IDENTIFYING TRADEOFFS BETWEEN RIVERINE CONNECTIVITY AND MUNICIPAL WATER SUPPLY IN PUERTO RICO

by

JESSICA CLAIRE CHAPPELL

(Under the Direction of CATHERINE M. PRINGLE)

ABSTRACT

Variability in freshwater supply creates challenges for managers balancing social and ecological water demand worldwide. Management actions are often focused on maintaining a consistent domestic water supply with minimal consideration of corresponding tradeoffs in freshwater ecosystems. This dissertation examines social, ecological, and engineering components through three socio-ecological studies focused on Puerto Rico's freshwater system. First, I evaluated long-term effects of low-head dams paired with water intakes on longitudinal riverine connectivity for freshwater shrimp (Chapter 2) and four other migratory animals (snails, gobies, mountain mullet, and American eel; Chapter 3) in northeastern Puerto Rico. An analysis of stream discharge and water intake rates show decreased habitat connectivity for native shrimp as a result of intakes and dams constructed over the last 3-4 decades. Conversely, other migratory animals, including snails, eel, and mullet, are limited by barrier height, which reduces their overall habitat availability but also makes their connectivity less sensitive to increases in water withdrawal rates. These studies emphasize the importance of water managers considering downstream and upstream temporal variation in connectivity, especially in relation to the sustainability of diadromous aquatic fauna metapopulations. Second, Puerto Rico's environmental flow law and practice are evaluated through interviews with state and federal water managers (Chapter 4). Mismatches in *de jure* (legislative directives) and *de facto* (execution of legislation) water management can be partially explained by the political accountability of state agencies, restricting environmental flow legislation enforcement. Third, a socio-ecological systems lens is applied to examine the resilience of stream ecosystems and the municipal water supply to extreme climatic events, specifically hurricanes and droughts (Chapter 5). Using systems models and historical event analysis, this study concludes that some historical management actions designed to increase social drought resilience has ultimately reduced overall system resilience when viewed relative to both infrastructure and ecological outcomes. Overall, this dissertation informs a holistic approach to freshwater management in Puerto Rico by examining the role of temporal variability in ecological connectivity; differences in water infrastructure effects across several taxa; limitations to water management legislation enforcement; and feedbacks across social and ecological processes that undermine and maintain system-wide resilience.

INDEX WORDS: Socio-ecological systems, Management tradeoffs, Longitudinal Riverine Connectivity Index, Environmental flows, Accountability, Stream community, Amphidromous species, Integrative conservation, Puerto Rico

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DEDICATION

I dedicate this dissertation to my family who supported me at my defense and throughout my life. To my dad, for telling me I could do anything. To my mom, for teaching me to always go after what I want. To my brother, for showing me how to open up and let people in. To my aunt Denise, for showing me how to make the life I want. To my grandmother Moddie, for being an inspirational pod leader and reminding me to be proud of hard work. To my cat Chevere, for showing me how to enjoy the simple things. And to Ashley Block, for being a fabulous human being and reminding me there’s always time to be kind and enjoy life.

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CHAPTER 1

INTRODUCTION

1. Introduction

Multiple competing users must share increasingly limited freshwater supplies (Poff et al. 2003, Hirsch et al. 2011), resulting in freshwater management decision tradeoffs (Gunderson 2000, Dahlberg and Burlando 2009). Human and stream ecosystem dependence on freshwater quantity creates a coupled socio-ecological system (SES) with multiple feedbacks across compartments (Wang and Blackmore 2009, Rodina 2019). The SES represent a framework for researchers and managers to explore solutions to achieve multiple outcomes (i.e. water supply and ecosystem sustainability) (McGinnis and Ostrom 2014). However, understanding how a freshwater SES can be sustainably managed requires an integrative approach to explore the potential tradeoffs involved with proposed management actions, especially to identify "hidden costs" that may not be immediately apparent (McShane et al. 2011, Huntsinger et al. 2017). In my dissertation, I use my integrative training to explore the social and ecological tradeoffs in water management and the associated implications for water SES resilience in Puerto Rico.

Balancing municipal water demand, while maintaining functioning stream ecosystems, is a complex issue. Municipal needs are prioritized when water resources are limited (Annear et al. 2004: 73); however, streams depend on some level of discharge to maintain their processes and functions (Arthington et al. 2010). One of the main processes that streamflow facilitates is hydrologic connectivity. Hydrologic connectivity is defined as "the water-mediated transport of matter, energy, and/or organisms within or between elements of the hydrologic cycle" (Pringle

2001). Hydrologic connectivity is especially important for diadromous aquatic animals, which migrate between freshwater and marine habitats to complete their life cycle, including amphidromous species. Amphidromous animals reproduce in freshwater, migrate to a saline environment to develop, and then return to freshwater (McDowall 2009). Dams and intake construction have reduced habitat connectivity for multiple amphidromous species (Holmquist et al. 1998, Morita and Yamamoto 2002, Cooney and Kwak 2013).

Although nearly all large river ecosystems worldwide have experienced fragmentation from dams and infrastructure (Nilsson et al. 2005), I focus on exploring water management tradeoffs in Puerto Rico given the importance of streams for freshwater supply and the dominance of diadromous species in the stream animal community. High spatial and temporal variability in water availability has led to the construction of reservoirs across the island that have associated ecological consequences (March et al. 2003, Greathouse et al. 2006). Although tradeoffs between human and ecological needs will not always result in a win-win scenario, management actions should ensure the most important properties of each sector are protected and maintained (Kareiva 2012). This dissertation informs effective water management practices by quantifying the effect of dams on temporal riverine connectivity for multiple diadromous species, evaluating enforcement limitations for environmental flow legislation, and identifying socio-ecological resilience tradeoffs of management actions in response to extreme climatic events.

2. Chapter overview

2.1 Chapter 2

My objective is to examine cumulative effects of low-head dams on longitudinal riverine connectivity for freshwater shrimps through time and space in northeastern Puerto Rico. I apply a modified version of the Index of Longitudinal Riverine Connectivity (ILRC; Crook et al. 2009) to man-made barriers in seven watersheds over a 37-year discharge dataset. I then scale the ILRC values to the region (i.e. across all seven focal watersheds) to better understand how shrimp habitat connectivity varies through time and space. Freshwater shrimps are genetically linked across large spatial scales (Cook et al. 2009), supporting an evaluation of habitat connectivity across watersheds. Additionally, I evaluate the effect of low-head dams on refugia habitat, as freshwater shrimp can escape from predators when upstream of barriers 5 m high (Covich et al. 2009). This chapter demonstrates the importance of considering temporal and spatial fluctuations in connectivity when evaluating ecological impacts.

2.2 Chapter 3

Building on analyses conducted for freshwater shrimp (Chapter 2), in this chapter I evaluate how altering water withdrawal rates influences habitat connectivity for five migratory taxa (i.e. shrimp, goby, snail, mountain mullet, and American eel) in four watersheds in northeastern Puerto Rico. I further modify the ILRC (Crook et al. 2009) to include characteristics important to the focal taxa (i.e. elevation and barrier height; Blanco and Scatena 2005, Cooney and Kwak 2013) and use a 31-year discharge dataset (1986-2016) to evaluate connectivity. I explore changes in connectivity for each taxonomic group for four water withdrawal management scenarios, accounting for differences among taxa in abilities to pass barriers and vulnerability to withdrawals. Finally, I evaluate which taxonomic groups experience the greatest losses in habitat connectivity with increasing water withdrawal rates. This chapter illustrates how altering water withdrawals may be an effective management option to increase habitat connectivity for some migratory taxa.

2.3 Chapter 4

Having quantified actual and potential effects of water withdrawals on migratory stream fauna, I shift gears to examine whether environmental flow legislation exists and how it is implemented in Puerto Rico. Specifically, I focus on two water management targets related to environmental flows: equitable allocation and water use efficiency. I use a framework to identify whether these targets are present in statutes (*de jure* management) and how the targets are achieved in practice (*de facto* management). I then evaluate whether mismatches exist between *de jure* and *de facto* water management (Fine et al. 2017) and use accountability for state agencies to create a network of influence (Black 2008) to understand implementation limitations. This chapter contributes to the ongoing conversation of what characteristics allow environmental flow legislation to be effectively implemented (Pahl-Wostl et al. 2013).

2.4 Chapter 5

In this chapter, I explore how water management actions taken to maintain municipal water supply interact with feedbacks in the water SES that maintain resilience to hurricanes and droughts. I explore the resilience of the water SES across Puerto Rico through a systems approach to understand how social and ecological components interact in response to hurricanes and droughts (Rodina 2019). Once I establish the SES feedbacks, I explore historical management actions used to increase social resilience to drought (i.e. maintain municipal water supply) and how these actions influence overall SES resilience. This chapter highlights the necessity of an integrative effort to understand how overall water SES resilience can be sustainably managed.

2.5 Chapter 6

Finally, I summarize my findings in Chapter 6 and propose future studies. I also reflect on my path to becoming an integrative conservation researcher and articulate how my research is integrative.

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CHAPTER 2

LONG-TERM (37 YEARS) IMPACTS OF LOW-HEAD DAMS ON FRESHWATER SHRIMP HABITAT CONNECTIVITY IN NORTHEASTERN PUERTO RICO[1]

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Abstract

Freshwater migratory shrimp in Puerto Rico depend on watershed connectivity, from stream headwaters to the ocean, to complete their life cycle. Moreover, shrimp populations in different watersheds have been shown to be connected in an island-wide metapopulation. However, low-head dams paired with water intakes on streams draining the El Yunque National Forest (EYNF) reduce streamflow, intermittently reducing shrimp habitat connectivity. Here, we examine the cumulative effects of low-head dams on habitat connectivity over a 37-year period across seven EYNF watersheds. We calculated *total* and *refugia habitat connectivity* (where refugia habitat is defined as predator-free upstream reaches above waterfalls > 5 m in height) at a monthly time step using a *habitat-weighted index of longitudinal riverine connectivity*, which incorporates location and operation of water intakes and streamflow variability. Findings indicate total and refugia habitat connectivity declined over 37 years (by 27% and 16%, respectively), as the proportion of water withdrawn increased relative to stream discharge. Total habitat connectivity was ~17% lower during drought years than in non-drought years and ~7% lower in dry seasons compared to wet. We also found that water intakes added over the study period were placed in lower reaches within individual watersheds, contributing to the observed temporal decline in habitat connectivity. Our study uses a long-term dataset to highlight how cumulative effects of low-head dams paired with water intakes have reduced shrimp habitat connectivity. These results underscore the importance of reducing existing withdrawal rates in EYNF, and siting intakes where effects on connectivity are minimal, if conserving shrimp habitat is a management objective.

1. Introduction

Nearly all large river ecosystems are experiencing decreased connectivity across multiple dimensions due to a variety of factors (Nilsson et al. 2005). For example, longitudinal connectivity (along the stream network) is significantly altered by dams, water withdrawals, and road networks (Ward and Stanford 1995, Pringle et al. 2000, Kondolf et al. 2006), while temporal connectivity (flow continuity) is influenced by natural or anthropogenic variables affecting stream discharge (Ward 1989, Fullerton et al. 2010). Loss of connectivity threatens migratory stream populations dependent on movement through stream networks to persist (Morita and Yamamoto 2002, Freeman et al. 2003, Pringle 2003). In many cases, these migratory organisms provide ecosystem services such as water filtration, nutrient cycling, and provisioning basal food resources (Willson and Halupka 1995, Pringle et al. 1999).

Longitudinal connectivity may be quantified through a variety of indices, which estimate both upstream and downstream passage probabilities for migratory or resident organisms (Cote et al. 2009, Bourne et al. 2011, Diebel et al. 2014). Some connectivity indices combine passage probabilities with habitat quality, quantity, or both, allowing stream habitat to be weighted by the impact on the organism of interest (McKay et al. 2013, Diebel et al. 2014). While many analyses assess connectivity as a time-averaged index, connectivity indices can be applied at smaller time-steps to allow for the detection of seasonal patterns, long-term trends, or periodic connectivity losses (Bourne et al. 2011, Jaeger et al. 2014).

Changes in longitudinal riverine connectivity are relevant to migratory organisms, including amphidromous freshwater shrimps. In Puerto Rico, for example, island-wide metapopulations of seven shrimp species are maintained by connectivity from the stream to the ocean in multiple basins (Cook et al. 2009). In this study, we examine temporal trends in shrimp habitat connectivity across seven watersheds in northeastern Puerto Rico by applying a habitat weighted index of longitudinal connectivity at a monthly time step. Barriers such as low-head dams and associated water intakes can limit habitat connectivity and lead to direct mortality of shrimp larvae through entrainment (Benstead et al. 1999). Conversely, natural barriers over 5 m tall (i.e. waterfalls) block upstream movement of shrimp predators, creating refugia habitat (Cooney and Kwak 2013) and leading to higher shrimp densities upstream of these natural barriers (Covich et al. 2009). Given the abundance of refugia habitat and lack of large dams in northeastern Puerto Rico (Snyder et al. 2011, Cooney and Kwak 2013), this area may serve as an important larval shrimp source for the island-wide metapopulation. In 2015, Puerto Rico experienced a severe drought which led to water supply shortages and accompanying proposals for expanded intake infrastructure. This paper provides a suite of methods for assessing cumulative impacts of low-head dams on migratory shrimp populations in light of existing municipal water intakes.

Our first objective was to: (1) examine the magnitude of regional decline in *total* and *refugia habitat connectivity* along streams draining EYNF over the last 37 years and (2) quantify effects of water withdrawal, dry season, and drought on both t*otal* and *refugia habitat connectivity*. We define *total habitat connectivity* as the proportion of EYNF stream habitat connected to the island-wide metapopulation (i.e., the ocean). We define *refugia habitat connectivity* as the proportion of EYNF refugia habitat (i.e., upstream from waterfalls > 5 m in height) connected to the island-wide metapopulation. We expected a temporal decline in both *total* and *refugia habitat connectivity* as a result of increased municipal population and associated water withdrawal over the same period, particularly during dry season and drought conditions. A secondary objective was to determine the effects of water intake location on connectivity by examining temporal and spatial patterns of water intake development within individual watersheds. We explored whether more recent water intakes were installed in downstream locations outside of EYNF, resulting in the disconnection of larger proportions of upstream habitat. Secondly, we expected more recent intakes would reduce *total habitat connectivity* but have a minimal effect on *refugia habitat connectivity* due to the location of waterfalls in the upper portions of watersheds. Finally, we explored the implications of regional reductions in *total* and *refugia habitat connectivity* and consequent reductions in export of larvae on shrimp metapopulation dynamics across the island.

2. Methods

2.1 Study site

Our study is based in northeastern Puerto Rico's El Yunque National Forest (EYNF), also known as the Luquillo Experimental Forest, which is managed by the U.S. Forest Service. EYNF spans over 110 km^2 of tropical rainforest (Weaver 2012) and includes the Luquillo Mountains, which rise from sea level to a maximum elevation of 1,074 m in less than 20 km (Figure 2.1). Within EYNF borders, there is an average rainfall of 3,860 mm yr^{-1} (García-Martinó et al. 1996), with higher precipitation typical from May through November (Larsen 2000). Eleven major rivers have headwaters originating from within EYNF boundaries, with numerous waterfalls greater than 5 m in height, and much of the island's northeastern population depends on these rivers for potable water and other uses (Crook et al. 2007). Additionally, the rivers contain a diverse community of migratory organisms, including fishes (Kwak et al. 2007), snails (Blanco and Scatena 2006), and shrimps (Covich and McDowell 1996). Eleven shrimp species from three families (Decapoda: Atyidae, Xiphocarididae and Palaemonidae) provide a variety of ecosystem services, such as nutrient cycling (Crowl et al. 2001) and regulating algal growth and community composition (Pringle 1996, Pringle et al. 1999). The most prevalent shrimp genera are: *Atya*, *Xiphocaris*, and *Macrobrachium* (Scatena and Johnson 2001).

We focused on seven rivers draining EYNF: Blanco, Canovanas, Espiritu Santo, Fajardo, Gurabo, Mameyes, and Sabana (Figure 2.1, Table 2.1). We included only watersheds with known intakes, withdrawal amounts, and publicly-available stream gage data. Our analysis is temporally limited to 1980 through 2016 (37 years), as this represents the most complete data set encompassing all seven watersheds. However, we excluded 17 months from our analysis when no discharge data were available at one or more gages: September 1990, October 2011 – September 2012, January 2016, and October – December 2016. We assumed shrimp presence in all focal watersheds based on long-term monitoring by the US Forest Service (Rios, unpublished data).

2.2 Calculating total habitat connectivity

We applied a modified index of longitudinal riverine connectivity for shrimp (ILRC; Crook et al. 2009) to each intake within our focal watersheds, using stream discharge and water withdrawals from intakes averaged for each month, 1980-2016. We then calculated the proportion of total habitat connected (summed across the seven watersheds) to the metapopulation (i.e., to the ocean) on a monthly time step based on the ILRC and stream habitat length. The ILRC is appropriate as Malvadkar et al. (2014) suggested an index based on flow for examining the impact of intakes on downstream larval migration. Our study builds on the study by Crook et al. (2009), which calculated an average ILRC over several years on a per barrier basis using a combination of estimated and permitted withdrawal amounts. However, our connectivity calculations differ by being: (1) calculated on a monthly time step; (2) weighted by shrimp habitat; (3) scaled up to the watershed level; and (4) based on permitted water withdrawal amounts obtained from the Puerto Rican Aqueduct and Sewage Authority.

We delineated each watershed using National Hydrography Dataset Plus (NHDPlus; EPA 2012) flowlines in ArcGIS (Figure 1). Two of our delineated watersheds (Canovanas and Gurabo) do not extend to the ocean as we could not identify all intakes and associated withdrawal amounts in the lower portion of the basins. To calculate the ILRC, we selected one USGS gage per watershed to estimate daily discharge at each intake, scaling discharge by drainage area at the intake in proportion to drainage area at the gage (Figure 1). We estimated each intake's daily total withdrawal amount based on data compiled from various agencies, including: the Puerto Rican Aqueduct and Sewage Authority (PRASA), the Department of Natural and Environmental Resources (DNER), and the US Forest Service (USFS). From these data, we identified 29 intakes with associated withdrawal amounts for our analysis.

The ILRC evaluates the probability a shrimp can move downstream to the ocean (as a larva) and return upstream to the same headwater location (as a juvenile). The index incorporates upstream and downstream passage rates at the barrier (i.e. intake) of interest and the cumulative effects of barriers downstream (see Crook et al. 2009 for formulas). We calculate the downstream passage rate at a given intake as the proportion of discharge remaining downstream of the barrier, as did Crook et al. (2009). Upstream passage is treated as binary, as juvenile shrimp can migrate over the face of barriers if water is flowing over the surface (Holmquist et al. 1998, Benstead et al. 1999). Thus, upstream passage only affects connectivity at a given barrier when water withdrawal equals or exceeds stream discharge, in which case both downstream and upstream passage probabilities are zero. As such, connectivity at a given time and barrier simplifies to the downstream passage probability.

We estimated cumulative connectivity for each barrier as the product of passage probabilities at that barrier and all (known) downstream intakes as in Crook et al. (2009). However, whereas Crook et al. (2009) estimated connectivity based on median annual flows and constant withdrawal amounts, we estimated cumulative connectivity on a daily time step using stream gage records and accounting for changing withdrawal amounts through time. From daily cumulative connectivity, we estimated a mean monthly ILRC value for each month, 1980-2016, for each of the 29 barriers in our study area.

To evaluate the proportion of available habitat in the study area connected to the island-wide metapopulation, we weighted monthly ILRC values by shrimp habitat quantity, which was calculated using NHDPlus (EPA 2012) flowlines. We estimated stream length upstream of each intake (extending to the next upstream intake or to the top of the watershed, as appropriate) and then multiplied by the intake's ILRC to estimate habitat connectivity upstream of the intake on a monthly time step. We summed connected habitat across all intakes to determine the shrimp habitat connectivity across the seven EYNF watersheds. We then divided this sum by total stream length across all seven study watersheds to get the proportion of total habitat connected as follows:

$$H_c = \frac{\sum_{k=1}^{m} ILRC_k * H_k}{H_t}$$

where k represents the intake of interest, $ILRC_k$ is the index of connectivity for intake k, H_k is the stream length upstream of intake k, m is the number of intakes, and H_t is the total stream length across the study area. This provides the proportion of total habitat connected for the entire study area (H_c) to the shrimp metapopulation.

Similar to total habitat calculations, we used NHDPlus flowlines to determine refugia habitat. Waterfalls over 5 m in height were identified using the DEM of Puerto Rico at 10m x 10m resolution and pinpointing where the difference in elevation between two adjacent raster cells was larger than 5 m along a stream in ArcGIS. We then marked the waterfall farthest downstream in each stream channel (Figure 2.1) and considered all upstream length to be refugia habitat. The calculations to determine the proportion of refugia habitat connected were the same as for the proportion of total habitat connected but using refugia stream length.

2.3 Statistical analysis

We used linear regression to test our predictions that total habitat connectivity and refugia habitat connectivity declined through time, and with droughts and dry seasons. Prior to analyses, we applied a logit transformation ($\log[H_c/(1-H_c)]$) to the monthly values for proportion habitat connected to better meet the assumptions for linear regression (Warton and Hui 2011). We modeled these transformed monthly values ($n = 427$ months per watershed) of habitat connectivity in relation to alternative predictor variables that represented our hypotheses. We represented time in the models with chronological month (i.e., 1-444, for 1980-2016), as we expected the response variable (habitat connectivity) to change through time. Because we also expected the proportion of water withdrawn in relation to total discharge to affect habitat connectivity, this variable (calculated for all intakes combined) along with month was included in all candidate models.

Additional binary variables indicated whether the month fell within a drought year or a dry season. The dry season in Puerto Rico is December to April, and the drought years were 1993-1995 (Weaver 2012, Jennings et al. 2014) and 2015. We did not include the number of intakes withdrawing water in a given month because this also increased through time (correlation with month = 0.84). Finally, we evaluated whether the effects of drought and dry season on habitat connectivity have changed over time by including interactions between each of these variables and month. We standardized the two non-binary predictors (proportion water withdrawn and month) by subtracting the mean and dividing by the standard error for each.

We fit a total of seven models for each response variable (total and refugia habitat connectivity) using lm in R (v. 3.3.2, R Development Core Team 2013; code is available on request from the authors). We used Akaike's information criterion (AIC) to assess relative support among our candidate models (Burnham and Anderson 2002). The best-supported model was identified as having the lowest AIC value. Additionally, we calculated the difference in AIC values between the best fit model and the alternative models.

2.4 Watershed analysis

To examine a potential spatial pattern in water intake development, we examined how the proportion of total and refugia habitat upstream of intakes changed through time. We evaluated a correlation between the average annual proportion of habitat upstream of intakes (i.e. habitat affected) and habitat connectivity (i.e. H_c). We also evaluated the timing and location of additional intakes, and their impact on the proportion of total and refugia habitat affected within the individual watersheds.

3. Results

The proportion of total habitat connected, averaged by month, ranged from 0.26 to 0.96, while the proportion of refugia habitat connected ranged from 0.18 to 0.90 over the study period (Figure 2.2). The best-supported model for total habitat connectivity included the proportion of water withdrawn, month, drought, and dry season (Table 2; adjusted R^2 = 0.63). The next best-supported model also included an interactive effect of dry season and drought (**Δ** AIC = 1.22;

Table 2.2). The best-supported model for refugia habitat connectivity only contained the proportion of water withdrawn and month (Table 2; adjusted R^2 = 0.66), while the second best-supported model included either dry season (**Δ** AIC= 1.88) or drought (**Δ** AIC = 1.9), but the parameter estimates for both variables included 0 (no effect, Table 3). Both total and refugia habitat connectivity were on average higher in months classified as wet and non-drought (0.78, 0.61, respectively, for total and refugia habitat), than in months classified as dry and drought (0.63, 0.47, respectively).

The lowest monthly values for habitat connectivity occurred near the end of the study period (2015) during wet and drought months (Figure 2.2). Fewer months were classified as dry (n=178) than wet (n=249). Additionally, the total months classified as drought (n=48) were much less than non-drought (n=379), with only two recognized droughts occurring between 1980 and 2016. Overall, habitat connectivity declined over time more for total (slope= -0.00029, adjusted R^2 = 0.09; Figure 2.2) than for refugia (slope= -0.00018, adjusted R^2 = 0.02275; Figure 2.2), partly as a result of seven new intakes added over the 37-year study period (Supplemental Figure 1). Additionally, refuge habitat connectivity averaged lower than total habitat connectivity at the beginning of the study period (0.64 and 0.86, respectively).

For both total and refugia habitat connectivity, the proportion of water withdrawn (i.e., relative to total discharge; Figure 2.3) had the largest modeled effect (Table 2.3 and 2.4). The proportion of water withdrawn varied from 0.02 to 0.59 across all watersheds and months (mean = 0.13, SD = 0.07). Increasing the proportion of water withdrawn by 0.07 (i.e. 1 SD), was associated with approximately the same decrease in habitat connectivity expected to occur over 369 months (i.e. ratio of parameter estimates = -0.46/ -0.16, multiplied by 128, 1 SD of month; Table 2.4). The relative effect of the proportion of water withdrawn to month on refugia habitat connectivity was even larger (Table 2.3, ratio of parameter estimates = 7.1). Though total water withdrawal amount and total number of intakes increased through time (Supplemental Figure 1), the proportion of water withdrawn did not (adjusted $R^2 = 0.003$; Figure 2.3).

The proportion of total habitat upstream from one or more intakes in the study area (i.e., habitat affected) increased from < 0.3 in 1980 to > 0.6 in 2015 (Supplemental Figure 2). This reflected the addition of intakes lower in the watersheds. In comparison, refugia habitat affected already exceeded 0.6 in 1980, and increased to 0.8 in 2016 (Figure 2.4). Total habitat affected reached a maximum of 0.73, indicating over 70% of shrimp habitat in northeastern Puerto Rico was upstream of an intake, while the maximum refugia habitat affected was 98% (Figure 2.4). Increased amounts of habitat upstream of intakes were associated with lower monthly habitat connectivity ($r^2 = -0.4193$; Supplemental Figure 2).

Two watersheds, Fajardo and Gurabo, had intakes added in the lower portions of the stream network during the study period. In the remaining watersheds, however, intakes were either removed (Blanco, Mameyes), installed upstream of existing intakes (Canovanas and Espirtu Santo), or constant (Sabana) (Supplemental Table 1 and 2). Though additional intakes in Fajardo and Gurabo increased the proportion of total habitat affected, the impact on affected refugia habitat was minimal. In Fajardo, the proportion of total habitat affected increased by 34%, while in Gurabo the affected total habitat increased by 72% over the study period. Conversely, refugia habitat impacted only increased by 12% in Gurabo and did not change in Fajardo, as all refugia habitat was already upstream of an intake in 1980 (Supplemental Table 1).

4. Discussion

Riverine connectivity within EYNF has decreased over 37 years as a result of increased water withdrawals associated with low-head dams, with total and refugia habitat connectivity declining by 27% and 16%, respectively. Losses in total habitat connectivity were exacerbated during a dry season or drought year by 7% and 17%, respectively. Additionally, our results indicate intakes added through the study period were lower down in the watershed. We also found intake placement within a river network, in relation to existing intakes, determines its impact on total and refugia habitat connectivity. Furthermore, as refugia habitat is particularly vulnerable to complete disconnection by the existence of even one intake within the watershed, exact intake placement within the watershed matters less in terms of the effect on refugia habitat connectivity.

The strongest driver of shrimp habitat connectivity in EYNF is the proportion of water withdrawn from intakes. This is in contrast to previous research conducted at other geographic locations, which identifies an increasing number of barriers as the main driver of lowered connectivity, particularly when cumulative effects arising from multiple barriers are considered (Diebel et al. 2014, Grill et al. 2014). However, these previous studies assumed barriers completely block upstream and downstream passage (i.e., passage rate = 0). In contrast, our study allows downstream passage rates to span from '0' to '1' in response to varying streamflow, as small changes in passage rates can magnify into large connectivity fluctuations (Shaw et al. 2016). Though the proportion of water withdrawn is the determinant variable of total habitat connectivity in our study, it does not vary through time.

The declining trend in total habitat connectivity results from an increase in the proportion of total habitat affected (i.e. habitat upstream of intakes). Intakes installed at downstream locations in the watershed increased the proportion of habitat affected by 60% over 37 years, reinforcing the importance of barrier placement as noted elsewhere (Cote et al. 2009, Ziv et al. 2012, McKay et al. 2013). While the development pattern of installing additional intakes lower in the watershed through time only occurred within Fajardo and Gurabo, both are large watersheds (38.6 km^2 and 57.8 km^2, respectively) and are likely driving this trend at the EYNF scale.

Refugia habitat connectivity demonstrated a more gradual temporal decline as refugia connectivity was initially lower. At the watershed scale, we observed intakes added during the study period did not decrease refugia habitat connectivity as much as total habitat connectivity. Initial barriers within a watershed are known to have the largest impact on connectivity (Cote et al. 2009), and this effect in our study is explained by the geomorphology of Puerto Rico; waterfalls are a result of the Luquillo Mountains which create refugia habitat in the headwaters.

For both total and refugia habitat, the lowest connectivity values were observed during low flow periods, such as the annual dry season (December – April) and extreme droughts (1993-1995 and 2015). During droughts, withdrawals from multiple intakes within EYNF reduced flow directly downstream to zero over several consecutive days (personal observation), severing connectivity to habitat upstream of the intake. Although periods of naturally occurring low flow can result in low habitat connectivity, water withdrawals from intakes paired with low-head dams further reduce habitat access.

This study emphasizes the importance of examining the cumulative effects of partial barriers in the context of long-term flow records. Though other studies have looked at partial barriers (King and O'Hanley 2014, Diebel et al. 2014) and long-term (20 years) studies have treated passage rates as dynamic (Bourne et al. 2011), here we also calculate connectivity on a monthly time-step to examine intra-annual connectivity patterns. We recognize there may be some barriers with associated intakes that were not included in this study, especially within the Canovanas and Gurabo watersheds, but adding additional barriers would lower habitat connectivity, indicating these connectivity values represent the best-case scenario. Our results illustrate even partial barriers, whose impact on habitat connectivity change with flow fluctuations, noticeably decrease connectivity over decades, a trend which may have been obscured at shorter time scales.

Refugia habitat is important as it is relatively predator-free and supports a high abundance of shrimps that can potentially contribute large numbers of larvae to the island-wide metapopulation. For example, in refugia habitat within the Espiritu Santo watershed, *Atya lanipes* and *Xiphocaris elongate*, the two dominant shrimp species, reach densities of 24 and 13 individuals/m^2, respectively (Covich et al. 2009). In contrast, below waterfalls in the Mameyes watershed, where the shrimp predator mountain mullet (*Agonostomus monticola)* occurs, *Atya* and *Xiphocaris* densities are smaller by an order of magnitude (0.94 and 0.61 individuals/m^2, Covich et al. 2009). Additionally, previous research suggests Rio Espiritu Santo has a mean daily larval drift rate 15% higher than Rio Mameyes (3.731 million larvae/day and 3.171 million larvae/day, respectively) (March et al. 1998), as the Espiritu Santo has 12% more refugia habitat above waterfalls. Periodic losses of refugia habitat connectivity could potentially result in large declines of larvae supplied to the island-wide metapopulation given the significant amount contributed by refugia habitat.

As a result of climate change, the Caribbean is predicted to receive less precipitation, resulting in drier *wet* seasons, drier *dry* seasons, and extended droughts (Jennings et al. 2014). As connectivity is highly correlated with streamflow, lower discharge will likely lead to a decline in habitat connectivity if withdrawal amounts are maintained. In the western U.S., which is also predicted to have reduced streamflow from the effects of climate change, models indicate longitudinal riverine connectivity could decrease by up to 14% during the dry season (Jaeger et al. 2014). A similar connectivity decline in EYNF would compound existing losses in total and refugia habitats.

The potential for further declines in habitat connectivity indicates the need to identify strategies that mitigate effects of intakes. Our results suggest intake impacts on total habitat connectivity can be minimized if withdrawals are installed upstream of existing intakes or if the proportion of water withdrawn in relation to streamflow is decreased. However, in terms of refugia habitat, any barrier within the stream network will likely lower connectivity. Cote et al. (2009) and King and O'Hanley (2014) note barriers placed high in watersheds without existing dams will have a smaller impact on migratory species, but this may not be true for shrimp taxa where refugia habitat is in the headwaters. If maintaining shrimp larval supply is a management priority, management plans that limit intakes in watersheds with ample refugia habitat should be considered.

Here, we provide a baseline of how connectivity in EYNF has been reduced by low-head dams and associated intakes. We do not include a barrier prioritization analysis, as this is not currently a priority in Puerto Rico given current levels of freshwater demand and other resource management challenges (e.g., Hurricane Maria recovery efforts). Establishing a long-term baseline of shrimp habitat connectivity provides critical context for natural resource managers conserving freshwater ecosystems. Just as managers of potable water supplies are proactively considering how climate change may affect freshwater supply, ecological management can proactively consider impacts to freshwater shrimp, resulting from a combination of naturally occurring low flow periods and human water withdrawals.

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Table 2.1. Characteristics of the seven EYNF watersheds studied. Gage drainage area indicates the drainage area of the gage as reported by USGS (km^2). Maximum number of intakes represents the highest number of intakes withdrawing water in a month. Maximum monthly water withdrawn represents the largest amount of water withdrawn in a month. Maximum habitat affected is the maximum proportion of habitat upstream of at least one intake over the study period. Earliest intake online is the first year a watershed has a known intake withdrawing water.

Watershed	Total Drainage Area (km^2)	USGS Gage Number	Gage Drainage Area (km^2)	Maximum Number of Intakes	Maximum Monthly Water Withdrawn (cms)	Maximum Habitat Affected (%)	Earliest Intake Online
Blanco	74.1	5007500	3.3	5	1.1	55.3	1928
Canovanas	46.4	50061800	25.5	5	0.2	93.7	1968
Espiritu Santo	91.4	50063800	22.3	7	0.8	91.3	1980
Fajardo	68.3	50071000	38.6	1	0.5	49.8	1966
Gurabo	179.0	50055750	57.8	5	0.7	84.4	1939
Mameyes	40.3	50065500	17.8	1	0.1	71.7	1998
Sabana	18.5	50067000	10.3	2	0.1	41	1956

Table 2.2. Linear regression models for predicting total habitat connectivity and refugia habitat connectivity. Models are listed based on increasing parameter number (K), along with AIC values relative to the best-supported model (Δ AIC).

Model	**K**	**Total Habitat Δ AIC**	**Refugia Habitat Δ AIC**
(1) Proportion Withdrawn, Month	4	7.8	0
(2) Proportion Withdrawn, Month, Dry	5	5.9	1.9
(3) Proportion Withdrawn, Month, Drought	5	2.8	1.9
(4) Proportion Withdrawn, Month, Dry, Month*Dry	6	7.4	2.9
(5) Proportion Withdrawn, Month, Drought, Month*Drought	6	4.8	3.9
(6) Proportion Withdrawn, Month, Drought, Dry	6	0	3.8
(7) Proportion Withdrawn, Month, Drought, Dry, Dry*Drought	7	1.2	5.5

Table 2.3. Refugia habitat connectivity parameter estimates (and standard errors) for predictor variables included in linear regression models. Models are listed based on increasing parameter size, as in Table 2.2. Model 1 (in bold) was the best-supported model.

Model	Intercept	Proportion Withdrawn	Month	Drought	Dry	Drought *Month	Dry* Month	Dry* Drought
1	**0.306 (0.018)**	**-0.503 (0.018)**	**-0.073 (0.018)**					
2	0.306 (0.018)	-0.502 (0.018)	-0.073 (0.018)		-0.006 (0.018)			
3	0.306 (0.018)	-0.502 (0.002)	-0.073 (0.018)	-0.006 (0.018)				
4	0.306 (0.018)	-0.503 (0.018)	-0.073 (0.018)		-0.007 (0.018)		-0.018 (0.018)	
5	0.306 (0.018)	-0.503 (0.018)	-0.073 (0.018)	-0.007 (0.018)		-0.018 (0.018)		
6	0.306 (0.018)	-0.504 (0.019)	-0.074 (0.018)	0.006 (0.019)	-0.006 (0.018)			
7	0.306 (0.018)	-0.503 (0.019)	-0.074 (0.018)	0.005 (0.019)	-0.006 (0.018)			0.009 (0.018)

Table 2.4. Total habitat connectivity parameter estimates (and standard errors) for predictor variables included in linear regression models. Models are listed based on increasing parameter number, as in Table 2.2. Model 6 (in bold) was the best-supported model.

Model	Intercept	Proportion Withdrawn	Month	Drought	Dry	Drought *Month	Dry* Month	Dry* Drought
1	1.125 (0.020)	-0.492 (0.020)	-0.162 (0.020)					
2	1.134 (0.020)	-0.483 (0.020)	-0.163 (0.020)		-0.040 (0.020)			
3	1.135 (0.020)	-0.474 (0.021)	-0.160 (0.020)	-0.056 (0.021)				
4	1.134 (0.020)	-0.484 (0.020)	-0.163 (0.020)		-0.041 (0.020)		-0.015 (0.020)	
5	1.135 (0.0120)	-0.474 (0.021)	-0.160 (0.020)	-0.056 (0.021)		0.002 (0.023)		
6	**1.135 (0.020)**	**-0.463 (0.021)**	**-0.161 (0.020)**	**-0.059 (0.021)**	**-0.044 (0.020)**			
7	1.135 (0.020)	-0.461 (0.022)	-0.162 (0.020)	-0.060 (0.021)	-0.045 (0.020)			0.017 (0.020)

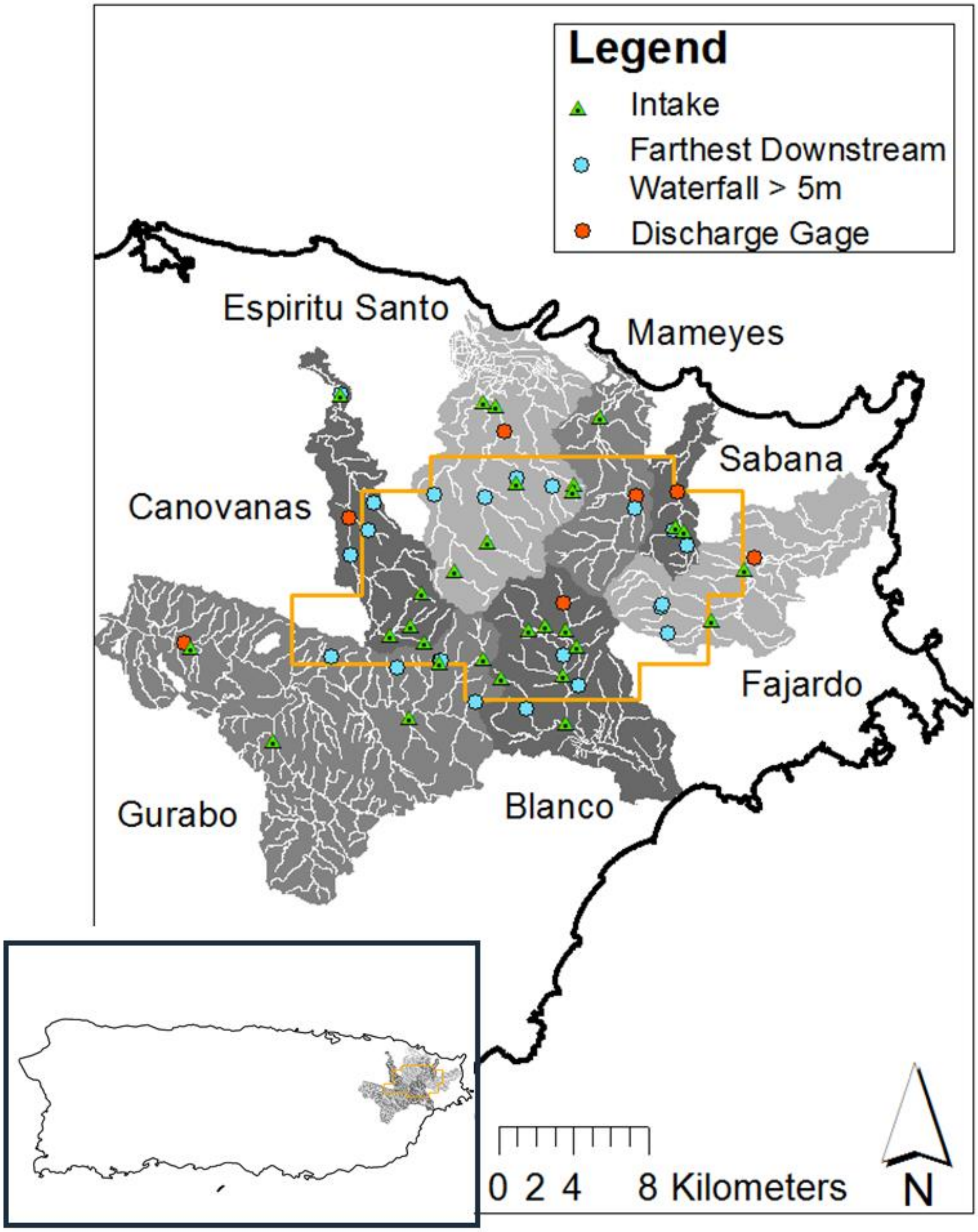

Figure 2.1. The seven study watersheds of EYNF. Each watershed is labeled, and the drainage area is colored in distinct shades of gray. The orange line represents the boundaries of EYNF. The white lines indicate the stream flowlines. The red points are the

USGS discharge gages, and the green triangles represent intakes. The blue circles indicate the farthest downstream waterfall over 5 m in height, which were identified using elevation differences between adjacent DEM raster cells in ArcGIS. Inset: Puerto Rico and the seven study watersheds.

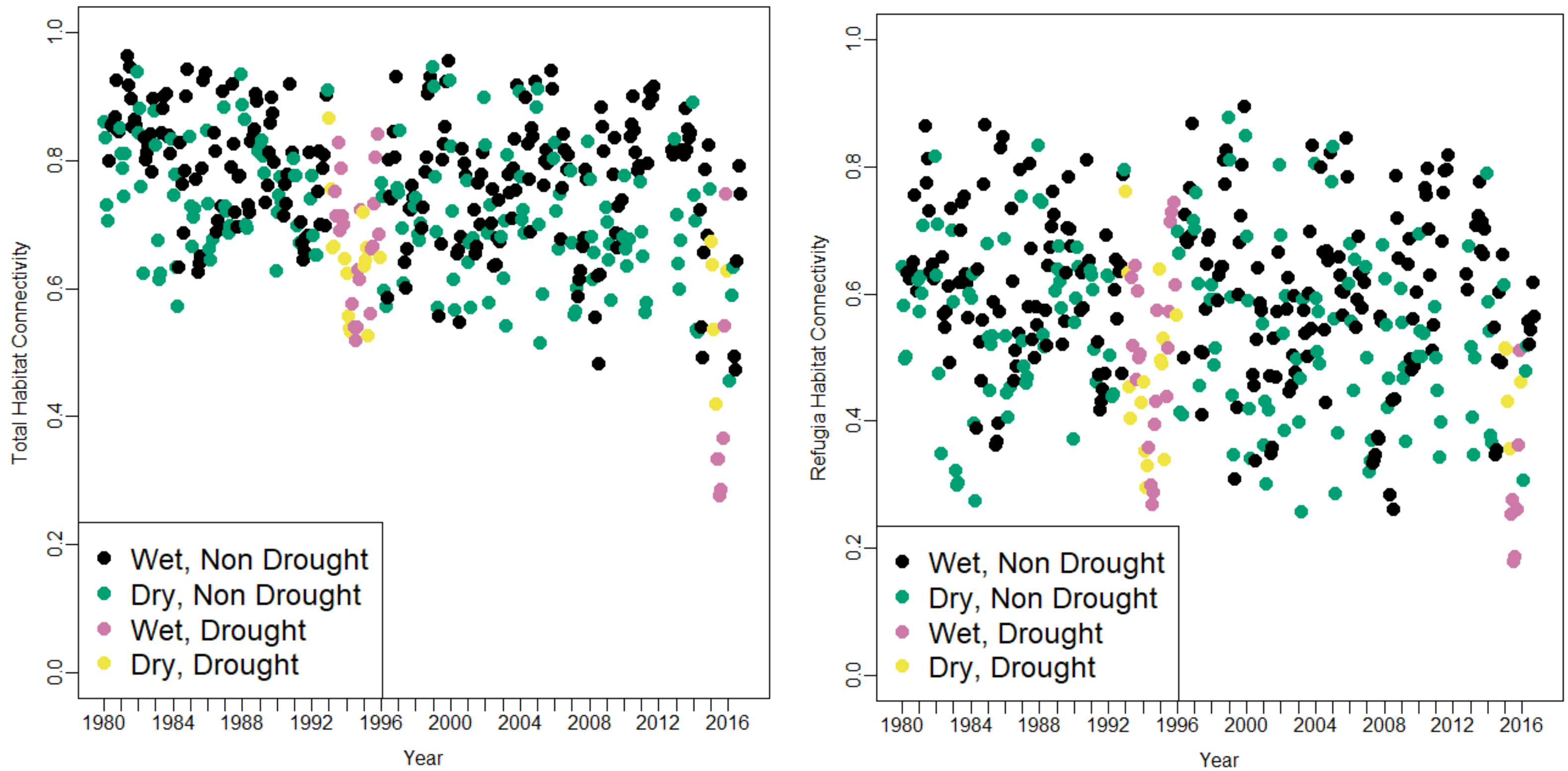

Figure 2.2. *Left*: Proportion of total habitat connected to shrimp metapopulation throughout EYNF (across the seven focal watersheds). *Right*: Proportion of refugia habitat connected across EYNF from 1980 – 2016. Seasons and drought years were classified based on previously accepted values. Habitat connectivity was calculated as the accessible habitat divided by the total stream length across all watersheds.

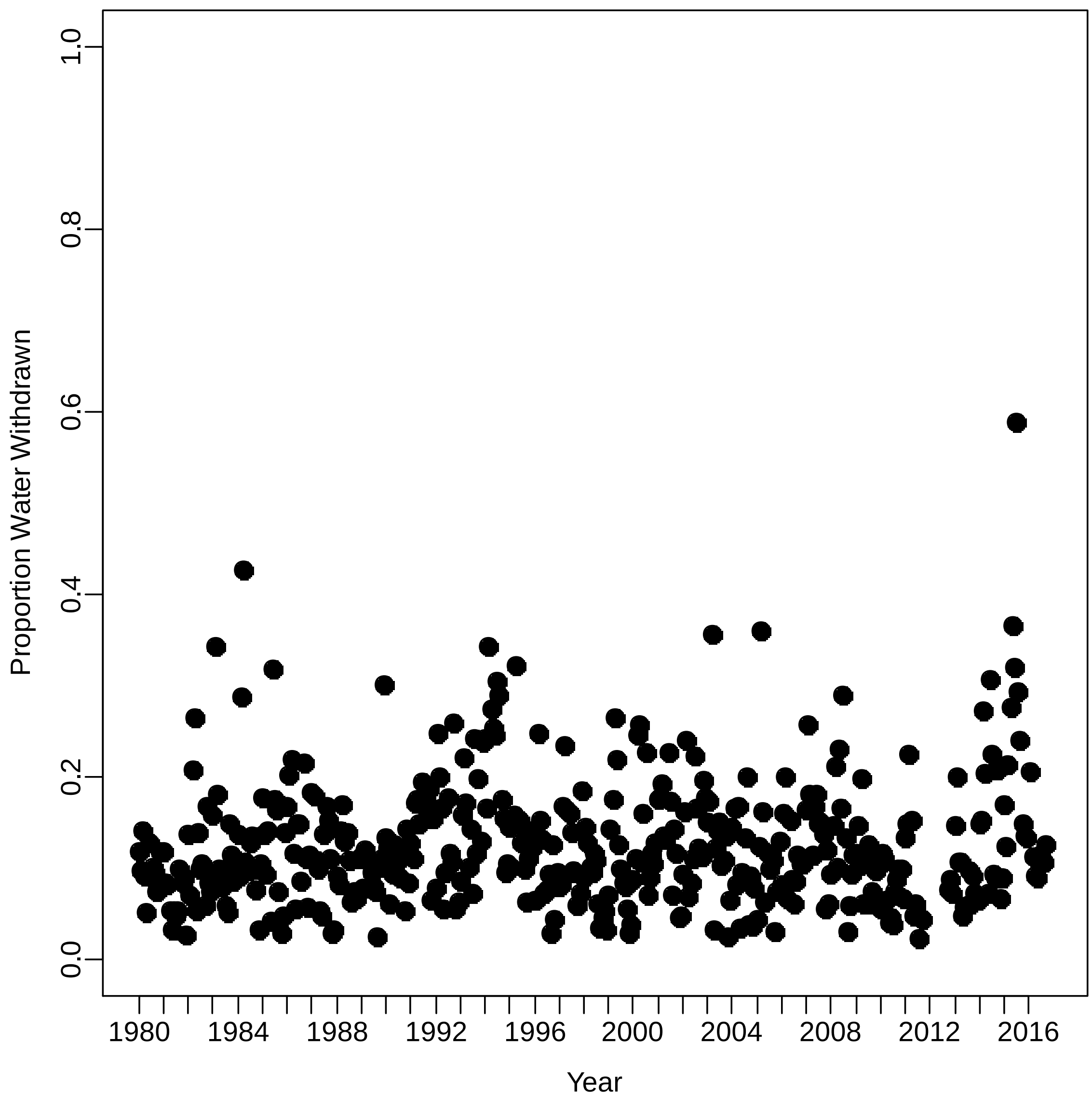

Figure 2.3. The relationship between the proportion of water withdrawn from within EYNF and month was not significant (adjusted R^2 = 0.003). We calculated the proportion of water withdrawn by dividing the total water withdrawn from all intakes by the total water amount reaching the ocean from the study area.

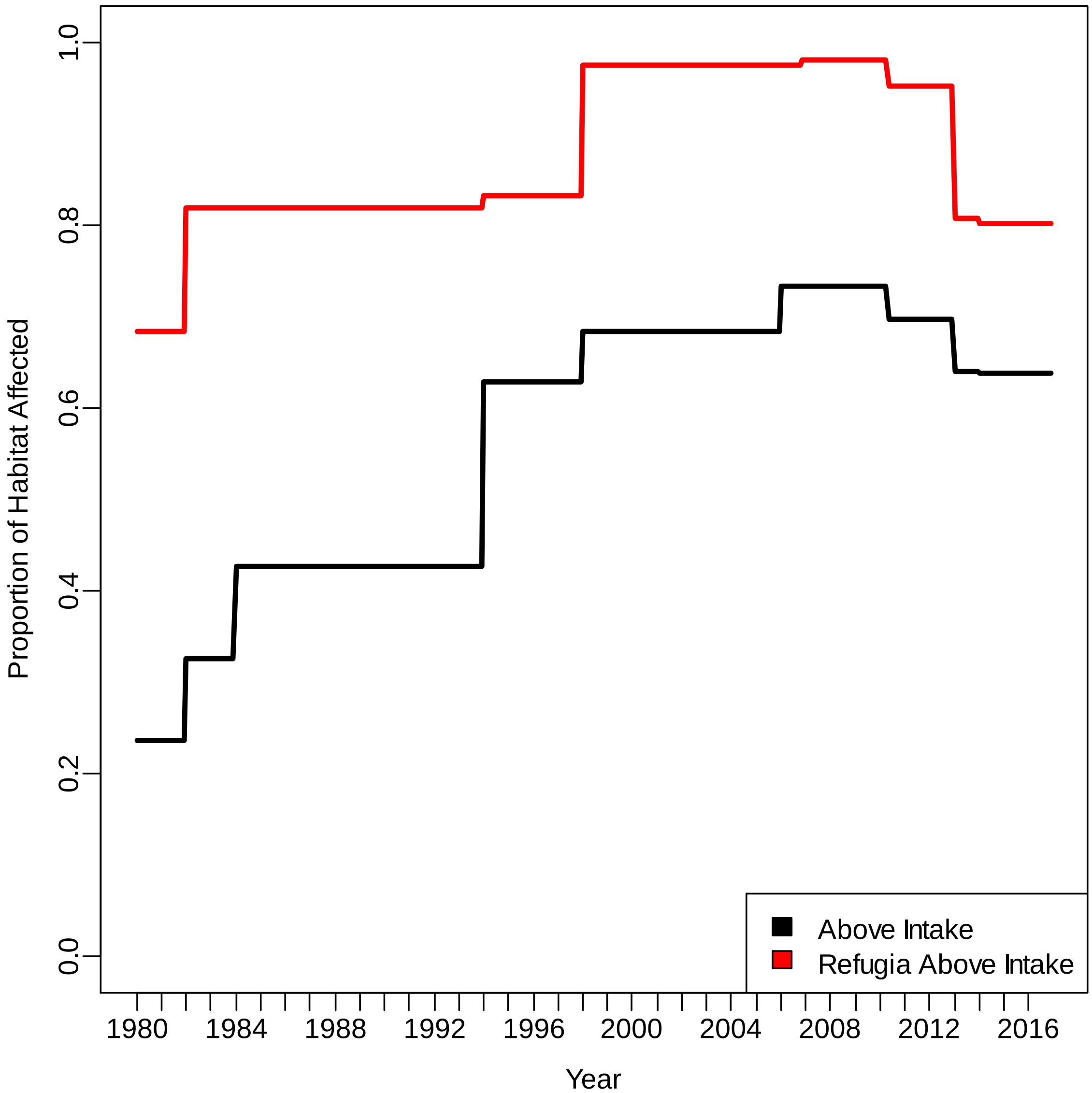

Figure 2.4. The proportion of habitat affected (i.e. habitat upstream of an intake) across the study area. We calculated the proportion of habitat affected by adding all stream habitat upstream of intakes across watersheds and dividing by the total stream habitat. Stream habitat was determined using NHDPlus flowlines.

CHAPTER 3

MANAGING HABITAT CONNECTIVITY FOR MULTIPLE MIGRATORY GROUPS USING ALTERNATIVE WATER WITHDRAWAL SCENARIOS[2]

[2]Chappell, Jessica, Mary Freeman, S. Kyle McKay, Cathy Pringle. To be submitted to *Ecological Applications*.

Abstract

Most research evaluating the impact of barriers on connectivity focuses on one migratory species or species group with the same movement physiology. Additionally, alternative management scenarios have been suggested as a solution to increase habitat connectivity if an intake associated with a barrier cannot be removed. However, tropical stream communities have multiple migratory species that have various life histories and how barriers paired with intakes affects species with different movement physiology is not well understood. Therefore, we evaluated how alternative water withdrawal scenarios influenced habitat connectivity for five migratory groups to determine whether there are management options that increase connectivity across movement physiologies. We conducted our analysis using four watersheds that drain the El Yunque National Forest in northeastern Puerto Rico. Our five focal groups included freshwater shrimps, gobies, snails, mountain mullets, and American eels. We used a 31-year discharge dataset (1986-2016) and four different water withdrawal scenarios based on the 2016 intake rate: 0%, 50%, 100% (2016 rate), and 200%. Our analysis found the groups that are limited in migration by barrier height (i.e. mullet and eel) experience a smaller rate of habitat connectivity gain as water withdrawals decrease. However, all groups have increased habitat connectivity when intakes do not have an associated instream barrier height. Additionally, our results illustrate the importance of including downstream passage for migratory animals connected to a larger metapopulation. Our findings suggest decreasing water withdrawal rates would result in increased habitat connectivity for shrimp and goby, but have little effect on snails, mullet, and eel. However, increased intake rates would decrease habitat connectivity for all groups.

1. Introduction

Instream barriers are reducing habitat connectivity worldwide for aquatic organisms (Vorosmarty et al. 2010). Effects of these stream channel alterations include blocked animal movement upstream to high-quality habitat (Sheer and Steel 2006) and alter downstream flow regimes (Poff and Zimmerman 2010). Migratory fish that must move either between freshwater and the ocean (diadromous) or within freshwater (potadromous) to complete their life cycle are especially vulnerable to declines in connectivity (Cote et al. 2009). The macroconsumer community of many tropical island streams are dominated by several taxa of migratory diadromous species including shrimps, fishes and snails (Covich and McDowell 1996, Bass et al. 2003), and it is not well understood how cumulative barriers affect the migratory community. An understanding of how barriers and their associated impacts (i.e. height and altered discharge) affect multiple taxa is essential to inform water resource management.

Historically, ecologists have characterized faunal passage at a barrier as a temporally static measure, often related to barrier height relative to animal jumping or climbing abilities (Cote et al. 2009, Cooney and Kwak 2013, Diebel et al. 2014). However, streamflow level, which characteristically varies through time, can also affect passage at barriers (Jaeger et al. 2014). In particular, diadromous freshwater shrimps require streamflow to move upstream over barriers (Crook et al. 2009). Shrimp larvae moving downstream also require streamflow, and their passage rate at barriers with water withdrawals is proportionately reduced (March et al. 1998, Benstead et al. 1999). Studies quantifying passage rates as variable through time reflect actual habitat connectivity within a system and can better inform management strategies (Bourne et al. 2011). Thus, passage rates at barriers should be considered as fluctuating through time.

In streams draining northeastern Puerto Rico, low-head dams (< 5 m in height) and associated water intakes, allow for partial longitudinal riverine connectivity that fluctuates with discharge (Crook et al. 2009, Chappell et al. 2019). Here we focus on streams draining the El Yunque National Forest (EYNF; x ha). Stream communities of the EYNF are dominated by diadromous species, including freshwater shrimps, fishes, and snails (Covich and McDowell 1996, Harris et al. 2012). Many of these taxa have been extirpated from Puerto Rican headwater streams outside the EYNF boundary due to large dams with no spillway discharge that completely block migratory movement (Snyder et al. 2011, Cooney and Kwak 2013). Though some movement barriers are natural and formed by waterfalls high up in the watershed (Covich et al. 2009), dams paired with water intakes to meet human water needs also block habitat for migratory animals (Greathouse et al. 2006a). Although these dams may be small, if there is no discharge downstream, diadromous species cannot complete their migration (Greathouse et al. 2006b). Intakes within streams draining EYNF represent a management challenge as 20% of the island population depends on freshwater originating from the forest (Crook et al. 2007), yet low-head dams can have ecological consequences for connectivity.

Although barrier removal has been shown to improve connectivity for migratory species (Hitt et al. 212), removing a barrier has associated tradeoffs that span ecological, social, and economic sectors (Roy et al. 2018). When deciding which barrier to remove from a watershed, removal cost is a major consideration for managers (McKay et al. 2016). Barrier removals can also have negative ecological consequences, such as allowing invasive species to spread their range (Jackson et al. 2010). Alternate management scenarios that allow for increased connectivity with the barrier remaining in place may be necessary to consider in some situations (Karieva et al. 2012) and allows for monitoring and adaptive management if the strategy is ineffective (Smith et al. 2017). Altering the streamflow regime may prove to increase connectivity for migratory fauna though barriers remain in place, especially if streamflow affects passage rates.

Objectives of this study are to evaluate effects of alternative water withdrawal operational scenarios on longitudinal habitat connectivity for migratory diadromous taxa in Puerto Rico. We focused on five migratory taxonomic groups (freshwater shrimps, gobies, American eels, mountain mullets, and snails) on a monthly time step across four EYNF watersheds over 31 years (1986-2016). These focal groups have varying life history characteristics that determine their downstream and upstream movements. We predicted that the amount of habitat connected within a watershed would differ between taxa and that shrimp would have the highest habitat connectivity across all watersheds given the movement restrictions of fishes and snails. Similarly, we predicted that shrimp would have the largest increase in habitat connectivity if water withdrawals were reduced given the strong effects of withdrawal on their downstream movement past intakes (Benstead et al. 1999). We also evaluated whether watershed characteristics influence habitat connectivity for taxa. Specifically, we expected watersheds with multiple barriers to have more habitat connectivity with lower numbers of water intakes.

2. Methods

2.1 Study Area

Our study site is located in northeastern Puerto Rico and includes El Yunque National Forest (EYNF). EYNF is also known as the Luquillo Experimental Forest and is the only rainforest managed by the U.S. Forest Service. EYNF receives an average annual rainfall of 3,860 mm yr^{-1} with higher elevations experiencing greater rainfall (García-Martinó et al. 1996). The national forest boundaries include the Luquillo Mountains, which rise from sea level to over 1,000 m in 8 km (Harris et al. 2012), creating headwater stream channels with steep gradients (Pike 2008). We focused our study in four watersheds within the EYNF: Blanco, Espiritu Santo, Fajardo, and Sabana (Figure 3.1, Table 3.1). These drainages were selected because each has watershed boundaries extending to the ocean, operational intakes in 2016, and man-made barriers. Our analysis used continuous daily discharge data from 1986 to 2016 (31 years), a time period that incorporates extreme fluctuations in measured discharge.

2.2 Migratory Strategies of Focal Taxa

The Puerto Rican stream community is composed of a variety of migratory aquatic animals that include fishes, snails, and freshwater shrimps. The migratory freshwater fish include two species of goby, sirajo (*Sicydium plumieri*) and river (*Awaous banana*), mountain mullet (*Agonostomus monticola*), and American eel (*Anguilla rostrata*). There are at least ten freshwater shrimp species, including *Xiphocaris elongata*, three *Atya* spp., and five *Macrobrachium* spp. (Harris et al. 2012). Over seven aquatic snail species have been found in EYNF (Harris et al. 2012), but *Neritina punctulata* and *Neritina virginea* are the two snail species most studied (Pyron and Covich 2003, Blanco and Scatena 2006). For our analysis, we focused on five major taxa: shrimp, goby, mountain mullet, American eel, and snail (Table 2). We grouped species based on similar life histories as have previous studies (Crook et al. 2009), although we recognize there are behavioral differences between species within a given taxa (e.g. *Macrobrachium* spp. tend not to migrate to the same high elevations as *Xiphocaris elongata*; March et al. 2001). We assumed these differences are negligible for our analysis because species tend to exhibit distinct characteristics upstream of the barriers we included (e.g. above 200 m).

The two diadromous life history strategies represented among groups are amphidromy and catadromy. Shrimps, gobies, mountain mullets, and snails are amphidromous, which means individuals move from freshwater to the ocean and back to freshwater to reproduce (McDowall 1992). Amphidromy is a life history strategy commonly used by aquatic animals on tropical islands, potentially as an adaptation to highly variable habitat resulting from frequent disturbances (McDowall 2009). The American eel is catadromous, meaning it reproduces in saltwater but migrates to freshwater as a juvenile to develop into an adult. The freshwater habitat that American eels can access stretches from Canada to Central America. Once adult eels are ready to spawn, which can take decades, they migrate from their freshwater habitat back to the Sargasso Sea (Jacoby et al. 2015). Although a diadromous life history allows individuals to rapidly colonize newly available habitat and potentially avoid extinction if a satellite population is lost, species persistence relies on connectivity between streams and the ocean.

The American eel faces multiple threats throughout its habitat range and is listed as endangered by the IUCN Red List (Jacoby et al. 2017). Generally, larger individuals are present higher in the watershed and smaller individuals congregate lower, closer to the ocean (DeMarte et al. 2016, Kwak et al. 2018), although eel movement upstream can be blocked by instream barriers, such as dams (Hitt et al. 2012, Cooney and Kwak 2013). Previous research suggests that stream discharge influences when eels migrate downstream (Smith et al. 2017) and upstream, because chemical cues present in streamflow may trigger upstream migration (Galbraith et al. 2017).

Although goby and mountain mullet are both amphidromous fish, they have distinct upstream movement mechanisms. Previous research suggests goby reproduction and upstream migration is influenced predominantly by river discharge (Fitzsimons et al. 2002, Engman et al. 2017). On the Caribbean island Guadeloupe, sirajo goby were found at 545 m (Fivete et al. 2001), supporting observations by Covich and McDowell (1996), who observed sirajo goby dominating the fish community at elevations above 600 m. Cooney and Kwak (2013) found that goby are able to move upstream over barriers up to 50 m in height. In contrast, the upstream movement of mountain mullet can be blocked by barriers over 5 m high (Cooney and Kwak 2013), and Caribbean populations seem to spawn late in the year (i.e. August – September), coinciding with the wet season (Aiken 1998).

Snail and shrimp represent the two non-fish taxa, but each is limited in its upstream migration by different stream characteristics. *Neritina punctulate* seems restricted to low elevations (Covich and McDowell 1996), specifically below 200 m (Pyron and Covich 2003). Upstream migration appears to be correlated with high discharge events during certain times of the year (May, August through November) (Blanco and Scatena 2005) and is potentially driven by predator avoidance and life span

(Pyron and Covich 2003). Shrimp also seem to move upstream to avoid predators because natural and man-made barriers over 5 m tall represent refugia habitat for shrimp (Covich et al. 2009, Cooney and Kwak 2013) since predators are excluded by these barriers. Shrimp species can climb over barriers of any height as long as there is discharge over the barrier (J.P. Benstead, J.G. March, P.J. Torres, *personal observations*). Although the mode of upstream movement varies by taxon (e.g. climbing, jumping), all amphidromous species larvae drift passively to the ocean (Benstead et al. 1999, McDowall 2009, Crandall et al. 2010; Table 3.2).

2.3 Calculating habitat connected

We used a modified version of the index of longitudinal riverine connectivity (ILRC) to determine connectivity based on a combination of downstream and upstream passage rates for the five taxa at all barriers in each study basin. We performed our calculations on a monthly time-step over the 31-year dataset using four different water withdrawal scenarios. We used the ILRC developed by Crook et al. (2009) and modified by Chappell et al. (2019) for shrimp species and included additional variables based on the movement characteristics of our focal taxa (Table 3.2).

The barriers in our connectivity analysis include man-made and natural barriers. Man-made barriers are dams that are either used for hydropower generation or paired with intakes for water withdrawal. We used 2016 intake withdrawal rates (Chappell et al. 2019) and barrier fish heights provided by Dr. Tom Kwak. For barriers without fish height data, we assumed barrier height was small (i.e., 1.5 m) as the barriers tended to be higher in the watershed and were overlooked. To identify waterfalls, we used the DEM of Puerto Rico at 10 m x 10 m resolution and identified sites where the difference in elevation between adjacent raster cells was larger than 5 m in ArcGIS. We only incorporate the most downstream waterfall in our connectivity calculations, as barriers of 5 m completely block movement of three of our focal taxa (snail, mullet, and eel). Overall, there were 12 waterfalls and 18 intake locations across the four watersheds. We delineated watersheds using the National Hydrography Dataset Plus (NHDPlus; EPA 2012), following methods in Chappell et al. (2019). To estimate discharge at each barrier, we scaled USGS gaged discharge data for each watershed by drainage area for each intake watershed area. We used the following USGS gages: 5007500 (Blanco), 50063800 (Espiritu Santo), 50071000 (Fajardo), and 50067000 (Sabana).

For all amphidromous taxa, we calculated downstream passage at each barrier as the proportion of discharge remaining downstream of the barrier versus the amount of water reaching the barrier from upstream. Downstream passage rate is assumed to be proportional to flow because drifting larvae are uniformly spread through the water column and entrained by intakes (Benstead et al. 1999). In contrast, we assumed a downstream passage rate of ‘1’ for the catadromous eel if downstream discharge was greater than zero, because adult eels migrating downstream likely require some streamflow. We considered the downstream passage rate for all taxa to be ‘1’ at waterfalls because no water is withdrawn.

Upstream passage rates varied across taxa. For shrimp, we calculated upstream passage as binary, assuming a rate of 1 if downstream discharge was greater than zero as did Chappell et al. (2019). Snail upstream passage rate was also binary, with upstream passage assumed to be zero if there was no downstream discharge, barrier height ≥ 5 m, or barrier elevation ≥ 200 m (Table 3.2). For goby, mullet, and eel, we calculated upstream passage rate based on a combination of downstream discharge and barrier height. If downstream discharge was zero (i.e. all water is withdrawn from the stream at the intake location), we assumed all fish were unable to move upstream (Table 2). For downstream discharges greater than zero, we fitted a logistic equation to estimate upstream passage probability as a function of barrier height. Using the data provided in Cooney and Kwak (2013), we estimated regression coefficients for goby (-1.80, 0.15), mullet (-2.08, 1.30), and eel (-9.82, 10.91).

We used downstream and upstream passage rates to determine the cumulative passage rates for each barrier. Cumulative passage rates are the product of passage rates at the barrier of interest and all identified downstream barriers. For our analysis, we calculated daily downstream and upstream cumulative connectivity for each taxon. We used a daily timescale because previous research has highlighted the importance of considering intake effects on a short timestep given the ability one or two high discharge events to elevate the average monthly discharge (Blanco and Johnson 2001, Christian et al. 2019). Additionally, we assumed all taxa are constantly moving upstream and downstream through time. Though previous studies have suggested temporal migratory patterns exist (Benstead et al. 1999, Engman et al. 2017), we believe further research is needed before these movement trends can be assumed to persist across watersheds.

Using the daily cumulative passage rates, we calculated the monthly average ILRC for all our study barriers (18 intakes, 12 waterfalls) for each month in our time series (1986-2016). The ILRC for barrier k is calculated as follows:

$$ILRC_k = PrDC_k * PrUC_k$$

where $PrDC_k$ is the probability individuals originating from upstream of intake k reach the ocean (i.e. cumulative downstream passage rate) and $PrUC_k$ is the probability individuals migrate above intake k originating from the ocean (i.e. cumulative upstream passage rate). We then estimated the habitat connected (in stream km) in the watershed. We used NHDPlus (EPA 2012) flowlines to estimate the stream length upstream each barrier to the next additional barrier or to the top of the watershed. We then calculated the habitat connected in the watershed using the following equation:

$$H_c = \sum_{k=1}^{m} ILRC_k * H_k$$

where k is the intake of interest, $ILRC_k$ is the index of connectivity for intake k, H_k is the stream length upstream of intake k, and m is the number of intakes in the watershed. This provides the habitat connected (in stream km) within a single watershed for one taxon. We calculated the habitat connected (H_c) using the same method for each of the five taxa in the four watersheds to determine the habitat connected by taxon and watershed.

2.4 Water Withdrawal Scenarios

We used four water withdrawal scenarios to evaluate whether changing the amount of water withdrawn had an effect on habitat connectivity for each taxon in the focal watersheds. We evaluated withdrawal rates of 0%, 50%, 100%, and 200%, using the 2016 withdrawal rate as the base scenario (i.e., 100%). The withdrawal rate of each individual intake was altered according to the scenario withdrawal rate (i.e., reduced to 0, halved, or doubled for the alternative scenarios). The 0% scenario allowed us to determine whether barrier height or withdrawal amount is driving habitat connectivity for certain taxa by isolating height-dependent and discharge-dependent movement processes.

The other scenarios allowed us to evaluate the consequences of substantive reduction or increase in water withdrawal for each taxon and watershed.

We compared the loss rate of habitat connectivity across all water withdrawal scenarios within individual watersheds to better understand the driving variables (e.g. height, discharge) of habitat connected by taxon. We compared habitat connected across watersheds to explore how intake location within a watershed affects habitat connected. To standardize comparisons of habitat connectivity across watersheds, we calculated the habitat accessible. The habitat accessible is calculated as the proportion of habitat that remains accessible for the specified taxon as water withdrawal rates are altered. We consider the habitat connected under the 0% water withdrawal scenario as the maximum habitat accessible within a watershed. Thus, a value of 1 would indicate that all habitat in the watershed remains accessible. We calculated the habitat accessible for each taxon in each watershed. We also determined the proportion of water withdrawn in each watershed. This allowed us to evaluate how different water withdrawal scenarios alter habitat connectivity for each taxon and the magnitude of change in water withdrawals necessary to increase habitat connectivity.

3. Results

The shrimp taxon had the highest habitat connected in all four watersheds when there was no water withdrawn (i.e. 0% scenario; Table 3.3). Similarly, shrimp experienced the highest loss rate of habitat connectivity as the proportion of water withdrawn increased (Figure 3.2) and had the highest proportion of habitat affected by barriers associated with water intakes within each watershed (Figure 3.3). Goby and snail taxa also demonstrated an elevated loss rate of habitat connectivity in response to increased water withdrawals, though this rate varied by watershed, with Espiritu Santo and Fajardo having the highest loss rates (Figure 3.2). Reducing water withdrawals to 50% increased the proportion of habitat accessible by shrimp, goby, and snail taxa across all watersheds, although the gain in habitat was smaller for gobies and snails in Blanco (Figure 3.2).

The taxa that climb over barriers (i.e. shrimp, goby) have proportions of habitat connected that vary over a wider range through time than fauna that cannot climb over barriers (Figure 3.4). Temporal variations in habitat connectivity for shrimp, goby, and snail likely result from stream discharge fluctuations because withdrawal rate and barrier height are constant, highlighting how the movements of these taxa are discharge dependent. The remaining taxa, mullet and eel, have less temporal variation in habitat connectivity, especially in the Blanco and Sabana watersheds (Figure 3.4). In these watersheds, a large proportion of the stream habitat (over 40%) is downstream of any barrier. Habitat connectivity for habitat downstream of a barrier can only be lowered if upstream intakes significantly reduce stream discharge (i.e. no discharge reaches the ocean). This occurs in Blanco and Fajardo in the 200% withdrawal scenario, where the farthest downstream intake reduced habitat connectivity downstream of the barrier (Figure 3.4). Thus, habitat connectivity for fauna that jump or ascend barriers have more constant connectivity through time as they experience limitations in habitat connected due to height, though large withdrawal amounts can decrease connectivity as well.

For eels, habitat connectivity response to changes in water withdrawals was mostly uniform across watersheds. Increasing water withdrawals did not decrease average habitat connected for eels until the 200% scenario, when connectivity declined in all watersheds except Sabana (Figure 3.2). In Blanco, most of the watershed is above a 20 m dam and that habitat will not be accessible to eel no matter how the water withdrawal scenario is altered. Eel habitat connectivity is not influenced by withdrawals unless discharge declines to zero.

The Espiritu Santo watershed highlights the importance of intake location in the watershed and associated habitat connectivity effects. The greatest range in habitat connected was in the Espiritu Santo watershed because it also has the highest number of intakes in a watershed. Additionally, those intakes farthest downstream have a high withdrawal rate such that the river can be overdrawn in the 100% scenario (Figure 3.3). The high number of intakes paired with barriers, as well as the largest intakes by volume near the river mouth, cause large fluctuations in habitat connectivity across all withdrawal scenarios for taxa that move upstream of barriers by climbing. Additionally, similarities in the median proportion of water withdrawn across watersheds (Figure 3.5) indicate that it’s not the volume of water withdrawn from a watershed, but where the intakes are located and whether there is an associated barrier height.

4. Discussion

Our analysis shows how reducing stream water withdrawal amount increases habitat connectivity for diadromous taxa that are discharge limited. In contrast, for taxa that are limited in their upstream movement by barrier height, there is a smaller positive effect. This distinction in movement restriction is highlighted by temporal connectivity patterns, because migratory taxa restricted by barrier height demonstrate static habitat connectivity, and taxa that are discharge dependent have connectivity that fluctuates through time. Additionally, intakes located closer to the ocean with an associated barrier and withdrawal have a larger effect on the proportion of habitat connected compared to intakes with no height or that are located farther upstream in the watershed. Our results emphasize the benefits of using previous life history knowledge of target taxa to develop management strategies that allow for social and ecological outcomes to be maintained (Smith et al. 2017), as barriers have varying effects across species (van Puijenbroek et al. 2018).

All our focal taxa had higher habitat connectivity when stream water intakes don't have associated instream barriers. Previous research recommended that intakes without a barrier should be installed to benefit shrimp populations so that constant longitudinal stream connectivity would be maintained (March et al. 2003). Our results suggest that intakes without barriers would benefit all migratory taxa, however, the eel would likely experience the largest gain in habitat connectivity. Because eels are catadromous, their movement upstream and downstream occurs at a late life stages and allows eel to avoid mortality from entrainment into water intakes, although they can still experience losses at hydropower dams (Smith et al. 2017). Additionally, intakes without barriers are more sensitive to changes in withdrawal rates. This is because barriers with partial passage can have connectivity more easily restored than barriers with zero passage probability (Diebel et al. 2014).

Similar to previous studies, we found that a higher number of barriers within a watershed tends to lower overall connectivity and that initial barriers affect connectivity more than additional barriers (Cote et al. 2009). This is especially true when considering the height of the initial barrier. Thus, management suggestions may need to be watershed specific. For example, a decrease in withdrawals in the Fajardo watershed (initial barrier height = 0 m) would increase habitat connectivity more than a similar withdrawal decrease in Espiritu Santo (initial barrier height = 1 m). Additionally, the amount of habitat that can be affected by an intake (i.e. upstream) should also be considered. If a higher amount of habitat is upstream of an intake where connectivity is determined by the discharge rate (i.e. no height), then that indicates that habitat will be lost at a higher rate with increasing water withdrawal. Additionally, a complete disconnection of the watershed resulting from downstream withdrawal effects is more likely when larger intakes are located lower in the watershed.

Previous research on connectivity for amphidromous taxa focused on their ability to move upstream above barriers (Fievet et al. 2001, Cooney and Kwak 2013); however, population persistence relies on downstream connectivity to the ocean for larval development. Research suggests amphidromous populations can be genetically connected across large spatial distances (Cook et al. 2009, Crandall et al. 2010, McMahon et al. 2012, USFS 2012). Thus, downstream passage rates should be considered when estimating habitat connectivity for amphidromous species, as reduced downstream connectivity has the potential to negatively impact the wider metapopulation. Not considering downstream passage rates may cause habitat connectivity to be overestimated for amphidromous species, especially in watersheds where the proportion of water withdrawn is nearly 100% for multiple days. Water withdrawals can also decrease habitat connectivity through lowered upstream passage rates, although discharge must decline to zero before habitat connectivity is affected.

Barrier removal prioritization analyses typically focus on increasing connectivity for one species (O'Hanley et al. 2005, Crook et al. 2009), even though multiple migratory species may exist in a watershed. Studies have begun to explore connectivity for multiple species (van Puijenbroek et al. 2018), but these analyses are still limited to one taxon (but see Neeson et al. 2015). Though focusing on one taxon may be appropriate in watersheds where there are a limited number of migratory species, tropical watersheds are often comprised of diverse migratory fauna that have different life history requirements (Bass 2003). In these systems a more effective approach to management might be to focus on barrier optimization strategies that assess how the entire community will be affected by changes in riverine connectivity. Previous studies have demonstrated a temporal migratory pattern for fauna movement in some watershed areas (March et al. 1998, Engman et al. 2017, Smith et al. 2017). If additional research confirmed these patterns occur throughout the watershed, more specific management recommendations could be suggested (i.e. maximize withdrawals during a specific moon phase).

Current stream water withdrawal rates could be reduced by fixing water leaks in pipes across the island, which are estimated to be 60% (Water Plan 2008). If leaks were even partially reduced, it is likely that the amount of water withdrawn would also decline. Additionally, given the large exodus of people (#) from the island in response to Hurricane Maria (REF), the decline in water demand may provide managers with some leeway to explore how to operate intakes in a way that supports ecological connectivity, especially given the potential for increased habitat connectivity for multiple taxa. The only taxon that would not experience gains in habitat connectivity (if stream water withdrawals were decreased) is the eel, because barrier height limits eel movement rather than the magnitude of water withdrawal amount. Though dams are typically discussed in relation to removal, altering water management operational schemes may present a more optimal solution in some cases, given the minimal tradeoffs (Smith et al. 2017) and financial investment needed. Additionally, dam removals may not always be effective in achieving desired ecological outcomes (Karieva 2000).

Our study builds on previous findings by Cooney and Kwak (2013) by exploring the effects of discharge and waterfalls on goby and nongoby habitat connectivity. By focusing on four watersheds, our results highlight that limited habitat connectivity for fish results from barrier height and discharge. Decreases in streamflow primarily lower downstream connectivity for larvae moving from headwaters to the ocean, but if intakes dry a stream, upstream passage will also be affected. Within the EYNF, water withdrawals can reduce stream discharge to zero (Benstead et al. 1999, Christian et al. 2019), especially during the dry period or drought years. Climate change predictions for this region suggest dry periods will become more frequent and intense (Jennings et al. 2014). If stream water withdrawal amounts remain static, the same withdrawal volume may have an even larger effect on habitat connectivity. The 200% scenario illustrates how a higher proportion of stream water withdrawal can result in lower connectivity across watersheds for shrimp, fish, and snail taxa.

Our analysis exploring how water withdrawal operations influence the proportion of habitat accessible across five taxa and four watersheds illustrates the importance of intake and barrier characteristics and location within a system. Specifically, our results show reduced habitat connectivity as a consequence of water intakes associated with barriers. It is also important to examine connectivity through time for those taxa that are limited in movement by discharge and not height. By examining how four different withdrawal scenarios alter temporal connectivity, we highlight watersheds with a small number of water withdrawals and no barrier height exhibit the largest gains in habitat when withdrawals are decreased. As managers attempt to maximize ecological benefits in relation to the water loss of decreasing withdrawals, watershed characteristics should also be taken into consideration.

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Table 3.1. Characteristics of the focal watersheds included in this study. We only include the barriers that are the closest to the ocean within each watershed. A withdrawal amount of zero indicates the intake was not operational in 2016, but the associated infrastructure remains in the watershed.

Watershed	Number of Intakes Online	Number of Waterfalls	Total Habitat (km)	Most Downstream Barrier/Intake	Height (m)	Withdrawal amount (cfs)	Distance to Ocean (km)
Blanco	4	3	110.5	Blanco River	0.9	0	10.85
				Waterfall	5	0	12
Espiritu Santo	5	4	80.6	Guzman- El Yunque	1	8.04	5.80
				El Verde	1.3	12.06	6.56
Fajardo	1	3	110	Fajardo Reservoir	0	18.27	11.32
Sabana	2	2	26.4	Sabana	1.5	2	13
				Cristal	1.5	2	14

Table 3.2. The five taxonomic groups we focused on to evaluate how habitat connectivity varies when water withdrawal operation patterns change.

	Shrimp	Goby	Snail	Mullet	Eel
Diadromous type	amphidromous	amphidromous	amphidromous	amphidromous	catadromous
Physiology/ movement mechanisms	Larvae: drift Juveniles and adults: crawl	Larvae: drift Adult: modified pelvic fin and mouth to climb barriers	Larvae: drift Juveniles and adults: crawl	Larvae: drift Adults: jump	Juveniles: Jump Adults: Swim using body; larger individuals tend to be older and stronger swimmers
Upstream passage qualitative factors	Discharge	Discharge Height	Discharge Elevation Height	Discharge Height	Discharge Height
Upstream passage equations	$\begin{cases} Q > 0 & 1 \\ Q = 0 & 0 \end{cases}$	$\begin{cases} Q > 0 & eqtn(coef) \\ Q = 0 & 0 \end{cases}$	$\begin{cases} E_{barrier} > 200m \; OR \; H_{barrier} > 5m \; OR \; Q_r = 0 & 0 \\ else & 1 \end{cases}$	$\begin{cases} Q > 0 & eqtn(coef) \\ Q = 0 & 0 \end{cases}$	$\begin{cases} Q > 0 & eqtn(coef) \\ Q = 0 & 0 \end{cases}$
Downstream passage qualitative factors	Discharge	Discharge	Discharge	Discharge	Discharge
Downstream passage equations	$\frac{Qbarrier}{Qdownstream}$	$\frac{Qbarrier}{Qdownstream}$	$\frac{Qbarrier}{Qdownstream}$	$\frac{Qbarrier}{Qdownstream}$	$\begin{cases} Q > 0 & 1 \\ Q = 0 & 0 \end{cases}$
Habitat limitations	Discharge < 0	Discharge < 0	Discharge < 0 Barriers > 5 m	Discharge < 0 Barriers > 5 m	Discharge < 0 Barriers > 3 m

		Barriers > 30m	Elevation > 200 m		
References	Engman et al. (2017), Benstead et al. (1999), March et al. (1998), Covich et al. (2009)	Covich and McDowell (1996), Cooney and Kwak (2013), Engman et al. (2017), McDowall (2009)	Blanco and Scatena (2005), Pyron and Covich (2003), Covich and McDowell (1996)	Cooney and Kwak (2013)	Cooney and Kwak (2003), Kwak et al. (2018), Smith et al. (2017), Righton et al. (2016), Galbraith et al. (2012)

Table 3.3. The habitat connected for each taxon. We found the habitat connected by taking the product of the index of longitudinal riverine connectivity (ILRC) and the habitat upstream of the intake (in km).

	Blanco (km)	Espiritu Santo (km)	Fajardo (km)	Sabana (km)
Shrimp	110.49	80.61	109.96	26.43
Goby	77.54	59.21	97.11	23.19
Snail	80.03	46.02	97.75	19.92
Mullet	66.17	30.51	89.40	17.94
Eel	60.78	11.33	92.76	15.59

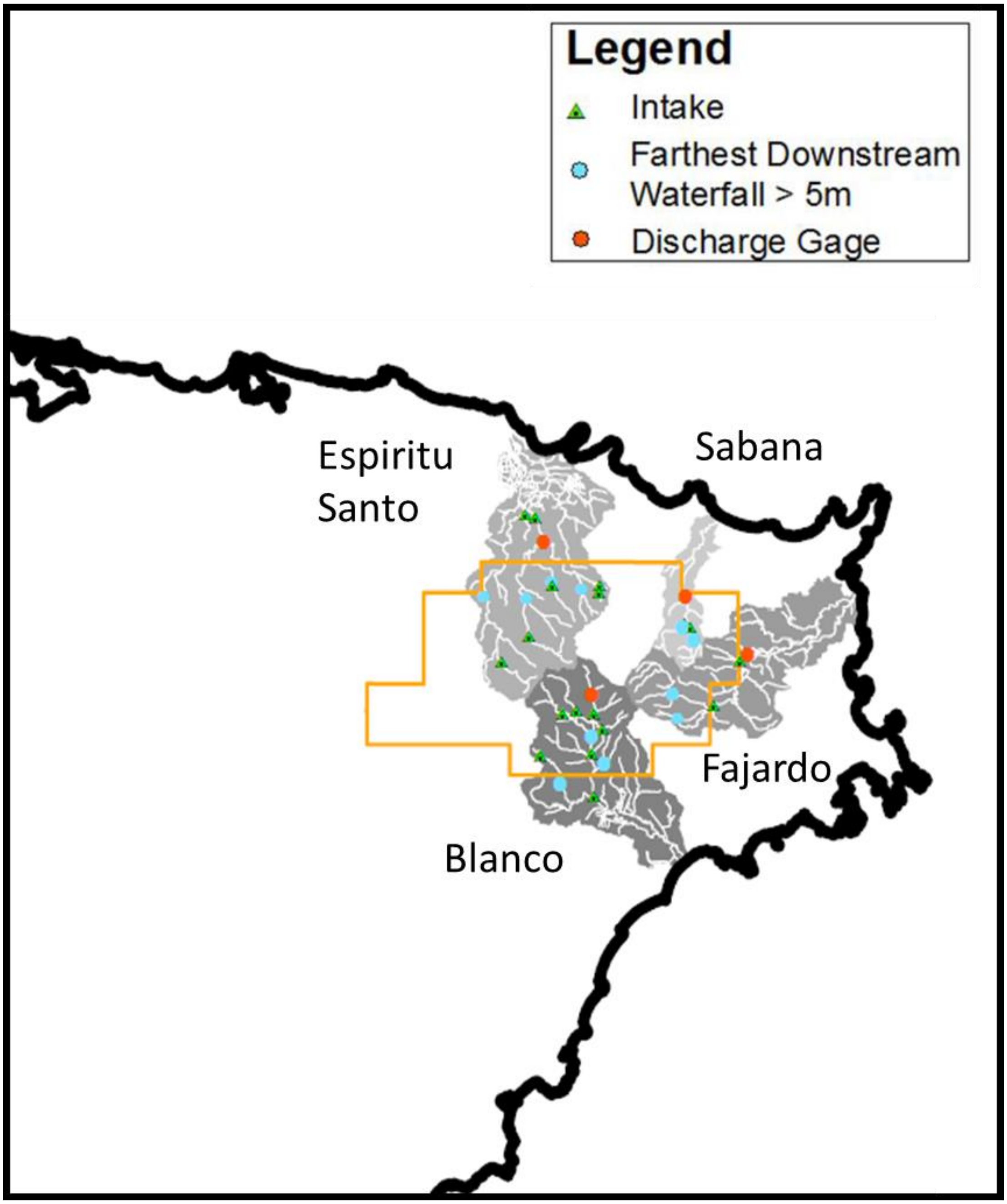

Figure 3.1. Map of four watersheds used in analysis. Each watershed is labeled, and the drainage area is colored in distinct shades of gray. The orange line represents the boundaries of EYNF. The white lines indicate the stream flowlines. The red points are the USGS discharge gages, and the green triangles represent intakes. The blue circles

indicate the farthest downstream waterfall over 5 m in height, which were identified using elevation differences between adjacent DEM raster cells in ArcGIS. Inset: Puerto Rico and the study watersheds.

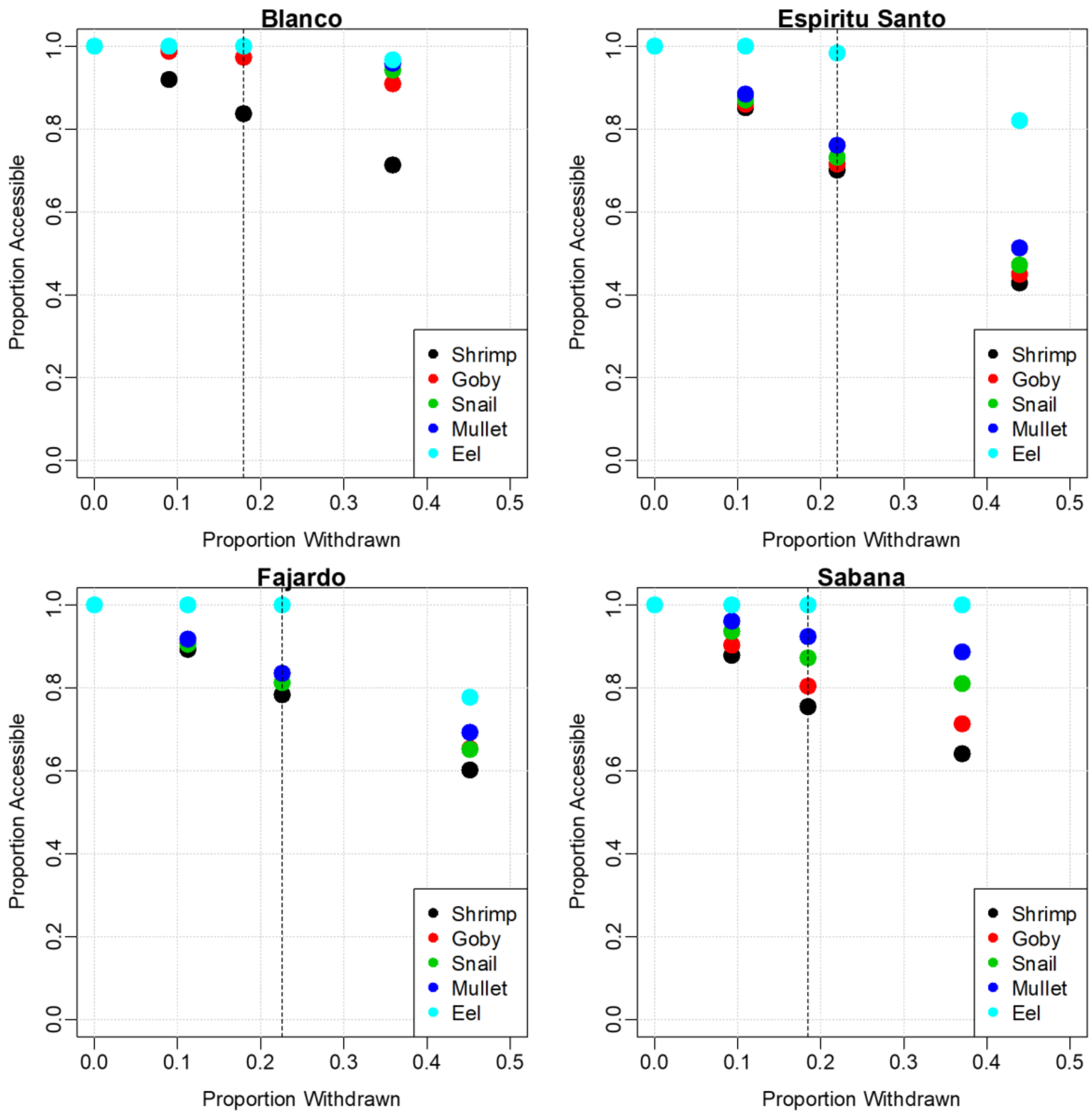
Blanco
Espiritu Santo
Fajardo
Sabana
Proportion Accessible
Proportion Withdrawn
0.0
0.2
0.4
0.6
0.8
1.0
0.0
0.1
0.2
0.3
0.4
0.5
Shrimp
Goby
Snail
Mullet
Eel

Figure 3.2. The average proportion of habitat accessible compared to the average proportion of water withdrawan for each taxonmic group and watershed. The proportion of water withdrawn is based on four different water withdrawal scenarios (0%, 50%, 100%, 200%). All taxonomic groups have a proportion of habitat accessible of 1 for 0% withdrawal because the maximum habitat connected is also accessible for the taxa within the waterhsed. The dotted line in each watershed represnts the proportion of water withdrawn at the 2016 intake rate (100% withdrawal scenario).

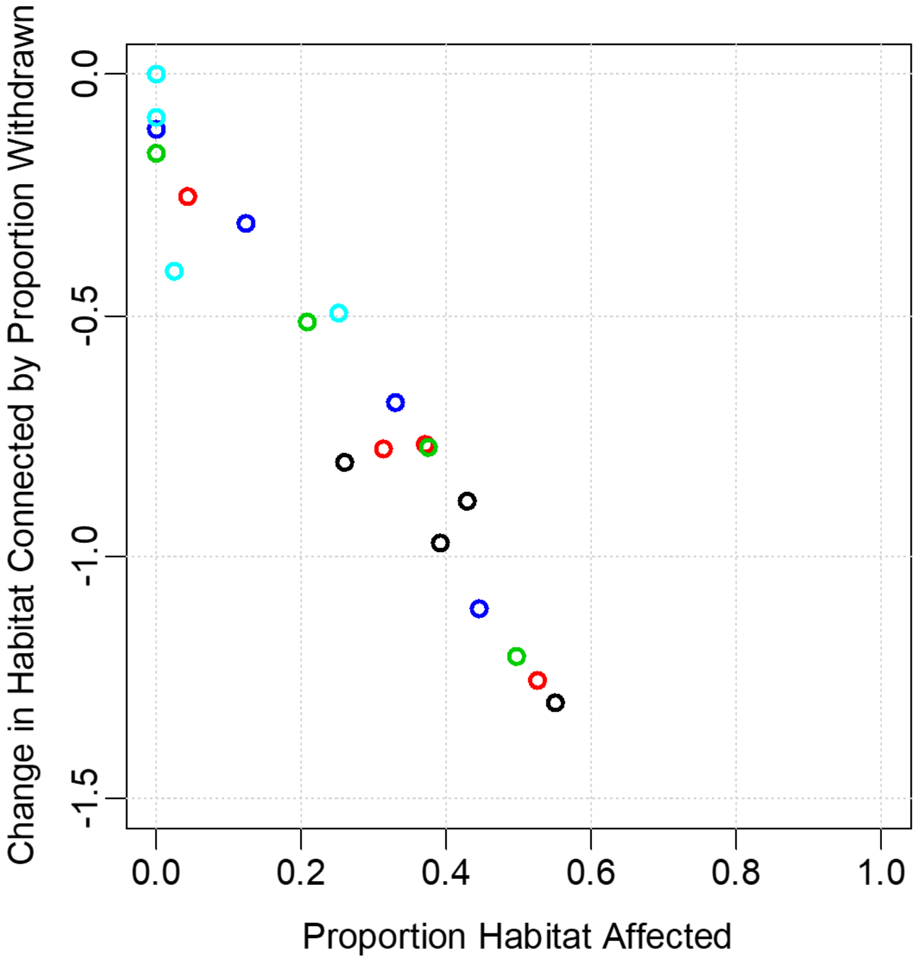

Figure 3.3. The rate of habitat loss with increasing withdrawal rates for five taxonomic groups (i.e. slopes from Figure 2) plotted in relation to the group-specific amount of habitat that is upstream from intake-associated barriers in each watershed. The colors represent the five groups: shrimp (black), goby (red), snail (green), mullet (dark blue), and eel (light blue).

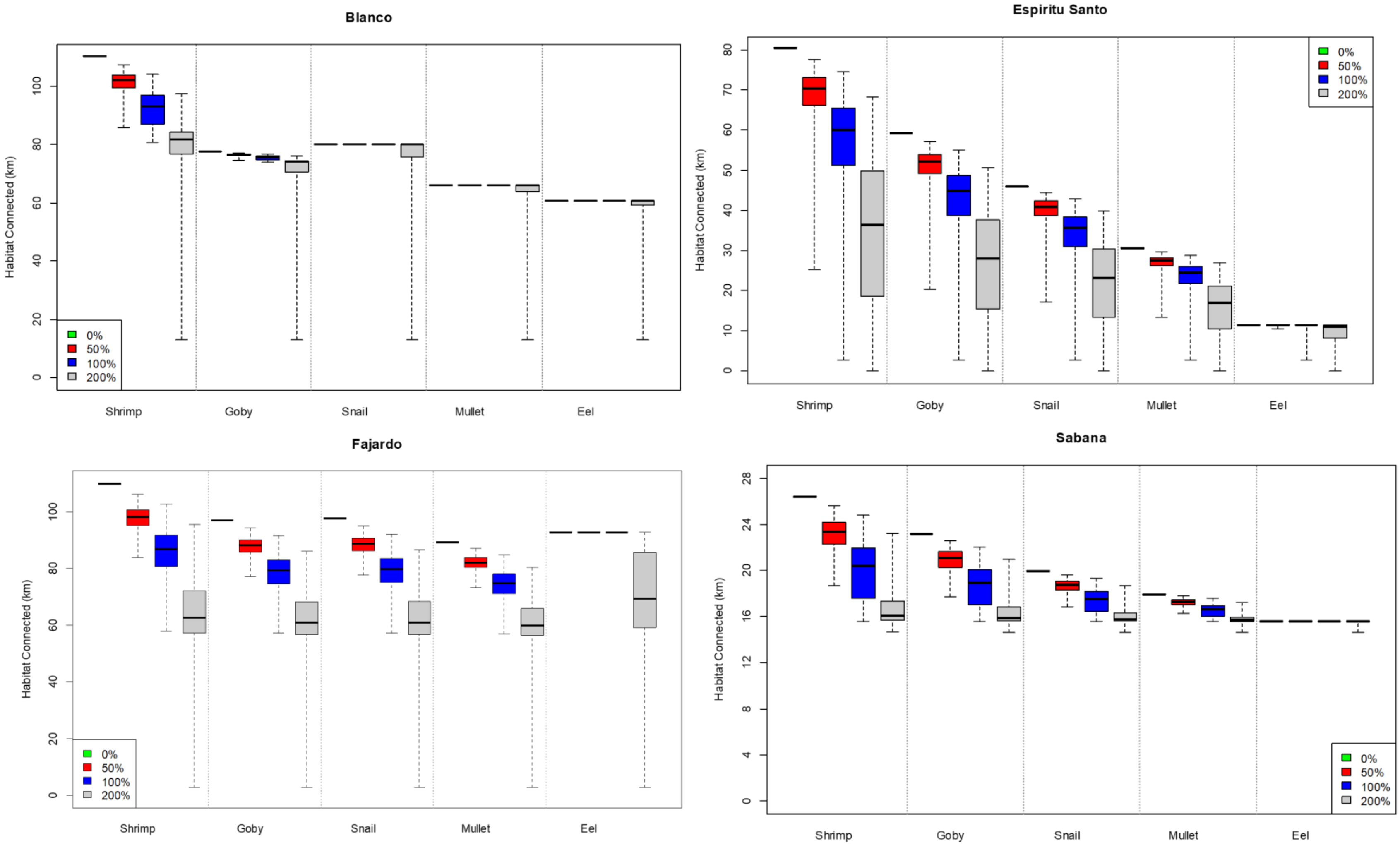
Blanco
Espiritu Santo
Fajardo
Sabana
Habitat Connected (km)
Shrimp
Goby
Snail
Mullet
Eel
0%
50%
100%
200%

Figure 3.4. Habitat connected (km) for each taxon and each water withdrawal scenario in four watersheds. For each box plot, the black line represents the median, the box extremes are the 25th and 75th quartile, and the whiskers are the maximum and minimum values calculated monthly over the 31-year dataset.

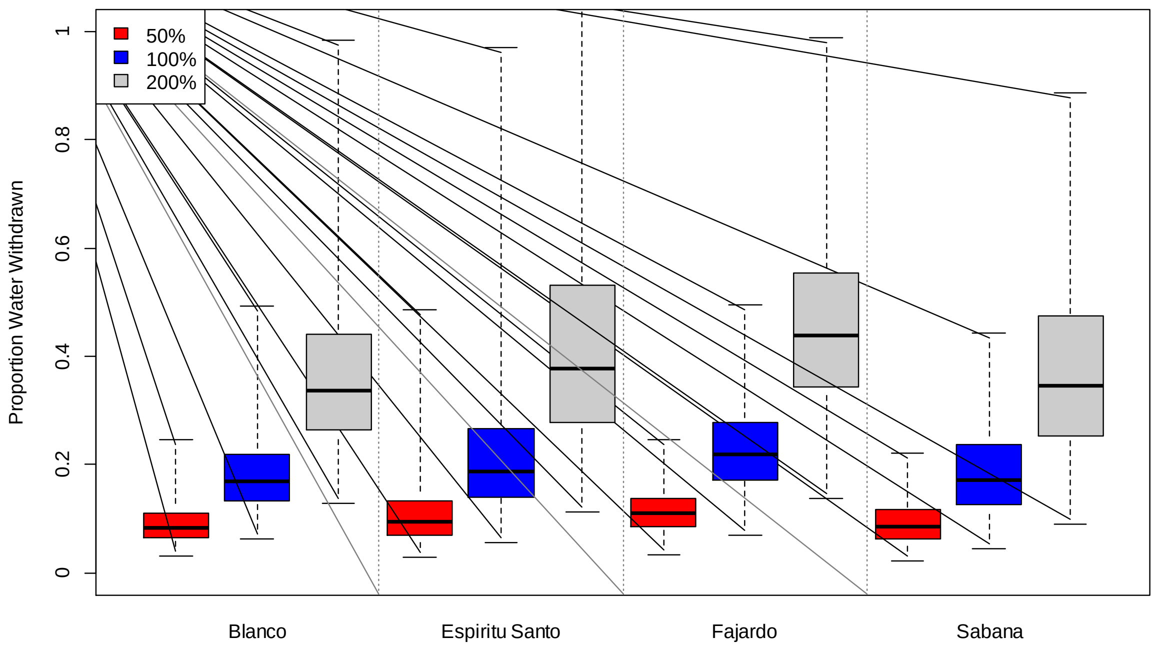

Figure 3.5. Proportion of water withdrawn from each watershed across three water withdrawal scenarios. For each box plot, the black line represents the median, the box extremes are the 25th and 75th quartile, and the whiskers are the maximum and minimum values calculated monthly over the 31-year dataset.

CHAPTER 4

EVALUATING ACCOUNTABILITY FOR ENVIRONMENTAL FLOWS: A FRAMEWORK AND APPLICATION TO PUERTO RICO[3]

[3]Chappell, Jessica, Laura German, S. Kyle McKay, Cathy Pringle. To be submitted to *Environmental Science and Policy*.

Abstract

Freshwater is essential to human communities and stream ecosystems, and governments strive to manage water to meet needs for both people and the ecosystems they depend on. Balancing competing water demands is challenging as freshwater resources are limited and its availability varies through time. One approach to ensure this balance is to legally mandate a specified amount of streamflow be maintained for stream ecosystems, also known as an environmental flow. But laws and regulations do not necessarily reflect what happens on the ground, potentially to the detriment of communities and natural systems. Through a case study of the island of Puerto Rico, we have investigated whether water management in practice matches legislative mandates and explored potential causes of this mismatch. Specifically, we seek to understand the formal and informal aspects of environmental flow enforcement. We focused on two governance targets, equitable allocation and water use efficiency, as their existence would promote environmental flow enforcement. We assessed whether the governance targets are enshrined in the law (*de jure*) and whether and how they manifest in practice (*de facto*). We then used an evaluation of water management agency accountability to better understand agency motivations. Specifically, we explored the accountability of agencies through evaluating agency structure and whether consequences are enforced for failing to carry out responsibilities. Our results indicate there are mismatches in how equitable allocation and water use efficiency are managed by law and in practice. Additionally, the structure of state agencies and domestic water use prioritization contribute to the misalignment in how water management is carried out on the ground versus what is legally mandated. By identifying two water targets in both legislation and practice, we developed a more holistic understanding of environmental flow management in Puerto Rico. Also, our

analysis suggests agency accountability may be an additional factor to consider when developing environmental flow legislation that will be effective in meeting ecological outcomes.

1. Introduction

Water management via dams, withdrawals, and other infrastructure reduces river flow and creates cascading effects such as altered ecological function, decreased biodiversity, and loss of ecosystem services (MA 2005, Poff and Zimmerman 2010). Over 750 scientists from 50 countries have articulated the need to maintain river flows for environmental protection in the Brisbane Declaration, which defines environmental flows as “the quantity, timing, and quality of water flows required to sustain freshwater and estuarine ecosystems and the human livelihoods and well-being that depend on these ecosystems” (Brisbane Declaration 2007). However, crafting environmental flow policies has proven to be difficult. Significant ecological complexity, including multiple ecological goals (e.g., maintaining fish populations, good water quality), a variety of methods to assess outcomes (i.e., multiple measurement methods and models), significant hydrologic variability across river systems (Arthington et al. 2006), and multiple dimensions of hydrologic processes (i.e. magnitude, frequency, duration, timing, and rate of change of river discharge or level) hampers the ability to prescribe environmental flows (Poff et al. 1997, Petts 2009, Poff and Zimmerman 2010).

In parallel with ecological complexity, social and political impediments also interfere with policy development, such as the misallocation of power and responsibilities within agencies (Ribot 2004), financial limitations (Le Quesne et al. 2010), and regulations lacking sufficient enforcement mechanisms (Fine et al. 2017). Additionally, as demonstrated by a growing body of literature addressing environmental flow governance issues (e.g., Pahl-Wostl et al. 2013, Brownson 2019, Baumgardner 2019), legislation implementation remains challenging. Even when flow protection policies are legally in place, there is little evidence of enforcement, e.g. in southeastern states of the

U.S. (Baer and Ingle 2016). This misalignment between governance targets enshrined in the law (a *de jure* perspective) and whether and how they manifest in practice (a *de facto* perspective) is a recognized factor affecting science policy outcomes (German et al. 2013, Fine et al. 2017, Kuyper et al. 2017).

To explore potential misalignment of legislation and practice, we developed a conceptual framework consisting of governance targets and evaluation criteria. We focused on two water governance targets: equitable allocation (water distributed reasonably among users) and water use efficiency (use of the minimum amount of water possible to accomplish a task). These two targets represent potential mechanisms to facilitate environmental flow legislation enforcement. Specifically, equitable allocation provides a basis for recognizing the legitimate water needs of multiple societal functions, including ecological functions supported by flow-dependent ecosystems. Maximizing water-use efficiency should increase the probability that there will be sufficient water available to share among users, including ecosystems.

Governance research has revealed that environmental management in practice often departs from formal governance systems established by law (Smith 1988, Schlager and Ostrom 1992). One factor that seems to limit federal agency policy implementation is lack of political oversight (Daley and Layton 2004). Indeed, policy research finds more complex legislation is generally less tractable to those responsible for implementation (Manna and Moffit 2019). In water management systems, lack of regulation implementation can result because the necessary components are not outlined by law, leaving major implementation decisions up to individual agencies (Kiparsky 2016, Cantor et al. 2018). To evaluate differences between legislation and practice in developing water governance regulations, a two-pronged *de jure* and *de facto* approach has been used to highlight mismatches between who participates and influences water policy formation (Jensen-Ryan 2017).

In this paper, we explored *de jure* and *de facto* water governance to better understand how environmental flows are managed in Puerto Rico, where high water variability resulting from annual dry seasons and periodic drought conditions limits the amount of available freshwater to meet human and ecological needs. We also examined whether the accountability of water management agencies at the state level influences the implementation of environmental flow legislation. Identifying accountability in a governance system helps establish a network of influence (Black 2008) and determines actors' motivations. Our findings contribute to the ongoing conversation of identifying factors that need to be addressed to move towards effective environmental flow governance (Pahl-Wostl et al. 2013, Horne et al. 2017).

1.1 Accountability

According to Bäckstrand (2008), accountability means "liable to be called into account; responsibility". Based on this requirement, at least two actors are needed for accountability to exist: one actor demonstrates they have met their responsibilities, and a separate actor enforces consequences if responsibilities are not met (Ribot 2002, Bovens 2007, Bäckstrand 2008, Kuyper et al. 2017). Many accountability types exist and are distinguished by "to whom" an actor gives account (Cedón 2000, Bovens 2007). Legal accountability is "based on specific responsibilities, formally or legally conferred upon authorities" (Bovens 2007). Political accountability refers to party allegiance and is defined by adherence to party expectations (Cedón 2000).

Accountability is useful as it illustrates which actors have influence over particular standards or expectations, such as environmental outcomes (Ribot 2004), highlights the relationship between actors, and ultimately increases our knowledge of how actors function within a governance system. For example, Agrawal and Ribot (1999) identified whether downward accountability of policy makers to citizens existed as an indicator of effective management decentralization. Though multiple mechanisms for identifying accountability exist (Agrawal and Ribot 1999, Mashaw 2006, Bovens 2007), we identified agency accountability through the existence of consequences for not following through with responsibilities (Cedón 2000).

2. The complexity of water management in Puerto Rico

Our case study is focused on Puerto Rico, a tropical island that is U.S. territory and that receives 3229 mm of annual rainfall on average. During the dry season (December to April) rainfall is 96 mm per month on average island-wide (Water Plan 2008). Droughts are relatively infrequent on the island, with only five major droughts (persisting for two years) occurring since 1899 (Larsen et al. 2000). Surface water is the main source of freshwater for the 3.195 million people that live on the island, and 27% of streams on the island are upstream of large dams (>5m; Snyder et al. 2011). Puerto Rico has 14 major reservoirs (Water Plan 2008), resulting in the island having one of the largest densities of dams over 15 m in the world (Greathouse et al. 2006). El Yunque National Forest (EYNF) has the highest rainfall on the island, receiving up to 4598 mm annually in some areas (Jennings et al. 2014), and it is an important water source for local communities with over 29 intakes on watersheds draining the forest (Chappell et al. 2019). Many aquatic organisms in the stream ecosystem are diadromous (i.e. migrate from rivers to the ocean) (Covich and McDowall 1996), and ecologists have identified the importance of leaving a minimal flow at dams and intakes to facilitate migratory shrimp movement (Benstead et al. 1999). Migratory shrimp maintain water quality by minimizing sediment and algal accrual (Pringle 1996, Pringle et al. 1999).

2.1 Water management agencies

Freshwater in Puerto Rico is managed similarly to states within the U.S.: federal agencies oversee water quality and navigation, and water quantity decisions are made by state agencies. Three state agencies are involved with managing water quantity in Puerto Rico: *Departmento de Recursos Naturales y Ambientales* (DRNA), *Autoridad de Energía Eléctrica* (AEE), and *Autoridad de Acueductos y Alcantarillados* (AAA). DRNA was created by the Commonwealth for managing, protecting, and sustainably developing the "natural, environmental and energy resources of the Island" (3a L.P.R.A § 2) and is led by a Governor-appointed Secretary (3a L.P.R.A § 1). DRNA is responsible for creating the Water Plan, which outlines how each water body is used along with water conservation and management goals.

The public energy corporation, AEE, legally controls the distribution of water within the hydropower reservoirs (22 L.P.R.A. § 196a) and is managed by a board of directors appointed by the Governor. Finally, AAA is the public water corporation created by the Commonwealth government and is responsible for setting and charging fees to finance the water and sewage services it provides to citizens (22 L.P.R.A. § 144). AAA can make water conservation regulations regarding domestic use and is charged with ensuring a consistently available water supply for its consumers (22 L.P.R.A § 159). AAA is managed by nine members on the board of directors, five of which are selected by the Governor and confirmed by the Senate (22 L.P.R.A. § 143). Though AAA has water resource regulatory responsibilities assigned by state law, it is also part of the regulated water user community. For example, AAA waste discharge must adhere to federal standards and regulations set by the Clean Water Act and state regulations.

Federal agency jurisdiction extends to water management issues related to barriers within navigable waters, federally listed endangered species, and water quality. If an instream barrier will be placed in a navigable river, permission must be obtained from the U.S. Army Corps of Engineers (ACE) and authorized by the Secretary of the Army (33 U.S.C.A. § 403). As per the Endangered Species Act (1972), the U.S. Secretary of the Interior coordinates interagency efforts to protect endangered species and their habitat (16 U.S.C.A. § 1533). The US Fish and Wildlife Service (USFWS) must also be consulted if the waters proposed to be impounded may affect federally listed endangered species (16 U.S.C.A. § 662).

Within the EYNF boundaries, the US Forest Service (USFS) has authority to grant access to freshwater resources. Waters within national forest boundaries are available for domestic purposes (16 U.S.C.A. § 481); however, the U.S. Secretary of Agriculture must review plans for the intended water use prior to granting a right-of-way (43 U.S.C.A. § 1761). In terms of water quality, the U.S. Environmental Protection Agency (EPA) is tasked with developing and enforcing water quality standards and issuing wastewater effluent permits under the Clean Water Act (33 U.S.C.A. § 1251d). The EPA also assists the ACE in issuing wetland disturbance permits (33 U.S.C.A. § 1344) and delegates the in-state implementation of water quality regulations to a state-level agency. In Puerto Rico, this agency is the Junta de Calidad Ambiental (JCA), and its board is appointed by the Governor (12 L.P.R.A. § 8002a).

2.2 The ecology of Puerto Rican streams

More than 30 years of stream studies in EYNF have highlighted the ecological importance of maintaining some flow within streams to support ecosystem functions (Holmquist et al. 1998, Pringle 2001, Crook et al. 2009). Evidence suggests even a trickle of water can be enough to allow for some migratory aquatic organisms to complete their life cycle (J. Benstead, J. March, P. Torres personal observation) and sustain stream ecological functions, underscoring the disproportionately large effect that maintaining streamflow can have on the ecosystem. A minimum flow regulation mandating that the Q99 (the flow exceeded in the river 99% of the time) must be maintained in certain circumstances has been in place since 2008 (Plan de Agua 2008). The importance of environmental flows ecologically, as well as the existence of a minimum flow regulation, provide a strong basis for examining environmental flow governance specifically.

We focused on whether environmental flows are enforced in Puerto Rico using the minimum flow requirement as a proxy throughout our analysis. Although it has long been recognized that maintaining flow magnitude is not sufficiently comprehensive to ensure stream function (Poff et al. 1997), the environmental flow policy in many states only protects minimum flows (Baer and Ingle 2016). Additionally, a minimum flow in Puerto Rico’s stream systems is ecologically significant in terms of maintaining aquatic fauna. Specifically, aquatic migratory species rely on flowing water to move up and downstream to complete their life history (Benstead et al. 1999, Torres personal communication), and these animals underpin many crucial ecosystem functions in the island’s streams (Greathouse et al. 2006). This includes reducing sediment accumulation and minimizing algal growth (Pringle 1996, Pringle et al. 1999).

3. Methods

We developed a conceptual framework to evaluate *de jure* and *de facto* environmental flow governance relative to two targets: equitable allocation and water use efficiency (Figure 4.1). Equitable allocation is the principle that every water user has the right to a reasonable amount of water proportional to its needs (Wohlwend 2001). Though applied to water shared across political boundaries, here we examine equitable allocation across different users or sectors (i.e. domestic, industry, agriculture, and the environment). Water use efficiency may be defined as “the accomplishment of a function, task, process, or result with the minimal amount of water feasible” (EPA 2016). When equitable allocation and water use efficiency exist by law and in practice, environmental flows are more likely to be achieved (Figure 4.1). Specifically, a legal platform ensuring users have water access (equitable allocation) and minimizing water waste (water use efficiency) can contribute to maintaining aquatic ecosystems (EPA 2016). We used minimum flow (i.e. the minimum quantity of water required to be left in a river) as an indicator of environmental flow protection, although we acknowledge minimum flows do not encompass a river’s entire flow regime and may not maintain river ecosystem functions (Poff et al. 1997).

We identified each water target in legislation (*de jure*) and in practice (*de facto*) using the criteria outlined in Table 4.1. We located statutes and regulations through Westlaw (Thomson Reuters Corporation 2017), using the following search terms: Puerto Rico, water, allocation, efficiency, water use, environmental flow, and minimum flow. To pinpoint the water targets in practice, we conducted interviews with key informants from federal and state agencies involved in Puerto Rico water management, specifically: DRNA, AAA, AEE, JCA, *Departamento de Agricultura* (DA), USFS, ACE, and USFWS. Semi-structured interviews were conducted in Puerto Rico from March 2016 to August 2016, with a list of pre-determined questions asked of each interviewee (Appendix 1). The final sample consisted of ten individuals who worked or provided consulting services for state agencies, and five individuals who worked or consulted for federal agencies. We identified the accountability of state agencies involved with environmental flow management using a combination of our *de jure* and *de facto* analysis results. We included specific questions related to inter-agency relationships into the interview checklist. The consequences agencies experienced if responsibilities were not met had to be documented within legislation or stated by interviewees.

4. Results

4.1 Equitable allocation: de jure

Puerto Rican legislation states "… the waters and bodies of water of Puerto Rico are the property and wealth of the People of Puerto Rico. The Commonwealth Government shall administer and protect this patrimony in the name and on behalf of the Puerto Rican people" (12 L.P.R.A. § 1115a). The Water Law assigns DRNA the responsibility for creating a water plan, which determines the uses of the island's freshwater for multiple users. These users include "the natural, social and economic systems which depend on the resource for their subsistence and development" (12 L.P.R.A. § 1115a; Table 1, #1). The Commonwealth is committed "to achieve the most equitable and fairest distribution of its waters" (12 L.P.R.A. § 1115a); however, the only user allocated a specific water amount is the environment (Table 1, #2). The Water Plan specifies the Q99 (which is the site-specific discharge that is exceeded 99% of the time, estimated on a daily time-step) be maintained downstream of new intakes (Plan de Agua 2008). There are no existing court cases that clarify how equitable allocation decisions are made in Puerto Rico (Table 1, #3).

DRNA is the agency that issues water withdrawal permits (or “franchises; 12 L.P.R.A. § 1115d). Withdrawal permits are only issued for up to ten years at a time, and the agency sets the permitting fee based on water volume (12 L.P.R.A. § 1115h). When a water user submits a permit request, the application must include evidence demonstrating there is substantial natural streamflow to support the withdrawal amount at the proposed site. DRNA must take into consideration the impact of the proposed water withdrawal on multiple entities, such as the economy and the environment (Table 1, #4). Domestic or agricultural users applying for small withdrawal amount permits may not have to comply with the application process or pay the associated fee, as the “water requirements inherent in domestic consumption and particularly human consumption, shall be satisfied with priority over any others” (12 L.P.R.A. § 1115n).

4.2 Equitable allocation: de facto

The interviewees indicated three areas where practice did not meet legislation. The first is poorly informed instream flow management, resulting from DRNA’s unkempt permitting process system. Intake information is not well tracked by the agency, including intake location, number of intakes per watershed, and water withdrawal amount. This makes it difficult for DRNA to accurately assess whether an additional intake installed on a river with multiple existing withdrawal permits will decrease streamflow below the minimum threshold (Table 1, #5). Though DRNA is the agency who approves the water withdrawal permits, the permit application is reviewed for different components (i.e. environmental impact, instream barrier construction, etc.) by multiple entities, including federal agencies (federal employee July 2016). USFWS, for example, determines whether the permit will impact endangered or threatened federally listed species. The USFS weighs in if a proposed intake is located on, or will affect, national forest land. In the early 1990s, a new intake was proposed for Rio Mameyes, a barrier-free river originating from EYNF. AAA submitted a proposal to install a dam paired with a water intake that would remove 15 MGD (million gallons per day). One interviewee walked us through USFS's response to this proposal.

> "We said… first of all, you don't have 15 [MGD], and we realized that the most that river could give in a safe way would be 5 MGD…And we said 'You are going to dry the river by your proposal.' And we're very cognizant of not drying the river and keeping always a base flow. So that's why we came up with [the] solution…put a limit on it to 5 MGD so that it will always maintain flow, it will never go dry. But in that case, what we did as an agency was to use the research…to defend our responsibility over the organisms that live inside the forest [but] depended on the [river] outside for their survival" (federal employee, June 2016).

Another mismatch in practice is ineffective enforcement of environmental flows during drought (Table 1, #6). During the 2015 drought, one of the most severe droughts in Puerto Rico over the last 100 years, DRNA was contacted by concerned citizens stating that Rio Blanco, a river flowing from EYNF to the ocean, was dry. This river contains a dam with a V-notch weir, intended to create an instream reservoir for AAA and maintain a minimum flow. However, AAA employees had deliberately obstructed the V-notch weir to fill the reservoir, as the drought-reduced streamflow was unable to do so naturally. This was in violation of AAA's water withdrawal permit (issued by DRNA), which states the minimum flow is 2.26 MGD (OPA June 22, 2015). DRNA forced AAA to remove the obstruction and gave AAA recommendations to maintain minimum flows in Rio Blanco, as even lower flows would stress the downstream ecosystem (OPA June 22, 2015). However, consistently maintaining minimum flows across the island appears to be challenging. For example, more water was withdrawn than was available in Rio Espiritu Santo (another river originating from EYNF) on at least 30 days during the period from 2005-2013 (Christian et al. 2019).

A final mismatch revealed through our interviews is the failure to ensure each water user receives an amount proportional to their needs. In southeastern Puerto Rico, AEE operates hydropower and irrigation systems involving multiple reservoirs. The hydropower reservoirs were established in the early 1900s partly to support agriculture, but AEE began to sell the excess water to AAA as the farming industry declined (state government employee, July 2016). Reservoir maintenance represents a significant financial loss to AEE, as it costs \$300 per acre foot to maintain the system and farmers pay \$2 per acre foot (state government employee, July 2016). When the DA pushed to increase the island's food security in 2012, DRNA informed the DA there was not enough water available to increase the agriculture sector's allocation (Table 1, #7). This situation is complicated, as the DA was trying to reclaim hydropower water now used by the domestic sector, which originally belonged to the farmers. In this case, DRNA does not seem to prioritize equitable allocation of Puerto Rico's waters to allow all users a reasonable amount to meet their needs, even when the water previously belonged to the user asking for more water.

Although DRNA may enforce the minimum flow regulation in extreme cases (i.e. when the stream is dry), we did not find evidence DRNA is held accountable for this responsibility (i.e. there are no agency consequences when Rio Espiritu Santo is overdrawn). DRNA also does not seem to be held accountable for achieving equitable allocation of water across users.

4.3 Water use efficiency: de jure

The Commonwealth is tasked with ensuring the island's water bodies are used in the public's interest "and with their best, most beneficial and most reasonable use…to protect our people from the adversities of the shortage, misuse, waste and pollution of such an essential resource" (12 L.P.R.A. § 1115a; Table 1, #8). Though DRNA, AAA, and AEE are all tasked with conserving water through legislation, DRNA is the agency legally responsible for creating criteria to evaluate the most beneficial and reasonable water use, as well as defining water "waste" (12 L.P.R.A. § 1115d). However, what constitutes a water waste is unclear, as the Water Plan states water should be used reasonably and in the public's best interest. Not surprisingly, violations for wasting water are not clearly stated in legislation or in the Water Plan (Table 1, #9). However, AAA is granted the authority to enforce rules to maintain its operations (22 L.P.R.A. § 144), indicating the agency can impose water use restrictions (or rationing) on AAA customers.

AAA is autonomous from the government, especially regarding financial obligations (22 L.P.R.A. § 144), and this separation was legally affirmed (U.S.C.A. No.

91-1602). Even though AAA argued “the Commonwealth’s significant financial support of [AAA]’s activities” indicates it is part of the state government, suggesting it can’t be sued, Puerto Rico’s own Supreme Court has “consistently concluded that [AAA] is not an alter ego of the central government” (Opinion, SELYA). This indicates public corporations, such as AAA and AEE, can be sued (Table 1, #10). AAA has also been sued by the EPA, in conjunction with JCA, as AAA wastewater treatment facilities were not in compliance with environmental standards set by the Clean Water Act (1977). Though AAA applied for leniency in meeting the secondary treatment standards, the requests were denied and the standards were upheld by the federal Court of Appeals (U.S.C.A. No. 93-2340). However, though AAA can be sued in court, we did not find any court case examples related to efficiency (Table 1, #11).

4.4 Water use efficiency: de facto

The interviewees indicated two areas of mismatch between water use efficiency legislation and what occurs in practice. One of these is concentrated regulatory responsibility, as AAA seems to be the main agency responsible for water use efficiency in practice (state government employee, July 2016). This is a result of the high water loss that occurs in transfer from AAA treatment facilities to domestic users. In 2014, AAA received 450 calls per day from individuals identifying leaky pipes, yet the agency responded to less than 25% on average (NotiCel July 13, 2014). Additionally, high water use efficiency directly benefits AAA’s profit margin as leaks represent a financial cost for the agency. For example, a leakage rate of 60% indicates AAA loses 60% of its product without any financial compensation (former AAA employee, July 2016).

An additional mismatch we identified is inconsistent enforcement of water use efficiency standards. Though it can be difficult to pin-point an example of water waste as it is not clearly defined, droughts force water managers to conserve the resource by identifying water wastes. Beginning in May 2015, AAA enforced water rationing for its 1.5 million domestic users in the island's north-central and northeastern regions. Residents were fined $250 for using potable water to wash patios, roofs, or cars, as this was considered a waste (AAA website June 17, 2015). Additionally, to conserve water, AAA limited the amount of time domestic users had running water by turning the water delivery system off and on; however, inconsistent pipe pressure exacerbated existing leaks and caused new ruptures. Multiple leaks were reported throughout the metropolitan water delivery zone during water rationing, including one along a major interbasin transfer, resulting in the "loss of millions of gallons of potable water" (El Nuevo Día July 24, 2015). Though water loss perpetuated by AAA actions possibly intensified or prolonged water rationing for domestic users, it was not formally identified as waste, suggesting a lack of clear examples of agencies wasting water (Table 1, #12). Additionally, no consequences were enforced for this major waste of water despite multiple regulatory agencies having legal responsibility for water conservation. Thus, it does not appear state agencies experience consequences for wasting water in practice (Table 1, #13).

Similar to equitable allocation, agencies responsible for maintaining high water use efficiency through limiting water waste do not seem to face consequences if these responsibilities are not carried out. For example, though AAA lost 60% of its treated water during a drought, the agency experienced no consequences. However, AAA was held accountable for its water quality responsibilities by the EPA, as the agency was forced to pay a fine for violating federal water quality regulations (U.S.C.A. No. 93-2340).

We also found evidence that state agencies are subject to political accountability as the Governor can enforce consequences if responsibilities aren't met. This opportunity is created from the gubernatorial appointment of agency leadership. All water management agencies are directed or overseen by individuals appointed by the Governor. Agencies seem expected to fulfill the Governor's desires, and consequences exist if this does not occur, indicating political accountability. One interviewee stated:

> "…the first Secretary [DRNA] had in 1976. He disobeyed, or did not comply with, a requirement of the office of the Governor. So, he got … a letter for his resignation. No choice. Sign or we fire you. So, he was fired. The head of [AAA]... he was fired…by the Governor" (AAA consultant, May 2016).

The Governor's influence is highlighted by exploring how domestic user water rates are set in practice. Although AAA is legally able to independently set water rates, the agency must have the Governor's support to charge more for water in practice (former AAA employee, July 2016). This is because citizens perceive the water rate is the Governor's responsibility, as AAA is managed by individuals appointed by the Governor. An increase in water rates thus triggers a decline in the Governor's public approval and the potential loss of a re-election. Due to the existence of consequences for going against the Governor, AAA does not increase water rates at the level necessary to finance its operations (former AAA employee, July 2016).

Political accountability to the Governor for all state agencies also shapes interagency relations. For example, we did not observe state agencies holding AAA accountable for maintaining water use efficiency. DRNA does not seem to impose consequences if the recommendations outlined in the Water Plan are not followed by fellow state agencies. As one interviewee put it, "How is it that the Governor would let one of his agencies put another agency head into jail? … the local agencies…work more on let's say a consensus basis, rather than a regulatory basis" (AAA consultant, July 2016). As a result of the hierarchical structure of agencies, agreements are reached without legal consequences. Though this informal enforcement leads to information sharing and a higher awareness of issues across agencies, limited interagency regulation has had unforeseen consequences. One interviewee attributes the reduced economic state of the island to this conflict:

> "… if [they] speak up then they don't have jobs...Everybody's on the take. So if you are on the take, and you speak up, then you're not going to get contacts. You're out of a job. [So] people don't say anything…everybody's just trying to do the dance. And slowly they take the whole country down" (federal employee, June 2016).

5. Discussion

Our *de jure* analysis revealed that equitable allocation and water use efficiency are water management targets enshrined by law in Puerto Rico; however, we found mismatches between what is written by law and what occurs in practice. Identifying accountability improved our understanding of environmental flow governance in Puerto Rico. Through our accountability analysis, it seems state agency accountabilities contribute to this misalignment. Agencies including DRNA, AAA, and AEE are not held accountable for enforcing their responsibilities related to water quantity management; however, there is evidence that consequences exist if state agencies do not fulfill their political responsibilities, potentially interfering with effective water management.

Our analysis highlights conflicting responsibilities legally assigned to DRNA, as the agency is responsible for maintaining a minimum flow to sustain ecological functions but also the prioritization of domestic human water needs. Prioritizing domestic users likely restricts DRNA’s effectiveness in maintaining a minimum flow, especially if multiple sectors require access to limited water resources (i.e. during drought conditions). This may be why most instream flow protection policies don’t include equitable allocation as an objective. In Arizona and Nebraska, for example, water for the environment is stated as important but comes second to domestic and agricultural water demand (Annear et al. 2004: 73). Although domestic water demand has been prioritized over environmental flows in the past in Puerto Rico (see Christian et al. 2019 for evidence of all discharge removed from streams), these competing mandates need to be resolved for effective environmental flow governance.

Historically, prioritizing human water demand over minimum flows incurred short-term ecological effects as stream systems seem to recover quickly from periodic low flows (Covich et al. 2003). However, long-term low or no flow events, such as when a large dam with no spillway is built, dramatically alter stream ecosystem functions (Holmquist et al. 1998, Greathouse et al. 2006). As climate change is expected to intensify water scarcity events in the region (Jennings et al. 2014), reductions in streamflow due to water withdrawals can be further magnified. The conflicting responsibilities of DRNA will likely need to be resolved to avoid the negative environmental impacts of reduced streamflow, especially in relation to diluting treated waste effluent to stream systems (Water Plan 2008).

The lack of consistent environmental flow enforcement by state agencies reflects what occurs in other states. Court rulings illustrate how citizen groups have sued agencies or water permit holders for environmental flows maintenance outlined by legislation. Groups in Washington state and California have argued for the necessity of environmental flow management using the public trust doctrine, forcing state governments to consider how private withdrawals will impact property held in trust (S.C.W. No. 87672-0; Annear et al. 2004: 64). As the water resources belong to the people and are entrusted to the Commonwealth for protection, the same legal argument could be made in Puerto Rico under the public trust doctrine. Legal recourses exist for Puerto Rican citizens to hold state agencies accountable for maintaining environmental flows, which could rectify the mismatch between *de jure* and *de facto* environmental flow governance.

An additional mismatch exists between AAA's legal autonomy to set domestic water rates and its inability to do so in practice. The reliance of AAA on the Governor's approval to increase rates is in opposition to our *de jure* analysis, which revealed that both the federal court and Puerto Rican Supreme Court view AAA as independent of the Commonwealth government. AAA's lack of autonomy over funding has likely contributed to its $4.2 billion debt (Slavin August 12, 2019). AAA's limited financial capital hampers the agency's ability to fulfill water management targets, such as repairing infrastructure in a timely manner to reduce leaks (Shultz August 23, 2017). This mismatch highlights the complex relationship between public corporations and the Commonwealth government, and how this relationship can undermine the agencies' abilities to fulfill environmental flow responsibilities.

Although our *de jure* analysis highlights existing legislation for the elements of our framework (equitable allocation and water use efficiency), what is outlined in the legislation does not consistently occur in practice. The Instream Flow Council cautioned that legislation may not be sufficient to ensure instream flows (Annear et al. 2004: 61), as there are additional challenges beyond creating statutes. Studies evaluating environmental flow implementation in countries world-wide have identified challenges such as limited political will, lack of funding, and institutional barriers (Le Quesne et al. 2010). State agencies in the southeastern U.S. also struggle with maintaining streamflow due to a lack of enforceable regulations, such as the minimum flow only existing as a recommendation or the flow amount not clearly stated (Baer and Ingle 2016). Funding limitations can also be a barrier to meeting water management goals (Baer and Ingle 2016).

Our analysis illustrates how agency accountability creates an additional challenge for state water management agencies trying to carry out legislative directives. In Puerto Rico, political accountability manifests through the Governor's appointment of agency heads and subsequent ability to remove the leaders at will. Accountability to the Governor interferes with state agencies' abilities to enforce regulations on each other (e.g. DRNA fining AAA for violating their permit's stated minimum flow requirement). State agencies tasked with water management targets instead fulfill political responsibilities. These competing lines of accountability are represented in Figure 4.2.

Actors (i.e. agencies) are known to be accountable to simultaneous forums, complicating the evaluation of an actor's motivations (Cedón 2000), and multiple accountabilities can lead to the ineffectiveness of actors (Koppell 2005, Brandsma and Schillemans 2013). In Puerto Rico, accountability to the Governor seems to limit the ability of state agencies to carry out their legally assigned responsibilities (i.e. raising water rates to increase water use efficiency). Politics impacting water governance is not endemic to Puerto Rico; e.g. politics drive water management decisions in the U.S. state of Georgia as well (Jensen-Ryan 2017). Though federal agencies are not accountable to the Governor and could therefore enforce regulations that may be counter to state political will, there are limited mechanisms allowing for federal agencies to engage in environmental flow governance.

Ecologists have recognized the importance of streamflow in Puerto Rican streams to maintain aquatic biota (Benstead et al. 1999, Greathouse et al. 2006). Indeed, research shows that prolonged minimum flows (i.e. below the flow exceeded 50% of the time) could be more detrimental to migratory animals than infrequent low flow periods (Scatena and Johnson 2001). This suggests that maintaining a minimum streamflow is not comprehensive enough to sustain migratory populations. If current streamflow regulations such as the Q99 cannot be effectively enforced on the island, it seems likely that any future regulations extending legal protection beyond a minimum flow will also be challenging to maintain.

Traditionally, ecologists focus on identifying the amount of flow necessary to maintain a functioning stream ecosystem (Arthington et al. 2010) and making recommendations useful to policy makers (Pahl-Wostl et al. 2013). However, the issue of whether ecological flow maintenance is occurring when protective legislation is in place is beginning to be addressed (Baumgardner 2019). Opportunities abound to understand whether the mandated flows are sufficient to maintain ecosystem functioning (Poff et al. 2010) and to identify the factors that make enforcement effective. Our identification of accountability for state agencies represents a novel approach to exploring limitations in effective environmental flow management, as accountability of agencies directly influences their actions (Ribot 2002). Though state agencies in Puerto Rico seem to meet the water governance targets (i.e. maintain minimum flows) when feasible, accountability for legislative directives may provide a necessary incentive to consistently enforce regulations. Though there are likely additional challenges to enforcing environmental flows, these findings advance the ongoing effort to determine how to effectively maintain environmental flows. We also contribute to the noted gap in identifying accountability relations within governance systems (Brandsma and Schillemens 2013). We can achieve a deeper understanding of how environmental flows are managed, facilitating ecologists' desire to make relevant policy recommendations (Arthington et al. 2006), by evaluating whether environmental flow management is assigned to an agency, determining agency structure, and examining inter-agency relationships.

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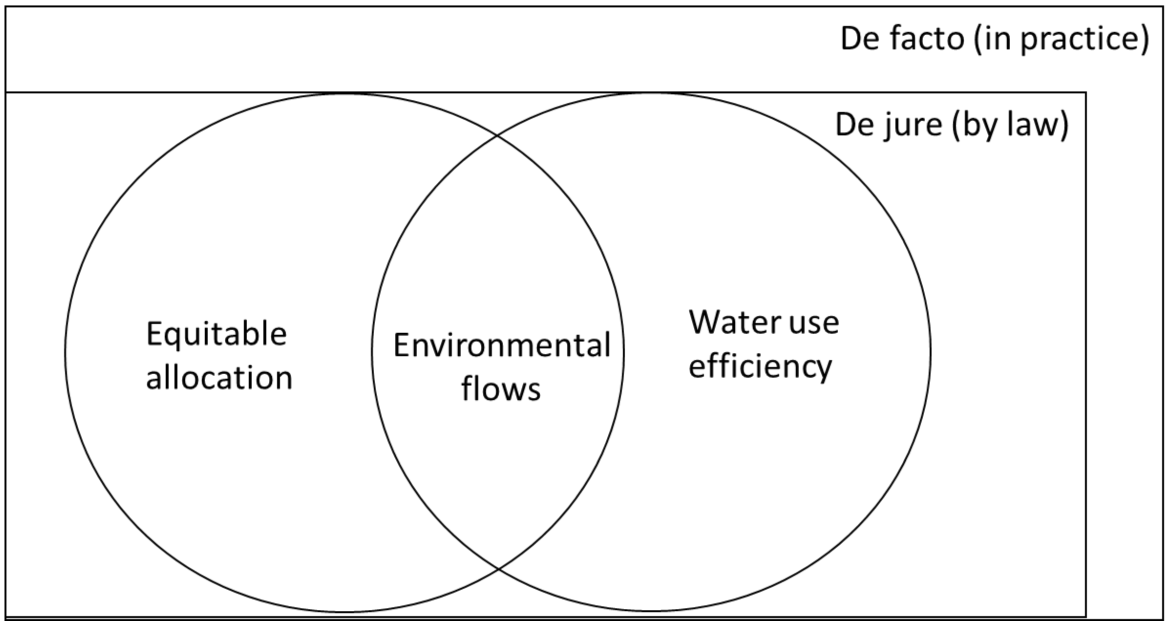

Figure 4.1. The framework we use to evaluate whether environmental flows governance exists in Puerto Rico. Two water governance targets (equitable allocation and water use efficiency) that exist in legislation and in practice indicate environmental flows can be effectively managed for by agencies.

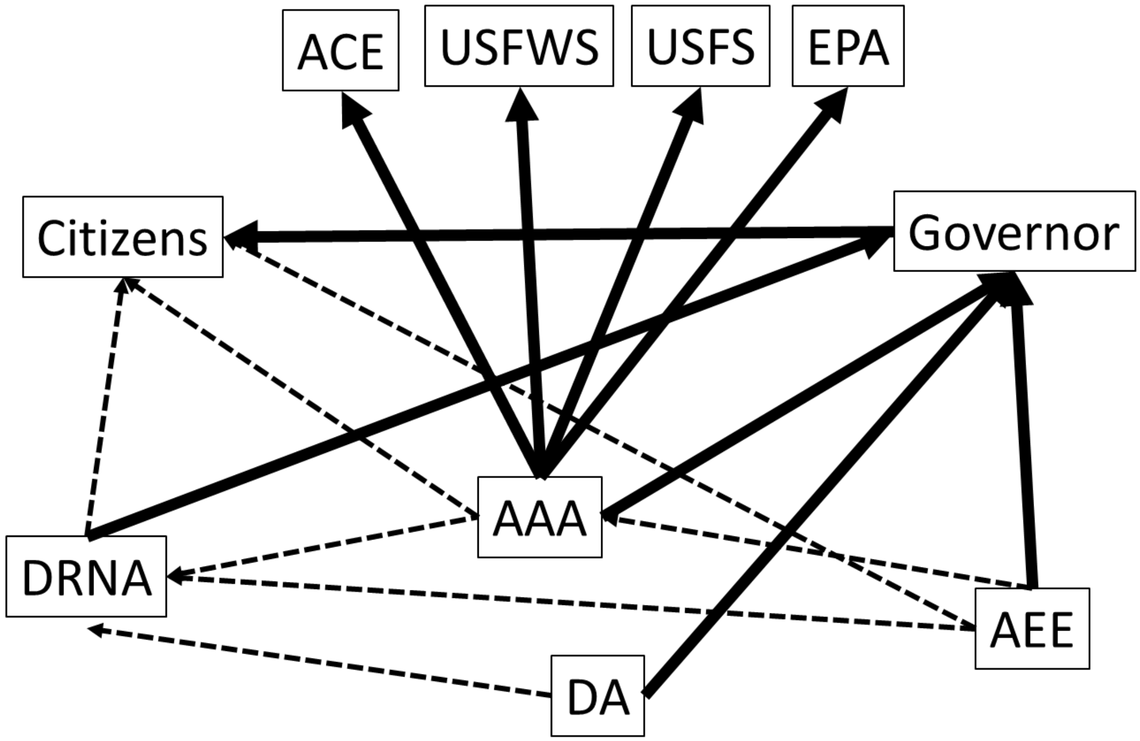

Figure 4.2. Accountability across state and federal agencies involved with Puerto Rican water management. The base of the arrow is the "who" and the arrow tip points to the "to whom" the agency is accountable. Solid lines represent interactions our analysis identified as having accountability by law and in practice, and dotted lines represent interactions that only have accountability in law. DRNA (Departmento de Recursos Naturales y Ambientales), DA (Departmento de Agricultura), AEE (Autoridad de Energía Eléctrica), AAA (Autoridad de Aqueductos y Alcalintarios), ACE (Army Corps of Engineers), USFWS (US Fish and Wildlife Service), USFS (US Forest Service), EPA (Environmental Protection Agency).

Table 4.1. Identification criteria used to determine whether two water governance targets, equitable allocation and water use efficiency, exist in legislation (*de jure*) and practice (*de facto*).

	De jure **Identification Criteria**	***De facto*** **Identification Criteria**
Equitable allocation	#1: Are water users or uses defined? #2: Are there statutes or regulations outlining how water gets apportioned? #3: Are there court cases providing clarity on how decisions on allocation are made? #4: What are the water permitting conditions?	#5: How is water permitted? #6: How is water allocated under drought conditions? #7: Are there priority uses or users?
Water use efficiency	#8: Does legislation encourage waste minimization? #9: Are consequences clear when a violation has occurred? #10: Can water management agencies be subject to legal recourse?	#12: Are there examples of water waste as outlined by the legislation? #13: Do agencies enforce consequences for water waste?

#11: Are there court cases providing clarity on efficiency?

CHAPTER 5

TRADEOFFS IN SOCIO-ECOLOGICAL RESILENCE OF WATER SYSTERMS IN PUERTO RICO[4]

[4]Chappell, Jessica, Elizabeth G. King, S. Kyle McKay, Laura German, Cathy Pringle. To be submitted to *Ecology and Society*.

Abstract

Freshwater resources are becoming increasingly variable in time and space, forcing difficult decisions regarding water allocation. Tradeoffs associated with these decisions affect social and ecological system processes and have the potential to impact the larger socio-ecological system (SES) resilience to extreme climatic events. However, we do not fully understand the baseline processes by which water-associated SES perform to maintain resilience. Here, we develop a baseline schematic of the water SES in Puerto Rico and explore how the social and ecological components interact to affect resilience to extreme climatic events. We then evaluate the historical context for implementing water supply management actions to mitigate drought effects for the social components. Finally, we explore how management actions have interacted with existing feedbacks to affect overall water SES resilience to droughts and hurricanes. Our analyses reveal that management actions targeting increasing water storage have resulted in decreased social resilience to drought and also lower ecological resilience. The findings suggest solutions focused on maintaining existing infrastructure (i.e. fixing leaks and reducing consumer water use) would allow ecological resilience processes to continue to function. Based on these analyses, we argue that engineering solutions should consider how to maintain overall SES resilience by evaluating the role of ecological systems and tightly coupled feedbacks between water infrastructure and ecosystem health. We argue that water supply managers should consider broadening their approach to increasing water supply resilience to include an evaluation of how proposed actions will interact with ecological resilience.

1. Introduction

The people of Puerto Rico depend on stream ecosystems for their drinking water supply, creating a tightly coupled socio-ecological system (SES) in which resilience of both the ecological and human infrastructure components depend on the availability and quality of surface water. Historically, surface water has been plentiful in Puerto Rico, flowing in more than 200 rivers and 500 named streams (Water Plan 2008), but highly variable in both time and space. Annual rainfall ranges from 3657-2801 mm, on the island's east to west side, respectively. More rain is received island-wide from May through November (127 to 201 mm per month) than December through April (66 to 126 mm per month) (Water Plan 2008). Human infrastructure and stream ecosystems are buffered against this variability by differing properties. In streams, migratory fishes and macroinvertebrates are able to reestablish populations following annual periods of water scarcity (Covich and McDowell 1996, McDowall 2009). Meanwhile, human communities have built dams to store water for municipal water supplies in times of scarcity, ensuring a constant supply of freshwater (Cashman 2010, Richter 2014). These behaviors of migration/recolonization by stream fauna and water storage by humans enhance the resilience of ecological and social systems to seasonal variation in water availability; however, it is unclear whether these "solutions" contribute, or in fact hamper, socio-ecological resilience to extreme climatic events that are less predictable in time, such as droughts and hurricanes.

Extreme climatic events (ECE), defined as "an episode or occurrence in which a statistically rare or unusual climatic period alters ecosystem structure and/or function well outside the bounds of what is considered typical or normal variability" (Smith 2011), have affected Puerto Rico over the past 25 years and include hurricanes and droughts.

These ECEs have altered social sectors, including economic output (GDP) (Lugo and Garcia-Martino 1996), the mental health of the island's citizens (Scaramutti et al. 2018), and the domestic water supply, which affects GDP and health. Specifically, the 2015 drought resulted in San Juan's residential and business sectors having restricted water access for over a month (Alvarez 2015), and residents across the island lost access to water for multiple months in the wake of Hurricane Maria in 2017 (Heredia Rodriguez 2018).

With climate change models suggesting that droughts and hurricanes will become more frequent in the future (Jennings 2014), water resource managers must identify management actions that will ensure a sustainable freshwater supply. However, management decisions have associated tradeoffs. Actions that enhance water supply resilience for human communities may lower resilience in stream ecosystems and trigger undesirable feedbacks. The history of multiple ECEs affecting the island, combined with the documented water management (social) responses and research on stream ecosystem (ecological) responses to disturbances, creates an ideal opportunity to evaluate the SES tradeoffs of management actions. Here, we ask: (1) Do feedbacks and interdependencies between water resource infrastructure and stream ecosystems lead to tradeoffs in social and ecological resilience? (2) How has historical water management affected overall SES resilience to droughts and hurricanes? and (3) Which water management actions hold promise for improving water supply and ecological resilience to both droughts and hurricanes?

The concept of resilience varies depending upon the disciplinary lens through which a system is assessed (Wang and Blackmore 2009). However, across disciplines, the concept can be bracketed by addressing two basic questions: resilience of what and resilience to what disturbance? (Carpenter 2001). Ecological resilience has been defined as “the ability of systems to absorb change of state variables and still persist” (Holling 1973), and Adger (2000) defines social resilience as “the ability of human communities to withstand external shocks to their social infrastructure.” In ecological, social, or coupled social-ecological systems, the definition of resilience is sometimes broadened to also include transformability, or the capacity of a system to reorganize into other configurations that maintain certain functions of interest, recognizing that an ecosystem may not return to its original state after a disturbance (Walker et al. 2004, Folke et al. 2010). A resilient SES can persist through a disturbance and can be characterized by its adaptability, which includes a capacity for learning and preparing for the next event (Folke 2006). Engineering resilience typically focuses on the ability to bounce back or the rate of recovery from disturbance (Wang and Blackmore 2009). A common technique to assess system resilience is through identifying proxy characteristics that generally confer resilience in complex systems, such as diversity or functional redundancy (Walker and Salt 2006, Liu 2014, Allen et al. 2018).

Here, we analyze water SES dynamics in Puerto Rico, including feedbacks between social and ecological components of water systems, in order to evaluate overarching SES resilience and to explore how management actions influence resilience. For our analysis, the water SES comprises water infrastructure as the social component, and stream ecosystems as the ecological component. Water infrastructure systems are “water supply, i.e., water resource systems and water distribution networks” (Shin et al. 2018). The stream ecosystem comprises those structures and functions that create, maintain, and inhabit stream channels, such as flow regime, aquatic species, and sediment flux (Palmer and Febria 2012). Thus, we can think of water SES resilience as "the ability of water systems to withstand a variety of water-related risks, as well as the ability of such systems to adapt or transform to new hydrologic regimes" (Rodina 2019). In Puerto Rico, a resilient water SES would have a water infrastructure system and stream ecosystems that can sustain their functioning during and after an ECE. Specifically, the water infrastructure system should show engineering resilience by being unlikely to fail and able to recover quickly (Wang and Blackmore 2009), and the stream ecosystem should show ecological resilience by having functions that persist and bounce back after a disturbance (Holling 2010). While recent studies have evaluated resilience to disturbance extremes in terrestrial ecosystems (Felton and Smith 2017) and infrastructure design (Curt and Tacnet 2018), the resilience of a tightly coupled water SES to contrasting extreme events and its implications for management actions is underexplored (but see Maxwell et al. 2018).

The overall objective of this paper is to understand the resilience of the water SES in Puerto Rico to hurricanes and droughts and evaluate how management actions alter system resilience. We assess SES resilience with three distinct analyses. First, we use a systems approach to identify state variables of the water SES, the relationships linking them, and feedbacks within and between SES components to construct a schematic network diagram of the water SES. Second, we explore the context of historical drought management decisions and actions, tracking the dynamics through the schematic of the water SES. Third, we assess water management actions and their relative benefits for water supply and ecological resilience to both droughts and hurricanes. Finally, we make recommendations of how to balance engineering solutions with ecological solutions to promote SES resiliency.

1.1 Study area

We focus our research on the island of Puerto Rico, a U.S. Commonwealth in the Caribbean Sea (population 3.2 million), and its capital city of San Juan (population 1.5 million). San Juan is located in the subtropical moist forest life zone, the dominant life zone on the island (Miller and Lugo 2009) and receives an annual average precipitation of 1675 mm (Daly et al. 2003). Over 70% of water extracted for human use in Puerto Rico originates from surface water (Water Plan 2009). Puerto Rico has 14 large reservoirs behind dams over 15 m in height (Water Plan 2009), resulting in one of the highest densities of large dams in the world (Greathouse et al. 2006). Municipal intakes on streams can also be coupled with small dams (less than 5 m tall) that create pools to submerge the intakes (Crook et al. 2009).

Although various agencies are responsible for water management on the island, the Puerto Rican Aqueduct and Sewer Authority (PRASA) plays a dominant role in domestic water supply management (AAA in Spanish: Chapter 4). Nearly all domestic water supply on the island is managed by this public corporation, though some communities have their own water supply (Jain et al. 2014). The Commonwealth government created PRASA in 1945, with the purpose of “providing and helping to provide for the citizens adequate drinking water, sanitary sewage service and any other service or facility proper or incidental thereto” (22 L.P.R.A. § 144). Though requiring PRASA to supply water to every citizen seems feasible given the small size of the island and abundant water resources, it has led to unforeseen consequences. Several densely populated communities have developed in the mountains, yet most water treatment plants are located along the coast. This misalignment in location of households and water treatment plants requires treated water to be pumped uphill to users using electricity. Additionally, over 60% of treated water does not reach paying customers, and regulators have identified leaky pipes as the dominant cause of this water loss (Water Plan 2008).

2. Methods

To explore how water management actions influence water SES resilience, we used three separate analytical approaches. In Analysis I, we used a qualitative systems modeling approach to describe how the SES functions in baseline, drought, and hurricane conditions. We constructed a baseline schematic of the water SES in Puerto Rico that specified state variables and stressors of interest. To identify ecological state variables within our system, we relied on peer-reviewed research focused on instream components. We selected the social state variables based on our understanding of how the water distribution system functions in Puerto Rico (Chapter 4). Our water SES schematic also included process relationships, which define the influences of variables on one another (Schlüter et al. 2014). We also elaborated on the timescale for SES functions to reestablish after ECE. Our systems approach allowed us to identify and understand feedbacks within the SES (Walker et al. 2002, Allison and Hobbs 2004), based on state variable interactions. Using our baseline water SES model and drawing on previous research, we evaluated how effects of drought and hurricanes may propagate through the system. As each drought and hurricane is unique and may not have the same characteristics, we conducted our evaluation with knowledge regarding the extreme droughts of 1994 and 2015, and hurricanes Irma and Maria in 2017.

In Analysis II, we used historical event analysis to describe five water management actions, and why they were implemented to increase social resilience to drought. In Analysis III, we explored how all five management actions have affected the social and ecological components and, ultimately, system resilience to drought and hurricanes. We also mapped the four management actions that most impacted resilience onto the baseline water SES model as a qualitative systems analysis to illustrate the SES feedbacks triggered by the water management actions.

For each of the three analyses, we relied on combined information from four sources: peer-reviewed literature, news reports, Puerto Rican legislation, and semi-structured interviews. We identified peer-reviewed literature by focusing on scientific publications produced by the Luquillo Long Term Ecological Research (LTER) Program, with an emphasis on studies that evaluate the impact of disturbances on stream ecosystems. Although this research was focused in El Yunque National Forest (EYNF), ecologists consider the system structure and functions to be representative of headwater streams island-wide prior to major human disturbance (Brokaw et al. 2012: 326). News reports came from a variety of print and online media sources (English and Spanish language) on the island (El Nuevo Día, Primera Hora) and from the U.S. mainland (New York Times), with a focus on articles describing the condition of the water distribution system (i.e. rationing, ruptured pipes, etc.). We identified Puerto Rican legislation through Westlaw (Thomson Reuters Corporation 2017). We focused on legislation related to water management, as we had prior knowledge that management actions occasionally stemmed from legally assigned responsibilities (Chapter 4). Semi-structured interviews were conducted in Puerto Rico from March to August 2016 and targeted individuals from key state and federal agencies involved in regulating or managing water on the island. These agencies included: PRASA, the Department of Environmental and Natural Resources (DNER), Puerto Rico Energy and Power Association (PREPA), Environmental Quality Board (EQB), Department of Agriculture (DA), US Forest Service (USFS), Army Corps of Engineers (ACE), and US Fish and Wildlife Service (USFWS). Interviews were semi-structured with a list of pre-determined questions asked of each interviewee. The final sample consisted of ten individuals who were involved with state agencies, and five individuals involved with federal agencies.

3. Results

3.1 Analysis I: SES components and ECE effect propagation

This analysis sought to describe how the SES functions in baseline, drought, and hurricane conditions using schematics of the water SES as a conceptual systems model. In the baseline water SES model (Figure 5.1a), precipitation amounts determine stream flow in Puerto Rican streams (link 1), as groundwater contributions are minimal (Crook et al. 2007). Streamflow is a major driver of the water SES, as both ecological and social components depend on water quantity to function. Connectivity is directly related to stream flow (link 2) and influences the movement of abiotic and biotic components (Pringle 2003). Although connectivity exists in four dimensions (Ward 1989), the focal dimensions in Puerto Rico are longitudinal and temporal (Crook et al. 2009). Longitudinal connectivity refers to the connection between stream headwaters and the ocean, while temporal connectivity refers to connectivity through time as mediated by variation in streamflow. Longitudinal connectivity affects both sediment accumulation (link 3) and migratory aquatic animal dynamics (link 4). When connectivity is high, sediment accumulation is low, as sediment is transported downstream to the ocean during high flow events. Additionally, because migratory stream animals depend on connectivity to complete their life cycle (Benstead et al. 1999, March et al. 2001), animal population dynamics are conferred a high degree of stability when connectivity is continuously high.

Freshwater shrimp are a key migratory animal in Puerto Rican streams, as they provide a multitude of ecosystem services. When shrimp are present in streams, leaf litter and algae are maintained at low levels (link 5). Some shrimp taxa are known to increase leaf litter breakdown in the absence of predators (March et al. 2001) and reduce algal cover and diversity when allowed to reach high densities (Pringle 1996). Additionally, filter-feeding shrimp remove sediment from the water column (Crowl et al. 2001) and scraper shrimp remove accumulated sediment from the stream bed (link 6, Pringle et al. 1999), reducing sediment accumulation and increasing water quality (link 7, Greathouse et al. 2006). Large wind events lower the amount of canopy cover over streams, resulting in an increase in light reaching the stream bed (link 8). The leaves and trees lost from the canopy enter streams as leaf litter (link 9, Covich et al. 1991). Higher levels of leaf litter and algae can contribute to decreased water quality (i.e. water clarity; link 10). Thus, freshwater shrimp can mitigate the effects of increased leaf litter or algal biomass when found at high densities through detrital processing and algal grazing.

Shifting to the social components, streamflow decreases in the presence of barriers, which are typically dams, and intakes (link 11). These barriers and associated reservoirs allow for increased water storage amount (link 12), which can increase stored water in response to precipitation (link 13). Intake volume is proportional to the amount of stored water (i.e. storage amount increases, volume increases; link 14). However, water intake volume also experiences system loss (i.e. leaks from pipes; link 15) in the transfer to the water treatment center.

As intake volume increases, treated water increases as well (link 16). Additionally, higher water quality will result in less expensive treated water (link 17), as it will take less time and money to perform the water treatment process. Treating water also relies on electricity to complete the process (link 18). Treated water is then available to be transferred to users (link 19), but water managers estimate that 60% of the treated water is non-revenue generating, meaning it is lost either through leaky infrastructure on its way to domestic consumers or is stolen by domestic users (link 20; Water Plan 2008). The water remaining in the pipe distribution system is delivered to domestic users at high and low elevations (links 21 and 22), to meet user demand (links 23 and 24). However, users at high elevations must have treated water pumped uphill, requiring a functional electrical system (link 25, federal employee interview, June 2016). The Puerto Rican electrical system can be negatively affected by wind (link 26), as hurricane-force winds can interrupt the power transmission system. Finally, municipal water intake volume increases with user demand (link 27) as more water must be withdrawn and treated to meet user needs.

Figure 5.1b highlights the effects of drought on the water SES, beginning with reduced precipitation and thus streamflow. In EYNF, migratory aquatic animal dynamics are sensitive to small changes in streamflow and connectivity, and seemingly slight decreases in connectivity can impact migratory animal dynamics (Benstead et al. 1999, Crook et al. 2009, Chappell et al. 2019). At the height of the 2015 drought, streamflow was reduced by over 50%, causing isolated pools to form (Gutiérrez-Fonseca et al. 2019), restricting freshwater shrimp access to food resources. Shrimp of two genera, *Atya* and *Xiphocaris*, decreased reproductive output as they congregated in high densities in headwater streams, while *Macrobrachium*, which prey on the other shrimp genera, tend to have low densities at high elevation sites during drought (Covich et al. 2003, 2006). Although connectivity limits shrimp movement, other factors such as food resources (Covich et al. 1996) and predator avoidance (Covich et al. 2009) can also influence shrimp distribution.

Other stream responses to drought include increases in leaf litter falling into the stream; however, during the 2015 drought, increased litterfall did not result in an observable increase in organic matter likely due to a combination of biotic and abiotic factors (Gutiérrez-Fonseca et al. 2019). Lowered water quality can result from high nutrient levels and low dissolved oxygen in isolated pools (Gutiérrez-Fonseca et al. 2019), which has the potential to affect water available to domestic users. Droughts also reduce water storage quantity, although the existing stored supplies can delay municipal water users from experiencing immediate water shortages.

When precipitation resumes, drought effects can dissipate in a matter of days in the social system (i.e. water rationing) but linger in the ecological system, as domestic water needs must be satisfied first before any other user (12 L.P.R.A. § 1115n). This indicates streamflow will be used for domestic consumption before satisfying stream ecosystem requirements. The persistence of low flow drought conditions delays the recovery of the ecological system in comparison to the social system.

The impacts of hurricanes are mapped in Figure 5.1c. Hurricanes, such as Hurricane Maria, can result in extremely high winds and precipitation; their main impacts are highlighted in green and blue, respectively, in Figure 5.1c. Elevated wind speeds lead to high leaf litter inputs, and high stream discharge moves leaf litter downstream quickly and scours algae from the streambed (Covich et al. 1991). High precipitation increases erosion and leads to elevated sediment accumulation in streams, reducing water quality; however, high streamflow and connectivity move the sediment to the ocean. High wind speeds can also cut off electricity.

While hurricanes can result in many immediate ecological effects (i.e. increased leaf litter and streamflow), most are temporary as the system re-sets itself within a matter of months (Covich et al. 1991). Hurricane winds can drastically reduce canopy cover, but increased light penetration facilitates algal growth (Covich et al. 1991) and canopy cover regrows if winds are not too extreme (Amandolare 9/11/2018). Higher precipitation associated with some hurricanes can elevate connectivity and allow for enhanced freshwater shrimp movement (Covich et al. 1991). Meanwhile, the water supply system may experience a disruption in meeting user demand for months, as a result of high sediment accumulation or electrical system failures.

3.2 Analysis II: Context of historical management actions

We used historical analysis to understand why five different water management actions were undertaken with the intent of increasing social water resilience. The management strategies (actions) we focused on include water redistribution (interbasin transfers), water storage (creating reservoirs), weather modification (cloud seeding), water conservation (reducing consumer water use), and water use efficiency (fixing infrastructure leaks). These actions have been implemented at various spatial scales in Puerto Rico (i.e., locally or island-wide). We describe the historical social context of why these particular management actions were selected.

Water redistribution

Interbasin transfers involve moving water to domestic users in another watershed. Water can either be transferred to another reservoir in the new basin or to a water treatment center. Interbasin transfers were proposed as a solution for infrastructure drought resilience in Puerto Rico as early as the 1950s, when the US Army Corps of Engineers developed a plan to connect all the reservoirs around the island. As drought periods typically affect only one area of the island at a time, interbasin transfers were seen as a viable way to increase domestic water supply resilience. All the reservoirs were never connected, in part due to the lack of demonstrated necessity (former federal employee May 28, 2016).

In response to the drought-induced water rationing, the Legislature "ordered" PRASA to construct part of the plan conceptualized by the US Army Corps: the North Coast Superaqueduct (22 L.P.R.A. § 451), an interbasin water transfer of 100 MGD (million gallons a day). The transfer required a large pipe to be installed, moving water from Dos Bocas Reservoir (13,200 acre feet; Water Plan 2008), 42 miles east to the San

Juan metropolitan area. The legislation demanded immediate approval by the Regulation and Permit Administration once the application was received (22 L.P.R.A. § 452), meaning the necessary environmental assessments were not completed (former PRASA consultant June 7, 2016). The project was rushed, as “the Legislature understands that in view of this crisis, the real, speedy and efficient alternative is the development of the North Coast Superaqueduct Project, which shall resolve the water supply problems of the North-central zone until the year 2050. This means that not only present deficiencies shall be resolved, but also those of the future” (Statement of motives: June 12, 1997, No. 19.).

Though legislators argued the Superaqueduct was necessary to ensure the people of Puerto Rico have access to water (22 L.P.R.A. § 1115a), its construction was a controversial management action seemingly driven by political motives and triggered a backlash from concerned citizens (PRASA consultant June 7, 2016). The Superaqueduct legislation was later ruled unconstitutional (146 D.P.R.64), and two politicians went to jail for their involvement in financially corrupt actions surrounding its construction (Quintero January 8, 2014). Nevertheless, the Superaqueduct continues to function as an interbasin transfer.

Water Storage

Creating additional water storage is a common management action to increase water infrastructure resilience (Richter 2014, Veldkamp et al. 2018). After the 1994 drought, existing river intakes were upgraded to withdraw more water and new instream barriers and intakes were added (Water Plan 2008). Not all proposed reservoirs were built, however. A reservoir was proposed for the Mameyes River, federally recognized as a Wild and Scenic River and one of the last free-flowing rivers originating in EYNF. The location for this new reservoir had been identified by surveys conducted in prior decades by engineers, but more recent land developments made the location obsolete; however, PRASA still pushed for a new reservoir at the old site, ultimately constructing a French drain with an off-channel reservoir that was abandoned within ten years of installation (federal employee; June 8, 2016). After the 2015 drought, a reservoir on the Mameyes River was again suggested (Misla July 16, 2015), and again not implemented. The difficulty of abandoning old, obsolete reservoir plans is not uncommon in water management (Reisner 1986).

Weather modification

At the onset of the 2015 drought, the Governor declared a state of emergency through an Executive Order (Boletin Administrativo Num OE-2015-011), setting into motion the government response to drought, including weekly meetings of drought-related committees. Though there are multiple committees and each plays a unique role in managing the island's freshwater resources for drought (Protocolo 2014), the Scientific Committee provides the scientific and technical assessment of recommended management actions and is composed of federal and state agencies that have access to relevant data. One management recommendation presented by PRASA to the Executive Committee was cloud seeding. The cloud seeding involved injecting calcium chloride and silver particles into clouds above three watersheds that contained the reservoirs with the lowest water levels (El Nuevo Día August 5, 2015). The Scientific Committee discussed multiple reasons why this option to increase water supply was not ideal, including lack of scientific evidence that cloud seeding induces rainfall, the mismatch in watershed scale between the island and those where cloud seeding is typically used (i.e. Texas, New

Mexico), and the limited financial resources available to mitigate drought effects. However, the Governor approved the recommendation, and PRASA paid over $500,000 for three months of cloud seeding (El Nuevo Día August 5, 2015).

Water Conservation

During both the 1994 and 2015 droughts, water conservation plans were implemented at multiple scales. The 1994 drought caused much more severe water rationing compared to previous droughts (Larsen 2000), triggering over two months of water rationing for more than 1.5 million people living in the San Juan area (Navarro July 3, 1994).

The 2015 drought caused water rationing that affected over 1.5 million people living in the San Juan area beginning in June (Primera Hora June 8, 2015). In August 2015, rationing became more severe for 140,000 clients who had water access for one day out of every four days (Rosario August 6, 2015), representing "the strictest rationing we've ever had" (Alberto M. Lazaro, president of PRASA in Alvarez July 14, 2015). Schools were forced to shut down during days of water rationing when there was no running water, limiting breakfast access for the 30,000 student participants in the *School Eaters Assistance program* (Bauzá August 3, 2015).

In 2015, PRASA shut off the domestic water supply flowing in underground pipes at certain intervals (i.e. water on for 24 hours, off for 36 hours). Additionally, fines were issued within the domestic sector for using water in ways considered "wasteful", such as washing a car or the driveway (Alvarez July 14, 2015).

Water Use Efficiency

Water loss through leaky pipes is an issue that many interviewees considered to be a major water management challenge on the island. However, there is no straightforward solution to the problem given existing infrastructure and maintenance limitations. The legislation requiring PRASA to supply water to every Puerto Rican citizen centralized water governance on the island, restricting the potential for community level management that could adapt to local conditions (Ribot 2004). PRASA's water treatment centers are typically located on flatter coastal lands, yet many domestic water users live in the mountains. This difference in location requires more pipes and electricity to pump water to high elevation users and has resulted in a sprawling water infrastructure system. Additional pipes increase the likelihood of leaks, especially if maintenance is not prioritized. One interviewee summed up the issue as

"So…they got water to every Puerto Rican, wherever they might be, no matter the cost, pumping water up the hill… they did it! Just like the law says. But in the process they created a system that is leaking all over the place, that is unsafe…unreliable…" (federal employee; July 19, 2016).

Leaks seems to be an inherent characteristic of the pipe infrastructure used to move water from treatment centers to households.

3.3 Analysis III: Assessing water management actions

Here, we evaluate how the five management actions discussed in analysis II elevate or reduce the resilience of the social and ecological components to ECE based on previous implementation within the system. We evaluate the effects creating a reservoir and interbasin transfers simultaneously, as the actions are inherently intertwined: there can be no large interbasin transfers without the existence of a reservoir. Mapping the water management actions onto the baseline water SES schematic (Figure 5.2), we saw the actions enter the system in four main spots: (1) intake volume, (2) water storage, (3) precipitation, and (4) available water. Thus, we considered reducing consumer water use and fixing infrastructure leaks jointly, as both actions focus on increasing the "available water" component of the SES. Cloud seeding is evaluated separately, as it is the only management action to target the precipitation component.

Creating reservoirs and interbasin transfers

Although creating reservoirs and adding interbasin transfers should increase social resilience to drought, these actions seem to be ineffective at times. Though the Superaqueduct and additional reservoirs were built in the late 1990s, water rationing was again implemented when the next major drought hit Puerto Rico. Even though the Superaqueduct was fast tracked in response to social pressure created by rationing, it did not alleviate the need for rationing. In fact, rationing increased after its implementation. However, since the early 2000s, there have been fewer new reservoir proposals (federal employee, July 15, 2016). This is probably the result of a combination of factors, including PRASA's lack of funding (state agency employee July 12, 2016), the recognition of cheaper management options that are more effective in maintaining water supply (e.g.

fixing infrastructure; multiple interviews), and the fact that adding intakes doesn't seem to help increase water infrastructure resilience.

This lack of increase in social resilience to drought could arise from unintended ecological consequences, such as reduced connectivity though streamflow blockage (Figure 5.2). Decreased connectivity results in sediment accrual behind dams. As Puerto Rico has high rates of sedimentation stemming from land use and large rainfall events, many of the island's large reservoirs have high storage loss rates. For example, the Loiza reservoir, a main water source for San Juan, is estimated to lose 2.57% of its storage capacity per year due to sedimentation (Soler-Lopez and Gomez-Gomez 2004). Sedimentation is of particular concern after a hurricane (Figure 5.2), as storms account for high sedimentation rates (Gellis 1993). For example, Hurricane Hortense (1996) and Hurricane George (1998) resulted in multiple reservoirs across Puerto Rico losing over 10% of their original storage capacity due to sediment loads (Soler-Lopez 2001). High sedimentation rates also plague the pipe system used to transfer water, sometimes completely clogging pipes with sediment and making water transfers unfeasible (drought committee 2015). Though water storage is intended to increase social resilience to drought, the reduction in reservoir volume during a hurricane may lower drought resilience (Figure 5.2).

Sedimentation can also affect water intakes on relatively small streams. For example, the instream French drain intake that was installed in 1998 on the Rio Mameyes in an attempt to maintain riverine connectivity (March et al. 2003) was abandoned after 10 years due to sedimentation (federal employee; July 19, 2016). After Hurricane Maria, a small town was without water for six days as sediment choked the water intake on the

Espiritu Santo River. Even once the intake was cleared, the tap water was undrinkable as the sediment impaired water quality (Beeler 11/9/2018). Although creating reservoirs increases water storage (positive social resilience), the associated dams negatively effects stream functions (negative ecological resilience), including water quality (negative ecological and social resilience) (Table 1). Dams also increase sediment accumulation, meaning that water storage volume is lost and the reservoirs provide less water during a drought (negative social resilience).

Additionally, declines in connectivity reduce accessible habitat for migratory aquatic animals (Crook et al. 2009, Cooney and Kwak 2013). Given the sensitivity of the aquatic community to changes in streamflow, researchers suggest continuous low flows caused by dams can be more harmful to the stream ecosystem than short term complete loses of connectivity (Scatena and Johnson 2001). However, if zero connectivity is maintained for an extended time period (years), migratory animal populations will eventually be extirpated (Holmquist et al. 1998), causing additional ecological effects. For example, complete extirpation of shrimp populations increases algal biomass by an order of magnitude and elevates nutrients in high-gradient streams (Greathouse et al. 2006), which has implications for water quality. Given the lack of functional redundancy, the loss of the migratory animal community represents a decrease in stream ecosystem resilience.

As interbasin transfers and creating reservoirs increase intake volume and water storage, respectively, any increases made in either of these are susceptible to system loss (Figure 5.2). In fact, more water transfers involving more pipes may inflate the leakage rate as the water distribution system is already susceptible to leaks (negative social resilience). As framed by one ex-federal employee, “The combination…of dredging and the Superaqueduct was supposed to solve the metro problems for the next 100 years…[PRASA] didn’t do anything about water losses that went up from 30% to 65%” (interview with retired federal employee; May 28, 2016).

Cloud seeding

A scientific study, which the Scientific Committee demanded, found that none of the cloud seeding attempts measurably increased precipitation (León June 11, 2016). This suggests there was no effect on the social or ecological resilience relating to our SES model (Table 5.1), and thus we did not trace the effects through the system. However, there are tradeoffs for choosing this water management action. By using PRASA funds to finance this action, it could limit PRASA’s ability to carry out other water management actions (e.g. fix leaks). Negative ecological effects are also possible in terms of inhibiting plant growth but seem to vary with plant species and silver concentration (Ouzounidou and Constantinidou 1999).

Reduce consumer water use and fix infrastructure leaks

The main method historically implemented to reduce consumer water use has been water rationing, though its effectiveness at maintaining the water supply during the drought is difficult to assess. During rationing periods when water was turned on, domestic water users filled pots, water barrels, trashcans, and cisterns in preparation for when their water would be restricted again (Alvarez July 14, 2015), suggesting rationing may have exacerbated water consumption. Additionally, turning the water off and on altered pressure within the underground water system, resulting in multiple pipe ruptures and increased water loss (PRASA consultant July 12, 2016). These examples suggest water rationing did not increase social resilience to drought. Although encouraging water conservation by mandating fines for water waste may have been an effective method for maintaining the water supply, these actions have not been implemented at timescales beyond when the island is experiencing a drought.

Increasing water use efficiency is one of the most common solutions interviewees suggested to enhance social resilience, perhaps because leaks have a disproportionately negative effect on drought resilience. During droughts, high leakage rate means that PRASA must withdraw nearly double the water amount needed to meet domestic demand (Figure 5.3). If leaks were reduced, the reservoir yield would be more sustainable (PRASA consultant; July 12, 2016). During the 2015 drought:

> "… [PRASA was] withdrawing 625 MGD from all of the sources. And they were delivering 300 MGD. Because it was losing 50% to leaks…but if PRASA were to reduce those losses to 30%...PRASA would have been able to manage the drought much more effectively with some rationing but not as intense..." (retired government employee; May 28, 2016).

Increasing water use efficiency by fixing infrastructure leaks is a major water management challenge on the island for multiple reasons. PRASA has recognized the importance of this action and is attempting to identify leaky pipe locations in San Juan (Arguinzoni August 5, 2015), but financial constraints restrict PRASA's ability to perform upkeep and maintenance of the existing system (former PRASA employee; July 12, 2016).

Leaky infrastructure could also be reduced if treatment facilities were in closer proximity to users (PRASA consultant; July 7, 2016). An additional benefit to this change in systems operation would be a reduction in the coupling of water and energy systems in Puerto Rico. In hurricanes, the misalignment between the location of water treatment centers and households results in mountain residents losing water services when the electricity is also cut. Decades of improper maintenance and financial limitations suggest this will be an ongoing issue for years to come, although PREPA, the electric utility company, is aware of the electrical system's vulnerability to high winds and is working on improvements (McKenna March 28, 2019).

Regardless of the mechanism, high leakage rates must be addressed, as they likely undermine the effectiveness of other water management actions. Engineers have suggested supplementing San Juan's Superaqueduct water supply with water stored in aquifers on the north coast via coastal wells (Arguinzoni August 5, 2015). Using two water supply sites would create functional redundancy within the water supply system and holds promise as a mechanism to increase social resilience to drought if leaks were reduced. Additionally, fixing leaky infrastructure has no negative effect on ecological resilience (Table 5.1), allowing the ecosystem to self-regulate its response to ECE.

4. Discussion

Our three analyses reveal the structure, interaction, and feedbacks in the tightly coupled water SES. Specifically, social and ecological resilience seem to be driven by the quantity and quality of waters and the dynamics of migratory aquatic animals. Our historical analyses indicated water management actions that were taken to increase infrastructure resilience to drought that targeted water storage or precipitation. Additionally, examples illustrated how management actions aimed at increasing social resilience to one ECE may reduce overall system resilience when the ecological response to different ECEs is considered. The combination of our three analyses illustrate how the water SES responds to ECEs on opposite sides of the spectrum and the resulting feedbacks when resilience management actions are implemented.

The examination of baseline water SES response to both hurricanes and droughts revealed two key findings. The first is the mismatch in drought conditions experienced/perceived by the ecological versus social system components. Droughts can exert an immediate effect on stream ecosystem, whereas the social component is buffered from experiencing initial impacts. This is partially an artifact of drought identified through ecological impacts such as declines in precipitation and streamflow (Larsen 2000), but also because water storage has been historically effective in ensuring short-term infrastructure resilience to drought. Puerto Rico experiences an annual dry season (Weaver et al. 2012), yet domestic water rationing occurs at a less frequent rate. A tradeoff of this effectiveness in increasing infrastructure resilience, however, is that the stream ecosystem experiences drought-like conditions (i.e. reduced water quantity) more frequently and for extended periods (Crook et al. 2009) compared to the infrastructure system. While Puerto Rican stream ecosystems historically recover quickly from brief droughts (i.e. weeks to months), altered streamflow for prolonged periods (i.e. years to decades) can have negative ecological consequences, including a loss of ecosystem function (Holmquist et al. 1998, Greathouse et al. 2006).

The social system seems to have less resilience to hurricanes than the ecological system in the baseline SES. Stream ecosystems in Puerto Rico have evolved to accommodate large variations in streamflow. High precipitation events occur annually in the wet season with named storms (i.e. Georges, Maria), contributing over 100 mm of rainfall, occurring roughly once every ten years since the late 1970s (Brokaw et al. 2012: 173); however, this frequency has increased since 2000. Previous research suggests that the frequency of streamflow alteration within tropical streams is the driving selection force for the migratory life history of stream animals (McDowall 2009). Meanwhile, the water infrastructure system has not adapted to these same conditions. The baseline water SES reveals that the factor limiting water distribution after a hurricane is the reliance on electricity to purify and deliver water to domestic customers. This dependence of the water infrastructure system on the electrical system makes water delivery vulnerable to any disturbance the electrical system may experience (Little 2002). As the electrical system is powered by imported carbon fuels (Mckenna; March 28, 2019), the system is vulnerable to multiple stressors. The vulnerability of the electrical system must be addressed through management actions to enhance water infrastructure system resilience to hurricanes.

Several water management actions enacted in Puerto Rico have targeted increasing water quantity and were reactive policies, as they attempted to reset the water supply system to a desirable state (Allison and Hobbs 2004). However, these actions either had no effect on drought resilience (cloud seeding) or were effective for short-term dry periods (water storage and interbasin transfers). The focus on augmenting water quantity as opposed to promoting better water use is a common management approach, as active strategies requiring spending money on a highly visible project seem to be more politically salient (Richter 2014, Reisner 1986). The other management actions (conservation and efficiency) aim to maintain the water present in the social system components and have the potential to increase water SES resilience. Water conservation would be more effective if it were encouraged year-round, especially before drought conditions take root, and avoided water rationing. Conservation actions could expand to include the promotion of low flow water faucets or water bill social comparisons (Ferraro et al. 2011). Management actions that use existing water storage efficiently without withdrawing more from the ecosystem reduce SES resilience tradeoffs.

Tradeoffs are inherent within management decision-making and exist within and across the social and ecological systems. In the social system, tradeoffs occur between community groups (Duit et al. 2010) or infrastructure development (Curt and Tacnet 2018), and within ecological system management, such as maximizing one ecosystem service while sacrificing others (Gordon et al. 2008), and also across social and ecological components (Gunderson et al. 2002, Hirsch et al. 2011). However, tradeoffs can be difficult to identify, especially across sectors, such as conservation and social systems (Hirsch et al. 2011). Previous research has highlighted the need to be aware of “unintended consequences” when managing a SES water system, including tradeoffs between wildlife and water conservation (Huntsinger et al. 2017). Ecosystem functions lost as a result of human modification may affect ecological resilience (Gunderson et al. 2002), and in turn influence human community resilience (Fedele et al. 2017). Though we recognize that tradeoffs will occur when making decisions related to SES management, our analyses highlight where tradeoffs exist to move the conversation forward regarding overall SES resilience to multiple ECEs.

Management actions that maintain a water supply without detracting from ecological resilience, such as increased water use efficiency, increase overall water SES resilience. This is exemplified in our case studies, which illustrate how actions such as increased water storage reduced ecological resilience and eventually result in lower infrastructure resilience. High sediment loading in reservoirs occurs during hurricanes, reducing water storage amount and lowering long-term infrastructure resilience to drought. Increased efficiency, however, ensures the water present in storage systems is used sustainably and does not require additional direct impacts on the ecosystem. Strategies that do not affect the ecological system will likely be more effective in increasing SES resilience, as unforeseen consequences involving SES feedbacks during an extreme event can be avoided.

We did not evaluate how different management strategies, such as adaptive management, might function within the system. An adaptive management strategy may allow infrastructure to be maintained (Huntsinger et al. 2017). For example, water managers have proposed regular dredging (e.g. bi-annually) to reduce the effect of sedimentation on water storage, though this has yet to be implemented in Puerto Rico. Another option is to open the sluice gates during a large precipitation event, allowing sediment to move downstream (Morris Sequia Committee). This would also increase water storage volume and may increase stream ecosystem resilience as sediment transport would be restored. Another solution that could facilitate increasing infrastructure resilience if applied at a larger scale is increased water use efficiency. Other countries, such as China, recognize the importance of high water use efficiency as a way to maintain the water supply to allow for adaptive management (Allan et al. 2013). Though challenges such as political support, financial resources, and infrastructure design (not all large dams have sluice gates) exist, these solutions represent realistic options to increase overall water SES resilience. Though these solutions were initially proposed to maintain infrastructure longevity, they will also benefit stream ecosystems by maintaining connectivity.

Though a sustainable water supply through diverse ECEs is a high-priority management objective, managers focused on infrastructure solutions should consider the associated feedbacks that may cause an overall decline in water SES resilience. Previous research targeting water infrastructure resilience typically focuses more on the ability of the infrastructure system to withstand, absorb, restore, and adapt after a disturbance occurs (Shin et al. 2018). Though these are important concepts, ecological systems must be able to carry out these four resilience dimensions in response to disturbances as well (Rodina et al. 2015). Water infrastructure resilience to drought hinges on water quantity and quality, both of which are underpinned by the resilience of the ecological system. If a management action reduces the water quality in the ecosystem, such as dams extirpating migratory populations or increasing sediment accumulation, this reduces overall SES resilience and thus infrastructure resilience. Considering tradeoffs in SES resilience is especially necessary in the Caribbean, as the onset of climate change represents an uncertain future for water availability (Cashman et al. 2010).

The increasing frequency of ECEs world-wide has made water managers aware of the need for solutions that maintain water supplies through disturbances (Hossain 2020: 89). We recommend infrastructure managers consider the sustainability and long-term impact of their decisions on SES resilience. While Puerto Rico stream ecosystems are characterized by some ecological variables that respond and reset quickly, the same is not true for all systems (Western Australian grasslands Allison and Hobbs 2004). We suggest that the tradeoffs associated with implementing water management actions should be carefully considered in management decisions to ensure that overall SES resilience is maintained to extreme climatic events, as their frequency is increasing. Based on these analyses, we argue that engineering solutions associated with increasing flexibility and adaptability are insufficient (e.g., Wang and Blackmore 2009, Linkov et al. 2014), and maintaining overall SES resilience necessarily requires solutions emphasizing the role of ecological systems and tightly coupled feedbacks between water infrastructure and ecosystem health.

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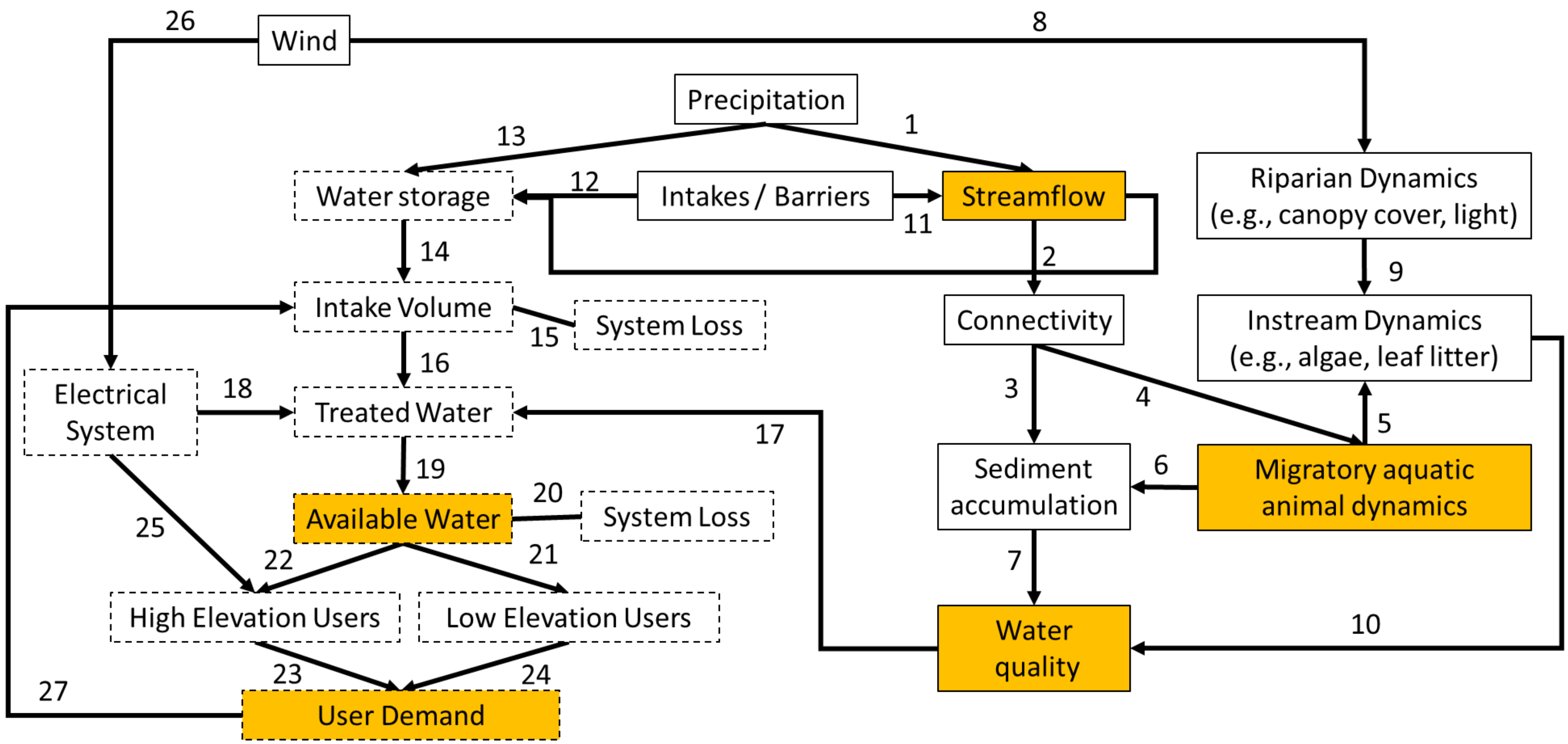

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Intake Volume
System Loss
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Connectivity
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(e.g., algae, leaf litter)
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Sediment
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Migratory aquatic
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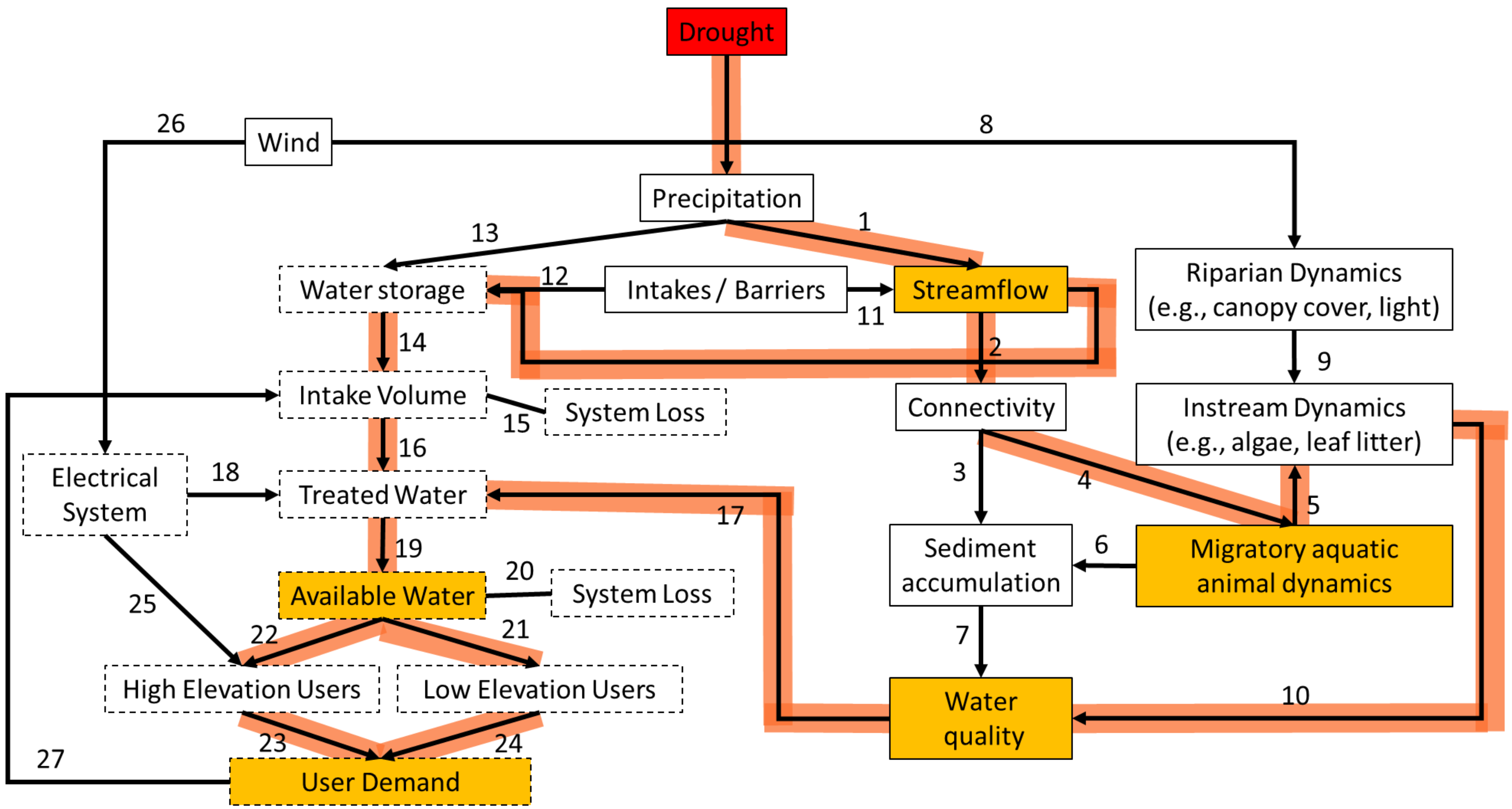

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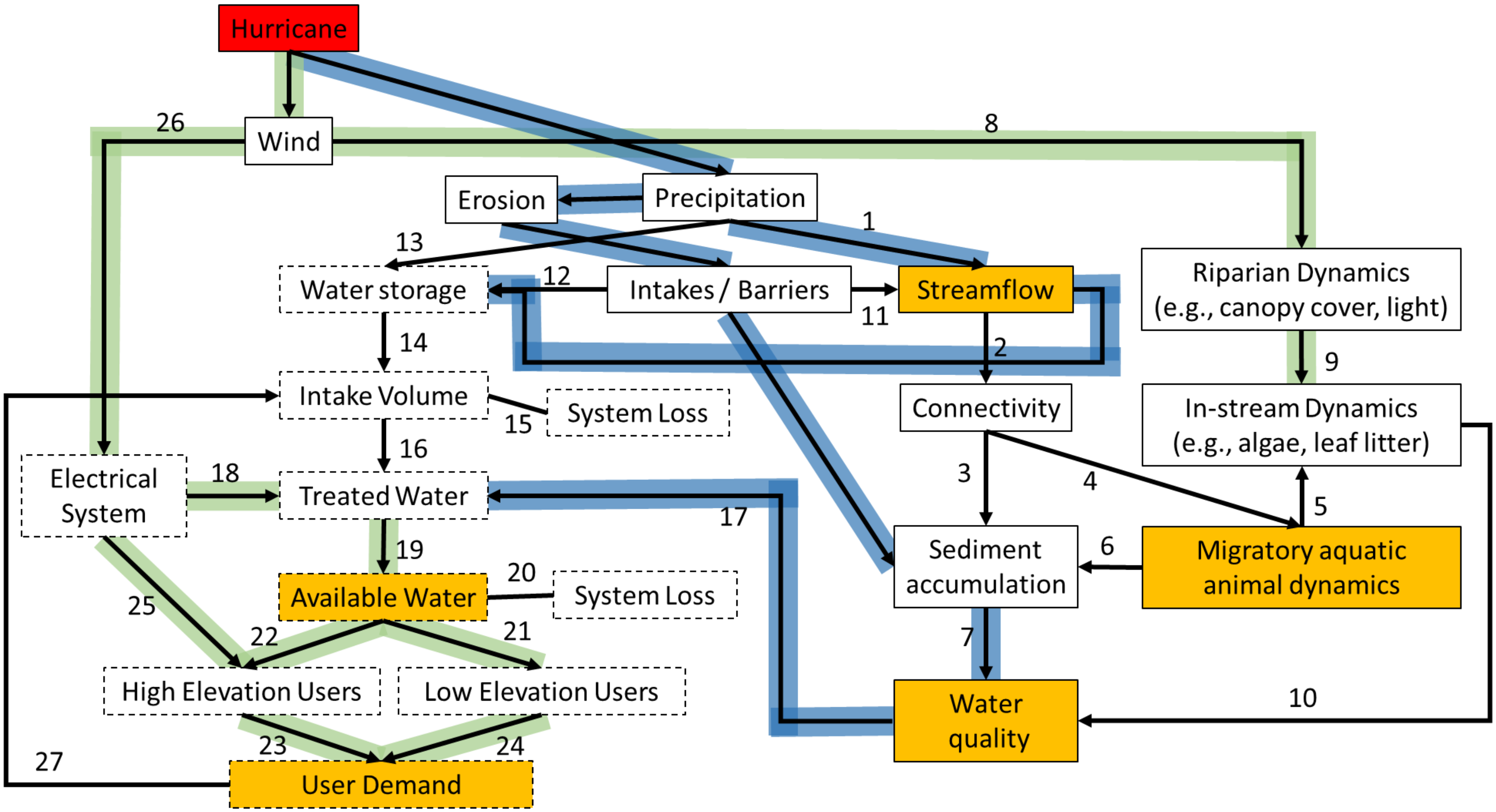
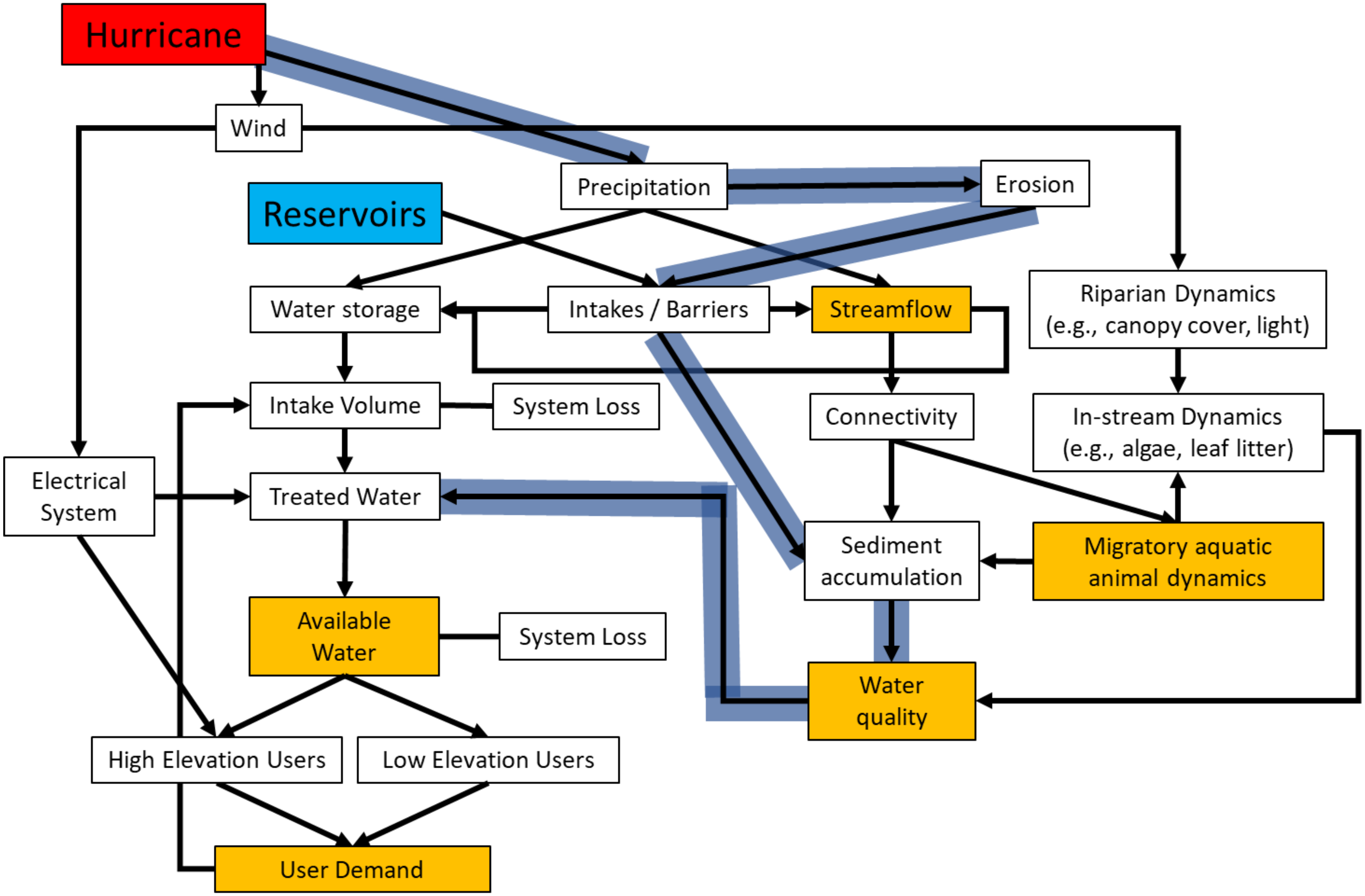


Figure 5.1. A baseline schematic of the water socio-ecological system in Puerto Rico. Each component contributes to overall system resilience. Solid boxes represent the ecological components and dotted boxes represent the social components. Orange boxes represent

key intermediate outcomes crucial to system feedbacks and all numbered links between components are explained in the text. Each figure shows: (a) baseline schematic of the water socio-ecological system, (b) how drought propagates through the system (pink lines), (c): how hurricanes characterized by high precipitation (blue lines) or high winds (green lines) affect the system.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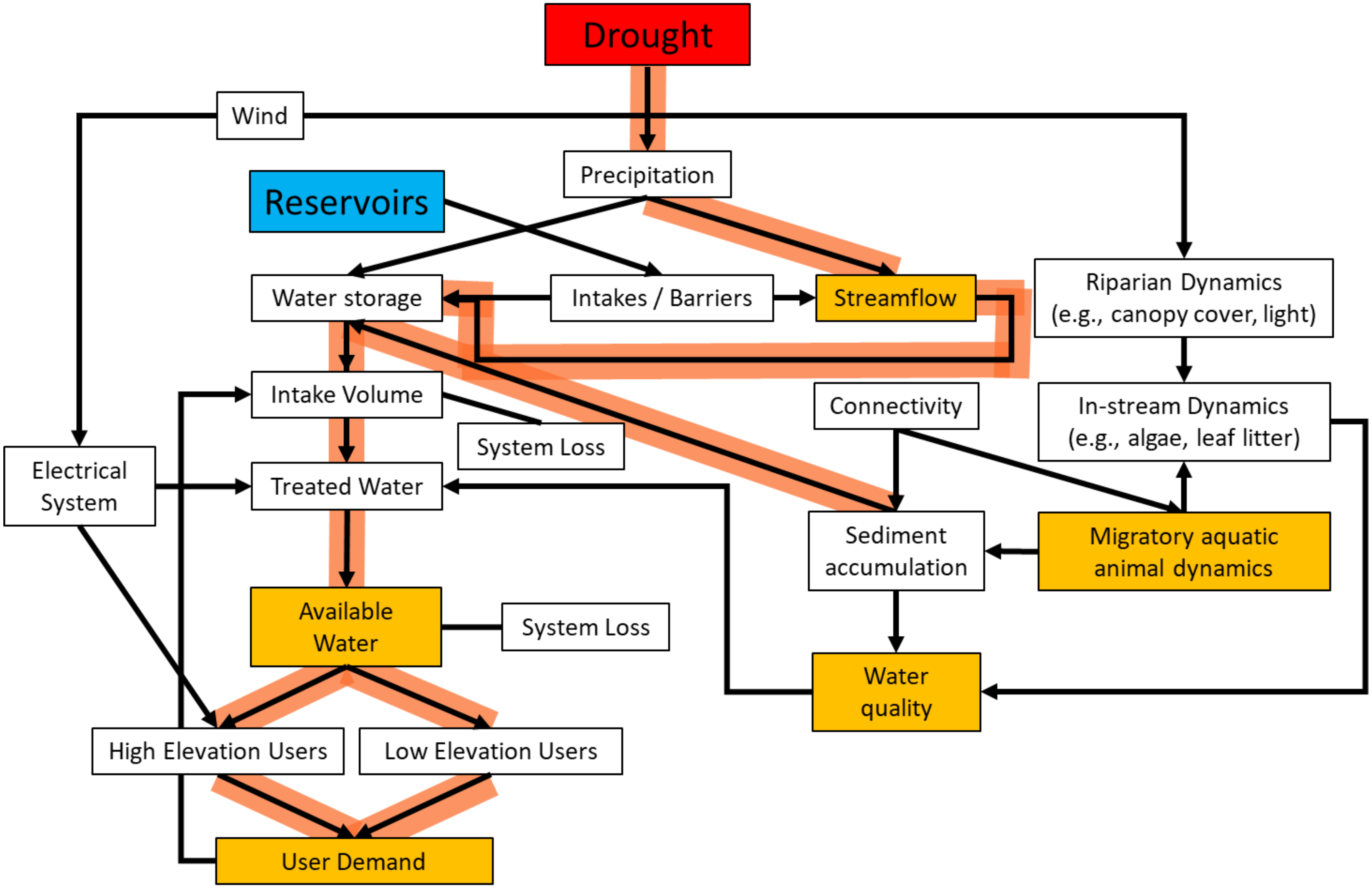

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Migratory aquatic
animal dynamics
Available
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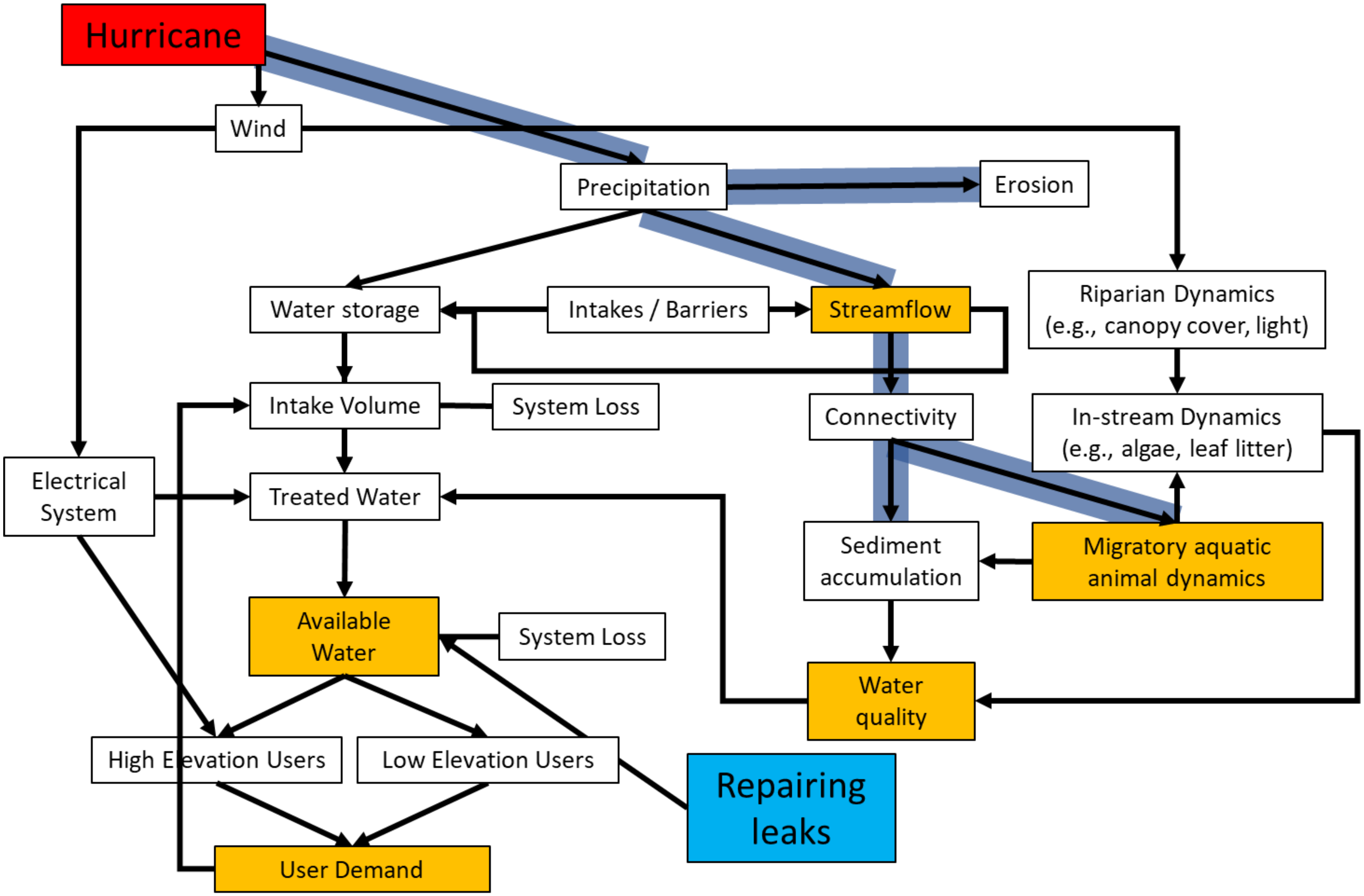
System Loss
Water
quality
High Elevation Users
Low Elevation Users
User Demand

Figure 5.2. Water SES response to two extreme climatic events with the addition of reservoirs. Increasing reservoirs would increase the number of barriers/ intakes present in the system. Blue arrows in the hurricane disturbance highlight the processes effected (top). Red arrows in the drought disturbance highlight the processes effected (bottom).



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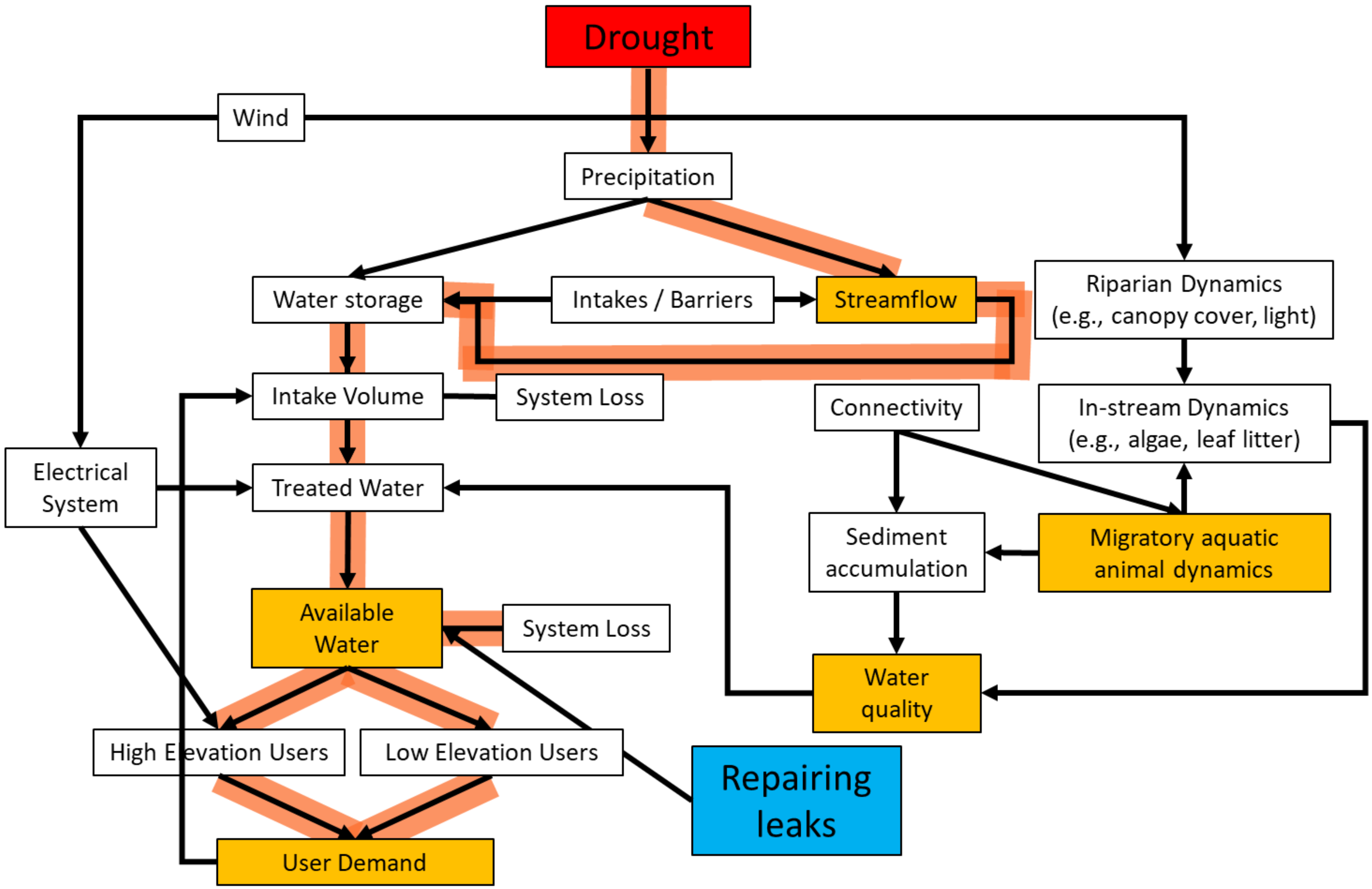
Low Elevation Users
Repairing
leaks
User Demand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animal dynamics
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High Elevation Users
Low Elevation Users
Repairing
leaks
User Demand

Figure 5.3. Water SES response to two extreme climatic events with leaks repaired. Fixing leaks would increase the available water present in the system. Blue arrows in the hurricane disturbance highlight the processes effected (top). Red arrows in the drought disturbance highlight the processes effected (bottom).

Table 5.1. Description of historical water management strategies, past actions, their outcomes, and the effect of ecological and social resilience where "+" indicates an increase in resilience, "-" is a decrease in resilience, and "0" means no effect.

Action	Description	Outcomes	Ecological resilience	Social resilience
Creating reservoirs	Construct a dam to store water in large reservoirs or small pools Reservoirs exist across the island	Preserve water for social use Reduces ecological connectivity, causing extirpation of migratory organisms and sediment accumulation	-	+/-
Interbasin transfer	Move water to domestic users in a separate watershed Superaqueduct: Constructed pipe to transfer 100 MGD to San Juan (42 mi)	Superaqueduct later declared unconstitutional but still functions Infrastructure leaks increase and water rationing returns in 2015 drought	-	+/-
Cloud seeding	Inject calcium chloride and silver into clouds to increase precipitation	No measurable increase in precipitation observed	0/-	0/-
Reduce consumer water use	Water rationing: PRASA enacts infrastructure shutdowns, limits for domestic users, and fines for wasteful water uses.	Domestic users hoard water during "water on" periods Changes in pressure cause underground pipe ruptures and leaks	0	+/-

Fix infrastructure leaks	Reduce leaks occurring from water distribution system Limit number of pipes needed in domestic water sector	Ongoing process due to financial and system configuration constraints Enhances sustainability of other water management actions Potential to lessen reliance on electrical system for water supply	0	+

CHAPTER 6

CONCLUSIONS

1. Introduction

In this chapter, I discuss an overview of my results across all chapters. I also evaluate how I see the results of each chapter fitting together to inform water management in Puerto Rico. I move on to discuss future research that would build on my work and extend my findings. I then reflect on my experience as an Integrative Conservation student, discussing challenges and giving advice to future students. Finally, I provide concluding remarks that include four main points that can be used to inform water management in Puerto Rico.

2. Summary of findings

2.1 Chapter 2

My objective for this chapter was to evaluate whether multiple low-head dams allowing for partial connectivity reduce longitudinal riverine connectivity for freshwater shrimps in northeastern Puerto Rico. I focused on seven watersheds draining El Yunque National Forest (EYNF) and determined how shrimp habitat connectivity varied through space and time (1980-2016). My findings illustrate how adding dams associated with intakes lower in the watershed (i.e. closer to the ocean) result in a decrease in shrimp habitat connectivity. I also evaluated the effect of dams on shrimp refugia habitat (i.e. habitat upstream of barriers over 5 m), because the lack of predators allows shrimp to reach higher densities. This analysis revealed that refugia habitat connectivity does not have as much temporal variation because nearly all refugia habitat occurs at higher elevations in the watershed (i.e. in the headwaters) and is impacted by even one dam in the watershed. Additionally, we expanded our results to consider how decreased shrimp habitat connectivity is likely decreasing the shrimp larval supply to the larger island-wide metapopulation. My main findings highlight how low-head dams paired with intakes have reduced shrimp habitat connectivity, especially refugia habitat, and the importance of considering connectivity as temporally variable.

2.2 Chapter 3

My objective for this chapter was to explore how changing water withdrawal rates affect habitat connectivity differently across a suite of native diadromous taxa (i.e. shrimp, goby, snail, mountain mullet, and American eel) in four watersheds in northeastern Puerto Rico. I selected watersheds that drained EYNF and whose boundaries extended to the ocean. I used a dataset that included 31 years of discharge data (1986-2016) and applied four different water withdrawal management scenarios to evaluate how habitat connectivity changed for each taxon. Results showed that connectivity for certain taxa was more affected by increased withdrawal rates than others. Taxa that are not limited in their movement by barrier height (i.e. shrimp, goby) demonstrated increased habitat connectivity loss rates as withdrawals increased, versus taxa that have their movements blocked by barriers (i.e. snail, mullet, eel). Additionally, this analysis highlighted the importance of considering downstream passage rates when calculating habitat connectivity for all migratory taxa. Historically, only upstream passage rates were used when calculating connectivity for migratory fish; however, downstream passage rates for larva should be considered especially for taxa that have large metapopulations (i.e. goby, mountain mullet, American eel). My main findings illustrate that decreased withdrawal rates will increase habitat connectivity for taxa that do not have their movement restricted by barrier height and highlight how larval mortality through entrainment can decrease habitat connectivity for amphidromous fish taxa.

2.3 Chapter 4

My objective for this chapter was to evaluate whether environmental flow legislation exists and how it is implemented in Puerto Rico. I developed criteria to identify two water targets (i.e. equitable allocation and water use efficiency) in legislation (*de jure*) and in practice (*de facto*). I found that both equitable allocation and water use efficiency are responsibilities assigned to multiple state agencies through legislation. My analysis of semi-structured interviews with water managers revealed mismatches in how environmental flow legislation was implemented in practice. In times of water scarcity (i.e. drought), water does not appear to be equitably allocated among users (i.e. domestic, industry, agriculture, environment) and water use efficiency is enforced at the individual level but not for the water utility company. My results also suggest the accountability of state-level water agencies may be restricting their ability to enforce environmental flow legislation. Specifically, state agencies are politically accountable to the Governor based on the structure of the state agencies created by legislation. My main findings suggest the accountability of water management agencies should be considered when creating environmental flow legislation, because this may limit effective enforcement and lead to mismatches in *de jure* and *de facto* water management.

2.4 Chapter 5

My objective for this chapter was to evaluate how water management actions have interacted with the water socio-ecological system (SES) feedbacks and influenced overall system resilience to droughts and hurricanes. I created a baseline schematic of the water SES in Puerto Rico to illustrate how feedbacks interact across the social and ecological components in response to droughts and hurricanes. I then evaluated the historical context for implementing five management actions to increase social (i.e. water supply) resilience to drought and discussed how each action interacted with existing water SES feedbacks to influence overall system resilience. My analysis highlighted how the ecological components are resilient to hurricane and drought events, but the water supply (i.e. social) components take longer to recover after hurricanes with strong winds resulting in electrical power loss. Additionally, management actions have historically been implemented in response to a drought and have not made the water supply system resilient to hurricanes or droughts. The inability of water management actions to maintain a water supply during extreme events partially results from ecological component feedbacks, such as increased sediment accumulation behind dams during hurricanes that lower water storage volume for droughts. Management actions can also result in lowered ecological resilience to disturbances. My main findings argue that water managers should consider the feedbacks that exist within the water SES and implement actions that result in overall SES resilience to extreme events.

3. Future research

Based on knowledge that I gained through developing and carrying out my dissertation research, I suggest that future research in the following four areas would benefit water management in Puerto Rico.

3.1 Ecosystem services/ conservation

One of the major justifications for maintaining shrimp populations is the importance of the ecosystem services that shrimp provide, as researchers have found filter-feeding shrimps maintain water clarity through preventing sediment and algal accrual (Pringle et al. 1996, Pringle et al. 1999). These processes no longer occur upstream of large dams that lack spillway discharge since shrimp populations have been extirpated (Holmquist et al. 1998, Greathouse et al. 2006). Researchers have suggested various management actions to reduce the effect of dams on shrimp populations, but the actions have not been widely implemented (Benstead et al. 1999, March et al. 2003).

To further shrimp population conservation, a monetary value could be placed on shrimp population extirpation. Water has less algae and sediment when withdrawn from streams with shrimp populations, likely reducing water treatment needs. If the cost of water treatment without shrimp present could be quantified, this could strengthen the reasoning behind maintaining shrimp populations. If state water management agencies are accountable to individuals who may not prioritize conservation, it may be necessary to frame the shrimp conservation argument using values shared by decision makers.

3.2 Connectivity theory

One topic for future study mentioned in Chapter 3 is determining the temporal component of migrations for different taxa in Puerto Rico. Existing literature on the timing of migratory movements for certain taxa suggests there is a temporal pattern (March et al. 1998, Engman et al. 2017, Smith et al. 2017). Many species seem to move upstream and downstream in a certain temporal pattern, however these trends need to be further studied to determine whether they persist throughout the watershed. Pinpointing when migratory species move may allow researchers to make more specific management recommendations (i.e. operate water withdrawals at 50% when the moon is changing from quarter to new phase, Smith et al. 2017). This research would contribute to reducing tradeoffs between social and ecological water demand.

Chapters 2 and 3 suggest that the impact on habitat connectivity can be determined by examining how water withdrawals affect the discharge rate. One of the major research gaps within these studies, as in many connectivity index research, is translating habitat connectivity into an ecological impact. For example, if connectivity ranged from 0.7-0.9 for decades, but then shifted to 0.4-0.6 after an additional low-head dam was installed, are there quantifiable effects on migratory species? Researchers have sampled migratory populations throughout EYNF streams, but not consistently through time and space. Patching together a dataset that includes historical abundances sampled by researchers from multiple institutions (e.g. US Forest Service, North Carolina State University, Luquillo LTER) with more recent estimates, and comparing changes in abundance to calculated connectivity index values could lead to a better understanding of whether decreasing temporal connectivity reduces migratory populations. The groundtruthing of reduced connectivity impacting migratory populations could serve to strengthen the results presented here and contribute to the wider connectivity literature.

3.3 Implementation

One area that could use further study is how environmental flow legislation is implemented. Chapter 4 explored one potential reason legislation may not be effectively implemented, but there are others that should be examined. An agency's power can influence its ability to carry out responsibilities (Ribot et al. 2004). The issue of power did come up in multiple interviews, with individuals explaining the water utility company received money from the Legislature, which was associated with having power. This concept should be further explored as it will likely contribute to our understanding of how water management decisions are made in Puerto Rico. The resource limitations of agencies (e.g. limited staff, lack of funding) should also be explored as agencies may be unable to carry out responsibilities due to financial and staffing constraints. A study evaluating how these issues influence environmental flow implementation would be useful to researchers attempting to effectively inform flow policy.

3.4 Conservation/ Motivations

An additional useful concept to explore is how EYNF stakeholders value streams and the services shrimp provide. Many people who enjoy EYNF streams are not aware of the role freshwater shrimp play in maintaining those benefits. As shrimp maintain water clarity, it is likely the places where shrimp are abundant are where people also enjoy recreating. Identifying traits valued by stakeholders and overlaying the ecosystem services shrimp provide (see Bennet et al. 2009) could result in a stronger argument for water management practices that promote sustainable shrimp populations. This approach would relate what stakeholders value to ecosystems services provided by shrimp.

Expanding on results from Chapter 5, more research is needed to understand why certain management actions are implemented. The historical analysis is a preliminary overview of events that influenced management actions, but a deeper understanding is needed. Thinking towards the sustainability of SES for future climate events, we need to understand the management actions considered and implemented in the past. For example, cloud seeding was implemented to mitigate the drought effects in 2015 although the science drought advisory committee suggested this solution would not be effective. A better understanding of what motivated decision makers to attempt weather modification could inform future recommendations from the scientific community. Additionally, ecologists should explore how management actions suggested by environmental engineers will influence ecosystems and overall SES resilience.

4. Integrative challenges

My experience in the Integrative Conservation (ICON) program was challenging but not without its rewards. My project would not have been complete without a social component. Learning more about the social issues surrounding my study site informed the questions I sought to answer as a researcher and affected the management recommendations that I made based on my results. For example, in Chapter 3 I explored whether altering water withdrawal rates can increase habitat connectivity based on my knowledge that dam removals are too costly for state agencies to consider. Additionally, I have learned the importance of defining terms. When working in integrative spaces, the same word can take on a different meaning to people. This was an important lesson, as almost all spaces are integrative in some form or another. Even when talking to other ecologists about connectivity, I now recognize we may be thinking of the term differently. I also don't use terms like "power," "resilience," and "authority" lightly. These words have been extensively researched and must be clearly defined. I will take these lessons with me throughout my career, and this knowledge makes me a more aware ecological researcher.

Being an ICON student is a constant exercise in humility. As we are engaging with theory in multiple disciplines, both ecological and social, it is challenging to think deeply about multiple disciplines simultaneously. Frequently I would think I finally understood one body of literature, only to come up short when my knowledge was compared with that of my committee members. I learned to enter meetings with an optimistic attitude, acknowledging my limitations before our discussions began. Ironically, this is the same skill academics advocate that research practitioners should use when approaching different forms of knowledge (Walley 2002). Perhaps a goal of the ICON program is to allow scientists to practice humility.

Additionally, within the ICON program, as potentially with all PhD programs, progress is infinitesimal yet infinite. The work put into the dissertation project often does not beget progress for weeks to months. However, after years of struggle, hard work is visible and rewarded. This process seems more difficult for ICON students, because even when you think you've achieved an understanding of the additional disciplines that you're engaging with, there is still so much to learn. Consequently, part of the learning process also requires you to make peace with yourself, to acknowledge that you'll never know as much as you think you "should know" by the time you are ready to graduate. There's always more.

Thinking about first- or second-year ICON students, I wish I could tell them that it gets easier to be vulnerable to criticism. To hear that you need to read more, think harder, know more. But in fact, it gets harder with time, because you think you've progressed. Again, this is part of learning how to be a humble researcher and to recognize your limits. You must be vulnerable and humble to be in the ICON program, qualities that also make for a good scientist.

5. Concluding remarks

In my dissertation, I explore water management issues in Puerto Rico from both an ecological and social perspective. Based on this research, I suggest four main points that can be used to inform water management in Puerto Rico:

- Low-head dams paired with intakes can reduce shrimp habitat connectivity under certain conditions, especially if the dams are located lower in the watershed (i.e. closer to the ocean);
- Altering water withdrawal scenarios have the potential to improve habitat connectivity for multiple migratory taxa if the intakes do not have an associated instream barrier;
- Environmental flow legislation should consider the accountability of agencies tasked with implementing regulations;
- Water managers developing actions targeting increased social resilience to drought should evaluate how the action will interact with feedbacks throughout the water SES.

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APPENDIX A

SUPPLEMENTARY FIGURES AND TABLES FOR CHAPTER 2

Table S1. Proportion of total habitat and refugia habitat upstream of intakes (i.e. affected) through time by watershed, shown as a percent. The proportion of habitat affected was calculated by dividing the proportion of each watershed upstream of an intake by the total stream length in the watershed. The proportion of refugia habitat was calculated by dividing the proportion of refugial habitat upstream of an intake by the total refugia stream length in the watershed.

	Total Habitat Affected (%)			**Refugia Habitat Affected (%)**		
Watershed	**1980**	**1998**	**2016**	**1980**	**1998**	**2016**
Blanco	55	55	27	98	98	84
Canovanas	94	94	94	100	100	100
Espiritu Santo	5	91	91	17	100	100
Fajardo	16	16	50	100	100	100
Gurabo	12	84	84	88	100	100

Mameyes	0	72	0	0	100	0
Sabana	41	41	41	100	100	100

Table S2. List of all intakes and their characteristics included in this study. Intake name represents the name of the intake given by an agency or the location of the intake. Stream length upstream of an intake was calculated using the NHDPlus flowlines and is in km. Year online is the earliest year the intake began withdrawing water. Year offline is the earliest year the intake was no longer withdrawing water. Average water withdrawn is the average amount the intake withdrew over its entire operating period in cms. The average is reported as some intakes fluctuated in the amount withdrawn through time.

Watershed	Intake Name	Stream Length Upstream Intake (km)	Year Online	Year Offline	Average Water Withdrawn (cfs)
Blanco	Rio Blanco	59.8	1978	2010	22.4
Blanco	Intake Blanco Reservoir	29.65	2010	NA	21.276
Blanco	La Mina	3.9	1951	NA	1.483
Blanco	Rio Sabana	3.6	1928	1992	0
Blanco	Rio Icacos	6.8	1928	NA	6.652
Blanco	Rio Prieto	2.8	1928	NA	6.652
Blanco	Cuchilla-Higuerillo	0.8	2006	2014	0.002
Canovanas	Canovanas River	60.0	1968	NA	4.859

Canovanas	Cubuy River	11.4	1980	NA	1.496
Canovanas	Camito	2.3	2004	NA	0.002
Canovanas	Los Santos	2.9	1980	NA	0.470
Canovanas	Raul Casanovanas	0.9	1997	NA	0.001
Espiritu Santo	Guzman El Yunque	32.8	1982	NA	10.393
Espiritu Santo	Guzman Arriba	4.0	1980	NA	1.666
Espiritu Santo	El Verde	40.8	1982	NA	15.588
Espiritu Santo	Morovis	4.9	2008	NA	0.537
Espiritu Santo	Jiminez	2.8	2008	NA	0.216
Espiritu Santo	El Zarzal-2	0.6	2008	2013	0.316
Espiritu Santo	El Zarzal-1	0.5	2008	2013	0.316
Fajardo	Fajardo Reservoir	54.7	2006	NA	14.659
Fajardo	Rio Fajardo	17.2	1966	2006	13.379
Gurabo	Ceiba Sur	273.6	1994	NA	4.514

Gurabo	Rio Valenciano	78	1984	NA	3.414
Gurabo	Pueblito del Rio	39.9	1951	NA	1.700
Gurabo	Juncos	11.5	1951	NA	0.957
Gurabo	Pasto Seco	4.4	2008	2013	0.130
Mameyes	Palmer	44.0	1998	2013	4.422
Sabana	Cristal	3.9	1956	NA	1.453
Sabana	Sabana	7.0	1956	NA	1.453

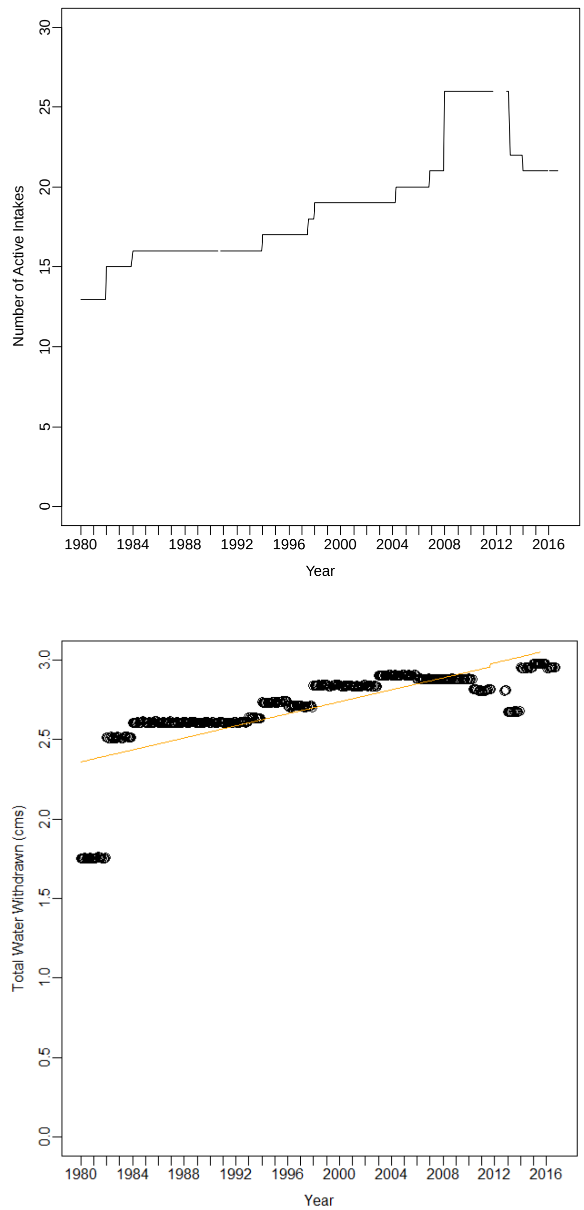

Figure S1. Temporal trends in additional explanatory parameters for total and refugia habitat connectivity. *Left*: Number of intakes actively withdrawing water across seven watersheds through time. Intakes were only included if a known water withdrawal amount could be identified. The maximum number of intakes actively withdrawing water over the course of this study was 27. *Right*: Total water withdrawn from El Yunque

National Forest through time (cms). We calculated the total water withdrawn by summing the water withdrawn from each active intake at each monthly time step.

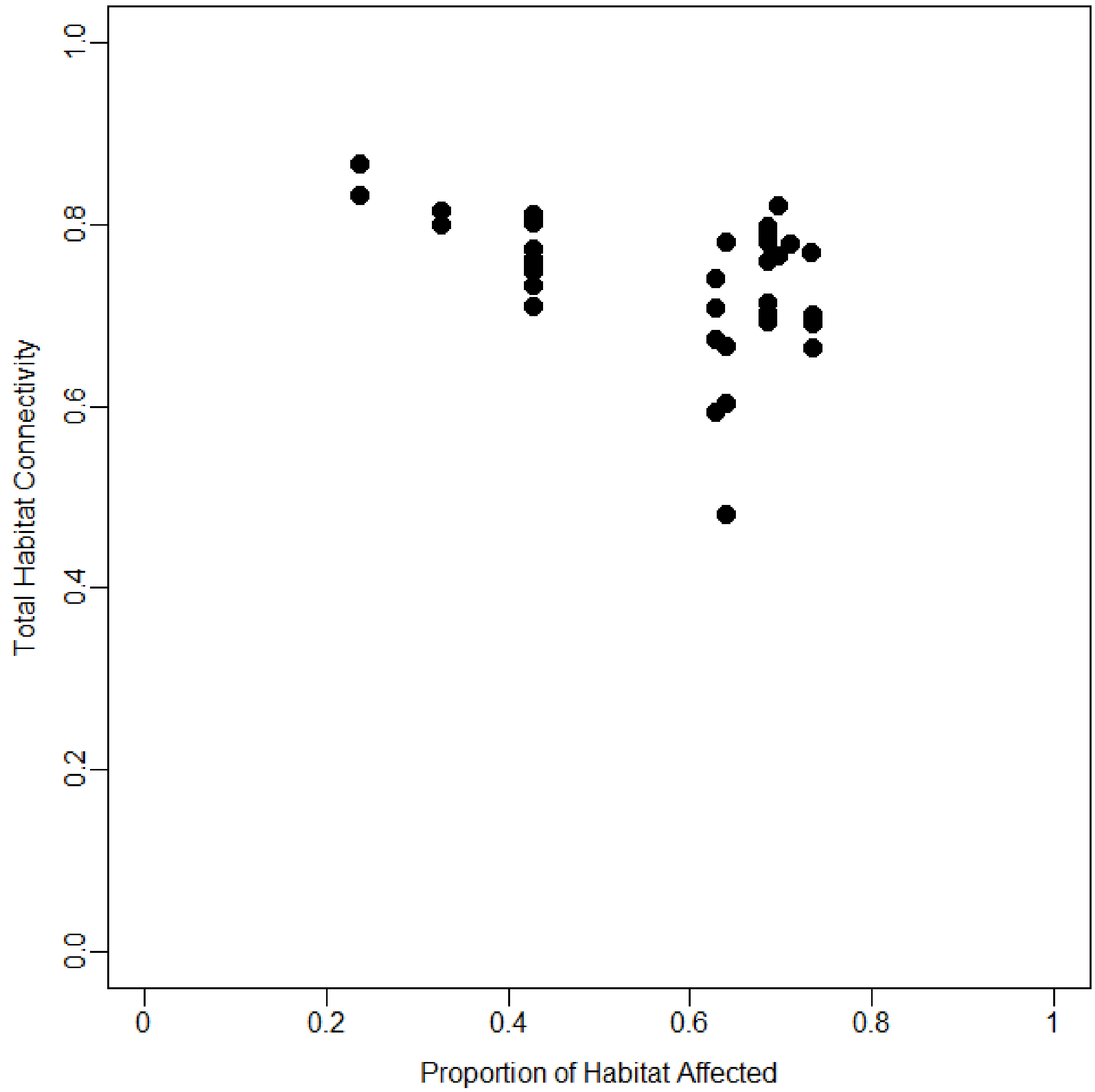

Figure S2. Average annual total habitat connectivity compared to the average annual proportion of habitat affected (i.e. habitat upstream of an intake) (r^2 = -0.4193) across all seven focal watersheds. We calculated stream habitat using NHDPlus flowlines.

APPENDIX B

INTAKE NAMES AND LOCATIONS USED IN CHAPTER 2 AND 3

Intake locations are in state plane coordinate system (meters).

Watershed	**Intake**	**Latitude**	**Longitude**
Blanco	Rio Blanco	268655.8473	242875.8086
Blanco	Intake Blanco Reservoir	268510.3002	245465.9258
Blanco	La Mina	266720.9958	247882.5117
Blanco	Rio Sabana	267569.1965	248067.0883
Blanco	Rio Icacos	268641.4107	247903.4508
Blanco	Rio Prieto	269197.0007	246990.2712
Blanco	Cuchilla-Higuerillo	265300.9945	245242.4849
Canovanas	Canovanas River	257120.3568	260299.4785
Canovanas	Cubuy River	259720.9518	247522.4304
Canovanas	Camito	261404.4853	247160.663
Canovanas	Los Santos	260749.9491	248076.2662
Canovanas	Raul Casanovas	261263.2797	249857.2531
Espiritu Santo	Guzman El Yunque	264367.7114	259952.0222
Espiritu Santo	Guzman Arriba	262943	251012.003
Espiritu Santo	El Verde	265040.4226	259792.5273
Espiritu Santo	Morovis	264616.9609	252552.9071
Espiritu Santo	Jimenez	266079.457	255693.2502
Espiritu Santo	El Zaral- 2	268987.4811	255644.4882
Espiritu Santo	El Zarzal- 1	268933.4831	255235.0463
Fajardo	Fajardo Reservoir	277653.079	251127.2165
Fajardo	Rio Fajardo	276013.6727	248430.7
Gurabo	Ceiba Sur	249539.8164	246846.6842
Gurabo	Rio Valenciano	253804.694	241848.8475
Gurabo	Unnamed Creek	256241.0613	235682.3325
Gurabo	Pueblito del Rio	260648.1849	243154.7705
Gurabo	Juncos	262179.1612	246127.2066
Gurabo	Pasto Seco	264458.2794	246349.3111
Mameyes	Palmer	270347.8758	259212.1322
Sabana	Cristal	274198.636	253337.7134
Sabana	Sabana	274574.6407	253133.6278

APPENDIX C

INTERVIEW QUESTIONS FOR CHAPTER 4

Interview questions for each interviewee by agency.

Each individual:

1. Here are the water management statutes in Puerto Rico we have identified. Does it look like we are missing anything? Are there any additional regulations we should examine?

2. What would you say your company/agency is in charge of in terms of water management in Puerto Rico? Why does your agency focus on this issue?

3. We are curious about equitable water allocation. Are there any events you feel illustrates this well in Puerto Rico? When was equitable water allocation the most difficult to address?

4. We're curious about how the decision to install the SuperAqueduct played out. I know this was a solution to meet the water demand of San Juan, but would you mind explaining to me whether other options were discussed?

5. Looking back on the 2015 drought, it seemed that water allocation was a difficult question there was no good solution for. What do you think were some of the key reasons for this? Also, which were the agencies involved in deciding how water would be allocated?

6. What issues should be discussed to move towards a more equitable water allocation across users including: domestic, agriculture, industry, and the environment? And who do

you think, based on your knowledge of the current water management agencies, should be in charge of enforcing this allocation?

7. In your opinion, who has the most say in determining how water is managed in Puerto Rico? Why?

AAA (water utility):

1. Making sure there is enough of a water supply to meet the needs of the people on the island seems like it would be a difficult task. Does AAA work with other agencies to achieve this goal?

2. Is it hard to maintain communication between AAA and these other agencies?

3. Would you say that AAA and DRNA generally have positive interactions? Do you feel like both agencies are able to work well together?

4. The 2015 drought seemed especially difficult for AAA, as they are the agency in charge of supplying water to the public. What are the different factors AAA considers when deciding when to start rationing water? Are there certain users (ex. homes, businesses, etc.) that get prioritized first over others?

AEE (electric utility):

1. Would you say AEE plays a key role in water management on the island? Why or why not?

2. It seems AEE was heavily involved with discussions surrounding the 2015 drought. What sort of information was AEE able to provide that other agencies were not?

DRNA (Department of Environmental and Natural Resources):

1. How do interactions with AAA function? Do you feel you work well together?

2. It seems like DRNA is tasked with enforcing nearly all the protections of the environment on the island of Puerto Rico. How does DRNA enforce these protections? Does DRNA work with other agencies to ensure enforcement? Does there seem to be certain restrictions where enforcement isn't possible?

3. Do you believe PRASA should be held accountable to increase efficiency of water use?

Federal agencies:

1. Are your interactions with AAA and DRNA generally positive?

2. Do you feel it's difficult for USFS to meet its management requirements while working with AAA?

3. Have there even been negative experiences with DRNA and/or AAA?