams are not reported in state and federal datasets (Chin et al. 2008). Thus, the fragmentation of rivers by reported or registered dams was a conservative estimate of the total amount of river fragmentation in the GSARS. To partially address this issue, I created a sixth iteration of the model that incorporated dams from all periods of completion (1800 – 2014) and the unedited stream network. The unedited stream network that included all flow lines regardless of whether they eventually connected to the main stem or not. Additionally, in the sixth iteration, I did not force FRNs to be associated with registered dams. The sixth iteration thus included FRNs caused by both registered dams and other sources of fragmentation inherently present in the NHD-HR dataset as flowline irregularities. The sixth iteration of the model provided a contrast between fragmentation created by registered dams alone and a more complete representation of fragmentation in the GSARS.

For each iteration of the model, the BAT produced two attribute tables and two complementary shapefiles which I joined to produce a point shapefile of snapped dams and a vector shapefile of the FRNs for the GSARS. The specific attributes of the sum length of the FRNs and the upstream length of the FRN from each dam were extracted for each iteration of the model for further analysis. I conducted this analysis using ArcGIS 10.3.1 (ESRI 2015).

I grouped the FRNs for each iteration of the model into five size classifications based on length: < 1 km, 1 - 10 km, 10-100 km, 100 – 1000 km, and > 1000 km. I then calculated the percentage of FRNs by size for each iteration. I calculated the average upstream FRN length for four dam sizes using the size classification modified by Graf (2005) from the Heinz Center (2002) (Table 2.1). For the sixth iteration of the model, I calculated the percent of FRNs caused by dams by dividing the number of dams by the total number of FRNs. I depicted the relationship between dam size, age, and FRN, using scatter plots.

2.3 RESULTS

2.3.1. Climatic & Geographic Trends

PET generally increased from east to west, with the highest PET values located in parts of the southwest (Table 2.2). Inversely, precipitation declined from east to west, with an average yearly precipitation range of 205.7 to 1562.1 mm (Table 2.2). There was a 1082 mm range of average yearly precipitation variables by basin, the Rio Grande 388.62 mm being the driest and the Neches-Trinity 1460.7 mm basin being the wettest (Table 2.2).

In general, river basins receiving less than 762 mm of average annual rainfall had larger percentages of dams and storage, with the exception of the Trinity River Basin. The Trinity River Basin contained the largest percentage of dams and storage, but had an average of 1051.6 mm (41.4 in) annual rainfall (Figure 2.2 and Table 2.2). Larger river basins contained a larger proportion of dams; again with exception to the Trinity River Basin and, in this case also the Rio Grande River Basin (Table 2.2). The Trinity contained nearly a fourth of all dams, but had only the fifth largest area (46,587 km^2, 6.7%), while the Rio Grande had the largest surface area of any basin (128,437 km^2, 18.5%) and less than five percent (N = 326) of the total number of dams (Table 2.2). Coastal basins, being among the smallest major river basins, contained less than one percent of the total number of dams in Texas (Table 2.2).

2.3.2. Further Analysis of Ten Selected Major River Basins

The ten major river basins chosen for further analysis are the ten largest river basins in Texas, excluding the San Fernando Creek Basin, a coastal basin with only 101 (1.4%) dams. The Trinity (N = 1787, 24.6%), Brazos (N = 1392, 19.1%), Colorado (N = 775, 10.7%), Red (N = 619, 8.5%), and Nueces (N = 456, 6.3%) river basins, combined contained 78.2% of dams (Table 2.2). The Rio Grande River Basins had the largest surface area in Texas, and 4.5% of dams (N=326). The Guadalupe (N = 215, 3%), and San Antonio (N = 160, 2.2%), Sabine (N = 335, 4.6%), and Neches (N = 308, 4.2%) river basins constituted another 14% of the total number of dams.

2.3.2. Dam Size

Medium-sized dams accounted for the largest occurrence. (N = 5588, 77.2%). Similarly, for each of the ten major river basins medium dams comprised more than 70% of the total : Trinity (N = 1426, 79.9%), Brazos (N = 1074, 77.5%), Colorado (N = 599, 77.7%), Red (N = 449, 72.6), Nueces (N = 365, 81.7%), Sabine (N = 365, 73.3%), Rio Grande (N = 266, 81.1%), Neches (N = 229, 74.5%), Guadalupe (N = 152, 71.3%), and San Antonio (N = 125, 78.6%). Small dams comprised the second largest proportion (N = 1415, 19.6%), and represented 18% to 23% of dams in each river basin. Large and very large dams represented the smallest portion of dams, and accounted for only 3.5% and 0.9%, respectively. Appendix 2.1. contains tables of theses descriptive statistics for all the basins.

While the amount of large and very large dams was low compared to medium and small dams, they accounted for nearly 95% of the total reservoir storage in Texas, and over 90% of the reservoir storage in each river basin (Appendix 2.2). Very large dams alone accounted for nearly fifty percent or more of the storage in each basin, and nearly 70 percent of the storage in Texas (Appendix 2.2).

2.3.3. Time Periods and Reservoir Storage

A general trend existed relative to the number of dams completed in Texas by each time period; dam construction increased during the first four time periods, and then declined in the 1980 - 2014 time period (Figure 2.3). In the 1800s, the most commonly built dams were small and medium sized with similar for both; only three large and no very large dams were built during this same time (Appendix 2.2). The majority of dams built for all other time periods were medium sized dams (Appendix 2.2). From 1900 to 1939 there was an increased number of large and very large dams commissioned. This trend continued from 1940 and 1979 during which the largest number of dams were built, including most of the large and very large dams. Specifically, 1940 – 1959 experienced the construction of 59 large and nine very large dams. An additional 72 large dams and 12 very large dams were constructed from 1960 to 1979 (Appendix 2.2).

Most of the reservoir storage capacity by volume was created between 1940 and 1979, with the largest percentage created between 1960 – 1979, and this same pattern applied to the Trinity, Brazos, Sabine, Rio Grande, Neches, and Guadalupe basins (Figure 2.4). However, in the Colorado and Red river basins the majority of reservoir storage was created between 1939 and 1940 (Figure 2.4).

The Nueces Basin gained over 60% of its reservoir storage during the 1980s, and for San Antonio, nearly half of the reservoir storage was built in the early 1900s (Figure 2.4).

The Trinity Basin experience the construction of one very large and ten large dams from 1940 to 1959, and between 1960 and 1979 an additional two very large and eight large dams were completed. The Brazos Basin gained 11 large dams and 3 very large dams from 1940 to 1959; and from 1960 to 1979, an additional 19 large and 2 very large dams were commissioned in this basin.

In the Sabine Basin, two of the three very large dams were built between the years of 1960 to 1979. These two dams were the Iron Bridge Dam, built in 1960 with a maximum storage of over a million af, and the Toledo Bend Dam built in 1967 with a maximum storage of over five million af. Together these two dams constituted over two thirds of the total reservoir storage in the basin (8,776,518 af).

The Rio Grande Basin had a total reservoir storage of 10,858,655 af, and only two very large dams, the International Falcon Lake Dam with a maximum storage of over four million af was built in 1954, and the International Amistad Dam with a maximum storage of over five and a half million af was built in 1969. The only two very large dams in the Neches basin were both commissioned during the 1960-1979 time period, and together had a maximum storage capacity of over seven and a half million acre feet. The Guadalupe Basin had one very large dam, Canyon Dam, built in 1964 with a maximum reservoir storage of over a million af. Canyon Dam was over eight times larger than the second largest dam in the river basin, and accounted for two thirds of the total reservoir storage.

The Colorado River Basin experienced the construction of a majority of its large dams from 1940 to 1959, with one very large dam built during this same period. The Denison Dam was commissioned on the Red River in 1944, with a maximum storage capacity of 10.6 $x10^9$ m^3 (8,600,000 af). The Denison dam was the largest dam in the Red Basin by a margin of over 9.3 $x10^9m^3$ (over seven and a half million af).

The only very large dam in the Nueces Basin was built in 1982, and with a storage capacity of over a billion meters cubed (over a million af), it had twice the maximum storage capacity of the second largest dam in the basin. The San Antonio Basin did not contain any very large dams but had five large dams. Two were built between 1900 and 1939, with a combined maximum storage of 349,220 af that only accounts for nearly half of the total reservoir storage in the basin, and three were commissioned during the 1960 to 1979 time period with a combined maximum storage of 148,787 af.

2.3.4. Owners

The largest percentage of owners in Texas were private entities (N = 4349, 60.2%), and the second largest group of owners were local governments (N = 2520, 34.8%). This was also true in each river basin, except for the Trinity and Colorado river basins, where local governments owned the majority of dams, 57.7% (N = 1031) and 56.6% (N = 440) respectively. Private entities owned 79.7 % (N = 1152) of small and 56.4% (N = 3150) of medium dams in the state, over 60% of small dams in each basin, and nearly 50 to over 90% of the medium dams in the majority of the river basins. Local governments owned the majority of large dams in all river basins, except in the Guadalupe where the state owned the majority of large dams. The federal government owned over 50% of very large dams in Texas, and this is fairly consistent across all river basins. Data shown in Appendix 2.3.

2.3.5. Primary Purpose

The most common primary purpose for all dams in Texas was flood control and storm water management (31.7%), followed by recreation (20.8%) and water supply (13.8%) (Figure 2.5). Only a small percentage of dams in Texas listed no purpose (7.1%) or "Other" as the primary purpose (6.1%) (Figure 2.7). The variety of primary purposes declined as dam size increased, and the sharpest decline occurred from large to very large dams.

For small dams, the most common primary purposes included recreation (28.6%) followed by fire protection, stock and farm pond (15%) (Figure 2.5). Most medium dams had flood control and storm water management (36.6%) as a primary purpose, followed by recreation (19.4%) and water supply (14.2%) (Figure 2.5). Only small and medium dams had fire protection, stock, and farm ponds as primary purposes, and there were no very large dams with recreation as the primary purpose (Figure 2.5). Very large dams were for flood control and storm water management (58.6%), followed by water supply (27.6%), irrigation (10.3%), and hydroelectric power generation (3.5%) (Figure 2.5). The number of dams without a reported purpose declined as the size of the dams increased, with over 17% of small dams having no listed purpose (Figure 2.5). All very large dams had a reported purpose (Figure 2.5).

The most common primary purpose for dams built between 1800 and 1899 was recreation (26%), flood control and storm water management (13.5%)

and irrigation (13.8%) (Figure 2.6). Recreation was the most prevalent primary purpose for dams completed during the 1900 – 1939 time period (36.4%), while irrigation (23.9%) and water supply (21%) were the second and third most common (Figure 2.6). Flood control and storm water management (28%) and recreation (24.5%) were the most common primary purpose for dams built from 1940 to 1959 (Figure 2.6). Dams completed between 1960-1979 had flood control and storm water management (36.8%), recreation (18.3%) and water supply (14.9%) listed as the top primary purposes (Figure 2.6). Similarly, dams constructed from 1980 to 2014 had the primarily purpose of flood control and storm water management (39.1%) reported most frequently, followed by recreation (16.9%) and irrigation (9.5%) (Figure 2.6). Most of the dams built in the 1800s had no listed purpose (20.8%), while only 3.3% of dams built between 1900-1939 did not include a purpose. Of the dams built during the most recent time period, 12.7% of had no listed purpose (Figure 2.6).

Primary purposes for dams in the Brazos, Red, and Guadalupe river basins generally followed state level trends (Figure 2.7). A noticeably larger proportion of the dams in the Trinity, Colorado, and San Antonio river basins reported flood control as their primary purpose, 56.9%, 46.2%, and 34.6% respectively (Figure 2.7). In the Sabine (43.3%) and Neches (50.8%) river basins recreation was the primary purpose for the majority of dams, while the majority of dams in the Nueces River Basin listed water supply (46.5%) (Figure 2.7). In the Rio Grande and Nueces river basins, there were a large number of dams without any purpose listed compared to the other river basins (25.9%, 18.1% respectively) (Figure 2.7).

2.3.6. Connectivity Model

The GSARS has been increasingly fragmented over time by the growing number of dams in the river system. A majority of this fragmentation has occurred in the mid-section of the river basins, where urban centers such as San Antonio, San Marcos, and Southeast Austin are located (Figure 1.2). Areas with a higher density of dams created smaller FRNs, and many of these appeared to be located along smaller tributaries and headwater streams.

Before the 1900s, when the San Antonio River network was largely connected, the mainstem river could be considered one large FRN, with only eleven other FRNs, all under 23 km in length, located on small tributaries (Figure 2.8, a). By this same time period, the Guadalupe River network, was already fragmented into three large FRNs (Figure 2.8, a). From 1900 to 1959 the mainstem San Antonio River remained largely connected as one FRN, apart from the fragmentation caused by the Medina Lake Dam, built in 1913 with a storage capacity of 327,250 af. The Medina Lake Dam divided the upper portion of the San Antonio River network from the rest of the river network creating the second largest FRN in the San Antonio River Basin (Figure 2.8, b). The San Antonio River network became increasingly segmented by dams, and all the FRN's are less than 200 km (Figure 2.8, c,d,e). Only in the sixth version of the model, when other sources of fragmentation were considered, is the San Antonio River network fragmented into several large FRNs.

By 1899, two dams already separated the majority of Guadalupe River network into three large FRNs (Figure 2.8, a). The Cuero Lake dam built on the Guadalupe River in 1898 with a storage capacity of 808 af, separated an FRN of

14,147.9 km. The Martindale Gin Dam was completed on the San Marcos River, a major tributary of the Guadalupe River, in 1892 with a storage capacity of 60 af and separated an FRN of 1,935.7 km.

In 1900, The Gonzales Dam was commissioned with a storage capacity of 650 af. This dam separated 3,087 km of river length from the larger Guadalupe River network (Figure 2.8, b). Additionally, in 1922 the Mission Valley Mills Lake Dam was built on the Guadalupe River with a storage capacity of 94 af that further fragmented the Guadalupe River network, severing an FRN of 5,736.9 km (shown in blue grey, Figure 2.8, b). Together the length of FRN disconnected by the Gonzales and Mission Valley Mills Lake dams shrunk the FRN associated with the Cuero Lake Dam by 8,826.987 km (shown in dark blue in Figure 2.8, b). The Guadalupe River network was further fragmented by dams through the 1900s, that divided the river network into hundreds of FRNs (Figure 2.8, c, d).

Canyon Dam, built in 1964, was the only very large dam in the GSARS (1,129,300 af). This dam separated 2,705.4 km of upstream river length, and shortened the existing FRN detached by the Mission Valley Mills Lake Dam to 650.1 km (Figure 2.8, d). The last major separation occurred when the Coleto Creek Dam was completed in 1980 on a tributary of the Guadalupe River, the dam (13,253 af) separated a length of 1,690.9 km of river network within the GSARS (Figure 2.8, e).

In the sixth iteration of the model, when additional causes of fragmentation were considered, 28.9% of FRNs were caused by registered dams. Considered separately, however, registered dams only accounted for 22.3% of the FRNs in the San Antonio River Basin, but 37.4% in the Guadalupe River Basin.

Registered dams caused most of the large FRNs in the Guadalupe River Basin, while in the San Antonio River Basin there were a number of large FRNs caused by other sources of fragmentation such as unregistered dams, artificial flowlines, knick points, etc (Figure 2.8, f).

As the number of dams and FRNs increased over time, the mean FRN size decreased by an order of magnitude from 1899 to 2014, and decreased by a second order of magnitude when considering additional causes of fragmentation in the GSARS (Table 2.3). The largest FRN in the GSARS that flows to the San Antonio Bay, has shrunk by over a 1,000 km over time due to dam building and other sources of fragmentation (Figure 2.8). The smallest FRN in the river system changed as new dams were built that separated smaller sections of river.

While the average length of FRNs shrunk over time, the number of large FRNs, particularly those over 1000 km in size remained relatively stable (Figure 2.9). Of the 30 FRNs over 100 km in size in the fifth iteration of the model, 29 were the result of registered dams, and six of these dams were large or very-large dams (Figure 2.10). The Medina Lake, Martindale Gin, Gonzales, Cuero Lake, and Mission Valley Mills Lake dams, alternatively demonstrated the ability of small- and medium-sized dams to separate large river segments. However, these FRNs represented less than one percent of the FRNs small- and medium-sized dams created in the GSARS, and dam size and FRN length were positively correlated (see below) (Figure 2.10).

The number of smaller FRNs, especially those between 1 and 100 km, increased dramatically (Figure 2.9). For the first three iterations of the model, the largest number of FRNs were those between 10 and 100 km in size (Figure 2.9).

In the fourth iteration of the model the largest number of FRNs were those between 1 and 10 km under 10 km in length (Figure 2.9). By 2014, the number of FRNs between 1 and 10 km represented nearly half of all FRNs disconnected by registered dams (Figure 2.9). This reflected the increasing number of FRNs that chipped away at the length of existing FRNs in the river system, and highlighted the substantial number of dams built between the years of 1960 and 1979.

In the sixth iteration of the model, the number of FRNs between 0.1 and 1 km rose by an order of magnitude, with 450 FRNs in this size category compared to 54 in the fifth iteration of the model (Figure 2.9). The number of FRNs between 1 and 10 km in size, saw a similar increase from 143 in the fifth iteration of the model to 482, while the number of FRNs between 10 to over 1000 km only slightly increased (Figure 2.9). This indicated that registered dams caused the majority of larger FRNs, while other sources of fragmentation caused smaller FRNs.

Most of the dams in the GSARS were small- and medium-sized dams (Table 2.4). Only 31 registered dams occurred in the GSARS as of 1899, but by 2014 there was a total of 359 dams (Table 2.4). 56% of the dams in this river system were built during the 1960 to 1979 time period, and only 49 additional dams were completed between 1980 and 2014 (Table 2.4). The mean length of FRNs decreased over time for all size categories of dams (Table 2.5). Large and very large dams caused correspondingly larger FRNs (Table 2.5).

Dam size and FRN length had a positive relationship, indicating that as dam size increased so did the length of the FRN they detached from the larger river network. There was a slight negative correlation between a dam's age and their associated FRN after removing dams built in 1800 (Figures 2.10, 2.11). This seemed counterintuitive, but was largely explained by the explosion of small- and medium-sized dams completed in the GSARS after 1960 (Figure 2.12).

2.4. DISCUSSION

Compared to the 2005 NID data presented in Chin et al. (2008), the number of dams in Texas increased for all sizes, except large dams (Appendix 2.1). This decline in large dams was accounted for by differences in state and federal data recording. The NID included dikes and levees and used average reservoir storage to classify size. Since 2005 the percentage of storage increased for medium and very large dams, but stayed the same for small dams. Small and medium sized dams continued to dominate by sheer numbers. Private entities owned most of the dams in Texas, but local governments owned most of the large dams and federal governments owned most of the very large dams. The primary purpose for most large and very large dams was flood control, while for small dams it was recreation and fire protection.

2.4.1. Climatic and Geographic Trends

As documented in a previous studies, dam distribution was related to the climate gradient and location of urban centers in Texas (Chin et al. 2008; Graf 1999). Precipitation decreased and PET increased generally from east to west, and most of the dams in Texas occurred in the wetter eastern portion of the state. Further, basins that receive 762 mm or less of average annual precipitation have a larger percentage of dams, indicating the importance irrigation plays in dam construction. Additionally, the Nueces River Basin had the highest PET, received less than 762 mm of water a year on average, and was the only river basin where the majority of dams were used for water supply. This may indicate the added importance of securing elusive water supplies in this west Texas river basin.

The Rio Grande River Basin contained nearly 20 percent of Texas' land mass, but less than 5 percent of its dams. The low number of dams compared to surface area in this river basin is most likely the result of extremely low precipitation and sparse population throughout much of the basin. In contrast to the Rio Grande River Basin was the Trinity River Basin which was less than 7% of Texas' surface area, but had nearly a fourth of all Texas dams. Additionally, local governments owned higher percentages of dams, and a much larger percentage of dams were primarily for flood control in this river basin. These trends are probably best explained by the eastern location of the river basin, which resulted in relatively higher amounts of precipitation, and the location of the Dallas – Fort Worth area with a population over a million in the upper portion of the river basin.

The Colorado River Basins had the second highest numbers of dams. As with the Trinity River Basin, local governments owned higher percentages of dams, a much larger percentage of dams were primarily for flood control, and a large urban area is located in it. The city of Austin with a population of nearly 800,000 is in the middle Colorado River Basin, so that here again the influence of population is demonstrated. In addition to receiving less than 762 mm of precipitation a year, the Colorado River Basin is in central Texas, one of the most flash flood prone areas in the United States according to the National Weather

Service. The increased chances for both floods and droughts, and the location of a large urban area within its borders demonstrates how both climate and population have led to increased numbers of dams in this river basin.

Texas has more dams than any other state (USACE 2016), and in a previous study, the Texas-Gulf water resource region had one of the highest ratios of storage capacity to drainage area (Graf 1999) further demonstrating the fragmented state of the Texas' rivers. The main hydrologic effect of medium and small-sized dams on river landscapes has been fragmentation (Chin et al. 2008), and 97% of the dams in Texas were small and medium-sized. River fragmentation has led to declines in fish and mussel populations (Richter et al. 1997; Wofford et al. 2005), and alter migration routes (Jager et al. 2001).

The amount of storage established in the United States rapidly increased during 1950s through 70s, with only minor increases after 1980 (Graf 1999). Texas has a pronounced history of flooding and drought (TWDB 2017), and the river basins of Texas have been documented as having some of the highest runoff to storage ratios (Graf 1999). In Texas, very large dams accounted for the smallest number, and unlike other size categories their temporal pattern of commissioning was not uniform across the different basins. The variation in number of very large dams by basin may reflect the amount of rainfall and runoff available to store in a basin, and the variation in the time of their construction is likely due to the large capital and planning required to build them. The time period when the bulk of storage is created in a basin is directly linked to when these very large reservoirs are built. This is particularly well demonstrated in the Red, Nueces, Rio Grande, and San Antonio river basins.

The construction of large and very large dams may also reflect the number of prime locations available for these structures in a river basin. The larger number of dams commissioned between 1900 and 1979 with water supply recorded as the primary purpose potentially reflects the increased scarcity of locations to build large water supply dams (Reisner 1986). Recreation was the main use for dams built before 1900 and most of these dams were small or medium-sized dams. The shift in the primary purpose to flood control for dams built in the 1940s and 1950s is potentially linked to increased population and the advancement of engineering capabilities required to build large flood control dams (Solomon 2010; Petts & Grunell 2005; Reisner 1986). Irrigation increased from 10.3% in the 1800s to nearly a fourth of all primary purposes for dams built during the mid-20th century. After this time period irrigation declined as a primary purpose potentially exhibiting the increased agriculture in the state of Texas during the 1800s and early 1900s, and then the impact of the drought of record in the 1950s on the industry.

The 1960s and '70s have often been referred to as the 'dam building era' in the United States, and the greatest increase in dams nationally occurred from the late 1950 to the late 1970s (Graf 1999). Similarly, in Texas, the majority of dams dated back to this time period. After 1980, the pace of dam construction slowed in the United States (Graf 1999) including Texas. However, 26 new major reservoirs are being built in Texas to secure the state's future water supply and help mitigate drought impacts (TWDB 2017). These new dams will increase the fragmentation of Texas water landscape, and make it even harder to maintain a balance between ensuring enough water for human water use and maintaining ecologically health rivers.

2.4.4. Connectivity Models in the GSARS

Fragmentation was concentrated around urban centers, where large and growing populations contributed to increasing numbers of dams (Chin et al. 2008; Graf 1999). The majority of fragmentation by dams resulted in small sections of the river network being separated, with many of these small FRNs located in headwaters and small tributary streams. Dams that segmented large FRNs occurred along major tributaries and main stem rivers.

Dams have fragmented the GSARS in two main ways. First, large sections of the river network were separated sporadically depending on when certain dams are completed in the GSARS. Large and very large dams in the GSARS corresponded to larger upstream FRNs, and dams with large storage capacities are generally built in locations where they will receive enough inflows to fill their associated reservoirs. Small- and medium-sized dams also separated FRNs over 100 km due to their location in the river network, but generally created proportionally smaller FRNs. These large separation events occurred infrequently, but were responsible for segmenting the GSARS into a series of disparate FRNs.

Second, the number of small- and medium-sized dams have separated hundreds of smaller FRNs from the river system. The vast majority of FRNs created by small- and medium-sized dams were under 10 km. This illustrates the dominating impact these dams have in the GSARS through the sheer number of discrete river segments they create. These small FRNs have gradually chipped away at the size of larger FRNs and resulted in a continuous reduction of connectivity over time.

Despite separating relatively smaller FRNs from the river system the substantial number of small and medium-sized dams can have dramatic impacts on the ecology of river systems. Small FRNs are located predominately on headwaters and/or minor tributaries of the GSARS, potentially separating source populations from downstream populations of aquatic biota. This separation can then lead to the extirpation of species in downstream river segments (McCluney et al. 2014). The isolation of source populations may have resounding effects on the population dynamics (McCluney et al. 2014), biodiversity (Freeman et al. 2007), and gene flow (Sterling et al. 20123) in the river system as cumulative numbers of source populations become isolated.

Registered dams accounted for a low percentage of the total fragmentation in a river system. Other sources of fragmentation include unregistered dams (Chin et al. 2008), road crossings and culverts (Park et al. 2008), and artificial pathways that disrupt the natural connectivity of a river. Some of these additional sources of fragmentation appear to be represented in the NHD as abrupt changes in river angle and/or elevation. Improved recording of artificial river barriers will increase the accuracy of connectivity models and allow for further scientific inquiry into how types of barriers effect connectivity.

2.4.5. Potential Applications of Connectivity Modeling

The methodology presented here allows for a way to analyze not only the extent but the temporal evolution of fragmentation in a river system. Determining when FRNs were separated by anthropogenic barriers in a river system can be used as a measure of isolation for specific populations and/or communities of aquatic biota. The age of barriers was an important variable in assessing species isolation (Roberts et al. 2013), and period of isolation has been linked to higher extirpation rates of stream-dwelling fishes from river segments (Morita and Yamamoto 2002). Anthropogenic fragmentation of rivers resulting in species isolation has also been found to result in genetically distinct populations (Sterling et al, 2012), and may potentially act as a catalyst for speciation over longer time periods. A method for determining the length of species isolation can thus facilitate research on and predictions of extirpation and speciation in river environments.

The models of connectivity presented in this research are largely constrained by data availability. The incorporation of additional data sources will produce more accurate depictions of river fragmentation. A substantial number of small- and medium-sized dams go accounted for in state and federal databases (Chin et al. 2008). These undocumented barriers and can located and georeferenced using high resolution aerial imagery to better capture the extent of fragmentation caused by dams in river systems. Culverts and road crossings also segment river systems (Park et al. 2008) and may greatly outnumber recorded dams in a river system (Januchowski-Hartley et al. 2012). Hanging culverts result in hydrologic changes that can impede native species migration, while favoring non-native species dispersal (Foster and Keller 2011). Additional data on road crossings and culverts can be acquired through aerial imagery and datasets available through state departments of transportation and incorporated into future models of connectivity using the methodology presented in this study.

Models of connectivity that incorporate drying events can also be generated using the BAT, by using data on the location of dry or potentially dry stream segments as barriers. While available data may be limited or difficult to acquire, incorporating fragmentation related to intermittent or ephemeral stream reaches can facilitate more accurate models of connectivity, and highlight the increasing occurrence of drying events caused by human alteration in river systems (Darty et al. 2014). Modeling the fragmentation caused by drying events may be particularly useful for demonstrating the effects of climate change on river systems (Jaeger et al. 2014; von Schiller et al. 2011).

The FRNs produced using the BAT are one way of characterizing connectivity in a river system, and can be overlain or refined by other measures of connectivity such as the dendritic connectivity index (Cote et al. 2009), graph-based approaches to connectivity (Eros and Grant 2012, 2015), and path counting metrics (Malvadkar and Leon 2015). Species specific connectivity metrics can also be used to further refine FRNs, such as dispersal ability (Malvadkar and Leon 2015; Perkin et al. 2013), and/or the importance of migration in species' lifecycles (Nunn and Cowx 2012). The number and type of connectivity measures used or combined should be determined by the research goals and/or species of consideration. Incorporating multiple measures of connectivity in future research will generate more accurate and hopefully useful representations of fragmentation in river systems.

Providing models of river connectivity can help inform river restoration, as restoration activities, especially dam removals, can be prioritized by cumulative length of resulting FRN. Targeted restoration and conservation activates can be prioritized using FRN based on the required or best suited FRN size, structure, or location for specific species or organisms. Restoring river connectivity has been shown to have a positive effect on multiple aquatic species (Bednarek 2001; Gottgens 2009; Hogg et al. 2015; Stanley and Doyle 2003). For example, reconnecting fragmented habitat has been shown to be an essential part of restoring metapopulation structure to charr and salmon (Tsuboi et al. 2010), and lengthening river fragments where possible has been promoted as the best management option to preserve Cutthroat Trout (Roberts et al. 2013a).

Additionally, river connectivity, particularly longitudinal river connectivity (de Araujo and Medeiros 2013) is a critical component of landscape connectivity (Bruzzi et al. 2014; Wetth et al. 2014). Rivers act as high-speed corridors that increase colonization speed of terrestrial environments. Dispersal limited plant species have been found to be efficiently transported over long distances and short periods by rivers (deAraujo and Medeiros 2013). Distance from permanent water sources has been found to be an important aspect of quality habitat for amphibians (Bruzzi et al. 2014), and genetic isolation of riparian plants can be caused by the fragmentation of rivers (Werth et al. 2014). Thus, the restoration and conservation of river connectivity also facilitates the restoration and conservation of landscape connectivity.

2.5. CONCLUSION

Since 2005, the number of dams in Texas has continued to grow. A basin-scale analysis of dams in ten major river basins, accounting for over 80% Texas' dams, produced similar results to a state scale analysis. However, primary purpose of dams varied between basins, as did, the timing for when the greatest storage was created in each basin. Climate factors, mainly precipitation, influenced dam placement. Population as a variable was not directly measured in this study, but nevertheless was a noticeable influence on the spatial distribution of dam placement and function.

Dams disrupt the natural connectivity of rivers, but the degree to which they impact the shape and size of FRNs appears to depend largely on their location and when they were completed. The spatial arrangement of dams in a river system creates unique patterns of fragmentation that change through time as new dams are built. Connectivity in a river system appears to gradually decrease as the construction of most dams separate only small FRNs. This is punctuated by sporadically substantial changes in the structure of a river systems connectivity by the completion of dams in locations that separate extensive lengths of river network. Such events can effectively split a river network in half, drastically altering the shape and size of aquatic ecosystems.

As registered dams account for only a portion of the total amount of fragmentation in a river system, models of river connectivity and fragmentation should continue to be refined and incorporated into river management and decision making about restoration and conservation strategies. Providing models for river connectivity in an entire river basin and/or system allows for better decision making in river basin management. Models of connectivity, such as the ones created in this research, allow the removal or completion of a dam to been seen within the greater context river system fragmentation.

The BAT allows for the analysis of the spatial and temporal changes in river connectivity and fragmentation at small and large scales. Modeling fragmentation at broader scales contributes to the understanding of dam impacts on whole river systems. The analysis of fragmentation at temporal scales generates a better understanding of how fragmentation has progressed and changed over time in a river system. The methodology presented here has multiple applications in both river research and management arenas.

2.6. TABLES AND FIGURES

Table 2.1 Size classifications based on Graf 2005.

Size Classification	Max. Reservoir Storage (m^3)	Max. Reservoir Storage (af)
Small	< 100,000	< 100
Medium	100,000 – 10,000,000	100 – 10,000
Large	10,000,000 – 1,000,000,000	10,000 – 1,000,000
Very Large	>1,000,000,000	>1,000,000

Table 2.2 Climatic and geographic variables for major Texas river basin.

	General	Precipitation (mm)			Potential ET (mm)			Area (km^2)		Dams		Total Reservoir Storage	
	Location	Mean	Min	Max	Mean	Min	Max	N	%	N	%	($^\wedge 10^9$ m^3)	%
Texas	---	728.4	206.7	1563.0	1496.5	1124.0	1878.0	695,620	100.0	7274	100.0	128.7	100.0
Cypress	Eastern	1224.8	1130.6	1309.6	1412.1	1373.0	1438.0	7,617	1.1	161	2.2	21.0	16.3
San Jacinto	Eastern	1258.8	1119.9	1425.8	1438.2	1316.0	1486.0	10,241	1.5	162	2.2	10.8	8.4
Sulphur	Eastern	1202.5	1100.4	1295.6	1390.2	1351.0	1427.0	9,302	1.3	162	2.2	10.6	8.2
Neches	Eastern	1305.8	1066.3	1527.2	1458.2	1359.0	1498.0	25,858	3.7	308	4.2	9.2	7.1
Sabine	Eastern	1261.0	1042.3	1560.6	1433.7	1355.0	1486.0	19,691	2.8	335	4.6	3.2	2.5
Trinity	Eastern	1050.8	765.7	1528.8	1439.6	1335.0	1508.0	46,587	6.7	1787	24.6	15.4	12.0
Canadian	Northern	495.0	382.0	622.8	1375.4	1302.0	1459.0	33,248	4.8	153	2.1	3.4	2.6
Red	Northern	683.9	463.4	1322.1	1422.0	1313.0	1530.0	63,027	9.1	619	8.5	18.3	14.2
Colorado	NW - SE	617.7	339.2	1215.6	1515.4	1298.0	1597.0	102,577	14.7	775	10.7	15.0	11.7
Brazos	NW - SE	760.7	442.5	1378.4	1466.0	1318.0	1544.0	111,457	16.0	1392	19.1	1.87	1.5
San Antonio	South-central	811.7	709.2	986.7	1526.8	1381.0	1589.0	10,866	1.6	160	2.2	1.94	1.5
Guadalupe	South-central	874.9	714.7	1026.7	1511.7	1391.0	1565.0	15,480	2.2	215	3.0	0.87	0.7
Nueces	Southwest	639.4	495.5	854.8	1608.7	1446.0	1712.0	43,380	6.2	456	6.3	2.25	1.7
Rio Grande	W - S	387.7	206.7	688.3	1602.8	1124.0	1879.0	128,437	18.5	326	4.5	13.4	10.4
Lavaca - Guadalupe	Coastal	1047.1	946.3	1117.2	1384.2	1238.0	1507.0	3,339	0.5	8	0.1	13.4	10.4
San Antonio - Nueces	Coastal	862.4	729.6	992.4	1466.3	1261.0	1575.0	7,855	1.1	10	0.1	0.18	0.1
Colorado - Lavaca	Coastal	1129.9	1030.9	1208.5	1380.7	1247.0	1478.0	3,289	0.5	11	0.2	0.05	0.04
Trinity - San Jacinto	Coastal	1419.6	1371.7	1441.9	1375.0	1321.0	1439.0	1,011	0.1	14	0.2	0.42	0.3
Neches - Trinity	Coastal	1471.2	1243.6	1547.1	1344.6	1168.0	1431.0	4,381	0.6	15	0.2	0.02	0.01
Lavaca	Coastal	1054.3	937.1	1177.0	1479.7	1353.0	1523.0	6,003	0.9	24	0.3	0.06	0.05
Brazos - Colorado	Coastal	1193.8	1051.7	1328.4	1426.9	1297.0	1514.0	4,846	0.7	26	0.4	0.32	0.2
San Jacinto - Brazos	Coastal	1335.1	1157.6	1469.1	1333.3	1147.0	1463.0	4,508	0.6	51	0.7	0.01	0.005
Nueces - Rio Grande	Coastal	637.5	511.6	897.3	1553.7	1260.0	1665.0	29,668	4.3	101	1.4	0.01	0.01

Table 2.3. Descriptive statistics for FRNs.

Model Iteration	FRN Length (km) Mean	Min	Max	Total FRNs
1800 - 1899	1618.3	0.02	19785.9	32
1800 - 1939	690.5	0.02	17457.6	53
1800 - 1959	180.5	0.02	17103.5	109
1800 - 1979	117.7	0.01	15738.4	311
1800 - 2014	101.7	0.01	13832.0	360
Unedited	33.8	0.004	8734.5	1240

Table 2.4. Dam sizes and percent of FRNs caused by registered dams.

	Dam Sizes								Total Dams
	Small		Medium		Large		Very Large		
Model Iteration	N	%	N	%	N	%	N	%	
1800 - 1899	20	64.5	11	35.5	0	0.0	0	0.0	31
1800 - 1939	25	48.1	22	42.3	5	9.6	0	0.0	52
1800 - 1959	34	31.5	69	63.9	5	4.6	0	0.0	108
1800 - 1979	66	21.3	235	75.8	8	2.6	1	0.3	310
1800 - 2014	79	22.0	268	74.7	11	3.1	1	0.3	359
Unedited	79	22.0	268	74.7	11	3.1	1	0.3	359

Table 2.5. Mean FRN length for each dam size classification.

	Mean FRN Length by Dam Size (km)			
Model Iteration	Small	Medium	Large	Very Large
1800 - 1899	113.6	1321.5	0.0	0.0
1800 - 1939	251.7	412.4	754.2	0.0
1800 - 1959	141.5	161.7	703.6	0.0
1800 - 1979	31.9	52.7	445.6	2795.3
1800 - 2014	36.8	46.6	481.5	2790.6
Unedited	39.8	42.4	497.4	2783.8

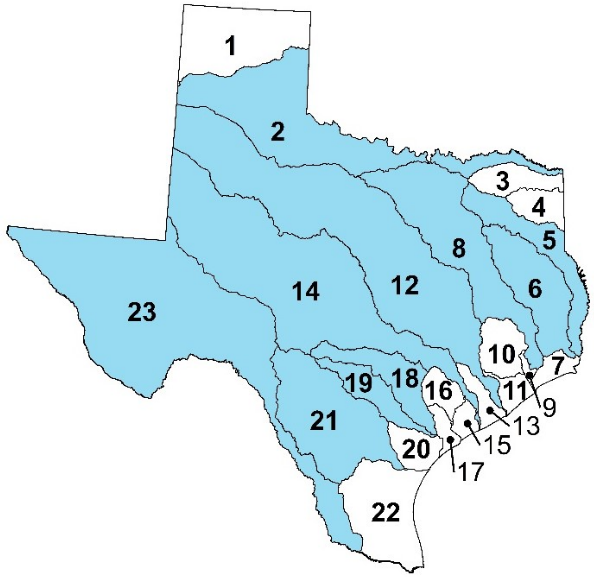

Figure 2.1. Map of Texas and 26 major river basins. River basins highlighted in blue represent the river basins analyzed in this study. Numbers correspond to river basin names: 1) Canadian, 2) Red, 3) Sulphur, 4) Cypress, 5) Sabine, 6) Neches, 7) Taylor Bayou, 8) Trinity, 9) Cedar Bayou, 10) San Jacinto, 11) Oyster Creek, 12) Brazos, 13) San Bernard River, 14) Colorado, 15) Tres Palacios Creek, 16) Lavaca, 17) Arenosa Creek, 18) Guadalupe, 19) San Antonio, 20) Arkansas River, 21) Nueces, 22) San Fernando Creek, 23) Rio Grande.

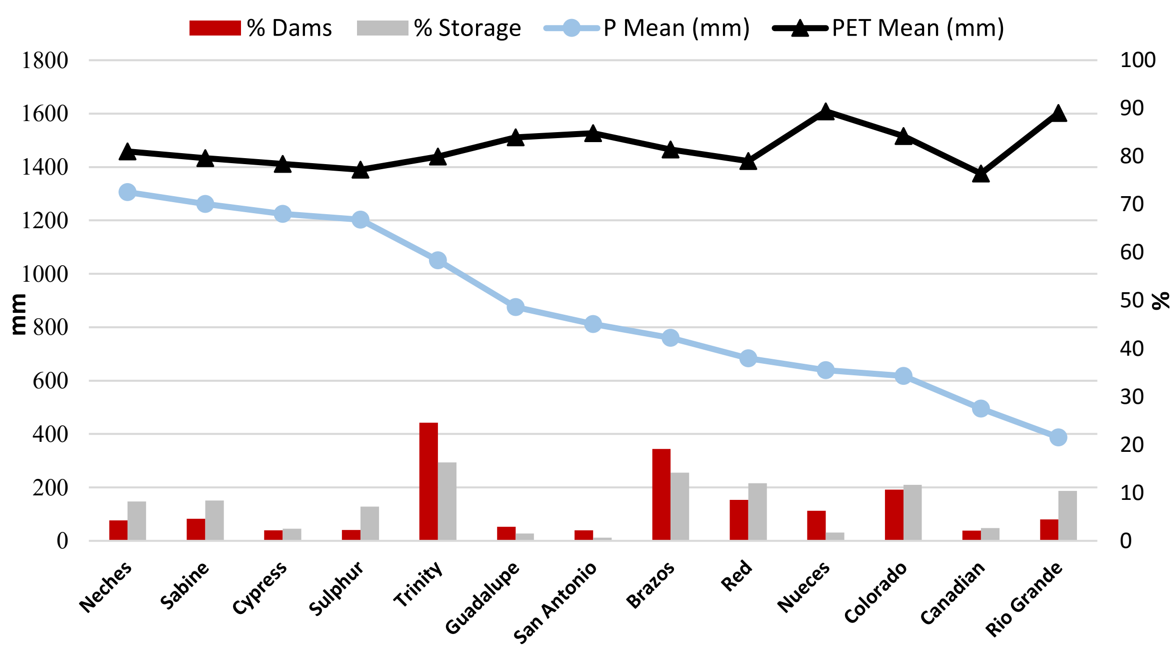

Figure 2.2. Climate and geographic distribution of dams. Percent of all Texas dams (red bars) and total reservoir storage (grey bars), with mean precipitation (blue line with circles) and potential evaporation (black line with triangles) in ten major river basins.

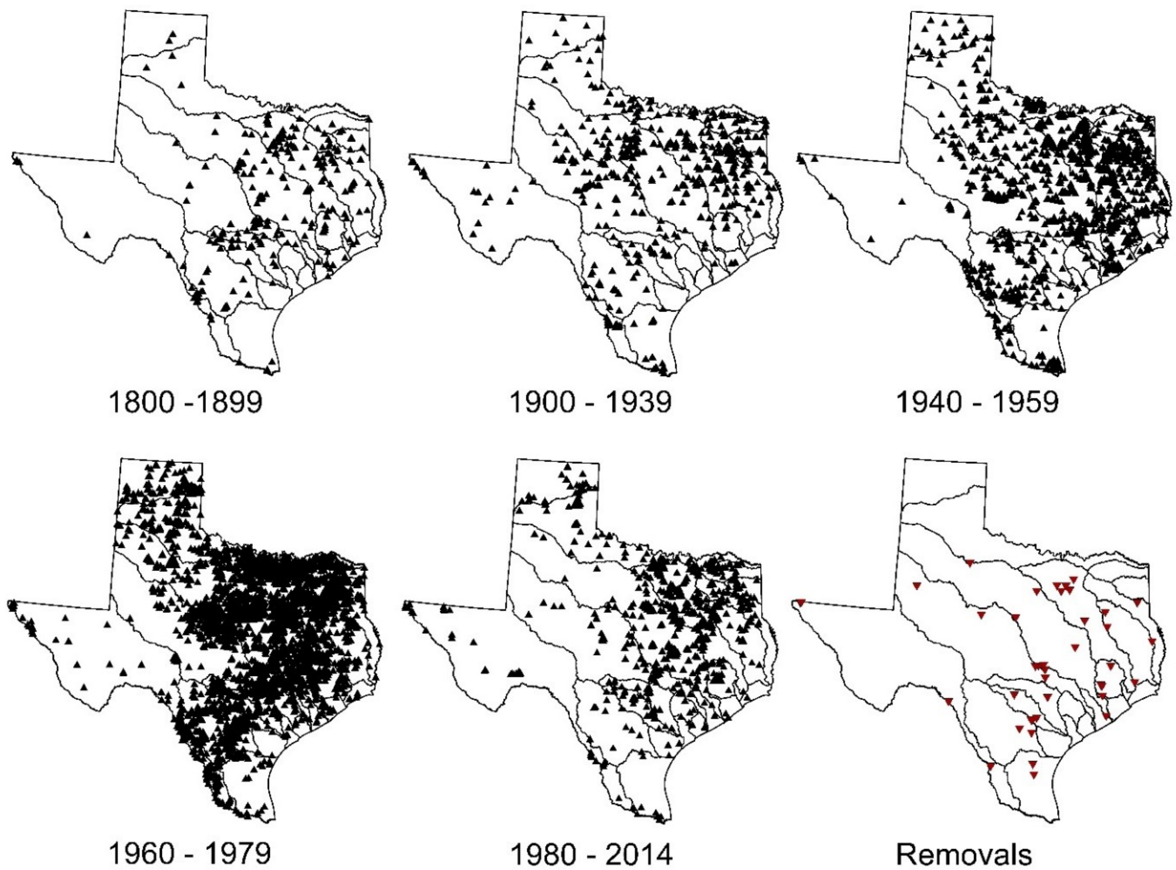

Figure 2.3. Location of dams completed during each time period and all dam removals in Texas.

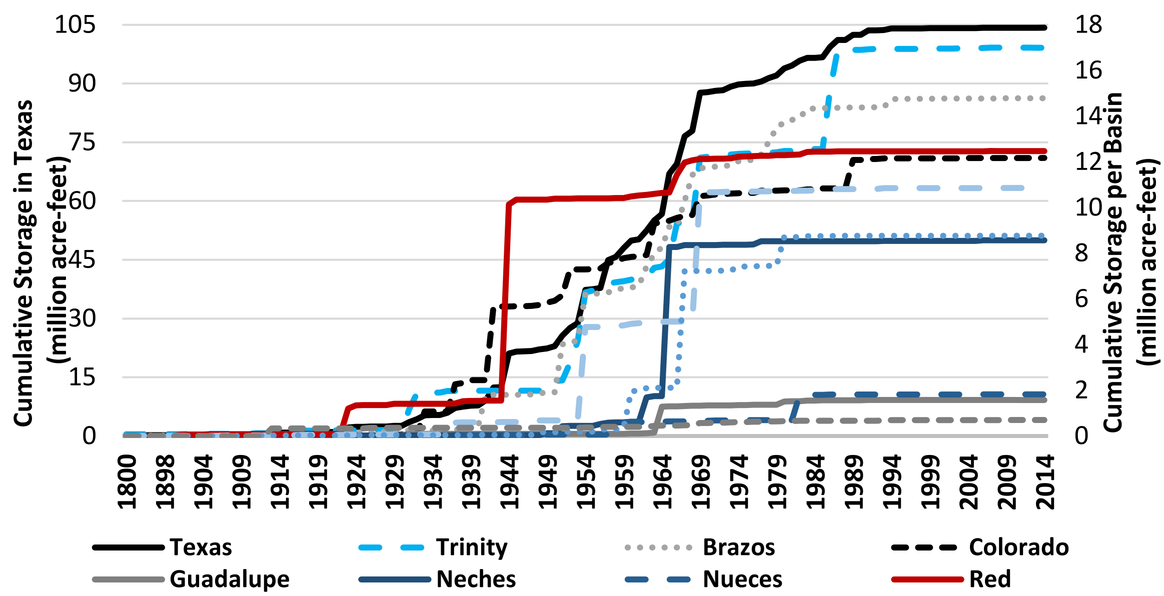

Figure 2.4. Total maximum cumulative reservoir storage in Texas and ten major river basins.

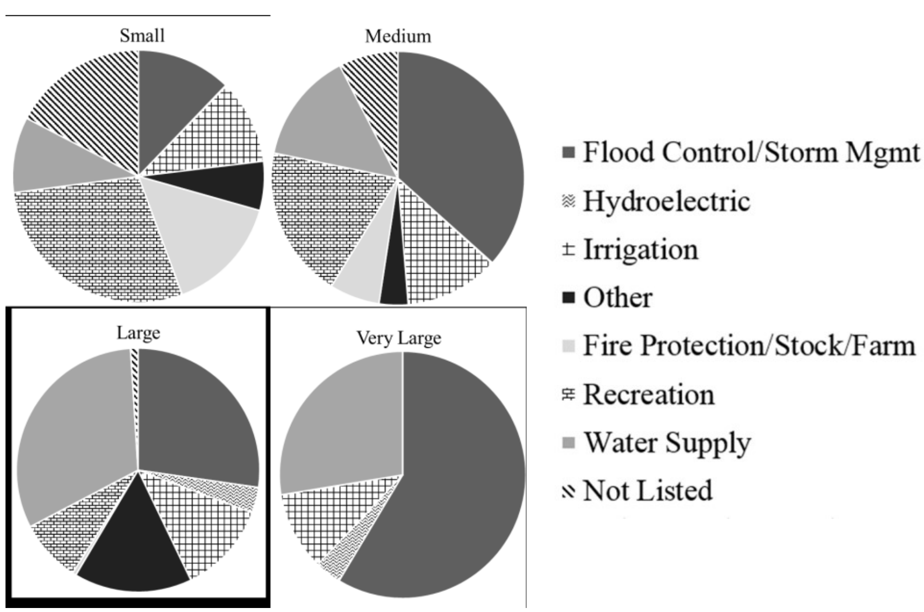

Figure 2.5. Primary Purpose for all dams in Texas based by dam size.

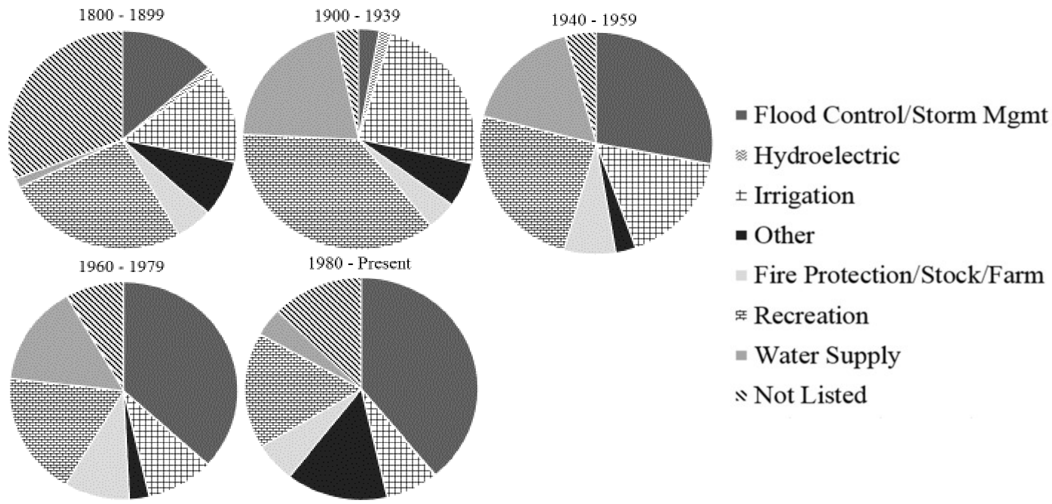

Figure 2.6. Primary purpose of dams by time period.

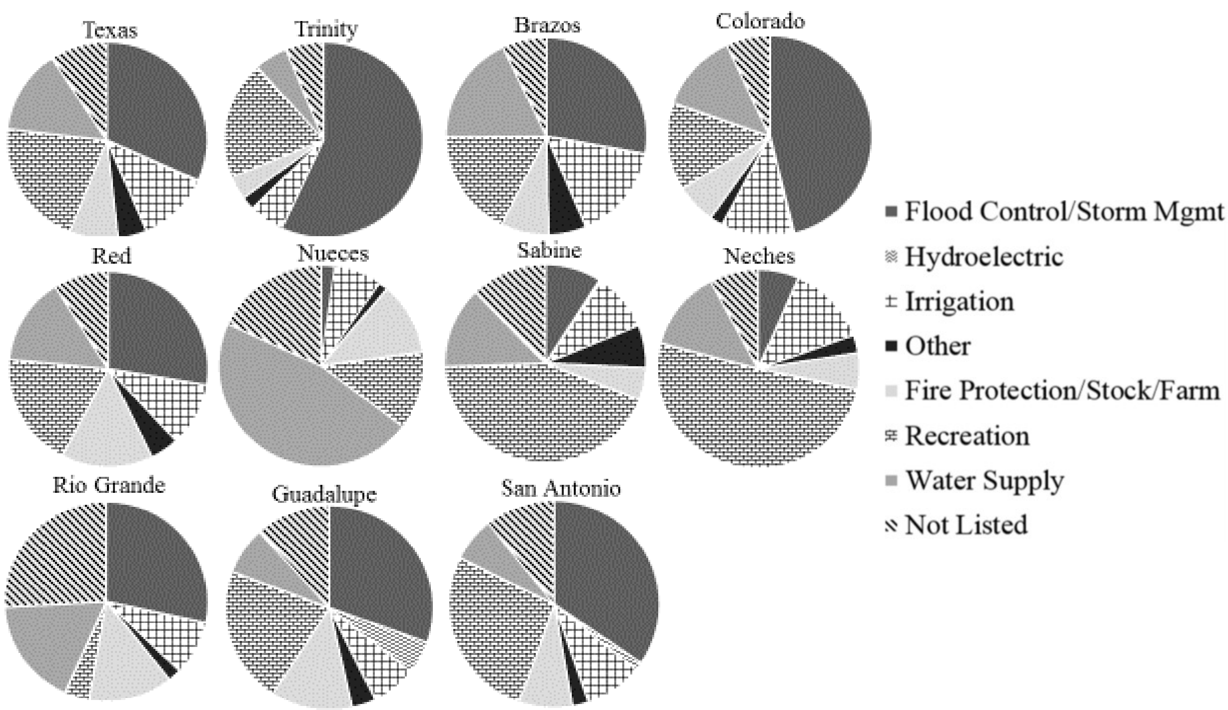

Figure 2.7. Primary Purpose of dams in ten major river basins.

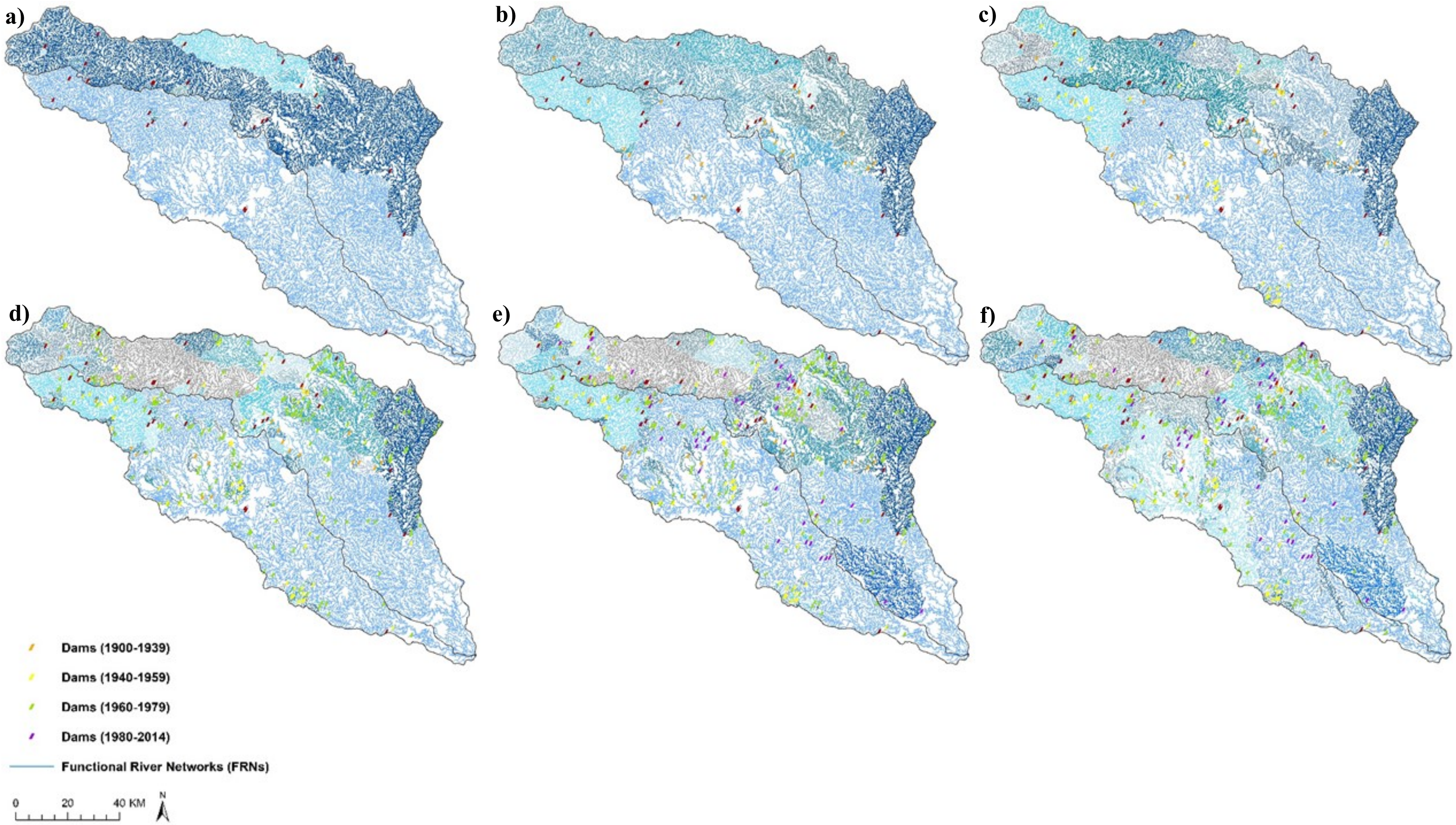

Figure 2.8. Connectivity models for the GSARS. FRNs modeled with edited flowlines and registered dams built between a) 1800 and 1899, b) 1800 and 1939, c) 1800 and 1959, d) 1800 and 1979, e) 1800 and 2014. The sixth iteration of the model (f) was created with registered dams built between 1800 and 2014 and unedited flowlines.

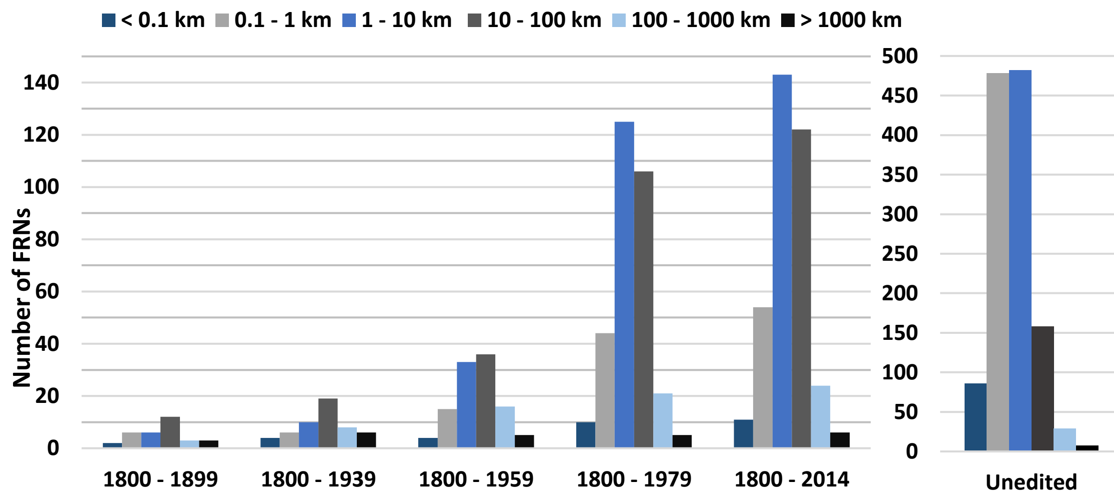

Figure 2.9. Number of FRNs by size classification for each iteration of the model.

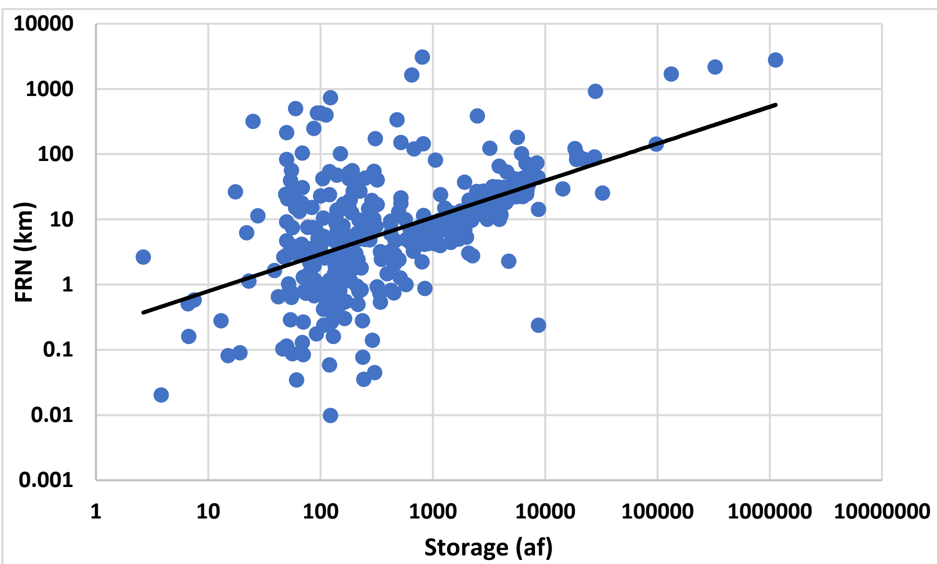

Figure 2.10. Scatter plot showing the relationship between maximum storage and FRN length for registered dams built between 1800 and 2014 in the Guadalupe – San Antonio River System.

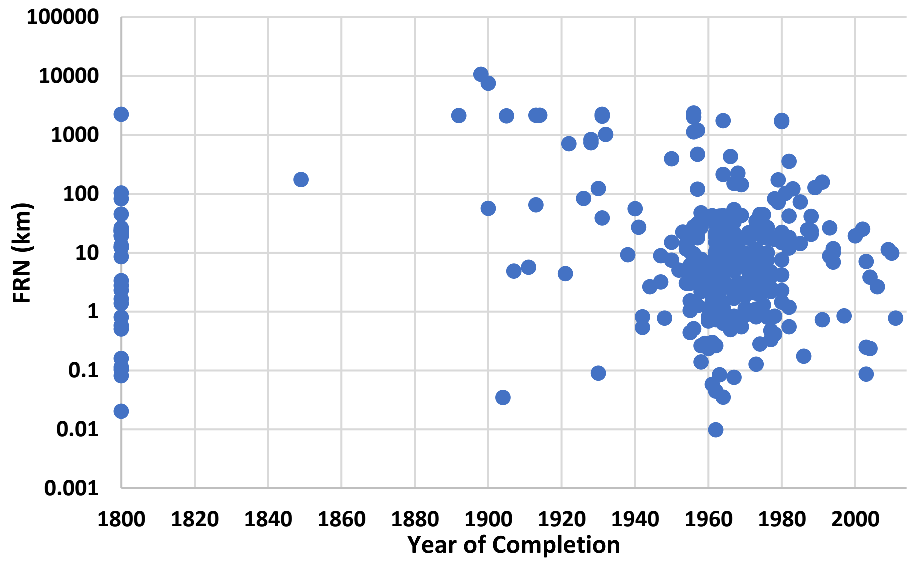

Figure 2.11. Scatter plot showing the relationship between year of completion and FRN length for registered dams built between 1800 and 2014 in the Guadalupe – San Antonio River System.

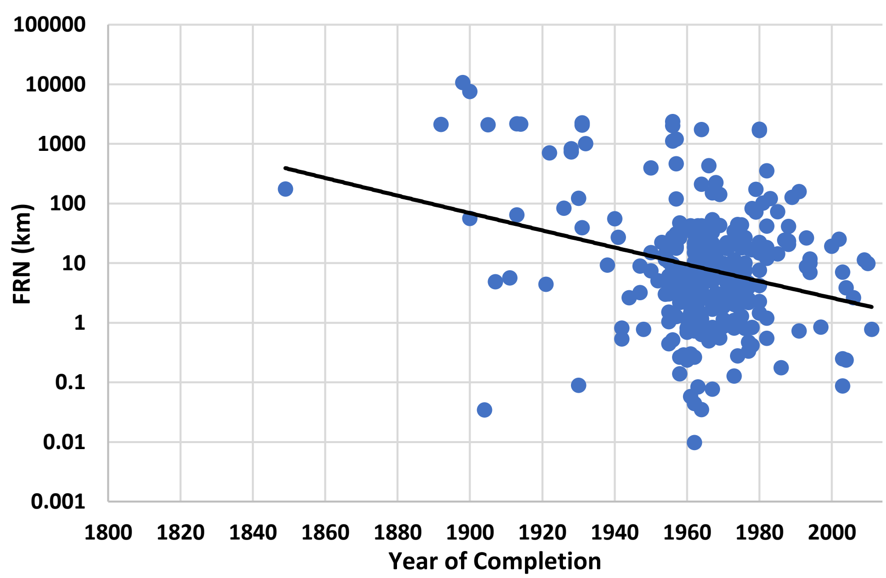

Figure 2.12. Scatter plot showing the relationship between year of completion and FRN length for registered dams built between 1800 and 2014 in the Guadalupe – San Antonio River System, with dams complete in 1800 removed.

3. SPATIAL ANALYSIS OF FRESHWATER MUSSELS

3.1. INTRODUCTION

The freshwater unionid mussel fauna of Texas consists of approximately 50 species, of which 15 are listed as threatened at the state level (TPWD 2016) (Appendix 1.1). Six of the 15 state-threatened species are also candidates for federal listing (TPWD 2016). Fourteen of the fifteen state-threatened freshwater mussel species are considered either regionally endemic or are endemic to the state of Texas (Burlakova et al. 2011). These endemic species are a critical component of the diversity and uniqueness of unionid communities in the state (Burlakova et al., 2011a). Additionally, Burlakova et al. (2011) identified 30 rare and very rare species indicating that 65% of Texas freshwater mussels are potentially imperiled.

The Guadalupe – San Antonio River System (GSARS) includes the Guadalupe and San Antonio river basins and contains 13 known species of freshwater mussels (Appendix 1.1). Two of these species, *Quadrula aurea and Quadrula petrina*, are considered endemic and currently listed as state-threatened. *Q. aurea* is found only in the GSARS, while *Q. petrina* is found only in the Guadalupe and Colorado rivers basins (Howells 1996). Both species are also currently candidates to be federally listed. River fragmentation by dams may be one factor limiting their populations. The GSARS contains 375 registered dams total with 215 in the Guadalupe River Basin and 160 in the San Antonio River Basin. These dams fragment the GSARS into over 300 disconnected river networks (Chapter II of this dissertation). This fragmentation can hinder dispersal and migration of aquatic organisms, and limit the amount of available habitat (Watters 1995; Sethi et al. 2004; Tiemann et al. 2007).

The first reports on Texas unionids were published over a century ago (Singley 1893; Strecker 1931), and conservation research intensified in the early to mid-2000s (Howells et al. 2000; Burlakova et al. 2007, 2011, 2011a; Randklev et al. 2010, 2013, 2013a, 2015; Karatayev et al. 2012, 2015; Ford et al. 2009, 2015). Freshwater mussel conservation has gained traction as an important topic in Texas during the last several years. In 2017, a Freshwater Mussel Workgroup was established by the Interagency Task Force on Economic Growth and Endangered Species, managed under the Comptroller's office. As part of the workgroup, stakeholders meet monthly to discuss and determine the best conservations strategies for freshwater mussels that both cooperate with federal regulations and protect the Texas economy. Understanding freshwater mussel distribution and its potential drivers in Texas is essential to aid and promote mussel conservation efforts throughout the state.

The objectives of this research were (1) characterize broad scale spatial patterns of freshwater mussels and their host fish distribution in Texas, (2) to compare broad scale patterns of freshwater mussel distribution and community composition in the Guadalupe – San Antonio River System, and (3) to examine the environmental and anthropogenic controls on freshwater mussel community composition in the (GSARS). I used spatial analyses to determine and compare the distribution of freshwater mussel and fish species in Texas and the GSARS. Then I used multivariate statistics to analyze the environmental and anthropogenic controls on the distribution of freshwater mussel species in the GSARS.

3.1.1. Background

Freshwater mussels are a globally threatened fauna (Haag 2012). In North America, the 'rainforest' of mussel diversity (Haag 2012), approximately 72% of all species are either imperiled or critically imperiled (Thorp and Covich 2010). Freshwater mussels are important components of aquatic ecosystems. They increase water clarity and remove nutrients from the water column, re-directing them to the benthos. They provide habitat and enhance the growth of other organisms such as benthic algae and macroinvertebrates (Vaughn et al. 2015; Atkinson and Vaughn 2014). Their limited mobility (Kappes and Haase 2012), complex life cycles (Stayer 2008), and long-life spans make them particularly sensitive to habitat disturbances, fragmentation, and non-native species (Haag 2012). Mussel species declines in riverine ecosystems can lead to altered nutrient dynamics, further impacting community composition and food-web dynamics (Atkinson and Vaughn, 2014). A better understanding of the processes and controls that shape their distribution is necessary for successful conservation of threatened and endangered mussel communities.

The geographic distribution of mussels depends on suitable habitat, available food, predators, and their dispersal ability via host fish (Strayer 2008). Because their larvae are obligate parasites on fish, the movement of their host fish during the parasitic stage is crucial for mussels' large-scale dispersal which can be over 100s of meters to kilometers (Newton et al. 2008). However, there is no consensus on the relative importance of host fish, compared to other factors, for the distribution of mussels, and it may differ regionally.

The importance of historical colonization via host fishes was documented for rivers in southern and Midwestern states in the U.S. (Vaughn 1997). Evidence for the important role of host fishes for the distribution of mussels was shown for rivers of south-western Ontario (Schwalb et al. 2012, 2015), the Ohio River System (Watters 1992), an Alabama River (Haag and Warren 1998), and for rivers in southern Oklahoma (Vaughn and Taylor 2000). However, a study of two Ohio rivers found only a weak correlation existed between mussel and fish richness (Krebs et al. 2010), and a study of the Pine Hills regions of southeast Louisiana found that mussel species richness increased with stream order, even though host fish species richness did not (Daniel and Brown 2013). In addition, the importance of host fish for the distribution of mussels may also depend on the mussels' host fish specificity and lure display (Haag and Warren 1998). A positive correlation between mussel densities and fish densities were detected for non-displaying host specialists, but not for lure displaying host specialists or host generalists (Haag and Warren 1998).

In addition to current distribution of host fishes, historical processes may have played a major role for current distributional patterns of freshwater mussels (Zanatta and Murphy 2008; Jones et al. 2014). For example, freshwater mussel distributions and community composition may reflect post-glacial invasion patterns and historic river connectivity (Graf 2002). They may have also been shaped by limited dispersal between river basins over larger time spans (i.e., 1000s of years; Schwalb et al. 2012). Genetic studies have shown that mussels can be traced back to glacial refugia from which mussels dispersed after the last glacial maximum about 10,000 years ago (e.g., Elderkin et al. 2007, 2008, Zanatta and Murphy 2008, Zanatta and Harris 2013).

While Texas was never glaciated, connectivity between river basins was affected by glacial expansions and retreats further north. Many, if not all of the Texas rivers that currently drain into the Gulf Coast may have been connected during glaciations of northern parts of North America (Conner and Suttkus 1986). The associated sea level drop may have produced a Mississippi River Basin that was more expansive than it is today, incorporating current Gulf Coast drainages. These rivers became isolated again during periods of glacial retreat or interglacial periods (Al-Rabab'ah and Williams 2004) during the Pleistocene epoch. The combination of historical connectivity during wetter and cooler periods and historical extinctions of species, especially in the more arid western regions, may have led to the current specific aquatic fauna in Texas' river basins.

Potential barriers to freshwater mussel dispersal include dams. Dams fragment rivers restricting dispersal and distributions of mussels and their host fish and create unsuitable habitat (Sethi et al. 2004; Tiemann et al. 2007; Watters 1995). Distance from dams can impact mussel community composition by decreasing species richness (Randklev et al. 2015; Troia et al. 2014) and abundance (Randklev et al. 2015). Upstream effects of dams include silt accumulation, increased water depth, stagnation, pollution accumulation, and decreased nutrient availability (Watters 2000; Ellis 1942). Many species are extirpated from impounded regions and/or replaced by species better adapted to lentic environments (Watters 2000). Dam impacts to downstream environments include flow and substrate alterations that cause extirpation and/or mussel population declines (Vaughn et al. 2015; Watters 2000). Shallow waters resulting from decreased flows (Watters 2000), or the fragmentation of habitat (Dycus et al. 2015), can result in the loss of buffering to extreme thermal events resulting in local extirpation. Even when extirpation does not occur, there may be a loss of recruitment through decreased growth rates, a lack of suitable habitat for juveniles, and a loss of thermal cues required for reproductive cycles (Gates et al. 2015; Watters 2000).

Most studies examining dam impacts on freshwater mussel populations focus on the proximity of dams, rather than the fragmentation they cause (Randklev et al. 2016; Troia et al. 2014; Watters 2000). Dams fragment river systems acting as physical barriers to dispersal and migration. Freshwater mussels are particularly vulnerable to the impacts of fragmentation (Haag 2012), which is often cited as a dominate reason for their declines (Strayer 2008; Vaughn 2012; Atkinson et al. 2012; Gailbraith et al. 2015). The length of functionally connected habitat (e.g. connected river segments accessible to migrating aquatic organisms) is potentially an important controlling variable of freshwater mussel distribution and community composition.

3.2. MATERIALS AND METHODS

3.2.1. Data

Fish data was aggregated from three sources. The Multistate Aquatic Resources Information System (MARIS), an online resource that contains compiled fish samples from over 20 state fish and wildlife agencies, provided fish data for 321 sites sampled from 1987 to 2013. This was supplemented with data from 18 sites collected by T. Bonner, sampled between 2008 and 2015, and 9 sites collected by D.F. Ford sampled between 2009 and 2015. The complete fish presence/absence data set included 348 individual sampling sites located on 148 rivers, in 17 river basins (Table 3.1). Maxwell (2012) and the Fishes of Texas Project Database (Hendrickson and Cohen 2015) provided information on the native ranges for endemic fish species at the sub-watershed level. All available fish data were used to address Objective 1, and subsets of the dataset were used to address Objective 2 and 3.

A mussel dataset included sites sampled along 22 rivers in 10 river basins in Texas from 2006 to 2015 (Appendix 3.1). Mussel sites where no mussel species were found were not considered in Objective 1 of this analysis. The final dataset included 98 mussel sites surveyed by L.E. Burlakova and A.Y. Karatayev (partly previously published in Burlakova et al. 2011, and Karatayev et al., 2012, 2015), 125 sites surveyed by D. Ford, and 5 sites surveyed by A.N. Schwalb. All mussel data were collected via timed searches and standardized by search effort (CPUE). The entire mussel dataset contained 228 mussel sites with mussel species present ≥ 1 at each site (Appendix 3.1). All mussel sites (n = 228) were used to address Objective 1, this included information for 41 freshwater mussel species.

A subset of the data located within the GSARS was used to address Objective 2. This included 22 mussel sites with species present, 10 mussel sites with no species present, and 39 fish sites. *Quadrula aurea* and *Quadrula* *houstonensis*, are currently state threatened species and candidates to be federally listed by their *Quadrula* names.

The Stream-Catchment (StreamCat) dataset provided anthropogenic and environmental variables at the landscape scale for mussel sites in the GSARS (Hill et al. 2016). StreamCat is and extensive collection of landscape metrics for over 2 million stream segments in the United States, available at watershed, catchment, and riparian buffer scales, that were made publicly available by the Environmental Protection Agency (EPA) (Hill et al. 2016). For this analysis, the 100-meter riparian buffer scale was used, as previous research has indicated that land cover variables at this scale best explain shifts in freshwater mussel community composition (Atkinson et al. 2012).

StreamCat variables were joined with mussel sites by first matching COMIDs to the National Hydrography Dataset Plus Version 2 Medium Resolution (NHDPlusV2). The NHDPlusV2 high resolution (1:12,000 and 1:24,000 scale) shapefile was developed through a joint effort by the United States Geological Society (USGS) and the EPA (McKay et al. 2012). The data set was created using a combination of USGS hydrologic digital line graph files for spatial accuracy and EPA reach files for attribute information, version 3.0 (McKay et al. 2012). Flowlines representing irrigation and/or drainage canals were removed from the NHDPlusV2 dataset before processing.

Each stream segment in the NHDPlusV2 has an assigned COMID. To determine the corresponding COMID for each mussel site, sites were snapped to the NHDV2Plus flowlines in a GIS. The StreamCat database provided 39 land cover and land use variables. Due to the small number of mussel sites available for analysis in the GSARS, correlations provided a means of reducing the number of land cover and land use classes. Variables that were related, for example land cover categories, and highly correlated (≥0.7) were eliminated or combined where possible. This created a final total of nine StreamCat variables. These metrics included watershed area, mean runoff, base flow index, nitrate ion deposition, percent of impervious surface area (ISA), percent of undeveloped land cover, percent of agricultural land cover, density of georeferenced dams, and the density of national pollutant discharge elimination systems (NPDES) (Table 3.1).

The Vogel extension package for the NHDPlusV2 provided discharge data in the form of mean annual flow estimates based on a regression technique developed by Vogel et al. 1999. COMIDs allowed the data in the Vogel extension to be linked to mussel sites. Maxwell (2012) provided fish data at the watershed scale (HUC 8), and this data was refined by potential host fish (Ford and Oliver 2015). The Barrier Assessment Tool (BAT) calculated the upstream FRN length for each mussel site. This was done by first mapping the FRNs in the GSARS, and then associating each mussel site with an FRN and determining the upstream FRN length from that site. For a complete description of the BAT and methods used to extract FRNs see Chapter II of this thesis. A complete list of the 12 variables used for analysis in the GSARS are listed in Table 3.2.

3.2.2. Analyses

Current distribution of mussel fauna across Texas basins constitute separate biogeographical provinces. Neck (1982) divides Texas into four biogeographical provinces: (1) northern Texas including the Canadian River

Basin, which was not considered in this study and has no known endemic mussel species (Haag, 2012); (2) East Texas, which includes the Sabine, Neches, Trinity, and San Jacinto river basins; (3) Central Texas, which includes the Brazos, Guadalupe, San Antonio, and Nueces river basins; and (4) West Texas, the Rio Grande River Basin (Figure 3.1).

To compare patterns of species richness and endemism for mussels and fishes across biogeographical provinces in Texas (i.e., east, central, west Texas) and river basins (Objective 1), a series of maps were created using ArcGIS Version 10.3.1 (ESRI, 2015) and a series of cluster analyses were conducted. The maps served to visualize the distributions of all fish and mussel species as well as the distribution of the endemic mussels and fishes across different river basins and regions in Texas. A second series of cluster analyses evaluates these patterns at the basin scale for the GSARS.

The optimized hot spot analysis tool statistically analyzed patterns of fish and mussel species richness by calculating a Getis-Ord Gi* statistic. The Getis-Ord Gi* statistic identified whether sites with significantly low or high values clustered together over the study area by evaluating the value of each site in relation to the values of neighboring sites (ESRI 2014a, 2016). The resulting statistic was returned as a Z score for each site. High, positive Z scores represented clusters of sites with statistically significant larger values, considered hot spots, and low, negative Z scores represented clusters of sites with statistically significant smaller values, considered cold spots (Table 3.3). For this research, hot spot analyses distinguished between statistically significant areas of high and low numbers of fish species, mussel species, endemic fish species, endemic mussel species, and the percentage of endemic mussel and fish species. Optimized hot spot analysis for endemic fish species utilized historic fish species ranges and were analyzed using sub-watersheds as opposed to individual sites, as data were only available in an aggregated format.

To satisfy sample size requirements ($n \geq 30$), the cluster analysis at the basin scale included sample sites with no mussel species present ($n = 10$). This resulted in a total sample size of 32 mussel sites in the GSARS. Because there are only eight HUC 8 sub-watersheds in the GSARS, patterns of fish species endemism were only analyzed at the state scale.

The alpha (α) and beta (β) diversity (Equation 3.1) of each mussel site, and species abundance, relative abundance, frequency, and relative frequency for each species provided additional information on the distribution and community composition of freshwater mussels in the GSARS. Alpha diversity is defined here as the total number of species at each site, while beta diversity is defined as a measure of compositional heterogeneity or the ratio of regional to local diversity, see Equation 1 (McCune and Grace 2002; Whittaker 1960). Species abundance is defined as the number of individuals per species, and relative abundance is defined as the percentage of individuals of a species at a site compared to the total number of individuals at a site, see Equation 3.2 (McCune and Grace 2002). Species frequency is defined as the portion or percentage of sites at which a species occurs, and relative frequency is defined as the portion of sites at which a species occurs compared to the frequency of all species, see Equation 3.3 (McCune and Grace 2002).

Equation 3.1.

$\beta = \gamma/\alpha - 1$

Where γ is gamma (the number of species at all sites) and α is the number of species at a site.

Equation 3.2.

Relative Abundance $_j$ % = (100 · number of individuals $_j$)/Total number of

individuals

Equation 3.3.

Relative frequency $_j$ % = (100 · frequency $_j$)/sum of frequencies

Where *j* is a particular species, and frequency is the proportion of sites at which a species occurs.

I used non-metric multidimensional scaling (NMS) ordination to determine the environmental and anthropogenic controls on mussel species distribution and community composition in the GSARS. Ordination graphically summarizes complex relationships by extracting dominant patterns (McCune and Grace 2002). Ordination techniques assume that species abundance varies with changes in environmental variables, and are commonly used in ecology to describe the strongest patterns of species composition (McCune and Grace 2002).

NMS is an increasingly utilized technique in community ecology, and was considered the most generally effective ordination method for ecological community data by McCune and Grace (2002). It is suited to data that is non-normal or is of questionable scale, and does not require linear relationships among variables (McCune and Grace 2002). NMS is a method of choice to assess the dimensionality of the dataset due to its flexibility and generality (Clarke 1993; McCune and Grace 2002).

I used PC-Ord software to perform the NMS (Mather 1976, Kruskal 1964) using Relative Sørenson as the distance measure and a random starting configuration for species abundance data at each mussel site. Relative Sørenson standardized distances between sites by the total number of sites, and shifted the emphasis of analysis to proportions of species at a site, as opposed to absolute abundance (McCune and Grace 2002). PC-Ord selected the dimensionality of the final solution by comparing the stress for the best solution for each dimensionality (McCune and Grace 2002). Additional dimensions were added if they reduce the final stress by 5 or more (0-100 scale), and the final stress had to be lower than $p \leq 0.05$ using a Monte Carlo test to be accepted as a solution (McCune and Grace 2002). For further detail on NMS procedures see McCune and Grace (2002), alternatively see Mather (1976) and Kruskal (1964).

Multi-response permutation procedures (MRPP), using Relative Sørenson as the distance measure, determined if the differences between sites located on different rivers were significant. MRPP is a non-parametric multivariate test of differences between groups, in this case rivers. MRPP first calculated the average distance and the weighted mean within group distance or delta (δ). Then determined the probability of achieving a value as small or smaller than the observed δ (McCune and Grace 2002).

3.3. RESULTS

3.3.1. Patterns of species richness and endemism

At the state level, the number of fish species at a site was generally higher compared to the number of mussel species (Figure 3.2a, b). General species richness of both mussels and fish species showed a decreasing trend from east to west Texas (Figure 3.2a, b, Table 3.3), as did mussel abundance (Figure 3.2c). The results of the hot spot analysis indicated significantly higher mussel species richness at sites in multiple rivers of the Neches River Basin, and significantly lower mussel species richness occurred in the Rio Grande River Basin along the Rio Grande River (Figure 3.2e, Table 3.3). Similarly, most of the areas with significantly higher species richness for fish occurred in East Texas, and areas of lower richness occurred at sites in the lower portion of the Colorado River Basin (Figure 3.2d). Species richness within river basins for both mussels and fish tended to be higher in the lower downstream reaches of river basins compared to upstream reaches. For fish species, this was most pronounced in the Brazos River, and somewhat in the Rio Grande and Guadalupe rivers (Figure 3.2a). For mussel species, this pattern occurred in the Sabine River and to a lesser degree in the Guadalupe River (Figure 3.2b), but was more obvious when sites with no mussels were included (data not shown).

The number of endemic mussel species also decreased from east to west (Figure 3.3c), yet the percentage of endemic mussel species decreased from west to east (Figure 3.3d). The number of endemic fish species increased from east to west (similar to the percentage of endemic mussel species, Figure 3.3a, d), observing the percentage of endemic fish species present in a watershed further heightened this pattern (Figure 3.3b). Hot spots of endemic species richness for mussels occurred almost exclusively in the Neches River Basin (Figure 3.3g, Table 3.3), which corresponds to the greater species richness in this river basin (Figure 3.2b, Table 3.3). Hot spots of endemic fish richness, however, occurred in Central and Western Texas river basins (Table 3.3), whereas significant clustering for low presence of endemic fish species was present in sub-watersheds of the Neches River Basin, upper portions of the Colorado and Blanco river basins, and sub-watersheds along the Gulf Coast (Figure 3.3e).

Percentages of endemic mussel species were significantly higher at sites on the Rio Grande River and Central Texas rivers, including the Llano and the Guadalupe, in the Colorado and Guadalupe river basins, respectively (Figure 3.3h, Table 3.3). Significant clusters of higher percentages of endemic fish were found almost exclusively in sub-watersheds within the upper Rio Grande River Basin, and significant clusters of lower percentages of endemic fish were present in sub-watersheds along the Gulf Coast (Figure 3.3f). Spatial patterns of endemism for mussels and fish were similar for percentages of endemic species, but not for endemic species richness across Texas.

At the basin scale, the optimized hot spot analysis revealed hot spots of mussel species richness in the lower reaches of the San Marcos River and in middle reaches of the Guadalupe River (Figure 3.4a). These hot spots corresponded to warm spots of fish species richness in these same areas (Figure 3.4b). Cold spots of fish species richness were located along multiple rivers, including the Cibolo Creek and San Antonio River, both in the San Antonio River

Basin. There were also warm spots of fish species richness in the lower reaches of the Guadalupe and San Antonio rivers (Figure 3.4b). The hot spot analysis for endemic mussel species richness exhibited a similar pattern to fish species richness, with hotspots in the lower reaches of the Guadalupe and San Antonio rivers, and cold spots in Cibolo Creek and middle reaches of the San Antonio River (Figure 3.4c). Locations of hot spots for percentage of endemic mussel species were the same at the state and basin scale (Figure 3.4d).

3.3.2. Patterns of species richness and endemism

There is a general increase in species abundance and richness in lower reaches of the GSARS, and most sites with no species were in the upper and middle portions of the GSARS (Figure 3.5). Species sites with higher alpha diversity and lower beta diversity occurred in the Guadalupe River Basin, particularly in the mid and lower reaches of the Guadalupe River and the lower reaches of the San Marcos River (Figure 3.5 b, Table 3.4). Similarly, the majority of sites with endemic/state threatened mussel species occur in the Guadalupe River Basin (Table 3.4). Most mussel sites in the GSARS had upstream FRNs of over a thousand km in length (Table 3.4). Two sites in the Guadalupe River Basin did have less than 1.5 km of upstream FRN length, and these sites had relatively high alpha diversity and low beta diversity (Table 3.4).

Q. aurea is the most abundant species in sites in reaches of the mid-Guadalupe River, while *Amblema plicata* is the most abundant in sites located in the lower Guadalupe River (Figure 3.5b). Sites located on the San Marcos River and middle reaches of the Guadalupe River are dominated by *A. plicata, Megalonaias nervosa, and Q. aurea* (Figure 3.5b). The only site where *Q. petrina* is the most abundant occurred on the San Marcos River (Figure 3.5b). In the San Antonio River Basin *Lampsilis teres* is the dominate species present at all sites, except where *Q. aurea* is also present (Figure 3.5b). *Quadrula verrucosa* occurs only in sites located on the San Marcos River and one site located in the middle course of the Guadalupe River.

A. plicata, L. teres, and *Q. aurea* were the most abundant and frequently occurring species in the GSARS (Table 3.5). *Arcidens confragosus* and *Uniomerus delivis* were the least abundant species, followed by *Pyganodon grandis* and *Quadrula apliculate* (Table 3.5). These four species also occurred with the least frequency, and only identified at one site (Table 3.5). *P. grandis* and *Q. apliculate* occurred at sites in the middle course of the Guadalupe River, and *A. confragosus* and *U. delcivis* at sites in the lower Guadalupe River (Figure 3.5b).

3.3.3. Multivariate Analysis

The NMS ordination (Figure 3.6) explained 96.7 % of the variation in the dataset, with 41.4% of the variation loaded on axis-1, 46.2% on axis-2, and 9.1% on axis-3 (Table 3.6). The final result was significant at the 0.05 level (p = 0.0480), using 200 runs of real data and 249 runs of randomized data. The final stress of the solution was 4.088, considered to be between good and excellent using Kruskal's rule of thumb (McCune and Grace 2002), and instability was below 0.00001.

Axis-1 demonstrated an inverse relationship between base flow index and mean annual discharge, with *Lampsilis hydiana, Q. aurea, M. nervosa, P. grandis,* and *Q. verrucosa* associated with higher discharge and lower base flow indexes (Figure 3.6a). Percent impervious surface area (ISA), base flow index (BI), and density of national pollutant discharge elimination systems (NPDES) were significantly positively correlated with the first axes, as was *L. teres* (Tables 4.7, 4.8). Runoff and percent undeveloped land cover, as was *Q. aurea* were significantly negatively correlated with axis-1 (Table 3.7, 3.8).

Axis-2 established a slight inverse relationship between the amount of ISA in a watershed and the variables of percent agricultural and undeveloped land cover, watershed area, and nitrate concentrations (Figure 3.6a). *L. teres* was associated with higher percentages of ISA, while *P. grandis, Q. apiculate, Cyrtonaias tampicoensis, A. plicata, A. confragosus, and U. delivis* were associated with higher percentages of agricultural and undeveloped land cover, as well as larger watersheds and higher concentrations of nitrate (Figure 3.6a). Mean annual discharge (MAQ), mean elevation, runoff, nitrate concentration, density of NPDES, and upstream FRN length were significantly positively correlated with the second axis (Table 3.7, 4.8). *A. plicata, C tampicoensis,* and *U. devlivis* were also significantly positively correlated with axis-2 *(Tabel 4.7, 3.8).* Percent ISA land cover and density of dams (NABD) were significantly negatively correlated with axis-2, as was *Q. petrina* (Table 3.7, 4.8).

Axis-3 showed an inverse relationship between number of host fish species and length of upstream FRN (Figure 3.6 b,c). *P. grandis* was most strongly associated with higher numbers of host fish, while *A. confragosus* was most strongly associated with smaller upstream FRNs (Figure 3.6 b, c). Watershed area and length of upstream FRN were significantly positively correlated with axis-3, as were *A. confragosus,* and *U. declivis* (Table 3.7, 4.8). Number of host fish species, and density of dams, as well as *L. hydiana, M. nervosa, Plectomerus dombeyanus, and P. grandis, and Q. verrucosa* were significantly negatively correlated with axis-3 (Table 3.7, 4.8).

The 22 mussel sites with species present in the GSARS were located on four rivers: the San Marcos River (SMR), Guadalupe River (GR), Cibolo Creek (CC), and San Antonio River (SAR). Sites on the same river exhibited more similar mussel community composition and differed between rivers (Figure 3.6). MRPP results indicated significant differences between the community composition of sites grouped by river (p = 0.0009, A = 0.24). Additionally, community composition appeared to vary less between sites in the same river basin, and more between sites in different river basins (Figure 3.6).

3.4. DISCUSSION

3.4.1. Patterns across Texas

Both mussel and fish species show similar large-scale patterns with an east to west gradient of decreasing species richness and abundance. Hot spots of mussel species richness occur almost exclusively within the Neches River Basin. This supports previous studies that have documented the Neches River Basin being the core of unionid species richness in Texas (Burlakova et al. 2011; Ford et al. 2014).

Though mussel distributions have not always been found to correlate to host fish availability (Rashleigh 2008; Krebs et al. 2010), there is evidence that host fishes are an important determinant of mussel community variability

(Schwalb et al. 2012; Watters 1992; Haag and Warren 1998; Vaughn and Taylor 2000). In the GSARS at the basin scale, increased fish species richness corresponded to both increased mussel species richness and endemic species richness. These corresponding patterns potentially indicate the importance of host fish to mussel species richness that has been documented in other studies (Vaughn 1997; Schwalb et al. 2012, 2015; Haag and Warren 1998; Vaughn and Taylor 2000). A lack of fish data makes it difficult to determine how patterns of fish species richness relate to the percentage of endemic species at sites in the Upper Guadalupe River, and deserves further study.

Similar to patterns of mussel distribution in the Mississippi Region, species richness in Texas is likely best explained by an east-west climate gradient (i.e., decreased rainfall), hydrologic/hydraulic disturbance intensity and frequency (i.e., flashiness of rivers), and distance from source population (Haag 2012). Texas has a pronounced climate gradient from the humid east to the arid west, with river basins in the east generally supporting larger numbers of mussel species. River basins in eastern Texas generally do not experience the severe dewatering events that characterize the river basins in central and west Texas, and tend to be less prone to flash flooding. This may allow more species to persist, where they would otherwise be washed away or stranded during extreme events, contributing to the generally greater richness of both mussel and fish species in these river basins.

The higher mussel species richness in river basins of East Texas may additionally reflect historical colonization processes via proximity to the Mississippi River Basin. The Mississippi River Basin encompasses rivers with the highest mussel richness in the United States (Haag, 2012), and aside from the endemic mussel species, all of the mussel species in East Texas river basins co-occur in the Mississippi Embayment Province (Haag 2012). During the Pleistocene, the Mississippi River Basin's expanse advanced and receded in accordance with the expansion and reduction of glacial ice (Conner and Suttkus, 1986; Al-Rabab'ah and Williams, 2004). As the expanse of the Mississippi River Basin shrank, it may have become separated from Gulf Coast Drainages in an eastward pattern. The Rio Grande River Basin would have been, potentially, the first to be disconnected from the Mississippi River Basin, followed by the separation of Central Texas river basins, and lastly those of East Texas. Thus, the faunal group similarity of the Mississippi and East Texas river basins, may reflect longer durations of connectivity between these river basins. This pattern may also contribute to the higher percentages of endemic mussel species exhibited in river basins of West and Central Texas. Historical processes should also be important for fish distribution as most fish species are restricted to their individual river basins. A likely confounding factor, if one attempts to explain differences in fish communities with historical processes, is the human introduction of a variety of fish species throughout Texas, including channel catfish and largemouth bass, many of which are often host fish to various mussel species (Ford and Oliver, 2015).

More recently, increased urbanization and human population growth likely led to declines of mussel populations and shifts in mussel community composition (Burlakova et al. 2011). Increased urbanization and population growth can result in decreased mussel species richness and abundance through increased pollution (Gillis 2012, Gillis et al. 2014) and increased magnitude and frequencies of high flows (Brown et al. 2010). The lower mussel species richness found in the Sabine and Trinity river basins, in relation to the Neches River Basin, may be due in part to human impacts affecting mussel distribution and abundances. A recent study has shown that mussel species richness and abundances decreased downstream of impoundments in the Sabine River, and community composition experienced shifts towards species with more opportunistic life history strategies (Randklev et al., 2015). The lower species richness in the Trinity River Basin is potentially explained by human use and modifications of rivers in the Trinity River Basin, as it encompasses several large population centres, including the Dallas-Fort Worth metroplex, and contains nearly a quarter of the dams (n = 1,787) in Texas (TCEQ 2014).

The higher percentage of endemism in West Texas, i.e. the Rio Grande River Basin, may be reflective of its proximity to different faunal systems i.e., the Panuc-Tamesi system as opposed to the Mississippian system (Neck 1982). It may also reflect larger periods of separation from the Mississippi River Basin during the Pleistocene, compared to river basins of East Texas (see above). These potentially longer and more frequent separations from the Mississippi River Basin may have acted as periods of genetic isolation and divergence that resulted in increased speciation. Additionally, the lower precipitation and higher temperatures of the Rio Grande River Basin may have resulted in increased isolation during droughts, potentially explaining the increased speciation and thus endemism (Hewitt, 2000; Davis and Shaw, 2001; Al-Rabab'ah and Williams, 2004).

The contribution of groundwater to stream flow in river basins of central and west Texas may also contribute to the higher rates of endemism seen in these river basins. Rivers with larger groundwater contributions to base flow in the Rio Grande River Basin and river basins of Central Texas may act as refuges during droughts, potentially explaining the larger percentages of endemic mussels that occur in these areas. The Edwards Plateau of Central Texas, an area that encompasses portions of the Colorado and Guadalupe river basins, in particular, is known as an area with many unique and endemic aquatic biota that are mostly found to inhabit subterranean systems, springs, or spring fed streams (Bowles and Arsuffi, 1993).

An increase of mussel species richness with stream size was already recognized at the beginning of the century, and applies especially to the Mississippian region, extending from the Great Lakes to the Gulf (Haag 2012). More recently, higher species turnover (Atkinson 2012) and increased species abundance (Cao et al. 2015) were found in larger streams farther from the headwaters. There was some indication at the state scale that within river basins both mussel and fish species varied along an upstream downstream gradient, with increased species richness downstream. For example, a vast majority of fish species hot spots were found in lower portions of Texas river basins (Figures 3.3 e,f), and this pattern was most obvious for mussels in the Sabine River.

3.4.2. Patterns in the GSARS

In the GSARS, there was a general increase in alpha and beta mussel species diversity in the downstream direction. Additional mussel data may further highlight this pattern, which is potentially the result of unidirectional flow and mussels' passive dispersal (Haag 2012), potentially lower rates of juvenile predations (Daniel and Brown 2013), and more stable (Haag and Warren 1998) and heterogeneous (Atkinson et al. 2012) downstream habitat. A notable exception was the relatively high mussel species diversity at sites G2 and G4 on the mid-course of the Guadalupe River. These sites occur near the confluence of the Guadalupe River and one of its major tributaries, the San Marcos River, and may reflect the importance of confluences as sources of habitat and biodiversity (Benda et al. 2004).

Increases in species richness associated with stream sizes (i.e. stream order) are accompanied by patterns of assemblage succession (Haag 2012). Such an assemblage shift is evident in the Guadalupe River Basin. *Q. aurea* is prevalent and predominante in mid-reaches of the Guadalupe River and reaches of the San Marcos River, but in lower reaches of the Guadalupe River it is *L. hydiana* that becomes the predominant mussel species. Similarly, *M. nervosa* is prevalent in the mid-reaches of the Guadalupe River and reaches of the San Marcos River, but disappears from assemblages in lower reaches of the Guadalupe River.

Evidence presented in this study demonstrates differences in mussel community composition by river; this may reflect differences in flow regimes, land use, and/or other landscape differences. *L. hydiana*, ubiquitous in the San Antonio River Basin, was associated with larger percentages of ISA. This may indicate this species' resilience or ability to adapt to increased urbanization, and the associated increased pollution (Gillis 2012, Gillis et al. 2014), and flashiness (Brown et al. 2010) that generally result in declines of other mussel species.

The higher species abundance and diversity in the Guadalupe River Basin is associated with non-ISA land cover categories and higher mean annual discharge. Land cover, such as wetlands, can influence community composition, because they moderate the effects of flow variability, and species that require more stable flows have been associated with catchments that have larger areas of wetlands (Atkinson et al. 2012). However, there is also an association of higher nitrate concentrations in the watershed of the Guadalupe River, that seems counterintuitive, but may again reflect the positive effects of less urban land cover in these watersheds.

There was a negative relationship between number of host fish and length of FRN, with the majority of mussel species more strongly associated with larger numbers of host fish species and smaller FRNs. This may reflect the actual amount of interaction between mussels and host fish. Smaller FRNs may provide more opportunities for contact with host fish and thus increased mussel propagation. It should be noted, that only registered dams were considered when calculating FRN length, and that host fish data was calculated at the HUC 8 watershed scale. Thus, both upstream FRN length and the number of host fish species present may be exaggerated, and this data may not necessarily reflect the actual number of host fish species within a particular FRN. A more complete dataset of instream barriers and linked fish and mussel sites within the same FRN may provide better insight to the relationship between host fish, FRN length, and mussel species abundance and diversity.

3.5. CONCLUSION

In Texas, 14 of the 15 state listed threatened mussels are considered either locally or regionally endemic, and several more species considered rare are critical components of mussel community uniqueness (Burlakova et al., 2011a). It often makes sense (e.g., due to budget constraints and other management challenges) to prioritize areas for protection to preserve these endemic and rare species. This study identified the Neches River Basin, and locations within the Colorado, Guadalupe, and Rio Grande river basins as areas of significantly higher mussel species richness and/or higher endemism (see Figures 2 and 3), and supports the conservation priorities previously set forth (Burlakova et al. 2011, 2011a, Karatayev et al. 2012, 2015). There are currently only two river segments listed as National Wild and Scenic River Systems in Texas, both along the Rio Grande River, comprising a total of 308 km (less than 200 miles) of Wild and Scenic River in Texas (WSR, n.d.). The Neches River, with its segments of un-impacted areas and the richness of its mussel fauna, especially endemic mussel species, has been recommended to be designated a National Wild and Scenic River System, and this study would support such an action.

Host fish are crucial for the reproduction of mussels, and mussel surveying and sampling should include or be coordinated with fish sampling. Additionally, the relationship between length of FRN, host fish, and mussel species diversity and abundance should be further explored with larger and more complete datasets. As more data on mussel distributions and community composition are obtained and become available, conservation and management actions can be better tailored to specific mussel species, communities and populations in Texas.

3.6. TABLES AND FIGURES

Table 3.1. The number of sampling sites for fish and mussel data and the number of rivers per river basins.

	Fish		Mussels	
River Basin	# Sites	# Rivers	# Sites	# Rivers
Brazos	96	36	13	5
Canadian	1	1	0	0
Colorado	126	16	10	2
Cypress	6	5	1	1
Guadalupe	14	6	16	2
Lavaca	3	2	0	0
Neches	18	12	71	4
Nueces	6	3	1	1
Red	13	10	0	0
Rio Grande	26	6	33	3
Sabine	20	7	62	1
San Antonio	33	7	6	2
San Jacinto	10	5	0	0
Sulphur	3	3	0	0
Trinity	58	22	15	1
Coastal	14	7	0	0
Total	447	148	228	22

*Coastal river basins refer to the following river basins Arenosa Creek, Oyster Creek, San Bernard River, San Fernando Creek, Taylor Bayou, and Tres Palacios Creek. The Arkansas River Basin and Cedar Bayou Basin had no fish or mussel sites located within them.

Table 3.2. Explanatory variables used in non-metric multidimensional scaling.

Variable	Description	Source
Tsp_Host	Total potential host fish present at the HUC 8 scale	(Maxwell 2012; Hendrickson and Cohen 2015)
MAQ	Mean Annual Flow at bottom of flowline as computed by Vogel Method (cfs). Standardize by watershed area.	(Vogel et al. 1999)
RunoffWs	Mean runoff (mm) within watershed. Standardized by watershed area.	StreamCat (Hill et al. 2016)
NO3	Annual gradient map of precipitation-weighted mean deposition for nitrate ion concentration wet deposition for 2008 in kg of NO3/ha/yr, within watershed.	StreamCat (Hill et al. 2016)
Area	Watershed area (square km) at NHDPlus stream segment outlet, i.e., at the most downstream location of the vector line segment.	StreamCat (Hill et al. 2016)
ISA	Percent of watershed area classified as developed, open space (<20% ISA), low - (20% - 49% ISA), medium - (50% - 70% ISA), and high - intensity (80% - 100% ISA) land use (NLCD 2011 class 21, 22, 23, and 24).	StreamCat (Hill et al. 2016)
BI	Base flow is the component of streamflow that can be attributed to ground-water discharge into streams. The BFI is the ratio of base flow to total flow, expressed as a percentage, within watershed.	StreamCat (Hill et al. 2016)
Undeveloped	Percent of watershed area classified as shrub/scrub, grassland, forest, and wetland land cover (NCLD 2011 class 52, 71, 41,42,43, 95, 90)	StreamCat (Hill et al. 2016)
Ag	Percent of watershed area classified as crop and pasture/hay land use (NLCD 2006 class 81, 82)	StreamCat (Hill et al. 2016)
NABD	Density of georeferenced dams within watershed (dams/ square km)	StreamCat (Hill et al. 2016)
NPDES	Density of permitted NPDES (National Pollutant Discharge Elimination System) sites within catchment and within a 100-m buffer of NHD stream lines (sites/square km)	StreamCat (Hill et al. 2016)
FRN	Upstream functional river network length in kilometers.	Calculated/BAT

Table 3.3. Z scores and corresponding *P* values for the optimized hotspot analysis.

GiZ-score (Standard Deviation)	Gi*P*-value (Probability)	Confidence Level
< -1.65 or > +1.65	< 0.10	90%
< -1.96 or > +1.96	< 0.05	95%
< -2.58 or > +2.58	< 0.01	99%

Table 3.4. Alpha and beta diversity, endemic species richness and percent, and ID and length of upstream functional river network for mussel sites in the Guadalupe San Antonio River System. Sites are roughly in the order they occur, going from up-to downstream for each river.

*Site		Diversity		Abundance	Endemic Species		FRN
		Alpha	Beta		N	%	(km)
SMR-8	a	3.00	1.00	8.30	2.00	66.67	615.56
SMR-6	b	3.00	3.33	5.71	0.00	0.00	1237.86
SMR-2	b	5.00	4.20	7.33	1.00	20.00	1283.89
SMR-5	b	6.00	2.67	72.00	1.00	16.67	1442.75
SMR-7	b	4.00	1.75	1.66	1.00	25.00	1283.89
GR-8	c	1.00	31.00	0.20	1.00	100.00	30.00
GR-9	c	1.00	30.00	0.40	1.00	100.00	372.92
GR-2	d	8.00	5.50	29.71	0.00	0.00	1.12
GR-4	b	5.00	7.20	5.33	1.00	20.00	0.58
GR-10	e	5.00	14.40	12.14	2.00	40.00	3079.29
GR-11	f	3.00	22.33	11.50	1.00	33.33	2923.14
GR-13	f	2.00	32.00	6.40	0.00	0.00	3031.69
GR-14	f	2.00	31.00	11.55	1.00	50.00	3617.26
GR-15	f	5.00	12.00	126.66	1.00	20.00	3707.22
GR-16	f	5.00	11.00	36.89	1.00	20.00	3722.56
GR-17	f	6.00	8.33	9.26	1.00	16.67	3752.38
CC-1	f	1.00	72.00	1.33	0.00	0.00	1070.50
CC-2	f	2.00	37.00	1.55	0.00	0.00	1380.26
SAR-2	f	1.00	29.00	0.33	0.00	0.00	2145.12
SAR-3	f	2.00	14.00	10.22	0.00	0.00	2635.56
SAR-4	f	2.00	13.00	92.66	1.00	50.00	4324.00
SAR-5	f	3.00	8.00	41.82	1.00	33.33	3843.21

*Sites with the same superscript letter indicate location on same functional river network.

Table 3.5. Abundance and frequency measures for the freshwater mussel species at 22 sites in the Guadalupe – San Antonio River System.

Species	Total Abundance	Relative Abundance	Frequency	Relative Frequency
A. confragosus	0.04	*0.008*	4.55	1.33
A. plicata	216.36	*40.91*	68.18	19.99
C. tampicoensis	14.90	*2.82*	31.82	9.33
L. hydiana	5.29	*1.00*	13.64	4.00
L. teres	123.75	*23.40*	72.73	21.33
M. Nervosa	27.87	*5.27*	31.82	9.33
P. dombeyanus	0.67	*0.13*	4.55	1.33
P. grandis	1.14	*0.22*	4.55	1.33
Q. apiculate	1.81	*0.34*	9.09	2.67
Q. aurea	122.03	*23.08*	68.18	19.99
Q. petrina	5.31	*1.00*	9.09	2.67
Q. verrucosa	9.24	*1.75*	18.18	5.33
U. declivis	0.42	*0.08*	4.55	1.33

Note: Data is standardized by person hour.

Table 3.6. Coefficients of determination for the correlations between ordination distances and distances in the original n-dimensional space.

	R-squared	
Axis	Increment	Cumulative
1	0.414	0.414
2	0.462	0.876
3	0.091	0.967

Table 3.7. Pearson and Kendall correlations with ordination axes for explanatory variables.

Variables	Axis											
	1				2				3			
	r	R-squared	τ		r	R-squared	τ		r	R-squared	τ	
Tsp_Host	0.085	0.007	0.132		0.203	0.041	-0.151		-0.692	0.479	-0.715	**
MAQ	-0.226	0.051	-0.048		0.621	0.386	0.441	**	0.194	0.038	0.197	
ELV	-0.218	0.048	-0.022		0.249	0.062	0.223	*	0.138	0.019	0.170	
Runoff	-0.754	0.568	-0.372	**	0.393	0.155	0.240	*	-0.128	0.016	-0.022	
NO3	-0.289	0.083	-0.188		0.688	0.474	0.389	**	0.066	0.004	0.197	
Area	-0.081	0.007	0.031		0.546	0.298	0.45		0.263	0.069	0.258	*
ISA	0.749	0.561	0.407	**	-0.376	0.141	-0.328	**	-0.058	0.003	-0.17	
BI	0.589	0.347	0.346	**	0.120	0.014	-0.092		-0.092	0.008	-0.144	
Und	-0.540	0.291	-0.407	**	-0.158	0.025	-0.022		-0.02	0.0001	0.118	
Ag	0.19	0.036	0.144		0.481	0.232	0.092		-0.181	0.033	-0.17	
NABD	0.059	0.003	-0.083		-0.334	0.112	-0.258	*	-0.236	0.056	-0.205	*
NPDES	0.382	0.146	0.346	**	0.227	0.052	0.240	*	0.006	0.0001	0.100	
FRN	0.162	0.026	0.139		0.141	0.020	0.252	*	0.546	0.298	0.330	**

* denotes significance at the 0.1 level.

** denotes significance at the 0.5 level.

Table 3.8 Pearson and Kendall correlations with ordination axes for species.

Variables	Axis 1				Axis 2				Axis 3			
	r	R-squared	τ		r	R-squared	τ		r	R-squared	τ	
A. confragosus	-0.067	0.005	0.072		0.294	0.086	0.244		0.192	0.037	0.245	*
A. plicata	-0.197	0.039	-0.191		0.359	0.129	0.509	**	0.196	0.038	0.0001	
C. tampicoensis	-0.152	0.023	-0.012		0.307	0.094	0.317	*	0.164	0.027	0.211	
L. hydiana	-0.116	0.013	0.068		-0.125	0.016	0.017		-0.282	0.08	-0.357	**
L. teres	0.413	0.171	0.438	**	-0.128	0.016	-0.077		0.015	0.0001	-0.212	
M. nervosa	-0.175	0.03	-0.106		-0.076	0.006	-0.023		-0.643	0.413	-0.481	**
P. dombeyanus	-0.072	0.005	0.014		-0.006	0.0001	0.043		-0.333	0.111	-0.244	*
P. grandis	-0.022	0.0001	0.129		0.152	0.023	0.072		-0.547	0.299	-0.302	**
Q. apiculate	-0.09	0.008	0.01		0.247	0.061	0.175		-0.398	0.159	-0.113	
Q. aurea	-0.27	0.073	-0.452	**	-0.223	0.05	-0.196		0.003	0.0001	-0.014	
Q. petrina	-0.37	0.137	-0.196		-0.236	0.056	-0.298	**	0.006	0.0001	-0.092	
Q. verrucosa	-0.129	0.017	-0.128		0.0001	0.0001	-0.037		-0.604	0.365	-0.517	**
U. declivis	-0.067	0.005	0.072		0.294	0.086	0.244	*	0.192	0.037	0.215	*

* denotes significance at the 0.1 level.

** denotes significance at the 0.5 level.

Figure 3.1. Map of Texas' river basins and biogeographic provinces (amended from Neck, 1982).

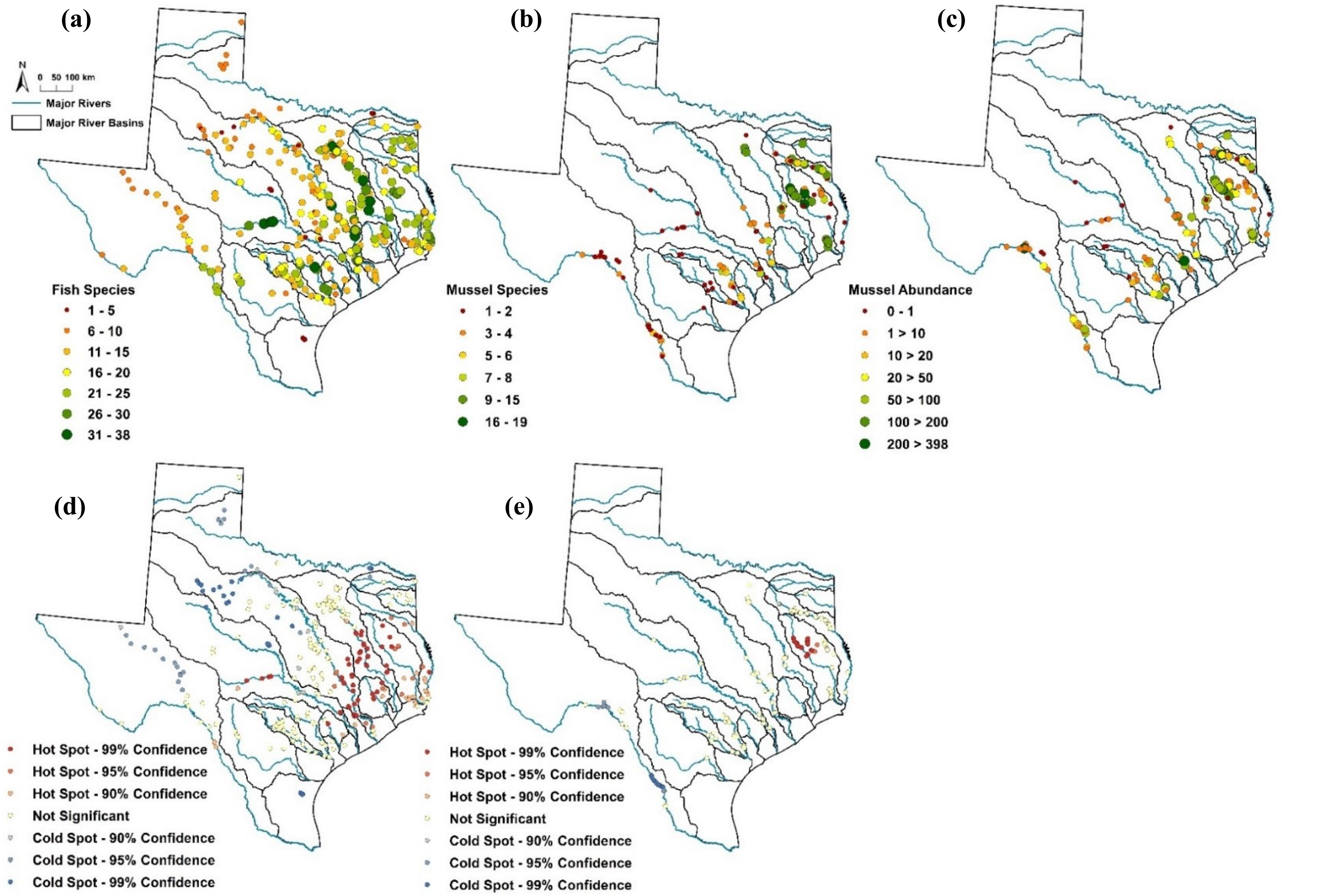

Figure 3.2. Texas mussel and fish species richness. Map of species richness for (a) fish species, (b) mussel species, (c) and total abundance of mussels (number of ind./p-h). Optimized hot spot analysis for (d) fish species, and (e) mussel species.

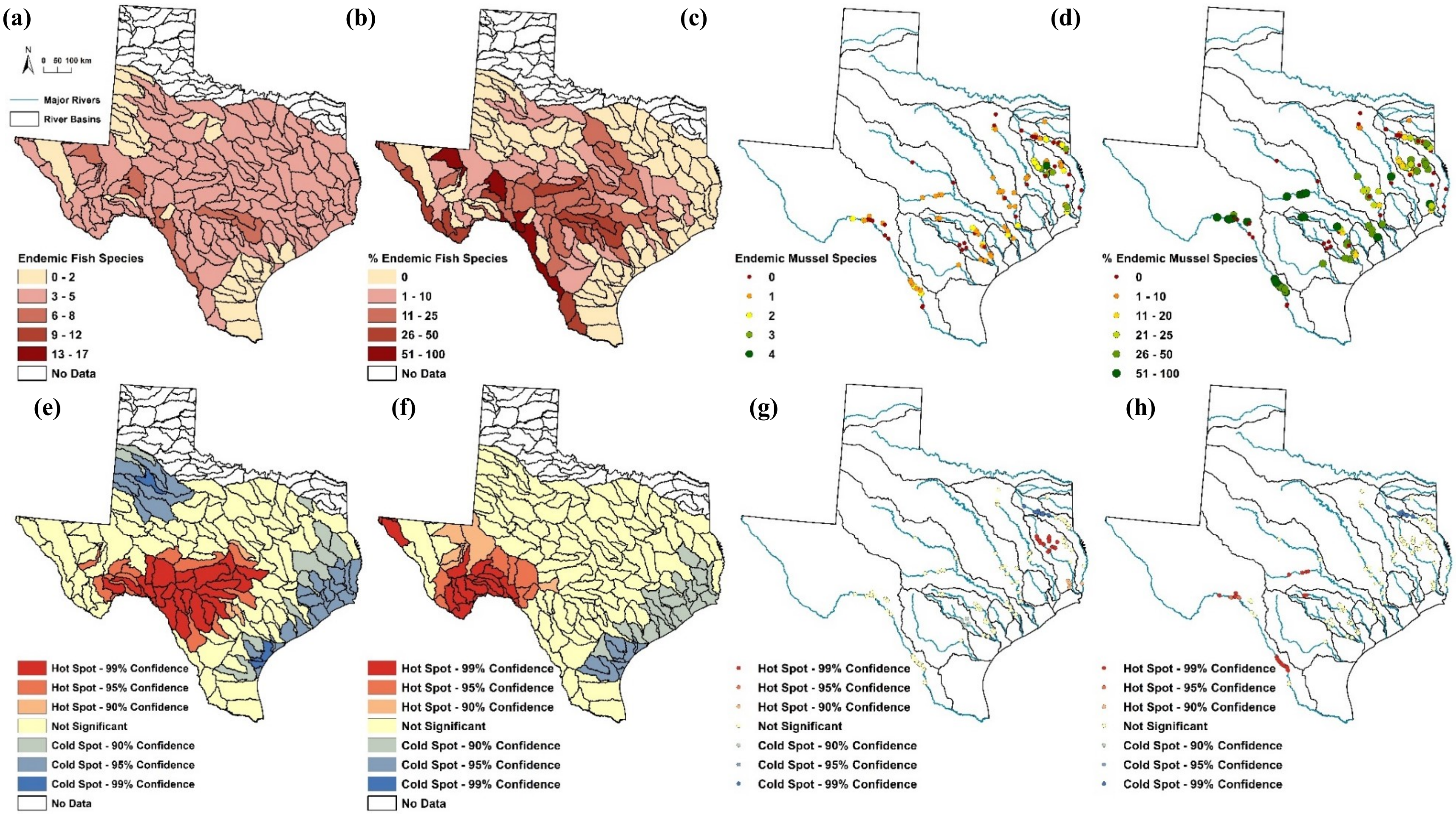

Figure 3.3. Texas endemic fish and mussel species. (a) Number of endemic fish species by sub-watershed, (b) percentage of endemic fish species by sub-watershed, (c) number of endemic mussel species, (d) percentage of endemic mussel species, (e) optimized hot spot analysis of the number of endemic fish species by sub-watershed, (f) optimized hot spot analysis of percentage of endemic fish species by sub-watershed, (g) optimized hot spot analysis of the number of endemic mussel species, (h) optimized hot spot analysis of percentages of endemic mussel species

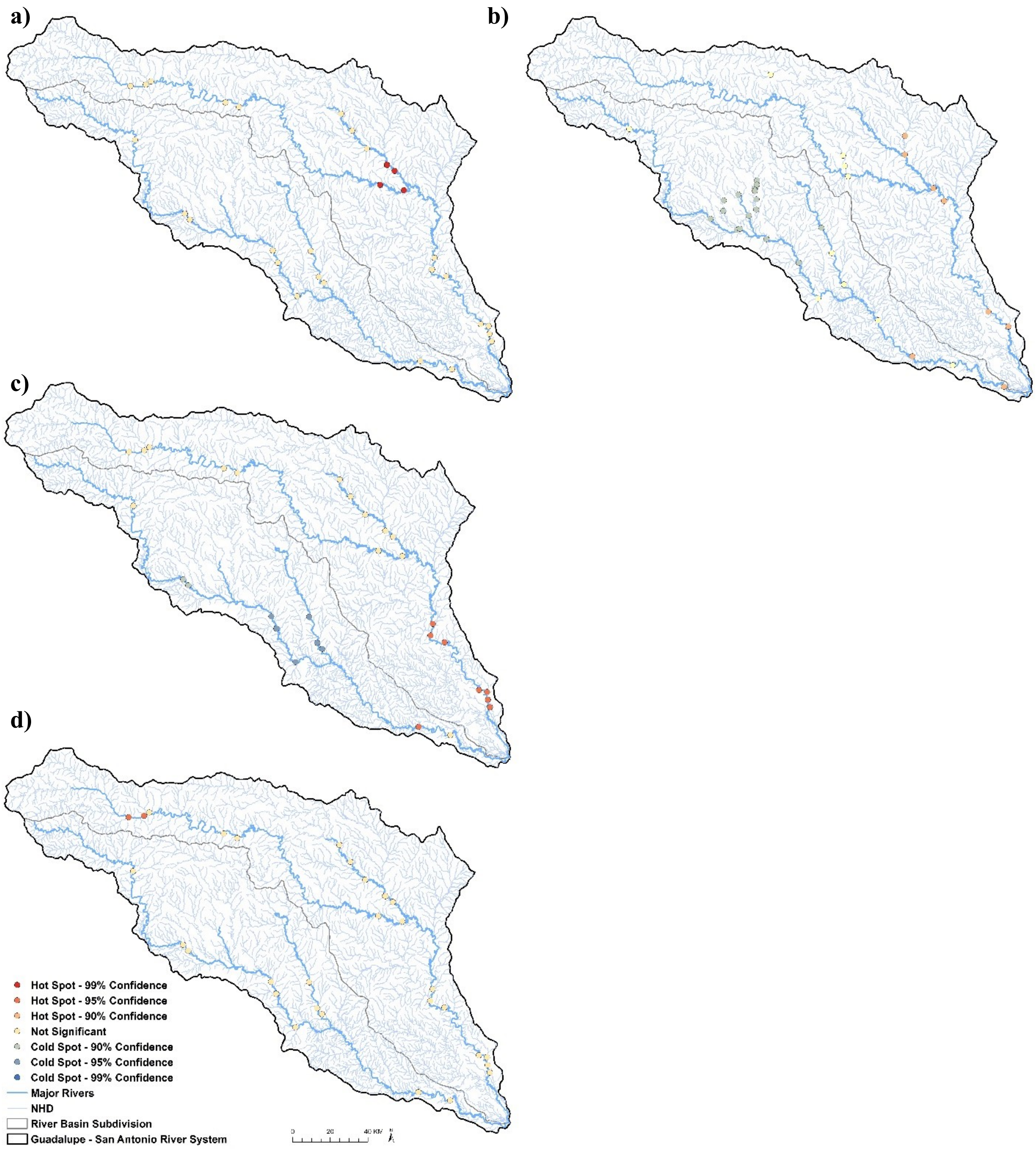

Figure 3.4. Optimized hot spot analysis of a) mussel species richness, b) fish species richness, c) number of endemic mussel species, and d) percent of endemic mussel species in the Guadalupe - San Antonio River System.

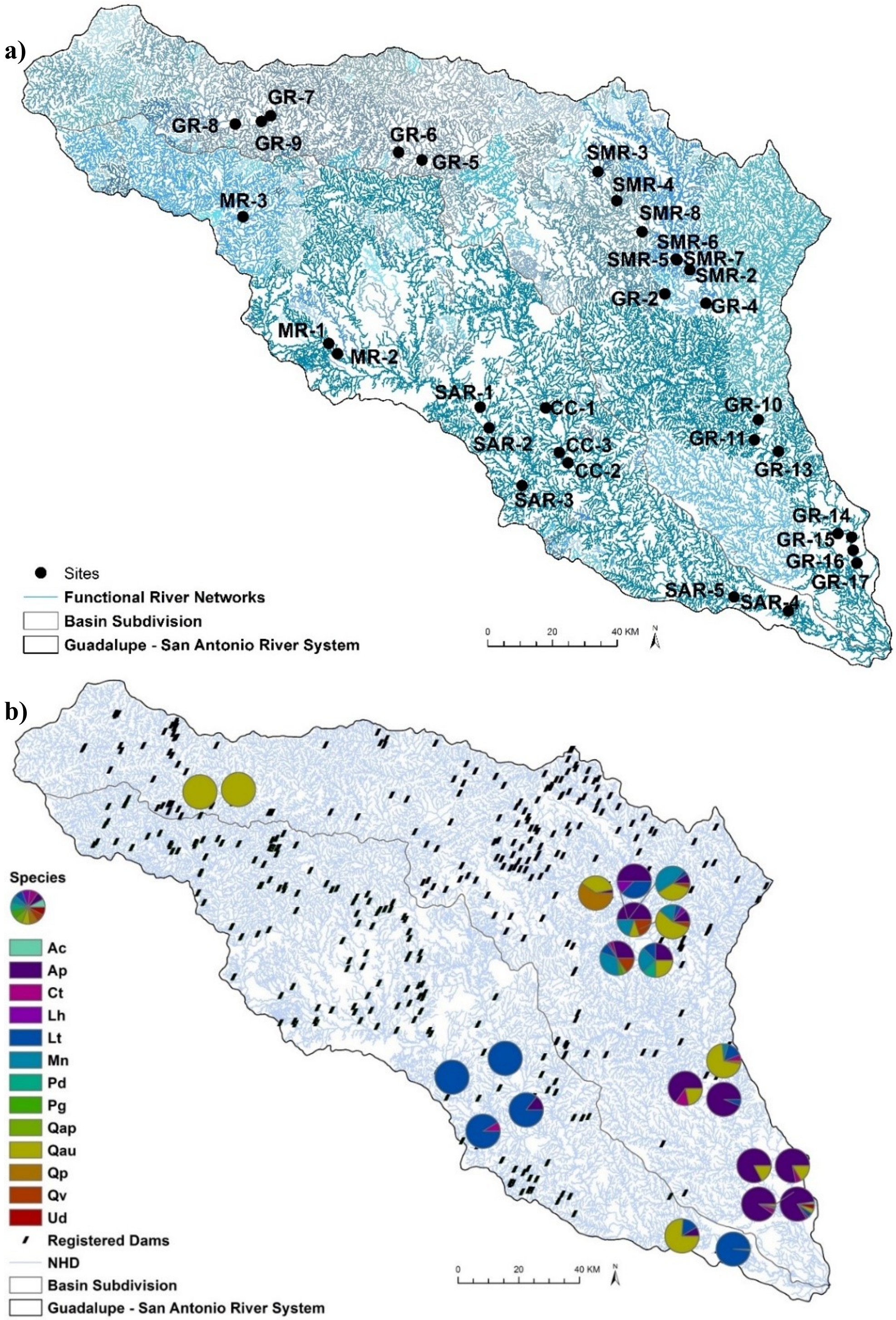

Figure 3.5. The Guadalupe – San Antonio River System with a) mussel sites and functional river networks shown, and b) species diversity and abundance for sites where species were present. Initials represent species Ac: *Arcidens confragosus*, Ap: *Amblema plicata*, Ct: *Cyrtonaias tampicoensis*, Lh: *Lampsilis hydiana*, Lt: *Lampsilis teres*, Mn: *Megalonaias nervosa*, Or: *Obliquaria reflexa*, Pd: *Plectomerus dombeyanus*, Pg: *Pyganodon (Anodonta) grandis*, Qap: *Quadrula apiculate*, Qau: *Quadrula aurea*, Qp: *Quadrula petrina*, Qv: *Quadrula verrucosa,* Ud: *Uniomerus declivis*.

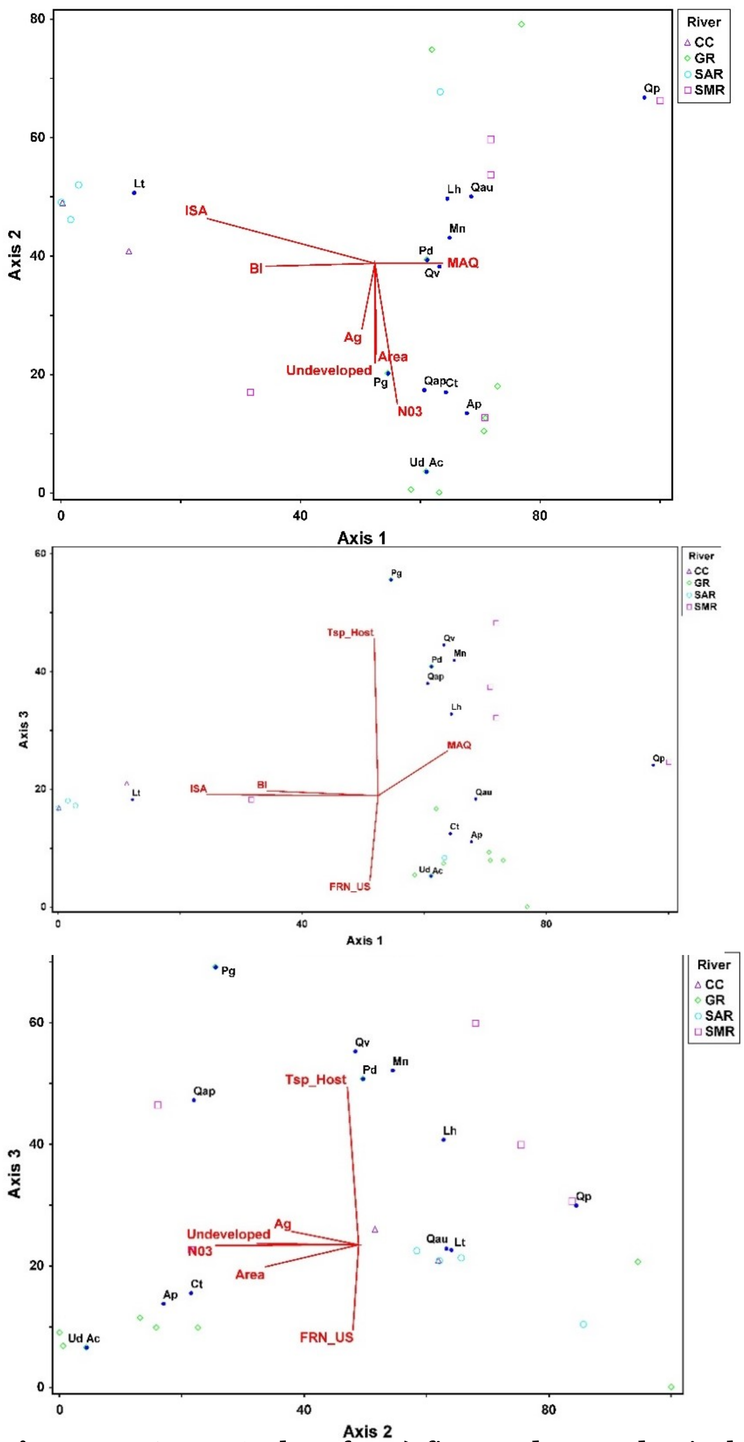

Figure 3.6. NMS plots for a) first and second axis, b) first and third axis, and c) second and third axes. Shapes represent sample sites, sites located on the same river have the same shape: San Marcos River: □, Guadalupe River: ◇, San Antonio River: o, and Cibolo Creek: Δ. Initials represent species Ac: *Arcidens confragosus*, Ap: *Amblema plicata*, Ct: *Cyrtonaias tampicoensis*, Lh: *Lampsilis hydiana*, Lt: *Lampsilis teres*, Mn: *Megalonaias nervosa*, Or: *Obliquaria reflexa*, Pd: *Piectomerus dombeyanus*, Pg: *Pyganodon (Anodonta) grandis*, Qap: *Quadrula apiculate*, Qau: *Quadrula aurea*, Qp: *Quadrula petrina*, Qv: *Quadrula verrucosa,* Ud: *Uniomerus declivis*.

4. DAM REMOVALS AND FRESHWATER MUSSEL CONSERVATION

4.1. INTRODUCTION

As of 2016, over 1,300 dams have been removed in the United States (American Rivers 2016), and this number is expected to increase as many dams in the U.S. reach the end of their usefulness (Doyle et al. 2003a). The increasing number of dam removals is emblematic of the paradoxical shift in the U.S. from trying to control and manipulate rivers, to attempting to restore them. The rate of dam removals has been climbing rapidly (Grant and Lewis 2015). In 2015 alone, 62 dam removals occurred (American Rivers 2016), which was nearly four times the number of new dams completed in the U.S. the same year (USACE 2016). Some states, such as Wisconsin and Pennsylvania, have removed well over a hundred dams (Bellmore et al. 2016).

The majority of dams removed in the United States have been small and older dams, which required repairs costlier than removal to continue operation (Stanley and Doyle 2003). Safety concerns are often cited as the main cause for dam removal, however the majority of dam removal projects do not actually provide a reason for removal (Pohl 2002). A review of stated justifications for dam removals showed environmental causes as the leading factor (Pohl 2002; Baish et al. 2002). While the majority of environmental efforts to remove dams focus on regions with anadromous fish populations (Baish et al. 2002; Pohl 2002), we can expect other species impacted by fragmentation, such as native freshwater mussels, to become targets for such restoration activities (Baish et al. 2002).

Freshwater mussels are a globally threatened fauna (Lydeard et al. 2004) that have severely declined in part due to habitat alterations, including the fragmentation of riparian habitat (Richter et al. 1997) and other dam impacts (Randklev et al. 2015; Troia et al. 2014; Tiemann et al. 2007; Sethi et al. 2004; Watters 1995). As we gain a better understanding of the effects of dam removal, the ability to use it as a conservation technique for threatened and endangered freshwater mussel species will increase. While dam removal is a promising restoration technique, it involves environmental tradeoffs (Doyle et al. 2003), and must be viewed within the appropriate context and carefully planned to yield the desired results. Future dam removals should be prioritized and implemented based on specific ecological and safety needs.

This study reviews dam removals in Texas, and then generates two dam removal prioritization models for the Guadalupe – San Antonio River System (GSARS) based on expert opinion and the existing literature-based information on dam removals. The research answered the following questions: (1) What are the spatial and temporal patterns of dam removal in Texas?, (2) How can these inform mussel conservation?, and (3) What are the criteria for prioritizing dam removals in the Guadalupe – San Antonio River System, and how should they be weighted?, and lastly (4) What dams have the highest and lowest priority for removal in the Guadalupe – San Antonio River System based on a dam removal prioritization model that prioritizes freshwater mussel conservation?

4.1.1. Background

While the majority of dam removals have involved smaller, older structures requiring expensive repairs (Bellmore et al. 2016; Stanley and Doyle

2003), the number of larger dam removals to restore fish habitat are increasing. In 2011, the largest dam removal in United States history took place with the removal of Condit Dam from the White Snake River in Washington (Gillman 2016). This was followed by the removal of two even larger dams on the Elwha River: the 210-foot-tall Glines Canyon Dam and the 108-foot-tall Elwha Dam, both also in Washington (Gillman 2016; American Rivers 2016). Four large dam removals are planned on the Klamath River (Gilman 2016; Gosnell and Kelly 2010), that will result in 482 km of reconnected river habitat (American Rivers 2016). Of the dam removals in the United States, over half of them have occurred during the last ten years (Grant and Lewis 2015). During this time, scientists have transitioned from requesting empirical and predictive environmental studies (Poff and Hart 2002; Bednarek 2001) to generalizing the geomorphic and ecological impacts of dam removals (Grant and Lewis 2015; Doyle et al. 2003a; Stanley and Doyle 2003; Bednarek 2001).

Dam removals can result in geomorphic and ecologic disturbances to already altered systems (Tullos et al. 2014; Riggsbee et al. 2007; Stanley and Doyle 2003), and result in a complex array of integrated biotic and abiotic process responses. These responses will likely occur over different time scales and need to be monitored and assessed accordingly (Bednarek 2001).

4.1.2 Geomorphic Response to Dam Removal

The main geomorphic challenge of dam removal is managing the amount of sediment and pollutants deposited within reservoirs (Riggsbee et al. 2007; Draut and Richie 2015; East et al. 2015; Grant and Lewis 2015; Warrick et al. 2015). Removing a dam increases the amount of sediment (and potentially nutrients and pollutants) transported and deposited downstream (Riggsbee et al. 2007; Draut and Richie 2015; East et al. 2015; Grant and Lewis 2015; Warrick et al. 2015). The amount of sediment transported from former impoundments can vary from 10 to 80 percent of the total reservoir sediment (Doyle et al.2003). The timing and style of dam removal are key determinates of the rate and volume of erosion (Grant and Lewis 2015).

The rate at which reservoir sediment becomes available for transport is controlled more by the dam removal process than watershed processes (Draut and Richie 2015). Staged dam removals (via slow dewatering) can result in slower erosion rates (Draut and Richie 2015; Grant and Lewis 2015), but produce a longer duration impact due to the large amounts of remaining sediment (Draut and Richie 2015). Once the initial amount of reservoir sediment is mobilized, the remaining sediment enter the system during floods and high flow events (Riggsbee et al. 2007; Draut and Richie 2015). The hydrology of the system will have a substantial impact on the long term geomorphological effects of dam removal.

The composition, grain size, and volume of reservoir sediment in relation to the transport competence and capacity, as well as the river's longitudinal profile and morphology, determine the downstream fate of eroded sediment (Grant and Lewis 2015). The composition and grain size of impounded sediments largely control the amount of sediment that is initially mobilized and transported through erosion. Saturated cohesive sediments erode the least, while non-saturated, non-cohesive sediments with higher percentages of sand (> 55%) erode quickly (Grant and Lewis 2015). Miscalculations in the percentage of stored sand can cause unpredictable sediment fluxes post removal, as seen in both the Elwha and Glines Canyon dam removals (Warrick et al. 2015). Sand requires higher stream power to transport than silt, and habitats impacted by sand deposition post removal may recover slower than those impacted by silt deposition, depending on the hydrology of the system (Cooper 2011).

Woody debris in reservoir sediments can help slow rates of erosion following dam removal. After large scale dam removal on the Elwha, woody debris reduced further erosion by 80% (Warrick et al. 2015). Additionally, the colonization of reservoir sediments by plants, regardless of type, may help river systems reach a quasi-equilibrium state by stabilizing sediment and decreasing runoff (Cool et al. 2011).

Once erosion begins, grain size also determines how far sediment is transported after a dam removal (Grant and Lewis 2015). Deposition of larger sediments can occur several kilometers downstream, while fine sediments are transported much farther downstream (Grant and Lewis 2015). Excessive sedimentary deposits are the product of imbalance between supply and transport capacity (Draut and Richie 2015; East et al. 2015). Substantial amounts of fine-grained sediment and low flows produced significant mud deposition in the Elwha River following dam removal (Draut and Richie 2015). In other systems, fine-grain sediments are quickly transported downstream with little morphological trace (Grant and Lewis 2015).

In addition to sediment considerations, the downstream channel and floodplain geometry will also affect the amount and rate of deposition (East et al. 2015; Grant and Lewis 2015). The ratio of reservoir width to the free-flowing channel width can provide an indicator of the amount of impounded sediment that will be transported to downstream reaches (Riggsbee et al. 2007). Slope is also an important variable determining erosion rates and downstream deposition, with lower slopes resulting in lower erosion rates (Burroughs et al. 2009). However, lower slopes may result in longer recovery periods post-removal, as more energy will be required to mobilize deposited sediment (Cooper 2011).

While the characteristics of individual dams and dam removals can be quite unique, there are certain trends in the geomorphological changes following dam removal. Removal can be considered as a geomorphological disturbance (Riggsbee et al. 2007), with a general recovery time of one to five years (Doyle et al. 2005). Channel morphology responses to dam removal can be considered within the framework of channel response to base-level lowering, and happen in sequential stages (Doyle et al. 2005; Doyle et al. 2003). Once a dam is removed, the water surface of the reservoir lowers and degradation begins, followed by continued degradation and widening, aggradation and widening, and finally quasi-equilibrium. These stages and the processes exhibited in each stage vary due to the specific site and situation of a dam removal.

4.1.3. Ecological Responses to Dam Removals

Removing a dam removes a physical barrier to riparian connectivity, allowing fish and other organism access to previously blocked habitats (Bednarek 2001). Fish assemblages often recover quickly after dam removal (Hogg et al. 2015; Gottgens 2009). Additionally, the coarser sediments exposed by the erosion of fines from the former reservoir-dominated channel (Bednarek 2001), can increase the available spawning habitat for migratory fish (Draut and Richie 2015; Konrad 2009; Doyle et al. 2003a).

After dam removal, lentic habitats transition back into lotic habitats (Stanley and Doyle 2003; Bednarek 2001). Organisms located in the reservoir prior to removal can be washed out or left stranded during dewatering (Stanley and Doyle 2003). Certain organisms can quickly recolonize the newly lotic system, others may need several years or decades to recover (Stanley and Doyle 2003). Former reservoir sediments can be quickly colonized by native plant communities after dam removal (Stanley and Doyle 2003), but there is also the risk of colonization by non-native and/or invasive species (Gangloff 2013; Stanley and Doyle 2003; Bednarek 2001).

Sediment deposition due to dam removal generally does not create detectable changes in algal or invertebrate communities, but fine-grained sediment can clog interstitial pores blocking hyporeic nutrient and oxygen exchange (Stanley and Doyle 2003). Some studies cite lower densities of macroinvertebrates downstream immediately following dam removal, with little change in the community composition of these communities (Renofalt et al. 2013; Orr et al. 2008; Thomson et al. 2005; Doyle et al. 2003). Other studies cite no significant differences in macroinvertebrate communities within one to two years of dam removal (Tullos et al. 2014; Stanley et al. 2002). However, impacts on macroinvertebrate communities may persist and/or increase with time (Renofalt et al. 2013), and the time scale of analyses can have a strong impact on the interpretation of ecological results post removal (Orr et al. 2008).

Long term ecological impacts can result from the remobilization of sediments by floods and high flows (Stanley and Doyle 2003). Additionally, the upstream migration of head cutting results in migrating sites of downstream deposition (Sethi et al. 2004). Post dam removal recovery rates of organisms are taxa and species specific. Some species may experience no deleterious effects, others may recover quickly, while others may continually decline (Renoflat et al. 2013; Morley et al. 2008), or require much longer periods to recover (Doyle et al. 2003a).

4.1.4. Freshwater Mussels and Dam Removal

Due to the importance of fish hosts in the life cycle of freshwater mussels, the positive response of fish to dam removal may result in an increase of native mussels (Gottgens 2009). After dam removal, deposition can occur too quickly for mussels downstream to migrate, burying them under sediment (Cooper 2011; Sethi et al. 2004). Mussels previously located in the impoundment of the dam may also experience high rates of mortality due to stranding, desiccation, and predation due to rapid dewatering (Sethi et al. 2004). High flows or floods may have a positive effect, creating suitable habitat for host fish spawning, or a negative effect, washing out extant mussels and/or burying them in sediment (Hauer 2015).

Freshwater mussels are sensitive to disturbance (Daniel & Brown 2013; Dycus et al. 2015), and have slow recovery rates (Sethi et al. 2004). While dam removal is a promising tool for restoration, it must be viewed within the appropriate context and carefully planned to yield desired results. For freshwater mussels, this will involve identifying the proximity of mussel populations to dams and making tradeoffs between increased habitat and potential mussel mortality rates as a result of dam removal.

4.1.5. Dam Removal Prioritization

Most dam removals are localized events that are opportunistic rather than strategically planned (Bellmore et al. 2016; Magilligan et al. 2016). However, strategic approaches would likely increase the rate and extent of functionally reconnected river network (FRRN) associated with dam removals (Magilligan et al. 2016). In general, a lack of well-developed decision analysis techniques has caused river restoration efforts to suffer (Corsair et al. 2009), and there is an increasing need for numerical tools to allocate resources for restoration activates (Branco et al. 2014).

Multiple criteria evaluation (MCE) provides a way to better define and incorporate multiple objectives while highlighting important trade-offs and can be useful to restoration planning (Corsair et al. 2009) especially when combined with Geographic Information System (GIS) as a support tool for complex decision making (Malczewski 2006). In the realm of river restoration, studies have used MCE to determine habitat suitability for species conservation (Kocovsky et al. 2008; Store and Kangas 2001), to measure the success of various restoration efforts (Huang and Zhang 2015), to decide how to spend funding for long term monitoring (Huang and Zhang 2015), and to evaluate management alternatives for individual dams with regards to imperiled migrating fish species (Mahmoud and Garcia 2000).

There has been increasing emphasis on applying MCE to improve dam removal decision making (Branco et al. 2014; Zheng et al. 2009; Kuby et al.

2005; Heinz Center 2002; Poff and Hart 2002). In one study, a single dam removal selected using MCE resulted in more connectivity than seven randomly removed dams (Branco et al. 2014). This highlights the benefit of using MCE and empirical methods to prioritize dam removals, and the importance of considering dam removals at a watershed scale as opposed to isolated events.

An important variable in most dam removal prioritization models is suitable habitat for migrating fish species (Hoenke et al. 2014; Martin et al. 2011; Mader & Maier 2008; Kuby et al. 2005). The Barrier Assessment Tool (BAT) calculated functional river networks (FRNs) at the watershed- and multi-state scales to prioritize dam removals for fish passage in two large dam removal initiatives in the United States (Martin and Apse 2011, 2013). These initiatives were spearheaded by The Nature Conservancy (TNC), a non-profit land and water conservation organization, and used expert workgroups to define data sources and the metrics for prioritizing dam removals. Anadromous and diadromous fish, resident fish, brook trout, cold water species, and other species of greatest conservation need (SGCN) were the ecological focus of these dam removal initiatives (Martian and Apse 2011, 2013).

4.2 METHODS AND MATERIALS

4.2.1. Data

To analyze patterns of dam removal in Texas, I obtained a list of dam removals from the Texas Commission on Environmental Quality (TCEQ). This dataset included information for 50 dam removals, and provided attributes and locations for the decommissioned dams. Additionally, this dataset provided the year and reason for removal.

Multiple data sources informed two dam removal prioritization models in the GSARS, and many of these have been previously discussed in Chapter 2-3 of this research. Data on registered dams was obtained from the TCEQ, and a full discussion and analysis of this dataset was provided in Chapter 2. Only registered dams located on the stream network and that could be associated with a COMID were prioritized for removal in the GSARS. This resulted in a reduced dataset of 273 out of 375 registered dams. Connectivity metrics were calculated with the BAT using TCEQ data on registered dams in conjunction with the National Hydrography Dataset High Resolution (NHD-HR) flowlines. For a full description of these methods please refer to the Modelling Connectivity section in Chapter 2.

Multiple sources provided biological data for fish and freshwater mussels. Biological datasets included 22 sampling sites of mussels in the GSARS and aggregated host fish data at the watershed scale (HUC 8). Ford and Oliver (2015) defined potential host fish species for Texas, and SGCN fish species were identified by the TCEQ Freshwater Fish SGCN database (2017). A full description of these biological datasets was provided in Chapter 3. The StreamCat database provided data for landscape and watershed scale variables including land use and cover. A more detailed description of the StreamCat database and how these variables were joined to the stream network was provided in Chapter 3.

Expert opinion defined metrics and their relative importance for inclusion in a dam removal prioritization model focused on freshwater mussel conservation. To gain expert opinion a one-day workshop, the Barrier Assessment Expert Workshop (BREW), was held on March 29, 2017 at the Meadows Center for Water and the Environment. The Graduate College at Texas State University provided funding for the workshop through the Graduate College Doctoral Research Support Fellowship. An application to host the workshop was submitted to the Texas State University Internal Review Board (IRB), and was approved at the exempt review level on February 24, 2017, application 2017384 (Appendix 4.1).

4.2.3 Barrier Removal Expert Workshop (BREW)

A total of 45 people from state and federal organizations, local and national non-profits, and academic institutions received e-mail invitations to BREW. Invitees were also provided a one-page workshop summary further describing the extent and purpose of the workshop (Appendix 4.2). Of the invited, 11 attended the workshop and participated in a collaborative process to decide on a completed list of ranked metrics. Workshop participants included representatives from government agencies, non-profit organizations, and academics specializing in the areas of fluvial geomorphology, instream flows, river restoration, and landscape change. Lacking from the workshop attendees were freshwater mussel ecologists; though they were invited, none could attend.

Workshop participants were provided with free lunch and parking, and offered a $30 Amazon gift card for their participation, though most were unable to accept the gift card given their employment category. All participants signed an informed consent form upon arrival (Appendix 4.1). At the workshop, participants were given a brief introduction to the topic background and proposed methods and a list of potential metrics to be included in the model (Table 4.1). A complete description of each potential metric is provided in Appendix 4.3.

Participants agreed on a final list of metrics, consisting of 24 variables assigned to five groups: biological metrics, connectivity improvement metrics, water quality metrics, landscape metrics, and dam attributes (Table 4.2). The biological metrics included the number of host fish, SGCN fish, and mussel species at the watershed scale, and mussel presence upstream and downstream of a dam. Additionally, biological metrics included the number of federally and state listed, and potential/candidate mussel species at the watershed scale and up- and downstream of a dam. Three connectivity improvement metrics were defined at and calculated by the BAT, these included: the total amount of potential reconnected stream length, the absolute gain of potentially reconnected stream length, and the relative gain of potentially reconnected stream network. BREW participants defined multiple water quality metrics for inclusion in the model including: ammonia and nitrate nitrogen instream concentrations, and high and low stream temperatures. Participants defined the Texas Clean Rivers Program (TCEQ 2017) as a potential data source for these metrics. An additional water quality metric, mean summer stream temperature, was included from the StreamCat database. Three landscape metrics from StreamCat were included in the model: total ISA land cover, undeveloped land cover, and percent riparian buffer. Dam attributes included reservoir length to storage ratio, average reservoir storage, and reservoir length. The BREW participants ranked individual metrics and metric groups on a scale of 1 to 9. For a detailed description of each metric please refer to Appendix 4.4.

4.2.2. Analysis of Dam Removals in Texas

I mapped and analyzed the dam removal dataset (2015) using a GIS framework in ArcGIS 10.3.1. I then used the NHD-HR in conjunction with aerial imagery to confirm the location of each dam removal, to determine if it was located on the river network, if the dam had been rebuilt, and to measure the length of resulting FRRN. The river network was considered functionally reconnected if the NHD-HR flowlines were connected and there was no registered dam located on the river network. Descriptive statics were used to summarize the dam removal dataset by river basin, height, owner, year built, year removed, reason for removal, and the calculated variable of FRRN.

4.2.4. Dam Removal Prioritization Models in the GSARS

The first iteration of the model incorporated the results of the BREW workshop. The final BREW model included a total of 15 metrics: eight biological metrics, three connectivity improvement metrics, and four landscape metrics (Table 4.4). While participants of the workshop assigned metrics to five groups, the model did not include the water quality metrics or dam attributes groups. Reservoir Length_Storage was the only metric in the dam attributes group (Table 4.2, 4.3). To properly weight and include this metric in the model, it was included in the landscape metrics group.

BREW participants recommended multiple metrics of water quality, and identified the Texas Clean Rivers Program (TCEQ 2017) as a data source. However, water quality data from the Texas Clean Rivers Program did not conform to extrapolation to the entire NHD network. Data was not available for the majority of NHD segments in the GSARS (Appendix 4.4). Mean Summer Stream Temperature was not included in the model, as it did not provide meaningful habitat suitability information.

Mean summer stream temperature was defined as the predicted mean summer stream temperature (July-Aug) for year 2014 (Hill et al. 2016, Table 4.2, 4.3). This was potentially a proxy for max daily stream temperature. A mean summer stream temperature of 35 degrees would indicate potential mussel extirpation; however, all mean summer temperatures were between 20.2 and 27.8 degrees. Additionally, federally listed freshwater mussel species were not present in the GSARS. This resulted in the exclusion of both federally listed mussels and federally listed upstream downstream mussels, from the model.

A second iteration of the model incorporated additional metrics based on the dam removal literature. This literature model built upon the BREW model in two ways. First, three additional metrics were included: dam age, or year of completion; dam owner, (1 = private ownership, 0 = non-private ownership), and dam size defined as maximum reservoir storage (Table 4.4). These metrics were included in a new metric group of dam attributes, and dam attributes group was ranked as being the most important metric group. Second, the literature model did not include Reservoir Length_Storage as a metric (Table 4.4). BREW participants recommended the metric as a measure of dam utility, and was defined as a ratio of a reservoir's length to the normal storage of that reservoir. The Reservoir Length_Storage metric weighted dams with larger reservoirs as better candidates for removal based on higher evaporation rates. Based on the available literature, larger dams are generally poorer candidates for removal, as they instigate political challenges, require more planning and resource to remove, and are often still functioning as source of hydroelectric power, water supply, flood control, etc.

4.2.5. Analytical Hierarchy Process (AHP)

I used analytical hierarchy process (AHP) as a numerical tool to weight each metric based on expert opinion (Saaty 1980). AHP is an MCE technique that ranks metrics on a scale of 1/9 to 9, so that each metric was 9 times to a ninth as important as any another metric. This was done by creating a reciprocal matrix for each group of metrics and using expert opinion from BREW to rank or compare individual metrics. An additional reciprocal matrix was created for ranking metric groups. This resulted in 4 reciprocal matrices for the BREW model.

I created a normalized matrix for each reciprocal matrix by multiplying the reciprocal matrix value by the sum of its column in the reciprocal matrix (Equation 4.1). The average of each row from the normalized matrix represents the priority vector or AHP weight (Equation 4.2). The AHP Weight is the number used to weight each metrics corresponding raster in the model.

I used Teknomo's method to ensure consistency. Multiplying each value in the reciprocal matrix by the corresponding priority vector resulted in a Weighted Vector (Equation 4.3). Then the sum of each row in the weighted vector matrix, i.e. the weighted sum vector (Equation 4.4), was divided by the corresponding priority vector to determine the consistency vector (Equation 4.5). The sum of the consistency vectors equaled λ max (Equation 4.6). The consistency index (CI) equaled λ max minus the number of metrics, divide by one less than the number of metrics (Equation 4.7). Lastly, the consistency ratio (CR) equaled CI divided by the random consistency ratio (RI) (Equation 4.8). If CR was below 0.1, then the results were considered consistent. This process was repeated for each reciprocal matrix. All AHP matrices are included in Appendix 4.6.

Equation 4.1

Normalized matrix = reciprocal matrix value ● Σ corresponding column

Equation 4.2

Priority Vector (AHP Weight) = average of row from normalized matrix

Equation 4.3

Weighted Vector = reciprocal matrix value ● Priority Vector

Equation 4.4

Weighted Sum Vector = Σ corresponding row in the Weighted Vector matrix

Equation 4.5

Consistency Vector = Weighted Sum Vector / Priority Vector

Equation 4.6

λ max = (Sum of consistency vectors)

Equation 4.7

Consistency Index (CI) = (λ max - n) / (n – 1)
Where n equals the number of metrics.

Equation 4.8

Consistency Ratio (CR) = CI/Random Consistency Index (RI)*
*RI is obtained from Table 4.4

4.2.5. Compromise Programing

I used compromise programing in a GIS to create raster files of each metric for the registered dams in in the GSARS. Compromise programming is a distance based technique that depends on the point of reference or "ideal" point and attempts to minimize the "distance" from the ideal solution for a satisficing solution. The closest distance to the ideal across all criteria is the compromise solution or compromise set.

I used Raster Calculator, a tool in the ESRI ArcGIS extension Spatial Analyst Toolbox (2015), to determine the minimal distance to the ideal alternative using Equation 4.9. This analysis involved two parts, first I used Raster Calculator to calculate and weight the relative Z value for each raster layer/metric using AHP weights (Equation 4.10, Table 4.3, 4.4). Second, I combined the individual weighted raster layers/metrics for each group of metrics using Equation 4.11. This resulted in a single raster file for each metric group, or three raster files for the BREW model and four for the literate based model.

I repeated these steps for the resulting raster layers/metric groups to create a single and final raster layer for each model (Figure 4.1). I extracted the resulting final raster value, or d score, by points i.e. registered dams in the GSARS. The d score represented the priority ranking of dams for removal on a scale of 1 to 0. Resulting d scores closer to 1 indicated less favorable dams for removal, while those closer to 0 indicated dams that were better candidates for removal based on the selected metrics.

Equation 4.9

$$dx, y = \left[\sum_{i=1}^{n} \frac{Wi}{\sum_{i=1}^{n} Wi} \left(\frac{Z_{\mathrm{i}}^{best} - Z_{x,y,i}}{Z_{i}^{best} - Z_{i}^{worst}} \right) ^\wedge \rho \right]^{1/p}$$

Equation 4.10

POWER (((Z_{best} – [raster layer]) / (Z_{best} – Z_{worst})),*p*)*AHP weight

Equation 4.11

POWER (([layer_1] + [layer_2] + ... [layer_n]), 0.5)

Where Wi is the weight assigned a given variable, Z is the value of that variable, and p is the scaling coefficient. Z^{best} indicates the most desirable value for a given variable, and Z^{worst} indicates the least desirable value for a given variable.

To compare model results, standard deviations from the mean d score defined five categories of dam removal priority. Dam removal priority categories were: very high (>-2.5 std. dev.), high (-2.5 - -1.5 std. dev.), mid-range (-1.5 – -0.5 std. dev), low (-0.5 – 0.5 std. dev.), and very low (0.5 – 1.2 std. dev.). Analysis of the top twenty dams ranked for removal and the distribution of d scores for each model allowed further model comparisons.

4.3. RESULTS

4.3.1. Dam Removals in Texas

There have been a total of 50 dam removals in Texas since 1983, resulting in a total of 1816.1 km of FRRN. There was a noticeable spike in dam removals between 1994 and 1996 (Figure 4.2). Four tailing ponds were removed in 1995, and another four oxidation dams were removed in 1996. These dams did not occur on the river network, and thus resulted in 0 km of FRRN. Dam Removals in

2006 and 2015 sharply increased the cumulative length of FRRN (Figure 4.2). The Patricio Lake Dam removal in 2007 had the second largest amount of FRRN, with 305.3 km, and was one of only two removals to result in over 100 km of FRRN (Figure 2.11). The Ottine Dam removal occurred on the San Marcos River in the Guadalupe Basin in 2016. This removal resulted in 1283 km of FRRN, 70.6% of the total FRNN.

Dams have been removed in 13 of the 26 major river basins in Texas, including three coastal basins, and the largest number of removals have occurred in the Colorado (N = 9), Rio Grande (N = 7), and Trinity (N = 7) river basins (Figure 4.3). Dams with unknown or unrecorded ages accounted for 26% of the removals (N = 13). Of the dams removed, most were built at least 37 years ago, between 1960 and 1979 (N = 17). (Figure 4.4). Over 70% of removed dams had a height of less than 9 meters, and nearly all were privately owned (N = 40, Figure 4.4). The main purpose for dam removals (N = 20) was the removal of a liability and state agency involvement (Figure 4.4). This was the reason for the removal of both the Ottine and Patricio Lake dams.

The removal of the Patricio Lake, Ottine, and the Tex Iron dams were responsible for 87.5% of the total FRRN. Average FRRN was 36.3 km, but the median was 0.2 km, revealing the strongly skewed distribution driven by the Ottine Dam removal, that was responsible for 70.6% of the total amount of FRRN. Nine dams were rebuilt, and 15 dam removals did not occur on the river network, so that 24 dam removals resulted in 0 km of FRRN (Figure 4.5). Of the dam removals that resulted in FRRN, the majority resulted in less than 10 km (N = 20), and nine of these dams resulted in less than 1 km of FRRN (Figure 4.5).

Additionally, the total amount of FRRN was likely over estimated as only registered dams were considered as river barriers in the study

4.3.2. Dam Removal Prioritization Models in the GSARS

The BREW model categorized two dams as being very high priorities and one as a high priority for removal, all three dams occurred in the Guadalupe River Basin (Table 4.6, Figure 4.6). The model categorized the majority of dams as low priorities (Table 4.6, Figure 4.6). Dams in the very low removal priority category occurred exclusively in the San Antonio River Basin, and clustered around the middle of the basin (Figure 4.6).

The BREW model assigned most dams a d score higher than 0.90, with a mean of 0.93 (std. dev. = 0.059) and range of 0.31 to 0.99 (Figure 4.7). Examination of the top twenty dams prioritized for removal in the BREW model includes 27 dams that are categorized as mid-range priorities. However, all of the top twenty dams had high values for biological metrics, and most, if removed, would result in over 1000 km of potentially reconnected stream length (Table 4.7). Additionally, the top twenty ranked dams had watersheds with less than 30% ISA land cover, and most of their associated watersheds had over 70% undeveloped land cover (Table 4.7). More than half of the top twenty dams had watersheds with more than 35% of a 100-meter buffer around the stream networks categorized as riparian buffer (Table 4.7). The year of completion ranged from 1989 to 2002, and 15 out of 20 had private owners. Six of the top twenty dams had reservoirs over 1000 af of storage (Table 4.7).

The literature model categorized 15 dams as very high removal priorities, and an additional five dams as high removal priorities (Table 4.6, Figure 4.8)

Eleven of these dams occurred in the Guadalupe River Basin, eight were classified as very high and three were high removal priority dams (Figure 4.8). Seven very high and two high removal priority dams occurred in the San Antonio River Basin (Figure 4.8). Groups of dams in the very low removal priority occurred in the middle of the Guadalupe and San Antonio river basins (Figure 4.8). The literature model categorized the majority of dams as low or very low priorities for removal (Table 4.3, Figure 4.8). There was a smaller range and a greater variation in d scores with the literature model, with a mean of 0.93 (std. dev. 0.047) and a range of 0.63 to 0.99 (Figure 4.9).

The top twenty dams in the literature model had relatively high values for fish species metrics, but lower values for metrics of mussel presence (Table 4.8). Only four of these dams had more than one mussel species present either upstream or downstream, and only three with a state listed and/or potential/candidate species present upstream or downstream. Of the top twenty dams, 16, if removed, would result in over 100 km of reconnected stream network, with seven resulting in more than 1000 km of FRRN. Four would result in less than 20 km of reconnected stream network. The top twenty dams had watersheds with less than 12% ISA land cover, and more than 70% undeveloped land cover. The watersheds for these twenty dams had 20% or more of the area surrounding the stream network (100-meter buffer) categorized as % riparian buffer (Table 4.8). All twenty dams had reservoirs less than 1000 af, private owners, and were built before 1900 (Table 4.8).

Both models prioritized the Cuero Lake Dam in the lower Guadalupe River Basin as the top removal priority, and this was the only dam to co-occur on the top twenty lists. (Table 4.7, 4.8). The dam was built in 1989, was listed as having a private owner and a reservoir of 808 af. This dam's watershed had less than 10 present ISA, and nearly half of the river network in the watershed had a riparian buffer. Additionally, there were potentially 37 host fish species present and eight mussel species occurred either upstream or downstream of the dam. Of these eight species two were listed as state threatened and candidate spices. *Q. aurea* and *Q. petrina* both occurred upstream of the dam, and *Q. aurea* also occurred downstream of the dam.

The BREW model categorized Canyon Dam as a mid-range priority for removal, while the literature model classified it as a very low priority. The removal of Canyon Dam would reconnect 426.3 km of river. At least one mussel species was located upstream of the dam, *Q. aurea*, and the dam's watershed has less than 10 percent ISA. This dam was also the largest dam in the GSARS with over a reservoir over a million af and is owned by the federal government.

4.4. DISCUSSION

4.4.1. Dam Removals in Texas

In the Trinity and Colorado river basins, dam removals appear to be grouped around major cities, i.e. Austin and the Dallas-Fort Worth Area, and are motivated by liability issues and development. This potentially reflected increasing population growth in these areas associated with increased land values. Other clusters of dam removals, such as those in the Sabine and Rio Grande river basins, were the result of ceased industrial operations where multiple dams were removed together. Dam removals that resulted in zero km of FRRN were mostly industrial use ponds. These industrial use ponds were connected to the river network through artificial canals, and when the ponds were no longer needed neither were the canals.

Dam removals in Texas generally follow national dam removal trends, with the majority of removals involving smaller, older structures (Bellmore et al. 2016, Heinz Center 2002; Stanley and Doyle 2003). Most of the dams in Texas are smaller, privately owned structures built before 1980. This indicates a potentially considerable number of outdated structures that likely require expensive upkeep or repairs, as prime candidates for removal (Heinz Center 200; Stanley and Doyle 2003). Additionally, removing these structures involves working with private individuals, as opposed to coordinating with multiple stakeholders.

The Ottine Dam removal reconnected over 1000 km of river, and is a powerful example of the ability of dam removals to restore river connectivity. However, most of the dam removals in Texas resulted in less than 1 km of FRRN, and three dam removals accounted for nearly 90% of the total FRRN. These results highlight the isolated and opportunistic nature of most dam removals (Bellmore et al. 2016; Magilligan et al. 2016), and further support the need for more strategic planning and management of dam removals (Magilligan et al. 2016).

Previous studies have called for more reliable record keeping and communication between organizations regarding dam removals (Bellmore et al. 2016; Heinz Center 2002). American Rivers (2016) only lists seven dam removals for Texas, as opposed to the 50 reported by the TCEQ. These additional removals potentially make Texas sixth in the nation for number of dams removed, but other states likely also have unreported dam removals and thus underreported totals. As permits are required to remove a dam (American River 2006), there is already a mechanism in place for obtaining data on dam removal. This data, however, unless voluntarily reported to American Rivers, is not collected or maintained in a national database.

A congressionally authorized national inventory of dam removals that assigns formal responsibility to a single agency, similar to the National Inventory of Dams maintained by the Army Core of Engineers (USACE) has been previously recommended (Heinz Center 2002). Such a national an inventory would provide a way to reliably maintain and organize data about dam removals. Additionally, a national database would likely help standardize record keeping and data reporting.

The United State Geological Survey (USGS) currently houses the USGS Dam Removal Science Database (Bellmore 2015). The USGS Dam Science Database is a collection of empirical monitoring data from 179 publications for 130 dam removals worldwide (Bellmore 2015). This data has been combined with the American Rivers Dam Removal Database to create an online database tool, the USGS Dam Removal Information Portal (DRIP) (Bellmore et al. 2016). Thus, the USGS would be a reasonable choice to maintain a national inventory of dam removals.

4.4.2 Dam Removal Prioritization Models in the GSARS

Expert opinion obtained at a one-day workshop defined metrics included in the BREW model. Workshop participants elected not to include any metrics pertaining to individual dam characteristics, such as size, age, or ownership. Without these metrics prioritization of dams did not consider feasibility of actual dam removal, but only freshwater mussel conservation.

A review of the literature and past dam removal in Texas indicated that small, older, dams that are privately owned are more likely candidates for removal (Bellmore et al. 2016, Heinz Center 2002; Stanley and Doyle 2003). These types of dams are more likely to be in disuse and disrepair, and are potentially cheaper to remove than repair (Stanley and Doyle 2002). Additionally, private owners can make decisions about removing their dams more readily than governments or other types of organizations that might require stakeholder input and/or additional authority/consensus.

The literature model built upon the existing BREW model by incorporating metrics of dam size, age, and ownership in a group of dam attribute metrics. The literature model improved upon the BREW model by accounting for the actual feasibility or likelihood of removal. This was demonstrated by the categorization change of Canyon Dam from a mid-range to a very low candidate for removal.

Canyon Dam was built primarily for flood control and water supply spurred by several floods in the 1930's and the drought of the 1950's, and created over a million af water storage (GBRA 2017). The federal government currently owns the dam and its associated reservoir. The USACE owns the flood control portion of the reservoir, while the Guadalupe Basin River Authority (GBRA) owns the right to the conservation storage (GBRA 2017). Additionally, in 1989 a hydroelectric plant was built with a six-megawatt capacity, and Canyon Dam became a source of hydroelectricity (GBRA 2017). Based on the current use, size, and ownership and management of Canyon Dam, removal was considered unrealistic, and the literature model's categorization of the dam as a very low candidate for removal as a noticeable improvement of the original BREW model.

By including dam attributes as metrics, the literature model better accounted for practicality of dam removal, but it did not prioritize freshwater mussel conservation as directly as the BREW model. The literature model categorized multiple dams as high or very high candidates for removal that would seemingly offer little value for freshwater mussel conservation. Particularly, the model categorized several dams in the upper and middle San Antonio River Basin where no mussel sites were located up- or downstream of the dam. Future renditions of the model should attempt to enhance the ability of the model to prioritize dams that are both practical candidates for removal and relevant to freshwater mussel conservation.

Data on the hazard classification of dams in Texas was unavailable through the TCEQ, but is potentially available through other sources such as published white papers and the National Performance of Dams Program (NPDP) maintained by Stanford University (2017). In Texas, 1,771 dams are classified as either high- or significant-hazard dams indicating probable or possible loss of life in the event of a dam failure, respectively (ASCE 2012). Old age and neglect intensifies a dam's vulnerability to failure (ASCE 2012), and the number of high-hazard dams in the United States is growing as dams age and development intensifies in areas downstream of dams (ASCE 2017, 2012). The Water Infrastructure Improvements for the Nation (WIIN) Act was signed into law in 2016, authorizing a dam rehabilitation and repair program to fund repair, rehabilitate, or remove non-federal, high-hazard dams (ASCE 2017). Incorporating hazard classifications into future renditions of the model may help prioritize dams for freshwater mussel conservation that are already being considered as candidates for removal.

Previous dam removal prioritization models focused on removing barriers to fish migration, and suitable habitat for migrating fish species was an important variable (Hoenke et al. 2014; Martin and Apse 2011, 2013; Mader & Maier 2008; Kuby et al. 2005). The models presented here included metrics of habitat quality for freshwater mussel species. However, host fish are vital to mussel species dispersal (Strayer 2008), and removals that prioritize freshwater mussel conservation should also benefit other aquatic organisms, such as fish that recover quickly after dam removal (Hogg et al. 2015; Gottgens 2009).

The Cuero Lake Dam in the lower Guadalupe River Basin was the top candidate for dam removal in both models. Based on broad scale analysis, removal of Cuero Lake Dam would reconnect several thousand km of stream network and remove a barrier to dispersal of mussel communities located up- and downstream of the dam. Further investigation of this dam, however, reveals that the right rim of the reservoir was breached in 2004, and in 2012 Small Hydro of Texas, Inc. filed to surrender their licensing exemption (FERC 2013).

The Guadalupe River currently circumvents the Cuero Lake Dam, and thus the dam no longer presents a barrier to migrating aquatic species. Despite these developments, Cuero Lake Dam is still currently listed in both the TCEQ dam database and the NID as a hydroelectric dam, and the current direction of the Guadalupe River around the dam is not reflected in the NHD-HR. This highlights the importance of evaluating dams on an individual basis after initial selection by the model, and the need for better record keeping of existing dam structures.

4.4.3. Model Limitations

There were only 22 sites available for this study to identify the presence of mussel species in the GSARS. This limits the ability of the model to prioritize dams based on freshwater mussel conservation, as it is unknown if mussels are located upstream and/or downstream of a dam. Certain dams were likely undervalued as candidates for removal due to the lack of available mussel data for the model.

Fish data for both species of host fish and SGCN was aggregated at the watershed scale (HUC 8), and likely caused the model to overestimate the rank of certain dams as priorities for removal. This was also true for the metrics of mussels, state listed mussels, and potential/candidate mussels, but since there was no mussel data available for large portions of the GSARS these metrics were included as auxiliary measures of mussel species presence. There are eight watersheds at the HUC 8 scale in the GSARS, this resulted in large numbers of dams having the same values for the metrics of Host Fish, SGCN Fish, Mussels, State Listed Mussels, and Potential/Candidate Mussels. This general lack of variation in biological metrics resulted in similar d scores for most dams in both models. Future versions of the model should consider additional data sources and/or aggregation to a finer watershed scale, such as HUC 10 or 12, to avoid this problem. Alternatively, these metrics could be grouped into subset of biological metrics and weighted less heavily to offset their effect on model results.

Noticeably absent from BREW were freshwater mussel ecologists. Their absence resulted in a model that did not include the opinions of freshwater mussel experts despite the importance of this input for freshwater mussel conservation. BREW participants felt strongly that mussel ecologists were a vital part of continuing the discussion about dam removal as a freshwater mussel conservation strategy.

The models presented in this study should be viewed as works in progress and a starting point to further discuss potential river connectivity improvements as a conservation strategy for freshwater mussels. While there are numerous limitations to modeling dam removal prioritization, particularly at the river basin scale, it provides a systematic way to evaluate and make decisions about dam removal. This study is the first attempt the author is aware of to prioritize dams for removal in Texas, and will hopefully encourage similar future scientific endeavors.

4.4.4. Refinement of Model Results

Specific metric gaps were discussed by BREW participants that were considered important components of prioritizing dam removals for freshwater mussel conservation, but were unable to be included in the model. This was due to either a lack of data or because data was not available in a usable format for the model. The current model provides a basin scale prioritization for dam removals that can be used to identify potential removal projects for freshwater mussel conservation in the GSARS. Additional data sources can then be used to further assess the feasibility and benefit to mussel species of particular dams removals. Data important to freshwater mussel conservation in the context of dam removals that are only available at finer scales or on a discrete basis are discussed below, and should be evaluated as site specific metrics before removing a dam.

Data related to instream flows and discharges as well as channel morphology was discussed at length, and considered a critical component of future discussions and inclusion in a model that prioritizes freshwater mussel conservation. Consensus was on the value of including a metric that described the potential change related to improved connectivity. Instream flows that are too low can result in sedimentation, burying and suffocating mussels, while flows that are too high can dislodge mussels (Vannote and Minshall 1982; Hartfield & Ebert 1986; Strayer 1999). After dam removal, high flows or floods may have a positive effect, creating suitable habitat for host fish spawning, or a negative effect, washing out mussels and/or burying them in sediment (Hauer 2015).

Data related to flow alterations such as reservoir operations, groundwater pumping, and diversions was considered important. Releases from dams can increase flow magnitudes and dislodge juvenile mussels (Hardison & Layzer 2001; Daraio et al. 2010a), create sediment scour that is harmful to mussels (Dennis 1984; Aldridge et al. 1987), and expose mussels to altered timing and temperature of flows (Galbraith and Vaugh 2011). Additionally, metrics concerning water use/withdrawals were discussed by the workgroup as a potentially important metric to include in the model.

The type of substrate available at particular locations was discussed as an important metric or set of metrics. The importance of substrate availability to mussel abundance and distribution varies by species (Vaughn et al. 2015; Watters 2000). Juvenile mussels require stable substrate and a well oxygenated interstitial zone (Hauer 2015; Scheder et al. 2015) as they will remain buried for several years (Watters 2000). Additionally, the type and gain size of impounded sediments is a major determinate of the amount of sediment that is initially eroded or mobilized from a reservoir after dam removal (Cooper 2011; Grant & Lewis 2015; Warrick et al. 2015). Siltation can smother mussels, especially mussels not adapted to soft substrates (Bogan 1993; Brim Box & Mossa 1999; Watters 2000), and excessive sand is erosive to mussel shells (Houp 1993). Large amounts of silt can also clog interstitial spaces creating a hardpan layer unsuitable for mussel habitation (Brim Box & Mossa 1999) and reduce the exchange of nutrients and oxygen in the interstitial zone (Watters 2000).

The amount of stream length required to support current and future freshwater mussel populations is an important metric to consider, and is likely species-specific and complicated by climate change. Increased duration and intensity of droughts caused by climate change result in larger sections of streams going and staying dry, this results in mussel die-offs through desiccation and stranding (Gates et al. 2015, Haag & Warren, 2008; Galbraith et al., 2010, Randklev et al., 2013). Climatic changes can also lead to more frequent and larger flood events that wash out mussels and/or their host organisms (Melillo et al., 2014).

BREW participants also recommended multiple metrics of water quality, but water quality data did not conform to the scale of the model and/or did not add a meaningful measure of habitat quality. Multiple factors of water chemistry are thought to affect distribution and abundance of freshwater mussel species. Pesticides have generally been linked to declines in freshwater mussel species, and ammonia, specifically unionized ammonia, is highly toxic to mussels (Haag 2012). Water temperature also an important factor, with most mussel species preferring stream temperatures under thirty degrees Celsius (Haag 2012).

The model results presented in this study can be further refined by evaluating the ecosystem serves associated with potential dam removals. Ecosystem services are the services ecological systems provide, such as water filtration and climate regulation (Costanza et al. 1997). They provide a way to evaluate the monetary benefits of river restoration, and played a role in the decision to remove two large dams on the Elwha River (Gowan et al. 2006). This role was minor however, and valuing ecosystem services is inherently based on societal values of these services (Gowan et al. 2006). Nonetheless, ecosystem services provide a way to appraise the losses and gains associated with dam removal, and allows for the monetization of the historical and cultural importance of a dam prior to removal.

4.4.5. Mussel Conservation Recommendations

The BREW and literature models categorized nearly all dams in and around urban centers as very low priorities for removal. In Texas, dam occurrence and urban centers are highly correlated (Chin et al. 2008). Larger numbers of dams increase river fragmentation, and dam removals in these areas would result in shorter potentially reconnected stream lengths. These areas also generally lacked data for mussels, potentially because they are poor sampling locations, and had lower numbers of host fish species.

Highly urbanized areas may provide poor habitat for mussels (Brown et al. 2010; Gillis 2012, Gillis et al. 2014), and urbanization may lead to declines in freshwater mussel richness and abundance (Burlakova et al. 2011). In highly urbanized watersheds, dam removal alone may be a poor strategy for freshwater mussel conservation. Regardless of whether a dam removal occurs, mussel species will not be able to recolonize a location if they have already been extirpated from both upstream and downstream river reaches. In such cases, mussel species may be reintroduced via breeding or relocation programs after suitable habitat has been restored. This would be a multi-step river restoration process and require more resources to accomplish than dam removal alone.

Before a dam is removed, it should be determined if any mussels are downstream of the dam, and such removals may require more careful planning, particularly if the mussel species have a protected status. In the case of downstream mussels, the percentage of sand and silt in the dam reservoir may be used to determine if the dam is a good candidate for removal. Reservoirs with over 55% sand are potentially hazardous as the sediment will erode quickly (Grant and Lewis 2015), and sand is erosive to mussel shells (Houp 1993). Planting native plants in the former reservoir may help stabilize sediment, and prevent mussel mortality from siltation. Alternatively, mussels may be relocated prior to dam removal. Dams with downstream mussel species will require more planning and potential mitigation before removal, but removing a dam that separates upstream and downstream mussels facilitates dispersal between these communities and ultimately result in increased mussel species richness and abundance (Gottgens 2009).

4.5. CONCLUSION

Previous studies have used multi-objective programming to evaluate dam removals under competing or alternative priorities, such as ecological and economic trade-offs (Kuby et al. 2005; Zheng et al. 2009). This study prioritizes dams for removal in two major Texas river basins for the sole purpose of freshwater mussel conservation, and represents a scale of analysis not normally addressed in the literature. Two large dam removal initiatives have prioritized dam removals over a state or multi-state geographic extents (Benner et al. 2011; Martin and Apse 2011), but most dam removal projects involve smaller geographic scales, either single dams or watersheds. Dam removal prioritization at the river basin scale allowed for the evaluation of 273 potential dam removals.

The dam removal prioritization models created in this research prioritized dam removal for a single purpose, i.e. freshwater mussel conservation, and were grounded in expert opinion. AHP was used as a numerical tool to translate the relative importance of metrics determined by workshop participants into standardized weights. Combining AHP with compromise programing produced a single solution that prioritized dams for removal based on multiple weighted metrics. Using compromise programming in a GIS provided a way to generate geographic representations of model results and was considered a strength of the methodology. The literature model was an adaptation of the original BREW model that incorporated additional literature derived metrics and weights. The literature model served as an example of how the methods described here can be used to continue to refine and adjust model results.

The methods and results presented in this study should be considered support tools for the complex decision making surrounding dam removal as a freshwater mussel conservation strategy. The visual and geographic nature of the output was designed to bolster effective communication of model results to multiple stakeholders. It is hoped that presenting model results in this way will facilitate more meaningful conversations about freshwater mussel conservation in the context of dam removal.

Better record keeping of existing and planned dam removals will improve the ability to assess their merit as conservation strategies for multiple aquatic organisms, such as freshwater mussel species. While dam removal may still have a negative connotation in Texas, 50 dams have been removed and future dam removals are likely based on the number of older, smaller dams that present liability issues. Dams that no longer serve a purpose, and are feasible to remove should be assed for the potential benefit their removal may yield for the conservation of riverine species, and at least one dam.

There are a limited number of studies of that investigate the costs and benefits of dam removals for freshwater mussels. The few that do exist have mainly highlighted the deleterious immediate impacts (Cooper 2001; Sethi et al. 2004). While, mussel species are particularly susceptible to disturbance, the removal of barriers to dispersal may lead to increased mussel species richness and abundance. There is a need for future research that examines the long-term recovery of freshwater mussels after dam removals in order to further evaluate their potential as a conservation strategy.

4.6. TABLES AND FIGURES

Table 4.1. List of potential metrics presented at BREW.

Dam Attributes	
Height	TCEQ 2014
Length	TCEQ 2015
Average Storage	TCEQ 2016
Age	TCEQ 2017
Connectivity Status Metrics	
Watershed Area	StreamCat (Hill et al. 2016)
Upstream Dam Density	StreamCat (Hill et al. 2016)
Downstream Dam Density	StreamCat (Hill et al. 2016)
Upstream Dam Density	StreamCat (Hill et al. 2016)
Upstream Dam Count	Calculated/BAT
Downstream Dam Count	Calculated/BAT
Stream Length Upstream	Calculated/BAT
Stream Length Downstream	Calculated/BAT
Connectivity Improvement Metrics	
Potential Recconected Stream Length	Calculated/BAT
Absolute Gain	Calculated/BAT
Relative Gain	Calculated/BAT
Ecological Metrics	
Fish	(Maxwell 2012, Hendrickson et al. 2015)
Endemic Fish	(Maxwell 2012, Hendrickson et al. 2015)
Mussels	Data presented in Dascher et al. 2017
Threatened Mussels	Data presented in Dascher et al. 2018
Upstream Mussels	Data presented in Dascher et al. 2019
Downstream Mussels	Data presented in Dascher et al. 2020
Threatened Upstream Mussels	Data presented in Dascher et al. 2021
Threateened Downstream Mussels	Data presented in Dascher et al. 2022
Flow Metrics	
Mean Yearly Discharge	EROM (Bondelid 2014)
Mean Monthly Discharge	EROM (Bondelid 2014)
Base Flow Index	StreamCat (Hill et al. 2016)
Water Quality Metrics	
Ammonium Ion Concentration	StreamCat (Hill et al. 2016)
Nitrate Ion Concentration	StreamCat (Hill et al. 2016)
Mean Pesticide Use in Watershed	StreamCat (Hill et al. 2016)
***Landscape Metrics**	
Kffactor	StreamCat (Hill et al. 2016)
Urban Land Cover	StreamCat (Hill et al. 2016)
Crop Land Cover	StreamCat (Hill et al. 2016)
Hay Land Cover	StreamCat (Hill et al. 2016)
Forest Land Cover	StreamCat (Hill et al. 2016)
Wetland Land Cover	StreamCat (Hill et al. 2016)
Mean Population Density	StreamCat (Hill et al. 2016)
Density of Roads	StreamCat (Hill et al. 2016)
Density of Road Stream Intersects	StreamCat (Hill et al. 2016)

Table. 4.2. Final list of metrics from BREW.

Metrics	Relative Importance (1-9)
Biological Metrics	**9**
Host Fish (HUC 8)	9
SGCN (HUC 8)	4
Mussels (HUC 8)	2
Federally Listed Mussels (HUC 8)	3
State Listed Mussels (HUC 8)	2
Potential/Candidate Mussels (HUC 8)	2
Upstream Downstream Mussels	3
Federally Listed US/DS Mussels	9
State Listed US/DS Mussels	7
Potential /Candidate US/DS Mussels	8
Connectivity Improvement Metrics	**8**
Potential Recconected Stream Length	9
Absolute Gain	5
Relative Gain	8
Water Quality Metrics	**7**
Ammonia Nitrogen Instream Concnetra	6
Nitatrate Nitrogen Instream Concentrati	3
High Stream Temperature	4
Low Stream Temperature	3
Mean Summer Stream Temperature	2
***Landscape Metrics**	**6**
Total ISA Land Cover	5
Undeveloped Land Cover	5
Riparian Buffer	7
Dam Attributes	**5**
Resevoir Length_Sorage	4
Average Storage	3
Resevoir Length	3

Table 4.3. Metrics and AHP weights for the BREW based dam removal prioritization model.

Criteria	Z *best*	Z *worst*	AHP Weight
Biological Metrics (d scores)	0.00	0.97	0.56
Host Fish	35.00	28.00	0.35
SGCN Fish	7.00	3.00	0.08
Mussels	11.00	9.00	0.05
State Listed Mussels	2.00	0.00	0.04
Potential/Candidate Mussels	2.00	0.00	0.03
Upstream Downstream Mussels	9.00	0.00	0.03
State Listed Upstream Downstream Mussels	2.00	0.00	0.21
Potential/Candidate Upstream Downstream Mussels	2.00	0.00	0.21
Connectivity Improvement Metrics (d scores)	0.36	1.00	0.32
Potential Reconnected Stream Length (km)	7271.28	0.18	0.57
Absolute Gain (km)	3078.794	0.009	0.10
Relative Gain (ratio)	0.50	0.00	0.33
Landscape Metrics (d scores)	0.31	0.95	0.12
Total ISA Land Cover (%)	0.00	98.57	0.19
Undeveloped Land Cover (%)	100.00	0.85	0.19
Riparian Buffer (%)	90.63	0.00	0.51
Reservoir Length _ Storage (ratio)	0.00	0.42	0.11

Table 4.4. Metrics and AHP weights for the literature dam removal prioritization model.

Criteria	*Z best*	*Z worst*	AHP Weight
Biological Metrics (d scores)	0.00	0.97	0.28
Host Fish	35.00	28.00	0.35
SGCN Fish	7.00	3.00	0.08
Mussels	11.00	9.00	0.05
State Listed Mussels	2.00	0.00	0.04
Potential/Candidate Mussels	2.00	0.00	0.03
Upstream Downstream Mussels	9.00	0.00	0.03
State Listed Upstream Downstream Mussels	2.00	0.00	0.21
Potential/Candidate Upstream Downstream Mussels	2.00	0.00	0.21
Connectivity Improvement Metrics (d scores)	0.36	1.00	0.15
Potential Reconnected Stream Length (km)	7271.28	0.18	0.57
Absolute Gain (km)	3078.79	0.009	0.10
Relative Gain (ratio)	0.50	0.00	0.33
Landscape Metrics (d scores)	0.31	0.95	0.10
Total ISA Land Cover (%)	0.00	98.57	0.20
Undeveloped Land Cover (%)	100.00	0.85	0.20
Riparian Buffer (%)	90.63	0.00	0.60
Dam Attributes (d scores)	0.32	0.99	0.47
Year of Completion	1800	2011	0.16
Owner (0,1)	1.00	0.00	0.30
Storage (af)	0.00	1129300.00	0.54

Table 4.5. Random Consistency Index (RI) values for Teknomo's method.

Number of Metrics	RI
1	0
2	0
3	0.58
4	0.9
5	1.12
6	1.24
7	1.32
8	1.41
9	1.45
10	1.49

Table 4.6. Comparison of model results using standard deviations from the mean to group dams into five removal priority categories.

Priority Rank		BREW Model		Literature Model	
Category	Standard Deviations from the Mean	d scores	Number of Dams	d scores	Number of Dams
Very High	> -2.5	0.31 - 0.84	2	0.63 - 0.78	15
High	-2.5 to -1.5	0.84 - 0.88	1	0.78 - 0.84	5
Mid-range	-1.5 to -0.5	0.88 - 0.93	34	0.84 - 0.90	24
Low	-0.5 to 0.5	0.93 - 0.98	195	0.90 - 0.96	109
Very Low	0.5 to 1.2	0.98 - 0.99	41	0.96 - 0.99	120

Table 4.7. Metric values for the top twenty dams prioritized for removal using the BREW model.

Priority Rank	Removal Category	d Score	Host Fish	SGCN Fish	Mussels	State Listed Mussels	Potential/Candidate Mussels	Upstream Downstream Mussels	State Listed Upstream Downstream Mussels	Potential/Candidate Upstream Downstream Mussels	Potential Reconnected Stream Length (km)	Absolute Gain (km)	Relative Gain	Total ISA Land Cover (%)	Undeveloped Land Cover (%)	Riparian Buffer (%)	Reservoir Length_Storage	Year of Completion	Owner (0,1)	Storage (af)
1	Very High	0.31	35	7	11	2	2	8	2	2	7271.28	3078.79	0.42	8.48	72.74	41.01	0.00	1898	1	808
2	Very High	0.72	35	7	11	2	2	7	2	2	4710.83	1632.03	0.35	9.04	77.21	39.48	0.00	1900	0	650
3	High	0.87	34	7	7	2	2	4	2	2	2426.29	794.26	0.33	8.50	78.06	40.17	0.36	1800	0	0
4	Mid-range	0.90	35	7	11	2	2	5	2	2	3080.75	1.95	0.00	6.94	41.85	53.13	0.01	1962	1	88
5	Mid-range	0.90	35	7	11	2	2	5	2	2	3079.64	0.85	0.00	3.22	35.00	24.90	0.00	1965	1	147
6	Mid-range	0.91	35	7	11	2	2	5	2	2	3082.75	3.95	0.00	12.45	43.39	13.92	0.00	1977	1	1163
7	Mid-range	0.91	35	7	11	2	2	5	2	2	3079.72	0.92	0.00	3.81	18.64	13.86	0.02	1962	1	80
8	Mid-range	0.91	35	7	11	2	2	5	2	2	3079.49	0.70	0.00	0.17	14.70	0.00	0.01	1960	1	104
9	Mid-range	0.91	35	7	11	2	2	8	1	1	4193.48	1.00	0.00	2.02	84.98	33.80	0.00	1962	1	576
10	Mid-range	0.91	35	7	11	2	2	8	1	1	4194.80	2.32	0.00	0.05	75.26	21.68	0.00	1800	1	182
11	Mid-range	0.92	35	7	11	2	2	8	1	1	4193.22	0.74	0.00	3.53	66.82	1.55	0.00	1961	1	336
12	Mid-range	0.92	35	7	11	2	2	8	1	1	4192.72	0.24	0.00	8.49	74.53	0.00	0.00	1960	1	107
13	Mid-range	0.92	35	7	11	2	2	8	1	1	4212.17	19.68	0.00	2.79	33.22	9.96	0.00	2002	1	2100
14	Mid-range	0.92	35	7	11	2	2	2	0	0	171.84	81.18	0.47	8.73	82.68	39.74	0.00	1914	1	1058
15	Mid-range	0.92	35	7	11	2	2	7	1	1	1722.70	90.67	0.05	8.67	81.99	39.68	0.00	1931	0	27450
16	Mid-range	0.92	35	7	11	2	2	1	0	0	17.75	7.50	0.42	5.08	81.82	57.98	0.01	1964	1	298
17	Mid-range	0.92	35	7	11	2	2	7	1	1	18.08	7.56	0.42	27.40	15.08	6.55	0.01	1962	1	106
18	Mid-range	0.92	35	7	11	2	2	8	1	1	1.26	0.27	0.21	2.77	93.99	0.00	0.01	1962	1	70
19	Mid-range	0.93	35	7	11	2	2	1	1	1	426.32	1.23	0.00	6.53	91.54	36.70	0.00	1964	0	1129300
20	Mid-range	0.93	35	7	11	2	2	1	0	0	45.26	17.28	0.38	26.58	73.12	60.08	0.01	1974	0	6911

Table 4.8. Metric values for the top twenty dams prioritized for removal using the literature model.

Priority Rank	Removal Category	d Score	Host Fish	SGCN Fish	Mussels	State Listed Mussels	Potential/Candidate Mussels	Upstream Downstream Mussels	State Listed Upstream Downstream Mussels	Potential/Candidate Upstream Downstream Mussels	Potential Reconnected Stream Length (km)	Absolute Gain (km)	Relative Gain	Total ISA Land Cover (%)	Undeveloped Land Cover (%)	Riparian Buffer (%)	Reservoir Length_Storage	Year of Completion	Owner (0,1)	Storage (af)
1	Very High	0.64	35	7	11	2	2	8	2	2	7271.28	3078.79	0.42	8.48	72.74	41.01	0.00	1898	1	808.00
2	Very High	0.70	31	6	0	0	0	1	0	0	0.18	0.02	0.11	0.17	99.83	58.56	0.02	1800	1	4.00
3	Very High	0.70	31	6	0	0	0	1	0	0	2180.10	0.16	0.00	0.17	99.83	58.56	0.02	1800	1	7.00
4	Very High	0.75	34	7	7	2	2	1	0	0	502.93	3.61	0.01	0.12	99.18	75.62	0.01	1800	1	85.00
5	Very High	0.76	35	7	11	2	2	8	1	1	4194.80	2.32	0.00	0.05	75.26	21.68	0.00	1800	1	182.00
6	Very High	0.76	35	7	11	2	2	1	0	0	429.03	3.94	0.01	2.79	97.21	60.50	0.30	1800	1	158.00
7	Very High	0.76	28	5	1	1	1	1	1	1	2866.74	82.91	0.03	2.05	97.92	50.68	0.01	1800	1	50.00
8	Very High	0.77	31	6	0	0	0	1	0	0	133.36	13.33	0.10	0.36	99.38	64.14	0.01	1800	1	65.00
9	Very High	0.77	33	3	4	1	1	5	1	1	4542.85	0.81	0.00	3.32	76.93	63.44	0.00	1800	1	148.00
10	Very High	0.77	31	6	0	0	0	1	0	0	192.69	19.02	0.10	0.30	99.61	38.11	0.02	1800	1	61.00
11	Very High	0.77	34	7	7	2	2	1	0	0	895.92	101.66	0.11	7.05	87.78	44.47	0.03	1800	1	150.00
12	Very High	0.78	28	5	1	1	1	1	0	0	353.58	104.27	0.29	1.06	98.83	40.21	0.01	1800	1	69.00
13	Very High	0.78	28	5	1	1	1	1	0	0	19.91	7.58	0.38	0.01	99.64	57.79	0.02	1800	1	140.00
14	Very High	0.78	31	6	0	0	0	1	0	0	2202.93	22.99	0.01	3.51	96.49	38.76	0.01	1800	1	100.00
15	Very High	0.78	30	3	0	0	0	3	0	0	2832.12	3.35	0.00	11.97	83.11	19.58	0.00	1800	1	72.00
16	High	0.79	28	5	1	1	1	1	0	0	356.32	19.90	0.06	0.00	100.00	66.21	0.00	1800	1	184.00
17	High	0.79	34	7	7	2	2	1	0	0	433.25	8.57	0.02	0.00	98.82	44.63	0.00	1800	1	420.00
18	High	0.79	28	5	1	1	1	1	0	0	15.49	0.53	0.03	9.21	90.03	50.03	0.05	1800	1	125.00
19	High	0.81	31	6	0	0	0	1	0	0	7.57	0.58	0.08	2.53	96.97	47.21	0.00	1800	1	627.00
20	High	0.81	31	6	0	0	0	1	0	0	174.25	0.58	0.00	2.53	96.97	47.21	0.02	1800	1	8.00

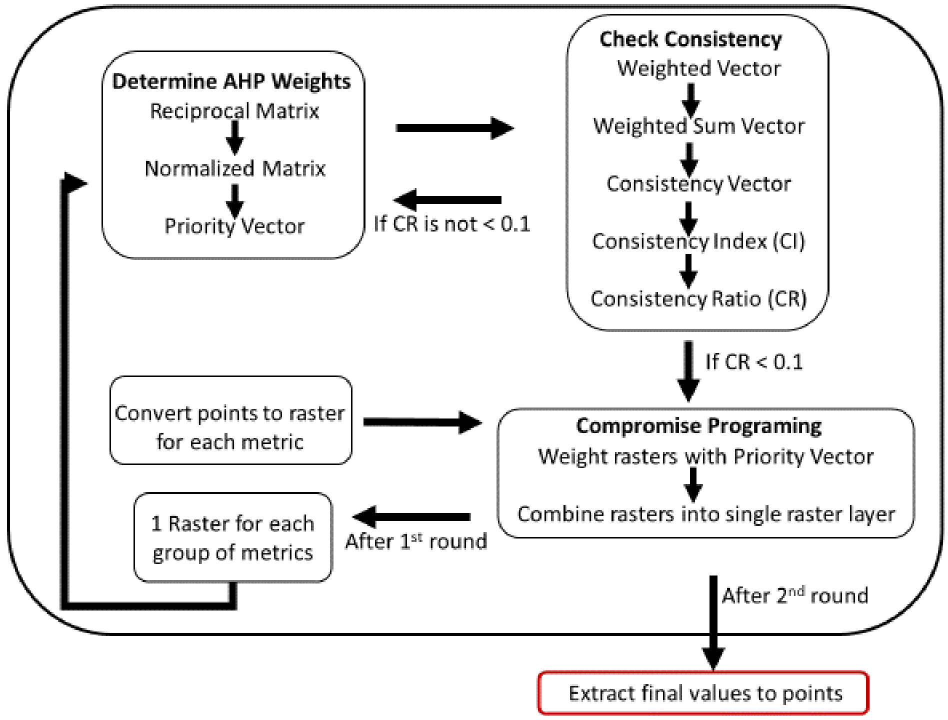

Figure 4.1. Flowchart of AHP and MCE methods used to create dam removal prioritization models.

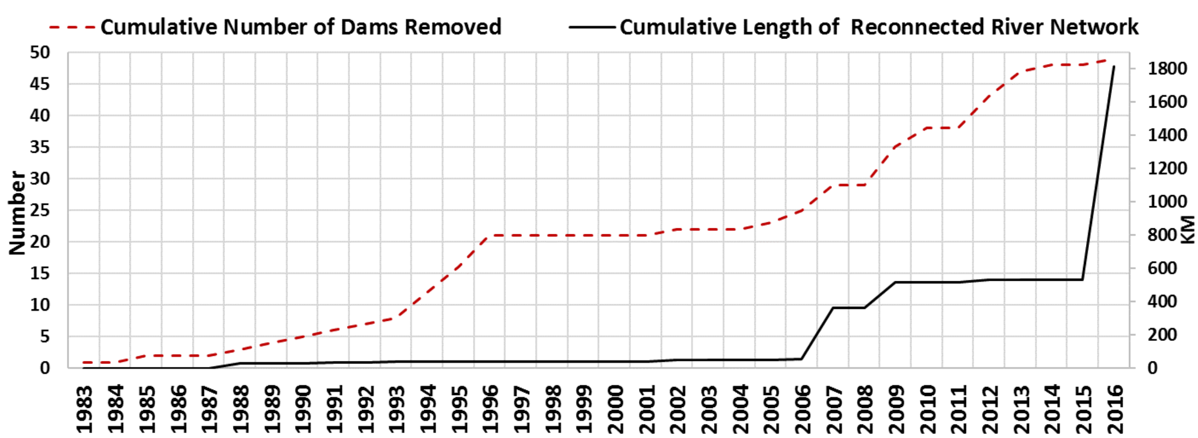

Figure 4.2. Cumulative number of dam removals in Texas and resulting functionally reconnected stream network (FRRN).

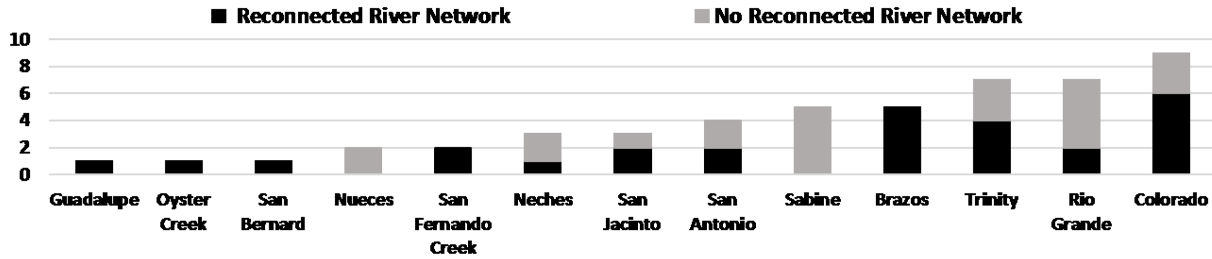

Figure 4.3. Number of dam removals by major river basin.

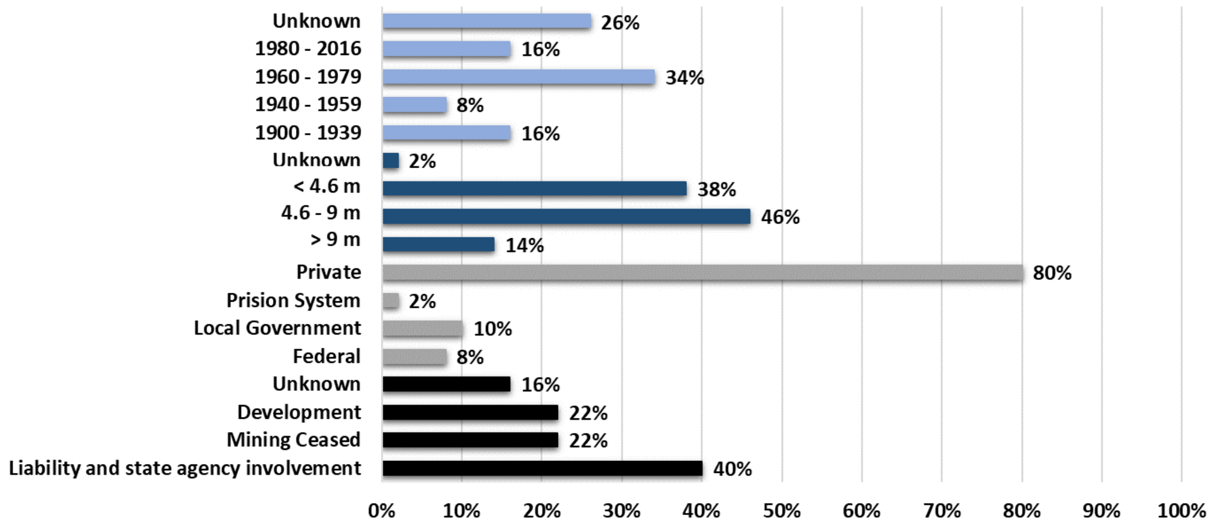

Figure 4.4. Percent of dam removals by time period of completion (relative age), height, owner, and reason for removal.

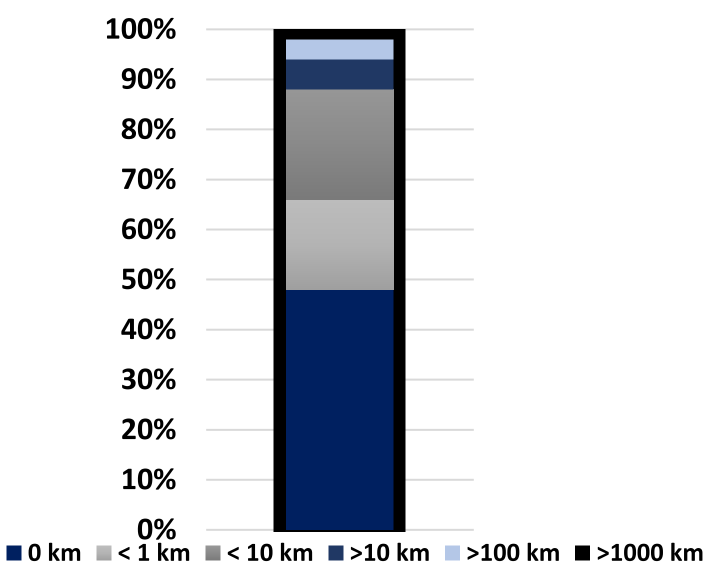

Figure 4.5. Number of dam removals by resulting length of FRRN.

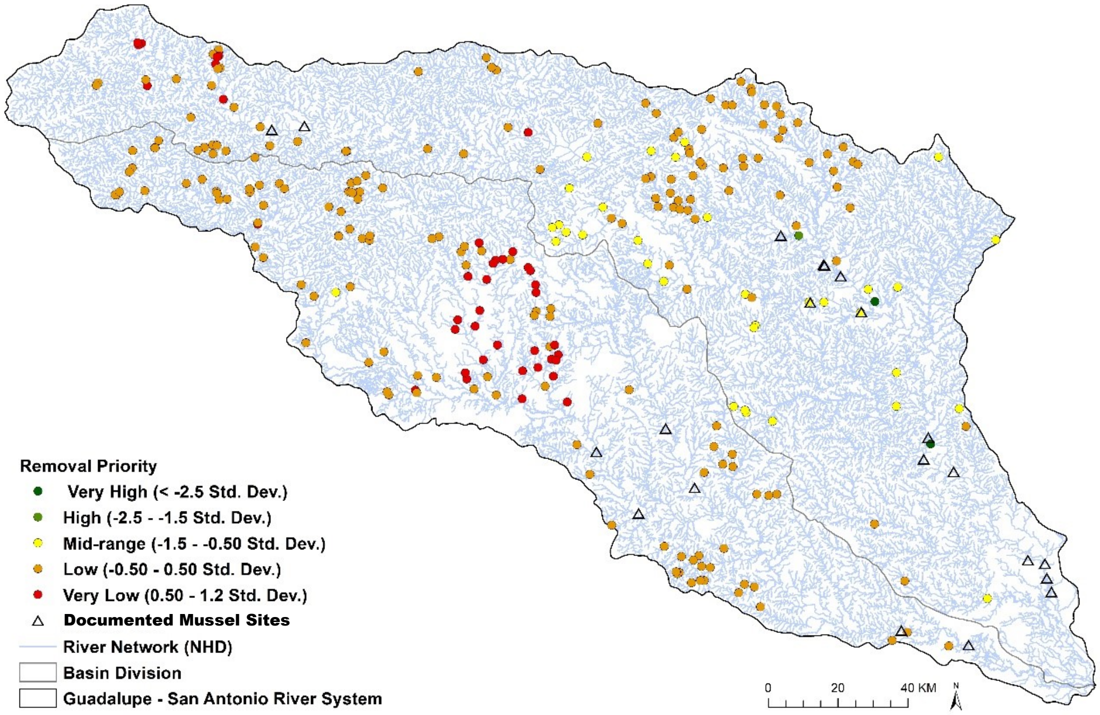

Figure 4.6. BREW model results using standard deviations from the mean d score to group dams into five removal priority categories.

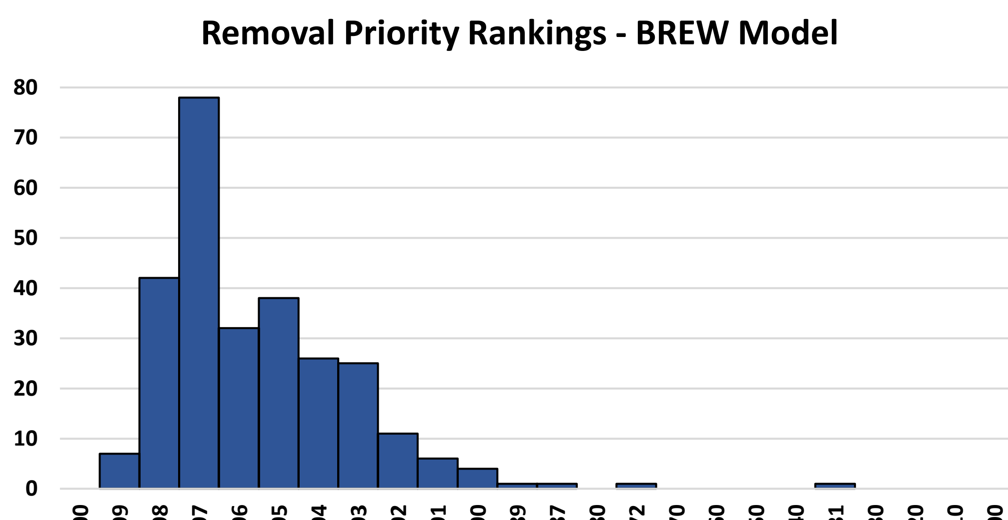

Figure 4.7. Dam removal priority rankings (d scores) for all dams in the GSARS for the BREW model.

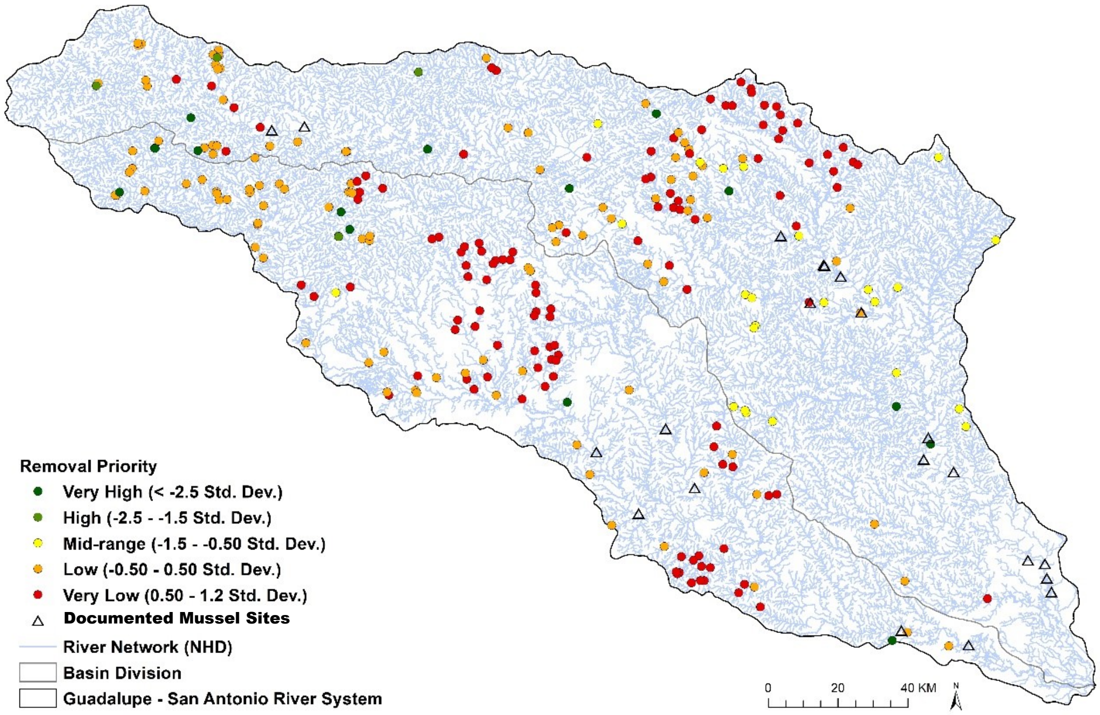

Figure 4.8. Literature model results using standard deviations from the mean d score to group dams into five removal priority categories.

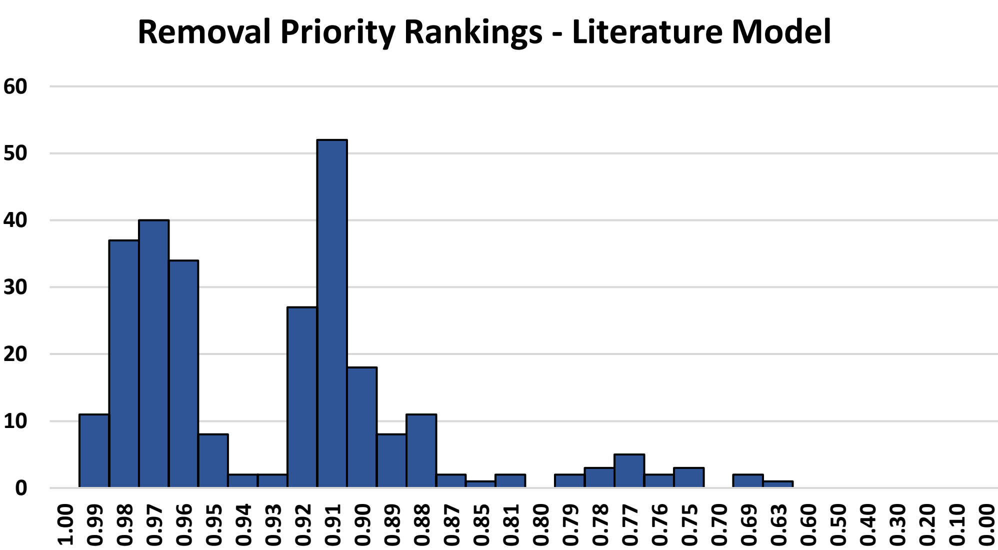

Figure 4.9 Dam removal priority rankings (d scores) for all dams in the GSARS for the Literature model.

5. CONCLUSIONS

This research examines the impacts of dams on longitudinal connectivity and the current distribution of freshwater mussels in Texas and the GSARS. Two dam removal prioritization models and recommendations for future freshwater mussel conservation are presented. Additional research is needed to further investigate the relationship between river fragmentation and freshwater mussel distribution and community composition. Removing a dam restores river connectivity, but may not always be the most viable option for freshwater mussel conservation.

Fragmentation is considered a leading cause of freshwater mussel declines, but there is an absence of research examining the specific impacts of fragmentation on freshwater mussel distribution and community composition. While the commissioning of a dam will often result in the abrupt decline and/or extirpation of freshwater mussel species directly up and downstream, fragmentation of river habitat due to multiple dams may result in less immediate or perceptible declines. Fragmentation inhibits the migration of aquatic species, and may result in either the separation of mussel species and their host fish and/or the separation of source populations of freshwater mussels. This in turn may result in the reduction of freshwater mussel dispersal, and limit the future recruitment of young mussels. Freshwater mussel communities may then age and decline without any new recruitment.

The connectivity models presented in this research highlight how river systems become increasingly fragmented over time due to dams. Future research can incorporate such connectivity models with historic and current data on freshwater mussels to examine how fragmentation has impacted their distribution and community composition. Fragmentation likely leads to older, isolated populations of mussels, and declines in species richness and abundance. However, these changes in freshwater mussel distribution and community composition may require multiple datasets that include detailed information on mussel age and past distribution patterns to be adequately investigated.

Dam removal is promoted in this research as a way to restore river connectivity and conserve freshwater mussel populations, but it is often associated with a negative connotation in Texas. An alternative approach to restore some of the functions of river connectivity and avoid the political tensions around dam removal, is to retro fit existing dams with fish passage. Fish passage provides a mechanism for allowing fish migration without completely removing a dam, and represents another strategy to restore or conserve freshwater mussel populations in Texas.

Fish passage may be considered a more viable conservation strategy for freshwater mussels compared to removal in Texas, particularly for larger dams that are important components of the state's flood control and water supply systems. The BREW model prioritizes dam removal for freshwater mussel conservation solely on the potential benefit to freshwater mussel populations. Since it does not incorporate metrics of dam removal feasibility, an alternative interpretation of this model is the prioritization of dams for fish passage improvements rather than removal. Canyon Dam, for example, is an unlikely candidate for removal, but is one of the top twenty dams prioritized in the BREW model. Shifting the focus of the model to fish passage, means that this dam should be considered for connectivity improvement measures rather than removal. If not already present, fish passage could be added to increase host fish migration and connectivity between mussel populations up- and downstream of Canyon Dam.

Better record keeping of dams and dam removals in general has been previously promoted in the literature. Existing state and national databases of dams should begin to include information on existing fish passage. The inclusion of this information would further improve connectivity models, and allow for an analysis of the actual hindrance of individual dams to species migration and dispersal. Cuero Lake Dam demonstrates the importance of maintaining better records on breeches and dam status, so that structures still listed in current databases can be more accurately evaluated in terms of their impacts to connectivity. In addition to the creation of a national inventory of dam removals, previously recommended in Chapter IV, this research highlights the need for similar national databases on instream barriers such as unregistered dams and road crossings. Such databases would allow for more accurate analyses of river connectivity and research on freshwater mussel conservation.

There are multiple researchers and agencies currently studying freshwater mussel distribution and conservation strategies throughout Texas. Researchers should consider combining existing ecological datasets where ever possible. This would enable researchers to more accurately determine the current and historical state of freshwater mussels in Texas and the driving forces and/or controls on their distribution and community composition. Without such collaborations, research over broad scales will continue to be plagued with issues of data availability and limited in their capacity to inform freshwater mussel management strategies.

APPENDIX SECTION

Appendix 1.1. Known Freshwater Mussel Species of Texas.

Scientific Name	Common Name	State Status	Federal Status	Endemic (0/1)	Guadalupe - San Antonio River System (0/1)
Amblema plicata	Threeridge	None	None	0	1
Andonata Imbecillis	Paper Pondshell	None	None	0	0
Andonata suborbiculata	Flat Floater	None	None	0	0
Arcidens confragosus	Rock-Pocketbook	None	None	0	1
Cyrtonaias tampicoensis	Tampico Pearlymussel	None	None	0	1
Elliptio dilatata	Spike	None	None	0	0
Fusconaia askewi	Texas Pigtoe	Threatened	None	1	0
Fusconaia flava	Wabash Pigtoe	None	None	0	0
Fusconaia lananensis	Triangle Pigtoe	Threatened	Petitioned	1	0
Quadrula (Fusconaia) mitchelli	Falsespike	Threatened	Petitioned	1	0
Glebula Rotundata	Round Pearlshell	None	None	0	0
Lampsilis bracteata	Texas Fatmucket	Threatened	Candidate	1	0
Lampsilis cardium	Plain Pocketbook	None	None	0	0
Lampsilis hydiana	Louisiana Fatmucket	None	None	0	1
Lampsilis satura	Sandbank Pocketbook	Threatened	None	1	0
Lampsilis teres	Yellow Sandshell	None	None	0	1
Lasmigona complanata	White Heelsplitter	None	None	0	0
Leptodea fragilis	Fragile Papershell	None	None	0	0
Logumia subrostrata	Pond Mussel	None	None	0	0
Megalonaias nervosa	Washboard	None	None	0	1
Obliquaria reflexa	Threehorn Watryback	None	None	0	0
Obovaria jacksoniana	Southern Hickorynut	Threatened	None	0	0
Plectomerus dombeyanus	Bankclimber	None	None	0	1
Pleurobema riddelli	Louisiana Pigtoe	Threatened	Petitioned	1	0
Popenaias popei	Texas Hornshell	Threatened	Candidate	1	0
Potamilus amphichaenus	Texas Heelsplitter	Threatened	Petitioned	1	0
Potamilus metnecktayi	Salina Mucket	Threatened	Petitioned	1	0
Potamilus ohiensis	Pink Papershell	None	None	0	0
Potamilus purpuratus	Bleufer	None	None	0	0
Anodonta (Pyganodon) grandis	Giant Floater	None	None	0	1
Quadrula apiculata	Soutehrn Mapleleaf	None	None	0	1
Quadrula aurea	Golden Orb	Threatened	Candidate	1	1
Quadrula couchiana	Rio Grande Monkeyface	None	None	0	0
Quadrula houstonensis	Smooth Pimpleback	Threatened	Candidate	1	0
Quadrula mortoni	Western Pimpleback	None	None	0	0
Quadrula nodulata	Wartyback	None	None	0	0
Quadrula petrina	Texas Pimpleback	Threatened	Candidate	1	1
Quadrula pustulosa	Pimpleback	None	None	0	0

Quadrula quadrula	Mapleleaf	None	None	0	0
Strophitus undulatus	Squawfoot	None	None	0	0
Toxolasma parvus	Lilliput	None	None	0	0
Toxolasma texasensis	Texas Lilliput	None	None	0	0
Tritogonia verrucosa	Pistolgrip	None	None	0	1
Truncilla cognata	Mexican Fawnsfoot	Threatened	Petitioned	1	0
Truncilla donaciformis	Fawnsfoot	None	None	0	0
Truncilla macrodon	Texas Fawnsfoot	Threatened	Candidate	1	0
Truncilla truncata	Deertoe	None	None	0	0
Uniomerus declivis	Tapered Pondhorn	None	None	0	0
Uniomerus tetralasmus	Pondhorn	None	None	0	1
Villosa lienosa	Little Spectaclecase	None	None	0	0

Appendix 2.1. Dams in Texas and ten major river basins by size and time period. *

	1800 - 1899		1900 - 1939		1940 -1959		1960 - 1979		1980 - 2014	
	N	%	N	%	N	%	N	%	N	%
Texas										
Small	143	49.5	64	12.3	180	13.0	774	18.6	218	26.8
Medium	143	49.5	415	80.0	1134	82.1	3296	79.3	556	68.4
Large	3	1.0	37	7.1	59	4.3	72	1.7	34	4.2
Very Large	0	0.0	3	0.6	9	0.7	12	0.3	5	0.6
Total	289	100.0	519	100.0	1382	100.0	4154	100.0	813	100.0
Trinity										
Small	35	60.3	9	13.0	25	6.8	177	16.4	63	33.9
Medium	22	37.9	52	75.4	332	90.2	893	82.7	115	61.8
Large	1	1.7	8	11.6	10	2.7	8	0.7	6	3.2
Very Large	0	0.0	0	0.0	1	0.3	2	0.2	2	1.1
Total	58	100.0	69	100.0	368	100.0	1080	100.0	186	100.0
Brazos										
Small	24	47.1	8	8.2	28	13.1	168	20.2	31	16.9
Medium	27	52.9	81	82.7	172	80.4	644	77.3	145	79.2
Large	0	0.0	9	9.2	11	5.1	19	2.3	7	3.8
Very Large	0	0.0	0	0.0	3	1.4	2	0.2	0	0.0
Total	51	100.0	98	100.0	214	100.0	833	100.0	183	100.0
Colorado										
Small	16	57.1	11	18.0	13	9.2	71	15.3	19	29.2
Medium	12	42.9	45	73.8	116	82.3	383	82.7	40	61.5
Large	0	0.0	3	4.9	11	7.8	8	1.7	5	7.7
Very Large	0	0.0	2	3.3	1	0.7	1	0.2	1	1.5
Total	28	100.0	61	100.0	141	100.0	463	100.0	65	100.0
Red										
Small	4	66.7	6	11.5	17	21.8	100	24.8	19	24.7
Medium	2	33.3	38	73.1	57	73.1	293	72.7	57	74.0
Large	0	0.0	7	13.5	3	3.8	10	2.5	1	1.3
Very Large	0	0.0	1	1.9	1	1.3	0	0.0	0	0.0
Total	6	100.0	52	100.0	78	100.0	403	100.0	77	100.0
Nueces										
Small	4	23.5	2	9.5	19	21.3	51	16.7	1	8.3
Medium	13	76.5	19	90.5	68	76.4	255	83.3	9	75.0
Large	0	0.0	0	0.0	2	2.2	0	0.0	1	8.3
Very Large	0	0.0	0	0.0	0	0.0	0	0.0	1	8.3
Total	17	100.0	21	100.0	89	100.0	306	100.0	12	100.0
Sabine										
Small	8	50.0	5	11.1	29	29.3	18	15.0	13	36.1
Medium	8	50.0	40	88.9	67	67.7	95	79.2	21	58.3
Large	0	0.0	0	0.0	3	3.0	5	4.2	1	2.8
Very Large	0	0.0	0	0.0	0	0.0	2	1.7	1	2.8
Total	16	100.0	45	100.0	99	100.0	120	100.0	36	100.0

Rio Grande										
Small	14	46.7	3	7.9	4	7.5	22	13.2	5	12.5
Medium	16	53.3	33	86.8	44	83.0	141	84.4	32	80.0
Large	0	0.0	2	5.3	4	7.5	3	1.8	3	7.5
Very Large	0	0.0	0	0.0	1	1.9	1	0.6	0	0.0
Total	30	100.0	38	100.0	53	100.0	167	100.0	40	100.0
Neches										
Small	12	60.0	6	14.0	12	13.3	24	7.8	12	35.3
Medium	8	40.0	37	86.0	74	82.2	84	27.5	21	61.8
Large	0	0.0	0	0.0	4	4.4	5	1.6	1	2.9
Very Large	0	0.0	0	0.0	0	0.0	2	0.7	0	0.0
Total	20	100.0	43	100.0	90	100.0	306	100.0	34	100.0
Guadalupe										
Small	11	55.0	3	23.1	6	28.6	23	17.7	7	29.2
Medium	9	45.0	7	53.8	15	71.4	106	81.5	14	58.3
Large	0	0.0	3	23.1	0	0.0	0	0.0	3	12.5
Very Large	0	0.0	0	0.0	0	0.0	1	0.8	0	0.0
Total	20	100.0	13	100.0	21	100.0	130	100.0	24	100.0
San Antonio										
Small	9	81.8	2	25.0	4	10.8	11	13.9	3	13.6
Medium	2	18.2	4	50.0	33	89.2	65	82.3	19	86.4
Large	0	0.0	2	25.0	0	0.0	3	3.8	0	0.0
Very Large	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0
Total	11	100.0	8	100.0	37	100.0	79	100.0	22	100.0

*Note: Dams without year complete and/or a maximum storage of zero listed were omitted.

Appendix 2.2. Storage and other attributes by dam size.

	Total Reservoir Storage (x10^{9} m^{3})	% of Total Storage	Total Number of Dams	Dam Density (dam/km^{2})	Area per Dam (km^{2}/dam)
*Texas (2005)					
Small	0.113	0.1	1368	0.002	500
Medium	6.91	5.9	5446	0.008	127
Large	36.5	31.4	212	0.000	3333
Extra Large	72.7	62.5	27	0.000	25641
Texas (2014)					
Small	0.111	0.1	1452	0.002	479
Medium	6.92	5.4	5588	0.008	124
Large	32.3	25.1	205	0.000	3393
Extra Large	89.3	69.4	29	0.000	23987
Trinity					
Small	0.02	0.1	323	0.007	144
Medium	1.62	7.7	1426	0.031	33
Large	7.90	37.7	33	0.001	1412
Extra Large	114	54.5	5	0.000	9317
Brazos					
Small	0.02	0.1	267	0.002	417
Medium	13.9	7.6	1074	0.010	104
Large	7.73	42.3	46	0.000	2423
Extra Large	9.11	49.9	5	0.000	22292
Colorado					
Small	0.01	0.1	144	0.001	712
Medium	10.6	7.1	599	0.006	171
Large	4.40	29.3	27	0.000	3799
Extra Large	9.55	63.6	5	0.000	20515
Red					
Small	0.01	0.1	147	0.002	429
Medium	0.51	3.3	449	0.007	140
Large	2.92	19.0	21	0.000	3001
Extra Large	11.9	77.6	2	0.000	31514
Nueces					
Small	0.01	0.3	87	0.002	499
Medium	0.21	9.5	365	0.008	119
Large	0.69	30.8	3	0.000	14460
Extra Large	1.34	59.4	1	0.000	43380

Sabine					
Small	0.00001	0.1	82	0.007	144
Medium	0.0002	2.3	241	0.031	33
Large	0.001	6.2	9	0.001	1412
Extra Large	0.008	91.5	3	0.000	9317
Rio Grande					
Small	0.000003	0.0	48	0.002	417
Medium	0.0004	3.5	266	0.010	104
Large	0.0007	6.8	12	0.000	2423
Extra Large	0.01	89.7	2	0.000	22292
Neches					
Small	0.000004	0.0	67	0.001	712
Medium	0.0001	1.7	229	0.006	171
Large	0.001	10.0	10	0.000	3799
Extra Large	0.001	88.3	2	0.000	20515
Guadalupe					
Small	0.000003	0.2	56	0.002	429
Medium	0.0002	12.9	152	0.007	140
Large	0.0002	15.2	6	0.000	3001
Extra Large	0.001	71.7	1	0.000	31514
San Antonio					
Small	0.000002	0.2	30	0.002	499
Medium	0.0002	28.9	125	0.008	119
Large	0.0005	70.9	5	0.000	14460
Extra Large	0	0	0	0.000	43380

*Note: Values for Texas (2005) are borrowed from Chin at el. 2008, Tabel 3, p.245.

Appendix 2.3. Ownership by dam size.

	Small		Medium		Large		Extra Large		Total	
	N	%	N	%	N	%	N	%	N	%
Texas										
Federal	22	1.5	49	0.9	26	12.7	16	55.2	113	1.6
State	9	0.6	56	1.0	9	4.4	4	13.8	78	1.1
Local government	190	13.1	2194	39.3	124	60.5	8	27.6	2516	34.6
Private	1152	79.7	3150	56.4	44	21.5	1	3.4	4347	59.8
Other	27	1.9	83	1.5	2	1.0	0	0.0	112	1.5
Not Listed	15	1.0	56	1.0	0	0.0	0	0.0	71	1.0
Trinity										
Federal	1	0.3	2	0.1	7	21.2	2	40.0	12	0.7
State	1	0.3	25	1.8	3	9.1	2	40.0	31	1.7
Local government	85	26.5	929	65.1	16	48.5	1	20.0	1031	57.7
Private	221	68.8	433	30.4	7	21.2	0	0.0	661	37.0
Other	9	2.8	25	1.8	0	0.0	0	0.0	34	1.9
Not Listed	4	1.2	12	0.8	0	0.0	0	0.0	16	0.9
Brazos										
Federal	16	6.0	25	2.3	5	10.9	4	80.0	50	3.6
State	2	0.8	7	0.7	1	2.2	0	0.0	10	0.7
Local government	24	9.0	352	32.8	28	60.9	1	20.0	405	29.1
Private	214	80.5	658	61.3	11	23.9	0	0.0	883	63.4
Other	3	1.1	11	1.0	1	2.2	0	0.0	15	1.1
Not Listed	1	0.4	21	2.0	0	0.0	0	0.0	22	1.6
Colorado										
Federal	3	2.1	2	0.3	2	7.4	1	20.0	8	1.0
State	0	0.0	3	0.5	0	0.0	0	0.0	3	0.4
Local government	26	18.3	386	64.4	23	85.2	3	60.0	438	56.5
Private	101	71.1	202	33.7	2	7.4	1	20.0	306	39.5
Other	8	5.6	6	1.0	0	0.0	0	0.0	14	1.8
Not Listed	2	1.4	0	0.0	0	0.0	0	0.0	2	0.3
Red										
Federal	1	0.7	5	1.1	4	19.0	1	50.0	11	1.8
State	2	1.4	5	1.1	0	0.0	0	0.0	7	1.1
Local government	10	6.8	160	35.6	13	61.9	1	50.0	184	29.7
Private	132	89.8	264	58.8	4	19.0	0	0.0	400	64.6
Other	1	0.7	13	2.9	0	0.0	0	0.0	14	2.3
Not Listed	1	0.7	2	0.4	0	0.0	0	0.0	3	0.5
Nueces										
Federal	0	0.0	1	0.3	0	0.0	1	100.0	2	0.4
State	2	2.6	0	0.0	0	0.0	0	0.0	2	0.4
Local government	2	2.6	15	4.1	3	100.0	0	0.0	20	4.5
Private	73	93.6	344	94.2	0	0.0	0	0.0	417	93.3
Other	1	1.3	4	1.1	0	0.0	0	0.0	5	1.1
Not Listed	0	0.0	1	0.3	0	0.0	0	0.0	1	0.2

Sabine										
Federal	0	0.0	1	0.4	0	0.0	0	0.0	1	0.3
State	0	0.0	1	0.4	0	0.0	2	66.7	3	0.9
Local government	3	3.9	30	12.4	5	55.6	1	33.3	39	11.9
Private	73	96.1	202	83.8	4	44.4	0	0.0	279	84.8
Other	0	0.0	3	1.2	0	0.0	0	0.0	3	0.9
Not Listed	0	0.0	4	1.7	0	0.0	0	0.0	4	1.2
Rio Grande										
Federal	0	0.0	2	0.8	2	16.7	2	100.0	6	1.8
State	0	0.0	3	1.1	0	0.0	0	0.0	3	0.9
Local government	12	25.0	77	28.9	8	66.7	0	0.0	97	29.6
Private	36	75.0	182	68.4	2	16.7	0	0.0	220	67.1
Other	0	0.0	2	0.8	0	0.0	0	0.0	2	0.6
Not Listed	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0
Neches										
Federal	0	0.0	2	0.9	1	10.0	1	50.0	4	1.3
State	0	0.0	3	1.3	0	0.0	0	0.0	3	1.0
Local government	3	4.5	32	14.0	9	90.0	1	50.0	45	14.7
Private	60	90.9	190	83.0	0	0.0	0	0.0	250	81.4
Other	1	1.5	2	0.9	0	0.0	0	0.0	3	1.0
Not Listed	2	3.0	0	0.0	0	0.0	0	0.0	2	0.7
Guadalupe										
Federal	0	0.0	0	0.0	0	0.0	1	100.0	1	0.5
State	2	3.7	4	2.6	4	66.7	0	0.0	10	4.7
Local government	10	18.5	55	36.2	2	33.3	0	0.0	67	31.5
Private	39	72.2	86	56.6	0	0.0	0	0.0	125	58.7
Other	1	1.9	5	3.3	0	0.0	0	0.0	6	2.8
Not Listed	2	3.7	2	1.3	0	0.0	0	0.0	4	1.9
San Antonio										
Federal	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0
State	0	0.0	0	0.0	0	0.0	0	0.0	0	0.0
Local government	2	6.9	59	47.2	5	100.0	0	0.0	66	41.5
Private	25	86.2	61	48.8	0	0.0	0	0.0	86	54.1
Other	0	0.0	5	4.0	0	0.0	0	0.0	5	3.1
Not Listed	2	6.9	0	0.0	0	0.0	0	0.0	2	1.3

*Note: Dams with a maximum storage of zero listed not included.

Appendix 3.1. Average abundance of mussel species for major river basins.

Scientific Name	Brazos	Colorado	Cypress	Guadalupe	Neches	Nueces	Rio Grande	Sabine	San Antonio	Trinity
Amblema plicata	14	29	7.7	13	2.3	0	0	0.4	0.6	1.3
Amblema plicata	0	0.2	1	0	0.4	0	0	0	0	0
Arcidens confragosus	0.03	0	0	0.01	0.1	0	0	0.4	0	0
Arcidens confragosus	0	0	0	0.6	5.5	0	0	13	0	1.1
Arkinsia wheeleri	0	0	0	0	0.1	0	0	0	0	0
Arkinsia wheeleri	0.04	0	0	0	0	0	0.1	0.02	0	0
Cyrtonaias tampicoensis	5.1	0.4	0	0.9	0	0	8.7	0	0.1	0
Cyrtonaias tampicoensis	0	0	0	0.02	0.2	0	0	0	0	0.2
Fusconaia askewi	0	0	0.7	0	5.5	0	0	9.4	0	1
Fusconaia askewi	0	0	0	5.6	0	3	0	0	5.5	0
Fusconaia lananensis	0	0	0	0	1.8	0	0	0	0	0
Fusconaia lananensis	4.8	3.7	0	0	0	0	0	0	0	0
Fusconaia mitchelli	0	0	1	0	14	0	0	2.5	0	4.2
Fusconaia mitchelli	0	0	0	0	0.2	0	0	0	0	1.6
Glebula Rotundata	0	0	0	0	0.5	0	0	0	0	0
Lampsilis bracteata	0	0.3	0	0	0	0	0	0	0	0
Lampsilis hydiana	0	0	0	0.3	0.9	0	0	0.3	0	0
Lampsilis satura	0	0	0	0	0.4	0	0	0.4	0	0
Lampsilis teres	3.5	0.2	12	0.9	0.8	4	0.2	5.1	18	2.7
Leptodea fragilis	0.2	0.1	0	0	0.4	0	0	1.8	0	1.7
Logumia subrostrata	0	0	1.3	0	0	0	0	0	0	0
Megalonaias nervosa	0.2	0	0.7	1.7	6.3	0	0.1	1.9	0	0.8
Obliquaria reflexa	0	0	0	0	3.7	0	0	2	0	6.3
Obovaria jacksoniana	0	0	0	0	0.1	0	0	0	0	0
Plectomerus dombeyanus	0	0	21	0.04	2.4	0	0	1.2	0	0.1
Pleurobema riddelli	0	0	0	0	2.1	0	0	0	0	0
Popenaias popei	0	0	0	0	0	0	8.7	0	0	0
Potamilus amphichaenus	0	0	0	0	0	0	0	0.5	0	0.1
Potamilus metnecktayi	0	0	0	0	0	0	1	0	0	0
Potamilus ohiensis	0.1	0	0	0	0	0	0	0	0	0
Potamilus purpuratus	0	0	0	0	1.1	0	0.4	2.4	0	2.6
Pyganodon grandis	0	0	1.7	0.1	0.2	0	0	0.3	0	1.1
Quadrula apiculata	1	0.1	0	0.1	1	0	2	4.2	0	1.9
Quadrula aurea	14	29	7.7	13	2.3	0	0	0.4	0.6	1.3
Quadrula houstonensis	0.03	0	0	0.01	0.1	0	0	0.4	0	0
Quadrula mortoni	0	0	0	0	0.1	0	0	0	0	0
Quadrula nodulata	5.1	0.4	0	0.9	0	0	8.7	0	0.1	0

Quadrula petrina	0	0	0.7	0	5.5	0	0	9.4	0	1
Quadrula quadrula	0	0	0	0	1.8	0	0	0	0	0
Scientific Name	0	0	0	0	0.02	0	0	0	0	0
Strophitus undulatus	0	0	0	5.6	0	3	0	0	5.5	0
Toxolasma parvus	4.8	3.7	0	0	0	0	0	0	0	0
Toxolasma texasensis	0	0	0	0	0.2	0	0	0	0	1.6
Tritogonia verrucosa	0	0	0	0.3	0	0	0	0	0	0
Truncilla cognata	0	0	7.3	0	0	0	0	0	0	0
Truncilla donaciformis	0	0	0	0	2.9	0	0	2.8	0	0.5
Truncilla macrodon	0	0	0	0	0	0	0	0	0	0.1
Truncilla truncata	0.04	0.7	0	0	0	0	0	0	0	0
Uniomerus declivis	0	0	0	0	0.02	0	0	0.01	0	0
Uniomerus tetralasmus	0	0	0	0	0.1	0	0	0	0	0
Villosa lienosa	0	0	0	0	0	0	0.2	0	0	0

Appendix 4.1. Approved IRB Packet.

TEXAS STATE UNIVERSITY
SAN MARCOS
The rising STAR of Texas

In future correspondence please refer to 2017384

February 24, 2017

Erin Dascher
Texas State University
601 University Drive.
San Marcos, TX 78666

Dear Ms. Dascher:

Your IRB application 2017384 titled "Barrier Removal Prioritization Workshop." was reviewed and approved by the Texas State University IRB. It has been determined that risks to subjects are: (1) minimized and reasonable; and that (2) research procedures are consistent with a sound research design and do not expose the subjects to unnecessary risk. Reviewers determined that: (1) benefits to subjects are considered along with the importance of the topic and that outcomes are reasonable; (2) selection of subjects is equitable; and (3) the purposes of the research and the research setting is amenable to subjects' welfare and producing desired outcomes; that indications of coercion or prejudice are absent, and that participation is clearly voluntary.

1. In addition, the IRB found that you need to orient participants as follows: (1) signed informed consent is required; (2) Provision is made for collecting, using and storing data in a manner that protects the safety and privacy of the subjects and the confidentiality of the data; (3) Appropriate safeguards are included to protect the rights and welfare of the subjects.

This project is therefore approved at the Exempt Review Level

2. Please note that the institution is not responsible for any actions regarding this protocol before approval. If you expand the project at a later date to use other instruments please re-apply. Copies of your request for human subjects review, your application, and this approval, are maintained in the Office of Research Integrity and Compliance. Please report any changes to this approved protocol to this office.

Sincerely,

Monica Gonzales

Monica Gonzales
IRB Regulatory Manager
Office of Research Integrity and Compliance

CC: Dr. Kimberly Meitzen

OFFICE OF THE ASSOCIATE VICE PRESIDENT FOR RESEARCH
601 University Drive | JCK #489 | San Marcos, Texas 78666-4616
Phone: 512.245.2314 | *fax:* 512.245.3847 | WWW.TXSTATE.EDU

This letter is an electronic communication from Texas State University-San Marcos, a member of The Texas State University System.

Recruitment Email Message Template

To: Expert
From: Erin D. Dascher (edd1@txstate.edu)
BCC:
Subject: Research Participation Invitation: Barrier Removal Prioritization Workshop

This email message is an approved request for participation in research that has been approved or declared exempt by the Texas State Institutional Review Board (IRB).

As part of my dissertation research I am hosting a barrier removal prioritization workshop, at the Meadows Center for Water and the Environment on March 2, 2017 from 9 am – 4 pm that will bring together up to 30 experts in the field together to discuss the criterion to be used for dam removal prioritization in the Guadalupe – San Antonino River System. Criteria will be chosen and weighted by importance for inclusion in a dam removal prioritization model.

Participants will receive a free lunch and an Amazon Gift card for attending the workshop.

To participate in this research or ask questions about this research please contact me at Erin D. Dascher, 512-245-4962, edd1@txstate.edu

This project 2017384 was approved by the Texas State IRB on February 24, 2017. Pertinent questions or concerns about the research, research participants' rights, and/or research-related injuries to participants should be directed to the IRB chair, Dr. Jon Lasser 512-245-3413 – (lasser@txstate.edu) or to Monica Gonzales, IRB administrator 512-245-2314 - (meg201@txstate.edu).

TEXAS STATE
Exempt Review
2/24/2017

TEXAS STATE

INFORMED CONSENT

Study Title: Barrier Removal Prioritization Workshop
Principal Investigator: Erin D. Dascher **Co-Investigator/Faculty Advisor: Dr. Kimberly Meitzen**
Sponsor: The Graduate College, Texas State University

This consent form will give you the information you will need to understand why this research study is being done and why you are being invited to participate. It will also describe what you will need to do to participate as well as any known risks, inconveniences or discomforts that you may have while participating. We encourage you to ask questions at any time. If you decide to participate, you will be asked to sign this form and it will be a record of your agreement to participate. You will be given a copy of this form to keep.

PURPOSE AND BACKGROUND

You are invited to participate in a research study to learn more about criteria for dam removals in the Guadalupe – San Antonio River System. The information gathered will be used as part of dissertation research designed to create a dam removal prioritization model. You are being asked to participate because you have been identified as an expert in the field via your occupation, publication history, or association with a particular government agency, non-profit organization, or other associated business.

PROCEDURES

We will invite 30 people to meet together to discuss their opinions on what criteria should be included, and how important individual criteria are, when determining the priority of dams in the Guadalupe – San Antonio River System for removal. The discussion topics include criteria of dam removal, i.e. dam attributes, habitat quality, and river biota/species. A member of the research team will help guide the discussion. To protect the privacy of focus group members, all transcripts will be coded with pseudonyms and we ask that you not discuss what is discussed in the focus group with anyone else. The focus group will last about eight hours.

RISKS/DISCOMFORTS

There are no foreseeable risks to participants. We will make every effort to protect participants' confidentiality. However, if you are uncomfortable answering any of these questions, you may leave them blank.

In the unlikely event that some of the survey or interview questions make you uncomfortable or upset, you are always free to decline to answer or to stop your participation at any time. Should you feel discomfort after participating and you are a Texas State University student, you may contact the University Health Services for counseling services at list 512-245-2208. They are located at 5-4.1 LBJ Student Center 601 University Drive, San Marcos, Texas 78666.

BENEFITS/ALTERNATIVES

There will be no direct benefit to you from participating in this study. However, the information that you provide will help inform future dam removals and avoid unnecessary environmental costs potentially associated with them while maximizing benefits. In addition, this research contributes to the growing body of literature on freshwater mussels in Texas, and will help inform future sampling and conservation efforts. Ultimately this research will provide new insights into the science of river restoration and conservation, and spur future actions that help protect imperiled species and valuable water resources.

EXTENT OF CONFIDENTIALITY

Reasonable efforts will be made to keep the personal information in your research record private and confidential. Any identifiable information obtained in connection with this study will remain confidential and will be disclosed only with your permission or as required by law. The members of the research team, the funding agency, and the Texas State University Office of Research Compliance (ORC) may access the data. The ORC monitors research studies to protect the rights and welfare of research participants.

Your name will not be used in any written reports or publications which result from this research. Data will be kept for three years (per federal regulations) after the study is completed and then destroyed.

PAYMENT/COMPENSATION

You will receive a gift card for Amazon.com in the amount of $30.00

PARTICIPATION IS VOLUNTARY

You do not have to be in this study if you do not want to. You may also refuse to answer any questions you do not want to answer. If you volunteer to be in this study, you may withdraw from it at any time without consequences of any kind or loss of benefits to which you are otherwise entitled.

QUESTIONS

If you have any questions or concerns about your participation in this study, you may contact the Principal Investigator, Erin D. Dascher: 267-242-4627 or edd1@txstate.edu.

This project 2017384 was approved by the Texas State IRB on February 24, 2017. Pertinent questions or concerns about the research, research participants' rights, and/or research-related injuries to participants should be directed to the IRB Chair, Dr. Jon Lasser 512-245-3413 – (lasser@txstate.edu) or to Monica Gonzales, IRB Regulatory Manager 512-245-2334 - (meg201@txstate.edu).

DOCUMENTATION OF CONSENT

I have read this form and decided that I will participate in the project described above. Its general purposes, the particulars of involvement and possible risks have been explained to my satisfaction. I understand I can withdraw at any time.

Printed Name of Study Participant	**Signature** of Study Participant	Date

Signature of Person Obtaining Consent	Date

Appendix 4.2. Workshop summary distributed to invitees of BREW.

Barrier Removal Prioritization Workshop – Executive Summary

The Guadalupe – San Antonio River System (GSARS) includes the Guadalupe and San Antonio river basins, and comprises an area of 26,346 km². Both basins drain south-east from the Texas Hill Country across the Balcones Escarpment, and eventually into the Gulf of Mexico. There are at least 13 different species of freshwater mussels that inhabit the Guadalupe – San Antonio River System, including two endemic species *Amphinaias aurea* found only in the GSARS and *Quadrula petrina* found only in the Colorado and Guadalupe drainages.

My research focuses primarily on river connectivity and mussel conservation, and as part of my dissertation I am developing a dam removal prioritization model for GSARS. The purpose of this workshop is to gather expert opinion on the metrics/indices used to prioritize dams for removal. Specifically, the goal of the workshop is to define and weight the variables that will be used in a dam removal prioritization model.

The GSARS contains 375 dams, 215 in the Guadalupe Basin and 160 in the San Antonio Basin (TCEQ 2014). Of the total, there are 86 small dams, 277 medium dams, 11 large dams, and only 1 is a very large dam. These dams fragment the river system into a series of dis-connected 'functional river networks' (FRNs). This fragmentation hinders dispersal and migration of aquatic organisims and the availability of quality habitat.

Dam removals are being increasingly used in the United States to restore river habitat and reconnect fragmented river networks. The dam removal prioritization model, informed from expert opinion attained at this workshop, will highlight dam removal as a restoration strategy for aquatic organisms in Central Texas river basins. Additionally, this research will provide insight for future management and conservation efforts of freshwater mussels.

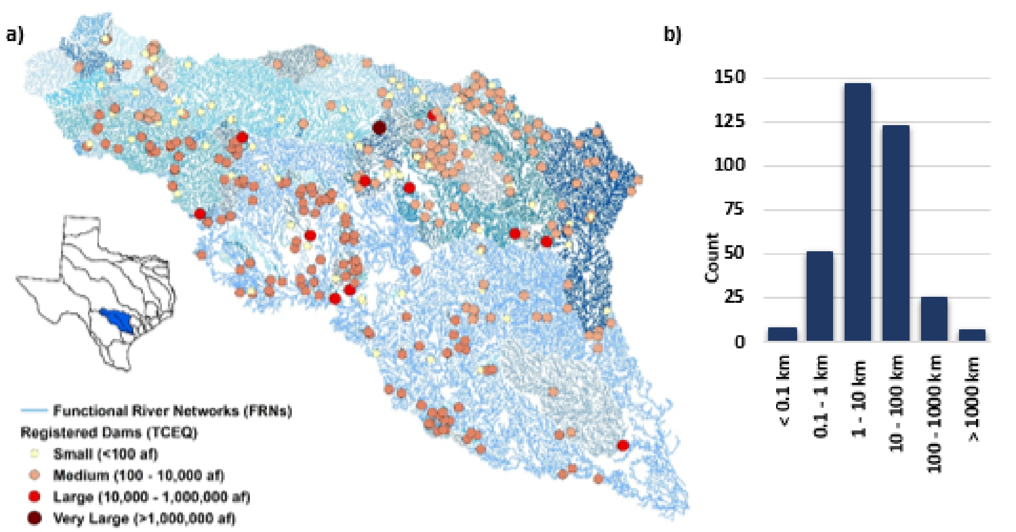

Figure 1. a) The Guadalupe – San Antonio River System showing the fragmentation created by registered dams alone. b) Number of FRNs by length in the Guadalupe – San Antonio River System created by registered dams.

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Appendix 4.3. List of potential metrics for dam removal prioritization model presented at BREW.

Dam Attributes
1. **Height** - Height of the dam, in feet to the nearest foot, which is defined as the vertical distance between the lowest point on the crest of the dam and the lowest point in the original streambed. (TCEQ 2014) 2. **Length** - Length of the dam, in feet, which is defined as the length along the top of the dam. This also includes the spillway, power plant, navigation lock, fish pass, etc., where these form part of the length of the dam. If detached from the dam, these structures should not be included. (TCEQ 2014) 3. **Avg_Storage** - Normal storage, in acre-feet, which is defined as the total storage space in a reservoir below the normal retention level, including dead and inactive storage and excluding any flood control or surcharge storage. If unknown, the value will be blank. (TCEQ 2014) 4. **Age** - Year when the original main dam structure was completed. If unknown, and reasonable estimate is unavailable 0 is used. (TCEQ 2014)

Connectivity Status Metrics
5. **Watershed Area (WsAreaSqKM)** - Watershed area (square km) at NHDPlus stream segment outlet, i.e., at the most downstream location of the vector line segment. (Hill et al. 2016) 6. **Upstream Dam Density - (batUSDty_H)** Density of dams in the watershed upstream from dam. Created using the Barrier Assessment Tool (BAT). 7. **Downstream Dam Density (batDSDty_H)** - Density of dams in the watershed downstream from dam. Created using BAT. 8. **Upstream Dam Count (batCntUS_H)** - Number of dams upstream on the same main line. Created using BAT. 9. **Downstream Dam Count (batCntDS_H)** - Number of dams between dam and mouth of the river, on the main line. Created using BAT. 10. **Stream Length Upstream (batFunUS)** - Upstream functional river network length in kilometers. Created using BAT. 11. **Stream Length Downstream (batFunDS)** - Downstream functional river network length in kilometers Created using BAT.

Connectivity Improvement Metrics

12. **Potential Reconnected Stream Length (batTotUSDS)** - total upstream and downstream functional river network, or the length of the functional river network if the dam were removed. Created using BAT.
13. **Absolute Gain (batAbs)** - This metric is the minimum of the two functional networks of a barrier. For example, if the upstream functional network was 10 kilometers and downstream functional network was 5 kilometers then the Absolute gain will be 5 kilometers. The distance values are in meters.

14. **Relative Gain (batRel)** - This metric is Absolute Gain divided by the total connected length. Created using BAT.

Ecological Metrics

15. **Fish (Tsp_Fish_HUC8)** - Total potentially present fish species (HUC 8). (Maxwell 2012)

16. **Endemic Fish (End_Fish_HUC8)** - Total potentially present endemic fish species (HUC 8). (Maxwell 2012)

17. **Mussels (Tsp_Muss_HUC8)** - Total potentially present mussel species (HUC 8). Calculated from data provided in Dascher et al. 2017.

18. **Threated Mussels (End_Muss_HUC8)** - Total potentially present threatened mussel species (HUC 8). Calculated from data provided in Dascher et al. 2017.

19. **Upstream Mussels (US_MUSS)** - Number of mussel species located upstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017.

20. **Downstream Mussels (DS_MUSS)** - Number of mussel species located downstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017.

21. **Threatened Upstream Mussels (US_EndMUSS)** - Number of threatened mussel species located upstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017.

22. **Threatened Downstream Mussels (DS_EndMUSS)** - Number of threatened mussel species located downstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017.

Flow Metrics

23. **Mean Yearly Discharge (Q0001E)** - Extended Unit Runoff Method (EROM) mean annual flow estimates for NHDFlowline features in the NHDPlus network. These flow estimates reflect the 1971 to 2000-time period. Provided by the NHDPlusV2 EROM extension.

24. **Mean Monthly discharge (Q0001E_month)** - Extended Unit Runoff Method (EROM) mean annual flow estimates for NHDFlowline features in the NHDPlus network. These flow estimates reflect the 1971 to 2000-time period. Provided by the NHDPlusV2 EROM extension.

25. **Base Flow Index (BFIWs)** - Base flow is the component of streamflow that can be attributed to ground-water discharge into streams. The BFI is the ratio of base flow to total flow, expressed as a percentage, within watershed. (Hill et al. 2016)

Water Quality Metrics

26. **Ammonium Ion Concentration (NH4_2008Ws)** - Annual gradient map of precipitation-weighted mean deposition for ammonium ion concentration wet deposition for 2008 in kg of NH4/ha/yr, within watershed (Hill et al. 2016)

27. **Nitrate Ion Concentration (NO3_2008Ws)** - Annual gradient map of precipitation-weighted mean deposition for nitrate ion concentration wet deposition for 2008 in kg of NO3/ha/yr, within watershed.

28. **Mean Pesticide Use in Watershed (Pestic97Ws)** - Mean pesticide use (kg/km2) in yr. 1997 within watershed. (Hill et al. 2016)

Landscape Metrics

29. **Kffactor (KffactWs)** - The Kffactor is used in the Universal Soil Loss Equation (USLE) and represents a relative index of susceptibility of bare, cultivated soil to particle detachment and transport by rainfall within watershed. (Hill et al. 2016)

30. ***Urban Land Cover (PctUrbTotal2011Ws)** - Percent of watershed area classified as developed, medium -, low -, and high – intensity land use (NLCD 2011 class 22, 23, and 24) (Hill et al. 2016)

31. ***Crop Land Cover (PctCrop2011Ws)** - Percent of watershed area classified as crop land use (NLCD 2011 class 82) (Hill et al. 2016)

32. ***Hay Land Cover (PctHay2011Ws)** - Percent of catchment area classified as pasture/hay land use (NLCD 2011 class 81) (Hill et al. 2016)

33. ***Forest Land Cover (PrctFst2011WS)** - Percent of watershed area classified as deciduous, evergreen, and mixed forest land cover (NLCD 2011 class 41, 42, 43) (Hill et al. 2016)

34. *** Wetland Land Cover (PctWet2011Ws)** - Percent of watershed area classified as herbaceous and woody wetland land cover (NLCD 2011 class 95, 90) (Hill et al. 2016)

35. ***Mean Population Density (PopDen2010Ws)** - Mean populating density (people/square km) within watershed (Hill et al. 2016)

36. ***Density of Roads (RdDenWs)** - Density of roads (2010 Census Tiger Lines) within watershed (km/square km) (Hill et al. 2016)

37. **Density of Road Stream Intersects (RdCrsWs)** - Density of roads-stream intersections (2010 Census Tiger Lines-NHD stream lines) multiplied by NHDPlusV21 slope within watershed (crossings*slope/square km) (Hill et al. 2016)

*Note: These variables are also available as the percent of a 100-meter riparian buffer of the NHD flow lines in the watershed, as opposed to percent of the total watershed area.

Metric Gaps

Hydrology

Water Use/Withdrawals

Climate Change

Appendix 4.4. Output from the Barrier Removal Expert Workshop (BREW).

Workshop Metrics Summary

Erin D. Dascher

This document includes a list of metrics discussed and agreed upon by the participants of BREW, a one-day workshop held on March 29, 2017 to determine the importance and weight of metrics for inclusion in a model I am creating to prioritize freshwater mussel conservation in the context of fragmented river systems and potential river connectivity improvements. Participants included representatives from government agencies, non-profit organizations, and academics specializing in the areas of fluvial geomorphology, instream flows, river restoration, and landscape change. Lacking from the workshop were the opinions of freshwater mussel ecologists, whose input is considered significant to this process, and I am currently working on how to incorporate their feedback moving forward. I will create a model using the metrics listed in this document that was produced from workshop input. The model will function as a decision support tool that highlights dam removal as a restoration strategy for aquatic organisms, i.e. freshwater mussels, in the Guadalupe – San Antonio River System. This contribution should be considered an initial iteration of an adaptive model designed to provide insight for future management and conservation efforts for freshwater mussels.

The list is organized by categorical order of importance for the five main categories: Biological, Connectivity, Water Quality, Landscape, and Dam metrics. Sub-section (a) describes the metric and how the values associated with the metric will be viewed relative to the 'best' to 'worst' scenario. Sub-section (b) provides the weight on a 1-9 scale determined by the work shop participants. The metrics in each category are not listed in any specific order.

1. Biological Metrics

38. **Host Fish (Host_Fish_HUC8)** - Total potentially present host fish species by HUC 8. Data provided by Ford and Oliver 2015, Fishes of Texas (Hendrickson et al. 2015) and Maxwell 2012.
    a. More potential host fish indicates potentially better habitat locations for freshwater mussels, as they require host fish to reproduce and propagate. The absence of any host fish species would represent the worst scenario, while the largest number of potentially present host fish species would represent the best scenario.
    b. Importance (1-9) = 9

39. **SGCN Fish (SGCN_Fish_HUC8)** – Total potentially present fish species of greatest conservation need by HUC8. Data provided by Fishes of Texas (Hendrickson et al. 2015), Maxwell 2012, and TPWD 2011.
    a. If a larger number of Species of Greatest Conservation Need (SGCN) fish are potentially present, than creating more habitat for freshwater mussels may also open up habitat for these fish, potentially providing an additional reason or emphasis on a particular restoration decision/strategy. The absence of any SGCN fish species would represent the worst scenario, while the largest number of potentially present SGCN fish species would represent the best scenario.
    b. Importance (1-9) = 4

40. **Mussels (Tsp_Muss_HUC8) -** Total potentially present mussel species by HUC 8. Calculated from data provided in Dascher et al. 2017.
    a. Since there is a limited amount of freshwater mussel data available, this measure acts as a safeguard or insurance policy accounting for any potentially present mussel species when there is not a known mussel location on a functional river network. The absence of any mussel species is the worst scenario, while the largest number of potentially present mussel species present would represent the best scenario.
    b. Importance (1-9) = 2

41. **Federally Listed Mussels (Fed_Muss_HUC8) -** Total potentially present federally listed mussel species by HUC 8. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. Since there is a limited amount of freshwater mussel available, this measure acts as a safeguard or insurance policy accounting for any potentially present federally listed mussel species when there is not a known mussel location on a functional river network. The absence of any federally listed mussel species is the worst scenario, while the largest number of potentially present federally listed mussel species present would represent the best scenario.
    b. Importance (1-9) = 3

42. **State Listed Mussels (St_Muss_HUC8) -** Total potentially present federally listed mussel species by HUC 8. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. Since there is a limited amount of freshwater mussel available, this measure acts as a safeguard or insurance policy accounting for any potentially present state listed mussel species when there is not a known mussel location on a functional river network. The absence of any state listed mussel species is the worst scenario, while the largest number of potentially present state listed mussel species present would represent the best scenario.
    b. Importance (1-9) = 2

43. **Potential/Candidate Mussels (Pot_Muss_HUC8)** - Total potentially present candidate or potentially listed mussel species by HUC 8. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. Since there is a limited amount of freshwater mussel available, this measure acts as a safeguard or insurance policy accounting for any potentially present potential or candidate mussel species when there is not a known mussel location on a functional river network. The absence of any potential/candidate mussel species is the worst scenario, while the largest number of potentially present potential/candidate mussel species present would represent the best scenario.
    b. Importance (1-9) = 2

44. **Upstream Downstream Mussels (USDS_MUSS)** - Number of mussel species located upstream and/or downstream on functional river network. Calculated in a GIS from data provided in Dascher et al. 2017.
    a. The largest number of mussel species located upstream and/or downstream on a functional river network is considered the best scenario, as it indicates quality habitat for freshwater mussel species. A lack of mussel species upstream and/or downstream on a functional river network is considered the worst scenario.
    b. Importance (1-9) = 3

45. **Federally Listed Upstream Downstream Mussels (USDS_FedMUSS)** - Number of federally listed mussel species located upstream and/or downstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. The largest number of federally listed mussel species located upstream and/or downstream on a functional rive network is considered the best scenario. A lack of federally listed mussel species upstream and/or downstream on a functional river network is considered the worst scenario.
    b. Importance (1-9) = 9

46. **State Listed Upstream Downstream Mussels (USDS_StMUSS)** - Number of State Listed mussel species located upstream and/or downstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. The largest number of state listed mussel species located upstream and/or downstream on a functional river network is considered the best scenario. A lack of state listed mussel species upstream and/or downstream on a functional river network is considered the worst scenario.
    b. Importance (1-9) = 7

47. **Potential/Candidate Upstream Downstream Mussels (USDS_PotMUSS) -** Number of potentially listed or candidate mussel species located upstream and/or downstream on functional river network. Calculated in a GIS. Calculated from data provided in Dascher et al. 2017 and TPWD 2016.
    a. The largest number of potential/candidate mussel species located upstream and/or downstream on a functional rive network is considered the best scenario. A lack of potential/candidate mussel species upstream and/or downstream on a functional river network is considered the worst scenario.
    b. Importance (1-9) = 8

2. Connectivity Improvement Metrics

48. **Potential Reconnected Stream Length (batTotUSDS) -** total upstream and downstream functional river network, or the length of the functional river network if the dam were removed. Created using the Barrier Assessment Tool (BAT).
    a. The greater the total amount of potentially reconnected stream length the more potentially available habitat is created for freshwater mussels. The largest amount of total reconnected stream network is the best scenario, and the least amount of total reconnected stream network is the worst scenario.
    b. Importance (1-9) = 9

49. **Absolute Gain (batAbs) -** This metric is the minimum of the two functional networks of a barrier. For example, if the upstream functional network was 10 kilometers and downstream functional network was 5 kilometers then the Absolute gain will be 5 kilometers. The distance values are in meters.
    a. The greater the absolute gain of potentially reconnected stream length the more potentially available habitat is created for freshwater mussels and host fish. The largest absolute gain is the best scenario, while the least absolute gain is the worst scenario.
    b. Importance (1-9) = 5

50. **Relative Gain (batRel) -** This metric is Absolute Gain divided by the total connected length. Created using BAT.
    a. The greater the relative gain of potentially reconnected stream length the more potentially available habitat is created for freshwater mussels and host fish. The largest relative gain is the best scenario, while the least relative gain is the worst scenario.
    b. Importance (1-9) = 8

51. **** Ammonia Nitrogen Instream Concentration (NH4_instream)** - Concentration of ammonia in a river segment, (Parameter code 610; Nitrogen, Ammonia, Total MG/L AS N). Data provided by Texas Clean Rivers Program (TCEQ 2017).
    a. A higher load of ammonia nitrogen would indicate poorer quality habitat for freshwater mussels, and the highest load of ammonia nitrogen is the worst scenario. Lower load would indicate higher quality habitat for freshwater mussels, and the lowest amount of ammonia nitrogen is the best scenario.
    b. Importance (1-9) = 6

52. **** Nitrate Nitrogen Instream Concentration (NO3_2Instream)** - Concentration of ammonia in a river segment, (parameter code 620; Nitrate Nitrogen, Total MG/L AS N). Data provided by Texas Clean Rivers Program (TCEQ 2017).
    a. A higher load of nitrate nitrogen would indicate poorer quality habitat for freshwater mussels, and the highest load of nitrate nitrogen is the worst scenario. Lower load would indicate higher quality habitat for freshwater mussels, and the lowest amount of ammonia nitrogen is the best scenario.
    b. Importance (1-9) = 3

53. **** High Stream Temperature (High_StrTemp) -** Twenty-four-hour maximum stream temperature. Data provided by Texas Clean Rivers Program (TCEQ 2017).
    a. A daily maximum stream temperature of 35 degrees Celsius or higher indicates poorer quality habitat for freshwater mussels. A daily maximum temperature of 35 degrees Celsius or higher is the worst scenario.
    b. Importance (1-9) = 4

54. **** Low Stream Temperature (Low_StrTemp) -** Twenty-four-hour minimum stream temperature. Data provided by Texas Clean Rivers Program (TCEQ 2017).
    a. Lower daily minimum stream temperatures may indicate poorer quality habitat for freshwater mussels and the lowest daily minimum temperature is the worst scenario. Higher minimum temperatures may indicate higher quality habitat for freshwater mussels, and is considered the best scenario.
    b. Importance (1-9) = 3

55. **Mean Summer Stream Temperature (MSST_2014)** - Predicted mean summer stream temperature (July-Aug) for year 2014. Data provided by StreamCat (Hill et al. 2016).
    a. This measure is considered a proxy for max daily stream temperature, and would only be included in the model if other stream temperature data is unavailable or unable to be included. A higher mean summer temperature may indicate poorer quality habitat for freshwater mussels, as they cannot tolerate water temperatures of 35 degrees Celsius or higher. A mean summer temperature of 35 degrees Celsius or higher is the worst scenario.
    b. Importance (1-9) = 2

4. Landscape Metrics

56. **Total ISA Land Cover (PctISATotal2011Ws)** - Percent of watershed area classified as developed, open space (<20% ISA), low - (20% - 49% ISA), medium - (50% - 70% ISA), and high - intensity (80% - 100% ISA) land use (NLCD 2011 class 21, 22, 23, and 24). Data provided by StreamCat (Hill et al. 2016).
    a. Watersheds with more impervious surface area may indicate poor quality habitat for freshwater mussels and host fish. The largest percentage of ISA in a contributing watershed is the worst scenario, while the lowest percentage of ISA in a contributing watershed is the best scenario.
    b. Importance (1-9) = 5

57. **Undeveloped Land Cover (PctUndTotal2011Ws)** - Percent of watershed area classified as shrub/scrub, grassland, forest, and wetland land cover (NCLD 2011 class 52, 71, 41,42,43, 95, 90) Data provided by StreamCat (Hill et al. 2016).
    a. Watersheds with more undeveloped land cover may indicate higher quality habitat for freshwater mussels. The largest percentage of undeveloped land cover in a contributing watershed is the best scenario, while the lowest percentage of undeveloped land cover in a contributing watershed is the worst scenario.
    b. Importance (1-9) = 5

58. **Riparian Buffer (Pct2011Rp100)** - Percent of watershed area classified as deciduous, evergreen, and mixed forest land cover (NLCD 2011 class 41, 42, 43) Data provided by StreamCat (Hill et al. 2016).
    a. Watersheds with larger riparian buffers may indicate higher quality habitat for freshwater mussels. The contributing watershed with the largest percentage of riparian buffer is the best scenario, while the contributing watershed with the lowest percentage of riparian buffer is the worst scenario.
    b. Importance (1-9) = 7

5. Dam Attributes

59. **Reservoir Length_ Storage (Lg_StorRatio)** - The ratio of a reservoir's length to the normal storage of a reservoir. Calculated in a GIS from TCEQ (2014) storage data and NHD (USGS 2006) waterbodies.
    a. A larger number would equate to a large impact or more evaporation from a reservoir, and the largest resulting number would represent the worst scenario.
    b. Importance (1-9) = 4

60. ***Avg_Storage (AvgStor)** - Normal storage, in acre-feet, which is defined as the total storage space in a reservoir below the normal retention level, including dead and inactive storage and excluding any flood control or surcharge storage. If unknown, the value will be blank. Data provided by the TCEQ (2014).
    a. Larger average storage may indicate a large amount of lentic habitat that couple potentially transition back to lotic habitat, and for this reason, larger average storage is considered a better scenario than lower average storage.
    b. Importance (1-9) = 3

61. ***Reservoir_Lg (ResLg)** - The total length of the existing reservoir measured using aerial imagery. Calculated in a GIS.
    a. Reservoir length is an indicator of the amount of lentic habitat that would transition back to lotic habitat, providing more habitat for freshwater mussels that prefer lentic habitats. The longest reservoir length is considered the best scenario, and the shortest reservoir length is considered the worst scenario.
    b. Importance (1-9) = 3

6. Metric Gaps

Multiple metrics were discussed in the work group that were important components of prioritizing freshwater mussel conservation and habitat restoration but were unable to be included in the model. This was due to either a lack of data or because data was not available in a usable format for the model. Below are the metrics that were discussed, but are unable to be included in the current model. Because of the importance of these metrics to mussels and their exclusion from the model, we recommend the model be viewed as a work in progress, and a starting point to further discuss potential river connectivity improvements as a conservation strategy for freshwater mussels.

Flow Metrics - Data related to instream flows and discharges, as well as, channel morphology was discussed at length and considered a critical component of further discussion and inclusion in a model that prioritizes freshwater mussel conservation. Consensus was on the value of including a metric that describe the potential change related to improved connectivity, but data providing this information is unavailable at the scale and resolution of the model.

Water Use/Withdrawals – Metrics concerning water use/withdrawals was discussed by the workgroup as a potentially important metric to include in the model, but a lack of data availability/access for water use or water withdrawals from particular locations along the stream network resulted in the exclusion of these metric(s) from the current model. Additionally, data related to flow alterations such as reservoir operations, groundwater pumping, diversions, etc. was similarly discussed, but due to a lack of data availability was unable to be included in the current model.

Substrate – The type of substrate available at particular locations was discussed as an important metric or set of metrics for a model prioritizing dam removals for freshwater mussel conservation. Data on substrate at the scale and resolution of this model was determined to be lacking or difficult to incorporate into the current model.

Effective Stream Length - The amount of stream length required to support current and future freshwater mussel populations was discussed as an important metric to consider for freshwater mussel conservation. The potentially species-specific nature of this metric and currently unknown or unavailable data relating to the amount of stream length required to support current or future mussel populations resulted in being unable to include this metric in the model at this time.

** Available data does not conform to extrapolating to entire NHD network, there are too many FRN segments with no data. These variables can be used to refine location specific considerations.

*** Correlation of length and storage was significant with ratio variable combining length and storage, thus only the ratio variable will be included.

Appendix 4.5. AHP Matrices.

1. Biological Metrics

Reciprocal Matrix

	Host_Fish_HUC8	SGCN_Fish_HUC8	Tsp_Muss_HUC8	St_Muss_HUC8	Pot_Muss_HUC8	USDS_MUSS	USDS_StMUSS
Host_Fish_HUC8	1.00	6.00	8.00	8.00	8.00	7.00	3.00
SGCN_Fish_HUC8	0.17	1.00	4.00	3.00	3.00	2.00	0.25
Tsp_Muss_HUC8	0.13	0.25	1.00	2.00	2.00	2.00	0.20
St_Muss_HUC8	0.13	0.33	0.50	1.00	2.00	2.00	0.17
Pot_Muss_HUC8	0.13	0.33	0.50	0.50	1.00	2.00	0.17
USDS_MUSS	0.14	0.50	0.50	0.50	0.50	1.00	0.20
USDS_StMUSS	0.33	4.00	5.00	6.00	6.00	5.00	1.00
Sum	2.52	17.42	25.50	28.00	29.50	27.00	5.48

Normalized Matrix

0.397163	0.344498	0.313725	0.285714	0.271186	0.259259	0.547112	0.345111
0.066194	0.057416	0.156863	0.107143	0.101695	0.074074	0.045593	0.034511
0.049645	0.014354	0.039216	0.071429	0.067797	0.074074	0.036474	0.024651
0.049645	0.019139	0.019608	0.035714	0.067797	0.074074	0.030395	0.024651
0.049645	0.019139	0.019608	0.017857	0.033898	0.074074	0.030395	0.024651
0.056738	0.028708	0.019608	0.017857	0.016949	0.037037	0.036474	0.028759
0.132388	0.229665	0.196078	0.214286	0.203390	0.185185	0.182371	0.345111
0.198582	0.287081	0.235294	0.250000	0.237288	0.222222	0.091185	0.172555
1	1	1	1	1	1	1	1

Priority Vector

Host_Fish_HUC8	0.345471
SGCN_Fish_HUC8	0.080436
Tsp_Muss_HUC8	0.047205
St_Muss_HUC8	0.040128
Pot_Muss_HUC8	0.033658
USDS_MUSS	0.030266
USDS_StMUSS	0.211059
USDS_PotMUSS	0.211776
	1.00

Weighted Sum Vector		**Consistency Vector**
HostFish_HUC8	2.652915434	7.679121846
SGCN_Fish_HUC8	0.705927873	8.776260785
Tsp_Muss_HUC8	0.38868482	8.233989939
St_Muss_HUC8	0.323571572	8.063514356
Pot_Muss_HUC8	0.259841253	7.719946063
USDS_MUSS	0.23172813	7.656311448
USDS_StMUSS	1.81759943	8.611798654
USDS_PotMUSS	1.897617796	8.960493711

Teknomo's method

8	n	8
	λ max	8.960493711
	CI	0.137213387
	RI	1.49
	CR	0.092089522

2. Connectivity Improvement Metrics

Reciprocal Matrix

	batTotUSDS	batAbs	batRel
batTotUSDS	1.00	5.00	2.00
batAbs	0.20	1.00	0.25
batRel	0.50	4.00	1.00
Sum	1.70	10.00	3.25

Normalized Matrix

0.588235294	0.5	0.615384615
0.117647059	0.1	0.076923077
0.294117647	0.4	0.307692308
1	1	1

Priority Vector

batTotUSDS	0.567873303
batAbs	0.098190045
batRel	0.333936652
	1.00

Weighted Vector

0.567873303	0.490950226	0.667873303
0.113574661	0.098190045	0.083484163
0.283936652	0.392760181	0.333936652

Weighted Sum Vector

batTotUSDS	1.726696833
batAbs	0.295248869
batRel	1.010633484

Consistency Vector

3.04063745
3.006912442
3.026422764

Teknomo's method

3	n	3
	λ max	3.04063745
	CI	0.020318725
	RI	0.58
	CR	**0.035032285**

3. Landscape Metrics (BREW)

Reciprocal Matrix

	PctISATotal2011Ws	PctUndTotal2011Ws	Pct2011Rp100	Lg_StorRatio
PctISATotal2011Ws	1.00	1.00	0.33	2.00
PctUndTotal2011Ws	1.00	1.00	0.33	2.00
Pct2011Rp100	3.00	3.00	1.00	4.00
Lg_StorRatio	0.50	0.50	0.25	1.00
Sum	5.50	5.50	1.92	9.00

Normalized Matrix

0.181818182	0.181818182	0.173913043	0.222222222
0.181818182	0.181818182	0.173913043	0.222222222
0.545454545	0.545454545	0.52173913	0.444444444
0.090909091	0.090909091	0.130434783	0.111111111
1	1	1	1

Priority Vector

PctISATotal2011Ws	0.189942907
PctUndTotal2011Ws	0.189942907
Pct2011Rp100	0.514273166
Lg_StorRatio	0.105841019
	1.00

Weighted Vector

0.189942907	0.189942907	0.171424389	0.211682038
0.189942907	0.189942907	0.171424389	0.211682038
0.569828722	0.569828722	0.514273166	0.423364076
0.094971454	0.094971454	0.128568292	0.105841019

Weighted Sum Vector		Consistency Vector
PctISATotal2011Ws	0.762992241	4.016955684
PctUndTotal2011Ws	0.762992241	4.016955684
Pct2011Rp100	2.077294686	4.039282664
Lg_StorRatio	0.424352218	4.0093361

Teknomo's method

4	n	4
	λ max	4.039282664
	CI	0.013094221
	RI	0.9
	CR	**0.014549135**

4. Dam Attributes

Reciprocal Matrix

	Age	Owner	Size
Age	1.00	0.50	0.33
Owner	2.00	1.00	0.50
Size	3.00	2.00	1.00
Sum	6.00	3.50	1.83

Normalized Matrix

0.166666667	0.142857143	0.181818182
0.333333333	0.285714286	0.272727273
0.5	0.571428571	0.545454545
1	1	1

Priority Vector

Age	0.163780664
Owner	0.297258297
Size	0.538961039
	1.00

Weighted Vector

0.163780664	0.148629149	0.17965368
0.327561328	0.297258297	0.269480519
0.491341991	0.594516595	0.538961039

	Weighted Sum Vector	Consistency Vector
Age	0.492063492	3.004405286
Owner	0.894300144	3.008495146
Size	1.624819625	3.014725569

Teknomo's method

3	n	3
	λ max	3.014725569
	CI	0.007362784
	RI	0.58
	CR	**0.012694456**

5. Landscape Metrics (Lit)

Reciprocal Matrix

	PctISATotal2011Ws	PctUndTotal2011Ws	Pct2011Rp100
PctISATotal2011Ws	1.00	1.00	0.33
PctUndTotal2011Ws	1.00	1.00	0.33
Pct2011Rp100	3.00	3.00	1.00
Sum	5.00	5.00	1.67

Normalized Matrix

0.2	0.2	0.2
0.2	0.2	0.2
0.6	0.6	0.6
1	1	1

Priority Vector

PctISATotal2011Ws	0.20
PctUndTotal2011Ws	0.20
Pct2011Rp100	0.60
	1.00

Weighted Vector

0.2	0.2	0.2
0.2	0.2	0.2
0.6	0.6	0.6

Weighted Sum Vector

PctISATotal2011Ws	0.6
PctUndTotal2011Ws	0.6
Pct2011Rp100	1.8

Consistency Vector

3
3
3

Teknomo's method

3	n	3
	λ max	3
	CI	2.22045E-16
	RI	0.58
	CR	**3.82836E-16**

6. BREW Metric Groups

Reciprocal Matrix

	Biological	CIM	Landscape
Biological	1.00	2.00	4.00
CIM	0.50	1.00	3.00
Landscape	0.25	0.33	1.00
Sum	1.75	3.33	8.00

Normalized Matrix

0.571428571	0.6	0.5
0.285714286	0.3	0.375
0.142857143	0.1	0.125
1	1	1

Priority Vector

Biological	0.557142857
CIM	0.320238095
Landscape	0.122619048
	1.00

Weighted Vector

0.557142857	0.64047619	0.49047619
0.278571429	0.320238095	0.367857143
0.139285714	0.106746032	0.122619048

Weighted Sum Vector		**Consistency Vector**
Biological	1.688095238	3.02991453
CIM	0.966666667	3.018587361
Landscape	0.368650794	3.006472492

Teknomo's method

3	n	3
	λ max	3.02991453
	CI	0.014957265
	RI	0.58
	CR	**0.025788388**

6. Literature model Metric Groups

Reciprocal Matrix

	Biological	CIM	Landscape	Dam_Attributes
Biological	1.00	2.00	3.00	0.50
CIM	0.50	1.00	2.00	0.25
Landscape	0.33	0.50	1.00	0.25
Dam_Attributes	2.00	3.00	4.00	1.00
Sum	3.83	6.50	10.00	2.00

Normalized Matrix

0.260869565	0.307692308	0.3	0.25
0.130434783	0.153846154	0.2	0.125
0.086956522	0.076923077	0.1	0.125
0.52173913	0.461538462	0.4	0.5
1	1	1	1

Priority Vector

Biological	0.279640468
CIM	0.152320234
Landscape	0.0972199
Dam_Attributes	0.470819398
	1.00

Weighted Vector

0.279640468	0.304640468	0.291659699	0.235409699
0.139820234	0.152320234	0.194439799	0.117704849
0.093213489	0.076160117	0.0972199	0.117704849
0.559280936	0.456960702	0.388879599	0.470819398

Weighted Sum Vector		**Consistency Vector**
Biological	1.111350334	3.974211392
CIM	0.604285117	3.967201866
Landscape	0.384298356	3.952877517
Dam_Attributes	1.875940635	3.984416622

Teknomo's method

4	n	4
	λ max	3.984416622
	CI	-0.005194459
	RI	0.9
	CR	**-0.005771621**

REFERENCES

Aldridge D.W., B.S. Payne, and A.C. Miller. 1987 The effects of intermittent exposure to suspended solids and turbulence on three species of freshwater mussels. *Environmental Pollution (Series B)* 45, 17–28.

Al-Rabab'ah, M. A. and C. G. Williams. 2004. An ancient bottleneck in the Lost Pines of central Texas. Molecular Ecology 13: 1075-1084

Ambers, R.K.R. 2007. Effects of a Small, Century-old Dam on a Second order Stream in the Virginia Piedmont. S*outheastern Geographer*, 47(2) 2007: pp. 181–201.

American Rivers. 2015. *62 Dams Removed to Restore Rivers in 2015.* Americanrivers.org. (last accessed 2 February 2016).

American Rivers. 2016. *American Rivers dam removal database*. http://www.americanrivers.org/initiative/dams/projects/2014-dam-removals. (Last accessed 1 March 2016)

American Society of Civil Engineers (ASCE). 2017. *Infrastructure Report Card.* http://www.infrastructurereportcard.org/wp-content/uploads/2017/01/Dams-Final.pdf (last accessed 11 July 2017).

American Society of Civil Engineers (ASCE). 2012. Dam Fact Sheet. In *Renewing Texas Infrastructure, 2012 Texas Infrastructure Report Card.* http://c.ymcdn.com/sites/www.texasce.org/resource/resmgr/Infrastructure_Report_Cards/2012%20ASCE%20Dam%20Fact%20Sheet%20FINAL3.pdf (last accessed 11 July 2017).

Atkinson, C. L. and C. C. Vaughn. 2014. Biogeochemical hotspots: temporal and spatial scaling of the impacts of freshwater mussels on ecosystem function. Freshwater Biology 60: 563-574.

Atkinson, C.L., J. P. Jullian, C. C. Vaughn. 2012. Scale-dependent longitudinal patterns in mussel communities. Freshwater Biology 57: 2272-2284.

Baish, S.K., S.D. David, W.L. Graff. 2002. The Complex Decision Making Process for Removing Dams. *Environment* May: 22-31.

Bednarek, A. T. 2001. Undamming Rivers: A Review of the Ecological Impacts of Dam Removal. *Environmental Management* 27(6): 803-814.

Bellmore, J.R., K.M. Vittum, J.J. Duda, S. Greene. 2015, USGS Dam Removal Science Database (ver. 1.3, July 2015): U.S. Geological Survey data release, https://doi.org/10.5066/F7K935KT

Bellmore, J.R. J.J. Duda, L.S. Craig, S.L. Greene, C.H. Torgersen, M.J. Collins, K. Vittum. 2016. Status and trends of the dam removal research in the United States. *WiREs Water doi: 10.1002/wat2.1164*

Benda, L.N., L. Poff, D. Miller, T. Dunne, G. Reeves, G. Pess, M. Pollock. 2004. The Network Dynamics Hypothesis: How Channel Networks Structure Riverine Habitats. BioScience 54(5): 413-427.

Benner, R., A. Barnett, A. Olivero. 2014. *North Carolina's Freshwater Resilience*. Durham, North Carolina: The Nature Conservancy.

Berg, M.D., S.C. Popescu, B.P. Wilcox, J.P. Angerer, E.C. Rhodes, J. McAlister, W.E. Fox. 2015. Farm Ponds: Overlooked Features wit Important Impacts on Watershed Sediment Transport. *Journal of the American Resources Association* 1-10.

Bogan. A.E. 1993. Freshwater Bivalve Extinctions (Mollusca: Unionoida): A Search for Causes. American Zoology 33: 599-609.

Bondelid, T. 2014. *EROM Monthly Flows*. Horizon Systems. ftp://ftp.horizon-systems.com/NHDplus/NHDPlusV21/Documentation/TechnicalDocs/EROM_Monthly_Flows.pdf

Bowles, D.E. and T.L. Arsuffi. 1993. Karst aquatic ecosystems of the Edwards Plateau region of central Texas, USA: a consideration of their importance, threats to their existence, and efforts for their conservation. Awuatic Conservation: Marine and Freshwater Ecosystems 3: 317-329.

Branco, P., P. Segurado, J.M. Santos, M.T. Ferreira. 2014. Prioritizing barrier removal to improve functional connectivity of rivers. *Journal of Applied Ecology* 51: 1197-1206.

Brim Box, J. & J. Mossa. 1999. Sediment, Land Use, and Freshwater Mussels: Prospects and Problems. Journal of the North American Benthological Society 18(1): 99-117.

Brown, K.M., G. George, W. Daniel. 2010. Urbanization and a threatened freshwater mussel: evidence from landscape scale studies. Hydrobiologia 655: 189-196.

Bruzzi Lion, M., A. Antonio Garda, C. Roberto Fonseca. 2014. Split distance: a key landscape metric shaping amphibian population and communities in forest fragments. Biodiversity Research, 20: 1245-1257.

Bunn, S.E. and A.H. Arthington. 2002. Basic principles and ecological consequences of altered flwo regimes for aquatic biodiversity. *Environmental Management* 30: 492-507.

Burlakova, L. E., and A. Y. Karatayev. 2007. The effect of invasive macrophytes and water level fluctuations on unionids in Texas impoundments. Hydrobiologia 586: 291-302.

Burlakova, L. E., A. Y. Karatayev, V. A. Karatayev, M. E. May, D. L. Bennett, M. J. Cook. 2011. Biogeography and conservation of freshwater mussels (Bivalvia: Unionidae) in Texas: patterns of diversity and threats. Diversity and Distributions 17: 393-407.

Burlakova, L. E., A. Y. Karatayev, V. A. Karatayev, M. E. May, D. L. Bennett, M. J. Cook. 2011a. Endemic species: Contribution to community uniqueness, effect of habitat alteration, and conservation priorities. Biological Conservation 144: 155-165.

Burroughs, B.A., D.B. Hayes, K.D. Klomp, J.F. Hansen, J. Mestak. 2009. Effects of Stonach Dam removal on fluvial geomorphology in the Pine River, Michigan, Unites States. Geomorphology 110:96-107.

Cao, Y., A. Stodola, S. Douglass, D. Shasteen, K. Cummings, A. Holtrop. 2015. Modelling and mapping the distribution, diversity, and abundance of freshwater mussels (Family Unionididae) in wadeable streams of Illinois, USA. Freshwater Biology 60: 1379-1397.

CGIAR-CSI Global Aridity and Global-PET Database (CGIAR-CSI). 2008. *Global-PET Annual*. http://www.cgiar-csi.org/ (last accessed 6 April 2014).

Chin, A., L.R. Laurencio, A. E. Martinez. 2008. The Hydrologic Importance of Small- and Medium-Sized Dams: Examples from Texas. *The Professional Geographer* 60(2): 238-251.

Clark, B.T. 2009. River Restoration in the American West: Assessing Variation in the Outcomes of Policy Change. *Society & Natural Resources* 22(5): 401-416.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology 18*(1): 117-143.

Conner, J. V. and R. D. Suttkus. 1986. Zoogeography of freshwater fishes of the Western Gulf Slope of North America. In C. H. Hocutt, E O Wiley (ed.) The Zoogeography of North American Freshwater Fishes. John Wiley and Sons, New York: 413-456.

Cooper, J.E. 2011. Chapter 3: Unioid Mussel Mortality from Habitat Loss in the Salmon River, New York, Following Dam Removal. In Advances in Environmental Research. 14. ed. J. A. Daniels, 351-364. Nova Science Publishers, Inc.

Corsair, H.J., J.B. Ruch, P.Q. zheng, B.F. Hobbs, J.F. Koonce. 2009. Multicriteria Decision Analysis of Stream Restoration: Potential and Examples. *Group Decisions and Negotiations* 18:387-417.

Costanza, R., R. D'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387(6630): 253-260.

Cote, D., D.G. Kehler, C. Bourne, Y.F. Wiersma. 2009. A new measure of longitudinal connectivity for stream network. *Landscape Ecology* 24:101-113.

Csiki, S.J. & B.L. Rhoads. 2014. Influence of four run-of-river dams on channel morphology and sediment characteristics in Illinois, USA. *Geomorphology* 206 (2014): 215-229.

Curtis, K.E., C.E. Renshaw, F.J. Magilligan, W.B. Dade. 2010. Temporal and spatial scales of geomorphic adjustments to reduced competency following fow regulation in bedload-dominated systems. *Geomorphology* 64: 105-117.

Daniel, W.M., K.M. Brown. 2013. Multifactorial model of habitat, host fish, and landscape effects on Louisiana freshwater mussels. Freshwater Science 32(1): 193-203.

Daraio J.A., L.J. Weber. and T.J. Newton. 2010a. Hydrodynamic modeling of juvenile mussel dispersal in a large river: the potential effects of bed shear stress and other parameters. *Journal of the North American Benthological Society* 29 838–851.

Darty, T., S. T. Larned, K. Tockner. 2014. Intermittent Rivers: A Challenge for Freshwater Ecology. BioScience 64(3): 229-235.

Davis, M. B. and R. G. Shaw. 2001. Range shifts and adaptive responses to Quaternary climate change. Science 292: 673–679.

de Araújo, J. C. & Medeiros, P. H. A. 2013. Impact of dense reservoir networks on water resources in semiarid environments. *Australian Journal of Water Resources* 17(1): 87-100.

Dennis, S.D. 1984. *Distributional Analysis of the Freshwater Fauna of the Tennessee River System, with Special Reference to the Possible Limiting Effects of Siltation*. Thesis. Virginia Polytechnic Institute and State University, Blacksburg, VA

Doyle, M.W., E.H. Stanley, C.H. Orr, A.R. Selle, S.A. Sethi, J.M. Harbor. 2005. Stream ecosystem response to small dam removal: Lessons from the Heartland. Geomorphology 71:227-244.

Doyle, M.W., E.H. Stanley, J.M. Harbor. 2003. Channel adjustments following two dam removals in Wisconsin. *Water Resources Research* 39(1): 1-15.

Doyle, M. W., J.M. Harbor, E.H. Stanley. 2003a. Toward Policies and Decision-Making for Dam Removal. *Environmental Management* 31(4): 453-465.

Duchemin, M. 2009. Water, Power, and Tourism: Hoover Dam and the Making of the New West. *California History* 86(4): 60-78, 87-89.

Draut A.E., & A.C. Richie. 2015. Sedimentology of new fluvial deposits on the Elwha River, Washington, USA, Formed during large-scale dam removal. River Research and Applications 31: 42-61.

Dycus, J.N., J.M. Wisniewski, J.T. Peterson. 2015. The effects of flow and stream characteristics on the variation in freshwater mussel growth in a Southeast US river Basin. Freshwater Biology 60: 395-409.

Dynesius, M., and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. *Science* 266: 753-762.

East, A.E., G.R. Pess, J.A. Bounty, C.S. Magirl, A.C. Richie, J.B. Logan, T.J. Randle, M.C. Mastin, J.T. Minear, J.J. Duda, M.C. Liermann, M.L. McHenry, T.J. Beechie, P.B. Shaforth. 2015. Large-scale dam removal on the Elwha River, Washington, USA: River channel and floodplain geomorphic change. *Geomorphology* 228: 765-786.

Elderkin, C. L., A. D. Christian, J. L. Metcalfe-Smith, D. J. Berg. 2008. Population genetics and phylogeography of freshwater mussels in North America, *Elliptio diltata* and *Actinonaias ligamentina* (Bivalvia: Unionidae). Molecular Ecology 17: 2149-2163.

Elderkin, C. L., A. D. Christian, C. C. Vaughn, J. L. Metcalfe-Smith, D. J. Berg. 2007. Population genetics of the freshwater mussel, *Amblema plicata* (Say 1817) (Bivalvia: Unionidae): Evidence of high dispersal and post-glacial colonization. Conservation Genetics 8: 355-372.

Ellis, M.M. 1942. Fresh-water Impoundments. *Transactions of the American Fisheries Societies* 71(1): 80-93.

ESRI. 2014. ArcGIS Desktop: Release 10.2.1. Environmental Systems Research Institute, Redlands (C.A.).

ESRI. 2014a. How Hot Spot Analysis (Getis-Ord Gi*) works. ArcGIS Pro, ESRI. (last accessed 10 May 2016). http://pro.arcgis.com/en/pro-app/tool-reference/spatial-statistics/h-how-hot-spot-analysis-getis-ord-gi-spatial-stati.htm

ESRI. 2015. ArcGIS Desktop: Release 10.3.1. Environmental Systems Research Institute, Redlands (C.A.).

ESRI. 2016. *Optimized Hot Spot Analysis*. Environmental Systems Research Institute, Inc., ERSI. (last accessed 12 December 2016) http://desktop.arcgis.com/en/arcmap/10.3/tools/spatial-statistics-toolbox/optimized-hot-spot-analysis.htm

Federal Energy Regulation Commission (FERC). 2013. Small Hydro of Texas, Inc.; Notice of Application Accepted for Filing, Soliciting Comments, Motions To Intervene, and Protests. *Federal Register, The Daily Journal of the United States Government.* https://www.federalregister.gov/documents/2013/05/06/2013-10611/small-hydro-of-texas-inc-notice-of-application-accepted-for-filing-soliciting-comments-motions-to (last accessed on 9 July 2017).

Folger, P., B.A. Cody, and N. T. Carter. 2012. Drought in the *United States: Causes and Issues for Congress*. (Report No. 7-7500). Congressional Research Service, www.crs.gov. (CRS Publication No. RL34580).

Ford, N. B., J. Gullet, M. E. May. 2009. Diversity and abundance of unionid mussels in three sanctuaries on the Sabine River in northeast Texas. Texas Journal of Science 61: 279-294.

Ford, N. B., K. Heffentrager, D. F. Ford, A. D. Walters, and N. Marshall. 2014. Significant Recent Records of Unionid Mussels in Northeast Texas Rivers. Walkerana 17: 8-15.

Ford, D. F. and A. M. Oliver. 2015. The Known Potential Hosts of Texas Mussels: implications for Future Research and Conservation Efforts. Freshwater Mollusk Biology and Conservation 18: 1-4.

Freeman, M.C., C.M. Pringle, C.R. Jackson. 2007. Hydrologic connectivity and the Contribution of stream Headwaters to Ecological Integrity at Regional Scales. *Journal of the American Water Resource Association* 43(1):5-14.

Galay, V.J. 1983. Causes of river bed degradation. *Water Resources Research* 19, 1057-1090.

Galbraith H.S., D.E. Spooner, and C.C. Vaughn. 2010. Synergistic effects of regional climate patterns and local water management on freshwater mussel communities. *Biological Conservation* 143: 1175–1183.

Galbraith, H.S. and C.C. Vaughn. 2011. Effects of reservoir management on abundance, condition, parasitism, and reproductive traits of downstream mussels. *River Research and Applications 27:* 193-201.

Gates, K.K., C.C. Vaugh, J.P. Julian. 2015. Developing environmental flow recommendations for freshwater mussels using the biological traits of species guilds. *Freshwater Biolog*y 60(4): 620-635.

Gillis, P.L. 2012. Cumulative impacts of urban runoff and municipal wastewater effluents on wild freshwater mussels (*Lasmigona costata*). *Science of the Total Environment* 431: 348-356.

Gillis, P. L., S. K. Higgins, and M. B. Jorge. 2014. Evidence of oxidative stress in wild freshwater mussels (Lasmigona costata) exposed to urban-derived contaminants. *Ecotoxicology and Environmental Safety* 102: 62-69.

Gilman, S. 2016. *This Will Be the Biggest Dam-Removal Project in History*. National Geographic. http://news.nationalgeographic.com/2016/04/160411-klamath-glen-canyon-dam-removal-video-anniversary/ *(last accessed on 4 March 2010).*

Gangloff, M. M. 2013. Taxonomic and ecological tradeoffs associated with small dam removals. *Aquatic Conservation: Marine and Freshwater Ecosystems* 23: 475-480.

Gangloff, M.M., E.E. Hartfield, D.C. Werneke, J.W. Feminella. 2011. Associations between small dams and mollusk assemblages in Alabama streams. *Journal of the North American Benthological Society* 30(4): 1107-1116.

Gosnell and Kelly. 2010. Peace on the River? Social-Ecological Restoration and Large Dam Removal in the Klamath Basin, USA. *Water Alterations* 3(2): 361-383.

Gottgens, J. 2009. *Impact of the Removal of the Secor Road Dam on the Fish Community Structure and Composition in the Ottawa River, Ohio*. Toledo Metropolitan Area Council of Governments.

Gowan, C., K. Stephenson, L. Shabman. 2006. The role of ecosystem valuation in environmental decision making: Hydropower relicensing and dam removal on the Elwha River. *Ecological Economics* 56(4): 508-523.

Graf, D. L. 2002. Historical biogeography and late glacial origin of the freshwater pearly mussel (Bivalvia: Unionidae) faunas of Lake Erie, North America. *Occasional Papers on Mollusks, The Department of Mollusks, Museum of Comparative Zoology, Harvard University, Cambridge (M.A.)* 6: 175-211.

Graf, W. L. 1999. Dam nation: A geographic census of American dams and their large-scale hydrologic impacts. *Water Resources Research* 35(4): 1305-11.

Graf, W.L. 2001. Damage Control: Restoring the Physical Integrity of America's Rivers. *Annals of the Association of American Geographers* 91(1): 1-27.

Graf, W. L. 2005. Geomorphology and American dams: The scientific, social, and economic context. *Geomorphology* 71: 336 – 360.

Graf W.L. 2006. Downstream hydrologic and geomorphic effects of large dams on American Rivers. *Geomorphology* 79: 336-360.

Grant, G. E. & S. L. Lewis. 2015. The Remains of the Dam: What Have We Learned from 15 Years of US Dam Removal?. *Engineering Geology for Society and Territory* 3: 31-35.

Grant, C.E., J.C. Schmidt, S.L. Lewis. 2003. *A geological framework fo interpreting downstream effects of dams on rivers*. In: J.E. O'Connor, G.E. Grant eds., A Peculiar River: Geology, Geomorphology, and Hydrology, of the Deschutes River, Orgon. American Geophysical Union Press, Washington, D.C., pp. 203-219.

Guadalupe River Blanco Authority (GBRA). 2017. *Canyon Reservoir*. http://www.gbra.org/canyon/default.aspx (last accessed 6 June 2017).

Haag, W. R. 2012. *North American Freshwater Mussels – Natural History, Ecology, and Conservation*. Cambridge University Press, New York.

Haag, W. R., and M. L. Warren. 1998. Role of ecological factors and reproductive strategies in structuring freshwater communities. *Canadian Journal of Fisheries and Aquatic Sciences* 55: 297-306.

Hardison B.S. and J.B. Layzer.2001. Relations between complex hydraulics and the localized distribution of mussels in three regulated rivers. *Regulated Rivers: Research & Management* 17: 77–84.

Hartfield P.D. and D. Ebert. 1986. The mussels of southwest Mississippi streams. *American Malacological Bulletin* 4: 21–23.

Hauer, C. 2015. Review of Hydro-morphological management criteria on a river basin scale for preservation and restoration of freshwater pearl mussel habitats. *Limnologica* 50: 40-53.

Heinz Center. 2002. *Dam removal: Science and decision making*. Washington, D.C.: The H. John Heinz Center for Science, Economics and the Environment.

Hendrickson, Dean A., and Adam E. Cohen. 2015. *Fishes of Texas Project Database* (Version 2.0) doi:10.17603/C3WC70. (last accessed 6 May 2017).

Henry, T. 2013. How Prop 6 Passed, and What's Up Next for Water Projects in Texas. *StateImpact*. https://stateimpact.npr.org/texas/2013/11/05/texas-water-fund-passes/ (last accessed 10 June 2017).

Hewitt, G. 2000. The genetic legacy of the Quaternary ice ages. *Nature* 405: 907–913.

Hill, R.A., M.H. Weber, S.G. Leibowitz, A.R. Olsen, D.J. Thornbrugh. 2016. The Stream-Catchment (StreamCat) Dataset: A Database of Watershed Metrics for the conterminous United States. *Journal of the American Water Association* 52(1): 120-128.

Hirsch, R.M., J.F. Walker, J.C. Day, R. Kallio. 1990. The influences of man on hydrologica systems. In: M.G. Wolman, H.C. Riggs, eds., *Surface Water Hydrology*. Geological Society of America, Boulder, CO, pp. 329-359.

Hoenke, K.M., M. Kumar, L. Batt. 2014 A GIS based approach for prioritization dams for potential removal. *Ecological Engineering* 64:27-36.

Hogg, R.S., S.M. Coghlan Jr., J. Zydelewski, C. Gardner. 2015. Fish Community Response to a Small-Stream Dam Removal in a Maine Coastal River Tributary. *Transactions of the American Fisheries Society* 144(3): 467-479.

Houp, R.E. 1993. Observations on Long-term Effects of Sedimentation on Freshwater Mussels (Mollusca: Unionidae) in the North Fork of Red River, Kentucky. *Transactions of the Kentucky Academy of Science* 54((3-4): 93-97.

Howells, R.G., C.M. Mather, and J.A.M. Bergmenn. 2000. Impacts of dewatering and cold on freshwater mussels (Unionidae) in B.A. Steinhagen Reservoir, Texas. *Texas Journal of Science* 52:93–104.

Huang, L. and Z. Zhang. 2015. The Stability of Revegetated Ecosystems in Sandy Areas: An Assessment and Predication Index. *Water* 7: 1969-1990.

IBM SPSS Statistics 22. 2014. Chicago, IL: IBM SPSS Inc.

International Commission on Large Dams (ICOLD). 2007. *Dams for Human Sustainable Development*. International Commission on Large Dams.

Jaeger, K.L., J.D. Olden, N.A. Pelland. 2014. Climate Change poised to threaten hydrologic connectivity and endemic fishes in dryland streams. *PNAS* 111(38): 13894 – 13899.

Jager, H.I., A. Chandler, K.B. Lepla, W.V. Winkle. 2001. A theoretical study of river fragmentation by dams and its effects on white sturgeon populations. *Environmental Biology* 60:347-361.

Januchowski-Hartley, S.R., P.B. McIntyre, M. Diebel, P.J. Doran, D.M. Infante, C. Joseph, J.D. Allan. 2012. Restoring aquatic ecosystem connectivity requires expanding inventories of both dams and road crossings. *Frontiers in Ecology and the Environment*. doi: 10.1890/120168

Jones, J.W., R.J. Neves, E.M. Hallerman. 2015. Historical demography of freshwater mussels (Bivalvia: Unionidawe): genetic evidence for population expansion and contraction during the late Pleistocene and Holocene. *Biological Journal of the Linnean Society* 144: 376-397.

Junk, W.J., P.B. Bayley, R.E. Sparks. 1989. The Flood Pulse Concepts in Tiver Floodplain Systems. *In* D. P. Dodge (ed.) *Proceedings of the International Large River Symposium. Canadian Special Publish Dish and Aquatic Science*. 106: 110 – 127.

Kappes, H. and P. Haase. 2012. Slow, but steady: dispersal of freshwater mollusks. *Aquatic Sciences* 74: 1-14.

Karatayev, A.Y., L.E. Burlakova, T.D. Miller, and M.F. Perrelli. 2015. Reconstructing historical range and population size of an endangered mollusc: long-term decline of Popenaias popeii in the Rio Grande, Texas. *Hydrobiologia* 1-17.

Karatayev, A.Y., T.D. Miller, and L.E. Burlakova. 2012. Long-term changes in unionid assemblages in the Rio Grande, one of the World's top 10 rivers at risk. Aquatic Conservation: *Marine Freshwater Ecosystems* 22: 206–219.

Katz, G.L., J.M. Friedman, S.W. Beatty. 2005. Delayed effects of flood control on a flood-dependent riparian forest. *Ecological Applications* 15: 1019-1035.

Kocovsky, P.M., R.M. Ross, D.S. Dropkin, J.M. Campbell. 2008. Linking Landscapes and Habitat Suitability Scores for Diadromous Fish Restoration in the Susquehanna River Basin. *North American Journal of Fisheries Management* 28: 906-918.

Konrad, C.P. 2009. Simulating the recovery of suspended sediment transport and river-bed stability in response to dam removal on the Elwha River, Washington. *Ecological Engineering* 35(2009): 1104-1115.

Krebs, R. A., W. C. Borden, E. R. Steiner, M. S. Lyons, W. Zawiski, B. M. Walton. 2010. Determinants of mussel diversity in Lake Erie tributaries. *Journal of the North American Benthological Society* 29: 506-520.

Kruskal, J.B. 1964. Nonmetric multidimensional scaling: a numerical method. *Psychometrika* 29:115-119.

Kuby, M.J., W.F. Fagan, C.S. ReVelle, W.L. Graf. 2005. A multiobjective optimization model for dam removal: an example trading off salmon passage with hydropower and water storage in the Willamette basin. *Advances in Water Resources* 28: 845-855.

Lautz, L.K., D.I. Siegel., R.L. Bauer. 2006. Impact of debris dams on hyporheic interaction along a semi-arid stream. *Hydrologic Processes* 20(1): 183-196.

Lydeard C., R.H. Cowie, W.F. Ponder, A.E. Bogan, P. Bouchet, S.A. Clark et al. 2004. The global decline of nonmarine mollusks. *BioScience*, 54, 321–330.

Mader, H., & C. Maier. 2008. A method for prioritizing the reestablishment of river continuity in Austrian rivers. *Hydrobiologia* 609:277-288.

Magilligan, F.J., B.E. Graber, K.H. Nislow, J.W. Chipman, C.S. Sneddon, C.A. Fox. 2016. River Restoration by dam removal: Enhancing connectivity at watershed scales. *Science of the Athropocene,* doi: 10.12952/

Magilligan, F.J. and K.H. Nislow. 2005. Changes in hydrologic regime by dams. *Geomorphology* 71: 61 – 78.

Magillian, F.J. and K.H. Nislow. 2001. Long-term changes in regional hydrologic regime following impoundment in a humid-climate watershed. *Journal of American Water Resources Association* 37: 1551-1569.

Mahmoud, M.R. & L.A. Garcia. 2000. Comparison of different multicriteria evaluation methods for the Red Bluff diversion dam. *Environmental Modeling & Software* 15: 471-478.

Malczewski, J. 2006. GIS-based multicriteria decision analysis: a survey of the literature. *International Journal of Geographical Information Science* 20(7): 703-726.

Malvadkar, U., F. Scatena, M. Leon. 2015. A Comparison of Connectivity Metrics on Watersheds and Implications for Water Management. *River Research and Applications* 31: 256-267.

Mantel, S.K., D.A. Hughes, N.WJ Muller. 2010. Ecological impacts of small dams on South African rivers Part 1: drivers of change – water quantity and quality. *Water SA (online)* 36 (3): 1-13. ISSN 1816-7950

Mantel, S.K., N.WJ Muller, D.A. Hughes. 2010a. Ecological impacts of small dams on South African rivers Part 2: Biotic response – abundance and composition of macroinvertebrate communities. *Water SA* 36(3): 361-370.

Martin, E. H. and Apse, C.D. 2013. *Chesapeake Fish Passage Prioritization: An Assessment of Dams in the Chesapeake Bay Watershed*. The Nature Conservancy, Eastern Division Conservation Science. http://maps.tnc.org/erof_ChesapeakeFPP (last accessed 6 June 2017).

Martin, E.H. and C. D. Apse. 2011. Northeast Aquatic Connectivity: An Assessment of Dams on Northeastern Rivers. The Nature Conservancy, Easter Freshwater Program.

Mather, P.M. 1976. *Computational methods of multivariate analysis in physical geography*. J. Wiley & Sons, London.

Maxwell, R. J., 2012. Patterns of endemism and species richness of fishes of the Western Gulf Slope. Thesis.

McCluney, K.E., N.L. Poff, M.A. Palmer, J.H. Thorp, G.C. Poole, B.S. Williams, M.R. Williams, J.S. Baron. 2014. Riverine macrosystems ecology: sensitivity, resistance, and resilience pf whole river basins with human alterations. *Macrosystems Ecology* 12(1): 48-58.

McCune, B., J.B. Grace, D.L. and Urban. 2002. *Analysis of ecological communities* (Vol. 28). Gleneden Beach, OR: MjM software design.

McKay, L., T. Bondelid, T. Dewald, J. Johnston, R. Moore, and A. Rea. 2012. *NHDPlus Version 2: User Guide* Washington, D.C. Environmental Protection Agency (EPA).

Melillo, J. M., T. C. Richmond, G. W. Yohe, Eds., 2014. Climate Change Impacts in the United States: The Third National Climate Assessment. U.S. Global Change Research Program, 841 pp. doi:10.7930/J0Z31WJ2.

Merrit, D.M. and E.E. 2006. Plant dispersal along rivers fragmented by dams. *River Research and Applications* 22: 1-26.

Morita, K. S. Yamamoto. 2002. Effects of Fragmentation by Damming on the Persistence of Stream-Dwelling Charr Populations. *Conservation Biology* 16(5).

Morley, S.A., J.J. Duda, H.J. Coe, K.K. Kloehn, M.L. McHenry. 2008. Banthic Invertebrates and Periphyton in the Elwha River Basin: Current Conditions and Predicted Response t Dam Removal. *Northwast Sceience 82*(sp1): 179-196.

Nathan, L. Lowe. 2012. The hydrologic impacts of farm dams. *Australian Journal of Water Resources* 16(1): 75-83.

Neck, R. W. 1982. Preliminary Analysis of the Regional Zoogeography of the Freshwater Mussels of Texas. *Proceedings of the Symposium on Recent Adjacent States.*

Nunn, A.D., I.G. Cowx. 2012. Restoring River Connectivity: Prioritizing Passage Improvements for Diadromous Fishes and Lampreys. *AMBIO*, 41: 402-409.

Orr, C.H., S.J. Kroiss, K.L. Rogers, E.H. Stanley. 2008. Downstream Benthic Responses to Small Dam Removal in a Coldwater Stream. *River Research and Applications* 24: 804-822.

Park, D., M. Sullivan, E. Bayne, G. Scrimgeour. 2008. Landscape-level stream fragmentation caused by hanging culverts along roads in Alberta's boreal forest. *Canadian Journal of Forest Resources*, 38: 566-575.

Perkin, J. S., Z. R. Shattuck, J. E. Gerken, and T. H. Bonner. 2013. Fragmentation and drought legacy correlated with distribution of Burrhead Chub in subtropical streams of North America. *Transactions of the American Fisheries Society* 142:1287-1298.

Petts, G.E. and A.M. Gurnell. 2005. Dams and geomorphology: Research progress and future directions. *Geomorphology* 71: 27 – 47.

Poff, N.L., J.D. Allen, M.B.Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E.Sparks, J.C. Stromberg. 1997. The Natural Flow Regime. American Institute of Biological Sciences 47(11):769-784.

Poff, L. N. & D. D. Hart. 2002. How Dams Vary and Why It Matters for the Emerging Science of Dam Removal. *BioScience* 52(8): 659-668.

Poff N.L., J.D. Olden, D.M. Merritt and D.M. Pepin. 2007. Homogenization of regional river dynamics by dams and global biodiversity implications. *Proceedings of the National Academy of Sciences of the United States of America* 104: 5732–5737.

Pohl, M.M. 2002. Bringing Down Our Dams: Trends in American Dam Removal Rationales. *Journal of the American Water Resources Association* 38(6): 1511 – 1519.

PRISM Climate Group, Oregon State University (PRISM). 2013. http://prism.oregonstate.edu (last accessed 5 April 2014).

Randklev, C. R., N. Ford, S. Wolverton, J.H. Kennedy, C. Robertson, K. Mayes, and D. Ford. 2015. The influence of stream discontinuity and life history strategy on mussel community structure: a case study from the Sabine River, Texas. *Hydrobiologia* 770(1): 1-19.

Randklev, C.R., E.T. Tsakiris, M.S. Johnson, J. Skorupski, L.E. Burlakova, J. Groce, and N. Wilkins. 2013. Is False Spike, Quadrula mitchelli (Bivalvia: Unionidae), extinct? First account of a very-recently deceased individual in over thirty years. *The Southwestern Naturalist* 58: 268-273.

Randklev C.R., M.S. Johnson, E.T. Tsakiris, J. Groce and N. Wilkins. 2013a. Status of the freshwater mussel (Unionidae) communities of the mainstem of the Leon River, Texas. *Aquatic Conservation: Marine and Freshwater Ecosystems* 23: 390–404.

Randklev, C.R., B.J. Lundeen, R.G. Howells, and J.H. Kennedy. 2010. Habitat preference and first account of a living population of Texas Fawnsfoot, Truncilla macrodon (Bivalvia: Unionidae) In the Brazos River. *The Southwestern Naturalist* 55:297-298.

Rashleigh, B. 2008. Nestedness in riverine mussel communities: patterns across sites and fish hosts. *Ecography* 31: 612-619.

Reinhardt, B.H. 2011. Drowned Towns in the Cold War West: Small Communities and Federal Water Projects. *Western Historical Quarterly* 42: 149-172

Renoflat, B.M., A.G.C. Lejon, M Jonsson, C. Nilsson. 2013. Long-Term Taxon Specific Responses of Macroinvertebrates to Dam Removal in a Mid-sized Swedish Stream. *River Research and Applications* 29: 1082-1089.

Richter B.D., D.P. Braun, M.A. Mendelson, L. L. Master. 1997. Threats to Imperiled Freshwater Fauna. *Conservation Biology* 11(5): 1081-1093.

Reisner, M. 1986. *Cadillac Desert: The American West and Its Disappearing Water*. New York, NY: Viking.

Riggsbee, J.A., J.P. Julian, M.W. Doyle, R.G. Wetzel. 2007. Suspended sediment, dissolved Organic carbon, and dissolved nitrogen export during dam removal process. *Water Resources Research* 43: 1-16.

Roberts, J.H., P.L. Angermeier, E.M. Hallerman. 2013. Distance, dams and drift: what structures populations of an endangered, benthic stream fish? *Freshwater Biology* 58: 2050-2064.

Roberts, J.J., K. D. Fausch, D.P. Peterson, M.B. Hooten. 2013a. Fragmentation and thermal risks from climate change interact to affect persistence of native trout in the Colorado River basin. *Global Change Biology* 19: 1383-1398.

Saaty, T.L. 1980. *The Analytic Hierarchy Process*. New York, NY: McGraw-Hill.

Sansom, A. 2008. *Water in Texas: An Introduction*. Austin, TX: University of Texas Press.

Sawyer, A.H., M.B. Cardenas, A. Bomar, M. Mackey. 2009. Impact of dam operations on hyporheic exchange in the riparian zone of a regulated river. *Hydrologic Processes* 23(15): 2129-2137.

Scheder, C., B. Lerchegger, P. Flodl, D. Csar, C. Gumpinger, C. Hauer. 2015. River bed stability versus interstitial: Depth-development accumulation of substances in freshwater pearl mussel (Margaritifera margaritifera L.) habitats in Austrian streams as a function of hydromorphologcia parameters. *Limnologica* 50: 29-39.

Schwalb, A. N., T. J. Morris, K. Cottenie. 2015. Dispersal abilities of riverine freshwater mussels influence metacommunity structure. *Freshwater Biology* 60: 911-921.

Schwalb, A. N., T. J. Morris, N. E. Mandrak, K. Cottenie. 2012. Distribution of unionid freshwater mussels depends on the distribution of host fishes on a regional scale. *Diversity and Distributions* 19: 446-454.

Sethi, S.A., A.R. Selle, M.W. Doyle, E.H. Stanley, H.E. Kitchel. 2004. Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin (USA). *Hydrobiologia* 525: 157-165.

Singley, J.A., 1893. Contributions to the natural history of Texas. Part 1. Texas Mollusca. Geological Survey of Texas, 4th annual report. pp. 299-343.

Smith, S.V., W.H. Renwick, J.D. Bartley, R.W. Buddemeier. 2002. *The Science of the Total Environment* 299: 21-36.

Solomon, S. 2010. *Water: the epic struggle for wealth, power, and civilization*. New York, NY: HarperCollins Publishers.

Stanford, J.A., J.V. Ward. 2001. Revisiting the Serial Discontinuity Concept. *Regulated Rivers: Research & Management* 17: 303-310.

Stanford University. 2017. *National Performance of Dams Program*. http://npdp.stanford.edu/home (last accessed 11 July 2017).

Stanley, E.H. and M.W. Doyle. 2003. Trading off: the ecological effects of dam removal. *Frontiers in Ecological Environments* 1: 15-22.

Stanley, E.H., M.A. Luebke, M.W. Doyle, D.W. Marshall. 2002. Short-term \ changes in channel form and macroinvertebrate communities following low head dam removal. *Journal of North American Benthological Society* 21(1): 172-187.

Sterling, K.A., D.H. Reed, B.P. Noonan, M.L. Warren Jr. 2012. Genetic effects of habitat fragmentation and population isolation on Etheostoma raneyi (Percidae). *Conservation Genetics* 13:859-872.

Store, R. & J. Kangas. 2001. Integrating spatial multi-criteria evaluation and expert knowledge for GIS-based habitat suitability modelling. *Landscape and Urban Planning* 55: 79-93.

Strayer, D. L., 2008. *Freshwater Mussel Ecology: A Multifactor Approach to Distribution and Abundance*. University of California Press, Berkley (N.Y.).

Strayer D.L. 1999. Use of flow refuges by unionid mussels in rivers. *Journal of the North American Benthological Society* 18: 468–476.

Strecker, J., 1931. The distribution of naiades or pearly fresh-water mussels of Texas. *Baylor University Museum Bulletin.*

Texas Commission on Environmental Quality (TCEQ). 2017. *The Clean Rivers Program*. https://www.tceq.texas.gov/waterquality/clean-rivers (last accessed 7 May 2017).

Texas Commission on Environmental Quality (TCEQ). 2014. Dams.gdb.

Texas Parks and Wildlife Department (TPWD). 2016. *Federally and State Listed Species in Texas*. Texas Parks and Wildlife Department, Austin (T.X.). https://tpwd.texas.gov/huntwild/wild/wildlife_diversity/nongame/listed-species/ (last accessed 8 March 2017).

Texas Water Development Board (TWDB). 2017. *Water for Texas 2017 State Water Plan*. Austin, TX: Texas Water Development Board (TWDB).

Texas Water Development Board (TWDB). 2012. *Water for Texas 2012 State Water Plan*. Austin, TX: Texas Water Development Board (TWDB).

Texas Water Development Board (TWDB). n.d. *Major River Basins*. http://www.twdb.texas.gov/mapping/gisdata.asp. (last accessed 16 November 2014).

Thorp, J.H. and A.P. Covich. 2010. Ecology and classification of North American freshwater invertebrates. 3. Academic Press, Elsevier, San Diego, California, USA.

Thomson, J.R., D.D. Hart, D.F. Charles, T.L. Nightengale, D.M. Winter. 2005. Effects of removal of a small dam on downstream macroinvertebrate and algal assemblages in a Pennsylvania stream. *Journal of the North American Benthological Society* 24(1): 192-207.

Tiemann, J.S., H.R. Dodd. N. Owens, D.H. Wahl. 2007. Effects of Lowhead Dams on Unionids in the Fox River, Illinois. *Northeastern Naturalist* 14(1): 125-138.

Troia, M. J., L. R. Williams, M. G. Williams, N. B. Ford. 2015. The process domains concept as a framework for fish and mussel habitat in a coastal plain river of southeastern North America. *Ecological Engineering* 75: 484-496.

Tsuboi, J., S. Endou, K. Morita. 2010. Habitat fragmentation by damming threatens coexistence of stream-dwelling charr and salmon in the fuji River, Japan. *Hydrobiologia* 650: 223-232.

Tullos, D.D., D.S. Finn, C. Walker. 2014. Geomorphic and Ecological Disturbance and Recovery from Two Small Dams and Their Removal. *PLOSone* 9(9): 1-18.

United States Army Corps of Engineers (USACE). 2016. *National Inventory of Dams (NID)*. Washington D.C. http://geo.usace.army.mil/pgis/f?p=397:1:1580576594309::NO (last accessed 5 May 2015).

United State Census Bureau (U.S. Census Bureau). 2010. *2010 Census of Population*. QuickFacts. https://www.census.gov/quickfacts/ (last accessed 6 June 2017).

United States Census Bureau (U.S. Census Bureau). 2016. *TIGER/Line® Shapefiles*. http://www.census.gov/cgi-bin/geo/shapefiles2013/main (last accessed 6 April 2016). *Water Plan*. Austin, TX: Texas Water Development Board (TWDB).

United States Drought Monitor (U.S. Drought Monitor). 2012. *Drought Monitor Archive*. http://droughtmonitor.unl.edu/archive.html (last accessed 19 September 2012).

United States Geologic Survey (USGS). 2017. *National Hydrography Dataset*. https://nhd.usgs.gov/NHD_High_Resolution.html (last accessed 10 May 2017).

Vannote R.L. and G.W. Minshall. 1982. Fluvial processes and local lithology controlling abundance, structure, and composition of mussel beds. *Proceedings of the National Academy of Science* 79: 4103–4107.

Vannote, R.L., G.W. Minshall, K.W. Cummins, J.R. Sedell, C. E. Cushing. 1980. The River Continuum Concept. *Canadian Journal of Fish and Aquatic Science*. 37: 130-137.

Vaughn, C. C., C. L. Atkinson, J. P. Julian. 2015. Drought-induced changes in flow regimes lead to long-term losses in mussel-provided ecosystem services. *Ecology and Evolution* 5: 1291-1305.

Vaughn, C.C. 2012. Life history traits and abundance can predict local colonization and extinction rates of freshwater mussels. *Freshwater Biology* 57: 982-992.

Vaughn, C. C. and C. M. Taylor. 2000. Macroecology of a host- parasite relationship. *Ecography* 23: 11-20.

Vaughn, C. C. 1997. Regional patterns of mussel species distributions in North American rivers. *Ecography* 20: 107-115.

Vogel, R.M., I.W. Wilson, and C. Daly. 1999. Regional Regression Models of Annual Streamflow for the United States. *Journal of Irrigation and Drainage Engineering* 125(3):148-157.

von Schiller, D., V. Acuña, D. Graeber, E. Martí, M. Ribot, S. Sabater, X. Timoner, K. Tockner. 2011. Contraction, fragmentation and expansion dynamics determine nutrient availability in a Mediterranean forest system. *Aquatic Science* 73:485-497

Ward, J.V. 1989. The Four-Dimensional Nature of Lotic Systems. *Journal of North American Benthological Society* 8 (1): 2 – 8.

Ward, J.V. and J.A. Stanford. 1983. The Serial Discontinuity of Lotic Ecosystems. In *Dynamics of Lotic Ecosystems,* ed. T.D. Fontaine, S.M. Bartell, 29-42. Ann Arbor, MI: Science Publications.

Watters, G.T. 2000. Freshwater mussels and water quality: A review of the effects of hydrological and instream habitat alterations. *Proceedings of the First Freshwater Mollusk Conservation Society Symposium* 1999: 261-274.

Watters, G.T. 1995. Small Dams as Barriers to Freshwater Mussels (Bivalviam, Unionoida) and their Hosts. *Biological Conservation* 75(1996): 79-85.

Watters, G. T. 1992. Unionids, fish and the species-area curve. *Journal of Biogeography* 19: 481-490.

Werth, S. M. Schödl, C. Scheidedder. 2014. Dams and canyons disrupt gene flow among populations of a threatened riparian plant. *Freshwater Biology* 59: 2502-2515.

World Commission on Dams (WCD). 2000. *DAMS AND DEVELOPMENT: A N NEW FRAMEWORK*. Sterling, VA: Earthscan Publications Ltd.

National Wild and Scenic Rivers System (WSR). n.d. *National Wild and Scenic Rivers System*. http://www.rivers.gov/index.php (last accessed 8 January 2016)

Warrick, J.A., J.A. Bountry, A.E. East, C.S. Magirl, T.J. Randle, G. Gelfenbaum, A.C. Ritchie, G.R. Pess, W. Leung, J.J. Duda. 2015. Large-scale dam removal on the Elwha River, Washington, USA: Source-tosinl sediment budget and synthesis. *Geomorphology* 246:729-750.

Whittaker, R.H. 1960. Vegetation od Siskiyou Mountains, Oregon and California. *Ecological Monograohs* 30: 279-338.

Williams, G.P. and M.G. Wolman. 1984. *Effects of Dams and Resevoirs on Surface-Water Hydrology; Changes in River Downstream from Dams*. U.S. Geologic Survey Professional Paper 1286, 83p.

Wofford, J.E., R.E., Gresswell, M.A. Banks. 2005. Influence of barriers to movement on within watershed genetic variation of coastal cutthroat trout. *Ecological Applications* 15(2): 628-637.

Wyrick, J.R., B.A. Rischman, C.A. Burke, C. McGee, C. Williams. 2009. Using hydraulic modeling to address social impacts of small dam removals in southern New Jersey. *Journal of Environmental Management* 90 (2009): 5270-5278.

Zanatta, D. T., A. T. Harris. 2013. Phylogeography and Genetic Variability of the Freshwater Mussels (Bivalvia Unionidae) Ellipse, *Venustaconcha ellipsiformis* (Conrad 1836), and Bleeding Tooth, *V. Pleasii* (Marsh 1891). *American Malacological Bulletin* 31:267-279.

Zanatta, D. T., R. W. Murphy. 2008. The phylogeographical and management implications of genetic population structure in the imperiled snuffbox mussel, *Epioblasma triquetra* (Bivalvia: Unionidae). *Biological Journal of the Linnean Society* 93: 371-384.

Zheng, P.Q., B.F. Hobbs, J.F. Koonce. 2009. Optimizing multiple dam removals under multiple objectives: Linkign tributary habitat and the Lake Erie ecosystem. *Water Resources Research* 45 1-14.

Zomer RJ, A. Trabucco, D.A. Bossio, O. van Straaten, L.V. Verchot. 2008. Climate Change Mitigation: A Spatial Analysis of Global Land Suitability for Clean Development Mechanism Afforestation and Reforestation. Agric. *Ecosystems and Environment* 126: 67-80.

Zomer RJ, D.A. Bossio, A. Trabucco, L. Yuanjie, D.C. Gupta and V.P. Singh. 2007. Trees and Water: *Smallholder Agroforestry on Irrigated Lands in Northern India (IWMI Research Report 122)*. Colombo, Sri Lanka: International Water Management Institute. p 45