# AN EXAMINATION OF THE INTERCONNECTED SOCIAL AND ECOLOGICAL DIMENSIONS OF STORMWATER MANAGEMENT

by

**Rachel Scarlett**

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# STATEMENT OF COMMITTEE APPROVAL

**Dr. Sara McMillan, Chair**

Department of Agricultural and Biological Engineering

**Dr. Jake Hosen**

Department of Forestry and Natural Resources

**Dr. Zhao Ma**

Department of Forestry and Natural Resources

**Dr. Suresh Rao**

Lyles School of Civil Engineering

**Approved by:**

Dr. Nathan Mosier

Dr. Linda Lee

*All good things must begin.*

- Octavia E. Butler

*To my mother. May she rest powerfully knowing that her seed has been sown.*

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# ABSTRACT

Land use change is a major cause of degradation to freshwater ecosystems. Excess nutrients and toxins, physical infrastructure, and habitat removal can lead to deleterious impacts on water quality, flooding, and biological integrity. The overarching inquiry of this dissertation was to assess how social and ecological dimensions of stormwater interact to influence stormwater and its management. A three-part study was conducted to investigate the ecological and social dynamics of aquatic ecosystems. In part one, I investigated the impacts of urbanization on stream metabolism— a fundamental ecological process. The proliferation of inexpensive water quality sensors has allowed researchers to investigate stream functional processes at a high temporal resolution. I used high-resolution dissolved oxygen data to estimate gross primary production (GPP) and ecosystem respiration (ER) across 12 urban creeks in Charlotte, North Carolina, USA. I used descriptive statistics and regression models to investigate the influence of light, temperature, and hydrological disturbances on GPP and ER. The results demonstrate that urbanization shifts metabolic regimes towards highly productive summers with substantial declines in GPP following summer storm events. My research shows that ER is associated with water temperature and is resistant to hydrological disturbances. These findings have management implications because as summer heat and storms intensify with climate change, my work suggests that stream organisms will become more vulnerable to scour and hypoxia.

In part two, I conducted a systematic literature review to identify salient social norms impacting water quality best management practice (BMP) adoption across urban and rural lands. Furthermore, I synthesized situational factors that mobilize and reproduce social norms associated with BMP adoption. The results demonstrate that social norms create expectations for conventional farming practices and manicured residential lawns, as well as a social responsibility for neighborly cohesion and environmental stewardship. Social norms supporting water quality BMPs were fostered during times of management uncertainty and in response to social sanctions and benefits. I found that social norms supporting water quality BMPs were more readily mobilized when supported by key community leaders, knowledge brokers, and institutional actors.

In part three, I examined if and how an individual's race, gender, and education level shape one's concern about and willingness to participate in stormwater management. Stormwater risks can be immediate burdens and at times life-threatening for marginalized people because

environmental injustices based on race, gender, and class can dictate exposure to and recovery from environmental risks like flooding and water pollution. Although marginalized groups bear the brunt of environmental risks, they are not likely to be perceived by others as highly concerned about the environment. I investigated differences, if any, in peoples' willingness to participate in stormwater management based on their race, gender, and educational level by analyzing community opinion surveys in Charlotte, North Carolina. Results suggest that socially marginalized individuals are more concerned about creek flooding than others and subsequently more likely to participate in conservation behaviors. This analysis calls attention to how adverse environmental conditions may shape the perspectives of those experiencing them and facilitate a greater willingness to engage in conservation practices. Collectively, this dissertation highlights the interconnectedness of human and ecological drivers of function and resilience in aquatic freshwater ecosystems with implications for future directions of freshwater management that prioritize social equity and sustain social infrastructures.

# CHAPTER 1. INTRODUCTION

Human development is a major cause of the degradation of freshwater ecosystems. Excess nutrients from urban and rural lands contribute to eutrophication and hypoxia of sensitive aquatic ecosystems (Carpenter et al., 1998; Howarth et al., 2000, 2011). Moreover, nuisance algal blooms and sediment pollution associated with urbanization and agriculture degrade ecological health and threaten our drinking water sources (Carmichael & Boyer, 2016; Gaffield et al., 2003). Impervious surface areas associated with urbanization have increased the frequency and magnitude of flooding events in watersheds with immense consequences for aquatic biota and ecological function (e.g., Walsh et al., 2005).

The changes to ecosystems wrought by human development impact ecosystem dynamics and have detrimental impacts on human lives. The cost of mitigating flood-related damage is increasing (Brody et al., 2007), and more importantly, people's livelihoods and well-being are threatened as a result of persistent flooding and water quality issues. Notably, such hazards disproportionately impact socially and economically marginalized groups (Liévanos, 2017; Qiang, 2019). Additionally, women, impoverished individuals, and racially marginalized individuals often face higher barriers to recovery from flooding and water quality hazards (Enarson & Fordham, 2000; Hendricks & Zandt, 2021).

Traditionally, water supply and treatment infrastructure have been technocratic, managed by a selective group of engineers and decision-makers (Finewood, 2016). This has led to water management that is largely separate from people and nature. The central theme of my dissertation is that freshwater ecosystems and society are mutually entangled and codependent on each other (Bakker & Bridge, 2006). I contend that the relationships between water and society are shaped through the physical infrastructures we build to manage the flooding and water quality problems disproportionately experienced in marginalized communities and the social connections that shape environmental decision-making.

Specifically, in Chapter 2, I demonstrate how human practices associated with urbanization—including but not limited to increasing impervious surface area, removing riparian vegetation, and applying excess fertilizer—uniquely shape ecological function of urban creeks to become both highly productive and susceptible to disturbances. I show how processes of human development alter functional properties of freshwater ecosystems and potentially shift these ecosystems into alternative functional regimes. Broadly, this chapter highlights how human development drives energetics of urban creeks, and this work has implications for targeted management practices at the stream and riparian scale.

In Chapter 3, I demonstrate how salient social norms drive human decision-making associated with water quality best management practices. I found that social connections, and more importantly, salient social expectations, shape environmental decision-making and practices. I show how social norms have been understudied in relation to water quality BMPs and highlight the importance of considering social norms when promoting conservation practices.

Chapter 4 shows how changes wrought by human development have inequitable consequences on marginalized groups of people; importantly, I connect disproportionate impacts of flooding and poor water quality to people's willingness to participate in stormwater management. Chapter 4 highlights how racism, classism, and sexism structure people's experiences with stormwater—leaving the most marginalized groups to the worst flooding conditions—and how these experiences heighten individuals' concern about flooding and willingness to participate in its management.

Overall, this work unravels a few interesting and intricate connections between society and freshwater ecosystems. In each chapter, you will observe an explicit focus on the interconnectedness of human and ecological drivers of function and resilience in aquatic ecosystems with implications for future water management directions across rural and urban lands.

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# CHAPTER 2. SHIFTS IN STREAM ECOLOGICAL FUNCTION WITH INCREASING URBANIZATION

## 2.1. Abstract

Stream metabolism, the coupling of gross primary production (GPP) and ecosystem respiration (ER), is a fundamental ecological process that captures the predominant bioenergetics of stream ecosystems. Frequent and intense flow events in urban creeks can scour stream beds and increase turbidity, which can constrain stream community development and reduce metabolic rates. Alternatively, increased light and energetic subsidies, including nutrients and allochthonous carbon, from the surrounding terrestrial landscape can amplify stream metabolism. In this study, I had three research questions: 1) which environmental factors control seasonal patterns in stream metabolism, 2) how does stream metabolism respond to hydrologic disturbances, and how does this response vary across watershed and stream characteristics, and 3) how does urbanization influence variability in stream metabolism? Using five years of high frequency dissolved oxygen, temperature, and discharge data, I estimated daily GPP and ER across 12 urban creeks in watersheds ranging in size from 15-168 $km^2$ in Charlotte, North Carolina, USA. I used fixed effects models to estimate the association between light, temperature, discharge, and stream metabolism. I used redundancy analysis to investigate the association between urbanization characteristics and seasonal GPP and ER. I additionally investigated the metabolic response to hydrologic disturbances by estimating the magnitude of change in GPP and ER following storms. The results demonstrate that seasonal GPP and ER were primarily driven by light and temperature regimes. Flow disturbances did not significantly influence seasonal patterns. GPP increased with watershed imperviousness and decreased with riparian and bank vegetation. At the storm time scale, I observed a persistent decrease in GPP following summer storms, and, in contrast, ER remained resistant to hydrologic disturbances. Overall, the results imply that urbanization shifts creeks towards highly productive regimes frequently disturbed by summer stormflows. This work shows that mitigation of both stream-scale (riparian and bank vegetation) and watershed-scale (imperviousness and nutrient inputs) stressors on ecosystem function is needed to reduce excess productivity and restore urban stream metabolic regimes.

## 2.2. Introduction

Urbanization has caused water quality and hydrological impairments in receiving lakes, estuaries, streams, and creeks. Increases in impervious surface cover, modified drainage networks, and pollution in urban watersheds lead to decreases in stream biodiversity (McCluney et al., 2014) and increases in frequency of flooding and pollutant loads (O'Driscoll et al., 2010; Walsh et al., 2005). The 'Urban Stream Syndrome' concept synthesizes the impacts of urbanization on streams, including increased magnitude and frequency of hydrologic disturbances (e.g., Bell et al., 2016; O'Driscoll et al., 2010), increased nutrients and toxins (e.g., de Jesús-Crespo & Ramírez, 2011; Hatt et al., 2004; Howarth et al., 2011), decreased channel complexity (e.g., Elmore and Kaushal, 2008; Pennino et al., 2014), and degradation of biotic communities (e.g., Konrad and Booth, 2005; Roy et al., 2016; Violin et al., 2011). While the impact of urbanization on hydrology, nutrients, and macro-organisms is well-studied, fewer studies have addressed urbanization's impact on basal energetics in stream ecosystems. This task has been cumbersome in the past due to methodological constraints; however, novel modeling techniques have provided researchers with an opportunity to study stream ecological functioning at high temporal scales (Bernhardt et al., 2018).

Gross primary production (GPP) and ecosystem respiration (ER) are two fundamental stream ecological functions. Primary producers perform photosynthesis, a biological process that converts inorganic carbon, in the form of $CO_2$, to organic carbon compounds that provide energy to all life forms. Ecosystem respiration is the subsequent mineralization of organic compounds (by both autotrophs and heterotrophs). At the global scale, streams play an integral role in regulating the carbon cycle (Drake et al., 2018). Streams receive terrestrial organic matter, and ultimately the fate of carbon follows one of 3 dominant paths: (1) burial of carbon within the stream bed, (2) outgassing of carbon in the form of carbon dioxide and methane, or (3) export of carbon to the ocean (Cole et al., 2007). Of the 5.1 Pg C $y^{-1}$ terrestrial carbon inputs to streams and inland waters, these waters outgas 3.8 Pg C $y^{-1}$ to the atmosphere, bury 0.6 Pg C $y^{-1}$, and export less than 20% of their inputs to the ocean (Drake et al., 2018). This indicates that there are substantial sinks and carbon transformations within streams. At the stream network scale, GPP and ER shape the carbon balance of aquatic ecosystems and ultimately provide a functional template that fuels or constrains the aquatic food chain (Bernhardt et al., 2018).

Light, nutrients, discharge, and temperature are primary drivers of stream metabolism (Bernhardt et al., 2018), and recent research has provided insight into how urbanization influences these drivers and subsequently shapes stream metabolism. Photosynthetically active radiation (PAR) is consistently a main control on GPP in streams (Mulholland et al., 2001; Savoy et al., 2019; Young & Huryn, 1999). Urbanization is often associated with riparian vegetation removal, which has resulted in elevated GPP in urban creeks with low canopy cover compared to their forested counterparts (Alberts et al., 2017; Smith & Kaushal, 2015). However, GPP in urban streams is elevated compared to reference streams, even in urban streams with closed canopies (Alberts et al., 2017). This suggests that high nutrient inputs, as well as light, contribute to the positive association often observed between urbanization and GPP (Bernot et al., 2010; Finlay, 2011). Conversely, studies show that suspended sediment and channel incision can constrain GPP in small urban creeks (Blaszczak et al., 2018; Larsen & Harvey, 2017).

Hydrological disturbances are essential to aquatic life, yet also stressful. Storm events represent a small fraction of flow conditions; however, they deliver an overwhelming majority of carbon subsidies to the stream relative to baseflow conditions (Moatar et al., 2017; Raymond et al., 2016; Zarnetske et al., 2018). There is extensive evidence that dissolved organic carbon (DOC) export increases with discharge (Zarnetske et al., 2018), and recent studies have shown that organic matter transported during these events fuels respiration (Demars, 2019; Reisinger et al., 2017). On the other hand, flashy hydrology can decrease bed stability (Blaszczak et al., 2018; Townsend et al., 1997), decrease water residence time— which limits carbon mineralization in streams (Casas-Ruiz et al., 2017)— and increase light attenuation through increased suspended sediment concentration (Larsen & Harvey, 2017). In urban streams, stormflows have been moderately associated with increases in ER (Larsen & Harvey, 2017; Qasem et al., 2019). On the contrary, GPP often decreases in storm events in urban streams and flood-prone forested reaches alike (Qasem et al., 2019; Reisinger et al., 2017; Roberts et al., 2007; Uehlinger, 2000, 2006). The type and intensity of storm events matter as well. GPP is less resistant to storms with rapid rates of change in discharge that likely contribute to scour (Qasem et al., 2019). Incised urban headwater streams can oscillate between hydrologic disturbances and hypoxia inducing a persistent stressed state for stream organisms (Blaszczak et al., 2018).

Less well understood is the relative importance of both hydrology and light in shaping temporal dynamics in urban stream metabolism. Changes in flow, especially flow extremes like droughts and storms, may decouple metabolic regimes from light and thermal regimes, creating highly variable and aseasonal metabolic regimes (Savoy et al., 2019). Given that flashy hydrographs and simplified channels characterize urban streams, it is yet understood whether urbanization decouples stream metabolism from thermal and light regimes due to chronic disturbances. Hydrologic disturbances can introduce stochasticity into the metabolic regime, possibly placing these streams in a persistent cycle of disturbance and recovery (Blaszczak et al., 2018; Clapcott et al., 2016; Reisinger et al., 2017).

This study investigated how stream metabolism responds to disturbances and assessed the relative importance of hydrological, light, and temperature regimes on stream metabolism in several urban creeks. I addressed three research questions: 1) which environmental factors control seasonal patterns in stream metabolism, 2) how does stream metabolism respond to hydrologic disturbances, and how does this response vary across watershed and stream characteristics, and 3) how does urbanization influence variability in stream metabolism? To address these questions, I investigated the influence of light, temperature, and discharge on GPP and ER in twelve urban creeks. I also explored the impacts of land use and stream characteristics on seasonal GPP and ER. To investigate the influence of storm events on GPP and ER, I used Boltzmann-Arrhenius models to quantify the mean response of gross primary production and ecosystem respiration to stormflows. Then, I used redundancy analysis and linear regression models to assess watershed and stream scale controls on metabolic resistance to stormflows.

## 2.3. Methods

I used high frequency dissolved oxygen time series data to model GPP and ER in urban creeks from 2013-2018. The study sites are located in Charlotte, Mecklenburg County, North Carolina, USA. I estimated stream metabolism using a single station open channel approach. I additionally modeled light at the stream surface using a biophysical model. I calculated seasonal means of GPP and ER at each site. To assess the influence of environmental factors on stream metabolism, I used fixed effects models to estimate the relationship between light, temperature, discharge, and stream metabolism. I additionally estimated the effects of watershed (i.e., impervious area, tree cover, watershed area) and stream characteristics (i.e., riparian vegetation, bank vegetation scores) on seasonal means of GPP and ER with redundancy analysis (RDA).

To assess metabolic responses to storm events, I investigated the impact of episodic storm events on daily mean GPP and ER. I used linear models of the Boltzmann-Arrhenius equation to estimate the change in GPP and ER following storm events while controlling for potential changes in temperature during events. I used RDAs to assess how metabolic responses to storm events differed across site characteristics. Lastly, to assess variability in metabolism across an urbanization gradient, I calculated the proportional variability (PV) index on daily means of GPP and ER at each site. I provide more detailed explanations of the methods in the following sections.

### 2.3.1. Study Sites

I examined twelve urban streams in the Charlotte, Mecklenburg County, North Carolina, USA metropolitan region (Fig. 2.1). The sites are located within two major river basins: the Catawba River Basin and the Yadkin/Pee Dee River Basin. The sites are a part of the Charlotte-Mecklenburg Storm Water Services (CMSWS) Continuous Monitoring and Alert Notification Network (CMANN), a network of automated water quality monitoring stations across Mecklenburg County. All sites are collocated with United States Geological Survey (USGS) long-term stream monitoring stations. I selected sites based on availability of dissolved oxygen, temperature, specific conductivity, and discharge data. Watershed size ranges from 15 square kilometers to 168 square kilometers. Impervious surface cover ranges from 9% to 47% (Table 2.1). All sites are located on 3$^{rd}$ and 4$^{th}$ order creeks. The climate in Charlotte-Mecklenburg County is humid subtropical with mean annual precipitation of 43.3 inches and mean annual temperature of 61°F.

I used hourly dissolved oxygen, temperature, turbidity, and specific conductivity data from CMANN from January 2013 – March 2018. Data were collected using YSI EXO sondes. The data was quality controlled by CMSWS. I downloaded instantaneous discharge data at a 15-minute interval from the USGS National Water Information System (NWIS). I downloaded precipitation data from USGS rain gages across Mecklenburg County at a 5-minute interval. I calculated area-weighted precipitation for each watershed using the Thiessen polygon method (Thiessen, 1911).

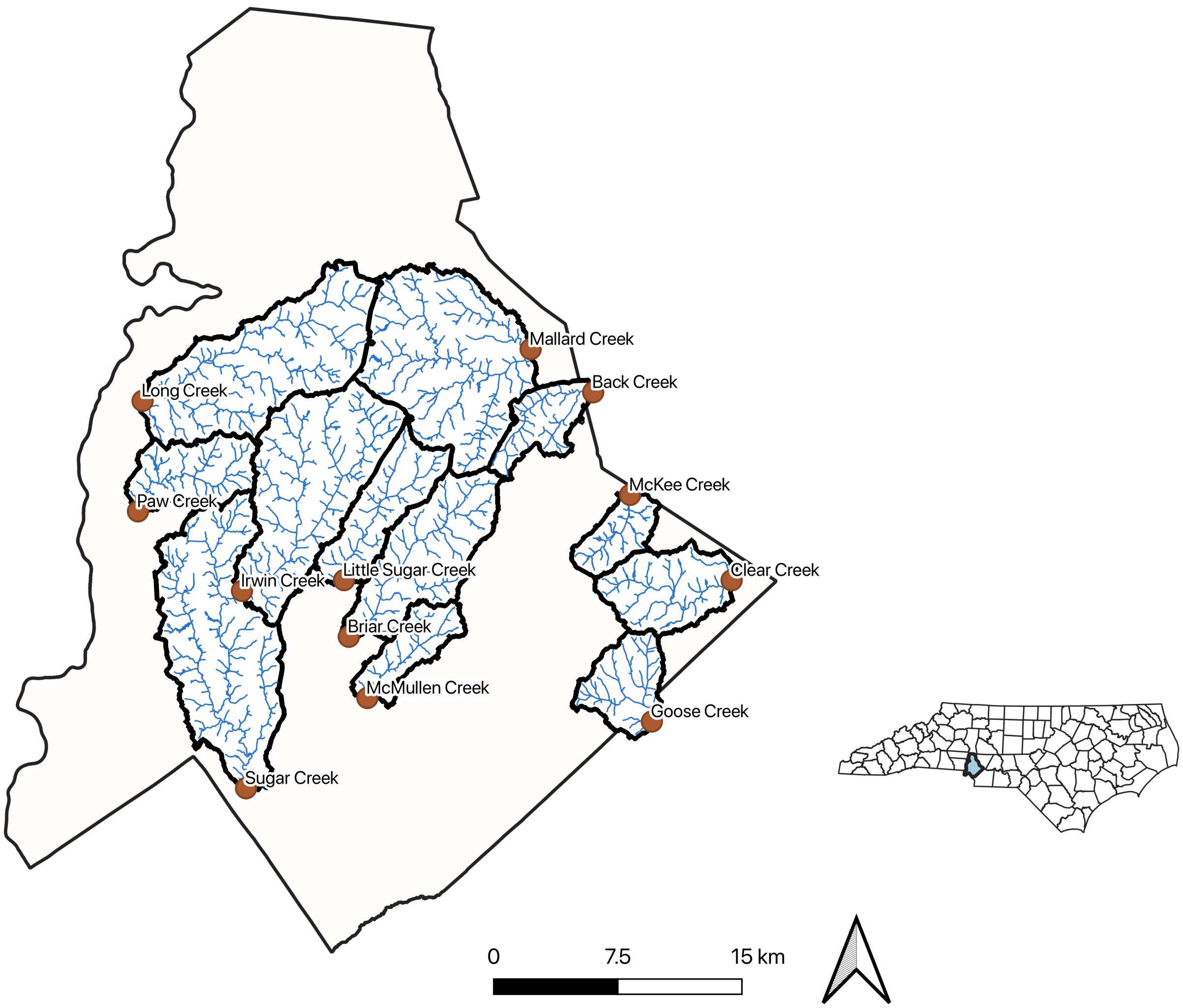


Figure 2.1. Map of study sites in Charlotte, Mecklenburg County, North Carolina.

Table 2.1. Site descriptions. TI is total imperviousness. UI is unmitigated imperviousness.

| Creek | Site Identifier | USGS ID | Watershed Area ($km^2$) | Channel Slope | Population Density ($/km^2$) | TI (%) | UI (%) | Tree Cover (%) |
|---|---|---|---|---|---|---|---|---|
| Clear Creek | CC | 212466000 | 32.3 | $3.96\times 10^{-3}$ | 184.8 | 8.8 | 8.8 | 59.4 |
| McKee Creek | MKC | 212430653 | 15.1 | $3.29\times 10^{-3}$ | 510.4 | 12.7 | 12.6 | 57.1 |
| Goose Creek | GC | 212467451 | 22.9 | $2.47\times 10^{-3}$ | 305.1 | 20.7 | 18.9 | 49.6 |
| Long Creek | LC | 214291555 | 82.2 | $1.00\times 10^{-5}$ | 507.8 | 21.9 | 20.7 | 46.2 |
| Paw Creek | PC | 214295600 | 27.2 | $1.00\times 10^{-5}$ | 584.2 | 23.5 | 20.9 | 49.4 |
| Mallard Creek | MC | 212414900 | 89.8 | $1.00\times 10^{-5}$ | 877.0 | 24.8 | 23.9 | 48.1 |
| Back Creek | BAC | 2124269 | 18.7 | $4.20\times 10^{-3}$ | 769.6 | 25.1 | 24.3 | 38.5 |
| McMullen Creek | MMC | 2146700 | 18.5 | $2.68\times 10^{-3}$ | 1058.9 | 26.3 | 25.4 | 56.6 |
| Briar Creek | BRC | 214645022 | 49.0 | $3.52\times 10^{-3}$ | 1430.7 | 28.2 | 25.8 | 52.1 |
| Sugar Creek | SC | 2146381 | 167.7 | $1.48\times 10^{-3}$ | 636.0 | 32.7 | 27.8 | 40.1 |
| Irwin Creek | IC | 2146300 | 78.9 | $1.63\times 10^{-3}$ | 828.7 | 33.9 | 33.8 | 39.7 |
| Little Sugar Creek | LSC | 2146409 | 31.4 | $1.00\times 10^{-5}$ | 1148.8 | 47.2 | 45.0 | 35.9 |

### 2.3.2. Watershed and Stream Characteristics

I delineated watershed drainage areas from a 1/9 arc-second digital elevation model (DEM) with the GRASS7 toolbox (GRASS Development Team, 2017). Stormwater pipes were burned into the DEM to ensure that water flow paths aligned with urban drainage networks. I used a 2012 land cover dataset from Charlotte-Mecklenburg County (http://maps.co.mecklenburg.nc.us/openmapping/). The land cover was derived from lidar point data at a 1-meter resolution and included tree, shrub, bare earth, water, and total impervious (TI) surfaces. Back Creek and Goose Creek watersheds did not have full coverage from the Mecklenburg County land cover dataset. For these watersheds, I mosaicked the Charlotte-Mecklenburg County land cover dataset with the 2013 National Land Cover Dataset (NLCD) to cover the missing land cover areas. The NLCD dataset has a resolution of 30-m; therefore, I resampled the NLCD dataset to a 1-m resolution then mosaicked the two rasters. Human population density in each watershed was calculated as an area-weighted population at the census block group scale.

The location and extent of wetlands were also acquired from Charlotte-Mecklenburg County. This dataset was derived from the National Wetland Inventory. The location and drainage areas of stormwater control measures were obtained from CMSWS. This dataset was last updated in 2017. Unmitigated imperviousness (UI) was calculated as the total impervious area that is untreated by stormwater control measures. I extracted channel slope from the National Hydrography Dataset High Resolution (NHDPlusHR) (https://www.usgs.gov/national-hydrography/nhdplus-high-resolution). I approximated bankfull channel width from USGS field measurements of wetted width during high flow. I additionally cross-referenced the width with hydraulic geometry relationships derived for urban streams in the Piedmont region (Doll et al., 2002).

Stream habitat assessments at each site were collected from the Mecklenburg County Land Use and Environmental Services Agency Water Quality Program (MCWQP). MCWQP conducts yearly (in late summer and early fall) habitat assessments along a 100-m reach near each CMANN site. MCWQP qualitatively assesses instream cover, epifaunal substrate, streambed substrate, sediment deposition, frequency of riffles, bank vegetative protection, bank stability, and vegetation buffer zone width. These variables were scored on a 0-20 scale. I created a summed scale of five instream habitat measures that represent habitat diversity: instream cover, epifaunal substrate, streambed substrate, sediment deposition, and frequency of riffles ($\alpha = 0.91$; Cronbach's $\alpha$ is a measure of internal consistency). Average habitat assessment scores, summarized across the study period, are listed in Appendix A.

### 2.3.3. Ecosystem Metabolism Modeling

I estimated daily mean rates of gross primary production and aerobic ecosystem respiration using a single-station open-channel approach. Open-channel metabolism estimation uses dissolved oxygen consumption and production as a proxy to derive GPP and ER (Odum, 1956). The estimation approach takes advantage of the way that GPP, ER, and air-water gas exchange rate (K600) affect dissolved oxygen concentration at different times of the day and in different directions. To model metabolism, I used Bayesian inverse modeling in the R package *streamMetabolizer* (Appling et al., 2018). The model seeks the most likely estimates of GPP, ER, and K600 that provide the best fit between modeled and measured $O_2$ data.

In *streamMetabolizer*, I estimated GPP, ER, and K600 by fitting the following model to dissolved oxygen data (Appling et al., 2018):

$$dO_{i,d}dt^{-1} = \left(\frac{GPP_d}{\bar{z}_{i,d}} \times \frac{PPFD_{i,d}}{\overline{PPFD}_d}\right) + \frac{ER_d}{\bar{z}_{i,d}} + f_{i,d}(K600_d)(Osat_{i,d} - O_{i,d}) \qquad (2.1)$$

where $O$ is the modeled dissolved oxygen concentration at time step $i$ and day $d$; $\bar{z}$ is average channel depth over the reach; $PPFD$ is photosynthetic photon flux density; $f_{i,d}(K600_d)$ is a function that models the gas exchange rate with $K600$, the gas exchange rate coefficient standardized to a common Schmit number; $Osat$ is the dissolved oxygen concentration at equilibrium. $GPP$, $ER$, and $K600$ are the parameters fitted by the model. I estimated average channel depth using piecewise linear model rating curves between discharge and water depth from USGS field measurements. In watersheds with multiple USGS discharge gaging stations, I combined discharge and water depth from all gaging stations to create a rating curve.

Within a Bayesian framework, *streamMetabolizer* integrates prior estimates of GPP, ER, and K600 to inform the model of uncertainty in the parameters. I used the *streamMetabolizer* default priors for GPP (mean: 3, SD: 6) and ER (mean: -7.1; SD: 7.1). I estimated K600 alongside GPP and ER. A setback of inverse modeling is equifinality—defined as many combinations of estimated parameters (GPP, ER, and K600) producing the same $O_2$ time series. *streamMetabolizer* reduces equifinality with the use of a Bayesian technique, partial pooling (Appling et al., 2018). Complete pooling uses the relationship between K600 and discharge (Q) to fix K600 on days with the same discharge. Bayesian partial pooling simultaneously estimates K600 with Markov Chain Monte Carlo (MCMC) estimation and the K ~ Q relationship. This method restrains outliers in K600 and therefore reduces equifinality (Appling et al., 2018). In the model, K600 was binned by discharge at six intervals of 0.2 natural log units of the observed range of Q at each site. I fit the model using MCMC in *rstan* (Stan Development Team, 2021). I used 1000 burn-in steps and 2000 saved steps. Further detail on this methods configuration, "b_Kb_oipi_tr_plrckm.stan", can be found in the *streamMetabolizer* documentation (http://usgs-r.github.io/streamMetabolizer/index.html). I ran the model with two years of data in a single model run. I removed any days with more than three hours of missing dissolved oxygen data, days with daily average discharge lower than the 5th percentile, and days with greater than 50% change in discharge as days with dynamic flow are not modeled well.

I performed model diagnostics to ensure that the model outputs were trustworthy. I visually assessed the relationship between ER and K600 to ensure that there was no significant relationship, which signals a problem with equifinality. Generally, I flagged models with ER ~ K600 relationships with an $R^2$ greater than 0.35. I removed all model outputs that signaled issues with equifinality. I removed all days with GPP less than 0.2, ER greater than -0.2, K600 greater than 75, $R^2$ between measured and modeled dissolved oxygen less than 0.6, rhat (a stan metric assessing model convergence) greater than 1.1, and effective sample size (n_eff) less than 100. Of the 27 potential CMANN sites, I successfully estimated stream metabolism at 12 sites. Fifteen sites showed strong equifinality, which is frequently observed in small streams with low GPP. Appendix B details the number of modeled days retained after diagnostics.

### 2.3.4. Light Model

Light at the stream surface was used to investigate relationships between light and GPP. I modeled photosynthetically active radiation (PAR) at the stream surface with the *StreamLight* R package (Savoy et al., 2021). *StreamLight* is a biophysical model that estimates PAR at the stream surface using total incoming shortwave radiation, stream canopy structure, and stream channel characteristics. I downloaded hourly total incoming shortwave radiation from the National Land

Data Assimilation System (NLDAS). I downloaded MODIS leaf area index (LAI) from the Application for Extracting and Exploring Analysis Ready Samples (AppEARS). Other input parameters include latitude, longitude, channel azimuth, channel width, bank height, bank slope, water level, tree height, canopy overhang, canopy overhang height, and leaf angle distribution. Channel width, bank height, and water level were estimated from USGS field measurements. Bank slope was set to 100 for all sites, which reflects steep slopes (Savoy et al., 2021). Tree height, canopy overhang, and canopy overhang height were estimated using the "extract_height()" function in *StreamLight*, which derives tree height with LiDAR estimates. In the most impervious watershed, Little Sugar Creek, I was unable to model PAR because LAI data are not available in highly urbanized landscapes. Instead, in Little Sugar Creek, I used incoming PAR (compared to PAR at the stream surface). Little Sugar Creek has little to no riparian cover; therefore, incoming PAR is a reasonable estimate of light reaching the stream surface.

### 2.3.5. Hydrology

To estimate the influence of discharge on seasonal patterns in stream metabolism, I calculated the seasonal mean unit discharge, time above the seasonal mean, and the number of storms with flow above the 80$^{th}$ percentile flow in each season. To explore the response of stream metabolism to storm events, I identified storm events with high turbidity. To define a flow event, I conducted hydrograph separation using a modified version of the constant line separation method (Bell et al., 2016; Hewlett & Hibbert, 1967). For each event, I calculated event precipitation as precipitation that occurred during an event and 1.5 hours prior to the event. Flow events with less than 1.5 mm of precipitation were considered as baseflow. I sought to identify events that influenced light attenuation and flow; therefore, I classified storm days as stormflow events with turbidity greater than the 80$^{th}$ percentile. Storm days started on the day of peak flow of the event and ended two days after the hydrograph subsided.

### 2.3.6. Statistical Analysis

All analyses were performed in the R language for statistical computing (R Core Team, 2013). I analyzed metabolism at the study sites between 2013 and 2018. The time series data had temporal gaps, which precluded the use of many time series analysis statistical techniques. To address the first research question, how environmental factors control metabolism, I calculated seasonal means of GPP, ER, PAR, temperature, and discharge and assessed the effect of environmental variables on GPP and ER. I used meteorological seasons where winter includes December, January, and February; spring includes March, April, and May; summer includes June, July, and August; and fall includes September, October, and November. Prior to calculating seasonal means of GPP and ER, I used linear interpolation to fill gaps in the time series. I filled missing seasonal temperature data with the mean of temperature in the same season of the prior and subsequent years. I performed all statistical methods on the absolute value of ER.

I built regression models from the general form of the Boltzmann-Arrhenius equation, which expresses the temperature sensitivity of biological processes (Arrhenius, 1889; Yvon-Durocher et al., 2012):

$$\ln GPP(T) = E_{GPP} \times \left(\frac{1}{kT_C} - \frac{1}{kT}\right) + \ln \mathrm{GPP}(\mathrm{T}_C) \tag{2.2}$$

$GPP(T)$ represents either GPP or ER, $E_{GPP}$ is the activation energy of the metabolic process, $k$ is the Boltzmann constant, and $T$ is the temperature. The equation describes the natural logarithm of GPP or ER as a function of standardized temperature, $(1/k\mathrm{T}_C - 1/k\mathrm{T})$, centered around $\mathrm{T}_C$ (15° C). The intercept of the model, $\ln\mathrm{GPP}(\mathrm{T}_C)$, represents the mean GPP or ER at $\mathrm{T}_C$. To test the effects of other known drivers of metabolism, I added the natural logarithms of the seasonal means of PAR and discharge as well as time above mean discharge and number of storms above the 80$^{th}$ percentile to the model as independent variables. I considered PAR, temperature, discharge, time above mean flow, and number of storms above the 80$^{th}$ percentile flow as predictors of GPP. I considered GPP, temperature, discharge, time above mean flow, and number of storms above the 80$^{th}$ percentile flow as predictors of ER. Fixed effects linear models were fit using ordinary least squares (OLS) estimation in the R package *fixest* (Bergé, 2018). Study site was the fixed effect in every model. I examined multicollinearity using variance inflation factors (VIF), and VIF above five was considered collinear. As expected, temperature and PAR were collinear; therefore, I ran the analysis with temperature and PAR in separate models. Due to the presence of serial correlation in time series data, I corrected the standard errors of the model using the Newey-West estimator (Newey & West, 1987).

I also assessed how watershed and stream characteristics influence seasonal mean GPP and ER. I used RDA to visually display the multivariate relationship between seasonal GPP and ER and site characteristics. I considered several watershed and stream characteristics as independent variables in the redundancy analysis, including unmitigated imperviousness (%), tree coverage (%), human population density (people km$^{-2}$), wetland coverage (%), channel width (m), channel slope, habitat diversity score, bank vegetation score, bank stability score, channel alteration score, and riparian vegetation score. When all variables were included in the model, the VIFs showed strong multicollinearity. I removed variables until VIF was below 10 for all variables, which is consistent with current recommendations for RDA (Borcard et al., 2018). I removed variables from the model based on known relationships between independent variables. For example, human population density and impervious surfaces in urban areas are known correlates. The final model included unmitigated imperviousness (UI), tree coverage (Tree), channel width (Width), channel slope (Slope), habitat diversity score (Hab), bank vegetation score (BV), bank stability score (BS), channel alteration score (ChanAlt), and riparian vegetation score (Rip). I performed partial redundancy analysis with seasonal means of GPP and ER as dependent variables. I performed RDAs in R package *vegan* (Oksanen et al., 2020). I conditioned the partial redundancy analysis on watershed area and year to control for variance in the response variables independently explained by changes in watershed area and trends in seasonal means over time. Redundancy analysis uses permutation to test the significance of the global model and each canonical axis (Borcard et al., 2018). I permuted the residuals of the full model (including conditional variables and covariates) and restricted the permutation using cyclic shifts to account for serial correlation between seasons at each site (Legendre et al., 2011; Simpson, 2020). RDAs were performed for summer and fall to match seasonal metabolism estimates with seasonal habitat assessment data. Models with global significance and at least one significant canonical axis were presented.

To assess the impact of storm events on stream metabolism, I used linear models of the Boltzmann-Arrhenius equation (Equation 2.2) (Arrhenius, 1889). To estimate changes in metabolism in response to storm events, I included stormflow and the interaction between temperature and stormflow as independent variables in the Boltzmann-Arrhenius equation:

$$\ln GPP(T) = E_{GPP} \times \left(\frac{1}{kT_C} - \frac{1}{kT}\right) + \ln GPP(\mathrm{T_C}) + Q + E_i(T \times Q) \tag{2.3}$$

$Q$ is a binary indicator of storm events. The parameter of the interaction term, $E_i$, represents potential changes in the slope, or activation energy, during storm events. I corrected the standard errors of the model using Newey West estimation (Newey & West, 1987). I initially fit four models to the daily data at each site: 1) the full model, 2) a model including temperature and discharge, 3) a model including temperature and the interaction effect, and 4) a model including temperature. I used the model F-statistic to determine whether model parameters were significantly different from zero. If the F-statistic for all models was insignificant, I did not fit any model to the site data. I used Wald tests to select the best fit model of the nested models. For each site, I used the fitted model to estimate the change in GPP and ER during storm events at 5°C, 10°C, 15°C, 20°C, and 25°C. This analysis was performed with the R packages *stats* and *lmtest* (Zeileis & Hothorn, 2002).

I used three metrics to describe the change in metabolism during storm events. The difference in GPP and ER between baseflow and stormflow is described as a magnitude of change ($M$):

$$M = GPP_{stormflow} - GPP_{baseflow} \tag{2.4}$$

I also described the change in metabolism between baseflow and stormflow using the natural logarithm of the difference ($M$) to account for changes in baseflow metabolic rates between sites. Finally, I calculated the absolute value of the difference in slopes (activation energy) of the Boltzmann-Arrhenius equation at baseflow and stormflow. A positive $M$ indicates that metabolic rates increased during stormflow. A negative $M$ indicates that metabolic rates decreased during stormflow. A large slope change indicates a large difference between baseflow and stormflow activation energy. A small slope change indicates a small difference between baseflow and stormflow activation energy. I performed partial RDAs to assess how the change in GPP and ER following storms varies with watershed and stream characteristics. The RDA included unmitigated imperviousness (UI), tree coverage (Tree), channel slope (Slope), habitat diversity score (Hab), bank vegetation score (BV), bank stability score (BS), and riparian vegetation score (Rip) as independent variables. I did not include channel width and channel alteration in an effort to keep the model parsimonious as the sample size is small (n = 12). I conditioned the RDA on watershed area and performed RDA at each temperature level (5°C, 10°C, 15°C, 20°C, and 25°C). I permuted the residuals of the full model to assess the significance of the model and the canonical axes.

To analyze the variability of the GPP and ER time series, I calculated the proportional variability (PV) index with daily metabolic rates at each site. The PV index determines the average proportional variability among every combination of values in a time series (Fernández-Martínez et al., 2018):

$$PV = \frac{2\sum z}{n(n-1)} \tag{2.5}$$

$z$ is calculated as

$$z = 1 - \frac{\min(z_i, z_j)}{\max(z_i, z_j)} \tag{2.6}$$

$n$ represents the sample size, and $z$ represents the pairwise comparisons between every combination of values in the time series. PV improves alternative measures of variation, such as the coefficient of variation (CV), because it is less dependent on and sensitive to the time series mean. PV is particularly useful when time series means are between 0 and 1, which can lead to biases in CV. I used Pearson correlation tests to assess the association between unmitigated impervious surface cover and PV.

## 2.4. Results

### 2.4.1. Two Distinct Productivity Regimes

I observed strong seasonal patterns in GPP and ER across all sites; however, there were differences in the timing of peaks and duration of high metabolic rates (Fig. 2.2; Fig. 2.3). GPP tended to be highest in spring; however, a couple of sites deviated from this pattern (LSC and BRC) (Table 2.2). I observed the highest seasonal mean GPP in Briar Creek (BRC), and GPP was, on average, highest in summer (Table 2.2). Some sites exhibited punctuated peaks and subsequent steep declines in GPP (MMC, BAC, MKC, PC, GC, and CC), while others displayed higher spring or summer peaks with more gradual declines (LSC, IC, BRC, LC, and MC) (Fig. 2.2). Generally, sites that exhibited a distinct peak associated with steep summer declines have watershed areas less than ~30 km$^2$ (MKC, MMC, BAC, GC, and PC) and lower unmitigated impervious cover than other sites (9%-25%). Generally, sites that exhibited gradual declines in GPP have watershed areas greater than ~30 km$^2$ (LSC, IC, BRC, LC, and MC), mean summer GPP similar to or higher than mean spring GPP (LSC, IC, BRC, LC, and MC), and higher unmitigated impervious cover than other sites (19%-45%) (Table 2.1; Table 2.2). Mean seasonal ER tended to be highest in the summer or fall (Table 2.2). Seasonality was evident in ER at all sites, and ER tended to be highest when temperature was highest in the summer (Fig. 2.3).

The results show that nearly all study creeks were net heterotrophic (Fig. 2.4). Little Sugar Creek (LSC) was an exception to this pattern. Little Sugar Creek showed a balance between GPP and ER across all seasons (Table 2.2). Little Sugar Creek is the most impervious watershed (45%) and is located in downtown Charlotte. This stream has the lowest riparian vegetation coverage and consequently high light reaching the stream surface (Appendix A).

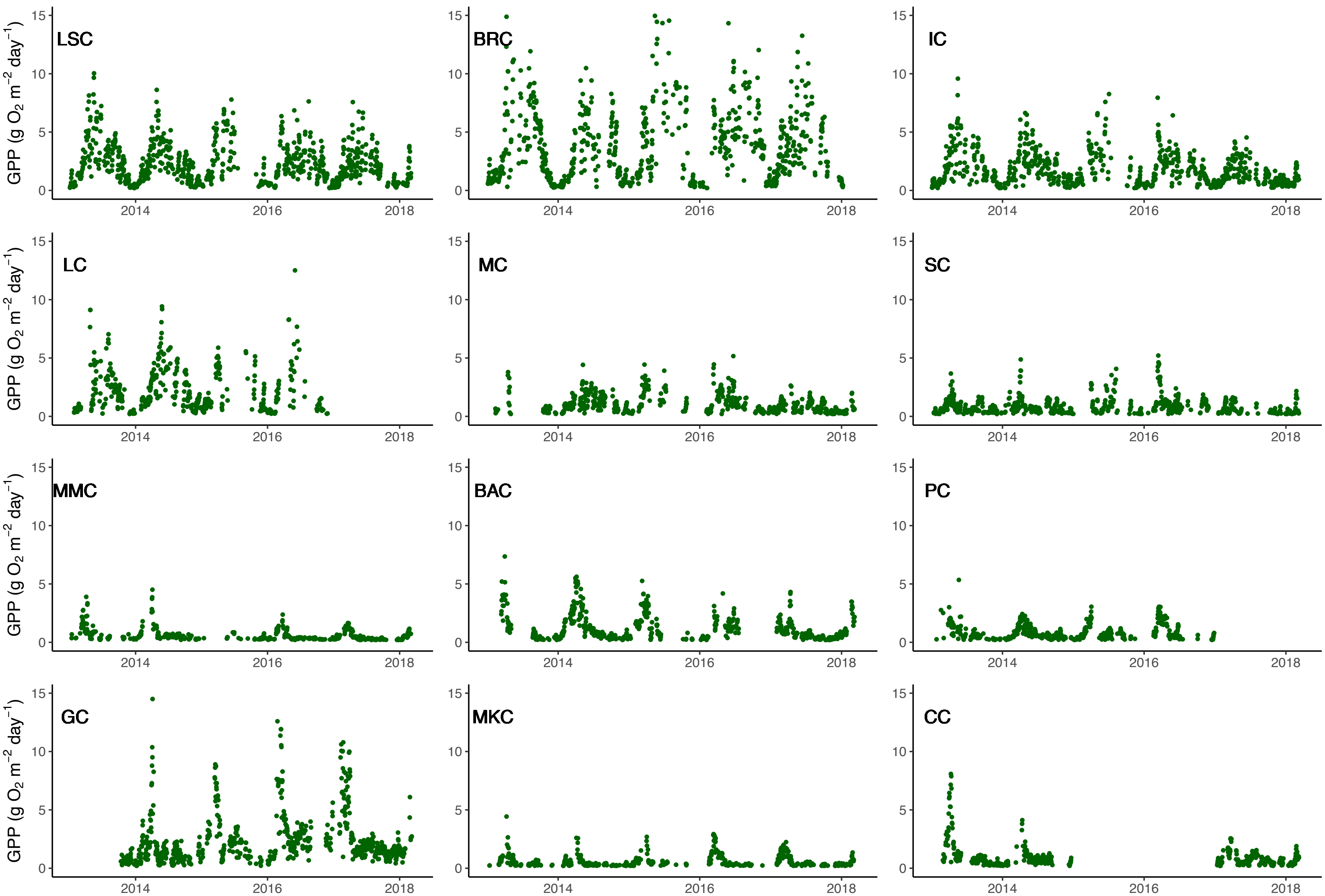


Figure 2.2. Time series plots of GPP at each site.

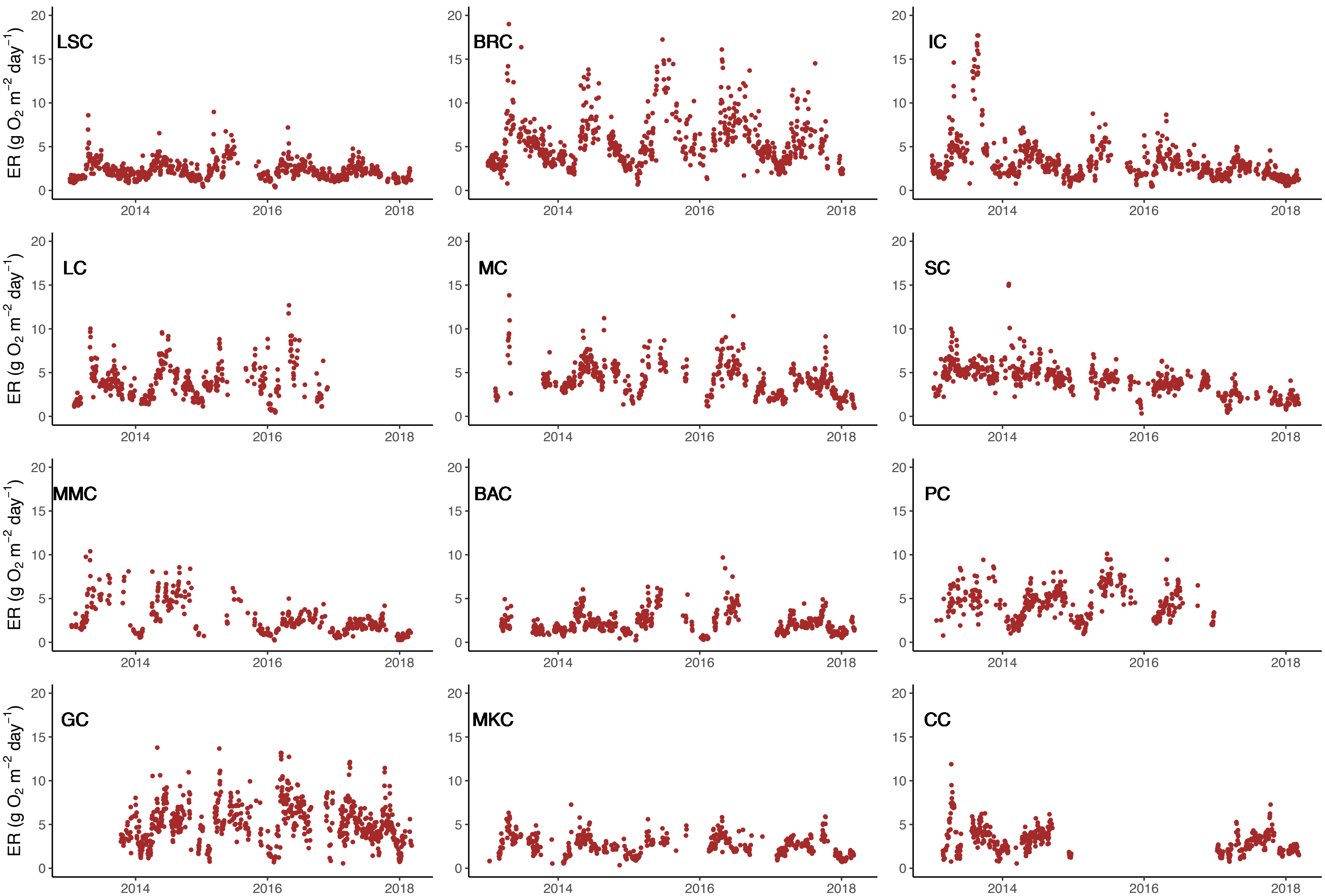


Figure 2.3. Time series plots of ER at each site.

Table 2.2. Average value of the seasonal mean GPP and ER summarized across the study period. The standard error is in parentheses. Bolded text indicates the highest mean seasonal metabolic rate.

| Site Identifier | GPP | | | | ER | | | |
|---|---|---|---|---|---|---|---|---|
| | Winter | Spring | Summer | Fall | Winter | Spring | Summer | Fall |
| CC | 0.77 (0.02) | **1.4 (0.16)** | 1.02 (0.11) | 0.68 (0.03) | 1.98 (0.15) | 2.64 (0.21) | 3.22 (0.27) | **3.59 (0.17)** |
| MKC | 0.61 (0.03) | **1.05 (0.09)** | 0.55 (0.02) | 0.63[a] | 1.73 (0.2) | 2.88 (0.33) | 2.96 (0.31) | **3.39[a]** |
| GC | 2.03 (0.43) | **2.98 (0.23)** | 1.73 (0.21) | 1.26 (0.24) | 3.51 (0.43) | 6.25 (0.59) | 6.28 (0.43) | **6.88 (0.57)** |
| LC | 1.06 (0.1) | **3.4 (0.33)** | 3.25 (0.33) | 1.6 (0.19) | 2.42 (0.13) | 4.56 (0.48) | **4.82 (0.27)** | 3.91 (0.25) |
| PC | 0.87 (0.13) | **1.42 (0.06)** | 0.72 (0.04) | 0.69 (0.04) | 3.95 (0.53) | 4.6 (0.51) | **5.61 (0.6)** | **5.61 (0.55)** |
| MC | 0.87 (0.07) | **1.69 (0.14)** | 1.52 (0.09) | 0.91 (0.06) | 2.81 (0.18) | 5.19 (0.52) | **5.39 (0.28)** | 4.84 (0.25) |
| BAC | 1.1 (0.07) | **2.22 (0.15)** | 0.97 (0.12) | 0.62 (0.03) | 1.48 (0.08) | 2.68 (0.37) | **2.74 (0.43)** | 2.46 (0.58) |
| MMC | 0.74 (0.07) | **1.11 (0.15)** | 0.69 (0.06) | 0.62 (0.01) | 1.45 (0.48) | 2.97 (0.6) | 5.35 (0.08) | **6.89 (0.88)** |
| BRC | 1.26 (0.1) | 4.62 (0.21) | **6.21 (0.64)** | 3.42 (0.27) | 3.12 (0.31) | 6.1 (0.41) | **8.01 (0.66)** | 5.08 (0.25) |
| SC | 0.76 (0.05) | **1.06 (0.09)** | 0.93 (0.13) | 0.84 (0.07) | 2.95 (0.44) | 3.68 (0.34) | **3.76 (0.34)** | 3.59 (0.28) |
| IC | 0.94 (0.07) | **2.61 (0.21)** | 2.46 (0.2) | 1.32 (0.09) | 1.98 (0.18) | 3.58 (0.37) | **3.92 (0.43)** | 2.7 (0.32) |
| LSC | 0.97 (0.06) | 2.96 (0.24) | **3.37 (0.27)** | 1.83 (0.17) | 1.58 (0.11) | **2.81 (0.21)** | **2.79 (0.17)** | 1.92 (0.13) |

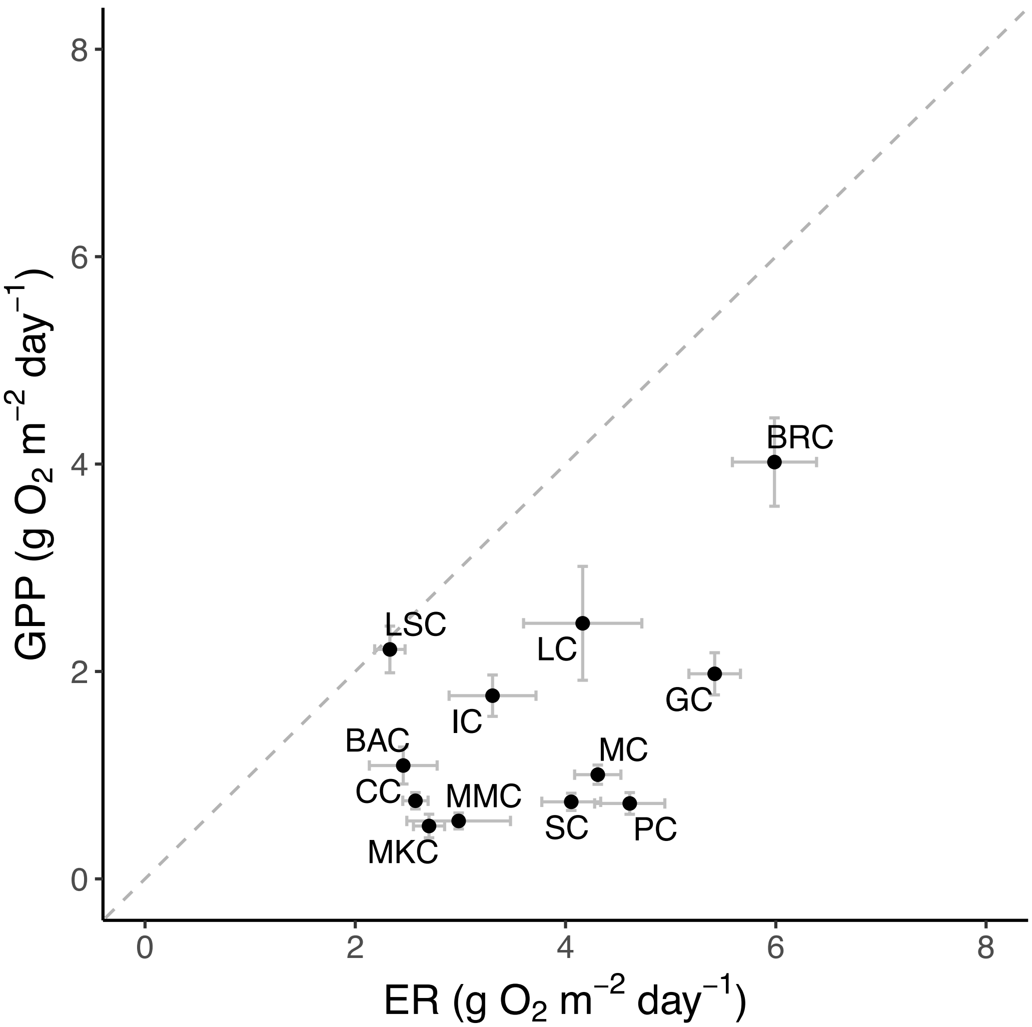


Figure 2.4. Mean GPP and ER over the study period (2013-2018). Grey error bars represent the standard error. The dotted grey line represents the 1:1 relationship between GPP and ER.

### 2.4.2. Light and Temperature Drive Seasonal Patterns in Metabolism

The results show that light and temperature regimes drive seasonal patterns in GPP and ER (Table 2.3). Mean seasonal PAR and temperature were positively associated with GPP ($\beta = 0.94, p < 0.001$; $\beta = 0.25, p < 0.001$, respectively; Table 2.3). Mean seasonal unit discharge was weakly positively associated with GPP, although this association was not significant at an alpha of 0.05 ($\beta = 0.09, p = 0.10$; $\beta = 0.13, p = 0.06$; Table 2.3). GPP was not associated with time above mean seasonal discharge ($\beta = -0.16, p = 0.84$; $\beta = 0.49, p = 0.57$; Table 2.3). Similarly, GPP was not associated with the number of storms above the 80th percentile of discharge ($\beta = 0.002, p = 0.86$; $\beta = 0.01, p = 0.28$; Table 2.3). ER was positively associated with temperature and GPP ($\beta = 0.25, p =< 0.001$; $\beta = 0.19, p < 0.001$, respectively, Table 2.3). ER was not significantly associated with mean seasonal unit discharge ($\beta = -0.01,\ p = 0.67$, Table 2.3), time above mean seasonal discharge ($\beta = 0.29,\ p = 0.73$, Table 2.3), and the number of storms above the 80th percentile of discharge ($\beta = -5.42 \times 10^{-6},\ p = 0.99$, Table 2.3)[1].

[1] Fixed effects models assume an equal slope parameter across sites. To be sure that the discharge effect was not significant across all sites, I ran multiple regressions with PAR, temperature, discharge, time above seasonal mean flow, and number of storms above the 80th percentile flow for each site. Two sites showed a significant relationship with discharge metrics. IC showed a negative association between time above mean flow and GPP. MKC showed a positive association between time above mean flow and GPP. MC showed a positive association between number of storms above the 80th percentile and ER. GC showed a positive association between time above mean flow and ER, and MMC showed a negative association between time above mean flow and ER. All other sites displayed no significant influence of flow related metrics on metabolism. This analysis revealed a few inconsistent relationships between seasonal metabolism and flow metrics. This additional analysis allowed the researchers to be more certain that flow metrics are largely unrelated to metabolism at the seasonal time scale.

Table 2.3. Coefficients and statistics of fixed effects models using environmental variables to predict seasonal mean GPP and ER.

| | log(GPP) | | | log(GPP) | | | log(ER) | | |
|---|---|---|---|---|---|---|---|---|---|
| | $\beta$ | SE | $p$ | $\beta$ | SE | $p$ | $\beta$ | SE | $p$ |
| log(PAR) | 0.94 | 0.12 | <0.001 | - | - | - | - | - | - |
| Standardized Temperature | - | - | - | 0.25 | 0.07 | <0.001 | 0.25 | 0.03 | <0.001 |
| log(Q) | 0.09 | 0.05 | 0.10 | 0.13 | 0.07 | 0.06 | -0.01 | 0.03 | 0.67 |
| Time above mean Q | -0.16 | 0.78 | 0.84 | 0.49 | 0.86 | 0.57 | 0.29 | 0.84 | 0.73 |
| No. storms above q.80 | $1.50\times10^{-3}$ | 0.01 | 0.86 | 0.01 | 0.01 | 0.28 | $-5.42\times10^{-6}$ | 0.01 | 0.99 |
| log(GPP) | - | - | - | - | - | - | 0.19 | 0.03 | <0.001 |
| S.E. type | Newey-West (L=8) | | | Newey-West (L=8) | | | Newey-West (L=8) | | |
| Observations | 250 | | | 250 | | | 250 | | |
| AIC | 376.06 | | | 439.87 | | | 144.27 | | |
| RMSE | 0.48 | | | 0.55 | | | 0.30 | | |
| Within $R^2$ | 0.33 | | | 0.13 | | | 0.43 | | |
| F-test | 10.21 | | | 6.73 | | | 8.09 | | |
| F-test, p-value | <0.001 | | | 0.0012 | | | <0.001 | | |

### 2.4.3. Elevated GPP Associated with Urbanization

The results show that watershed and stream metrics of urbanization were positively associated with GPP in summer and fall (Fig. 2.5). The variables used in the RDA explained 53% of the variation in seasonal mean GPP and ER in the summer and 51% in the fall. In the summer model, the global model and axis 1 significantly explained variation in GPP and ER ( $F_{global} = 7.15,\ p_{global} < 0.001,\ F_{axis1} = 56.74,\ p_{axis1} < 0.001,\ F_{axis2} = 20.19,\ p_{axis2} = 0.25$ ). In the fall model, the global model and the first two axes significantly explained variation in GPP and ER ($F = 6.36, p < 0.001,\ F_{axis1} = 41.17,\ p_{axis1} < 0.001,\ F_{axis2} = 27.23,\ p_{axis2} = 0.04$). The results were similar in the summer and fall. When controlling for watershed area, GPP was strongly positively correlated with unmitigated imperviousness, bank stability, and channel alteration (Fig. 2.5). In urbanized watersheds, stream banks are often stabilized with rip rap to prevent erosion, and urban channels are often altered through channelization and the colocation of other urban infrastructures like pipes and bridges. Likewise, GPP was negatively correlated with riparian vegetation, tree cover in the watershed, and bank vegetation (Fig. 2.5). Compared to GPP, ER was not strongly associated with urbanization metrics (Fig. 2.5). ER was negatively correlated with channel width across seasons; however, the shorter arrow shows that channel width is less important than the other variables. ER was strongly negatively correlated with stream habitat diversity. The correlation between ER and tree cover was stronger in the fall (Fig. 2.5).

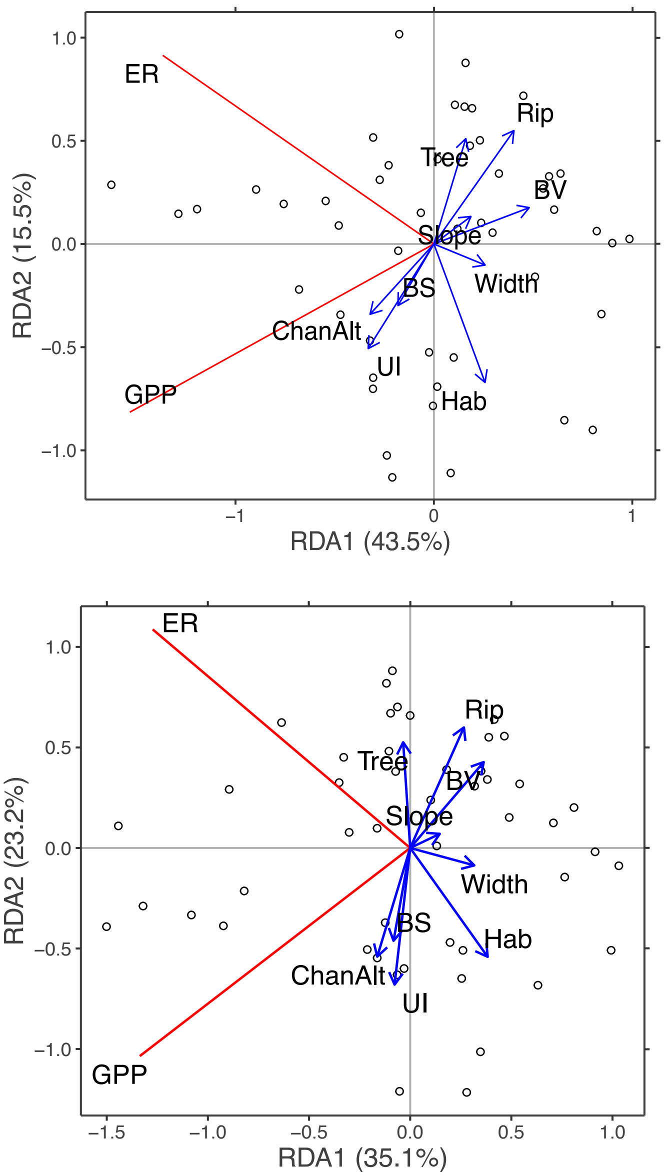


Figure 2.5. Correlation triplot of the RDA analysis using watershed and stream characteristics as predictor variables and GPP and ER as response variables. Summer (top), and fall (bottom) RDAs are shown. Each point represents a site for one year. There are pseudo-replicates for each site representing the yearly data over the study period (2013-2018). *Tree* is tree cover. *Rip* is the riparian vegetation score. *BV* is the bank vegetation score. *Slope* is the channel slope. *Width* is the channel width. *Hab* is the habitat diversity score. *BS* is the bank stability score. *UI* is unmitigated imperviousness. *ChanAlt* is channel alteration.

### 2.4.4. Summer Storms Reduced GPP; ER Resistant to Storms

When controlling for temperature, stormflows were associated with a decrease in GPP in six sites; moreover, the reduction was most pronounced on warmer days (Fig. 2.6; Appendix C, Table C.1). Four sites exhibited a significant interaction between temperature and discharge, highlighting the stronger storm effect on warmer days (SC, LSC, BRC, and MC) (Fig. 2.6; Appendix C, Table C.1). Seven sites exhibited a positive relationship between GPP and temperature (SC, LSC, IC, BRC, LC, MC, and CC), while the other sites showed no association between GPP and temperature (Fig. 2.6; Appendix C, Table C.1). ER was strongly positively associated with temperature across all sites (Fig. 2.7; Appendix C, Table C.2). In most sites, stormflows were not significantly associated with ER (Fig. 2.7; Appendix C, Table C.2). In a few sites, stormflows were associated with a small increase in ER on cooler days (LSC, MMC, BAC, and MKC) (Fig. 2.7; Appendix C, Table C.2).

The magnitude of change in GPP during storms varied along an urbanization gradient (Fig. 2.8). The redundancy analyses at 5°C, 10°C, 15°C, and 20°C were statistically insignificant; therefore, I only present the RDA at 25°C which represents the change in GPP during warmer storms. The variables used in the RDA explained 74% of the variation in the response metrics representing the change in GPP during stormflows (Fig. 2.8). The global model and first axis were significant ( $F_{global} = 13.8$, $p_{global} = 0.003$; $F_{axis1} = 163.12$, $p_{axis1} = 0.006$; $F_{axis2} = 61.42$, $p_{axis2} = 0.13$). When controlling for watershed area, the magnitude of change in GPP during storms on warmer days was negatively associated with unmitigated imperviousness and stream bank stability (Fig. 2.8). That is, the sites least resistant to stormflows had more impervious cover and relatively stable banks. Sites with greater riparian and bank vegetation exhibited a smaller magnitude of change in GPP than others (Fig. 2.8). All metrics of GPP change with stormflow show similar patterns with independent variables (Fig. 2.8).

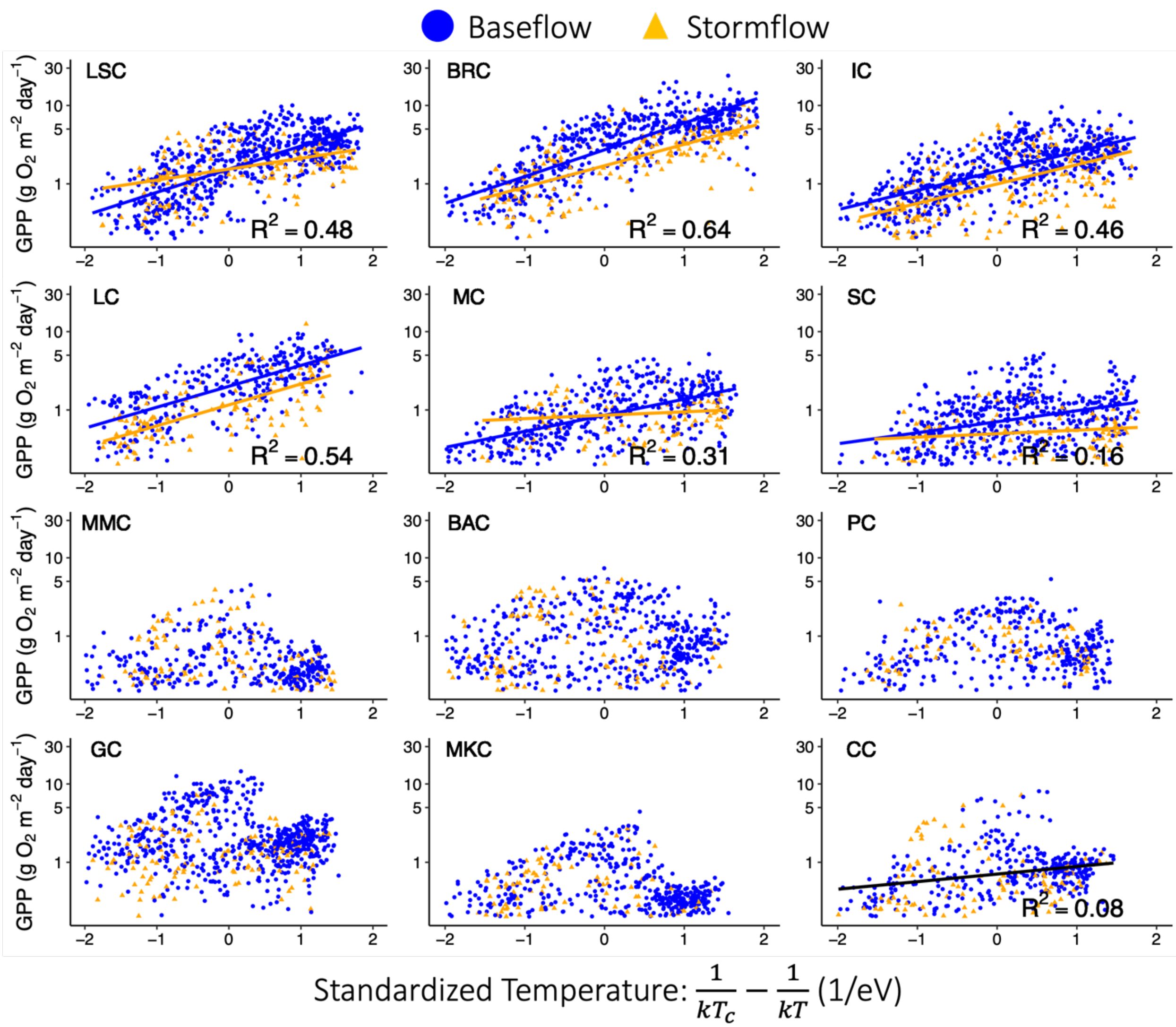


Figure 2.6. Arrhenius plots of GPP at all study sites. When storm events were a significant factor, regression lines were plotted separately for baseflow and stormflow. Sites are labeled in the top left corner of each panel.

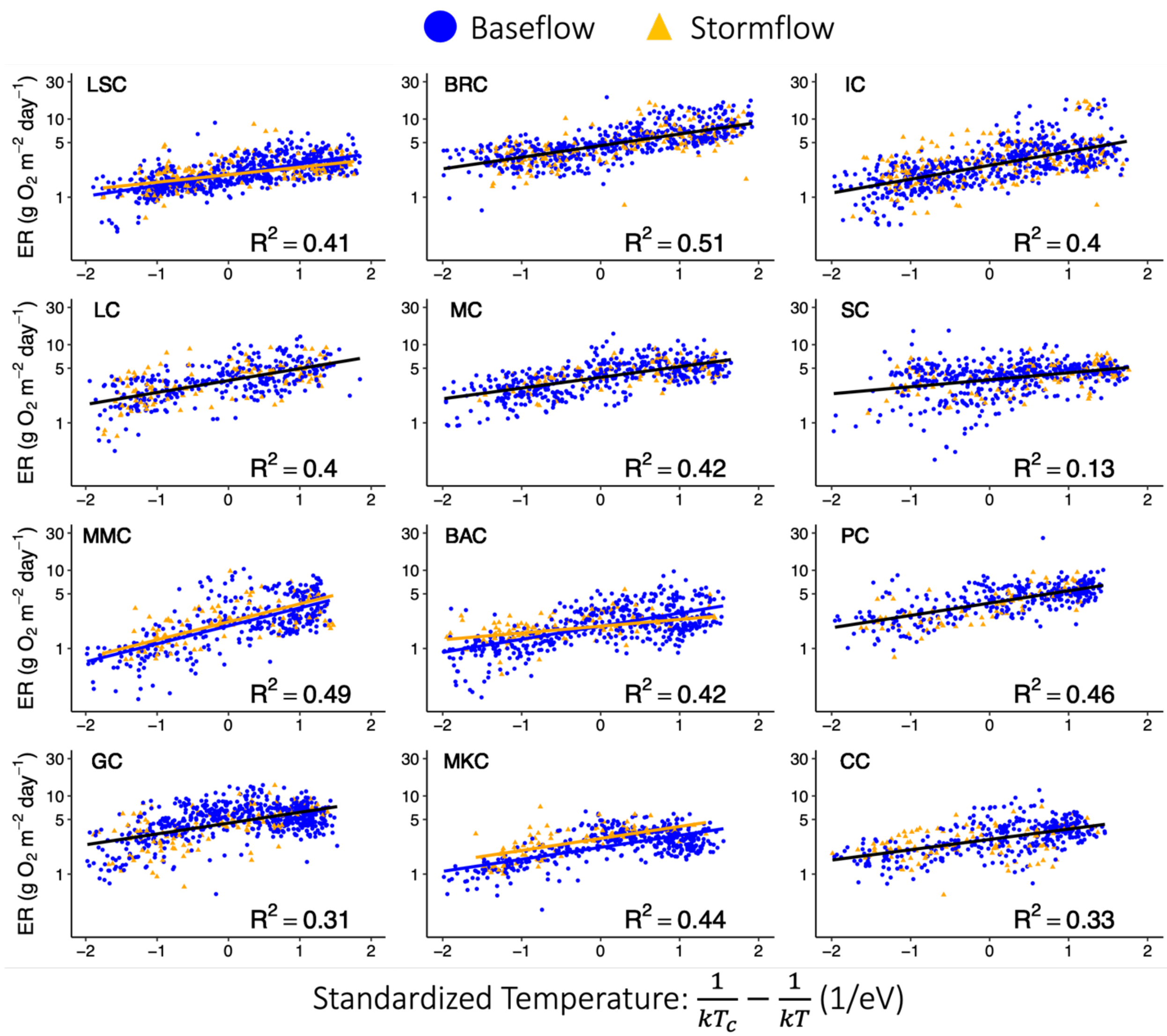


Figure 2.7. Arrhenius plots of ER at all study sites. When storm events were a significant factor, regression lines were plotted separately for baseflow and stormflow. Sites are labeled in the top left corner of each panel.

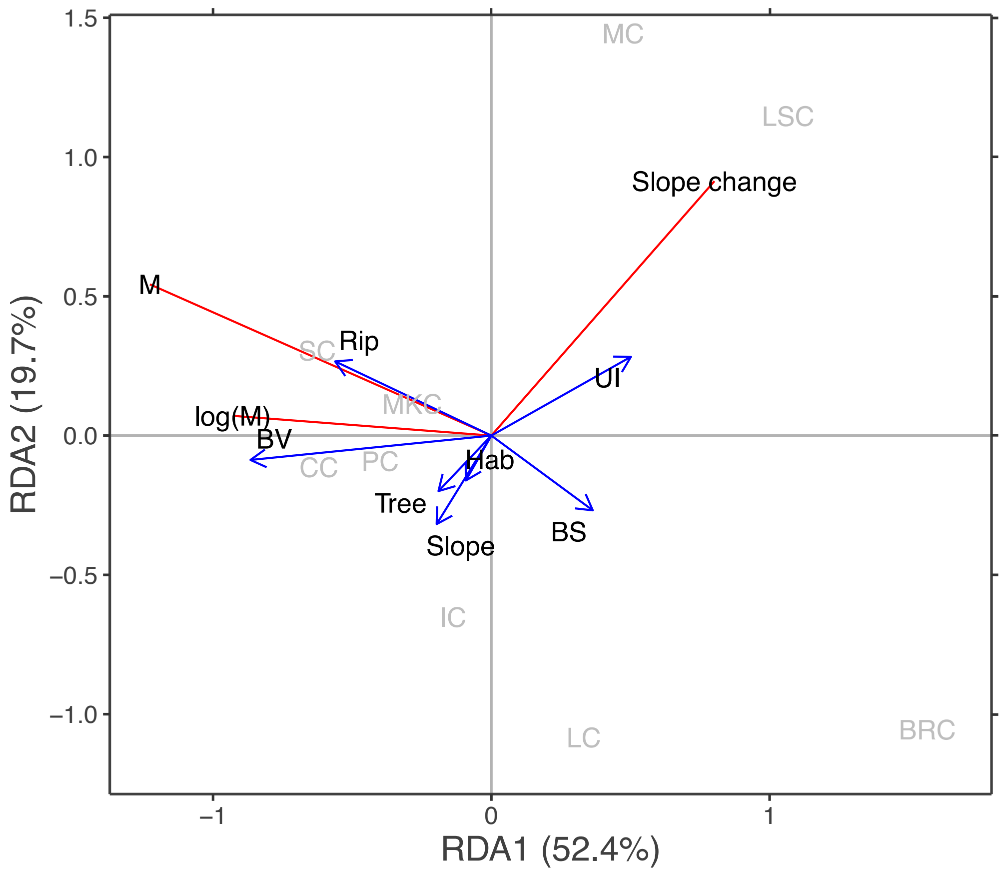

Figure 2.8. Correlation triplot of the RDA analysis using watershed and stream characteristics as predictor variables and metrics of magnitude of change in GPP during and following storms as response variables. The RDA was performed with estimates of magnitude of change between baseflow and stormflow at 25°C. Sites are labeled in grey. Sites that had no magnitude of change in GPP with stormflow overlapped near the origin on the triplot. We jittered each site enough so that the reader can see the label for each site. Still, we were unable to fit all sites on the plot and accurately represent the site loadings. Therefore, MMC, BAC, and GC were not included in the plot. *Tree* is tree cover. *Rip* is the riparian vegetation score. *BV* is the bank vegetation score. *Slope* is the channel slope. *Hab* is the habitat diversity score. *BS* is the bank stability score. *UI* is unmitigated imperviousness.

### 2.4.5. Imperviousness Associated with Increased Variability in GPP, but not Significant

For GPP, the PV index increased with unmitigated imperviousness; however, Pearson's correlation coefficient was not significant at an alpha of 0.05 ($r = 0.44, p = 0.15$; Fig. 2.9). Figure 2.9 shows that variability in daily GPP increased with unmitigated imperviousness, but the relationship is merely suggestive. For ER, the PV index was not associated with unmitigated imperviousness ($r = -0.05, p = 0.89$; Fig. 2.9). GPP tended to be more variable than ER over the study period (Fig. 2.9).

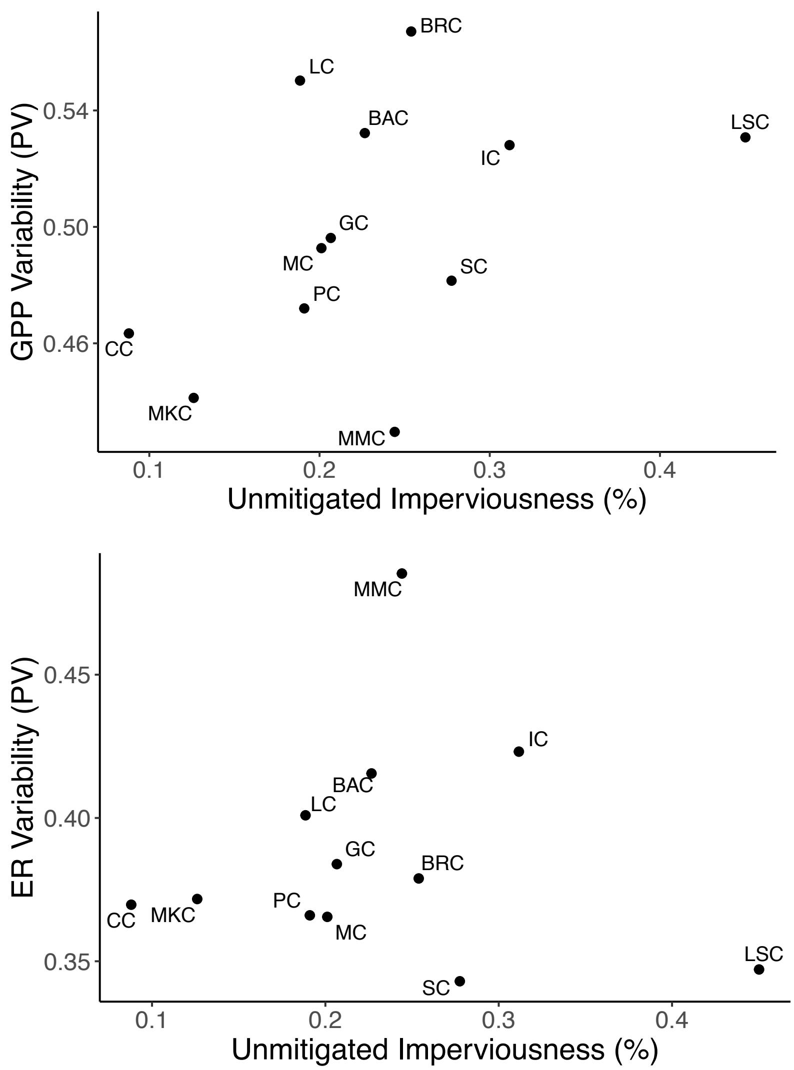


Figure 2.9. The proportional variability (PV) index vs. unmitigated impervious cover. The top graph shows the relationship for GPP. The bottom graph shows the relationship for ER.

## 2.5. Discussion

The findings demonstrate distinct seasonal patterns in GPP and ER across 12 urban streams (Figs. 2.2 and 2.3). I found that seasonal patterns in metabolism are primarily driven by temperature and light regimes (Table 2.3). I found little influence of hydrologic disturbances at the seasonal time scale; however, storm events introduce short-term decreases in GPP in warmer months while ER was relatively resistant to events (Figs. 2.6 and 2.7). Mean seasonal GPP was positively associated with urbanization metrics at the watershed and stream scale (Fig. 2.5). Additionally, GPP was more variable over time than ER (Fig. 2.9).

I observed two distinct patterns of timing and magnitude of productivity across the study sites (Fig. 2.2). Half of the sites showed relatively high spring peaks and productive summers. The other half showed punctuated spring peaks in GPP and relatively unproductive summers. The productivity regimes observed in this study align with predominant productivity regimes observed across 47 rivers in the United States (Savoy et al., 2019). Savoy et al. (2019) associated punctuated spring peaks in GPP with smaller watersheds and lower discharge. These watersheds often had short periods of optimal conditions for GPP during the spring when temperature was high, and canopy cover was low. Rivers with high canopy cover tend to exhibit sharp declines in GPP as light at the stream surface declines (Roberts et al., 2007). Conversely, rivers that exhibited productive summers were associated with larger watersheds and higher discharge (Savoy et al., 2019). These rivers peaked in the summer, coinciding with light and temperature peaks (Savoy et al., 2019). Notably, my study sites have substantially smaller watershed areas than the rivers exhibiting highly productive summers identified in Savoy et al. (2019). My study sites with highly productive summers range from 23-90 $km^2$ (Table 2.1) which is notably smaller than the mean watershed area of $1840 \pm 3248$ $km^2$ associated with summer productivity regimes (Savoy et al., 2019). I suggest land use change and riparian vegetation removal associated with urbanization shift these small urban creeks towards high summer productivity regimes that are normally more characteristic of larger rivers and desert streams (Koenig et al., 2019). The presence of two distinct productivity regimes across 12 urban creeks suggests that there may be a threshold of urban development beyond which creeks shift into high summer productivity regimes. Future studies can elucidate this pattern with longer time series data or reference sites (to compare with urbanized sites). Importantly, high productivity during warm temperatures in low gradient channels can leave these creeks more susceptible to eutrophication and hypoxia.

While ER displayed seasonal patterns, I did not observe distinctive regimes across sites (Fig. 2.3). ER was strongly associated with temperature at the seasonal and daily scale (Table 2.3; Fig. 2.7). Seasonal highs in ER were asynchronous with highs in GPP—which generally displayed the highest mean rates in the spring. ER tended to have the highest seasonal means in the summer when temperature tends to peak (Table 2.2). Though, notably, the sites that do peak in the fall (CC, MKC, GC, PC, and MMC) are located in watersheds with greater than ~50% tree cover (Table 2.1). This descriptive result, along with the association between tree cover and ER in fall, suggests that ER is associated with allochthonous organic matter. Overall, ER was less variable over time than GPP (Fig. 2.9), reflecting that ER occurs in the more protected hyporheic zone (Boulton et al., 1998). The hyporheic zone may be less susceptible to changes in the terrestrial landscape and hydrological disturbances than the stream bed.

Although urbanization is associated with increases in GPP, most of the study sites were net heterotrophic (Fig. 2.4). Urban streams were previously hypothesized to be net autotrophic because of high light availability as riparian vegetation is removed (Bernot et al., 2010; Finlay, 2011); however, several recent students using high temporal resolution data have demonstrated that urban streams, like their forested counterparts, exhibit net heterotrophy (Blaszczak et al., 2018; Larsen & Harvey, 2017; Smith & Kaushal, 2015). Little Sugar Creek (LSC) is a notable exception to this pattern. Little Sugar Creek is the most urbanized watershed in this study and shows a balance between mean annual GPP and ER (Fig. 2.4). Little Sugar Creek is located in downtown Charlotte (45 % unmitigated imperviousness) and has the lowest riparian vegetation score. Likewise, channelization can limit instream habitat and make this creek more susceptible to storms (Fig. 2.6). The balance in annual means of GPP and ER suggests that ER is largely driven by autotrophic respiration. Although, notably, GPP is decoupled from ER during storms, as evident in the decrease in GPP and stability of ER to storm events in LSC (Fig. 2.6). LSC likely represents a near endmember of metabolic regimes in urban creeks. Metabolic regimes in creeks with highly impervious watersheds are tightly coupled with light, temperature, and elevated nutrients, which drives these creeks towards net autotrophic conditions (Fuß et al., 2017; Kaushal et al., 2014). My study highlights this condition in the most impervious urban watershed, LSC; however, I show considerable heterogeneity in metabolic regimes across urban watersheds, highlighting a broad range of possibilities in the metabolic response to urbanization.

I hypothesized that flashy hydrology in the studied urban creeks (Bell et al., 2016) would decouple metabolic regimes from temperature and light regimes. However, the results show that the distinct seasonal patterns observed in the study sites are correlated with light and temperature and unassociated with stream hydrology (Table 2.3). PAR explained more variation in GPP than temperature. ER had a strong relationship with temperature and was associated with GPP. Additionally, the magnitude of mean seasonal GPP was positively associated with an urbanization gradient and negatively associated with vegetation that shades the stream (Fig. 2.5). The results suggest that urbanization is associated with elevated GPP through increased light to the stream. I show that local controls on light, including riparian and bank vegetation, are negatively associated with GPP and likely constrain GPP. Although not measured in this study, it is also important to note that nutrient inputs to urban streams have also been linked to increased productivity in urban streams (Fuß et al., 2017). Increased nutrients and light in urban streams can play a role in elevating GPP along urbanization gradients. Conversely, seasonal mean ER was unassociated with urbanization metrics. While ER is driven by GPP, the results show substantial heterotrophic respiration in the studied streams reflected in the imbalance between GPP and ER across sites. This suggests that urbanization has a stronger impact on GPP than ER. Previous studies have shown that ER is concentrated in the hyporheic zone, which could buffer heterotrophs from urbanization impacts. Additionally, GPP is known to be more responsive to light and nutrient conditions than ER (which is more responsive to temperature) (Bernhardt et al., 2018), so the unbalanced response could reflect the fact that the primary disturbances associated with urbanization in these creeks are increased light and possibly nutrient enrichment.

The results also demonstrate that ER was negatively associated with stream channel width, and GPP was unassociated with width (Fig. 2.5). The negative association between channel width and ER aligns with the River Continuum Concept (RCC) in that smaller streams have higher and more direct inputs of allochthonous carbon from the riparian zone (Vannote et al., 1980). I additionally found a positive association between tree cover and ER in the fall, supporting this notion. On the contrary, GPP is predicted to increase in wider streams due to increased light availability, but I did not observe this pattern. Many of the small streams have low riparian vegetation scores (Appendix A). Additionally, two of the smallest watersheds, GC and LSC, exhibit the highest mean annual GPP over the study period (Fig. 2.2; Fig. 2.4). My study provides further support that open canopy urban streams disrupt expected patterns in GPP with stream network position.

On the daily scale, I observed a pronounced decline in GPP associated with summer storm events (Fig 2.6). I attribute this decline in GPP during events to scour, abrasion by sediment, substratum loss, and light attenuation limiting primary production (Biggs & Close, 1989; Katz et al., 2018). Urban streams and flood-prone rivers often show low resistance of GPP to storm events (Qasem et al., 2019; Reisinger et al., 2017; Uehlinger, 2006). In the studied sites particularly, the storm response was more pronounced in summer events. The summer months had the highest number of storms above the 80$^{th}$ percentile and the lowest time above mean flow compared to other seasons, indicating more flashy hydrology. I suggest that the seasonal response stems from intense summer storms— characteristic of the Piedmont region—with rapid water velocities on the rising stage of the hydrograph. Rapid rising rates increase the probability of scouring primary producers (Biggs & Close, 1989). Winter storms in Charlotte tend to be frontal systems with low-intensity rainfall, which is less likely to cause rapid hydrological responses. My work aligns with recent work that associates hydrological flashiness with GPP declines during storms (Qasem et al., 2019), but my work shows a distinct seasonal impact of storm events.

There are several supplementary explanations for the seasonal storm response in GPP. It is possible that winter storms transport detritus from the stream bed downstream which provides more light to benthic primary producers and balances the influences of scour and enhancement due to increased PAR (Roberts et al., 2007). Additionally, this pattern can be linked to increased susceptibility of autotrophs to storms with increasing biofilm density (Biggs & Close, 1989). In the winter, the baseflow community can be too poorly established to be susceptible to flood disturbances.

I also observed that storms impacted GPP most in urbanized watersheds and creeks with low bank vegetation (Fig. 2.8). A recent study in a few of the same watersheds as ours showed that total imperviousness is associated with higher peak flows in urban streams across Charlotte (Bell et al., 2016). I suggest that hydrological flashiness and high peak flows explain the GPP decline in more urbanized streams. The analysis additionally shows that, at the local channel scale, bank vegetation plays a role in protecting primary producers from abrasive flows. Bank vegetation can protect biomass by reducing the shear stress induced by high water velocities during storms (Chambers et al., 1991). Bank vegetation is also negatively related to bank stability and channel alteration (not included to improve parsimony), highlighting the role that practices such as rip rap stabilization and channelization play in increasing water velocity and decreasing resistance of primary producers to storms.

Alternatively, another process that could be important in urbanized streams is the susceptibility of high biomass mats to scour during stormflows (Biggs & Close, 1989). The storm response tends to be significant in creeks with highly productive summers (Fig. 2.6). High concentrations of biomass in more mature communities can be more loosely adhered to substrate leading to massive scour even with low magnitudes of changes in discharge (Biggs & Close, 1989). In both scenarios, urbanization plays a role either by enhancing biomass accrual between storms or increasing the magnitude and flashiness of storm events, leading to scouring. Both scenarios can also coexist and interact.

While storms explained variability in GPP, ER tended to be resistant to storms (Fig. 2.7). I observed small increases in ER in the winter in a few creeks (Fig. 2.7). Several studies have shown that ER is more resistant to stormflows than GPP (Beaulieu et al., 2013; Qasem et al., 2019; Uehlinger, 2006). Others have shown that ER was stimulated during stormflows (Larsen & Harvey, 2017; Roberts et al., 2007; Uehlinger, 2006). The concentration of respiration in the hyporheic zone—as opposed to the benthic riverbed—could explain the resistance of ER to storm events (Naegeli & Uehlinger, 1997).

Finally, I show that variability in GPP is positively associated with urbanization, but the correlation is not significant (Fig. 2.9). It is possible that the sample size is too small to detect this relationship. However, it is also likely that land use and creek changes do not exclusively increase variability in metabolic rates. I observed two factors primarily link urbanization to metabolic regimes: light (and possibly nutrients) and discharge. Increased light and nutrients to the creek represent a press disturbance that supports high biomass during baseflow throughout seasons (Bender et al., 1984). On the other hand, hydrologic disturbances are better characterized as pulse disturbances that have punctuated responses on short time scales. The combination of these two types of disturbances likely creates a non-linear relationship between urbanization and the variability of metabolic responses.

## 2.6. Implications

I observed strong seasonal patterns across all twelve urban creeks under study. Two distinct productivity regimes emerged across twelve urban creeks in Charlotte, North Carolina. One was characterized by distinct spring peaks and unproductive summers. The other was characterized by late spring and summer peaks and highly productive summers. Urbanization was associated with elevated GPP as well as lower resistance of GPP to flow disturbances, particularly during the summer. The streams in this study were all relatively small urban creeks— and based on stream size, I infer that these sites would be largely unproductive in summers under natural (non-urbanized) conditions. I attribute high productivity in the more urbanized watersheds to high light and nutrient conditions often observed in urban creeks (Fuß et al., 2017; Smith & Kaushal, 2015).

Seasonality in GPP and ER was largely associated with light and temperature regimes; however, hydrologic disturbances resulted in short-term declines in GPP during summer storms. Given that metabolic regimes represent basal energetics of stream ecosystems, such variability in carbon dynamics likely controls the type of biota that can survive hydrologic disturbances that also decrease energy supply. Overall, urbanization results in increases in productivity and frequent yet short-term variability associated with storms which likely impacts the type of macro-organisms that can adapt to such conditions (Bernhardt et al., 2018).

While hypoxia was not observed in the study creeks, it is important to note that high productivity, especially in the summer, creates conditions conducive to hypoxia (Mallin et al., 2006). There is increasing recognition of hypoxic conditions as key features of stream ecosystems in the Piedmont region (Blaszczak et al., 2019; Carter et al., 2021), and hypoxic conditions can negatively affect macro-organisms. In low-gradient and incised urban streams, like those observed in this study, hypoxia can manifest in deep pools because of stagnant water, eutrophication, high productivity, and subsequent decomposition of algal blooms. Increasingly hot summers and high nutrient loads can lead to a regime shift whereby summer hypoxic conditions are more dominant in these reaches. Frequent summer storms likely play a role in preventing hypoxia by moving water and nutrients through the stream network; however, projected climate regimes marked by summer drought and more extreme precipitation events could facilitate alternating hypoxia and hydrologic disturbance, placing these creeks in a persistent stressed state. The high productivity I observed could be an early warning sign of a critical transition towards prolonged hypoxic states in the summer.

Given the evidence presented here that urbanization shifts streams to high productivity regimes along with the potential for productive regimes to quickly become hypoxic, stream functional processes must be central to stream and watershed management. At the seasonal scale, it is important to control light reaching the stream surface. The results suggest that bank and riparian vegetation play an important role in shading streams. It may also be important to reduce nutrient inputs at the watershed scale and use stormwater control measures to promote nutrient storage and transformation in the terrestrial landscape.

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# CHAPTER 3. MAKING CONSERVATION NORMAL: A SYSTEMATIC REVIEW OF SOCIAL NORMS IN PROMOTING WATER QUALITY BEST MANAGEMENT PRACTICES

## 3.1. Abstract

Intensive land use practices in urban and agricultural watersheds significantly contribute to nonpoint source pollution. Given that water quality improvement relies on voluntary adoption of best management practices (BMPs), it is critical that we understand the factors that influence land managers' adoption of BMPs. While individual attributes are frequently studied in conservation adoption literature, the social context within which individuals are situated is important yet seldom examined. Social norms are key influences on decision-making, but they are seldom measured as a predictor of conservation adoption. I systematically reviewed studies addressing the influence of social norms on BMP adoption. Specifically, I identified salient social norms influencing BMP adoption and situational factors that mobilize and reproduce social norms. This review indicates that social norms create expectations for conventional farming practices and manicured residential lawns as well as a social responsibility for neighborhood cohesion and environmental stewardship. I found that norms supporting water quality BMPs were fostered during times of uncertainty, when attached to social sanctions and benefits, and in natural spaces. The most influential people to the establishment and reproduction of social norms were opinion leaders, bridging actors, and institutional actors. I suggest that practitioners promote descriptive norms supporting BMP adoption and leverage existing social networks and associated social norms to promote BMP adoption.

## 3.2. Introduction

Nonpoint source pollution from urban and agricultural land uses is detrimental to the water quality of rivers, lakes, reservoirs, and coastal zones (Carpenter et al., 1998; Foley et al., 2005; Howarth et al., 2011). Farmers, ranchers, and herders manage upwards of 40% of the world’s terrestrial biome, and excess inputs of nitrogen (N) and phosphorus (P) through fertilization and manure production cause nutrient saturated soils, subsequent leaching of nutrients to waterways, and over-enrichment of receiving water bodies (Carpenter et al., 1998; Foley et al., 2005).

Overenrichment of N and P causes eutrophication (excessive plant growth) and widespread hypoxia of water bodies at a global scale (Howarth et al., 2011), which are leading causes of water quality and aquatic ecosystem degradation (Smith & Schindler, 2009). Likewise, urban sources of N and P from atmospheric deposition and fertilization are quickly transported to water bodies via impervious surfaces and subsequently lead to impaired urban streams (Grimm et al., 2008; O'Driscoll et al., 2010; Paul & Meyer, 2001; Yang & Lusk, 2018). Other important causes of water quality degradation linked to urban and agricultural land uses are sediment and pesticides. Pesticide contamination is widespread (Aktar et al., 2009) and detrimental to stream macroinvertebrates (Rosi-Marshall et al., 2007). Sediment pollution contributes to multiple stressors that reduce algal biomass and abundance of aquatic organisms (Matthaei et al., 2010). Overall, urban and agricultural land management practices are predominant contributors to the degradation of water quality, and in the absence of strict regulation on nonpoint source pollution, governments rely on voluntary adoption of best management practices (BMPs) to ameliorate water quality. As these problems become increasingly difficult to manage, a primary challenge is to engage land managers in best management practices (BMPs), identify the most prominent influences on their decision-making, and promote decision-making supportive of water quality improvements.

BMPs are designed to reduce the load of pollutants and treat runoff before it enters receiving waters. Structural BMPs include, but are not limited to, grass swales, rain gardens, permeable pavement, riparian buffers, and cover crops (Yang & Lusk, 2018). Non-structural BMPs include low-P fertilizer, conservation tillage, and pesticide bans. BMPs reduce nutrient pollution at the household, neighborhood, and watershed scale (e.g., Collins et al., 2010; Dietz and Clausen, 2008; Line and White, 2016; Pennino et al., 2016); however, BMP effectiveness is highly dependent on the level of mitigation within a watershed (Bell et al., 2016; Jefferson et al., 2017; Pennino et al., 2016). Oftentimes, treatment of small fractions of impervious or agricultural land fails to yield measurable results on water quality, and Jefferson et al. (2017) suggest that critical thresholds must be attained to improve water quality.

Unlike point source pollution, nonpoint source pollution is not heavily regulated; therefore, pollutant reduction is primarily addressed through voluntary adoption of BMPs. Residential turfgrass is one of the primary sources of nutrient pollution in urban watersheds (Carey et al., 2012; Hobbie et al., 2017), and, in the United States, there are no federal laws regulating fertilizer application. The states are primarily responsible, and only a handful of states have implemented restrictions (Carey et al., 2012). Likewise, there is a heavy reliance on voluntary compliance with BMP adoption in the agricultural sector (Patterson et al., 2013; Shortle et al., 2001; Weitman, 2010). Voluntary conservation programs, like the United States Department of Agriculture (USDA) Farm Bill program, encourage landowners to adopt BMPs, provide financial and technical assistance, and sometimes, partially or fully fund BMP adoption. Such voluntary programs have shown little success, but widespread adoption of BMPs has not been evident (Ribaudo, 2015; Wade et al., 2015).

Given the reliance on voluntary adoption of BMPs, it is critical that we understand the primary drivers of land managers' BMP adoption. Most quantitative studies on farmer BMP adoption investigate how farmers' individual attributes— rather than their social and institutional context— influence BMP adoption (Yoder et al., 2019). Land managers are situated in social and institutional relations, and individuals' decision-making is shaped or constrained by others in their network, making possible the achievement of mutual benefits (Putnam, 2000). Social networks have a positive influence on conservation adoption of agricultural BMPs (Prokopy et al., 2008), but this structural dimension of social relationships says little about the social aspects that make these relationships meaningful to conservation practice. Social norms are a predominant lens through which we can understand the social context surrounding conservation decisions within social networks. Consistent operationalization and measurement of social norms are needed (Prokopy et al., 2019; Yoder et al., 2019), and to do that, we need to know the salient public and peer pressures that BMP adopters face.

Social norms— informal rules and understandings on how to behave in a social context (Cialdini et al., 1991; Mcdonald & Crandall, 2015; Morris et al., 2015)— are powerful predictors of human decision-making and specifically environmentally conscious behaviors (Farrow et al., 2017; Niemiec et al., 2020). The two major types of social norms are descriptive and injunctive social norms, and differentiating the two is important to identify different motivations. Descriptive norms refer to one's perception of the commonality of a behavior or "what is done" (Cialdini et al., 1991). Descriptive norms motivate human behavior by providing information on what actions will yield desired impacts or indicating behaviors that help one fit in (Nyborg, 2018). Injunctive norms refer to one's perception of the appropriate and socially approved behavior or "what ought to be done" (Cialdini et al., 1991). Injunctive norms motivate human behavior through the perceived promise of social rewards or threat of social sanctions (Cialdini et al., 1991). A form of injunctive norms are subjective norms, and subjective norms refer to people's motivation to comply with important others like family and friends (Cialdini & Jacobson, 2021; Lapinski & Rimal, 2005). Social norms are predominant predictors of human behavior and are features of widely cited behavioral models, including the value-belief-norm theory (Stern et al., 1999), theory of reasoned action (Ajzen & Fishbein, 1980), and theory of planned behavior (Ajzen, 1991).

Social norms are a prominent influence on the likelihood of adopting environmentally conscious behaviors. Social norms influence eco-friendly consumer choices, energy conservation, recycling, sustainable food management, sustainable land management, water conservation, and civic environmental action (e.g., Cialdini and Jacobson, 2020; Clarke et al., 2021a, 2021b; Farrow et al., 2017; Niemiec et al., 2020). For example, normative messaging detailing neighbors' household energy consumption and judgment of individuals' household energy consumption significantly reduced energy consumption for high consumers (Schultz et al., 2007). Regarding land management, social norms can be prominent influences on farming practices. A recent review showed that subjective social norms can both motivate and hinder the adoption of conservation BMPs and that farmers' observations of their neighbors influence farmers' practices (Ranjan et al., 2019). Additionally, in an Australian state that prohibits the clearing of native vegetation, farmers continued the practice, and those that complied failed to report offenders due to an established social norm that native plant clearing is appropriate and beneficial to their farms. Compliant farmers feared that reporting their neighbors would cause distrust between neighbors (Minato et al., 2012). When social norms are included in studies on conservation behavior, they are important and consistent predictors of intentions to adopt BMPs (Niemiec et al., 2020). Although social norms are established predictors of environmentally conscious behavior, social norms are inconsistently operationalized and infrequently measured in the conservation literature (Prokopy et al., 2019). However, there is a stronger focus on social norms in qualitative and mixed methods studies (Ranjan et al., 2019). It is critical that we understand the nuanced underpinnings of social support or disapproval of BMP adoption and the mechanisms that mobilize and reproduce norms supporting BMP adoption, especially in the context of water quality improvement.

Norm-based interventions and marketing strategies are gaining in popularity as a strategy to enhance BMP adoption and improve water quality (Hay et al., 2018; Riggs et al., 2010; Warner & Diaz, 2020). Additionally, fertilizer and chemical companies have competing, and often conflicting, marketing campaigns to shape community attitudes and norms towards heavier fertilizer and pesticide use (Robbins, 2007). A shift in norms towards social acceptance of conservation behavior could enhance BMP adoption; however, widespread use of social norms as an intervention strategy requires knowledge of the salient social norms influencing decision-making on land management and, more importantly, the pathways through which conservation norms become accepted and standardized. A synthesis of predominant normative influences on water quality BMPs could generate insight into conventional and emerging social norms that hinder or facilitate water quality improvement, identify influential referents and contextual factors that moderate normative influences, and facilitate the design of interventions that target social norms to enhance BMP adoption.

I systematically reviewed literature on social norms in relation to BMP adoption with the goal of water quality improvement. Specifically, this systematic review asked the following: 1) What are the salient normative influences on the adoption of water quality improvement BMPs? 2) What factors promote or inhibit the influence of social norms supporting water quality BMPs? 3) Who are the most influential people and organizations affecting the adoption of BMPs for improving water quality? 4) Can social norms be further leveraged to encourage BMP adoption, and if so, how? I addressed these questions by conducting a systematic search and reviewing 65 research articles focused on the influence of social norms on BMP adoption for water quality improvement.

## 1.1. Methods

In this review, we specifically focused on studies that discussed the role of social norms in the adoption of BMPs that address water quality impairment from nonpoint sources in both rural and urban settings. The following methods were conceptualized and performed by Dr. Zhao Ma, Jenn Domenech, Je'Nae Johnson, and me. We used electronic databases and snowball sampling to identify relevant literature on the topic. We conducted electronic database searches in Scopus, Web of Science, Agricola, Wiley, and JSTOR because each database includes some different disciplines and journals but together they provide a comprehensive coverage of peer-reviewed journal publications. We performed three electronic searches in October 2017, May 2020, and August 2021. Search terms are presented in Table 3.1. We searched the title, abstract, keywords, and full-text as appropriate for each database. We restricted our search to peer-reviewed journal articles when this restriction was available in the database search options. We acquired additional articles through snowball sampling of in-text references. After the final electronic database search, we updated our search with recent publications using Google Scholar.

We initially screened the electronic search results by removing articles that were not peer-reviewed journal articles and removing duplicate articles across the databases. We then developed and applied exclusion criteria to screen the abstracts and full-text of the remaining articles. We applied the following exclusion criteria and removed articles that: 1) had no mention of search terms for water quality, norms, or BMPs/LIDs, 2) referred to normative water quality standards or BMP design standards, not social norms, 3) included term “norm” or highlights social influences, but the social influence was not evaluated in the specific context of social norms, 4) did not examine norms specifically in the context of motivation to participate in conservation practices, and 5) addressed conservation behaviors unrelated to improving water quality such as reducing water consumption. In total, we screened 785 articles, and after applying the exclusion criteria, our final reviewed set included 65 articles.

We compiled a database of the final reviewed set and extracted the following data from each article: the overall purpose of the study, study type (e.g., theoretical/conceptual, empirical, and review), type of empirical data collected (e.g., primary or secondary), study design, data collection tool (e.g., survey, interview, secondary documents), type of analysis conducted (e.g., qualitative, quantitative, or mixed), unit of observation, unit of analysis, the country where the study was conducted, land use (rural, urban, mixed), norm focus (descriptive, injunctive, both), social influence (e.g., block leader, farmer peer, neighbor), water quality focus (e.g., nitrogen, phosphorus, pesticides), best management practice, main results, and significance of normative influence.

Table 3.1. Terms for electronic searches.

| **Water** | | **AND** | **Best Management Practices** | | **AND** | **Theory** | |
|---|---|---|---|---|---|---|---|
| Water Qual* | **OR** | | Best management practice | **OR** | | Norm | **OR** |
| Nonpoint source pollution | | | Conserv* | | | Norm* motivat* | |
| NPS | | | Cover crop | | | Intrinsic motiv* | |
| | | | No-till | | | Descriptive norm | |
| | | | Grass waterway | | | Injunctive norm | |
| | | | Riparian buffer | | | Enviro* norm | |
| | | | Manure stor* | | | Social norm | |
| | | | Filter strip | | | Norm* behav* | |
| | | | Conservation till* | | | Normative | |
| | | | Contour crop* | | | Peer influen* | |
| | | | Cover crop* | | | Peer persuas* | |
| | | | Water conserv* practice | | | Social influ* | |
| | | | Low impact development | | | Peer educat* | |
| | | | Bioretention | | | Block leader | |
| | | | Grass swale | | | Peer to peer | |
| | | | Rain garden | | | Peer-to-peer | |
| | | | Green infrastructure | | | Social network | |
| | | | Permeable pavement | | | Social learning | |
| | | | Green roo* | | | Peer network | |
| | | | Green space | | | | |
| | | | Rain barrel | | | | |
| | | | Stormwater control | | | | |
| | | | Agricult* practice | | | | |

## 1.2. Results and Discussion

Of the 65 reviewed articles, 40 articles explicitly conceptualized, measured, and/or analyzed the impact of social norms on BMP adoption. The rest of the articles often measured social relations/connections and speculated that social norms were important to social relations/connections. Several articles also mentioned social norms as important predictors of BMP adoption in the introduction, without substantially conceptualizing or analyzing the ways in which social norms impacted BMP adoption. Therefore, I focused the analysis on the 40 studies that explicitly conceptualized, measured, and/or analyzed social norms impact on BMP adoption. Table 3.2 provides descriptive metrics of the selected studies. Of the 40 articles, 34 of the studies were published between 2011-2021, which aligns with the documented exponential growth of the conservation adoption research field (Yoder et al., 2019). An overwhelming majority of the studies were conducted in the United States; notably, the review was limited to studies written in English. Twenty-two studies were conducted in rural landscapes, and eight studies were in urban landscapes. There were 18 quantitative studies, 16 qualitative studies, and 6 mixed methods studies. Twenty studies focused on injunctive norms, six studies on descriptive norms, and five studies focused on both. Nine studies did not specify the type of normative influence and only used the general terms "norms" or "social norms" in the articles, which indicates a lack of conceptual clarity. The heavy focus on injunctive norms stemmed from the use of theoretical frameworks that include injunctive norms such as the Theory of Planned Behavior (Ajzen, 1991) and Theory of Reasoned Action (Ajzen & Fishbein, 1980). The social referents most often analyzed were neighbors and peers (Table 3.2).

In the following sections, I present the major themes that emerged from this systematic review and address the research questions regarding salient social norms, contextual factors with which social norms interact, influential people and institutional actors, and social norms as an intervention strategy (Table 3.3). I also discuss future directions for research on social norms in conservation adoption and the implications of this synthesis on building and sustaining social infrastructures that promote widespread adoption of BMPs for improving water quality.

Table 3.2. Overview of extracted data from reviewed studies.

| | | Number of Articles |
|---|---|---|
| **Year of Publication** | Before and During 2010 | 6 |
| | Post 2010 | 34 |
| **Country** | Australia | 4 |
| | China | 1 |
| | England | 2 |
| | France | 1 |
| | Germany | 1 |
| | Ireland | 2 |
| | Italy | 1 |
| | United States | 25 |
| | Global | 1 |
| | Not specified | 2 |
| **Land Use** | Rural | 22 |
| | Urban/suburban | 8 |
| | Both | 8 |
| | Not specified | 2 |
| **Study Type** | Empirical | 35 |
| | Review/Conceptual | 5 |
| **Type of Data** | Qualitative | 16 |
| | Quantitative | 18 |
| | Mixed | 6 |
| **Social Norm Type** | Descriptive | 6 |
| | Injunctive | 20 |
| | Both | 5 |
| | Not specified | 9 |
| **Primary Social Influence** | Neighbors/Peers | 17 |
| | Farmer Peers | 16 |
| | Family/friends | 7 |
| | People important to individual | 7 |
| | Advisors | 6 |
| | Organizations | 3 |
| | Suppliers | 3 |

Table 3.3. Emerging themes from the review addressing each research question.

| Research Question | Theme | Articles Reviewed |
|---|---|---|
| What are the salient normative influences on the adoption of water quality improvement BMPs? | (i) Social Pressure for Conventional Practices – High Input, High Output Agriculture | Inman et al. 2018; McGuire et al. 2013; Mills et al. 2017; Taylor et al. 2015 |
| | (ii) Social Pressure for Conventional Practices – The Manicured Lawn | Eisenhauer et al. 2016; Hu et al. 2017; Nohner et al. 2018; Peterson et al. 2012 |
| | (iii) A Shared Responsibility to Care for Neighbors and Water Quality in Urban and Rural Lands | Atwell et al. 2009a; Atwell et al. 2009b; Yoder and Chowdhurry 2018; McGuire et al. 2013; Evans et al. 2020; Warner et al. 2018; Shaw et al. 2011; Mills et al. 2017; Warner et al. 2021; Doran et al. 2020; Daxini et al. 2018; Daxini et al. 2019; Fielding et al. 2005; Wang et al. 2020; Eanes et al. 2020 |
| | (iv) Conflicting Conservation and Production Norms | McGuire et al. 2013; Skaalsveen et al. 2020 |
| What factors promote or inhibit the influence of social norms supporting water quality BMPs? | (i) Ambiguous Contexts | Atwell et al. 2009a; Yoder and Chowdhurry 2018; Daxini et al. 2019 |
| | (ii) Social Sanctions and Rewards | Atwell et al. 2009a; Daxini et al. 2018; Taylor et al. 2015; Yoder and Chowdhurry 2018; Welch and Marc-Aurele 2001 |
| | (iii) Proximity to Nature and Conservation | Warner et al. 2020; Mills et al. 2017 |
| Who are the most influential people and organizations affecting the adoption of BMPs for improving water quality? | (i) Opinion Leaders and Influencers | Skaalsveen et al. 2020; McGuire 2013 |
| | (ii) Bridging Roles | Skaalsveen et al. 2020; Daxini et al. 2019; Yoder and Chowdhurry 2018 |
| | (iii) Institutional Support | Fielding et al. 2005; Larson et al. 2010; Persaud et al. 2016; Warner et al. 2021 |
| Can social norms be further leveraged to encourage BMP adoption, and if so, how? | (i) Nudging farmers increases BMP adoption | Kuhfuss et al. 2016; Peth et al. 2018 |

### 3.2.1. What are the salient normative influences on the adoption of water quality improvement BMPs?

Social norms are most likely to motivate behavior when social expectations and behavioral commonalities are made salient to an individual (Cialdini et al., 1991). Furthermore, past studies have shown that multiple social norms across different in-groups can co-exist and conflict, and such conflict can motivate individuals' behavior (McDonald et al., 2014). In this section, I discuss the salient and often conflicting social norms influencing water quality BMP adoption. Three salient social norms emerged, including peer pressure to maintain conventional and high-input farming practices, social pressure to support manicured lawns, and a shared responsibility to care for neighbors and the environment (Table 3.3). Here, I also discuss the co-existence of conflicting social norms and how they drive water quality conservation behavior.

*(i) Social Pressure for Conventional Practices – High Input, High Output Agriculture*

In farming communities, there was a prevalent social norm to maintain conventional farming practices (e.g., high fertilizer and pesticide inputs and soil tillage) (Inman et al., 2018; McGuire et al., 2013; Mills et al., 2017; Taylor & Grieken, 2015). The motivational underpinning was that conventional practices lead to productive farming and thus high yields. Specifically, productivist, profit-based, normative beliefs have been linked to farmers' perceived social responsibility to produce as much as they can to address food insecurity (Mills et al., 2017). Compliance with these norms is motivated by social sanctions and rewards. Farmers noted fears that being out-of-step with their peers by implementing new BMPs would lead to them being "hung-out" as expressed in this farmer's account:

> In our group, all of us went the same way [change of row spacings] and we had our own harvesting group. Now that started to fall apart a bit ... I didn't realize this, but one of our blokes was debating strongly whether he was going to go back to the old row spacing [because] he didn't want to get hung-out suddenly finding he was the only person to get his cane cut, in a group he wanted to be in, that was different. (Taylor et al., 2015, p. 16)

The loss of social relationships, especially in cooperatives, can lead to financial impacts as farmers are often dependent on those relationships to meet their daily needs (Taylor and Grieken, 2015). Farmers also noted peer pressure for their fields to "look as good" (McGuire et al., 2013, p. 65) as their neighbor's, and new practices bring uncertainty in their ability to maintain the aesthetic and assumed productivity that follows. In contrast to social sanctions, farmers also recognized productive, high-yielding farms as status symbols, and productivity was the main metric by which they were judged as 'good farmers' by their peers (Inman et al., 2018; Mills et al., 2017). Such recognition and status are not often equally sought for BMP adoption (Inman et al., 2018).

*(ii) Social Pressure for Conventional Practices – The Manicured Lawn*

Previous studies discuss how residents' intensive upkeep of culturally normative turfgrass lawns is perceived as a social obligation and civic responsibility to be good neighbors and promote neighborhood cohesion (Robbins, 2007; Robbins & Sharp, 2003). The articles I reviewed discussed how rural and urban residents perceived social pressure to maintain a manicured lawn was associated with lower likelihood to adopt lawn BMPs (e.g., fertilizer reduction, native planting, littoral planting) (Eisenhauer et al., 2016; Hu et al., 2017; Nohner et al., 2018; Peterson et al., 2012). For example, shoreline property owners in Michigan who perceived social pressure to maintain a manicured lawn were less likely to accept shoreline conservation easements (Nohner et al., 2018). Likewise, when considering shoreline vegetation of stormwater ponds, residents in southwest Florida expressed shared preferences for manicured shorelines and open views of clear (algae-free) water which reflected their normative preference for turfgrass lawns (Hu et al., 2017). Residents were aware of the potential consequences of their pond preferences on water quality yet still prioritized normative aesthetic preferences (Hu et al., 2017). The normative preference for manicured lawns is so deeply rooted in residential communities that residents incorrectly assume the high prevalence of manicured lawns reflects their neighbors' personal preferences for lawns—in this case, the most preferred landscape was actually mixed lawn and native planting (Peterson et al., 2012). The ubiquity of turfgrass lawn in residential neighborhoods and institutional enforcement of the norm elicits an injunctive norm for residents to upkeep this practice. Lawns are often inherited from previous owners, and strong neighborhood pressure to maintain them drives conformity.

*(iii) A Shared Responsibility to Care for Neighbors and Water Quality in Urban and Rural Lands*

Farmers' perceived obligation to care for their neighbors and steward the land was evident in several qualitative studies (Atwell et al., 2009b, 2009a; Evans et al., 2020; McGuire et al., 2013;

Mills et al., 2017; Yoder & Chowdhury, 2018). Caring for neighbors is a rural ideal and often a social expectation in rural communities (Atwell et al., 2009a, 2009b; Evans et al., 2020). For example, neighbors appreciate a heads-up when farmers spray pesticides, given the toxicity of the chemicals (Atwell et al., 2009b). Likewise, one farmer in Wisconsin recalled, “It comes down to neighbor relations. It is embarrassing. It is like, we have got mud in the lawn of the neighbor because it washed out of our field. Put a few of those up to, ‘Oh it happened,’ but it happens too many times, so it is like, we need to do something different” (Evans et al., 2020, p. 423). Farmers are expected to upkeep neighborly relations, and upholding such relations can elicit conservation practices, such as pesticide use reduction and soil conservation. The impact of social norms supporting neighborhood cohesion on BMP adoption for water quality improvement has not been explored in the literature; however, reframing conservation as an issue that threatens neighborly and communal relations as well as water quality could resonate with the established norm for good neighborly relations.

Farmers also expressed an explicit perceived obligation to care for the land and water. Farmers expressed their desire to be seen as good stewards of the land and how they *ought* to implement BMPs that help improve downstream water quality (Atwell et al., 2009a). Such expectations become particularly salient when farmers are made aware of the consequences of conventional farming practices on water quality. When facing public scrutiny and being adjacent to nature reserves that practice conservation, farmers felt an obligation to be seen ‘doing the right thing’ and contribute to conservation efforts (Mills et al., 2017). Under more strict nutrient regulation, farmers were under peer pressure to not be the “bad guys” who contributed most to phosphorus loads (Yoder and Chowdhury, 2018, p. 358). In the Florida everglades, farmers compared monitoring data from each other’s farms to pressure the biggest polluters to use less fertilizer. This became a competition where awards were used as social recognition of conservation farming. Soon, normative expectations of being a ‘good farmer’ entailed not only high yields but also conservative inputs and low contribution to nutrient loading in the watershed (Yoder & Chowdhury, 2018). These qualitative accounts are confirmed by quantitative studies showing that subjective norms of neighbor and peer support for BMPs positively influences willingness to adopt BMPs (Daxini et al., 2018, 2019); although, the effect of social norms on BMP adoption was inconclusive in others (Doran et al., 2020; Fielding et al., 2005).

Urban and rural studies show a shared responsibility for water quality in residential communities. Studies showed high social support for fertilizer reduction and rain garden installation in residential neighborhoods (Shaw et al., 2011; Warner et al., 2018), and residents with higher social approval are more likely to implement BMPs (Shaw et al., 2011; Warner et al., 2021). Other studies show inconclusive evidence that social norms supporting conservation impact the likelihood of BMP adoption (Eanes & Zhou, 2020; Wang et al., 2020).

*(iv) Conflicting Conservation and Production Norms*

Different, even conflicting social norms about conservation and production can co-exist. Social norms are informal rules anchored in social groups, and disparate groups can maintain different norms (McDonald et al., 2014). There is emerging evidence that conservation-oriented farmers form like-minded social networks with other conservation-oriented farmers, and they use these networks for peer-to-peer learning about conservation practices (Skaalsveen et al., 2020). These conservation networks often expand beyond the traditional rural communities to global networks and help farmers alleviate unpleasant judgment from neighbors (i.e., conventional farmers) who disapprove of their conservation practices (Skaalsveen et al., 2020).

Social norms supporting conservation and production can also operate simultaneously within the same community of practice. McGuire et al. (2013) detail such phenomena using the identity control model from Burke (1991). The model posits that farmers adjust their 'good farmer' identity by reflecting on their social situation and what they see others doing (i.e., descriptive norms). Iowa farmers were made aware of their detrimental impact on local water quality, and under the simultaneous influence of presumably conflicting conservation and production social norms, farmers began to align conservation and production goals; conventional farmers adjusted their farming practices by looking to conservation-oriented farmers to learn from their practices (McGuire et al., 2013). Conventional farmers emphasized the economic efficiency of conservation practices as strong motivators for adoption, and conservation-oriented farmers drew on this socially acceptable motivation to encourage adoption by emphasizing financial savings and performance enhancement of BMPs. In this situation, conventional farmers still saw production and profit goals as main drivers of their actions, but when both conservation and production norms were salient, an opportunity emerged for farmers to seek new information from the conservation social network and develop an appropriate response to water quality issues. Farmers adjusted their practices through discourse with other farmers, where injunctive norms supporting BMP adoption were established, and through observation of conservation leaders.

### 3.2.2. What factors promote or inhibit the influence of social norms supporting water quality BMPs

Previous studies have recognized that situational factors can influence the relationship between social norms and individuals' behaviors (Morris et al., 2015). Regarding water quality, landowners and managers often face threats of stricter government regulation and uncertain decision domains as novel water quality issues emerge and courses of action are unclear. Some studies also posit that locational context is important to the impact of social norms on adoption of BMPs. The following themes emerged from this systematic review and outline several situational contexts that promote or inhibit establishment of social norms supporting water quality BMPs (Table 3.3).

*(i) Ambiguous Contexts*

Past research indicates that ambiguous decision domains, where the appropriate action is unclear to an individual, trigger individuals to look for social cues to either gather information or learn about the effects of new actions (Cialdini et al., 1991; Lapinski & Rimal, 2005). In these situations, individuals look to influential referents for new information about what they ought to do. Individuals in familiar situations and contexts often do not need to reference their social situation for cues on the appropriate behavior. In an article I reviewed, a farmer dealing with recurrent flooding and poorly drained soils described the following:

> Well, one of the neighbors up north here, he was kind of making fun of me one day. I was complaining about all these waterways coming down here, and he goes, 'You know, if you were smart, you'd put that in wetlands.' He said it kind of abusive. And I sat around and thought, 'You know, you're right.' ... That was the best thing I'd ever done. Oh, I'd had to fight those fields! (Atwell et al., 2009b, p. 9)

The farmer looked to a neighbor to provide information and validate their behavior when their course of action to deal with flooding was unclear, and in this situation, normative information supporting restored wetlands provided the farmer with a solution to their problem as well as a practice that improves water quality.

Seeking out others can also adjust individuals' ideas of perceived behavioral control (Lapinski & Rimal, 2005). Daxini et al. (2019) further support the argument that injunctive norms are a source of information for new behaviors by showing that subjective norms supporting BMP adoption positively influence farmers' attitudes about BMP adoption and their perceived behavioral control. The authors suggest that farmers use important referents as an easy and accessible source of information to evaluate the advantages, disadvantages, and efficacy of BMP adoption.

For farmers, uncertainty also arises as their conventional behaviors are judged by the public as problematic. These situations can cause individuals to seek new information to inform their decisions and address the problem (McDown, 2005). For example, farmers facing government regulation and public scrutiny drew on published farm monitoring data (i.e., descriptive norms) to seek out farmers that implemented practices to control their phosphorus pollution and learn about new practices to reduce pollution (Yoder & Chowdhury, 2018).

*(ii) Social Sanctions and Rewards*

Several studies have shown that injunctive norms are more salient than descriptive norms because of social sanctions (Daxini et al., 2018; Yoder & Chowdhury, 2018). Social sanctions are a motivating factor for individuals to comply with social expectations (Cialdini et al., 1991; Schwartz, 1977). Of the studies I reviewed, it was clear that early adopters of conservation practices felt greater social pressure from regulators to adopt BMPs—it was potentially the fear of potential regulation that drove their decisions (Daxini et al., 2018). Individuals often adhered to social norms supporting conservation to avoid potential social sanctions (i.e., heavier regulation) (Daxini et al., 2018; Yoder & Chowdhury, 2018). In the same vein, individuals may conform to conservation with the expectation of social rewards and pay-offs (Cialdini et al., 1991; Morris et al., 2015). In one study, conforming to conservation norms was rewarded with social accolades to incentivize participation (Yoder & Chowdhury, 2018), and farmers often sought social recognition and praise for their compliance with conservation norms (Atwell et al., 2009b; Taylor & Grieken, 2015). Farmers may also adhere more to conservation norms when made aware of the potential financial savings (McGuire et al., 2013).

*(iii) Proximity to Nature and Conservation*

I found two studies that linked proximity to nature reserves and water to social norms supporting water quality conservation BMPs. In one case, those with higher exposure to aquatic ecosystems (e.g., stormwater ponds, lakes, rivers, etc.) perceived significantly greater injunctive norms for good fertilizer practices (Warner et al., 2020). Past studies have shown that urban park use is linked to support for conservation through social interactions (Dean et al., 2019). It is possible that amount, condition, and use of natural spaces are associated with greater exposure to social norms and opportunities for social interaction—information sharing, information seeking, observing others, and mimicry— and thus the activation and spread of conservation norms. In another case, farmers adjacent to nature reserves felt an obligation to 'do the right thing' and adopt conservation practices (Mills et al., 2017, p. 293). One farmer noted, "It is easier to have the margin because on the other side of the ditch the land belongs to an ecological trust and they have trees and fancy grass and bird boxes and all that and I thought it might look like I was doing my bit as well" (Mills et al., 2017, p. 293). While not explicated in these examples, proximity to conservation cues can be a proxy for one's perception of prevalent descriptive norms. In the case of the farmers adjacent to nature reserves, persistent descriptive cues of conservation practices from neighboring reserves could make conservation norms more salient. It is also possible that farmers feel that their behavior is now more readily observed and judged by their conservation-oriented neighbors, which can make injunctive norms supporting conservation more powerful (Andreoni & Bernheim, 2009; Vesely & Klöckner, 2018).

### 3.2.3. Who are the most influential people and organizations affecting the adoption of BMPs for improving water quality?

Social capital theory argues that social connections facilitate and foster social norms (Putnam, 2000). People are more likely to conform to social norms when they feel included in referent groups and share a strong affinity with the group members (Lapinski & Rimal, 2005). Moreover, some social connections may be more likely to impact the development and mobilization of social norms than others (Coleman, 1988). In this section, I discuss how opinion leaders, bridging roles, and institutional actors foster social norms related to BMP adoption (Table 3.3).

*(i) Opinion Leaders and Influencers*

Influential individuals play a central role in the diffusion of new ideas and actions (Rogers, 1983). Opinion leaders are socially situated in central positions in social networks and tend to have high technical competence. Opinion leaders motivate others to change their behavior by acting as innovators and trusted sources of advice on the new behavior (Rogers, 1983). In some of the studies I reviewed, opinion leaders were early adopters of conservation practices and prompted others to adopt BMPs (McGuire et al., 2013; Skaalsveen et al., 2020). For example, opinion leaders were early adopters of no-till conservation practices in England and central to the social networks of conservation-oriented farmers (Skaalsveen et al., 2020). Conservation leaders also often had a strong social media presence; hence, they were social media influencers (Skaalsveen et al., 2020).

Opinion leaders are also often those who comply with social norms and are involved in norm-enforcement; however, they also take risks and are innovative when they perceive positive outcomes (Morris et al., 2015; Rogers, 1983). When influential group members stop complying with and enforcing the social norm, change can ripple through the social network (Morris et al., 2015). In the studies I reviewed, this phenomenon was observed when a single conventional farmer in Iowa adopted manure conservation practices that resulted in the highest yield in the local coop. This feat was seen as innovative, and discussion of the farmer's practices rippled through the farming community (McGuire et al., 2013). Influencers like this farmer likely facilitate the spread of social norms supporting adoption of BMPs by reducing the perceived social cost of new behaviors for others.

*(ii) Bridging Roles*

Bridges, or knowledge brokers, are typically peripheral to the local community and serve as connectors between the local community and new sources of knowledge. In the Diffusions of Innovations theory, bridges are termed change agents, and they are often technical experts that increase information flow (Rogers, 1983). Several bridges were identified in the articles I reviewed (Daxini et al., 2019; Skaalsveen et al., 2020; Yoder & Chowdhury, 2018). Bridges identified in Skaalsveen et al. (2020) were farmers who were connected with the scientific community and communicated new scientific information about no-till farming to other farmers. Knowledgebroking farmers that span social boundaries were important to connecting farmers to new sources of information that could help strengthen conservation norms.

Extension agents were also highlighted as key informants that helped farmers understand the costs and feasibility of new practices and ultimately encouraged a shift in social norms around good farm management towards practices that save money and benefit the water quality (Yoder & Chowdhury, 2018). While both technical experts and peers were important to the development of farmers' conservation norms, technical experts exerted a *greater* influence on nutrient management (Daxini et al., 2019). This shows the importance of bridging roles and knowledge brokers to the development of injunctive conservation norms. Bridging roles did not unanimously facilitate social norms supporting conservation practices. Bridging roles as a limitation to the uptake of conservation norms were illustrated as extension agents reflected the widespread public criticism and pressure that farmers perceive as unfair, which ultimately discouraged participation in BMP adoption (Yoder & Chowdhury, 2018).

*(iii) Institutional Support*

Institutional actors play a critical role in the crystallization of social norms. As institutional actors play the role of norm enforcers, social norms often become more rigid, and compliance becomes more uniform (Morris et al., 2015). While most studies in this review focused on peers as the normative influencers, a few studies placed greater focus on institutional actors (Fielding et al., 2005; Larson et al., 2010; Persaud et al., 2016; Warner et al., 2021). Compliance with norms requires that individuals feel affinity to the social group or organization. Fielding et al. (2005) attempted to identify the most influential referents in an elicitation study that asked respondents to identify the most influential individuals or groups that would judge their riparian management. The authors used these groups in the full survey to test for normative influence. Strong intenders to adopt riparian BMPs had greater normative beliefs and more willingness to comply with the local watershed organizations, conservation groups, and government agencies (Fielding et al., 2005). Other studies that I reviewed discussed residential homeowner associations (HOAs) as norm-enforcing institutions— institutional actors that enforce the social norm for a manicured lawn through a series of social sanctions and rewards (Larson et al., 2010; Persaud et al., 2016; Warner et al., 2021). However, only one study, Warner et al. (2021), tested the influence of living in an HOA on the intention to adopt BMPs. Interestingly, Warner et al. (2021) found that living in an HOA— and presumably being influenced to over-fertilize one's lawn— supported engagement in fertilizer BMPs. The authors suggested that HOAs, as influential institutional actors, can be supportive of water quality BMPs and that these organizations should be targeted to promote fertilizer BMPs.

### 3.2.4. Can social norms be further leveraged to encourage BMP adoption, and if so, how?

There is a trend towards soft policies— in contrast to strict regulation— to encourage behavioral change for the public good (van Deun et al., 2018). Social norms have been used as a social marketing tool to encourage BMP adoption (Hay et al., 2018; Riggs et al., 2010; Warner & Diaz, 2020). In this section, I discuss experimental and longitudinal studies that used normative information in intervention strategies to increase farmers' likelihood to adopt BMPs (Table 3.3).

*(i) Nudging farmers increases BMP adoption*

It is clear that social norms impact conservation BMP adoption broadly and people's willingness to adopt water quality improvement BMPs specifically, but can social norms be leveraged to change conservation behavior? The use of social norms to nudge behavioral change has been increasingly explored as a policy tool for motivating behaviors from healthy eating to water and energy conservation (e.g., Abrahamse and Steg, 2013; Goldstein et al., 2008; Lede et al., 2019; van Deun et al., 2018). Two experimental studies in this review examined the influence of descriptive social norms on farmers' likelihood to adopt BMPs (Kuhfuss et al., 2016; Peth et al., 2018). Kuhfuss et al. (2016) compared maintenance of agricultural BMPs after a period of regulation ended between a control group of farmers and a treatment group who received information on the prevalence of BMP adoption in the region. They found that the likelihood of maintaining BMPs was two times higher in the group given information on descriptive norms (Kuhfuss et al., 2016). In a business management game experiment, Peth et al. (2018) found that nudging farmers with descriptive and personal norms decreased the prevalence of farmers' noncompliance with a mandate for riparian buffers. It should be noted that descriptive norms are more effective at nudging behaviors when the prevalence of the desired behavior is already high (Cialdini et al., 1991). In both of these contexts, farmers had already experienced strict regulation that required the adoption of BMPs, and the studies tested whether the farmers would maintain these practices after the regulation period or comply with mandates (Kuhfuss et al., 2016; Peth et al., 2018, respectively). Overall, these studies suggest that social norms can be leveraged to encourage late adopters to practice conservation.

Alternatively, social norms can be shifted through peer-to-peer discourse and interactions that challenge existing norms supporting conventional practices. I reviewed a longitudinal study in which researchers interviewed farmers as farmers went through a performance-based environmental management intervention (McGuire et al., 2013). Farmers received information on their management outcomes in relation to environmental metrics and had the opportunity to discuss their practices and outcomes with other farmers. These discussions provided farmers with feedback from their peers. This feedback process successfully challenged well-established social norms supporting conventional practices and provided social support for farmers seeking out conservation practices (McGuire et al., 2013). This approach relies on providing farmers with information and social support that facilitates sound decision-making.

### 3.3. Future Directions for Research

This synthesis included both qualitative and quantitative studies; however, I noticed that many qualitative studies provided rich contextual information that was often missed in the quantitative studies. Social norms are informal and unwritten rules—these subtle constructs often emerge in discourse. Many of the quantitative studies used generic measures of social norms (Appendix D) that fail to capture normative motivations and contextual underpinnings. Through interviews, the qualitative studies provided nuanced and complex accounts of salient social norms that were not explicitly tested for in the quantitative, survey-based studies. For example, while many of the quantitative studies measured social norms by asking whether one's neighbors think the individual *should* implement water quality BMPs (Appendix D), the qualitative studies more often explored underlying normative beliefs (e.g., productivity and environmentalism), perceptions of social sanctions and rewards, and influential referents, which are all important conceptual refinements that improve our understanding of how social norms influence behavior. I also reviewed a few studies that used sequential mixed-methods to first identify the salient normative influences through interviews and then operationalize the concepts in survey item measures (Fielding et al., 2005). Given that social norms change with contextual factors, I suggest that mixed-methods designs can balance the rich contextual information that qualitative studies provide and the generalizability of quantitative studies.

Descriptive norms were not a predominant focus of the studies I reviewed (Table 3.2), but I present evidence that descriptive norms play a central role in water quality BMP adoption. Land managers use information about what others are doing to inform their decisions. This is clearly evident in the two experimental studies that show how descriptive norms can be used to enhance the likelihood of BMP adoption (Kuhfuss et al., 2016; Peth et al., 2018). The synthesis also indicated that descriptive norms are powerful influences on the development of subjective norms supporting manicured lawns (Peterson et al., 2012). Descriptive norms likely play a larger role than the body of literature reflects, and understanding this role may lead to more effective outreach and intervention on BMP adoption.

I identified three emerging topics that could assist in our understanding of social norms impact on the adoption of water quality BMPs. First, this synthesis showed that conflicting social norms supporting conservation and productivity could co-exist and inform decision-making. We need more research investigating if and how people navigate and negotiate through conflicting norms and how this process informs their decision-making. McDonald et al. (2014) suggest that norm conflicts surrounding environmental behaviors are associated with enhanced willingness to adopt environmental practices. If this pattern holds for water quality improvement BMPs, it would be beneficial to bridge communities supporting different social norms to increase the flow of normative information supporting conservation. Second, this synthesis suggests more explicit links between social norms and social structures. I found opinion leaders, bridging roles, and institutional actors are key facilitators of social norms. Researchers should explore the role that HOAs, conservation leaders, and extension agents play in facilitating social norms supporting water quality conservation. Third, a fruitful avenue for future scholarship could be a theoretical understanding of how proximity to aquatic spaces and nature reserves facilitates the development of social norms supporting conservation. Such knowledge would help develop programs targeting spaces such as nature reserves and stormwater ponds as effective places to promote conservation norms.

There was little to no discussion on how racism, sexism, and other systemic inequities impact social norms supporting water quality conservation. Race, gender, and sexuality are dividing lines in agricultural resources and social support (Leslie et al., 2019; Williams, 2018).

Many of the reviewed studies studied predominantly White and male samples. Given that agriculture and residential neighborhoods alike are highly segregated, different social norms likely exist in minoritized social groups. Social norms are place- and community-based; therefore, use of norm interventions may differ by community.

### 3.4. Implications for Water Quality Improvement

This synthesis provides evidence that social norms are important predictors of the adoption of water quality improvement BMPs. I highlight that social norms are precursors to individuals' attitudes and perceived efficacy of BMPs (Daxini et al., 2018), showing that individuals' social environment, and specifically social norms, cannot be ignored when encouraging BMP adoption to improve water quality.

Before I dive into policy implications, it is important to stress that social norms are context-dependent. In this review, I highlighted patterns in social norms across many studies; however, practitioners and researchers should engage in conversation with their local communities to elicit norms specific to their locale. As I have shown throughout this review, social norms are implicit, unwritten, and often subconscious drivers of behaviors, which requires deep community engagement and reliance on existing social networks to understand salient norms.

It is important that land managers are aware of the consequences that their practices have on receiving water bodies. Several studies revealed that social norms supporting conservation were more salient when there was an awareness of consequences. Water quality problems are often externalities to residential and farm management practices; therefore, problem identification and particularly sharing this information with farmers and residents is an important influence on the development of social norms (Bamberg & Möser, 2007). However, problem awareness alone may not be enough to facilitate social norms supporting conservation. This is evident among residential communities who acknowledge water quality impairment related to pesticide and fertilizer use yet continue such practices due, at least in part, to social norms supporting manicured lawns as essential to neighborhood cohesion (Hu et al., 2017; Robbins et al., 2001).

I also noted a need to identify and recognize social norms that are debilitating to conservation efforts but also take advantage of the plasticity of such norms. Using an assets-based approach (Kretzmann & McKnight, 1993), organizations and practitioners can place greater focus on foundational social norms that are compatible with water quality abatement. In this review, several studies noted normative definitions of ‘good farmers’ and ‘good neighbors’ that were rooted in values of reciprocity with peers and stewardship of the environment (Atwell et al., 2009b, 2009a; Evans et al., 2020; McGuire et al., 2013; Mills et al., 2017; Yoder & Chowdhury, 2018). When these social norms are called into question—as individuals and communities are instead labeled as polluters—a series of negotiations may proceed whereby individuals elicit feedback from their social environment to reestablish themselves as good peers (McGuire et al., 2013). It is through this process that new expectations— those supportive of BMP adoption— to be a ‘good farmer’ and a ‘good neighbor’ emerge.

Being a good neighbor is an overarching norm common to many communities and cultures, and norms supporting neighborly relations can be linked to expectations to perform specific management practices (Minato et al., 2010). For instance, in Indigo Valley, Australia, norms supporting reciprocity between neighbors facilitated a sense of obligation to control invasive plant species for newcomers in the community because invasive plants have a negative impact on their neighbors (Minato et al., 2010). In the case of water quality, it may be beneficial to highlight negative impacts that poor water quality, and more importantly, poor management practices have on the community as a whole. This approach harnesses existing community assets (e.g., existing social norms supporting reciprocity) that are compatible with water quality BMP adoption.

Along with the assets-based approach, it is important to recognize that productivity and conversation norms are not completely incompatible. Production-oriented farmers tended to comply with conservation norms when their peers drew on socially-acceptable motivations, including social and financial rewards (e.g., community awards, high yields, financial savings). Additionally, production-oriented farmers tended to look towards like-minded farmers to gauge what they ought to do regarding conservation. Like-minded yet innovative peers can provide the spark that others need to adopt BMPs (McGuire et al., 2013; Skaalsveen et al., 2020). These individuals increase perceived efficacy of adoption and lower the social sanctions associated with being out-of-step with peers (Morris et al., 2015). This highlights the importance of existing social networks, even if initially unsupportive of conservation goals, to the facilitation of conservation norms.

Existing social networks, such as farmer cooperatives and neighborhood associations, are structures through which social norms can be fostered. BMP adoption focused on individual farmers and individual residents often fail to recognize the individuals as being embedded in these communities of support. Taylor et al. (2015) suggest that conservation practices should be promoted and implemented for groups as well as individuals, which reduces the likelihood of farmers feeling ‘hung out’ when they choose to implement BMPs. Group-level BMP adoption reduces social sanctions driven by noncompliance with group practices. This approach recognizes the social infrastructure that communities have built to sustain and improve their livelihood and addresses barriers to BMP adoption.

It is pertinent to identify key individual and institutional actors involved in the facilitation and enforcement of social norms. I found that key individuals facilitated conservation norms by demonstrating their practices to others and communicating how land management ought to be done (McGuire et al., 2013; Skaalsveen et al., 2020). It may be beneficial to target these innovative residents and farmers and connect them with extension agents that can act as a bridge of information between the scientific community and land managers. That way, key individuals are equipped to share the most relevant information. Institutional actors also play a key role in the facilitation of conservation norms, and such institutional actors should be identified and supported. Daxini et al. (2019) found that HOAs are important facilitators of lawn-care norms, and they may not be as supportive of excessive fertilization as presumed. Again, it is important to understand the local context. Targeted interventions with unsupportive HOAs may also prove fruitful as these organizations are well-established and often trusted sources of information for local residents. Once institutional support is garnered, there is a higher likelihood that conservation norms will diffuse throughout the community through well-established systems of social sanctions and benefits.

## 3.5. Conclusion

This review reveals salient social norms influencing BMP adoption to improve water quality. I also identify situational factors and key individuals and organizations important to reproducing social norms supporting BMP adoption. Importantly, I highlight processes through which social norms supporting water quality conservation are mobilized. I hope that this review spurs further investigation into conflicting social norms supporting conservation and production, the intersection between social networks and facilitation of social norms, and mechanistic understanding of the association between natural spaces and the fostering of social norms supporting BMP adoption. I suggest that practitioners with a goal to increase BMP adoption should aim to work with the existing social networks and identify social norms that align with conservation goals. Future research can build from this work to develop policy instruments and intervention strategies that utilize social norms to promote water quality improvement.

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# CHAPTER 4. STORMWATER ON THE MARGINS: INFLUENCE OF RACE, GENDER, AND EDUCATION ON WILLINGNESS TO PARTICIPATE IN STORMWATER MANAGEMENT

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## 4.1. Abstract

Stormwater has immense impacts on urban flooding and water quality, leaving the marginalized and the impoverished disproportionately impacted by and vulnerable to stormwater hazards. However, the environmental health concerns of socially and economically marginalized individuals are largely underestimated. Through regression analysis of data from three longitudinal surveys, this article examines if and how an individual's race, gender, and education level help predict one's concern about and willingness to participate in stormwater management. We found that people of color, women, and less-educated respondents had a greater willingness to participate in stormwater management than White, male, and more-educated respondents, and their concern about local stormwater hazards drove their willingness to participate. Our analysis suggests that physical exposure and high vulnerability to stormwater hazards may shape an individual's concern about and willingness to participate in stormwater management.

## 4.2. Introduction

Urban stormwater has drawn water managers' attention because of its deleterious impacts on flooding and water quality in surrounding streams, rivers, lakes, and coastal zones (e.g., Meyer et al., 2005; O'Driscoll et al., 2010). Additionally, nuisance algal blooms and increased sediment loads can threaten drinking water reservoirs (Carmichael and Boyer, 2016; Gaffield et al., 2003). The magnitude of this problem is underscored as millions of people experience flood-related damage yearly, and the cost of mitigating stormwater externalities escalates (Brody et al., 2007). While flood and water quality hazards are a serious threat to urban communities worldwide, notably, these hazards disproportionately affect socially and economically marginalized communities. Women, the impoverished, and racially marginalized individuals are at the highest risk of flooding and impaired water quality and often have the highest barriers to recovery from emergencies (Enarson and Fordham, 2000; Liévanos, 2017; Qiang, 2019). Such an inequitable distribution of environmental degradation triggers broad social concerns about flooding, water quality, and ecological integrity and creates a unique socioecological problem that requires solutions across social and technical viewpoints.

While social inequalities are persistent in urban water systems, conventional management of stormwater is technocratic— centered on engineering strategies that convey water, sediment, and nutrients out of sight (Finewood, 2016). Technocratic governance hides stormwater's socioecological complexity, reinforces the public perception that stormwater governance is expert-driven, and, ultimately, isolates stormwater management from the public (Dhakal and Chevalier, 2016). The focus on technological solutions ignores the structural and institutional drivers of inequitable impacts and ultimately can perpetuate inequality throughout the socioecological system. Municipalities need broader approaches to stormwater management that engage communities across socioeconomic backgrounds— approaches that will improve access to stormwater management services and address the growing threats of climate change, urban growth, and socioeconomic inequality.

Scholars have shown that individuals' social and economic status is an important predictor of their concern about and participation in environmental management broadly, but stormwater management has been overlooked. A recent study examining a broad range of environmental concerns illustrates that diverse segments of the American public underestimated the environmental concerns of racially marginalized and impoverished individuals (Pearson et al., 2018). Despite public perception, scholars have illustrated that people of color, the impoverished, and women tend to be just as or more concerned about environmental issues than more socioeconomically privileged groups, especially about issues related to environmental racism and risk exposure (Lazri and Konisky, 2019; Macias, 2016a). Stormwater hazards align with traditional environmental racism issues, like toxic waste, regarding the inequitable distribution of vulnerability and outcomes (Debbage, 2019). Yet, notably, the interplay between inequitable experiences and technical decision-making and knowledge presents an additional complexity in stormwater management that is not completely understood. Daily experiences of stormwater hazards can raise public awareness of stormwater problems; alternatively, inaccessible technical knowledge and management can lead to the perception that stormwater is not a social and environmental problem and certainly not one that engages the public.

Using secondary data from a survey of individuals conducted in Charlotte, North Carolina, USA, we examined whether and how individuals' race, gender, and education level help predict their willingness to participate in stormwater management. Additionally, we investigated how these patterns change based on different forms of participation in stormwater management, including individuals' willingness to volunteer for stream cleanups and willingness to pay more in stormwater fees.

## 4.3. Literature Review

### 4.3.1. Social Marginalization and Concern for Environmental Management

Numerous empirical studies have shown that a person's race, class, and gender are important predictors of their environmental concern. Environmental concern is a broad construct that is often conceptualized as a general attitude towards environmental protection. More recently, sense of environmental risk has been included as a key facet of environmental concern in recognition of the direct influence of environmental threats on individuals' attitudes towards the environment (Mohai and Bryant 1998; Macias, 2016a).

Early literature on this topic has suggested that Black people are less concerned about environmental degradation than White people (Hershey and Hill, 1977; Hohm, 1976; Kreger, 1973). For instance, Hohm (1976) conducted a survey on the relationship between one's race and concern for air pollution and found that White respondents had a higher perception of the severity of air pollution and related health risks than Black respondents. The author's explanation relied on Maslow's hierarchy of needs theory, which supports the claim that economically disadvantaged groups—assumed to be the case for Black respondents— lack concern for the environment because they focus on fundamental needs like food, housing security, and healthcare (Maslow, 1954). Critics, however, have argued that Maslow's hierarchy of needs fails to recognize the dependence of basic needs on environmental conditions (Mohai and Bryant, 1998). Water pollution and stormwater flooding can threaten one's housing security and access to safe drinking water. Others find a lack of support for Maslow's hierarchy of needs because race is a significant determinant of environmental concern regardless of socioeconomic status (Hershey and Hill, 1977). With regard to a survey of young adults on concern for litter, land preservation, and endangered species protection, Hershey and Hill (1977) instead argued that White youth are more concerned about environmental pollution than Black youth due to disparate subcultural norms. At the time of the study, the mainstream environmental movement advocated economic downscaling, which seemed to threaten economic advancement goals in the Black community. Researchers suggested that environmental support in Black communities would decline during economic downturns because the economy would be prioritized. However, Jones and Carter (1994) challenged this claim by showing that Black and White people equally supported higher national spending on environmental protection throughout the 1970s and 1980s, and this support was unaffected by economic downturns.

A more recent wave of literature has challenged the conclusion that racially marginalized people are less concerned about the environment than White people. Empirical studies began to illustrate that Black people were just as or more concerned about the environment than their wealthy and White counterparts (Caron, 1989; Jones, 1998; Mohai and Bryant, 1998). Through an empirical analysis, Mohai and Bryant (1998) investigated three theories that could explain an environmental concern gap between Black and White Americans. The first is the environmental deprivation theory—communities of color experience greater environmental burden, which subsequently increases their environmental concern. The second hypothesis is hierarchy of needs, and the third considers cultural differences between Black and White people. Cultural differences refer to disparate sociocultural experiences of nature—for example, the Black community's (assumed) negative environmental attitudes are conditioned on their lack of access to natural spaces like national forests and beaches due to racial segregation (Finney, 2014; Taylor, 1989). Mohai and Bryant (1998) tested whether these theories applied to a range of environmental concerns, including nature preservation, global warming, plastic waste, and air pollution. Their findings did not support the hierarchy of needs hypothesis or the cultural difference hypothesis because African Americans and low-income respondents were equally concerned as White and wealthy respondents about most environmental issues. Rather, their findings support the environmental deprivation hypothesis. They found that Black and White people are similarly concerned about most environmental issues, but in reference to issues that disproportionately affect Black populations, like industrial waste, Black people's proximity to these problems drove their heightened concern.

Some scholars have explored the influence of environmental injustices on concern about the environment (Jones and Rainey, 2006; Lazri and Konisky, 2019). Environmental justice activists and scholars have not only revealed that people of color and those with low-income are disproportionately burdened by environmental degradation but also stressed that structural forms of racism, classism, and sexism create and sustain inequitable patterns (Arp and Boeckelman, 1997; Bullard, 2008; Hines, 2001). Moreover, environmental degradation often reflects legacies of structural racism that uniquely advantage White and wealthy people, such as housing discrimination and historical redlining practices (Pulido, 2000). Jones and Rainey (2006) explored the impact of feelings of environmental injustice on environmental concern. Their findings support previous empirical studies that illustrate a heightened environmental concern in Black respondents compared to White respondents. Furthermore, they illustrate that feelings of environmental injustice drive concern: residents who felt that they were unfairly exposed to detrimental environmental conditions were more concerned. Such experiences of poor environmental conditions in communal settings can cause people to be more conscious of environmental injustices and subsequently participate in reporting and challenging these injustices (Young and Subramaniam, 2017).

While most empirical studies focus on race, a gender dimension reveals patterns at the intersection of gender and race. Researchers have quantified that women tend to have greater environmental concern than men (Blocker and Eckberg, 1997; Chakraborty et al., 2017; Gifford and Nilsson, 2014; Tikka et al., 2000; Uyeki and Holland, 2000). In an investigation of race and gender, Kalof et al. (2000) found that White respondents reported significantly lower environmental values and beliefs than Black and Latino respondents, but they also found that gender differences in the National Environmental Paradigm scale (NEP; a measure of general environmental attitudes) only existed for White respondents. White women scored significantly higher on the NEP scale than White men. Others have found a specific "White male effect" (Brent, 2004) on environmental concern and attribute this to White men's perception that their environmental risk is low and their institutional support is high, which reduces their concern about environmental protection and risk.

The most recent wave of research extends the literature to include multiple racial and ethnic groups in nationally representative samples. Scholars have found that people of color (including multi-ethnic Latino, Asian, and African Americans) tend to be more concerned than White people about environmental issues related to environmental risks (Macias, 2016a) and environmental racism (Lazri and Konisky, 2019), such as air and water pollution. However, outside of environmental risk, people of color show similar or greater concern for locally and globally relevant environmental issues compared to White individuals (Lazri and Konisky, 2019). These studies also show that higher income (Lazri and Konisky, 2019; Macias, 2016a) and education levels (Lazri and Konisky, 2019) are correlated with lower environmental concern, and women show higher levels of environmental concern than men (Macias, 2016b). Even when controlling for other socioeconomic factors, race is still a significant predictor of environmental concern (Macias, 2016a), highlighting the interconnected but differentiated effects of marginalization based on race, gender, and class.

Few studies support the claim that racially marginalized and low-income individuals are less concerned about the environment; however, this perception is popular in the American public (Pearson et al., 2018). This public perception likely stems from conflation of environmental concern and perceived participation in the environmental movement. The mainstream environmental movement is largely White and upper class (Taylor, 2015), leading to the inference that environmental values and concerns are also White and upper class. Distortion of environmental interest in marginalized communities largely undermines the growing popularity of the environmental justice movement and places these communities in positions where stakeholders assume their disinterest in environmental governance (Finewood, 2016). Moreover, these assumptions can substantially derail coalition building and equitable decision-making.

### 4.3.2. From Concern to Participatory Intentions

Are differences in concern extended to participation? Substantially fewer studies have examined which individuals, based on gender, race, and education level, are willing to participate in environmentally conscious ways. Attitudes and behaviors exhibit a tenuous relationship, and research to date is inconclusive regarding the influence of social marginalization on environmental behaviors. Scholars have discussed both participation and willingness to participate in environmentally conscious behaviors. Here we conceptualize participation as actions that individuals take to improve environmental quality or mitigate environmental problems and willingness to participate as an intention to perform these behaviors. Those who express the intention to perform environmentally conscious behaviors are more likely than others to actually perform those behaviors (Hines et al., 1987).

Content validity on measures of participation and willingness to participate in environmentally conscious behaviors has been a consistent issue. Environmentally conscious behaviors frequently examined in the literature often tap into underlying issues of disproportionate economic opportunity as well as social and physical resources. For example, the frequency of recycling is often higher with individuals who are White, earn a higher income, and have a higher education level (Johnson et al., 2004; Macias, 2016a). However, Laidley (2013) determined that a significant predictor of recycling behavior is access to curbside recycling programs highlighting accessibility as an underlying issue. Likewise, individuals with higher income and education levels are often more willing to pay for environmental management (Chui and Ngai, 2016; Macias, 2016a; Newburn and Alberini, 2016). Other measures, such as purchasing chemical-free products, organic foods, and electric vehicles, show similar trends with an individual's income and education level (Laidley, 2013; Macias, 2016b). These high-status consumptive behaviors are strongly tied to social class and can just as easily align with attitudes of class distinction as they do with environmental concerns (Kennedy and Givens, 2019).

Much of the research on race and environmentally conscious behavior has shown that historically marginalized individuals participate less in environmentally conscious behaviors; however, the limitations of current measures do not capture the complexity of these behaviors. A literature review illustrates that Black and Latino people show high concern for national parks and natural resources, but they are highly underrepresented in outdoor recreational activities in parks (Roberts and Rodriguez, 2008). Likewise, marginalized individuals are grossly underrepresented in mainstream environmental groups, with most members and leaders being White and upper class (Taylor, 2014). A more nuanced look shows that Black and Latino individuals tend to lack awareness of recreational opportunities in national parks and perceive these spaces as unwelcoming and discriminatory towards communities of color (Roberts and Rodriguez, 2008). Similarly, exclusionary practices within mainstream environmental groups and failures to address the needs of working-class communities of color deter participation in mainstream environmentalism (Clarke and Agyeman, 2011; Hoover, 2017). Ultimately, measures of participation can capture broader constructs than intended, which can lead to significant biases.

Given the scant body of work on this topic and the complexity of measuring environmentally conscious behaviors and intentions to participate in those behaviors, we measure intention to participate in environmentally conscious behaviors in two ways: willingness to pay and willingness to volunteer. While willingness to pay taps into issues of economic opportunity, willingness to volunteer provides an alternate measure of participation that does not require economic investment. Hands-on and practical stewardship activities help communities closely relate to their local environments but are often ignored in discussions of public participation (but *see* Ando et al., 2020). Such actions broaden the scope of what we imagine as participation in environmental management (Eden and Bear, 2012).

### 4.3.3. Urban Stormwater Perceptions, Experiences, and Participation

Stormwater impacts and recovery are not distributed evenly, often with the most marginalized experiencing the greatest harm and/or vulnerability. In the United States, the impoverished, unemployed, and underinsured are more likely to live in flood zones than outside, and this pattern is more prominent in inland areas than coastal zones (Qiang, 2019). These populations are at a higher risk of physical exposure to stormwater hazards. In addition to physical exposure, we also consider vulnerability to stormwater hazards, which accounts for people's ability to recover from disasters. Physical exposure and high vulnerability to stormwater hazards can lead to high stormwater risk perception in socially and economically marginalized communities.

Scholars have found that women, those with low-income, and people of color have a higher flood risk perception than men, those with high-income, and White people across multiple urban regions (Harlan et al., 2019). Higher risk perception and accounts of flood-related experiences in marginalized communities reflect their social vulnerability. Recovery from large storms can consume expendable income, and damage to transportation systems can leave many without transit to work and school—a loss of income that can be debilitating. Furthermore, gender role disparities in the private/domestic sphere lead to women bearing the brunt of flood recovery tasks. Women are often expected to care for sick and elderly family members, apply for aid from public services, and women-dominated service industries are less likely to provide job security, childcare, and uninterrupted paychecks during flood events (Enarson and Fordham, 2000; Walker and Burningham, 2011). Moreover, communities of color that are historically underserved by the government can lack trust in government-issued recovery services (Harlan et al., 2019; Pradhananga et al., 2019). This lack of trust is often a result of oppressive relationships with government officials that fail to meet the community's basic needs. Lack of equitable collaboration between institutions and communities leaves these communities isolated and vulnerable to stormwater risks.

Conventional stormwater management focused on technical solutions to flooding and water quality issues assumes that engineering approaches will result in equitable service provision (Carriquiry et al., 2020). Such a historic top-down model ignores the multiple social and environmental objectives of stormwater management and can drive a wedge between managers and the public. Some scholars have shown that water-related knowledge is positively associated with environmentally conscious behaviors suggesting that lack of knowledge can be a barrier to participation in water management (Dean et al., 2016). For instance, there is a lack of understanding about how the public's actions, like pet waste and lawn fertilizing, negatively impact water quality (Giacalone et al., 2010). Whereas others note how stormwater governance relies heavily on technical expertise, which can impede broader forms of public participation in decision-making and adoption of mitigative practices on privately owned land (Cousins, 2018). For example, stormwater agencies in Seattle, WA, Portland, OR, Philadelphia, PA, Chicago, IL, and Syracuse, NY, privileged technical expertise related to hydrological control of water and lacked formal structures for residents to participate in decision-making (Dhakal and Chevalier, 2016).

The fact that stormwater impacts and benefits are not equitably distributed calls for social and political processes to be incorporated into sustainable stormwater management programs (Hillman, 2004). There is now recognition that multiple stakeholders need to be involved in stormwater management, including residents, homeowner associations, scientists and engineers, and regulatory officials to ensure sustainable and equitable distribution of stormwater risks and benefits (Carriquiry et al., 2020). Lack of community participation in decision-making has been cited as one of the most identified barriers to building sustainable stormwater management systems (Brown and Farrelly, 2009). Community participation, especially that of the most marginalized and vulnerable individuals, is pivotal to sustainable and equitable stormwater management.

In this paper, we examined whether an individual's race, gender, and education level help predict their willingness to pay more in stormwater fees and willingness to volunteer for stream cleanups. We predicted that this work would support the environmental deprivation theory (Mohai and Bryant, 1998)—socially and racially marginalized peoples will be more exposed to stormwater hazards leading to greater concern about stormwater, and their heightened concern will lead to an intention to alleviate their conditions through participatory activities. We recognize that some environmentally conscious behaviors have higher barriers to implementation than others; therefore, we predicted that behaviors with lower barriers to action would garner more support from socially and economically marginalized peoples. For instance, environmental behaviors that require financial support will have lower buy-in from residents with lower expendable income. In contrast, activities that require hands-on participation, such as stream cleanups, directly align with other individual and community-level needs such as physical activity, environmental education, and community beautification and will garner greater support from marginalized communities.

## 4.4. Methods

### 4.4.1. Study Area and Data Collection

Three longitudinal surveys conducted in Charlotte, North Carolina, in 2014, 2016, and 2017 provide data for this study. Charlotte has a growing urban population and economy; however, in contrast to the growing prosperity, in 2015, 17% of Charlotte's residents lived below the poverty line, and Black, Native, and Latino Americans are overrepresented in this population (U.S. Census Bureau, 2015). Compared to other cities in the contiguous U.S., Charlotte has the 5th flashiest streamflow, which is indicative of the high frequency of flash flooding events (Smith and Smith, 2015). Charlotte is predicted to have a higher risk for drought and more extreme storms in future climate change scenarios (Kunkel et al., 2020). Charlotte residents who are historically marginalized by their race, gender, and class are increasingly at risk of and vulnerable to floods and stormwater pollution because they are more likely to reside in flood zones (Debbage, 2019).

The data for this paper was drawn from surveys conducted by Charlotte-Mecklenburg Storm Water Services (CMSWS), in the three years noted above (2014, 2016, and 2017), on community perception and opinion of stormwater in Mecklenburg County and the city of Charlotte. In 2014, the University of North Carolina (UNC) Charlotte's Energy and Environmental Assistance Office administered phone surveys by randomly sampling a list of purchased landline and cell phone numbers of Mecklenburg County residents. Survey administers sampled until 400 surveys were 100% complete. We do not have access to the number of attempted phone calls, and therefore we cannot calculate a response rate. The 2016 and 2017 surveys were administered by The Jackson Group, a private survey company. We accessed the data from Charlotte Stormwater Services in the summer of 2018.

### 4.4.2. Variables and Measures

Details of the key measures used to operationalize each variable in our analysis are discussed below. The complete list of survey measures can be found in Appendix E.

*Outcome Variables*

We used three measures as outcomes: *willingness to volunteer*, *willingness to pay*, and *concern for flooding*. Willingness to volunteer is a single item related to willingness to clean up a local stream. Willingness to pay is a summed scale of two item measures ($\alpha_{2014}$= 0.69; $\alpha_{2016}$= 0.83; $\alpha_{2017}$= 0.85; Cronbach's α is a measure of internal consistency). Each item relates to willingness to pay more in stormwater fees to improve flooding or water quality. Willingness to volunteer was only measured in 2014, and willingness to pay was measured in 2014, 2016, and 2017. *Concern for flooding* is a summed scale of two item measures that represent respondents' concern with local flooding of buildings and roads ($\alpha_{2014}$= 0.77; $\alpha_{2016}$= 0.77; $\alpha_{2017}$= 0.69). These variables were originally measured on a Likert-scale (1= don't know, 2= strongly disagree, 3= disagree somewhat, 4= agree somewhat, 5=agree strongly). We recoded all variables to a 4-point scale (1= strongly disagree, 2= disagree somewhat, 3= agree somewhat, 4=agree strongly) prior to analysis. "Don't know" responses were not included in our analysis.

*Predictor Variables*

The predictor variables of interest in the study are *race/ethnicity*, *gender*, and *education level*. Race/ethnicity was measured as a nominal variable— including non-Latino White (reference level), Latino of any race, non-Latino Black/African American, non-Latino Asian American and Pacific Islander, non-Latino multi-racial, and other race/ethnicity. Gender was measured as a dichotomous variable—female (reference level) and male. Education level was measured as an ordinal variable that varies from "less than high school" to "graduate study."

Control variables are *exposure to stormwater ads, knowledge of stormwater, age, homeownership, residence in a flood zone,* and *time*. Exposure to stormwater ads is a summed scale of five item measures related to whether respondents have seen or heard recent Charlotte stormwater advertisements ($\alpha_{2014}$ = 0.61; $\alpha_{2016}$ = 0.83; $\alpha_{2017}$ = 0.80). This scale measures exposure to informal awareness-raising campaigns that often have short and digestible messages to the public about stormwater. CMSWS runs educational advertisements on stormwater and flood awareness, such as the "Turn Around, Don't Drown" campaign. Stormwater advertisement campaigns were not the same content-wise across all years; therefore, exposure to advertisements represents a count of advertisements that a respondent has seen in a given year. Knowledge of stormwater is a measure of a respondent's technical knowledge of stormwater treatment. We developed a summed scale of two items representing awareness that stormwater directly drains to local streams without treatment facilities ($\alpha_{2016}$= 0.64; $\alpha_{2017}$= 0.61). Knowledge of stormwater was only considered in our analysis for 2016 and 2017 due to inconsistent item measurements, poorly worded survey items, and lack of reliability in 2014.

We replaced missing values in the independent variables if the variables had less than 10% of values missing. We replaced missing values for education level, exposure to stormwater ads, age, residence in a flood zone, homeownership, and knowledge of stormwater. We performed multiple imputations using chained equations to replace "missing at random" (MAR) values of the independent variables (Graham, 2009).

### 4.4.3. Modeling and Analytic Strategy

Data were analyzed using R Programming (R Core Team, 2013) and PROCESS in SPSS Statistics 26 (Hayes, 2017). First, we examined whether socially marginalized groups reported higher exposure to flood zones using a test of equal proportions. Two additional objectives were assessed using mediation analysis: 1) the direct and indirect effects of race, gender, and education level on an individual's willingness to participate, and 2) whether the type of participation influences the association between participation and race, gender, and education.

We used a simple attitude-behavior model to frame our investigation. Attitudes influence behavioral intentions (Ajzen, 1991), and knowledge as well as correlates of knowledge modify attitudes rather than directly influence behavioral intentions (Kollmuss and Agyeman, 2002). In this framework, knowledge (knowledge of stormwater and exposure to educational advertisements) informs attitudes (concern for flooding), and attitudes predict willingness to participate (Fig. 4.1). Then, we used a more exploratory approach, rooted in empirical evidence of the predictive pathways, to test for direct and indirect relationships between race, gender, and education level and willingness to participate.

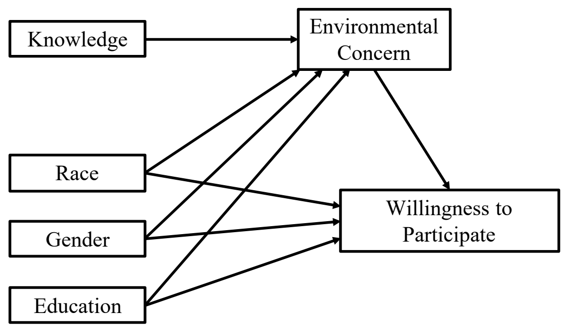


Figure 4.1. Conceptual model of willingness to participate based on a simple attitude-behavior model. An individual's knowledge of stormwater informs their concern about stormwater, and concern about stormwater influences their willingness to participate. Using this framework, we test for direct and indirect influences of an individual's race, gender, and education level on their willingness to participate.

Prior research has shown that an individual's race, gender, and education level influence *both* environmental attitudes and intentions to participate in environmentally conscious behaviors, and path models allowed us to investigate both direct and indirect pathways. We tested for two possibilities: (1) race, gender, and education level have an indirect effect on willingness to participate, which implies that their effect is mediated by differential concerns for flooding; or (2) the effect is predominantly direct, which implies that the effect is largely unrelated to one's concern for flooding. Empirical studies point towards the first explanation (Botetzagias et al., 2015); however, a significant direct effect can also imply that we did not capture the specific attitude (an unmeasured mediator) that mediates the relationship between individuals' race, gender, and education and their willingness to participate.

We performed mediation analysis via OLS (Ordinary Least Squares) path analysis— a causal model in which a series of multiple regressions are estimated to examine the effect of a set of variables on a specified outcome via multiple mechanisms. The general equations of an OLS path analysis are as follows (Hayes, 2017):

$$M = i_M + a_1X_1 + a_2X_2 + \ldots + a_kX_k + e_M \tag{4.1}$$

$$Y = i_Y + c_1X_1 + c_2X_2 + \ldots + c_kX_k + bM + e_Y \quad (4.2)$$

where $M$ is the mediating effect, $i$ is the constant, $X_i$ are predictor variables, $e$ is error, and $Y$ is the outcome variable. Equations 4.1 and 4.2 estimate the direct effect coefficients that predict $M$ and $Y$ outcome variables. The indirect effect of $X$ on $Y$ through $M$ is the product of two effects: $a_ib$. We investigated two paths: one where race, gender, and education directly influence respondents' willingness to participate, and another where race, gender, and education indirectly influence respondents' willingness to participate through a mediator variable, concern for flooding. While we included all independent variables and controls in the model, we only calculated the indirect effect of race, gender, and education level as these are the predictor variables of interest. We estimated bootstrapped confidence intervals for the indirect effect based on 5,000 bootstrapped samples of the indirect effects. We performed two mediation analyses with our outcome variables of interest: willingness to pay and willingness to volunteer. Willingness to volunteer was only assessed in the 2014 survey, and willingness to pay utilizes a pooled dataset including 2014, 2016, and 2017 surveys. We ran an additional path analysis with a pooled dataset, including 2016 and 2017 samples, because this dataset has a reliable measure of stormwater knowledge.

To ensure that our model choice was a good fit, we checked the assumptions of multiple regression (Appendix F). We also compared the results of the multiple regression models to proportional odds models to ensure that our assumption to treat ordinal response variables as continuous would not significantly influence our results (Appendix G).

## 4.5. Results

### 4.5.1. Sampling Characteristics

Respondents in the 2014 sample resembled the demographics of Mecklenburg County in terms of race (47% White, 13% Latino of any race, 31% Black, 6% Asian, 2% Multi-racial), gender (52% Female), and education level (average educational attainment is "some college or associate's degree") (U.S. Census Bureau, 2019). In 2016 and 2017, there was a higher proportion of White respondents (64%) compared to 2014 (58%), and Latino, Black, and Asian American respondents were underrepresented (Table 4.1). The 'Other' racial category includes individuals who refused to respond to the race and ethnicity questions and individuals belonging to racial/ethnic groups with cumulatively fewer than 20 representatives across the three surveys. Gender and education level of the respondents were representative of the population in all surveys. The mean age of survey respondents increased over time from 35-44 to 45-54 years old. Additionally, in 2014, 70% of respondents were homeowners, which is higher than the city average of 56% (U.S. Census Bureau, 2019).

Table 4.1. Sample characteristics in 2014, 2016, and 2017 surveys.

| | 2014 | | | | 2016 | | | | 2017 | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Range | Mean | SD | N | Range | Mean | SD | N | Range | Mean | SD | N |
| *Outcome Variables* | | | | | | | | | | | | |
| Willingness to Pay | 1-7 | 4.86 | 1.87 | 402 | 1-7 | 3.69 | 1.83 | 393 | 1-7 | 3.57 | 1.82 | 363 |
| Willingness to Volunteer | 1-4 | 3.02 | 1.06 | 403 | | | | | | | | |
| Concern for flooding | 1-7 | 4.28 | 1.95 | 397 | 1-7 | 4.90 | 1.49 | 409 | 1-7 | 4.80 | 1.38 | 394 |
| *Predictor Variables* | | | | | | | | | | | | |
| Race | | | | | | | | | | | | |
| White | 0-1 | 0.58 | | 233 | 0-1 | 0.64 | | 264 | 0-1 | 0.64 | | 255 |
| Latino | 0-1 | 0.07 | | 30 | 0-1 | 0.07 | | 29 | 0-1 | 0.03 | | 12 |
| Black/African American | 0-1 | 0.24 | | 98 | 0-1 | 0.18 | | 76 | 0-1 | 0.11 | | 44 |
| Asian or Pacific Islander | 0-1 | 0.05 | | 20 | 0-1 | 0.04 | | 15 | 0-1 | 0.02 | | 9 |
| Multi-racial | 0-1 | 0.04 | | 17 | 0-1 | 0.02 | | 7 | 0-1 | 0.03 | | 12 |
| Other | 0-1 | 0.01 | | 5 | 0-1 | 0.05 | | 22 | 0-1 | 0.17 | | 68 |
| Gender (Female = 0) | 0-1 | 0.47 | | 402 | 0-1 | 0.52 | | 413 | 0-1 | 0.52 | | 359 |
| Education | 1-5 | 3.61 | 1.11 | 403 | 1-5 | 3.67 | 1.08 | 413 | 1-5 | 3.83 | 1.05 | 359 |
| *Control Variables* | | | | | | | | | | | | |
| Exposure to ads | 1-6 | 2.77 | 1.43 | 403 | 1-6 | 2.37 | 1.71 | 412 | 1-6 | 2.05 | 1.51 | 385 |
| Knowledge of stormwater | | | | | 1-3 | 2.46 | 0.75 | 411 | 1-3 | 2.47 | 0.75 | 395 |
| Age | 1-6 | 3.96 | 1.55 | 402 | 1-6 | 4.37 | 1.47 | 413 | 1-6 | 4.63 | 1.39 | 358 |
| Residence in Flood Zone | 0-1 | 0.06 | | 403 | 0-1 | 0.04 | | 413 | 0-1 | 0.05 | | 390 |
| Homeownership (Rent =0) | 0-1 | 0.70 | | 403 | | | | | | | | |

### 4.5.2. Flood Zone Residence

A test of equal proportions revealed that racially marginalized respondents were significantly overrepresented in flood zones (Fig. 4.2). Non-White respondents represented roughly half of the residents that claimed to live in a flood zone, while they were 30% of the population outside of flood zones. There were no significant differences in the proportion of respondents that resided in flood zones compared to that outside of flood zones by gender and education level.

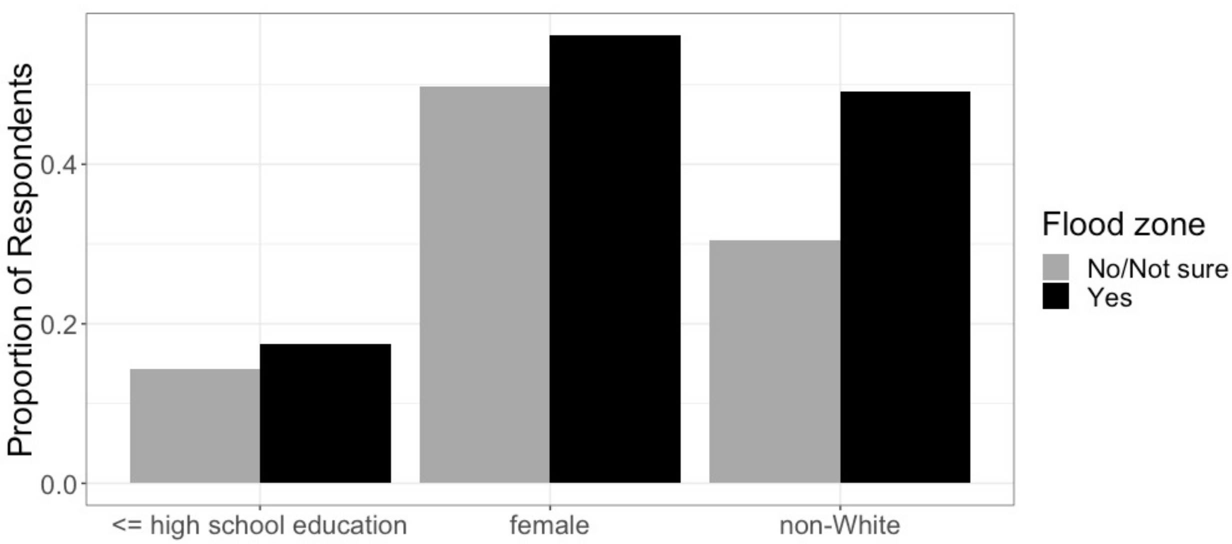


Figure 4.2. Proportion of respondents who reported residence in a flood zone by sociodemographic variables. We performed a test of equal proportions to assess significant differences. Non-White respondents are significantly overrepresented in flood zone residences (49% vs. 30%, $p < 0.01$). There were no significant differences between the proportions of women within and outside of flood zones (56% vs. 48%, $p = 0.42$) and the proportions of respondents with less than or equal to high school education level within and outside of flood zones (18% vs. 14%, $p = 0.63$).

### 4.5.3. Dependent Variable 1: Willingness to Volunteer

The mediation analysis indicated that racially marginalized respondents, on average, were more concerned for flooding and more willing to volunteer than White respondents; however, respondents' concern for flooding was not significantly associated with their willingness to volunteer. Latino, Black, and multi-racial respondents were more concerned for flooding near their homes and businesses compared to White respondents (reference level) when all other variables are held constant ( $a = 1.34, S.E. = 0.39, p < 0.01$; $a = 0.84$, $S.E. = 0.23, p < 0.01$; $a = 0.86$, $S.E. = 0.47$, $p = 0.07$, respectively; Table 4.2). Latino respondents, on average, were more willing to volunteer for stream cleanups than White respondents ($c = 0.84, S.E. = 0.22, p < 0.01$; Table 4.2). However, respondents' concern for flooding was not significantly associated with their willingness to volunteer ($b = 0.04, S.E. = 0.03, p = 0.15$; Table 4.2), resulting in a predominantly direct effect of race on willingness to volunteer. This indicates that the heightened willingness to volunteer in Latino respondents was not associated with their concern for flooding. We did not find a consistent response across all racial groups. Asian American and Pacific Islander respondents were, on average, just as concerned for flooding as White respondents ( $a = 0.35, S.E. = 0.44, p = 0.42$; Table 4.2). Those with lower education, on average, were more concerned for flooding ( $a = -0.17, S.E. = 0.09, p = 0.06$ ). Gender was not significantly associated with respondents' concern for flooding or willingness to volunteer ($a = -0.10, S.E. = 0.19, p = 0.60$; $c = 0.08, S.E. = 0.11, p = 0.45$, respectively; Table 4.2).

Our results illustrate that race and education level were associated with respondents' concern for flooding and willingness to volunteer after controlling for other factors. Among the controls, concern for flooding was significantly higher among older respondents, those living within a flood zone, renters, and respondents who have seen more stormwater advertisements (Table 4.2). In Charlotte, stormwater fees are decided by the amount of impervious land on one's property; therefore, it is likely that homeowners pay more in stormwater fees than renters. Also, homeowners receive stormwater bills directly, while for some renters, water and sewage fees are included in the rental payment. Willingness to volunteer was significantly higher among respondents living within a flood zone and younger respondents (Table 4.2).

Table 4.2. OLS regression coefficients. Following equations (1) and (2), concern for flooding is the mediating variable (M), and willingness to volunteer (Volunteer) is the outcome variable (Y). Standard errors of the direct effect are presented in parentheses. Indirect effect coefficients ($a_i b$) are presented with the bootstrapped 95% confidence interval in brackets.

| Independent Variable (IV) | Direct Effects | | | | Indirect Effects |
|---|---|---|---|---|---|
| | IV → Concern | | IV → Volunteer | | IV → Concern → Volunteer |
| | $a$ | $p$ | $c$ | $p$ | $a_i b$ |
| Race (White = 0) | | | | | |
| Latino | 1.34 (0.39) | <0.01*** | 0.84 (0.22) | <0.01*** | 0.05 [-0.03, 0.15] |
| Black/African American | 0.84 (0.23) | <0.01*** | 0.03 (0.13) | 0.83 | 0.03 [-0.02, 0.09] |
| Asian or Pacific Islander | 0.35 (0.44) | 0.42 | 0.01 (0.24) | 0.99 | 0.01 [-0.02, 0.06] |
| Multi-racial | 0.86 (0.47) | 0.07* | -0.01 (0.26) | 0.96 | 0.03 [-0.02, 0.11] |
| Other | -0.47 (0.83) | 0.58 | 0.03 (0.46) | 0.95 | -0.02 [-0.12, 0.07] |
| Gender (Female = 0) | -0.10 (0.19) | 0.60 | 0.08 (0.11) | 0.45 | -0.004 [-0.03, 0.01] |
| Education | -0.17 (0.09) | 0.06* | 0.07 (0.05) | 0.18 | -0.007 [-0.02, 0.004] |
| Age | 0.15 (0.07) | 0.03** | -0.11 (.04) | <0.01*** | |
| Flood zone | 1.11 (0.40) | <0.01*** | 0.55 (0.22) | 0.01*** | |
| Homeownership (Rent = 0) | -0.38 (0.23) | 0.09* | 0.08 (0.12) | 0.51 | |
| Exposure to ads | 0.25 (0.07) | <0.01*** | -- | -- | |
| Concern for flooding ($b$) | -- | | 0.04 (0.03) | 0.15 | |
| $R^2$ | 0.14 | | 0.10 | | |
| MSE | 3.36 | | 1.03 | | |
| F Statistic | 5.89*** (df = 11) | | 3.68*** (df = 11) | | |
| N | 396 | | 396 | | |

*Note:* *p<0.1; **p<0.05; ***p<0.01
**Bold** indicates that the confidence interval for the indirect estimate does not contain zero
Letters *a, b*, and *c* indicate coefficients displayed in Equations (1) and (2)

### 4.5.4. Dependent Variable 2: Willingness to Pay

We found that racially marginalized respondents were more concerned for flooding, and, in contrast to willingness to volunteer, respondents' concern for flooding was positively associated with their willingness to pay increased stormwater fees (Table 4.3). On average, Latino, Black, and multi-racial respondents were more concerned about flooding compared to White respondents when all other variables are held constant ($a = 1.18, S.E. = 0.22, p < 0.01$; $a = 0.51$, $S.E. = 0.13, p < 0.01$; $a = 0.49$, $S.E. = 0.27$, $p = 0.07$, respectively; Table 4.3). In turn, respondents who were more concerned about flooding expressed a greater willingness to pay ($b = 0.24, S.E. = 0.03, < 0.01$). Results also indicate that the effect of race on willingness to pay was fully mediated by concern for flooding. Full mediation occurs when an indirect effect is present without a significant direct effect (Zhao et al., 2010), and in this case, indicates that Latino and Black respondents' heightened concern for flooding was significantly and positively associated with their willingness to pay. We also found that women (reference level) and those with lower education, on average, were more concerned about flooding ($a = -0.22, S.E. = 0.10, p = 0.02$; $a = -0.15$, $S.E. = 0.05, p < 0.01$, respectively; Table 4.3), and, in turn, their concern for flooding was positively associated with their willingness to pay. The effect of gender on willingness to pay was fully mediated by concern for flooding, as indicated by the presence of an indirect effect without a significant direct effect.

We observed competitive mediation in reference to education level: direct ($c = 0.08, S.E. = 0.05, p = 0.11$; Table 4.3) and indirect ($ab = -0.04$, $[upper\ limit, lower\ limit] = [-0.06, -0.01]$; Table 4.3) effects exist but in opposite directions (Zhao et al., 2010). This result suggests multiple mechanisms by which education influences a respondents' willingness to pay. The indirect effect suggests that respondents with lower education were more concerned about flooding, and their concern positively influenced their willingness to pay. After controlling for the indirect effect of concern for flooding, there remains an effect of education on willingness to pay. This effect works in the opposite direction: those with higher education were more willing to pay, but their concern for flooding did not drive their willingness to pay. Of the controls, older respondents, those living within flood zones, and respondents with greater exposure to stormwater advertisements were more concerned about flooding. Willingness to pay was significantly higher among younger respondents (Table 4.3). Interestingly, we found that concern for flooding increased over time ($a = 0.25, S.E. = 0.04, p < 0.01$), but willingness to pay decreased over time ($c = -0.43, S.E. = 0.05, p < 0.01$).

We also conducted an OLS path analysis with the pooled samples from 2016 and 2017, excluding data from 2014. The purpose of this analysis was to test the influence of knowledge about stormwater on respondents' concern for flooding and willingness to pay. The results of this analysis were similar to the findings reported in Table 4.3 and additionally illustrated that knowledge about stormwater was not significantly associated with respondents' concern for flooding ($a = 0.08, S.E. = 0.07, p = 0.28$; Appendix H).

Table 4.3. OLS regression coefficients. Following equations (1) and (2), concern for flooding is the mediating variable (M), and willingness to pay (WTP) is the outcome variable (Y). Standard errors of the direct effect are presented in parentheses. Indirect effect coefficients ($a_i b$) are presented with the bootstrapped 95% confidence interval in brackets.

| Independent Variable (IV) | Direct Effects | | | | Indirect Effects |
|---|---|---|---|---|---|
| | IV → Concern | | IV → WTP | | IV → Concern → WTP |
| | $a$ | p | $c$ | p | $a_i b$ |
| Race (White = 0) | | | | | |
| Latino | 1.18 (0.22) | $<0.01^{***}$ | 0.02 (0.25) | 0.95 | **0.28 [0.15, 0.43]** |
| Black/African American | 0.51 (0.13) | $<0.01^{***}$ | 0.11 (0.15) | 0.45 | **0.12 [0.05, 0.20]** |
| Asian or Pacific Islander | 0.27 (0.25) | 0.27 | 0.01 (0.28) | 0.98 | 0.07 [-0.04, 0.18] |
| Multi-racial | 0.49 (0.27) | $0.07^{*}$ | -0.29 (0.31) | 0.35 | 0.12 [-0.03, 0.27] |
| Other | -0.05 (0.24) | 0.83 | -0.56 (0.27) | $0.04^{**}$ | -0.01 [-0.13, 0.10] |
| Gender (Female = 0) | -0.22 (0.10) | $0.02^{**}$ | -0.06 (0.11) | 0.60 | **-0.05 [-0.10, -0.007]** |
| Education | -0.15 (0.05) | $<0.01^{***}$ | 0.08 (0.05) | 0.11 | **-0.04 [-0.06, -0.01]** |
| Age | 0.10 (0.03) | $<0.01^{***}$ | -0.22 (0.04) | $<0.01^{***}$ | |
| Flood zone | 0.69 (0.22) | $<0.01^{***}$ | 0.05 (0.25) | 0.83 | |
| Time | 0.25 (0.04) | $<0.01^{***}$ | -0.43 (0.05) | $<0.01^{***}$ | |
| Exposure to ads | 0.09 (0.03) | $<0.01^{***}$ | -- | | |
| Concern for flooding ($b$) | -- | | 0.24 (0.03) | $<0.01^{***}$ | |
| $R^2$ | 0.10 | | 0.16 | | |
| MSE | 2.50 | | 3.18 | | |
| F Statistic | $11.64^{***}$ (df = 11) | | $18.42^{***}$ (df = 11) | | |
| N | 1115 | | 1115 | | |

*Note:* $^{*}p<0.1$; $^{**}p<0.05$; $^{***}p<0.01$
**Bold** indicates that the confidence interval for the indirect estimate does not contain zero
Letters *a, b*, and *c* indicate coefficients displayed in Equations (1) and (2)

## 4.6. Discussion

In this paper, we examine if and how an individual's gender, race, and education level help predict their concern for flooding and willingness to participate in stormwater management. Consistent with previous literature (Lazri and Konisky, 2019; Macias, 2016a), we found that racially marginalized individuals, women, and those with a lower education level reported higher concern for local flooding compared to White, male, and higher educated respondents. Moreover, a heightened concern for flooding was an essential pathway through which socially and economically marginalized individuals developed their increased willingness to participate. The racial disparity in concern for flooding was greatest between White and Latino participants as Latinos were, on average, 1.2 units higher in their concern for flooding on a scale from 1-7 (Table 4.3). The disparity in concern for flooding and willingness to participate remained even after considering the effects of flood zone residence and stormwater knowledge and awareness.

We illustrate that racial disparities exist in flood zone residence, with non-White respondents being overrepresented in flood zones (Fig. 4.2). Likewise, a recent study showed that Black, Latino, and impoverished communities in Mecklenburg County are more likely to reside in flood zones than White and wealthier populations (Debbage, 2019). Our work takes this one step further in that racial disparities in concern for flooding and willingness to participate remain even after considering flood zone residence (Tables 4.2 and 4.3). Our results suggest that it is not only physical risk exposure but also differential vulnerability to stormwater hazards that drives risk perceptions of flood-related hazards (Hale et al., 2018). Like physical exposure, vulnerability to hazards is also driven by structural forms of racism, classism, and sexism that create and sustain debilitating patterns of unequal wealth distribution, access to loans, and access to transportation (Masozera et al., 2007). As Jones and Rainey (2006) pointed out, a perceived lack of social and financial support to address environmental hazards can trigger concerns about one's capacity to adapt. Importantly, the link we show between race, concern for flooding, and willingness to participate suggests that heightened concerns from racially marginalized individuals can lead to intentions to improve stormwater management.

Our results also highlight the complexity in willingness to pay measures, which tap into economic opportunity. We found that less-educated individuals had higher concerns for flooding and were, in turn, more willing to pay to improve flooding. However, education level still explained a portion of willingness to pay that was independent of concern for flooding: people with higher education were more willing to pay to improve flooding, which aligns with previous empirical studies (Chui and Ngai, 2016; Dietz et al., 2007; Macias, 2016a; Newburn and Alberini, 2016). Identifying such relationships highlights the complexity of willingness to pay measures, especially when associated with education level. High exposure to flooding hazards can drive low-income individuals' concern about stormwater and thus increase their willingness to participate in improving their conditions; however, a lack of expendable income or access to pertinent resources may deter willingness to participate in activities involving payment for services. Notably, this association was less evident with regard to willingness to volunteer (Table 4.2), implying that barriers that exist for willingness to pay may not substantially deter willingness to volunteer. These results highlight the need for multiple measures of environmentally conscious participation that capture the complexities of behaviors and barriers to performing them.

Lack of knowledge about stormwater has been cited as a key barrier to participation; however, we found an inconsistent effect of knowledge about stormwater on concern for flooding and willingness to participate. While technical knowledge about stormwater was not associated with concern for flooding, exposure to educational advertisements was positively associated with concern for flooding. Our results suggest that technical expertise of watershed processes does not influence concern about stormwater; rather, as Mobley (2016) pointed out, informal education—driven by educational campaigns and everyday experiences with and observations of flooding and water quality— likely drives concern about and participation in stormwater management. This finding has implications for understanding the importance of environmental experiences, rather than formal knowledge, on participation in water management. For example, pervasive segregation of U.S. cities, driven by legacies of discriminatory housing segregation policies, can dictate differential environmental experiences by socioeconomic status. These differences in experiences possibly facilitate different understandings of flooding and water quality impacts and thus different concerns about and participation in stormwater management.

Future research will be needed to further explore the implications of the current study. First, future work could explore how sociodemographic variables can be included in classic models of environmental behavior, such as the Theory of Planned Behavior (Ajzen, 1991). Given the growing evidence that race, gender, and class significantly influence concern and behavior, more researchers should aim to sample and gather data in representative ways along these lines. Second, our measurement of willingness to participate is limited by our ability to only consider willingness to pay and volunteer. We observe differences in how these activities are related to race, gender, education, and age, which calls for researchers to expand measurements of participation and willingness to participate to include other activities such as reporting infrastructure failures to local agencies and joining advocacy organizations. This research would also benefit by measuring actual behavior rather than the intent to participate. By measuring actual levels of participation, we can develop a more comprehensive understanding of barriers to action. Third, given our results, we should expand our conceptualization of "knowledge" in survey instruments. As shown in this study, technical expertise of watershed processes does not influence concern about flooding, but targeted campaigns like CMSWS "Turn Around, Don't Drown" and people's experiences with local flooding are more influential.

There are some notable shortcomings of the surveys used in our analysis. First, each year that the survey was conducted, administrators sampled until 400 surveys were complete. There was a lack of attention towards survey response rates, which signals low response rates and likely biased the survey participants towards those who are more opinionated about stormwater. Second, in the 2014 survey, respondents' gender was recorded without explicitly asking their gender identity. Survey administrators determined gender by the voice of the respondents. This practice not only introduced biases into the gender measurement but also stripped participants of agency to define their gender. This practice was not repeated in 2016 and 2017. Third, while our sample yielded enough cases to be a valid representation of racial groups, Black, Latino, and Asian American respondents were underrepresented compared to their representation in Mecklenburg county. Future surveys should address this bias by conducting stratified random sampling to ensure multiple population characteristics are represented in the sample. Fourth, we recommend that future surveys measure respondents' experiences with flooding because it can be an important predictor of concerns for flooding (*see* Hale et al., 2018). Urban flooding is spatially heterogeneous due to stormwater infrastructure and impervious surfaces and can deviate from riverine floodplains considerably. Given that flood zones are not well known, flooding experiences could be a more accurate measurement of flood exposure than residence within a 100-year floodplain. Despite the shortcomings mentioned above that often come with secondary data, we find that these surveys provide unique and timely information about willingness to participate in stormwater management based on an individual's race, gender, and education level.

### 4.7. Concluding Thoughts

We evaluated the relationship between social and economic marginalization and stormwater management. We found consistent and significant associations between race, gender, and education level and individuals' concern about and willingness to participate in stormwater management across three longitudinal surveys. More underserved groups were more willing to participate in stormwater remediation, and willingness to participate was associated with their heightened concern for flooding. These analyses have considerable implications for how we theorize the interplay between inequitable distributions of environmental conditions and actions to remediate those conditions. Even in highly technical spaces where public participation is unconventional, we saw that socially and racially marginalized individuals were driven by their concerns for flooding to be more involved in shaping their future relationship with stormwater. These patterns should alert policymakers to recognize this heightened concern and facilitate ways in which these communities can articulate their experiences and be involved in stormwater decision-making and planning.

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# CHAPTER 5. CONCLUSIONS

## 5.1. Limitations and Future Work

### 5.1.1. Shifts in Stream Ecological Function with Increasing Urbanization

This study lacked data on nutrients (nitrogen, phosphorus, and carbon). Lack of nutrient data limited my ability to understand the role that nutrients play in increasing GPP in more urbanized watersheds. Nutrients have been associated with increased productivity in urban watersheds (Alberts et al., 2017; Fuß et al., 2017). In this study, I cannot draw conclusions on the role of nutrients. This, in turn, limits the ability to inform specific management practices. If nutrients were the primary control on GPP, management should be focused on nutrient load reduction in the watershed. In contrast, if light was the primary control, management should focus on the riparian and channel scale to reduce shade in the creeks. I hope that future research will be able to inform the primary mechanism at play in urban watersheds.

Additionally, I was unable to control for watershed size in our site selection, and size is a major driver of metabolism. Future work would benefit from a large synthesis of data from urban watersheds that controls for watershed size.

However, even without this working knowledge, it is clear that urbanization shifts small creeks to highly productive systems, especially in the summer. We observed two predominant productivity regimes across the study creeks: productive and unproductive summers. Productivity was associated with watershed imperviousness. Highly productive summers, along with slow-moving and low baseflow, can increase the likelihood of eutrophication in these creeks. Management practices focused on reducing light and nutrient inputs to the stream will likely help improve conditions.

### 5.1.2. Making Conservation Normal: A Systematic Review of Social Norms in Promoting Water Quality Best Management Practices

This work implies that the current predominant focus on changing individual attributes to motivate conservation practices may be oversimplified. Individuals are socially connected to others and these connections, and importantly social expectations that form through these relationships, inform decision-making. One avenue of my future research will be to explicate the relationship between freshwater ecosystems and the development and adherence to social norms supporting conservation (Warner et al., 2020). Warner et al. (2020) found an important connection between water spaces and social norms supporting conservation but the mechanism that mobilizes social norms in these spaces was not clear. If better understood, this link between social norms and aquatic ecosystems can help build management tools that utilize local aquatic ecosystems, such as stormwater parks and river walks, to promote and mobilize social norms supporting conservation.

### 5.1.3. Stormwater on the Margins: Influence of Race, Gender, and Education on Willingness to Participate in Stormwater Management

In this chapter, we only used two metrics of participation and measured participation at the individual levels. Throughout my experiences with community engagement in stormwater, I've seen that participation can look differently in disenfranchised communities who must use alternative channels to communicate their grievances with local decision-makers. Payment for services may not be the best option, and volunteering often takes time that many working-class Americans do not have. Civic engagement in places that are already frequented by community members and have cultural meaning are cited in the literature as pathways to participation in water management in Black communities (Paolisso et al., 2012). A primary goal of my future work is to gain a better understanding of how people adapt to and mitigate flooding and water quality issues while also navigating institutional constraints maintained by structural racism. Procedural environmental justice will be a key conceptual foundation for this future work. Histories of racialized exclusion can lead to the development of alternative modes of participation.

We also saw that concern for flooding was associated with greater willingness to participate in stormwater management. Concern for flooding was higher in Black and Latino communities—communities that are at a higher risk for flooding Charlotte-Mecklenburg County (Debbage, 2019). Recent works have shown that in the face of racism and segregation, Black communities have created intimate relationships with water and local knowledge of freshwater ecosystems (Roane, 2018). These communities develop unique forms of environmental engagement that may not look like mainstream environmentalism. In the context of this dissertation and my future work, it is important to recognize and support the resilient adaptations already taking place in communities as they manage changing environmental conditions.

## 5.2. The Bigger Picture: My Perspectives on the Future of Stormwater Management

Chapter 2 demonstrated that human development is shifting stream metabolism towards highly productive summers with substantial instability following summer storm events. I identified specific land cover and stream habitat changes likely associated with increased productivity, including increased unmitigated impervious surfaces and low riparian and stream bank vegetation. I suggested that management focus on stream channel shading and broader watershed projects that disconnect impervious surfaces from the stream and reduce nutrient pollution. Impervious surfaces quickly convey nutrients from the watershed to the stream, which suggests that stormwater control measures— including ponds, wetlands, and bioretention cells— and nutrient load reduction play a role at the watershed scale. As overviewed in Chapters 3 and 4, management of stormwater involves an assemblage of individual decision-makers who are embedded in intricate socio-ecological relationships. I contend that future water management would benefit from incorporating social equity into management practices and sustaining social infrastructures that support watershed restoration.

Chapters 3 and 4 call for watershed conservation to move beyond the scale of the individual towards practices that recognize that individuals are 1) embedded within social networks that drive expectations for conservation and 2) influenced by socio-historical processes that inform their decision-making. On the former, I demonstrate that social norms are important drivers of conservation decision-making. I stress the importance of recognizing existing social norms surrounding water quality conservation and promoting conservation practices through these networked relationships. I recognize that conservation is not an isolated practice performed by one individual; rather, it is embedded within socially and biologically interconnected landscapes. Conservation practices occur within elaborate social networks where social norms driving the behavior of communities are established, contested, and reproduced. Conservation managers should develop practices that recognize these social networks and work through established networks to promote change.

Foundational to the incorporation of social equity into water management is the recognition that water is constituted by social and ecological processes (Bakker & Bridge, 2006; Linton & Budds, 2014). While waterways certainly are products of ecological processes, they are also "artifacts" of past and present social inequalities— they embody exclusionary practices. In U.S. cities, racist practices were codified into spatial realities through segregation, housing covenants, gerrymandering, and redlining whereby "people of different races are relegated to differential spaces" (Lipsitz, 2007). The legacy of these practices resides not only in the present-day fragmentation of people based on race and class (Delmelle, 2019) but also in economic divestment from Black, Indigenous, and other communities of color, privatization of essential water, health, housing, and transportation services, and, importantly, biophysical patterns of flora, fauna, and pollution of rivers (Schell et al., 2020). Such socio-ecological fragmentation can lead to differential experiences of and vulnerabilities to flooding and water pollution. These vulnerabilities create unique experiences that may facilitate willingness to participate in stormwater management, as demonstrated in Chapter 4. It is important to recognize the willingness of marginalized communities to be involved in water management. Beyond recognition of these communities as water stewards, water management should work towards **building practices** that value the unique knowledges created from these experiences as assets to conservation (Roane, 2018) as well as reducing institutional barriers that systematically exclude marginalized communities from participating in water management.

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# APPENDIX A. HABITAT ASSESSMENT SCORES 2013-2018

**Table A.1**. Summary Table of Average Habitat Assessment Score 2013-2018. Channel alteration, bank vegetation, bank stability, and riparian vegetation scores range from 0-20. Habitat diversity scores range from 0-100.

| Creek | Site Identifier | Channel Alteration (ChanAlt) | Habitat Diversity (Hab) | Bank Vegetation (BV) | Bank Stability (BS) | Riparian Vegetation (Rip) |
|---|---|---|---|---|---|---|
| Clear Creek | CC | 10 | 66 | 11 | 11 | 6 |
| McKee Creek | MKC | 8 | 48 | 11 | 12 | 11 |
| Goose Creek | GC | 6 | 58 | 12 | 13 | 10 |
| Long Creek | LC | 9 | 50 | 13 | 13 | 5 |
| Paw Creek | PC | 10 | 14 | 12 | 12 | 8 |
| Mallard Creek | MC | 10 | 18 | 10 | 9 | 13 |
| Back Creek | BAC | 2 | 82 | 15 | 17 | 17 |
| McMullen Creek | MMC | 13 | 54 | 13 | 17 | 11 |
| Briar Creek | BRC | 17 | 40 | 4 | 17 | 4 |
| Sugar Creek | SC | 8 | 40 | 13 | 14 | 14 |
| Irwin Creek | IC | 15 | 60 | 12 | 17 | 5 |
| Little Sugar Creek | LSC | 19 | 65 | 8 | 17 | 1 |

# APPENDIX B. DIAGNOSTIC SUMMARY

**Table B.1.** Total days modeled in *streamMetabolizer* and number of days retained after removing days with poor model fit.

| Creek | Site Identifier | Total modeled days | Total days retained | Percent Retained (%) |
|---|---|---|---|---|
| Clear Creek[a] | CC | 2227 | 1169 | 52 |
| McKee Creek | MKC | 2173 | 517 | 24 |
| Goose Creek | GC | 1738 | 1155 | 66 |
| Long Creek[b] | LC | 1628 | 785 | 48 |
| Paw Creek[b] | PC | 1905 | 829 | 44 |
| Mallard Creek | MC | 2074 | 1236 | 60 |
| Back Creek | BAC | 1671 | 1004 | 60 |
| McMullen Creek | MMC | 1847 | 443 | 24 |
| Briar Creek | BRC | 2119 | 1346 | 64 |
| Sugar Creek | SC | 2109 | 1037 | 49 |
| Irwin Creek | IC | 2166 | 1311 | 61 |
| Little Sugar Creek | LSC | 2125 | 1341 | 53 |

[a] GPP and ER estimates in 2015 and 2016 were removed due to signs of equifinality
[b] GPP and ER estimates in 2017 and 2018 were removed due to signs of equifinality

# APPENDIX C. ARRHENIUS MODEL RESULTS

**Table C.1.** Regression coefficients and model statistics for the Boltzmann-Arrhenius Equation. The response variable is mean daily GPP.

| | **Standardized Temperature** | | | **Q** | | | **Q×T** | | | **Model Statistics** | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Site** | ***β*** | **SE** | ***p*** | ***β*** | **SE** | ***p*** | ***β*** | **SE** | ***p*** | **N** | **Adj. R-squared** | ***F*** | ***p*** | **AIC** |
| CC | 0.22 | 0.07 | <0.001 | -- | -- | -- | -- | -- | -- | 453 | 0.08 | 38.68 | <0.001 | 938.43 |
| MKC | -0.08 | 0.07 | 0.27 | -- | -- | -- | -- | -- | -- | 517 | 0.01 | 6.89 | 0.01 | 1036.90 |
| GC | 0.04 | 0.07 | 0.62 | -- | -- | -- | -- | -- | -- | 738 | <0.001 | 1.39 | 0.24 | 1728.82 |
| LC | 0.62 | 0.07 | <0.001 | -0.55 | 0.11 | <0.001 | -- | -- | -- | 436 | 0.54 | 260.95 | <0.001 | 800.16 |
| PC | 0.09 | 0.09 | 0.30 | -- | -- | -- | -- | -- | -- | 418 | 0.01 | 5.72 | 0.02 | 856.50 |
| MC | 0.47 | 0.06 | <0.001 | -- | -- | -- | -0.37 | 0.10 | <0.001 | 552 | 0.31 | 126.50 | <0.001 | 1010.17 |
| BAC | 0.08 | 0.08 | 0.29 | -- | -- | -- | -- | -- | -- | 594 | 0.01 | 5.91 | 0.02 | 1478.35 |
| MMC | -0.08 | 0.08 | 0.33 | -- | -- | -- | -- | -- | -- | 439 | 0.01 | 6.12 | 0.01 | 847.31 |
| BRC | 0.79 | 0.05 | <0.001 | -0.47 | 0.11 | <0.001 | -0.16 | 0.07 | 0.04 | 730 | 0.64 | 434.00 | <0.001 | 1317.30 |
| SC | 0.32 | 0.07 | <0.001 | -0.35 | 0.11 | <0.001 | -0.22 | 0.08 | 0.01 | 613 | 0.16 | 38.84 | <0.001 | 1217.27 |
| IC | 0.57 | 0.04 | <0.001 | -0.38 | 0.09 | <0.001 | -- | -- | -- | 796 | 0.46 | 338.78 | <0.001 | 1466.41 |
| LSC | 0.68 | 0.05 | <0.001 | -- | -- | -- | -0.36 | 0.08 | 0.00 | 807 | 0.48 | 368.09 | <0.001 | 1517.08 |

**Table C.2.** Regression coefficients and model statistics for the Boltzmann-Arrhenius Equation. The response variable is mean daily ER.

| | **Standardized Temperature** | | | **Q** | | | **Q×T** | | | **Model Statistics** | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Site** | $\beta$ | **SE** | $p$ | $\beta$ | **SE** | $p$ | $\beta$ | **SE** | $p$ | **N** | **Adj. R-squared** | $F$ | $p$ | **AIC** |
| CC | 0.30 | 0.04 | <0.001 | -- | -- | -- | -- | -- | -- | 453 | 0.33 | 221.82 | <0.001 | 431.93 |
| MKC | 0.35 | 0.04 | <0.001 | 0.26 | 0.07 | <0.001 | -- | -- | -- | 517 | 0.44 | 207.72 | <0.001 | 412.72 |
| GC | 0.32 | 0.06 | <0.001 | -- | -- | -- | -- | -- | -- | 738 | 0.31 | 329.56 | <0.001 | 918.46 |
| LC | 0.35 | 0.06 | <0.001 | -- | -- | -- | -- | -- | -- | 436 | 0.40 | 287.32 | <0.001 | 487.74 |
| PC | 0.36 | 0.04 | <0.001 | -- | -- | -- | -- | -- | -- | 418 | 0.46 | 358.87 | <0.001 | 311.48 |
| MC | 0.32 | 0.04 | <0.001 | -- | -- | -- | -- | -- | -- | 552 | 0.42 | 402.63 | <0.001 | 405.69 |
| BAC | 0.39 | 0.05 | <0.001 | -- | -- | -- | -0.19 | 0.07 | 0.01 | 594 | 0.42 | 216.19 | <0.001 | 703.14 |
| MMC | 0.53 | 0.06 | <0.001 | 0.12 | 0.09 | 0.20 | -- | -- | -- | 439 | 0.49 | 207.52 | <0.001 | 681.58 |
| BRC | 0.34 | 0.03 | <0.001 | -- | -- | -- | -- | -- | -- | 730 | 0.51 | 766.62 | <0.001 | 524.25 |
| SC | 0.21 | 0.07 | <0.003 | -- | -- | -- | -- | -- | -- | 611 | 0.13 | 92.71 | <0.001 | 764.74 |
| IC | 0.41 | 0.06 | <0.001 | -- | -- | -- | -- | -- | -- | 796 | 0.40 | 534.30 | <0.001 | 1049.78 |
| LSC | 0.32 | 0.03 | <0.001 | | | | -0.09 | 0.04 | 0.02 | 807 | 0.41 | 285.10 | <0.001 | 536.76 |

# APPENDIX D. SUMMARY OF SOCIAL NORMS MEASURES

**Table D.1.** Summary of social norm measures and effect of social norms on adoption of water quality BMPs in the empirical quantitative studies. **We specifically isolated empirical, quantitative studies that used regression methods to measure the influence of social norms on BMP adoption.** A significant positive effect of social norms on likelihood of BMP adoption is indicated with "+", a significant negative effect is indicated with "-", and no effect is indicated with "N". Inconclusive studies are those with multiple models with differing effects of social norms, and these are indicated with "INC".

| **Publication** | **Land Holder Category** | **Construct Measured (as explicitly detailed in study)** | **Measure(s) of Social Norms** | **Measurement Scale** | **Outcome Measure** | **Effect on outcome** |
|---|---|---|---|---|---|---|
| Shaw et al. (2011)[a] | Residential Homeowners/ Renters | Normative belief | If I build a rain garden in my yard, my neighbors would:<br><br>If I build a rain garden in my yard, my family would:<br><br>If I build a rain garden in my yard, my friends would: | Strongly disapprove – strongly approve | Willingness to adopt BMP | + |
| | | Subjective norm evaluation | What my neighbors recommend is:<br><br>What my family recommends is:<br><br>What my friends recommend is: | Not important to me – Very important to me | Willingness to adopt BMP | |
| Welch et al. (2001) | Farm owner/operator | Regulatory Pressure | If this program fails strict regulations may follow | Strongly disagree – strongly agree | Probability of early adoption | + |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | The Whole Farm Plan is my regulatory insurance policy | | Late adopter (1), Early adopter (2) | |
| | | Community Pressure | The watershed community will recognize that I am doing everything I can to protect Skaneateles Lake<br><br>It is important to me that my peers in the agricultural community recognize that I am doing the best I can | Strongly disagree – strongly agree | | - |
| Eisenhauer et al. (2016) | Residential Homeowners/ Renters | Normative preferences for lawn appearance | having no weeds on my lawn<br><br>having my lawn as dark green as possible<br><br>having the grass be as thick as possible<br><br>having my lawn be clover free<br><br>having a pest-free lawn;<br><br>having a ‘‘golf course’’ quality lawn | Not important – very important | Willingness to adopt BMP | - |
| Nohner et al. (2018) | Residential Homeowners/ Renters | Social pressure | It is important to me that my neighbors maintain a manicured lawn and shoreline. | Strongly disagree – strongly agree | Willingness to adopt BMP | - |
| Warner et al. (2021) | Both | Subjective norms | The people who are important to me: | Very unlikely – very likely | Willingness to adopt BMP | + |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | would expect that I use good fertilization practices<br><br>would approve if I applied fertilizers appropriately<br><br>expect that I will read the fertilizer label before applying fertilizer<br><br>expect that I carefully apply fertilizer according to plants' needs | | | |
| Kuhfuss et al. (2016)[b] | Farm owner/operator | Positively framed information | In a previous survey, 80% of the respondents stated that they would maintain the new practices they had adopted, even without renewal of their contract. After your period of agreement ends, do you plan to maintain these changes without renewal of the contract? | NA | Likelihood to maintain BMP adoption | + |
| | | Negatively framed information | In a previous survey, 20% of the respondents stated they would not maintain the new practices they had adopted without renewal of their contract. After your period of agreement ends, do you plan to maintain these changes without renewal of the contract? | NA | | + |

| | | | | | | |
|---|---|---|---|---|---|---|
| Doran et al. (2020) | Farm owner/operator | Perceived social norm | The next question is designed to help us understand who (friends and/or family, neighbors, or other farmers) may most strongly influence your decision to adopt conservation practices. Under each conservation practice, please tell us how strongly you agree or disagree that friends and family, neighbors,<br>or other farmers think you should adopt that practice, if applicable. | Strongly disagree – strongly agree | Willingness to adopt BMP | INC[c] |
| Daxini et al. (2018) | Farm owner/operator | Subjective norm | Think that I should [apply fertilizer on the basis of soil test results] do so<br><br>Encourage me to [apply fertilizer on the basis of soil test results] do so<br><br>Would approve if I [apply fertilizer on the basis of soil test results] do so<br><br>Most farmers I am aware of base fertiliser application on recommendations from soil test results | Strongly disagree – strongly agree | Willingness to adopt BMP | INC[d] |

| | | | | | | |
|---|---|---|---|---|---|---|
| Daxini et al. (2019) | Farm owner/operator | Subjective norm | When it comes to following a NMP [nutrient management plan], most people whose opinion I value regarding farming: would approve if I do so?<br><br>When it comes to following a NMP, most people whose opinion I value regarding farming: encourage me to do so?<br><br>When it comes to following a NMP, most people whose opinion I value regarding farming: think that I should do so? | Strongly disagree – strongly agree | Willingness to adopt BMP | + |
| Fielding et al. (2005) | Farm owner/operator | Normative beliefs | "…respondents were asked to assess how much they thought each of the referents would think they should manage the riparian zones on their property" (p. 15).<br><br>"The salient referents were (1) Fitzroy Basin Association, (2) Landcare, environmental, and conservation groups, (3) Department of Primary Industries, (4) other government agencies (e.g. | Extremely unlikely – extremely likely | Willingness to adopt BMP | N |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | Department of Natural Resources and Mines), (5) urban Australians" (p. 15) | | | |
| | | Motivation to comply with referents | "…asking respondents how willing they were to do what these groups wanted them to do on their property" (p. 15). | Not at all – very much | | + |
| Wang et al. (2020)[e] | Residential Homeowners/ Renters | Subjective norm | "The people who are important to me think that citizens should pay for SCP" (p. 6)<br><br>"Local government thinks that citizens should pay for SCP" (p. 6)<br><br>"Local media thinks that citizens should pay for SCP" (p. 6)<br><br>"The people whose opinions I value would like me to pay for SCP" (p. 6)<br><br>"The people who are important to me expect that I will pay for SCP" (p. 6)<br><br>"Local government expects that I will pay for SCP" (p. 6) | Strongly disagree – strongly agree | Willingness to pay for BMP initiative | N |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | "Local media expects that I will pay for SCP" (p. 6) | | | |
| Peterson et al. (2012) | Residential Homeowners/ Renters | Neighbors Support | Measured perception of neighbors support for different residential land cover designs including 0% native plant gardens, 50% native plant gardens, 75% native plant gardens, and 100% native plant gardens | Strongly oppose – strongly support | Preference for native plant garden | + |
| Slagle et al. (2015) | Residential Homeowners/ Renters | Informational subjective norms | People who are important to me would expect me to stay on top of information regarding local streams (p. 826)<br><br>People who are important to me would expect me to be knowledgeable about actions to improve local stream conditions (p. 826)<br><br>The people I spend most of my time with are likely to seek information related to local stream conditions (p. 826) | NA | Information seeking about local stream conditions | + |
| Eanes et al. (2020)[f] | Residential Homeowners/ Renters | Injunctive norm | Neighbors on my street generally think I should use lawncare BMPs (p. 741) | Strongly disagree – strongly agree | Adoption of BMP | INC |

| | | | | | | |
|---|---|---|---|---|---|---|
| | | | My close friends generally think I should use lawncare BMPs (p. 741)<br><br>People in my neighborhood generally think I should use lawncare BMPs (p. 741) | | | |
| | | Descriptive norm | Neighbors on my street generally use lawncare BMPs (p. 741)<br><br>My close friends generally use lawncare BMPs (p. 741)<br><br>People in my neighborhood generally use lawncare BMPs (p. 741) | Strongly disagree – strongly agree | | INC |
| | | Social norm influence | “How influential are the following entities on your current lawncare practices?” (p. 741)<br><br>My friends<br>Neighbors on my street<br>People in my neighborhood (p. 741) | not influential - extremely influential | | NA |
| Peth et al. (2018)[b] | Farm owner/operator | Social nudge | Treatment group provided with consequences of non-compliant behavior with minimum-distance-to-water rule and information about neighbors compliance with rule | NA | Adoption of BMP | + |

*a – composite measure created by multiplying normative beliefs by evaluation
*b – experimental design study
*c – social norm influence always positive but effect significance differs by BMP
*d – social norms are significant in the national sample and sample of mandated farmers. Social norms not significant in voluntary sample of farmers
*e – Wang et al. (2020) used a higher threshold for significance $p<0.01$
*f – subjective and descriptive norms were influential on fertilizer pesticide avoidance but not mulching and chemical bans
*Significance measured at $p < 0.05$

# APPENDIX E.  SURVEY VARIABLES AND MEASURES

| Variables | Questions |
|---|---|
| Willingness to Pay[a] Summed Scale 1 ($\alpha_{2014} = 0.69$) | I would be willing to pay more in storm water fees if it would be used to clean up polluted creeks and streams in Charlotte-Mecklenburg. |
| | I would be willing to pay more in storm water fees if it would reduce flooding in Charlotte-Mecklenburg. |
| Willingness to Pay[a] Summed Scale 2 ($\alpha_{2016} = 0.83;\ \alpha_{2017} = 0.85$) | I would be willing to pay more in stormwater fees if it would be used to clean up polluted creeks, streams, and lakes in Charlotte and Mecklenburg County |
| | I would be willing to pay more in stormwater fees if it would be used to reduce flooding in Charlotte and Mecklenburg County. |
| Willingness to Volunteer[a] | I would volunteer to work with a group of volunteers, twice a year, to help clean up polluted creeks and streams in or near my neighborhood. |
| Concern for Flooding[a] Summed Scale 1 ($\alpha_{2014} = 0.77$) | During times of heavy rain, I am concerned that the creeks in Charlotte-Mecklenburg will flood roads. |
| | During times of heavy rain, I am concerned that the creeks in Charlotte-Mecklenburg will flood buildings |
| Concern for Flooding[a] Summed Scale 2 ($\alpha_{2016} = 0.77;\ \alpha_{2017} = 0.69$) | During times of heavy rain I am concerned about stormwater flooding on roads. |
| | During times of heavy rain I am concerned about flooding in homes and buildings. |
| Exposure to stormwater advertisements[b] Summed Scale 1 ($\alpha_{2014} = 0.61$) | Have you heard or seen any information about storm water in the past year? |
| | Advertisements have been run on multiple mediums letting people know it's so easy to volunteer.  Do you recall seeing or hearing any of these ads in the past year? |
| | Advertisements have been run on multiple mediums encouraging citizens to Turn around Don't drown with the tag line your car is not a boat. Do you recall seeing or hearing any of these ads in the past year? |
| | Local emergency responders and flood experts have started an educational effort called 'Build an Ark' about flood awareness, responsibility and knowledge. Have you heard or seen anything about the Build an Ark campaign? |

| | |
|---|---|
| | Advertisements have been run on multiple mediums encouraging citizens to be a water watcher and report storm water or creek pollution. Do you recall seeing or hearing any of these ads in the past year? |
| Exposure to stormwater advertisements[b]<br>Summed Scale 2 ($\alpha_{2016} = 0.83;\ \alpha_{2017} = 0.80$) | Have you heard or seen any information from Charlotte-Mecklenburg Storm Water Services about flood zone maps and flood insurance in the past 12 months? |
| | Have you heard or seen any information from Charlotte-Mecklenburg Storm Water Services about volunteer opportunities to reduce pollution in and around our creeks, streams, and lakes in the past 12 months? |
| | Have you heard or seen any information from Charlotte-Mecklenburg Storm Water Services about reporting pollution in stormwater, creeks, streams, and lakes, such as the "Water Watcher" program, in the past 12 months? |
| | Have you heard or seen any information from Charlotte-Mecklenburg Storm Water Services about stormwater pollution and how it flows to creeks, streams, and lakes in the past 12 months? |
| | Have you heard or seen any information from any other entity about stormwater pollution, flooding, volunteer opportunities, etc.? |
| Knowledge of Stormwater[a]<br>Summed Scale 1 ($\alpha_{2014} = 0.12$) | Water that runs into storm drains is treated and cleaned before being released into creeks, lakes and ponds. |
| | People in Charlotte-Mecklenburg can experience severe flooding even if their property is not in a flood zone. |
| | Water that runs into storm drains flows directly to local creeks and lakes. |
| Knowledge of Stormwater[c]<br>Summed Scale 2 ($\alpha_{2016} = 0.64;\ \alpha_{2017} = 0.61$) | Stormwater that goes into storm drains is cleaned by a treatment facility before it goes into creeks, streams, and lakes. |
| | Water that flows into storm drains goes directly to local creeks, streams, and lakes. |
| Residence in a Flood Zone[b] | Is the property where you currently live in a flood zone? |
| Home Ownership[d] | Do you own or rent your home? |

*Notes on coding:*

[a] 1 = Don't know, 2 = Disagree strongly, 3 = Disagree somewhat, 4 = Agree somewhat, 5 = Agree strongly

[b] 1 = Yes, 2 = No, 3 = Don't know

[c] 1 = False, 2 = True

[d] 1 = Own, 2 = Rent, 3 = Other, 4 = Refused to answer

# APPENDIX F. ASSUMPTIONS OF MULTIPLE REGRESSION

We checked multiple regression assumptions for each model displayed in the main text (Tables 2-4). We checked the assumptions of the direct effect model with participation (willingness to volunteer, willingness to pay, and overall willingness) as the outcome variable. We checked for multicollinearity using the generalized variance inflation factor (GVIF), normality using a histogram of the regression residuals, and homoscedasticity by plotted residual vs. fitted values of the regression model.

**Table F.1.** The generalized variance inflation factor for predictor variables in the regression displayed in Table 4.2 of the main text. Willingness to volunteer is the outcome variable.

| Variable | GVIF | $GVIF^{(1/2Df)}$ |
|---|---|---|
| Race | 1.46 | 1.04 |
| Education | 1.24 | 1.12 |
| Gender | 1.05 | 1.03 |
| Exposure to Ads | 1.19 | 1.09 |
| Concern for Flooding | 1.17 | 1.08 |
| Home Ownership | 1.26 | 1.12 |
| Age | 1.29 | 1.13 |
| Exposure to flood zone | 1.07 | 1.04 |

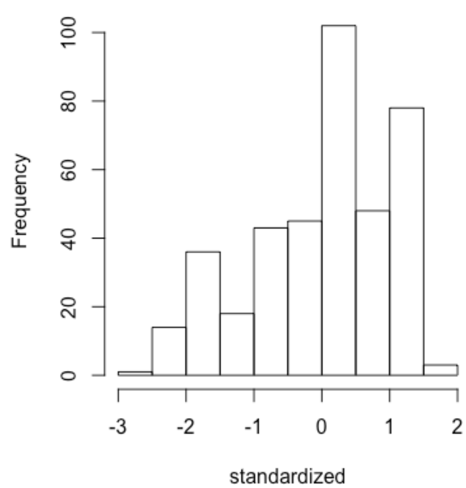


**Figure F.1.** Histogram of the standardized residuals of the regression displayed in Table 4.2 of the main text. Willingness to volunteer is the outcome variable.

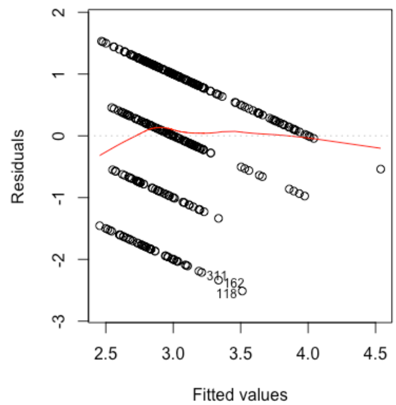


**Figure F.2.** Residuals vs. fitted values of the regression displayed in Table 4.2 of the main text. Willingness to volunteer is the outcome variable.

**Table F.2.** The generalized variance inflation factor for predictor variables in the regression displayed in Table 4.3 of the main text. Willingness to pay is the outcome variable.

| Variable | GVIF | $GVIF^{(1/2Df)}$ |
|---|---|---|
| Race | 1.32 | 1.03 |
| Education | 1.13 | 1.06 |
| Gender | 1.03 | 1.02 |
| Exposure to Ads | 1.09 | 1.04 |
| Concern for Flooding | 1.12 | 1.06 |
| Time | 1.17 | 1.04 |
| Age | 1.17 | 1.08 |
| Exposure to flood zone | 1.03 | 1.01 |

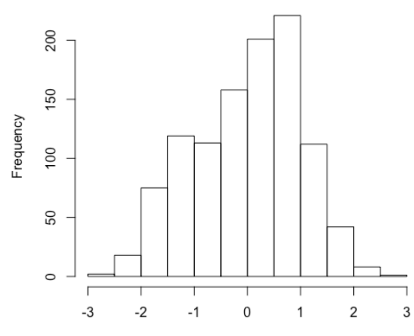


**Figure F.3.** Histogram of the standardized residuals of the regression displayed in Table 4.3 of the main text. Willingness to pay is the outcome variable.

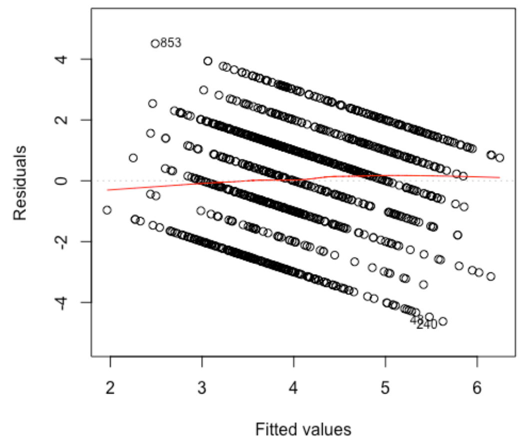


**Figure F.4.** Residuals vs. fitted values of the regression displayed in Table 4.3 of the main text. Willingness to pay is the outcome variable.

**Table F.3.** The generalized variance inflation factor for predictor variables in the regression displayed in Appendix H. Willingness to pay is the outcome variable.

| Variable | GVIF | $GVIF^{(1/2Df)}$ |
|---|---|---|
| Race | 1.32 | 1.03 |
| Education | 1.14 | 1.07 |
| Gender | 1.07 | 1.03 |
| Exposure to Ads | 1.05 | 1.03 |
| Concern for Flooding | 1.08 | 1.04 |
| Time | 1.03 | 1.02 |
| Exposure to flood zone | 1.02 | 1.01 |
| Age | 1.15 | 1.07 |
| Knowledge | 1.07 | 1.03 |

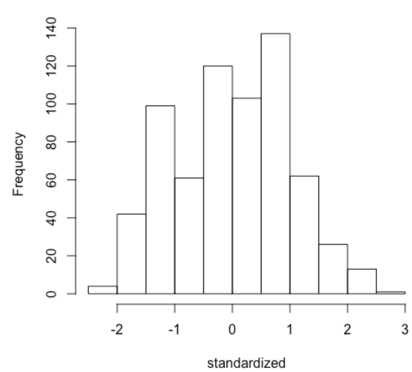


**Figure F.5.** Histogram of the standardized residuals of the regression displayed in Appendix H. Willingness to pay is the outcome variable.

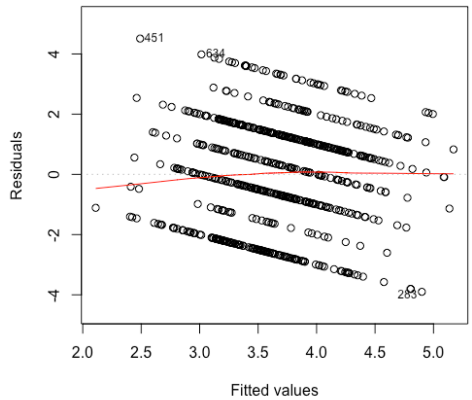


**Figure F.6.** Residuals vs. fitted values of the regression displayed in Appendix H. Willingness to pay is the outcome variable.

# APPENDIX G. PROPORTIONAL ODDS MODEL

**Table G.1.** We performed several proportional odds models to test whether our assumption of continuous response variables significantly impacted the model results. This model is comparable to Table 4.2 in the main body of the text. The outcome variable is willingness to volunteer. The significant variables and the direction of the effect is the same in both the proportional odds model shown below and the multiple regression shown in Table 4.2 of the main text.

| Independent Variable | Odds Ratio |
|---|---|
| Race | |
| Latino | 10.87*** |
| Black/African American | 1.14 |
| Asian or Pacific Islander | 0.96 |
| Multi-racial | 0.85 |
| Other | 1.58 |
| Gender (Female = 0) | 1.19 |
| Education | 1.14 |
| Age | 0.84** |
| Flood zone | 3.36*** |
| Home Ownership (Rent = 0) | 1.09 |
| Exposure to ads | 0.99 |
| Concern for flooding | 1.09 (p = 0.12) |
| Observations | 396 |
| Residual Deviance | 953.5 |
| AIC | 983.5 |
| LR Chi$^2$ | 48.7 |
| Pr(>Chi2) | <0.01 |
| *Note:* | $^{*}$p<0.1; $^{**}$p<0.05; $^{***}$p<0.01 |

**Table G.2.** We performed several proportional odds models to test whether our assumption of continuous response variables significantly impacted the model results. This model is comparable to Table 4.3 in the main body of the text. The outcome variable is willingness to pay. The significant variables and the direction of the effect is the same in both the proportional odds model shown below and the multiple regression shown in Table 4.3 of the main text.

| Independent Variable | Odds Ratio |
|---|---|
| Race | |
| Latino | 1.15 |
| Black /African American | 1.20 |
| Asian or Pacific Islander | 0.94 |
| Multi-racial | 0.75 |
| Other | 0.58** |
| Gender (Female = 0) | 0.93 |
| Education | 1.09* |
| Age | 0.81*** |
| Flood zone | 1.03 |
| Time | 0.28*** |
| Exposure to ads | 0.99 |
| Concern for flooding | 1.32*** |
| Observations | 1115 |
| Residual Deviance | 3906.3 |
| AIC | 3944.3 |
| LR $Chi^2$ | 213.3 |
| Pr(>Chi2) | <0.01 |
| *Note:* | $^{*}$p<0.1; $^{**}$p<0.05; $^{***}$p<0.01 |

# APPENDIX H. OLS PATH ANALYSIS WITH KNOWLEDGE VARIABLE

**Table H.1**. OLS regression coefficients. This analysis utilizes the 2016 and 2017 datasets. Following equations (1) and (2) in the main body of the text, concern is the mediating variable ($M$) and willingness to pay ($WTP$) is the outcome variable ($Y$). Standard errors of the direct effect are presented in parentheses. Indirect effect coefficients ($a_i b$) are presented with the bootstrapped 95% confidence interval in brackets.

| Independent Variable (IV) | Direct Effects | | | | Indirect Effects |
|---|---|---|---|---|---|
| | IV → Concern | | IV → WTP | | IV → Concern → WTP |
| | $a$ | p | $c$ | p | $a_i b$ |
| Race (White = 0) | | | | | |
| Latino | 1.17 (0.25) | <0.01*** | -0.35 (0.33) | 0.28 | **0.22 [0.09, 0.37]** |
| Black/African American | 0.24 (0.16) | 0.13 | 0.13 (0.20) | 0.52 | 0.04 [-0.02, 0.12] |
| Asian or Pacific Islander | 0.18 (0.30) | 0.62 | 0.02 (0.38) | 0.95 | -0.03 [-0.10, 0.17] |
| Multi-racial | 0.31 (0.33) | 0.34 | -0.39 (0.42) | 0.35 | 0.06 [-0.10, 0.22] |
| Other | -0.05 (0.23) | 0.84 | -0.52 (0.29) | 0.07 | -0.01 [-0.10, 0.08] |
| Gender (Female = 0) | -0.29 (0.11) | <0.01*** | -0.03 (0.13) | 0.81 | **-0.05 [-0.11, -0.01]** |
| Education | -0.16 (0.05) | <0.01*** | 0.01 (0.07) | 0.87 | **-0.03 [-0.06, -0.01]** |
| Age | 0.08 (0.04) | 0.05** | -0.28 (0.05) | <0.01*** | |
| Flood zone | 0.35 (0.25) | 0.16 | 0.12 (0.32) | 0.70 | |
| Knowledge | 0.08 (0.07) | 0.28 | -- | | |
| Exposure to ads | 0.03 (0.03) | 0.45 | -- | | |
| Concern for flooding | -- | | 0.19 (0.05) | <0.01*** | |
| $R^2$ | 0.07 | | 0.08 | | |
| MSE | 1.95 | | 3.15 | | |
| F Statistic | $5.05^{***}$ (df = 11) | | $5.74^{***}$ (df = 10) | | |
| N | 720 | | 720 | | |

*Note:* $^{*}$p<0.1; $^{**}$p<0.05; $^{***}$p<0.01
**Bold** indicates that the confidence interval for the indirect estimate does not contain zero
Letters *a, b*, and *c* indicate coefficients displayed in Equations (1) and (2)