Relations between Landscape Structure and a Watershed's Capacity to Regulate River Flooding

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Abstract

Climate and human activities impact the timing and quantity of streamflow and floods in different ways, with important implications for people and aquatic environments. Impacts of landscape changes on streamflow and floods are known, but few studies have explored the magnitude, duration and count of floods the landscape can influence. Understanding how floods are influenced by landscape structure provides insight into how, why and where floods have changed over time, and facilitates mapping the capacity of watersheds to regulate floods. In this study, I (1) compared nine flood-return periods of 31 watersheds across North Carolina and Virginia using long-term hydrologic records, (2) examined temporal trends in precipitation, stream flashiness, and the count, magnitude and duration of small and large floods for the same watersheds, and (3) developed a methodology to map the biophysical and technological capacity of eight urban watersheds to regulate floods. I found (1) floods with return periods ≤ 10 years can be managed by manipulating landscape structure, (2) precipitation and floods have decreased in the study watersheds while stream flashiness has increased between 1991 and 2013, (3) mapping both the biophysical and technological features of the landscape improved previous efforts of representing an urban landscape's capacity to regulate floods. My results can inform researchers and managers on the effect of anthropogenic change and management responses on floods, the efficacy of current strategies and policies to manage water resources, and the spatial distribution of a watershed's capacity to regulate flooding at a high spatial resolution.

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Attribution

Chapters 1 and 2: Chapter 1 will be submitted to the Journal of Environmental Management and Chapter 2 will be submitted to the Journal of the American Water Resources Association. The co-authors of both these manuscripts, in order, are Dr. Emmanuel A. Frimpong, Andrew B. Hoegh and Dr. Paul L. Angermeier. Dr. Frimpong is an Associate Professor in the Department of Fish and Wildlife Conservation at Virginia Tech. He developed and provided the code for the statistical analyses conducted in this manuscript. Andrew Hoegh is a PhD student in the Department of Statistics at Virginia Tech and a Lead Collaborator in Virginia Tech's LISA (Laboratory for Interdisciplinary Statistical Analysis) group. Andrew ran the temporal and spatial autocorrelation models for climatic and flood metrics, providing relevant information (e.g., tables, figures, writing) for this section. Dr. Angermeier is a Professor in the Department of Fish and Wildlife Conservation at Virginia Tech and an Assistant Leader for the Virginia Cooperative Fish and Wildlife Research Unit of the U.S. Geological Survey. Dr. Angermeier contributed financial support, provided input in study design and structure, and edited the manuscripts for publication.

Chapter 3: Chapter 3 will be submitted to the Ecosystem Services journal or to the Ecological Indicators journal. The co-authors in this manuscript are Dr. Amy M. Villamagna, Dr. Emmanuel A. Frimpong, and Dr. Paul L. Angermeier. Dr. Villamagna is an Assistant Professor in the Department of Environmental Sciences and Policy at Plymouth State University. Dr. Villamagna guided the development of the ecosystem service framework and edited the manuscript for publication. Dr. Frimpong developed and provided the code for the statistical analyses conducted in this manuscript. Dr. Angermeier contributed financial support, provided input in study design and structure, and edited the manuscript for publication.

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General Introduction

Floods occur naturally and aquatic ecosystems have adapted to these events (Black 2012); however many people dislike floods because of the casualties and damage they cause. Despite improvements in flood forecasting and global annual water-related infrastructure investments exceeding US$500 billion (Milly et al. 2002), floods continue to exact tolls on society (Pielke et al. 2002, Sutherland et al. 2013). As flood occurrences are spatially heterogeneous and inevitable, understanding their drivers and developing strategies to diminish damage is warranted. Inland floods are primarily driven by precipitation patterns (Lecce and Kotecki 2008, Tran et al. 2010), but landscape changes and human intervention can exacerbate or reduce flooding and damage (Eng et al. 2013b). Understanding the effects of anthropogenic changes and management responses on floods is imperative to assess water management strategies.

Germane to flood management is the role landscapes play in regulating floods. Natural and anthropogenic landscape features can substantially affect inland floods, but their effectiveness decreases as flood size increases (Tollan 2002, Bloschl et al. 2007). Depending on the flood, landscape processes can be more or less important in dictating the degree of flooding (Kundzewicz 1999). Smaller floods are more responsive to landscape structure, with lower and longer floods in forested watersheds but higher and shorter floods in urban watersheds (Magilligan and Stamp 1997, Findlay and Taylor 2006, Hawley and Bledsoe 2011). Similarly, artificial impoundments affect flooding only up to the point where runoff equals their storage capacity (Sordo-Ward et al. 2012). A clearer understanding of the flood-size threshold, above which watersheds with different landscape characteristics respond the same, is necessary to help design effective flood reduction strategies and to weigh the risks of urban development in flood-prone areas.

Changes in precipitation typically drive changes in streamflow (Patterson et al. 2012), including flood regimes. Unlike national-scale trends, in the southeastern US, precipitation and streamflow have decreased since the 1970s (Andreadis and Lettenmaier 2006, Patterson et al. 2012). Despite decreasing precipitation, North Carolina and Virginia rank in the top 12 states for flood damage from 1983 to 1999 (Pielke et al. 2002), and are among the top 25 states with the highest number of major disaster and emergency declarations from 1952 to 2014 (FEMA 2014). This pattern suggests there are non-climatic causes of floods, such as land cover change, construction of flow-regulating features, irrigation and groundwater pumping, sewer system design, and changes in soil management (Ye et al. 2003, Arrigoni et al. 2010, Wang and Hejazi 2011, Deasy et al. 2014). Understanding which human activities and interventions contribute to flooding (or do not) can guide water resource management.

In the context of urban or suburban development, the ability of landscapes to regulate floods is a globally important service, whether it is provided by natural ecosystems or technologic features. Mapping flood regulation can provide insight into the spatial distribution of service capacity (i.e., the level of potential service), thereby helping managers identify areas where flood regulation capacity can be enhanced (Villamagna et al. 2013). Most of the previous studies mapping flood regulating services only included biophysical features (soils, vegetation, land use/cover) (Posthumus et al. 2010, Nedkov and Burkhard 2012, Schulp et al. 2012). However, technological features are ubiquitous across the landscape, particularly in urban and semi-urban environments (Smith et al. 2002b, Downing et al. 2006, Ignatius and Jones 2014). Furthermore, previous flood-regulation mapping efforts have mapped the landscape processes that regulate floods, but have not used hydrologic records to assess the importance among these processes. Weighting all features the same can provide misleading spatial representations of the service. Developing a methodology that includes both technological and biophysical features, and uses long-term hydrologic records to assess the importance of those features would significantly improve analysis of flood regulation.

For this study, I selected small, non-coastal watersheds with long-term gage records and representing a wide range of topographies and human settlement patterns across Virginia (VA) and North Carolina (NC). VA and NC present an opportunity to conduct this research, as they have an extensive flooding history while sharing a similar climate (Patterson et al. 2012). North Carolina had 14 major flooding disasters declared from 1962 to 2013, while Virginia had 27 major disaster declarations from 1957 to 2009 (FEMA 2014). In an effort to further understand the capacity of watersheds to regulate floods via natural and artificial features, my study addressed three main questions: (1) What inland floods can be managed by changing the landscape structure?, (2) How, where and why have flood regimes changed from 1991 to 2013 given changes in landscape structure?, and (3) How can interactions among biophysical and technological features of watersheds, along with long-term hydrologic data, be integrated to map the capacity of landscapes to regulate inland floods?

Chapter 1: What inland floods can be managed?

Abstract

Human-induced changes in landscape structure directly alter the magnitude, timing, frequency and rate of change of riverine flows and floods, with direct implications for people and aquatic environments. For the most part, studies that define the types of floods that landscape features (topography, land cover and impoundments, among others) can curtail are lacking. This study defines parameters of floods for which changes in landscape structure can have an impact. We compare nine flood return periods of 31 watersheds across North Carolina and Virginia using long-term hydrologic records (≥ 20 years). We also assess the performance of flow-regulating features (best management practices and artificial water bodies) on selected flood regime metrics across urban watersheds. Our results suggest that only those floods with return periods ≤ 10 years can be managed by manipulating landscape structure. Overall, urban watersheds exhibited larger but quicker floods than non-urban watersheds. However, urban watersheds with greater flow-regulating features had lower flood magnitudes, but similar flood durations compared to urban watersheds with few flow-regulating features. Our analysis provides insight into the magnitude, duration and count of floods that can be curtailed by landscape structure. This knowledge can help ensure that investment in flood management is cost-effective and that a balance is maintained between managing floods to reduce their socio-economic costs and maintaining healthy aquatic environments.

Introduction

Floods occur naturally and aquatic ecosystems have adapted to these events (Black 2012); however humans are in dissonance with floods because of the casualties and damage they cause. Patterns of flooding determine stream morphology (Newbury and Gaboury 1993), which in turn influences the distribution, abundance and diversity of stream organisms (Poff et al. 1997). Many stream biota depend on natural flooding patterns to persist (Bunn and Arthington 2002). In contrast, humans are often negatively affected by floods. Floods, inland and coastal, make up 45% of all natural disasters worldwide, and close to 20% of all economic damage (International Federation of Red Cross and Red Crescent Societies 2013). The socioeconomic impacts of damaging floods are not only a result of changing climatic patterns, but also of land use policy and floodplain occupancy (Pielke et al. 2000). Inland flood fatalities exceed those of coastal floods, as the area affected by inland floods is generally greater (Rappaport 2000).

Both aquatic ecosystems and humans are affected by the timing, variability and quantity of inland floods. Some of the key metrics used to describe flooding regimes across spatial and temporal scales include flood duration, magnitude and count (Poff et al. 1997, Olden and Poff 2003). Flood duration is the length of time a particular flood exceeds a certain flow threshold. Flood magnitude is the amount of discharge passing a fixed location, and flood count is the number of floods exceeding a certain flow threshold. Aquatic ecosystems are sensitive to the amount, variability and timing of recurrent floods (Poff and Ward 1989), while humans are impacted mostly by high magnitude floods (Yen 1995). Extended flooding significantly lowers property values, in addition to immediate damages to property (Filatova et al. 2014).

Inland floods are primarily driven by precipitation patterns (Kochenderfer et al. 2007, Lecce and Kotecki 2008, Tran et al. 2010), although other natural and anthropogenic features can alter their characteristics (Eng et al. 2013b). Vegetation, especially forest, plays an important role in regulating the runoff rate to streams through evapotranspiration, interception and infiltration (Lana-Renault et al. 2011, Brown et al. 2013). These mechanisms delay and reduce the amount of water that reaches streams. Floodplains receive the excess flow over stream banks, regulating the amount and timing of water flowing downstream (Golet et al. 2006, Moss 2007). Soil type and topography also influence inland floods. Sandy and loamy soils reduce water runoff through infiltration, while clay and silt soils enhance flooding. In flat terrain, close proximity between surface and ground water can cause overland flooding by a rise in the water table and aquifer saturation (Barron et al. 2011), but steeper slopes make water run off faster which can contribute to flooding downstream (Wardrop et al. 2005).

Natural and anthropogenic landscape features can substantially affect inland floods, but their effectiveness decreases as flood return period increases (Tollan 2002, Bloschl et al. 2007). Depending on the flood, landscape features can be more or less important in dictating the degree of flooding (Kundzewicz 1999). Smaller floods are more responsive to landscape structure (i.e., landscape features that are within the management realm of humans, not climatic or topographic features), with lower-peaked and longer duration floods in forested watersheds but higher-peaked and shorter duration floods in urban watersheds (Magilligan and Stamp 1997, Findlay and Taylor 2006, Hawley and Bledsoe 2011). Sturdevant-Rees et al. (2001) found no evidence of forested watersheds reducing peak runoff volumes for the 100-year flood. Similarly, artificial water bodies affect flooding only up to the point where runoff equals their storage capacity (Sordo-Ward et al. 2012). These studies suggest there is a flood-size threshold above which watersheds with different landscape characteristics respond the same (Figure 1.1).

Changes in land use and cover, driven by economic development (Brody et al. 2011), impact the natural capacity of landscapes to regulate floods. Loss of vegetation, in particular forest and wetland cover, decreases evapotranspiration and water retention, enhancing surface runoff (Bradshaw et al. 2007, Brown et al. 2013). Many floodplains have lost their capacity to buffer floods as encroachment of development and agriculture ruptures the lateral and longitudinal connectivity (Moss 2007). Intensive agriculture often compacts and erodes the soil, reducing water infiltration (Raper 2005), in addition to reducing the water retained in the soil (Van Wie et al. 2013). Impervious surfaces (e.g., roads, roofs) enhance surface runoff more than any other land cover conversion (Beighley and Moglen 2003, Huang et al. 2008), leading to flashier, higher-peaked and shorter floods (Leopold 1968). Reductions in the ability of landscapes to regulate floods exacerbates stream bank erosion, with consequences for transportation infrastructure (Dutton 2012) and water quality, particularly for downstream users (Brabec et al. 2002).

Flood control structures, a common management tactic, are selectively placed in the landscape to prevent or reduce flooding (Lehner et al. 2011b). Such structures act locally in the sense that their specific location is where peak flows are controlled. Levees and channelization keep water in-channel, limiting the lateral extent of flooding (Keller and Ketcheson 2011). These structures increase water conveyance from the immediate surroundings, thereby reducing local flooding by pushing river flow downstream. Impoundments, on the other hand, reduce local peak flows and flooding downstream by impounding water in or out of river channels. Other flood management tactics include bioretention areas and constructed wetlands that enhance infiltration and remove pollutants (Wossink and Hunt 2003). Building flood control structures is a technological response to the loss of the natural capacity of landscapes to regulate floods (e.g., forests, wetlands, floodplains), particularly common in agricultural and urban areas.

If managing floods were easy or straightforward, it would not be an issue. Investment in infrastructure for flood control has, in some cases, decreased causalities and damages. For example, the United States Army Corps of Engineers (USACE) invested $30 billion up to 1993 in structural flood control, avoiding $170 billion in damage since 1983 (Lovelace et al. 1994). For every dollar invested in reducing flood risk, the USACE has prevented $7.17, controlling for inflation, in damage from 1928-2009 (USACE 2011). However, property damage from flooding continues to increase (Milly et al. 2002, Patterson and Doyle 2009, Highfield et al. 2014). Extreme floods exact significant tolls on society. For example, since January 1978, the claims and payments from river flooding in North Carolina and Virginia amount to $970 and $600 million respectively (FEMA 2013a), amounting to 4% of flood claims in the US. North Carolina has had 14 major flooding disaster declarations spanning from 1962 to 2013, while Virginia has had 27 major disaster declarations from 1957 to 2009 (FEMA 2014). Inland flooding nationwide has exposed human's vulnerability to floods, and identified the importance of policies and land use planning to manage hazards and risks (Kaźmierczak and Cavan 2011). Mixed evidence on the effectiveness of flood control structures, in addition to high installation and maintenance costs (Thurston et al. 2003), legislative and institutional barriers (Roy et al. 2008), and continued adverse impacts on aquatic environments (Booth et al. 2002) raise important questions about the efficacy of current flood control strategies.

To reduce the socio-economic damage and environmental impacts of inland flooding, a greater understanding is needed regarding which floods can be curtailed by landscape structure. Understanding which floods can be managed is a necessary first step to examine the performance of flow-regulating features in regulating floods. In this study, we examine floods at nine return periods in selected watersheds throughout North Carolina and Virginia using data from long term gaging stations. Our objectives are to (1) define watershed types in relation to flooding, 2) identify a threshold of manageable floods based on flood magnitude, duration and count, 3) assess effects of flow-regulating features on flooding in urban watersheds, and 4) provide new information to help manage floods more effectively.

Methods

Study Area

The 31 gaged watersheds selected in this study represent diverse landscapes across Virginia (VA) and North Carolina (NC), yet share a similar climate (Patterson et al. 2012). Localized population growth patterns in recent times have concentrated in urban areas (Young 2014, Borders 2014) with little change in non-urban areas (Mogollón et al. n.d.). These spatially explicit patterns of growth or lack thereof allow us to compare the flood hydrology among watershed types for the past twenty years. Widely varying topographic and land use configurations were obtained as watersheds were distributed throughout major physiographic provinces, with most watersheds in the Piedmont region and others in the Coastal Plain, Valley and Ridge, and Blue Ridge regions (Appendix A; Figure 1.2).

We selected these watersheds based on size ($\leq 80 km^2$) and availability of instantaneous discharge record (period $\geq$ 20 years). We selected instantaneous records, as opposed to daily averages, to capture the peaks of floods that might be obscured in daily average data (Rice and Hirsch 2012). We limited the analysis to relatively small watersheds to highlight the effect of land cover on flooding, as the influence of anthropogenic disturbance on stream flow strongly decreases with increasing watershed size (Tollan 2002, Bloschl et al. 2007, Chang and Franczyk 2008, Petrow and Merz 2009).

Each watershed was delineated in ArcGIS 10.1 (ESRI 2012) using the National Hydrography Database plus version 2, 30-m flow accumulation and direction layers; pour points were derived from the gage location provided in the United States Geological Survey's (USGS) flow records.

Hydrologic Analysis

We downloaded peak and instantaneous discharge records from the USGS Water for the Nation Database (USGS 2013). Using the peak annual discharge records, we used USGS's PeakFQ program to derive the discharge values for 1.005- (henceforth 1-year flood), 1.5-, 2-, 5-, 10-, 20-, 50- and ≥100-year floods and manually estimated 80% discharge of a 1-year flood. We defined the lowest flood as 80% of a 1-year flood, which is a flood that happens, on average, multiple times a year. This arbitrary threshold enabled us to assess small changes in the flood regime over time (Huang et al. 2008) at a flow below the bank-full discharge (Poff and Ward 1989; Bunn and Arthington 2002). Striving to represent trends in hydrology, as opposed to yearly variation in precipitation, we used at least 21 years of peak discharge records to derive return periods in PeakFQ.

Available periods of instantaneous records ranged from 20 to 28 years, but we limited the analysis to 1991-2013, as this period held a continuous record for most watersheds. We defined a complete water-year discharge record as one having at least 300 days of data. We tabulated the number, magnitude and duration of floods (independent events); a flood could not last < 24 hours (Appendix B). Since we tabulated flood metrics by water year, all floods had to be assigned to a single water year. We had 29 events that spanned water years (1.6% of the records). We split these into independent floods by water year. For example, if a flood occurred from September 30th to October 1st, we counted it as two events (flood count = 1 for each water year), and summed the corresponding duration and magnitude for each event.

Based on the nine flood return-interval thresholds derived from the peak discharge records, we compiled the number, magnitude and duration of independent floods for each return period per water year. We define flood count as the number of times per water year the discharge of an independent event equaled or exceeded 80% of the discharge of a 1-year flood. Flood magnitude is the discharge above 80% of a 1-year flood. Flood duration is the length of time when discharge exceeds the flood threshold per flood event.

Twelve watersheds had one to seven years of missing discharge data, for a total of 44 years missing (6% of the entire record). One watershed had seven consecutive years of missing data (gage failed in 2006), one had six consecutive years (1992-1997), three gages had four consecutive years, four gages had two or three consecutive years, and three had one year of missing data. We interpolated the missing years for a given watershed, instead of discarding these watersheds, by taking the average of the two years preceding and succeeding the missing values. We verified that the interpolated records fell within the range of values for watersheds that had discharge records for each water year.

Land Cover Analysis

To classify each watershed by its dominant land cover, we first assembled comparable and publicly available land cover databases for the five time periods available. The 1992-Enhanced database is the product of merging the Land Use Data and Analysis 1970-1985 database, known as GIRAS (Geographic Information Retrieval and Analysis System) and the National Land Cover Database (NLCD) 1992 (Nakagaki et al. 2007). The 1992-Enhanced database provides a representation of land cover for the early 1990s. The different classification methodologies used between NLCD 1992 (i.e. unsupervised) and NLCD 2001 (i.e. supervised) make these two layers directly incomparable. However, the retrofit products of 1992 and 2001 make the data for these two time periods compatible (Fry et al. 2009). We also used the 2006 (Fry et al. 2011) and 2011 (Jin et al. 2013) NLCD.

We aggregated land cover classes from the NLCD for five time periods (1992-Enhanced, 1992-Retrofit, 2001-Retrofit, 2006 and 2011) into three major groups: forest, urban and agriculture (Appendix C). These groups accounted for 97-100% of the land cover in the study watersheds, except one with 15% barren land; other land covers were mainly water and barren lands. In aggregating major land cover classes by watershed, we minimized the error produced in comparing the land cover databases (Price et al. 2003). To classify each watershed by its dominant land cover, we conducted a hierarchical cluster analyses based on the k-means procedure using the percentage of forest, urban and agriculture for the five time periods in JMP® Pro 10 (SAS Institute Inc. Cary, NC).

Spatial and Temporal Autocorrelation

Before examining relations between flood metrics and landscape structure, we first assessed the temporal and spatial correlation structure of our flood regime metrics. Not controlling for such correlation can reduce the effective number of samples available to test trends (Douglas et al. 2000). We visually inspected plots of temporal autocorrelation (Appendix D). When strong autocorrelation is present the covariance function between previous time periods will be large and consistently exceed the critical values for the null hypothesis of no autocorrelation. In our case correlations attenuated quickly; information from the previous year provides little knowledge about the next period. Additionally, we used the Durbin –Watson (DW) statistic to assess the strength of autocorrelation; a null hypothesis of no autocorrelation (DW ≈ 2) was tested. DW statistics indicated no significant temporal autocorrelation among years of flow data (Appendix D, Table 1.1).

We tested the spatial correlation structure of the flow data by running a single set of Markov chain Monte Carlo (MCMC) simulations for the flood regime metrics. We used the following spatial model,

$$\tilde{Y}_t = X_t\beta + Z\theta + \epsilon_t,$$

where $\tilde{Y}_t$ is a transformed flood regime metric, X_t is a matrix containing watershed characteristics and β is a vector of estimated coefficients relating those characteristics to the flood regime metrics. The matrix Z corresponds to a vector of random effects, θ, which quantifies the effects of individual watersheds. Finally, the error term, $\epsilon_t \sim N(0, \sigma^2 H(\phi) + \tau^2 I)$, where $N(\mu,\sigma^2)$ denotes a normal distribution with mean = μ and variance $=\sigma^2$ and I is an identity matrix. The spatial error structure is contained in $\sigma^2 H(\phi)$, where $H(\phi)$ is a spatial covariance matrix such as $exp(-d/\phi)$, where d = Euclidean distance between two points.

We assessed the effect of spatial correlation in the model using two parameters that control the covariance induced by the spatial structure, ϕ and σ2, and compared them to τ2 which is the variance. Hence, the spatial structure is controlled by the function of two parameters, σ2H(ϕ). Discerning the spatial structure requires considering them jointly as interpreting them individually can be misleading. For example, ϕ can be large relative to the distance between sites which means spatial correlation is high; however, if σ2 is small the spatial covariance will be small. The appropriate comparison is between the covariance from the spatial structure, σ2H(ϕ), and the observation variance, τ2. Given that τ2I is several orders of magnitude larger than the elements of σ2H(ϕ) (Table 1.1), we found no significant spatial correlation structure in the data. The lack of significant temporal or spatial autocorrelation in our data enabled us to treat the yearly records as independent observations.

Flood Return Period Threshold

We used generalized linear mixed models (GLMMs) to evaluate the flood threshold for which changes in landscape structure can influence flooding. GLMMs handle non-normal data and incorporate random effects (Bolker et al. 2008). To examine how different flood return periods respond to variation in dominant land cover type, we ran one GLMM per flood return period (80% of a 1-, 1-, 1.5-, 2-, 5-, 10-, 20-, 50- and ≥ 100-year floods) per flood metric (count, duration and magnitude). We used gage identification number (ID) as a random effect, and dominant land cover (from the cluster analysis) and flood interval as categorical fixed effects. For each return period, we compared the least-squares means of the watersheds with different dominant land cover types to determine the direction and magnitude of the effect of dominant land cover on flood count, duration and magnitude using Bonferronni-adjusted confidence intervals.

Before running the GLMMs, we examined the flood magnitude, duration and count distributions for each flood return periods for goodness-of-fit to a number of possible distributions. We used the overdispersion parameter (i.e., $\hat{c}$ = Pearson Chi-Square/df) of the intercept-only model to test the distribution fit of flood metrics at each return period using distributions from the exponential family, and to test the assumption of parameter homogeneity (Anderson 2008). Violating the parameter homogeneity assumption leads to a small sample variance which results in a false sense of precision (Anderson 2008). A small overdispersion parameter ($\hat{c} \leq 1$) suggests the data suitably fits the selected distribution. For small return periods (≤ 2 years), we found that lognormal (link=identity) and Poisson (link=log) distributions best describe flood duration and count, while the negative binomial distribution best describes flood magnitude. Greater return periods (> 2 years) were zero-heavy; thus, the negative binomial distribution (link=log) best describes flood duration, while negative binomial (link=log) and Poisson (link=log) distributions best describe flood magnitude and count (Appendix E). In selecting an appropriate distribution, we can with greater certainty make statistical inference about our flood metrics.

Other Landscape Features

Since other climatic and landscape features also influence flows and floods (Wilby and Keenan 2012) we summarized the precipitation, hill slope and soil type for each watershed land cover category from our cluster analysis. We derived the mean annual precipitation from the PRISM database from 1991 to 2012 (PRISM Climate Group 2013), mean hill slope from the National Hydrography Database (NHD 2013), and the percentage of sandy and loamy soils (A and B soil hydrologic group) from SSURGO (NRCS 2012). We processed this information in ArcGIS version 10.1 (ESRI 2012).

Flow-regulating Features and Flooding

We assessed the effect of flow-regulating features on flood magnitude and duration by comparing urban watersheds ($\geq$ 30% urban cover) with different areal proportions of BMPs (e.g. wet and dry ponds, bioretention areas, stormwater wetlands, and sand filters) and artificial water bodies (e.g. farm ponds, golf course ponds, water supply reservoirs; henceforth, AWBs) to forested watersheds. We selected flood records based on the maximum flood return period for which landscape structure can curtail floods (derived from the first study objective).

We derived locations of BMPs and AWBs (jointly, flow-regulating features) from multiple sources: the United States Army Corps of Engineers' National Inventory of Dams (USACE 2012), Global Reservoirs and Dams Database (Lehner et al. 2011a), National Anthropogenic Barrier Dataset (Ostroff et al. 2012), Federal Emergency Management Agency's National Flood Hazard Layer (FEMA 2013b), National Hydrograph Database on Waterbodies (USGS 2012), the North Carolina Department of Environment and Natural Resources' dam inventory (NCDENR 2013), Virginia Database (personal communication Mark Bradford, Virginia Department of Conservation and Recreation), and county BMP databases (from contacts at individual counties). We used Google Earth aerial imagery to verify the existence of, map the surface area of, and date BMPs and AWBs. We used surface area as a surrogate for volumetric capacity, as information on volumetric capacity was largely unavailable. We standardized BMP and AWB area across urban watersheds by calculating the cumulative percentage of BMP (% BMP) and AWB (% AWB) surface area by watershed area per year from 1991 to 2013. We weighted flow-regulating capacity of AWBs and BMPs equally, because although AWBs are not constructed to control floods, they store water during floods, and are ubiquitous across the landscape (Smith et al. 2002; Ignatius & Jones 2014).

To compare urban watersheds with different proportional coverage of flow-regulating features, we summed %BMPs and %AWB, and divided the sums into three categories based on natural breaks in the data: low (≤ 0.02%), medium (0.16% to 0.22%), and high (0.43% to 2.04%). Watersheds did not change category during the period of study. We compared the three flow-regulating feature categories in urban watersheds to our nine forested watersheds (≥89% forested). We examined the effect of these four categories (Low, Medium, High and Forested) on selected flood metrics with GLMMs. We used gage ID and watershed size as random effects, and the four categories (urban-low, urban-medium, urban-high, and forested) and mean annual precipitation as fixed effects. Prior to running the GLMMs, we examined the appropriate distribution of flood metrics and determined that a lognormal transformation (link=identity) best described magnitude and mean duration. We compared the least-square means of the four categories using Bonferronni-adjusted confidence intervals.

Results

Land Cover Analysis

Our cluster analysis of watershed land-cover composition revealed four major watershed types: forested (n=9; 89-98% forest), semi-forested (n=7; 50-80% forest), rural (n=6; 28-53% agriculture; 32-64% forest) and urban (n=9; 40-100% urban; 0-53% forest) (Appendix F). Based on the cluster analysis distances, urban watersheds were the most different from other watershed types; forested and semi-forested watersheds were the most similar. Of the 31 watersheds, only three changed types over the 23 years. Two watersheds changed from 1990 to 1992, one from semi-forested to urban (02142900) and the other from rural to urban (02099000). The third watershed shifted from rural to semi-forested (01673550) from 2001 to 2006.

Flood Return Interval Threshold

Flood Magnitude

We found urban watersheds had significantly higher flood magnitudes up to the 2-year flood compared to non-urban watersheds, and this trend continued, albeit non-significant, for the 5- and 10-year floods (Table 1.2; Figure 1.3). Flood magnitude differences between urban and non-urban watersheds diminished as return period increased. At 80% of a 1-year flood, flood magnitude was four times greater in urban watersheds than non-urban watersheds, nearly three times greater for the 1-year flood, and two times greater for the 1.5- and 2- year floods. The 5- and 10-year floods were 1.5 and 1.0 times greater in urban watersheds, respectively, than in non-urban watersheds, yet these differences were not significant.

Surprisingly, forested, semi-forested and rural watersheds shared similar flood magnitude responses across flood return periods, particularly for small floods (Figure 1.3) but different features seem to drive the similarity. Our forested watersheds overall receive higher mean annual rainfall (1308 mm) and have steeper slopes (25.6°), probably due to orographic effects, as these watersheds are mainly in the mountainous Blue Ridge and Valley and Ridge physiographic provinces. Steeper slopes and greater rainfall would result in rapid surface runoff, contributing to higher flood magnitudes. In contrast, semi-forested and rural watersheds receive lower mean annual precipitation (1121mm and 1085mm respectively) and have flatter slopes (9.8° and 9.6°, respectively), as they are mainly in the Piedmont and Coastal Plain provinces, resulting in lower surface runoff and potentially less flooding. Semi-forested and rural watersheds also have greater flow-regulating features (0.83% and 0.90%, respectively), than forested watersheds (0.13%), yet forested watersheds have more extensive forest vegetation, which also regulates surface flow. Higher precipitation and steeper slopes in forested watersheds, and the presence of flow-regulating features in semi-forested and rural watersheds help explain the similarities in response of flood magnitude among forested, semi-forested and rural watersheds (Table 1.3).

For floods larger than the 10-year flood, there were no differences or trends in flood magnitude among watershed types, except for the ≥100-year flood. We do not attribute observed differences in magnitude of ≥100-year floods among the four watershed types to different land cover types. Rather, we think they simply reflect where these rare events happened during our 23-year study. There were three ≥100-year floods, two in semi-forested watersheds in 1996 and 2008 and one in an urban watershed in 2010. The significant differences among urban, rural, semi-forested and forested watersheds for the ≥100-year flood do not follow the same pattern observed for small floods among the four watershed types (Table 1.2).

Flood Duration

Flood duration was significantly shorter in urban than non-urban watersheds for floods smaller than the 10-year flood (Table 1.2; Figure 1.3). The difference in flood duration between urban and non-urban watersheds was similar for all floods below the 10-year flood. On average, non-urban watersheds exhibited flood durations nearly three times longer than urban watersheds.

We found no significant differences among non-urban watersheds. Semi-forested watersheds had longer floods up to the 2-year flood, while rural watershed had longer floods for the 5- and 10-year floods. A greater areal extent of flow-regulation features in rural and semi-forested watersheds, in addition to the topographic and climatic differences discussed above, explains the slight, albeit non-significant, differences in flood duration among non-urban watersheds.

We found no differences in flood duration among watershed types for the 20- and 50- year floods; however, the ≥100-year flood did differ. As we found for flood magnitude, differences among urban, rural, semi-forested, and forested for the ≥100-year flood do not follow the same patterns observed for small floods among the four watershed types (Table 1.2).

Flood Count

The number of floods for most of the flood return periods was not different among watershed types. Urban watersheds had greater counts of the 80% of a 1-year flood, semi-forested watersheds had greater counts of the 1-year flood, and rural watersheds had greater counts of the 1.5-, 2- and 5- year floods. The number of floods per watershed type was similar for subsequent return periods, except for the 50-year flood, where differences among urban, rural, semi-forested and forested were found (Table 1.2). These differences are not likely related to land cover, but instead seem to reflect the five 50-year floods that occurred in forested and urban watersheds during our study period; four occurred in urban watersheds in 1996, 1997, 2010 and 2011, and one occurred in a forested watershed in 2004.

Flow-regulating Features and Flooding

Flow-regulating features had measurable impact on some floods in urban watersheds. We chose to compare the magnitude and duration of >5-year floods between forested and urban watersheds with low, medium, and high percentage of flow-regulating features so we could use a flood threshold that was present for both flood metrics. The extent of flow-regulating features was negatively related to magnitude and positively related to duration in urban watersheds but watersheds with extensive flow-regulating features still had larger, shorter floods than forested watersheds (Table 1.4). Urban watersheds with a low percentage of flow-regulating features had significantly larger floods than watersheds with a high percentage of flow-regulating features, but non-significant differences in flood duration. The differences in flood magnitude and duration between forested and urban watersheds decreased with greater flow-regulating features. Forested watersheds had flood magnitudes 14% as large as urban watersheds with a low percentage of flow-regulating features and 33% as large as urban watersheds with a high percentage of such features. Flood duration of forested watersheds was nearly seven times longer than urban watersheds with a low percentage of flow-regulating features and nearly three times longer than urban watersheds with a high percentage of such features.

Discussion

Which Inland Floods Can We Manage?

Our study shows that land cover can affect the duration and magnitude of floods up to a 10-year flood, while the number of floods, driven by precipitation patterns, is not impacted by land cover (Figure 1.3). Urban watersheds have higher and shorter floods than non-urban watersheds, despite ubiquitous flow-regulating features. Land cover is ineffective at regulating floods larger than the 10-year flood; these floods seem to be determined by other large scale drivers such as precipitation. Our results provide insight into which floods landscape managers can expect to curtail to decrease the socio-economic costs and environmental impacts of inland flooding, and the floods that are largely out of our control at the watershed-scale.

Results presented here provide empirical evidence, using a straightforward approach, of the flood return period where landscape structure can curtail flooding. Our results are consistent with other studies that found the greatest hydrologic response difference between natural and transformed landscapes is at small flood return periods (Leopold 1968, Hollis 1975, Smith et al. 2002a, Wissmar et al. 2004, Kochenderfer et al. 2007). Magilligan and Stamp (1997) modeled hydrologic alterations in a small watershed in Georgia by reconstructing past land cover, and found greater variability at the 2-year flood than at the 100-year flood through time. Findlay and Taylor (2006) report 2-year floods happening more frequently with increasing levels of urbanization. Hawley and Bledsoe (2011) found urbanization impacted the magnitude and duration of flows up to the 5-year flood. These studies are consistent with our findings of greater flood regulation capacity of the landscape for small floods (Figure 1.3).

In this study, we limited our watersheds by drainage size (< 80km^2) and restricted it to NC and VA that share a similar climate. Other studies, examining watersheds up to 250km^2, have also found differences in the magnitude and duration of small floods between watershed types (Smith et al. 2002a, Hawley and Bledsoe 2011). Sturdevant-Rees et al. (2001) found no evidence of forested watersheds reducing large peak runoff volumes for the 100-year flood in watersheds ranging from 200 to 7800km^2. These studies suggest that regardless of watershed size, small floods respond differently to land cover than larger floods. Further studies comparing flood return periods in larger watersheds are needed to confirm the applicability of our results at greater spatial extents. However, in expanding the spatial extent of the study area, there are greater differences in climatic and physiographic patterns.

Relying on long-term hydrologic records limited our ability to choose what watersheds were the most interesting to study based on land cover changes and physiographic characteristics. Ideally, we would have controlled for topographic and rainfall characteristics, yet then we would have been confronted with a small sample. In our study, almost all forested watersheds were in the mountainous region, while all urban watersheds were in the low elevation regions. For our study, the vast differences in the magnitude and duration of ≤10 year floods between forested and urban watersheds, confirms the great impact impervious surfaces have on floods, even when forested watersheds appear to be more susceptible to flooding (e.g., greater rainfall, steeper slopes) than urban watersheds. Further, even with ubiquitous flow regulating features in urban watersheds, these had flashier floods than forested watersheds. The impact of impervious surface on flows and flooding has been widely documented (Beighley and Moglen 2003, Moglen and Kim 2007), and our results suggest that reducing these impacts by increasing forest and wetland cover can provide an effective strategy to increase the infiltration, evapotranspiration and retention capacity of urban watersheds, and in this way reduce flashy floods.

Our ability to detect differences among non-urban watersheds is confounded by the differences in land cover, physiographic characteristics, and flow-regulating feature patterns. We would have expected rural, followed by semi-forested, watersheds to have shorter and higher floods than forested watersheds, as these had lower forest cover (Lana-Renault et al. 2011). Agriculture-dominated watersheds have been shown to have 8- 33% magnification of flood peaks compared to forested watersheds (Poff et al. 2006). However, similar flood magnitude and duration among forested, semi-forested and rural watersheds, suggests that widespread flow-regulating features in rural (0.90%) and semi-forested (0.83%) watersheds, counteracts the effects of urbanization (average of 13% and 8% urban cover, respectively) and agriculture (average of 41% and 18% agricultural cover) in these watersheds. Our inability to detect differences among non-urban watersheds suggests that land cover is not the only predictor of flooding, but that topography, soils, precipitation and flood control structures also play important roles. Future studies on non-urban watersheds warrant controlling for physiographic characteristics, to understand the role of land cover and flow regulating features on flooding.

Managing Urban Floods

Storage-based mechanisms to lower runoff have dominated stormwater management since the 1990s because rapid surface runoff decreases water quality via pollutant mobilization and channel erosion (Balascio and Lucas 2009). Within our study watersheds, the majority of BMPs (97%) store water: wet ponds (76%), flood control dams (11%) and dry ponds (10%); with 60% of them in-channel. We show that urban watersheds with extensive flow-regulating features exhibit longer duration and lower-peaked floods than watersheds with few flow-regulating features, suggesting BMPs do lower peak flows (Table 1.4; Figure 1.1). However, urban watersheds had, on average, ≤ 2-year floods that were three times larger than but only 33% as long as floods in non-urban watersheds. Great differences in small floods between urban and non-urban watersheds suggest limited effectiveness of flow-regulating features in urban watersheds (0.78%). Our results suggest that urban stormwater management to date has not entirely counteracted the effects of expanding impervious surfaces on flood magnitude and duration.

The major land cover difference between urban and non-urban watersheds is the extent of impervious surface, which alter the magnitude and duration, but not the number, of floods. Storage-based stormwater management intends to retain runoff from impervious surfaces, but these structures only marginally decrease watershed-wide peak flows, while significantly impacting watershed health (Booth and Jackson 1997, Emerson et al. 2005). Complementing storage-based efforts with infiltration-based mechanisms (e.g., rainwater harvesting, green roofs, permeable pavement, bioretention areas) can reduce surface runoff and impacts on aquatic environments (Deutsch et al. 2005, Williams and Wise 2006, Davis 2008, Czemiel Berndtsson 2010, Damodaram et al. 2010, EPA 2013), especially for small storms. However, we were unable to detect differences between storage- and infiltration- based BMPs because bioretention areas and sand filters were the only infiltration-based technologies that we documented, and these were rare (covering < 0.02 km^2). Although we expect widespread installation of more infiltration-based BMPs across urban areas to increase the duration of small floods, our results showed no effect of these BMPs, perhaps due to their rarity in our study area.

Finding the most effective set of stormwater management tactics, may require greater monitoring and documentation efforts. We found no single database that contained all the flow-regulating features in the landscape. County stormwater management offices, particularly in urban areas, stated that unifying this data was one of their priorities (selected city stormwater management offices, pers. comm.). A detailed database on flow-regulating features, describing the capacity to retain, evaporate or infiltrate water, would allow stormwater managers to understand the landscape configuration and distribution of flow-regulating features in relation to sources of runoff and areas with high flood risk across watersheds (Strecker et al. 2001). In contrast, most studies assess BMP effectiveness on a site-by-site basis (Davis 2008, Hancock et al. 2010), which provides limited insight into how an entire stormwater management strategy contributes to improving water quality and stream health (Harrell and Ranjithan 2003, Emerson et al. 2005).

Management and Policy Implications

Knowing which floods can be managed helps clarify two public misconceptions of flood management that continue to impose significant costs on society (Pielke 1999, Tran et al. 2010). The first misconception is that engineered structures prevent river flooding (Tobin 1995). Numerous historical events (e.g., 1927, 1937, 1973, 1993, and 2011 Mississippi River floods, Hurricanes Katrina and Sandy, Kissimmee River flooding) have been followed by construction of flood control structures, yet many of these structures have exacerbated damage (Koebel 1995, Pielke 1999, Criss and Shock 2001, Tollan 2002). While > 10-year floods have been contained in the areas benefited by these structures, in many cases the designed flood return period (e.g., 100-year flood for levees and dams) give a false sense of security, which encourages development in high-risk areas (Highfield et al. 2013). For example, under the National Flood Insurance Program, lands behind a 100-year flood levee are considered protected against floods, which has facilitated construction on these lands, as they are perceived as safe (Ludy and Kondolf 2012). Such flood control structures have failed occasionally, causing widespread damage locally and downstream (Doyle et al. 2008; Pielke 1999).

The second misconception is that damages from catastrophic floods are consequences of the loss of natural ecosystems (e.g., forests and wetlands) (Lecce and Kotecki 2008). Such was the case with eastern North Carolina’s 1999 flood, labeled as greater than a 500-year flood (Bales et al. 2000), where estimated damages ranged from $3 to $6 billion dollars (Pasch et al. 2014). Lecce & Kotecki (2008) found no relation between human-induced land cover changes and flood severity in prior decades by comparing the hydrologic record to population growth, number of housing units, and area under cultivation from 1930 to

2000. Other studies have shown that natural ecosystems have a marginal impact in mitigating catastrophic flooding, despite public perception to the contrary (Calder and Aylward 2006, Tran et al. 2010). The perception that natural ecosystems mitigate extreme floods has significant implications for land use; particularly for upstream communities that are blamed for flood damages downstream (Tran et al. 2010). Although rainfall patterns largely drive extreme floods (Öztürk et al. 2013; Tran et al. 2014), millions of dollars have been spend to regulate floods by changing land use (e.g., afforestation projects), which has marginal impact on flood damage (Calder and Aylward 2006).

Damage from catastrophic floods is well documented; however, more recurrent small floods can also cause significant damage, particularly in high-density urban areas as they are the flows that control channel dimensions (Green and Penning-Rowsell 1989, Doyle et al. 2007, Lantz et al. 2012). In Charlotte, NC, the number of private properties reached by a 25-year flood (i.e., flood with 4% chance of occurring in a given year), is significant in residential and commercial areas (Figure 1.4). Managers there are delimiting floods as small as the 2-year flood to develop sound floodplain management programs and determine flood insurance rates (FEMA and State of North Carolina 2014). Our study shows that purposeful changes to the landscape can alter the magnitude and duration of ≤ 10-year floods (Table 1.2). Such up-slope management actions could complement floodplain management to further reduce property damage during small floods. Aside from damage to private property, floods greater than the bank-full discharge (1- to 3-year floods), have the potential to disrupt transportation systems (e.g., flooded roads) during floods, as well as incrementally destabilize stream banks alongside roads and bridges (Dutton 2012).

Conclusions

Few studies have examined the river flows for which changes in landscape structure can curtail floods across a large number of watersheds. Our study shows that for floods recurring at intervals > 10 years (e.g., large floods), flood magnitude and duration do not differ among watersheds with different land cover compositions. For floods recurring at intervals ≤ 10 years (e.g., small floods), urban watersheds generate larger and shorter floods, than non-urban watersheds. However, adding flow-regulating features, particularly storage-based features, to urban watersheds can significantly reduce flood magnitudes but not flood durations for floods smaller than a 5-year flood. These patterns suggest that management efforts to decrease the socioeconomic and environmental impacts of small floods can be grouped into two main strategies: 1) increase the incidence of infiltration- and storage-based flow-regulating features throughout urban watersheds, where space is limited, and 2) increase forest cover of watersheds to reduce runoff by enhancing water retention, infiltration and evapotranspiration. Large floods seem to be beyond the influence of conventional management interventions. Thus, a greater focus on managing flood risk (e.g., land use planning and zoning, education and outreach, and early warning systems) might be the most effective strategy to lower the socioeconomic costs of inland flooding (Hansson et al. 2008, Brody et al. 2011).

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References

Anderson, D. R. 2008. Model Based Inference in the Life Sciences: A Primer on Evidence. Springer, New York City.

Balascio, C. C., and W. C. Lucas. 2009. A survey of storm-water management water quality regulations in four Mid-Atlantic States. Journal of environmental management 90(1):1–7.

Bales, J. D., C. J. Oblinger, and A. H. J. Sallenger. 2000. Two Months of Flooding in Eastern North Carolina , September – October 1999 Hydrologic, Water Quality, and Geologic Effects. Page Water Resources Investigations Report 00–4093. Raleigh.

Barron, O. V., D. Pollock, and W. Dawes. 2011. Evaluation of catchment contributing areas and storm runoff in flat terrain subject to urbanisation. Hydrology and Earth System Sciences 15(2):547–559.

Beighley, R. E., and G. E. Moglen. 2003. Adjusting measured peak discharges from an urbanizing watershed to reflect a stationary land use signal. Water Resources Research 39(4):1–11.

Black, P. E. 2012. The U.S. Flood Control Program at 75: Environmental Issues. Journal of the American Water Resources Association 48(2):244–255.

Bloschl, G., S. Ardoin-Bardin, M. Bonell, M. Dorninger, D. Goodrich, D. Gutknecht, D. Matamoros, B. Merz, P. Shand, and J. Szolgay. 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes 1247(January):1241–1247.

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J.-S. White. 2008. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology & Evolution 24(3):127–135.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38(3):835–947.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detection, and the limits of mitigation. Journal of the American Water Resources Association 33(5):1077–1090.

Borders, S. 2014, March 30. Census data shows No.Va. drives Virginia growth. The Cavalier Daily. Charlottesville, VA.

Brabec, E., S. Schulte, and P. L. Richards. 2002. Impervious surfaces and water quality: A review of current literature and its implications for watershed planning. Journal of Planning 16(4):499–514.

Bradshaw, C. J. A., N. S. Sodhi, K. S.-H. Peh, and B. W. Brook. 2007. Global evidence that deforestation amplifies flood risk and severity in the developing world. Global Change Biology 13(11):2379–2395.

Brody, S. D., W. E. Highfield, and J. E. Kang. 2011. Rising Waters: The Causes and Consequences of Flooding in the United States. Page 195. Cambridge University Press.

Brown, A. E., A. W. Western, T. A. McMahon, and L. Zhang. 2013. Impact of forest cover changes on annual streamflow and flow duration curves. Journal of Hydrology 483:39–50.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30(4):492–507.

Calder, I. R., and B. Aylward. 2006. Forest and Floods : Moving to an Evidence-based Approach to Watershed and Integrated Flood Management. Water International 31(1):1–13.

Chang, H., and J. Franczyk. 2008. Climate change, land use change, and floods: Towards an integrated assessment. Geography Compass 2(5):1549–1579.

Criss, R. E., and E. L. Shock. 2001. Flood enhancement through flood control. Geology 29(10):875.

Czemiel Berndtsson, J. 2010. Green roof performance towards management of runoff water quantity and quality: A review. Ecological Engineering 36(4):351–360.

Damodaram, C., M. H. Giacomoni, C. Prakash Khedun, H. Holmes, A. Ryan, W. Saour, and E. M. Zechman. 2010. Simulation of Combined Best Management Practices and Low Impact Development for Sustainable Stormwater Management. Journal of the American Water Resources Association 46(5):907–918.

Davis, A. P. 2008. Field Performance of Bioretention: Hydrology Impacts. Journal of Hydrologic Engineering 13(2):90–95.

Deutsch, B., H. Whitlow, M. Sullivan, and A. Savineau. 2005. A Green Roof Vision Based on Quantifying Storm Water and Air Quality Benefits. Pages 379–384 in The Cardinal Group Toronto, editor. Proceedings of the 3rd North American: Green Roof Conference on Greening Rooftops for Sustainable Communities. Washington, DC.

Douglas, E. M., R. M. Vogel, and C. N. Kroll. 2000. Trends in floods and low flows in the United States: impact of spatial correlation. Journal of Hydrology 240:90–105.

Doyle, M. W., D. Shields, K. F. Boyd, P. B. Skidmore, and D. Dominick. 2007. Channel-Forming Discharge Selection in River Restoration Design. Journal of Hydraulic Engineering 133(7):831–837.

Doyle, M. W., E. H. Stanley, D. G. Havlick, M. J. Kaiser, G. Steinbach, W. L. Graf, G. E. Galloway, and J. A. Riggsbee. 2008. Aging Infrastructure and Ecosystem Restoration. Science 319:286–287.

Dutton, P. D. 2012. Factors influencing roadside erosion and in-stream geomorphic stability at road-stream crossings for selected watersheds, North Shore, Minnesota, USA. Master's Thesis Natural Resources Science and Management, University of Minnesota.

Emerson, C. H., C. Welty, and R. G. Traver. 2005. Watershed-Scale Evaluation of a System of Storm Water Detention Basins. Journal of Hydrologic Engineering 10(June):237–242.

Eng, K., D. M. Wolock, and D. M. Carlisle. 2013. River flow changes related to land and water management practices across the conterminous United States. The Science of the Total Environment 463-464:414–22.

EPA. 2013. Proposed National Rulemaking to Strengthen the Stormwater Program. Alexandria, VA. http://cfpub.epa.gov/npdes/stormwater/rulemaking.cfm.

ESRI. 2012. ArcGIS Desktop. Environmental Systems Research Institute, Redlands, CA.

FEMA. 2013a. Claim Information by State (1978- current month). http://www.fema.gov/policy-claim-statistics-flood-insurance/policy-claim-statistics-flood-insurance/policy-claim-13.

FEMA. 2013b. Map Service Center. https://msc.fema.gov/webapp/wcs/stores/servlet/info?storeId=10001&catalogId=10001&langId=-1&content=productDFIRM&title=DFIRM Databases&parent=productInfo&parentTitle=Product Information.

FEMA. 2014. Disaster Declarations. http://www.fema.gov/disasters.

FEMA, and State of North Carolina. 2014. Flood Insurance Study. Pages 1–93. Charlotte, NC.

Filatova, T., O. Bin, B. A-, T. Street, and N. Carolina. 2014. Advances in Social Simulation 229:249–258.

Findlay, S. J., and M. P. Taylor. 2006. Why rehabilitate urban river systems? Area 38(3):312–325.

Fry, J. A., M. J. Coan, C. G. Homer, D. K. Meyer, and J. D. Wickham. 2009. Completion of the National Land Cover Database (NLCD) 1992 – 2001 land cover change retrofit product. Page 18.

Fry, J., G. Xian, S. Jin, J. Dewitz, C. Homer, L. Yang, C. Barnes, N. Herold, and J. Wickham. 2011. Completion of the 2006 National Land Cover Database for the Conterminous United States. Photogrammetric Engineering and Remote Sensing 77(9):858–864.

Golet, G. H., M. D. Roberts, E. W. Larsen, R. a Luster, R. Unger, G. Werner, and G. G. White. 2006. Assessing societal impacts when planning restoration of large alluvial rivers: a case study of the Sacramento River project, California. Environmental management 37(6):862–79.

Green, C. H., and E. C. Penning-Rowsell. 1989. Flooding and the Quantification of "Intangibles". Water and Environment Journal (8):27–30.

Hancock, G. S., J. W. Holley, and R. M. Chambers. 2010. A Field-Based Evaluation of Wet Retention Ponds: How Effective Are Ponds at Water Quantity Control? Journal of the American Water Resources Association 46(6):1145–1158.

Hansson, K., M. Danielson, and L. Ekenberg. 2008. A framework for evaluation of flood management strategies. Journal of environmental management 86(3):465–80.

Harrell, L. J., and S. R. Ranjithan. 2003. Detention Pond Design and Land Use Planning for Watershed Management. Journal of Water Resources Planning and Management 129:98–106.

Hawley, R. J., and B. P. Bledsoe. 2011. How do flow peaks and durations change in suburbanizing semi-arid watersheds? A southern California case study. Journal of Hydrology 405(1-2):69–82.

Highfield, W. E., S. D. Brody, and R. Blessing. 2014. Measuring the impact of mitigation activities on flood loss reduction at the parcel level: the case of the clear creek watershed on the upper Texas coast. Natural Hazards:1–18.

Highfield, W. E., S. A. Norman, and S. D. Brody. 2013. Examining the 100-year floodplain as a metric of risk, loss, and household adjustment. Risk analysis : an official publication of the Society for Risk Analysis 33(2):186–91.

Hollis, G. E. 1975. The effect of urbanization on floods of different recurrence interval. Water Resources Research 11(3):431–435.

Huang, S., S. Cheng, J. Wen, and J. Lee. 2008. Identifying peak-imperviousness-recurrence relationships on a growing-impervious watershed, Taiwan. Journal of Hydrology 362(3-4):320–336.

Ignatius, A., and J. Jones. 2014. Small reservoir distribution, rate of construction, and uses in the upper and middle Chattahoochee basins of the Georgia piedmont, USA, 1950–2010. International Journal of Geo-Information 3(2):460–480.

International Federation of Red Cross and Red Crescent Societies. 2013. World Disasters Report 2013 Focus on Technology and the future of humanitarian action. Geneva.

Jin, S., L. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment 132:159–175.

Kaźmierczak, A., and G. Cavan. 2011. Surface water flooding risk to urban communities: Analysis of vulnerability, hazard and exposure. Landscape and Urban Planning 103(2):185–197.

Keller, G., and G. Ketcheson. 2011. Storm Damage Risk Reduction. Transportation Research Record: Journal of the Transportation Research Board 2203(-1):211–218.

Kochenderfer, J. N., M. B. Adams, G. W. Miller, and J. D. Helvey. 2007. Factors Affecting Large Peakflows on Appalachian Watersheds: Lessons from the Fernow Experimental Forest. Pages 1–24. Research Paper NRS-3 Newtown Square, PA.

Koebel, J. W. 1995. An historical perspective on the Kissimmee River Restoration Project. Restoration Ecology 3(3):149–159.

Kundzewicz, Z. W. 1999. Flood protection — sustainability Issues. Hydrological Sciences Journal 44(4):559–571.

Lana-Renault, N., J. Latron, D. Karssenberg, P. Serrano-Muela, D. Regüés, and M. F. P. Bierkens. 2011. Differences in stream flow in relation to changes in land cover: A comparative study in two sub-Mediterranean mountain catchments. Journal of Hydrology 411(3-4):366–378.

Lantz, V., R. Trenholm, J. Wilson, and W. Richards. 2012. Assessing market and non-market costs of freshwater flooding due to climate change in the community of Fredericton, Eastern Canada. Climatic Change 110(1-2):347–372.

Lecce, S. A., and E. S. Kotecki. 2008. The 1999 Flood of the Century in Eastern North Carolina: Extraordinary Hydro-Meterological Event or Human-Induced Catastrophe? Physical Geography 29(2):101–120.

Lehner, B., C. R. Liermann, C. Revenga, C. Vorosmarty, B. Fekete, P. Crouzet, P. Doll, M. Endejan, K. Frenken, J. Magome, C. Nilsson, J. C. Robertson, R. Rodel, N. Sindorf, and D. Wisser. 2011a. Global Reservoir and Dam Database, Version 1 (GRanDv1): Reservoirs, Revision 01. http://sedac.ciesin.columbia.edu/data/set/grand-v1-reservoirs-rev01.

Lehner, B., C. R. Liermann, C. Revenga, C. Vörösmarty, B. Fekete, P. Crouzet, P. Döll, M. Endejan, K. Frenken, J. Magome, C. Nilsson, J. C. Robertson, R. Rödel, N. Sindorf, and D. Wisser. 2011b. High-resolution mapping of the world's reservoirs and dams for sustainable river-flow management. Frontiers in Ecology and the Environment 9(9):494–502.

Leopold, L. B. 1968. Hydrolog for Urban Land Planning - A guidebook on the hydrologic effects of urban land use. Pages 1–18. Washington, D. C.

Lovelace, J. T., B. R. Arthur, D. M. Coleman, D. Busse, J. D. Bartles, C. N. Strauser, S. L. Redington, R. J. Kopsky, R. J. Rapp, C. Engineering, and I. Global. 1994. Flood-Control Structures Have Their Place. Civil Engineering 64(12):27.

Ludy, J., and G. M. Kondolf. 2012. Flood risk perception in lands "protected" by 100-year levees. Natural Hazards 61(2):829–842.

Magilligan, F. J., and M. L. Stamp. 1997. Historical land-cover changes and hydrogeomorphic adjustment in a small Georgia watershed. Annals of the Association of American Geographers 87(4):614–635.

Milly, P. C. D., R. T. Wetherald, K. A. Dunne, and T. L. Delworth. 2002. Increasing risk of great floods in a changing climate. Nature 415(6871):514–7.

Moglen, G. E., and S. Kim. 2007. Are Threshold-Based Policies a Good Idea? Journal of the American Planning Association 73(2):161–172.

Mogollón, B., E. A. Frimpong, A. Hoegh, and P. L. Angermeier. (n.d.). Recent changes in stream flashiness and flooding in North Carolina and Virginia. In Prep.

Moss, T. 2007. Institutional drivers and constraints of floodplain restoration in Europe. International Journal of River Basin Management 5(2):121–130.

Nakagaki, N., C. V. Price, J. A. Falcone, K. J. Hitt, and B. C. Ruddy. 2007. Enhanced National Land Cover Data 1992 (NLCDe 92). Sacramento, CA. http://water.usgs.gov/GIS/metadata/usgswrd/XML/nlcde92.xml.

NCDENR. 2013. North Carolina Dam Inventory. http://portal.ncdenr.org/web/lr/dams.

Newbury, R., and M. Gaboury. 1993. Exploration and rehabilitation of hydraulic habitats in streams using principles of fluvial behaviour. Freshwater Biology 29:195–210.

NHD. 2013. Digital elevation model, flow accumulation and direction rasters. http://nhd.usgs.gov/data.html.

NRCS. 2012. Soil Survey Geographic database (SSURGO). http://websoilsurvey.nrcs.usda.gov/.

Olden, J. D., and N. L. Poff. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications 19(2):101–121.

Ostroff, A., D. Wieferich, A. Cooper, and D. Infante. 2012. National Anthropogenic Barrier Dataset. https://www.sciencebase.gov/catalog/item/get/512cf142e4b0855fde669828.

Öztürk, M., N. K. Copty, and A. K. Saysel. 2013. Modeling the Impact of Land Use Change on the Hydrology of a Rural Watershed. Journal of Hydrology 497:97–109.

Pasch, R. J., T. B. Kimberlain, and S. R. Stewart. 2014. Hurricane Floyd 7-17 September, 1999. http://www.nhc.noaa.gov/1999floyd.html.

Patterson, L. a., and M. W. Doyle. 2009. Assessing Effectiveness of National Flood Policy Through Spatiotemporal Monitoring of Socioeconomic Exposure. Journal of the American Water Resources Association 45(1):237–252.

Patterson, L. A., B. Lutz, and M. W. Doyle. 2012. Streamflow Changes in the South Atlantic, United States During the Mid- and Late 20th Century. Journal of the American Water Resources Association 48(6):1126–1138.

Petrow, T., and B. Merz. 2009. Trends in flood magnitude, frequency and seasonality in Germany in the period 1951–2002. Journal of Hydrology 371(1-4):129–141.

Pielke, R. A. 1999. Nine fallacies of floods. Climatic Change 42(2):413–438.

Pielke, R. A., M. W. Downton, and S. I. Group. 2000. Precipitation and Damaging Floods: Trends in the United States, 1932 – 97. Journal of Climate 13:3625–3637.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47(11):769–784.

Poff, N. L., B. P. Bledsoe, and C. O. Cuhaciyan. 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology 79(3-4):264–285.

Poff, N. L., and J. . Ward. 1989. Implications of Streamflow Variability and Predictability for Lotic Community Structure: A regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805–1817.

Price, C., N. Nakagaki, K. Hitt, and R. Clawges. 2003. Mining GIRAS: Improving on a national treasure of land use data. ESRI 2003 International User Conference. San Diego, California.

PRISM Climate Group. 2013. Precipitation grids. http://prism.oregonstate.edu.

Raper, R. L. 2005. Agricultural traffic impacts on soil. Journal of Terramechanics 42(3-4):259–280.

Rappaport, E. N. 2000. Loss of Life in the United States Associated with Recent Atlantic Tropical Cyclones. Bulletin of the American Meteorological Society 81(9):2065–2073.

Rice, K. C., and R. M. Hirsch. 2012. Spatial and temporal trends in runoff at long-term streamgages within and near the Chesapeake Bay Watershed: U.S. Geological Survey Scientific Investigations Report 2012–5151. Page 56 p.

Roy, A. H., S. J. Wenger, T. D. Fletcher, C. J. Walsh, A. R. Ladson, W. D. Shuster, H. W. Thurston, and R. R. Brown. 2008. Impediments and solutions to sustainable, watershed-scale urban stormwater management: lessons from Australia and the United States. Environmental Management 42(2):344–59.

Smith, J. A., M. L. Baeck, J. E. Morrison, P. Sturdevant-Rees, D. F. Turner-Gillespie, and P. D. Bates. 2002a. The Regional Hydrology of Extreme Floods in an Urbanizing Drainage Basin. Journal of Hydrometeorology 3:267–282.

Smith, S. V., W. H. Renwick, J. D. Bartley, and R. W. Buddemeier. 2002b. Distribution and significance of small, artificial water bodies across the United States landscape. The Science of the Total Environment 299(1-3):21–36.

Sordo-Ward, A., L. Garrote, F. Martín-Carrasco, and M. Dolores Bejarano. 2012. Extreme flood abatement in large dams with fixed-crest spillways. Journal of Hydrology 466-467:60–72.

Strecker, E. W., M. M. Quigley, B. R. Urbonas, J. E. Jones, and J. K. Clary. 2001. Determining urban storm water BMP effectiveness. Journal of Water Resources Planning and Management 127(127):144–149.

Sturdevant-Rees, P., J. A. Smith, J. Morrison, and M. L. Baeck. 2001. Tropical storms and the flood hydrology of the central Appalachians. Water Resources Research 37(8):2143–2168.

Thurston, H. W., H. C. Goddard, D. Szlag, and B. Lemberg. 2003. Controlling Storm-Water Runoff with Tradable Allowances for Impervious Surfaces. Journal of Water Resources Planning and Management (129):409–418.

Tollan, A. 2002. Land use change and floods: what do we need most, research or management? Water Science & Technology 45(8):183–190.

Tran, P., F. Marincioni, and R. Shaw. 2010. Catastrophic flood and forest cover change in the Huong river basin, central Viet Nam: a gap between common perceptions and facts. Journal of Environmental Management 91(11):2186–200.

USACE. 2011. Appendix B: Corps of Engineers Civil Works Program Statistics from the National Water Resources Challenges Facing the U.S. Army Corps of Engineers. Washington, DC.

USACE. 2012. National Inventory of Dams. http://geo.usace.army.mil/pgis/f?p=397:1:0.

USGS. 2012. National Hydrography Dataset Waterbodies. http://nhd.usgs.gov/data.html.

USGS. 2013. National Water Information System. http://nwis.waterdata.usgs.gov/nwis.

Wardrop, D. H., J. A. Bishop, M. Easterling, K. Hychka, W. Myers, G. P. Patil, C. Taillie, P. State, C. Wetlands, and E. Statistics. 2005. Use of landscape and land use parameters for classification and characterization of watersheds in the mid-Atlantic across five physiographic provinces. Environmental and Ecological Statistics 12:209–223.

Van Wie, J. B., J. C. Adam, and J. L. Ullman. 2013. Conservation tillage in dryland agriculture impacts watershed hydrology. Journal of Hydrology 483:26–38.

Wilby, R. L., and R. Keenan. 2012. Adapting to flood risk under climate change. Progress in Physical Geography 36(3):348–378.

Williams, E. S., and W. R. Wise. 2006. Hydrologic impacts of alternative approaches to storm water management and land development. Journal of the American Water Resources Association 6450(04108):443–455.

Wissmar, R. C., R. K. Timm, and M. G. Logsdon. 2004. Effects of changing forest and impervious land covers on discharge characteristics of watersheds. Environmental management 34(1):91–8.

Wossink, A., and B. Hunt. 2003. The economics of structural stormwater BMPs in North Carolina. Raleigh.

Yen, B. C. 1995. Hydraulics and effectiveness of levees for flood control. Hydrometeorology, Impacts and Management of Extreme Floods. U.S.-Italy Research Workshop, Perugia, Italy.

Young, W. 2014, March 27. N.C.'s growth concentrated in a small group of counties. News and Record. Winston-Salem, NC.

Tables and Figures

Table 1.1. Selected statistics to assess the temporal and spatial correlation structure of stream flow data. For the temporal correlation analysis, the Durbin-Watson (DW) statistic and P-value are used to assess data for the presence of no autocorrelation, where values near 2 suggest no autocorrelation. Three spatial correlation statistics are reported with the mean and 95% confidence interval (in parentheses); τ^2 is the variance and ϕ and σ^2 are parameters that control the covariance induced by the spatial structure. The parameters ϕ and σ^2 need to be considered jointly, as if ϕ is large relative to the distance between sites spatial autocorrelation is high; however, if σ^2 is small the contribution from the spatial structure is insignificant.

Flood Regime Metrics	**Temporal Correlation**		**Spatial Correlation**		
	DW Statistic	P-Value	τ^2	σ^2	ϕ
Count	2.27	0.75	0.043 (0.039, 0.048)	0.0001 (<0.0001, .0002)	23.7 (1.31, 48.4)
Duration	2.17	0.66	0.044 (0.04, 0.05)	0.0001 (<0.0001, 0.0002)	17.5 (0.2, 48.4)
Magnitude	1.92	0.42	0.51 (0.45, 0.56)	0.004 (<0.0001, 0.005)	25.0 (0.7, 48.9)

Table 1.2. Least square means and Bonferroni confidence intervals for nine flood return periods in four watershed types for three flood regime metrics. Means followed by the same letter are not significantly different. Gray text indicates no differences among the four watershed types were found.

Flood Return period	Count (number of floods)				Magnitude (m^3/s)				Duration (days)			
	Urban	Rural	Semi-Forested	Forested	Urban	Rural	Semi-Forested	Forested	Urban	Rural	Semi-Forested	Forested
80% Q1	2.23 ± 1.1 a	2.14 ± 1.1 a	2.19 ± 1.1 a	1.73 ± 1.1 a	20.5 ± 1.1 a	4.69 ± 1.1 b	3.57 ± 1.1 c	5.05 ± 1 b	0.04 ± 1.4 a	0.1 ± 1.3 b	0.16 ± 1.3 b	0.13 ± 1.4 b
Q1	5.76 ± 1.2 a	6.51 ± 1.2 a	6.65 ± 1.2 a	4.56 ± 1.2 a	107.7 ± 1.2 a	39.35 ± 1.2 b	34.05 ± 1.2 b	39.48 ± 1.2 b	0.25 ± 1.2 a	0.6 ± 1.2 b	0.86 ± 1.2 b	0.65 ± 1.3 b
Q1.5	0.57 ± 1.2 a	0.68 ± 1.2 a	0.53 ± 1.2 a	0.59 ± 1.1 a	25.64 ± 1.2 a	10.1 ± 1.2 b	7.52 ± 1.2 b	12.53 ± 1.2 b	0.44 ± 1.3 a	1.22 ± 1.3 b	1.54 ± 1.3 b	1.22 ± 1.3 b
Q2	1.34 ± 1.1 a	1.55 ± 1.1 a	1.26 ± 1.1 a	1.29 ± 1.1 a	37.56 ± 1.2 a	17.25 ± 1.3 b	12.41 ± 1.3 b	16.95 ± 1.2 b	0.47 ± 1.3 a	1.62 ± 1.3 b	1.97 ± 1.3 b	1.68 ± 1.3 b
Q5	0.10 ± 1.3 a	0.16 ± 1.3 a	0.15 ± 1.3 a	0.08 ± 1.3 a	9.34 ± 1.7 a	5.92 ± 1.8 a	6.32 ± 1.8 a	4.27 ± 1.6 a	0.07 ± 4.6 a	0.49 ± 7.1 b	0.27 ± 4.6 b	0.16 ± 4.6 c
Q10	0.06 ± 1.3 a	0.07 ± 1.4 a	0.05 ± 1.5 a	0.07 ± 1.3 a	7.04 ± 2 a	4.25 ± 2.3 a	2.84 ± 2.2 a	6.48 ± 2 a	0.05 ± 1.3 a	0.22 ± 1.9 b	0.15 ± 1.2 b	0.23 ± 1.3 b
Q20	0.01 ± 1.8 a	0.01 ± 2.8 a	0.02 ± 1.9 a	0.02 ± 1.8 a	1.71 ± 4.5 a	0.18 ± 6.2 a	1.55 ± 5.2 a	2.42 ± 5.7 a	0.00 ± 3 a	0.00 ± 4.6 a	0.02 ± 2.9 a	0.01 ± 2.6 a
Q50	0.01 ± 2.2 a	0.00 ± 2.2 b	0.00 ± 2.2 c	0.004 ± 3.3 a	0.00 ± 102 a	0 ± 2.1E+9 a	0 ± 4.9E+9 a	0 ± 3.E+4 a	0.00 ± 3.9 a	0.00 ± 0.0 a	0.00 ± 0.0 a	0.00 ± 5.5 a
≥Q100	0.01 ± 2.7 a	0.00 ± 0.0 a	0.01 ± 2 a	0.0 ± 0.0 a	0.00 ± 46.6 a	0.0 ± 46.6 b	0.00 ± 122 a	0.00 ± 46.6 c	0.00 ± 4.7 a	0.00 ± 4.7 b	0.03 ± 2.1 a	0.00 ± 4.7 c

Table 1.3. Summary of landscape features influencing flood regime metrics for four watershed types. Percentages derived for sandy and loamy soils, artificial water bodies and best management practices are based on watershed area.

Watershed Type	**Mean Annual Precipitation (mm)**	**% of Sandy and Loamy Soils**	**Mean Hill Slope (degrees)**	**% Artificial Water bodies**	**% Best Management Practices**
Forested	1308	71	25.61	0.09	0.04
Semi-Forested	1121	57	9.87	0.68	0.15
Rural	1085	63	9.58	0.78	0.12
Urban	1067	57	7.33	0.41	0.37

Table 1.4. Least square means and Bonferroni-adjusted confidence intervals for two flood regime metrics in four watershed types that differ in aerial extent of flow-regulating features, with low (≤ 0.02%), medium (0.16% to 0.22%), and high (0.43% to 2.04%) categories. Means with the same superscript are not significantly different.

Watershed Type	**Magnitude (m^3/s)**	**Duration (days)**
Low	359.7 ± 1.3^{a}	0.23 ± 1.6^{a}
Medium	296.4 ± 1.3^{ab}	0.44 ± 1.6^{a}
High	154.6 ± 1.2^{b}	0.55 ± 1.3^{a}
Forested	49.7 ± 1.1^{c}	1.49 ± 1.3^{b}

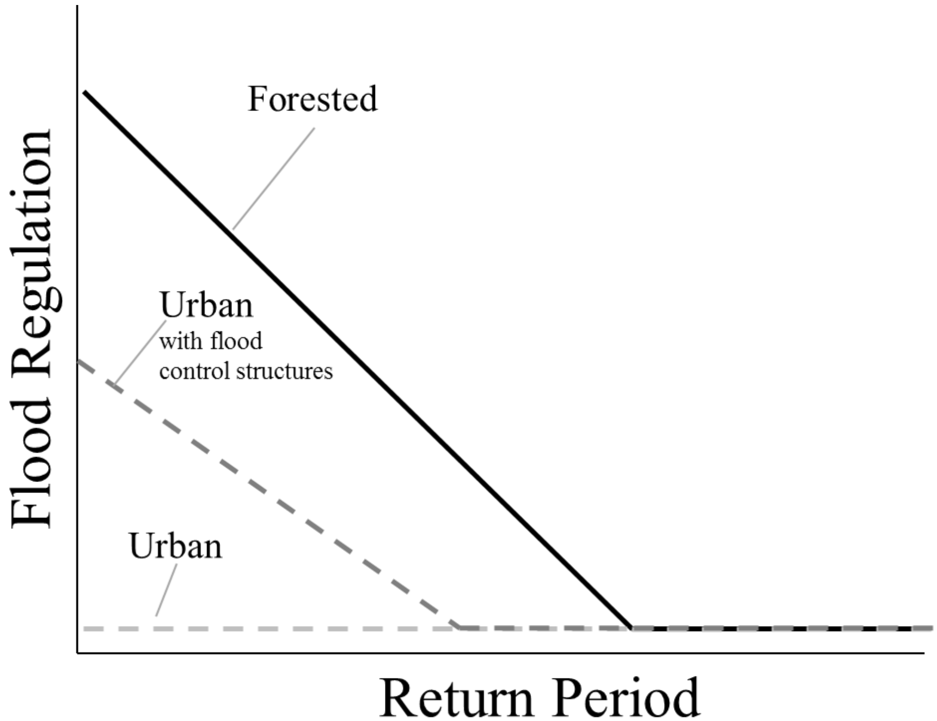

Figure 1.1. Conceptual relations between a landscape's capacity to regulate floods and return period for hypothetical urban, urban with flood control structures and forested watersheds. For large floods, all watersheds have little flood regulation capacity.

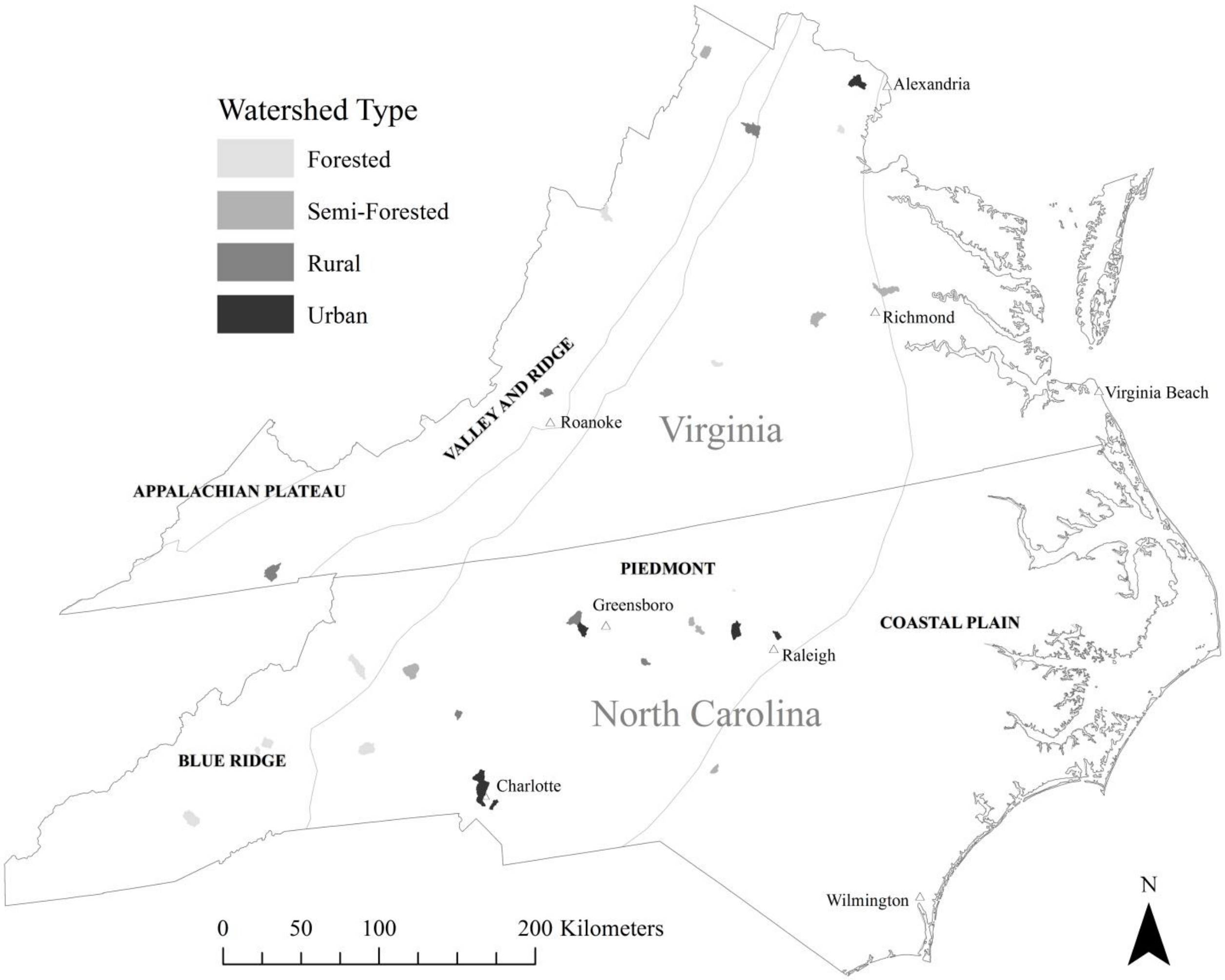

Figure 1.2. Map illustrating the type and location of the 31 study watersheds by physiographic province across North Carolina and Virginia.

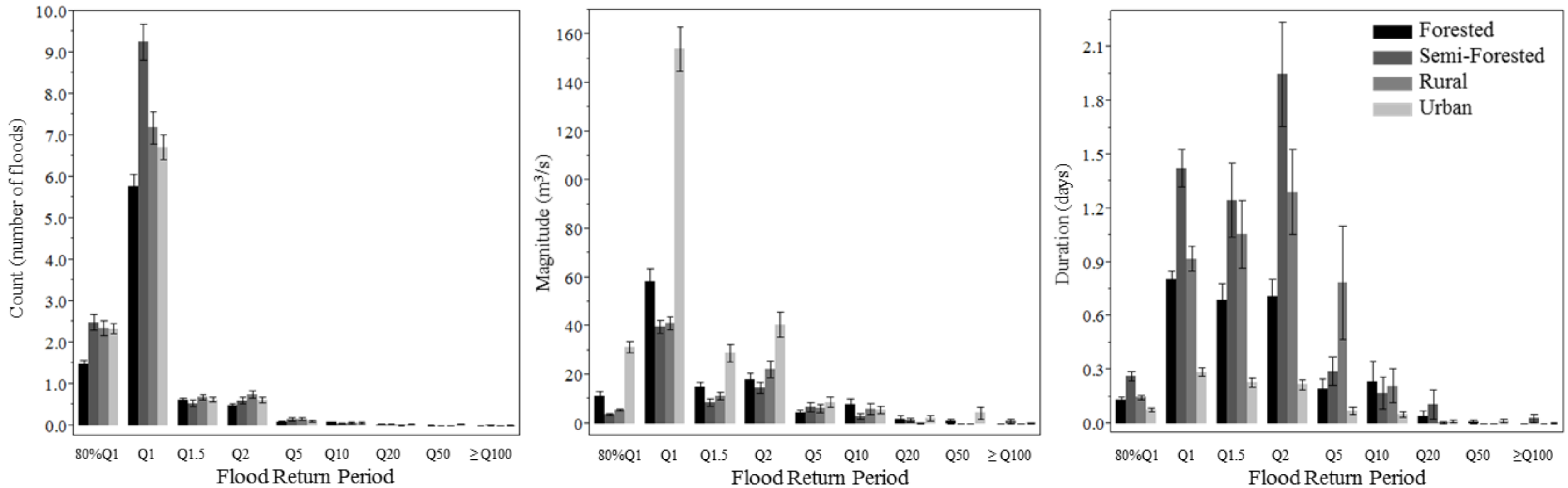

Figure 1.3. Mean flood count, magnitude, and duration, with standard error bars, for four watershed types across nine flood return periods, increasing from left to right on the x-axis (80%Q1 = 80% of a 1-year flood; Q1 = 1-year flood; Q1.5 = 1.5-year flood; Q2 = 2-year flood; Q5 = 5-year flood; Q10 = 10-year flood; Q20 = 20-year flood; Q50 = 50-year flood and ≥Q100 = greater than or equal to the 100-year flood).

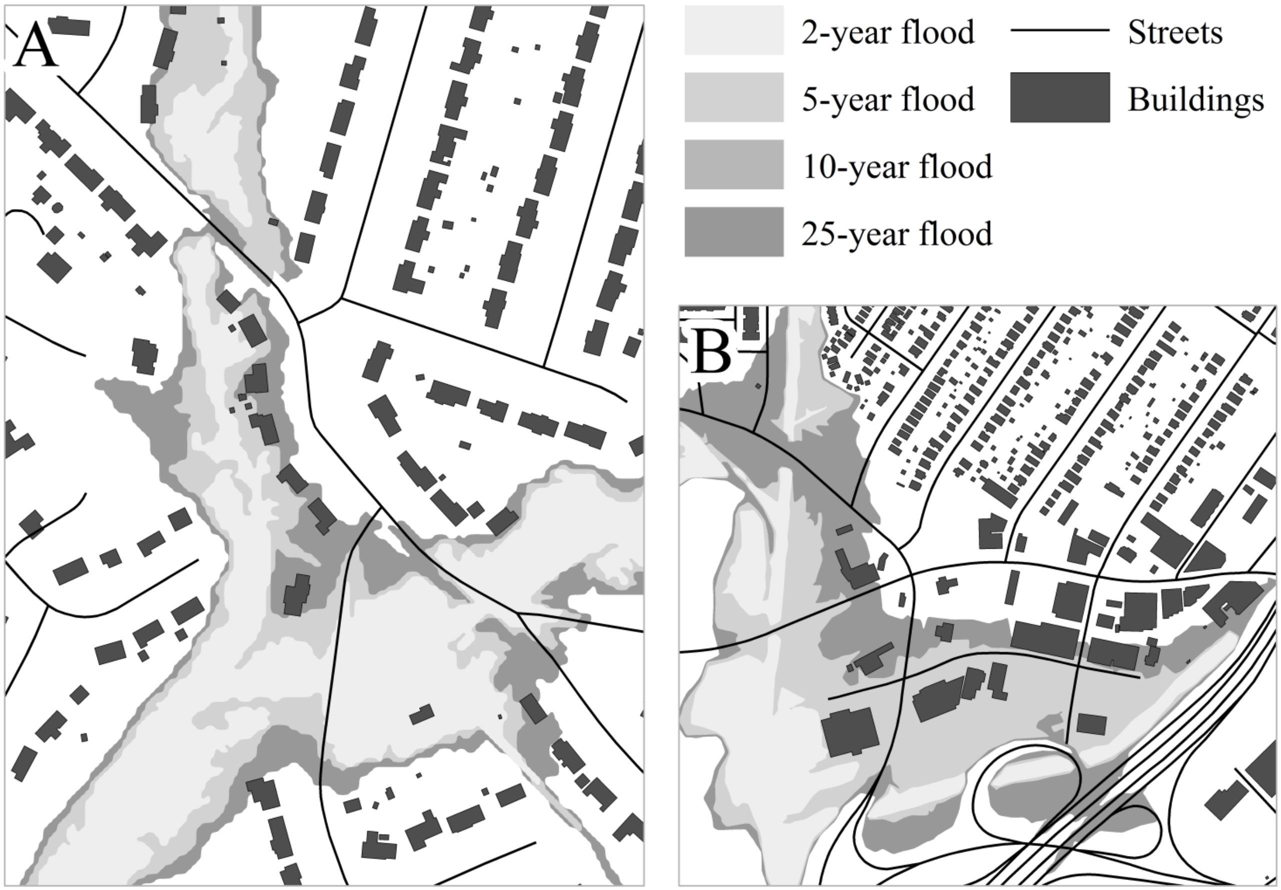

Figure 1.4. Maps illustrating the extent of inland flooding for the 2-, 5-, 10- and 25-year flood in two of the study watersheds in the city of Charlotte, North Carolina. Panel A shows flooding in a residential area (watershed 02146700) and Panel B shows flooding in a commercial area (watershed 02146300). These flooding zones were delimited from a joint effort between the Federal Emergency Management Agency and Charlotte-Mecklenburg Storm Water Services (FEMA and State of North Carolina 2014).

Chapter 2: Recent changes in stream flashiness and flooding in North Carolina and Virginia

Abstract

Climate and human activities impact the timing and quantity of streamflow and floods in different ways; while climate acts regionally, humans act locally. The southeastern US has undergone anthropogenic changes with the potential to increase (e.g., urbanization) and decrease (e.g., afforestation, reservoir construction) stream flashiness and floods. Insight into such change can inform researchers and managers on the efficacy of current strategies and policies to manage water resources. In this study, we examined long-term trends in precipitation, the count, magnitude and duration of small and large floods, and stream flashiness (via the flashiness index) for watersheds representing a gradient of human activity across North Carolina and Virginia from 1991 to 2013. In particular, we assessed the relative influence of land cover and flow-regulating features (e.g., best management practices and artificial water bodies) in stream flashiness of individual watersheds. We found that precipitation decreased, which coincided with decreasing trends in flood duration, count and magnitude for both large and small floods. In contrast, stream flashiness increased during the period of study. Upon closer examination, of 31 watersheds studied, 20 showed stable stream flashiness, while five increased and six decreased in flashiness. Urban watersheds were among those that increased or decreased in flashiness. Watersheds that increased in stream flashiness gained more urban cover, lost more forested cover and had fewer best management practices (e.g., detention ponds, flood control reservoirs) installed than urban watersheds that decreased in stream flashiness. Our results show that if land cover conversion does not include managing surface runoff, streams will become flashier. For watershed managers, flashiness index is a valuable and straightforward metric to identify changes in streamflow, and can be used to assess the efficacy of management interventions. Our study provides additional evidence against the assumption of stationarity (i.e., climate and land cover are unchanging through time) that pervades water policy and infrastructure design to date.

Introduction

Climate and landscapes are dynamic, with profound effects on the timing and quantity of streamflow (Ye et al. 2003, Arrigoni et al. 2010, Wang and Hejazi 2011). Understanding past and present trends in climate is imperative for water resource planning (Wilby and Keenan 2012); tracking landscape changes informs the assessment of human impacts and management interventions. Insight into human-induced change can inform water management strategies.

Understanding trends in streamflow and floods is important to society and aquatic environments. Flashy (i.e., rapidly changing) streamflow and recurrent small floods can exacerbate stream bank erosion, with undesirable consequences for transportation infrastructure (Dutton 2012), and water quality, particularly for downstream users (Brabec et al. 2002). In addition, flashy flows and recurrent floods jeopardize the ability of aquatic environments to provide suitable habitat for native biota (Paul and Meyer 2001) and conditions for fishing, wildlife watching and esthetically pleasing environments (Villamagna et al. 2014). Large floods are low-probability, high-consequence events that directly harm people, property and infrastructure; property damages from floods in the US average $2 billion per year (Water Science and Tecnology Board 2009).

Changes in precipitation typically drive changes in streamflow (Patterson et al. 2012), including flood regimes. Unlike national-scale trends, in the southeastern US precipitation and streamflow have decreased since the 1970s (Andreadis and Lettenmaier 2006, Patterson et al. 2012). Despite decreasing precipitation North Carolina and Virginia rank in the top 12 states for flood damage from 1983 to 1999 (Pielke et al. 2002). This pattern suggests there are non-climatic causes of floods, such as land cover change, construction of flow-regulating features, irrigation and groundwater pumping, sewer system design, and changes in soil management (Ye et al. 2003, Arrigoni et al. 2010, Wang and Hejazi 2011, Deasy et al. 2014) . Understanding which human activities and interventions contribute to flooding (or do not) can guide water resource management.

With decreasing precipitation trends in the southeastern US, Virginia (VA) and North Carolina (NC) provide an interesting case study regarding landscape changes in the past twenty years that presumably affect streamflow and floods in different ways. Increases in surface runoff due to expanding impervious surfaces from urban and suburban growth have been met with advances in stormwater management that reroute and retain surface runoff (Brown et al. 2005). The rate of reservoir construction was highest between 1980 and 1990, particularly in suburban areas, directly affecting streamflow and floods (Ignatius and Jones 2014). Conversion from farmlands to forests in the piedmont region has also impacted streamflow (Brown et al. 2005, Kim et al. 2014). The multiple changes across NC and VA landscapes suggest the response of streamflow and flooding to human-induced changes is spatially heterogeneous and warrants further investigation (Vogel et al. 2011).

Common metrics used to describe streamflow and flood regimes include the duration, magnitude and count of floods, and the flashiness index (Poff et al. 1997, Olden and Poff 2003, Baker et al. 2004). Flood duration is the mean amount of time a particular flood exceeds a certain flow threshold. Flood magnitude is the amount of discharge passing a fixed location, and flood count is the number of floods exceeding a certain flow threshold over a certain timeframe. Flood responses to landscape changes (anthropogenic or natural) are greatest for small floods, as large floods are driven primarily by climatic events (Mogollón et al. In Prep., Hawley and Bledsoe 2011). Flashiness index measures the rate of change in discharge, and has been used to assess anthropogenic disturbance over time (Baker et al. 2004).

Our goals in this study are to understand how streamflow and floods have changed from 1991 to 2013, and which human-induced changes in the landscape have impacted streamflow. We selected watersheds that represent a gradient of human activities with long-term hydrologic records and compiled changes in land cover and flow-regulating features (e.g., best management practices and artificial water bodies) for the same time period. Our objectives are to (1) examine trends in precipitation, flood count, duration and magnitude, and stream flashiness in non-coastal watersheds across North Carolina and Virginia, and (2) determine where and how stream flashiness has changed for each watershed and infer possible causes of that change based on landscape features. We conclude by assessing the utility of the flashiness index as a measure of human impact and as a guide for assessing landscape planning.

Methods

Study area

The 31 gaged watersheds selected in this study represent diverse landscapes across VA and NC. Most watersheds are in the Piedmont region, with others in the Coastal Plain, Valley and Ridge, and Blue Ridge regions; they represent a wide range of topographies and human settlement patterns (Appendix G; Figure 2.1). The watersheds used in this analysis are the same as those in Mogollon et al. (In prep.). We will refer to the four watershed types in Mogollon et al. (In prep): forested (89-98% forest), semi-forested (50-80% forest), rural (28-53% agriculture; 32-64% forest), and urban (40-100% urban).

We selected these watersheds based on size ($\leq 80 km^2$) and availability of instantaneous discharge record ($\geq$ 20 years). We selected instantaneous records, as opposed to daily averages, to capture the peaks of floods that might be obscured in daily average data (Rice and Hirsch 2012). We limited the analysis to small watersheds to highlight the effect of land cover on flooding, as the influence of anthropogenic disturbance on stream flow strongly decreases with increasing watershed size (Tollan 2002, Bloschl et al. 2007, Chang and Franczyk 2008, Petrow and Merz 2009).

Each watershed was spatially delineated in ArcGIS 10.1 (ESRI 2012) using the National Hydrography Database version 2, 30-m flow accumulation and direction layers; pour points were derived from the gage location provided in the United States Geological Survey's (USGS) flow records.

Trends in floods, stream flashiness and precipitation

Hydrologic and climatic data

Flood metrics (count, magnitude and duration) were derived using instantaneous discharge records, while the stream flashiness index was derived using mean daily records. Below, we will first describe how we derived the flood metrics and stream flashiness index, and then describe the compilation of precipitation data.

We downloaded peak and instantaneous discharge records from the USGS Water for the Nation Database (USGS 2013). Using annual peak discharge records, we used USGS's Peak FQ program to derive the discharge values for 1.005- (henceforth 1-year flood) and 5-year floods, and manually estimated 80% discharge of a 1-year flood. We defined the lowest flood as 80% of a 1-year flood, which is a flood that happens, on average, multiple times a year. This arbitrary threshold enabled us to assess small changes in the flood regime over time (Huang et al. 2008) at a flow below the bankfull discharge (Poff and Ward 1989; Bunn and Arthington 2002). Striving to represent trends in hydrology, as opposed to yearly variations in precipitation, we used at least 21 years of peak discharge records to derive the 1-year flood in PeakFQ.

Available periods of instantaneous records ranged from 20 to 28 years, but we limited the analysis to 1991-2013, as this period held a continuous record for most watersheds. We defined a complete water year discharge record as one having at least 300 days of data. We tabulated the count, magnitude and duration of floods (independent events); a flood could not last < 24 hours (Figure 2.2). Since we tabulated flood metrics by water year, all floods had to be assigned to a single water year. We had 29 events that spanned water years (1.6% of the records). We split these into independent floods by water year. For example, if a flood occurred from September 30th to October 1st, we counted it as two events (flood count = 1 for each water year), and summed the corresponding duration and magnitude for each event.

We distinguished between large (≥ 5-year flood) and small (< 5-year flood) floods, as the former are driven by climatic events wherein the landscape plays a negligible role, while the latter can be significantly altered by landscape features (Mogollon et al., In Prep; Hawley and Bledsoe 2011). Based on the discharge above 80% of a 1- and 5-year flood, we compiled the count, magnitude and duration of independent floods per water year for small and large floods respectively (Figure 2.2). We define flood count as the number of times per water year the discharge of an independent event equaled or exceeded the discharge of 80% of a 1- and 5- year flood. Flood magnitude is the amount of discharge above 80% of a 1- and 5-year flood. Flood duration is the length of time when discharge exceeds 80% of a 1- and 5-year flood, divided by the number of floods per water year.

Twelve watersheds had one to seven years of missing discharge data, for a total of 44 years missing (6% of the entire record). One watershed had seven consecutive years of missing data (gage failed in 2006), one had six consecutive years (1992-1997), three gages had four consecutive years, four gages had two or three consecutive years, and three had one year of missing data. We interpolated the missing years for a given watershed, instead of discarding these watersheds, by taking the average of the two years preceding and succeeding the missing values. We verified that the interpolated records fell within the range of values for watersheds that had discharge records for each water year.

To derive the stream flashiness index, we downloaded mean daily flows from the USGS Water for the Nation Database (USGS 2013) from 1991 to 2013 to match the previous flood metrics' time period. The flashiness index (also known as the Richards-Baker Index) is derived by summing the absolute difference of day-to-day changes in daily discharge over the sum of daily discharge volumes by water year. The index ranges from 0 to 2, where high values are associated with greater flashiness due to landscape features rather than precipitation trends (Baker et al. 2004).

Eight watersheds had from two to seven years of missing discharge data, for a total of 35 years missing or 5% of the entire record. One watershed had seven consecutive years of missing data (gage failed in 2006), one had six consecutive years, three had four consecutive years, and three had two or three consecutive years of missing data. We interpolated the missing years for a given watershed by taking the average of the two years preceding and succeeding the missing values.

We downloaded mean annual precipitation (mm) for 4-km pixel grids from the PRISM Climate Working Group (PRISM Climate Group 2013) for our study watersheds. In ArcGIS, we used the Zonal Statistics tool to derive yearly average precipitation estimates from 1991 to 2012. Due to unavailability of the 2013 precipitation record, we used the 2012 data to estimate 2013 precipitation.

Spatial and temporal autocorrelation

Before examining trends in hydrology and precipitation through time, we first assessed the temporal and spatial correlation structure of our hydrologic metrics and precipitation data. Not controlling for such correlation can reduce the effective number of samples available to test trends (Douglas et al. 2000). We visually inspected plots of temporal autocorrelation (Figure 2.3). When strong autocorrelation is present the covariance function between previous time periods will be large and consistently exceed the critical values for the null hypothesis of no autocorrelation. In our case, correlation attenuated quickly; information from the previous year provides little knowledge about the next period. Additionally, we used the Durbin –Watson (DW) statistic to assess the strength of autocorrelation; a null hypothesis of no autocorrelation (DW ≈ 2) was tested. DW statistics indicated no significant temporal autocorrelation among years of flow data (Figure 2.3, Table 2.1).

We tested the spatial correlation structure of the flow data by running a single set of Markov chain Monte Carlo (MCMC) simulations for the hydrology metrics and precipitation. We used the following spatial model,

$$\tilde{Y}_t = X_t\beta + Z\theta + \epsilon_t,$$

where $\tilde{Y}_t$ is a transformed flood regime metric, X_t is a matrix containing watershed characteristics and β is a vector of estimated coefficients relating those characteristics to the flood regime metrics. The matrix Z corresponds to a vector of random effects, θ, which quantifies the effects of individual watersheds. Finally, the error term, $\epsilon_t \sim N(0, \sigma^2 H(\phi) + \tau^2 I)$, where $N(\mu,\sigma^2)$ denotes a normal distribution with mean = μ and variance $=\sigma^2$ and I is an identity matrix. The spatial error structure is contained in $\sigma^2 H(\phi)$, where $H(\phi)$ is a spatial covariance matrix such as $exp(-d/\phi)$, where d = Euclidean distance between two points.

We assessed the effect of spatial correlation in the model using two parameters that control the covariance induced by the spatial structure, ϕ and σ^2, and compared them to τ^2 which is the variance. Hence, the spatial structure is controlled by the function of two parameters, $\sigma^2 H(\phi)$. Discerning the spatial structure requires considering them jointly as interpreting them individually can be misleading. For example, ϕ can be large relative to the distance between sites which means spatial correlation is high; however, if σ^2 is small the spatial covariance will be small. The appropriate comparison is between the covariance from the spatial structure, $\sigma^2 H(\phi)$, and the observation variance, τ^2. Given that $\tau^2 I$ is several orders of magnitude larger than the elements of $\sigma^2 H(\phi)$ (Table 2.1), we found no significant spatial correlation structure in the data. The lack of significant temporal or spatial autocorrelation in our data enabled us to treat the yearly records as independent observations.

Trend analysis

Our objective was to examine trends in precipitation, flood metrics (i.e., count, magnitude and duration of small and large floods), and stream flashiness (i.e., flashiness index) from 1991 to 2013. We used generalized linear mixed models (GLMMs) to test the direction and significance of change through time. GLMMs handle non-normal data and incorporate random effects (Bolker et al. 2008). The monotonic trend approach is the most appropriate for analyzing many stations at the same time (Hirsch et al. 1991). For precipitation, we ran a GLMM using gage identification number (ID) as a categorical random effect, and year as a continuous fixed effect. Prior to running the GLMM, we used the over-dispersion parameter (i.e., $\hat{c}$ = Pearson Chi-Square/df) of the intercept-only model to test the assumption of parameter homogeneity (Anderson 2008). Violating the parameter homogeneity assumption leads to a small sample variance which results in a false sense of precision (Anderson 2008). A small over-dispersion parameter ($\hat{c} \leq 1$) suggests the data suitably fits the selected distribution. Precipitation best fit a lognormal distribution and an identity link ($\hat{c} = 0.02$).

We ran GLMMs for duration, count and magnitude of small and large floods, and a GLMM for flashiness index. We used gage identification number (ID) as a categorical random effect, and year and mean annual precipitation as continuous fixed effects. For small floods, duration ($\hat{c} = 0.63$), count ($\hat{c} = 0.28$) and magnitude ($\hat{c} = 0.43$) followed lognormal distributions (link=identity). For large floods, duration ($\hat{c}$ = 0.44) and magnitude ($\hat{c} = 0.12$) followed lognormal distributions (link=identity), while flood count ($\hat{c}$ = 0.99) followed a negative binomial distribution (link=log). Flashiness Index followed a lognormal distribution ($\hat{c} = 0.04$; link=identity).

Changes in streamflow and landscape structure

Direction of changes in stream flashiness

We used flashiness index, as opposed to the other flood metrics, to examine the direction of streamflow change for each watershed because this index handles high interannual variability inherent in streamflow data, and is sensitive to landscape changes, but not to climatic trends (Baker et al. 2004). While the GLMM described previously used all the flashiness index records for each watershed per year in the same model, herein we assessed stream flashiness one watershed at a time using the non-parametric Kendall's tau statistical trend test. Kendall's tau is widely used in hydrology and is resistant to outliers (Helsel and Hirsch 2002). We derived a Kendall's tau value for each watershed using the flashiness index and water year. Kendall's tau ranges from -1 to 1, where values of -1 indicate a perfect negative association, 1 indicates a perfect positive association, and 0 indicate no association (Göktaş and İşçi 2011). Tau values of ≥ 0.7 and ≤ -0.7 are considered strong linear correlations (Helsel and Hirsch 2002).

Changes in the landscape

Different land management practices can make stream flashiness increase or decrease. To assess the causes of change, we derived information on land cover and flow-regulating features for the period of study. We recognize that other factors, such as irrigation, groundwater pumping, waste water flows, ditches and channels, sewer system, and soil management practices can also alter stream flows, but these data were largely unavailable for our study watersheds. We tested the effects of land cover and flow-regulating features on stream flashiness using Kendall's tau with a GLMM. We used the percentage change in urban cover and flow-regulating features from 1991 to 2013 as the model's fixed effects. Kendall's tau followed a normal distribution ($\hat{c}$ = 0.04; link=identity). Below we describe how we derived land cover and flow-regulating feature data.

Land cover

To characterize change in land cover for our study watersheds, we used the 1992-retrofit and 2011 National Land Cover Databases (NLCDs). The different classification methodologies used between NLCD 1992 (i.e. unsupervised) and more recent land cover databases make these two layers directly incomparable. However, the 1992-retrofit version is comparable to more recent land cover databases (Fry et al. 2009, Jin et al. 2013). We aggregated land cover classes from the NLCD into seven groups: water, forest, wetlands, urban, grasslands, crops, and barren (Table 2.2). In aggregating major land cover classes by watershed, we minimized the error produced in comparing the land cover databases (Price et al. 2003). To assess changes from 1992 to 2011, we subtracted the aggregated classes of 1992 from those of 2011, such that a gain in one particular land cover class was positive and a loss was negative.

Flow-regulating features

We assessed changes in flow-regulating features of best management practices (e.g. wet and dry ponds, flood control dams, bioretention areas, stormwater wetlands, sand filters, and infiltration devices; henceforth BMPs) and artificial water bodies (e.g. farm ponds, golf course ponds, water supply reservoirs;

henceforth AWBs) from 1991 to 2013. We included AWBs in our study because although AWBs are not designed to control floods, they store water during floods, and are ubiquitous across the landscape (Smith et al. 2002b, Downing et al. 2006, Ignatius and Jones 2014). We derived locations of BMPs and AWBs from multiple sources: the United States Army Corps of Engineers' National Inventory of Dams (USACE 2012), Global Reservoirs and Dams Database (Lehner et al. 2011a), National Anthropogenic Barrier Dataset (Ostroff et al. 2012), Federal Emergency Management Agency's National Flood Hazard Layer (FEMA 2013b), National Hydrograph Database on Waterbodies (USGS 2012), the North Carolina Department of Environment and Natural Resources' dam inventory (NCDENR 2013), Virginia Database (personal communication, Mark Bradford, Virginia Department of Conservation and Recreation), and BMP databases maintained by individual counties. We used Google Earth aerial imagery to verify the existence of, map the surface area of, and date BMPs and AWBs. We used surface area as a surrogate for volumetric capacity, as information on volumetric capacity was largely unavailable. We standardized flow-regulating features across the watersheds by dividing the BMP and AWB surface area by watershed drainage area for 1991 and 2013. We then subtracted the areal percentage of BMPs and AWBs present in 1991 from that of 2013, to estimate the change in these features during the study period.

Results

Trends in precipitation and hydrologic metrics

Precipitation and flood metrics mainly decreased during the period of study, while flashiness index increased (Table 2.3). Most flood metrics for, both small and large floods, decreased, but duration of small floods showed the only significant downward trend. The exception to the downward trends in flood metrics was the magnitude of large floods, which showed a positive, non-significant, trend. Flashiness index significantly increased during the period of study. The GLMMs showed that mean annual precipitation was significantly positively associated with flood count, magnitude and duration, and flashiness index. These results provide insight into the general trends for all study watersheds, regardless of their landscape evolution, topographic or geologic characteristics.

Changes in stream flashiness and landscape structure

Stream flashiness

In individually analyzing the study watersheds, we found that 17 decreased and 14 increased in stream flashiness based on Kendall's tau values during our period of study (Figure 2.4). The majority of watersheds (65%) had tau values between -0.19 and 0.19, which suggests little to no change in stream flashiness. Six watersheds decreased (tau between -0.2 and -0.36), and five watersheds increased in stream flashiness. Of the five watersheds that increased in flashiness, four had tau values between 0.25 and 0.35, while one watershed (02146211) had a tau value of 0.7, suggesting a substantial increase in stream flashiness. Watersheds showing substantial changes in stream flashiness (tau > 0.2 or < -0.2) were in NC, primarily in the urban centers of Charlotte, Raleigh, Durham, and Winston-Salem or in rural southwestern NC (Figure 2.4). Of the five watersheds that increased in stream flashiness, four were urban and one was forested. Three of the six watersheds that decreased in stream flashiness were urban, while the other three were forested. Changes in stream flashiness were spatially heterogeneous. The Charlotte (NC) area encompassed watersheds that increased and decreased in stream flashiness, which suggests influences of local activities and management interventions exceeded those of climatic factors.

Land cover

In the past 20 years, the study watersheds experienced losses of cropland and forested areas, and gains in urban, wetland and grassland areas (Table 2.4). Overall, the greatest loss was in croplands (21% per year), and the greatest gain was in grasslands (17% per year). North Carolina had a cropland loss and grassland gain similar to Virginia, but on average NC gained 2.6% more in urban cover per year and lost 3% more in forests compared to VA. Among the four watershed types (forested, semi-forested, rural and urban), the greatest increase in urban cover occurred in urban watersheds, followed by rural, semi-forested and forested watersheds. The greatest loss of cropland and greatest gain in grassland was in rural watersheds, followed by semi-forested, urban and forested watersheds. Urban watersheds lost the most forest, followed by forested, rural and semi-forested watersheds. The greatest gain in wetland was in forested watersheds, followed by semi-forested, rural and urban watersheds.

Flow-regulating features

On average, over three times more AWBs were constructed in our study watersheds than BMPs (Table 2.4). Overall, NC had more BMPs and AWBs than VA. BMPs were installed at a higher annual rate in urban watersheds, followed by semi-forested and rural watersheds. Forested watersheds had no BMPs constructed during the period of study. Semi-forested and rural watersheds had the greatest annual rate of AWB construction, followed by urban and forested watersheds. The majority of our BMPs were wet ponds (76%), followed by flood control dams (11%) and dry ponds (10%). The majority of our AWBs were reservoirs (60%), followed by farm ponds (34%) and golf course ponds (6%). Based on surface area, AWBs made up 76% of all flow-regulating features.

Stream flashiness and landscape structure

Watersheds that increased in stream flashiness (tau ≥ 0.2) gained, on average, 1.5 times more urban cover and lost 1.3 times more forest cover than watersheds that decreased in flashiness (tau ≤ -0.2) (Table 2.4; Figure 2.5). In addition, watersheds that decreased in stream flashiness had over twice as many BMPs constructed over the period of study as watersheds that increased in flashiness, despite the fact that watersheds with increasing flashiness had over twice as many AWBs as watersheds that decreased in flashiness. Watersheds with little to no change in stream flashiness (tau between -0.19 and 0.19) showed less than half as much forest-to-urban cover conversion and a greater increase in AWBs than watersheds that increased and decreased in flashiness. Our GLMM showed urban cover was significantly positively and BMPs were significantly negatively related to Kendall's tau flashiness index (Table 2.5). AWBs were non-significantly negatively related to Kendall's tau.

Discussion

Precipitation has significantly decreased since 1991 in selected watersheds across North Carolina and Virginia. This trend largely coincides with decreasing trends in the count, magnitude and duration of small and large floods, but not with the increasing trend in stream flashiness. A general upward trend in stream flashiness suggests that human activities have changed the timing and magnitude of discharge. A closer look at trends by individual watersheds suggests that the majority of watersheds changed little in flashiness. However, watersheds that increased in stream flashiness gained more urban cover and lost more forest cover, while implementing significantly fewer BMPs, than watersheds that decreased in stream flashiness. Below, we discuss our results, the implications of landscape structure changes in relation to stream flashiness findings, and the value of the flashiness index in providing evidence of human activities on streamflow.

Shifts in precipitation and flood regime

Since the 1970s, precipitation and streamflow have decreased in the southeastern United States (Patterson et al. 2012), which is consistent with our findings. As precipitation mainly drives patterns of streamflow and floods (Wilby and Keenan 2012), we expected the magnitude, duration and count of large floods to decrease through time. While we do find downward, non-significant trends in the count and duration of large floods, we find an increasing, but non-significant trend in the magnitude of large floods. This pattern of large floods might be due to an increase in the intensity of precipitation (Kunkel et al. 1999). Groisman et al. (2004) showed increases in heavy (above the upper 10 percentile) and very heavy (above the upper 0.3 percentile) precipitation events for the contiguous US during the twentieth century. Greater spatiotemporal resolution of precipitation records would be needed to examine the relation between extreme precipitation events and the increasing trend in the magnitude of large floods.

For small floods, the decreasing trends in magnitude, duration and count of floods coincide with precipitation patterns, although only flood duration showed a significant trend. We suspect the decreasing, non-significant trends in magnitude and count of small floods might be confounded by the wide range of human settlement patterns and degrees of disturbance in our study watersheds. A similar study assessing the causes of runoff in watersheds varying in land cover composition had inconclusive results (Frans et al. 2013). Further research is needed to understand whether the decreasing trend in small floods is due to decreasing precipitation, increases in flow-regulating features, or increases in land cover types (e.g., wetland or forest) that provide greater flood regulation.

In contrast to decreasing trends in precipitation and flood metrics, stream flashiness significantly increased over the 23-year record, which provides evidence of the influence that human activities have on streamflow. Our analyses of individual watersheds revealed that 55% of the study watersheds decreased in flashiness. Of the 45% of watersheds that increased in flashiness, one became significantly flashier (tau > 0.7). This shows that watersheds vary widely in stream flashiness, and how that flashiness responds to changes in climate and landscape structure.

Effects of landscape structure on flood regime

Trends in stream flashiness by watershed provided insight into the causes of change given land cover and flow-regulating feature patterns. Twenty of the 31 watersheds showed little to no change in stream flashiness, but were constant for different reasons. Four watersheds, three of which were forested underwent small changes in land cover (<5%) and in BMPs and AWBs (<0.1%) during the 23 years. Fourteen of the 20 watersheds underwent the greatest land cover change from cropland to grassland in addition to gaining a substantial number of AWBs. Of all the possible changes in land cover types that affect stream flashiness, the cropland to grassland change probably has the smallest hydrological impact. Despite a decrease in cropland in the eastern US from intensification of agriculture and conversion to urban areas (Brown et al. 2005; Kim et al. 2014), we suspect that most of the changes reported in this study from cropland to grassland are misclassifications between land cover databases (Wardlow and Egbert 2003).

Of the 20 watersheds that underwent small to no changes in flashiness, two watersheds (01654000 and 01673550) gained the most urban cover (> 10%). While watershed 01673550 constructed 0.6% in BMPs and AWBs, watershed 01654000 constructed < 0.1%. The former seems to have offset its development with a gain in flow-regulating features; the latter appears to have gained few flow-regulating features without significant changes in stream flashiness. Of all urban watersheds, watershed 01654000 in northern Virginia did not have a compiled database of flow-regulating features as other urban watersheds did. For this watershed, we relied on national-scale databases and our digitizing capacity from aerial photographs. However, we suspect that to show so little change in stream flashiness, this watershed must have had a greater number of flow-regulating features that were not captured in our records.

Of the watersheds that underwent changes in stream flashiness (tau ≤ -0.2 or ≥ 0.2), a pattern emerges in the trade-offs between urbanization, forest loss and construction of flow-regulating features. The urban watersheds that became flashier had greater expansion of urban cover, greater loss of forest cover and fewer BMPs constructed during the period of study. In contrast, the urban watersheds that became less flashy, had less expansion of urban cover and forest conversion, and twice the number of BMPs constructed during the period of study. Although twice as many AWBs were constructed in watersheds that increased in flashiness, these did little to affect stream flashiness. Our results show that BMPs are much more effective than AWBs in regulating floods (Table 2.5). The biggest difference between AWBs and BMPs is that the former are usually filled to near capacity (e.g., for water supply or recreation) while the latter are intended to retain water only after storms (Downing et al. 2006).

Our results are evidence against the assumption of climate and land cover stationarity, as we found that both precipitation and landscapes are dynamic, and changes are spatially heterogeneous. This assumption has been questioned since the 1960s in light of population growth and climate change (Mandelbrot and W 1968, Milly et al. 2008, Rootzén and Katz 2013), which has implications for water-related infrastructure and policy now. The future design of water-related infrastructure and policy will rely not only on past records, but also on models predicting trends in climate and hydrology, and scenarios that explore levels of uncertainty and risk (Lawrence et al. 2013, Rosner et al. 2014).

Utility of flood metrics for managers

The flashiness index and other flood metrics enable watershed researchers and managers to understand hydrologic change through time due to human activities and management interventions, unaffected by climatic trends (Baker et al. 2004, Dow 2007). While the application of the flashiness index is limited to gaged watersheds, it can be applied over a monthly or annual timeframe. The information derived from these analyses can be used to monitor management efficacy, particularly in urban watersheds that are susceptible to flashy flows due to widespread impervious surfaces. Furthermore, comparing watersheds of similar landscape characteristics can provide insight into the effectiveness of flood-control interventions so that learned lessons can be disseminated elsewhere. While the flashiness index summarizes changes in stream flashiness, it does not provide insight into the natural flood regime needed to sustain healthy aquatic environments. Whereas decreasing stream flashiness may be a desired outcome, monitoring its change has to be complemented with other hydrologic metrics, where comparison with more natural watersheds is imperative. For example, aquatic ecosystems have adapted to floods, where suitable floods, in terms of their count, magnitude and duration, are needed (Poff et al. 1997, Bunn and Arthington 2002). In managing to improve water quality, a manager might make better use of the flood count, magnitude and duration metrics rather than the stream flashiness index.

Conclusion

Landscape structure and climate are dynamic, with important consequences for flood regimes. A key implication of this dynamism is that effective flood management requires scientific knowledge of the relations among hydrology, landscape structure, and potential management actions. Our study shows decreasing trends in precipitation coincide with decreasing trends in the magnitude, duration and count of small and large floods, but increasing trends in stream flashiness. A closer look at stream flashiness trends by watershed reveals that the majority of watersheds throughout NC and VA have remained stable despite having different landscape configuration and degrees of disturbance. Watersheds that increased in stream flashiness generally gained more urban cover, lost more forested cover and implemented fewer BMPs than watersheds that decreased in stream flashiness. These results suggest that nearby watersheds, with similar climate, can be vastly different in stream flashiness due to variation in landscape structure and management. Our findings indicate that flood-control BMPs, integrated across a watershed, can substantially affect the flood regime. The flashiness index provides a useful metric for researchers and managers to understand alterations in streamflow that are not driven by changes in precipitation. This index can be used alongside other flow regime metrics to characterize a watershed's hydrologic patterns. Knowledge of relations among hydrologic processes and landscape management can inform policies to effectively manage water resources.

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References

Anderson, D. R. 2008. Model Based Inference in the Life Sciences: A Primer on Evidence. Springer, New York City.

Andreadis, K. M., and D. P. Lettenmaier. 2006. Trends in 20th century drought over the continental United States. Geophysical Research Letters 33(10):1–4.

Arrigoni, A. S., M. C. Greenwood, and J. N. Moore. 2010. Relative impact of anthropogenic modifications versus climate change on the natural flow regimes of rivers in the Northern Rocky Mountains, United States. Water Resources Research 46(12):1–16.

Baker, D. B., R. P. Richards, T. T. Loftus, and J. W. Kramer. 2004. A new flashiness index: characteristics and applications to midwestern rivers and streams. Journal of the American Water Resources Association 40(2):503–522.

Bloschl, G., S. Ardoin-Bardin, M. Bonell, M. Dorninger, D. Goodrich, D. Gutknecht, D. Matamoros, B. Merz, P. Shand, and J. Szolgay. 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes 1247(January):1241–1247.

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J.-S. White. 2008. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology & Evolution 24(3):127–135.

Brabec, E., S. Schulte, and P. L. Richards. 2002. Impervious surfaces and water quality: A review of current literature and its implications for watershed planning. Journal of Planning 16(4):499–514.

Brown, D. G., K. M. Johnson, T. R. Loveland, and D. M. Theobald. 2005. Rural land-use trends in the conterminous United States, 1950-2000. Ecological Applications 15(December):1851–1863.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30(4):492–507.

Chang, H., and J. Franczyk. 2008. Climate change, land use change, and floods: Towards an integrated assessment. Geography Compass 2(5):1549–1579.

Deasy, C., A. Titman, and J. N. Quinton. 2014. Measurement of flood peak effects as a result of soil and land management, with focus on experimental issues and scale. Journal of Environmental Management 132:304–312.

Douglas, E. M., R. M. Vogel, and C. N. Kroll. 2000. Trends in floods and low flows in the United States: impact of spatial correlation. Journal of Hydrology 240:90–105.

Dow, C. L. 2007. Assessing regional land-use/cover influences on New Jersey Pinelands streamflow through hydrograph analysis. Hydrological Processes 21:185–197.

Downing, J. A., Y. T. Prairie, J. J. Cole, C. M. Duarte, L. J. Tranvik, R. G. Striegl, W. H. McDowell, P. Kortelainen, N. F. Caraco, and J. M. Melack. 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnology and Oceanography 51(5):2388–2397.

Dutton, P. D. 2012. Factors influencing roadside erosion and in-stream geomorphic stability at road-stream crossings for selected watersheds, North Shore, Minnesota, USA. Master's Thesis Natural Resources Science and Management, University of Minnesota.

ESRI. 2012. ArcGIS Desktop. Environmental Systems Research Institute, Redlands, CA.

FEMA. 2013. Map Service Center. https://msc.fema.gov/webapp/wcs/stores/servlet/info?storeId=10001&catalogId=10001&langId=-1&content=productDFIRM&title=DFIRM Databases&parent=productInfo&parentTitle=Product Information.

Frans, C., E. Istanbulluoglu, V. Mishra, F. Munoz-Arriola, and D. P. Lettenmaier. 2013. Are climatic or land cover changes the dominant cause of runoff trends in the Upper Mississippi River Basin? Geophysical Research Letters 40(6):1104–1110.

Fry, J. A., M. J. Coan, C. G. Homer, D. K. Meyer, and J. D. Wickham. 2009. Completion of the National Land Cover Database (NLCD) 1992 – 2001 land cover change retrofit product. Page 18.

Göktaş, A., and Ö. İşçi. 2011. A comparison of the most commonly used measures of association for doubly ordered square contingency tables via simulation. Metodoloski zvezki 8(1):17–37.

Groisman, P. Y., R. W. Knight, T. R. Karl, D. R. Easterling, B. Sun, and J. H. Lawrimore. 2004. Contemporary Changes of the Hydrological Cycle over the Contiguous United States : Trends Derived from In Situ Observations. Journal of Hydrometeorology 5:64–85.

Hawley, R. J., and B. P. Bledsoe. 2011. How do flow peaks and durations change in suburbanizing semi-arid watersheds? A southern California case study. Journal of Hydrology 405(1-2):69–82.

Helsel, D. R., and R. M. Hirsch. 2002. Statistical Methods in Water Resources Techniques of Water Resources Investigations. Page 522 TechnometricsBook 4. U.S. Geological Survey.

Hirsch, R. M., R. B. Alexander, and R. A. Smith. 1991. Selection of methods for the detection and estimation of trends in water quality. Water Resources Research 27(5):803–813.

Huang, S., S. Cheng, J. Wen, and J. Lee. 2008. Identifying peak-imperviousness-recurrence relationships on a growing-impervious watershed, Taiwan. Journal of Hydrology 362(3-4):320–336.

Ignatius, A., and J. Jones. 2014. Small reservoir distribution, rate of construction, and uses in the upper and middle Chattahoochee basins of the Georgia piedmont, USA, 1950–2010. International Journal of Geo-Information 3(2):460–480.

Jin, S., L. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment 132:159–175.

Kim, Y., L. E. Band, and C. Song. 2014. The influence of forest regrowth on the stream discharge in the North Carolina piedmont watersheds. Journal of the American Water Resources Association 50(1):57–73.

Kunkel, K. E., K. Andsager, and D. R. Easterling. 1999. Long-term trends in extreme precipitation events over the conterminous United States and Canada. Journal of Climate 12:2515–2527.

Lawrence, J., A. Reisinger, B. Mullan, and B. Jackson. 2013. Exploring climate change uncertainties to support adaptive management of changing flood-risk. Environmental Science & Policy 33:133–142.

Lehner, B., C. R. Liermann, C. Revenga, C. Vorosmarty, B. Fekete, P. Crouzet, P. Doll, M. Endejan, K. Frenken, J. Magome, C. Nilsson, J. C. Robertson, R. Rodel, N. Sindorf, and D. Wisser. 2011. Global Reservoir and Dam Database, Version 1 (GRanDv1): Reservoirs, Revision 01. http://sedac.ciesin.columbia.edu/data/set/grand-v1-reservoirs-rev01.

Mandelbrot, B. B., and J. R. W. 1968. Noah, Joseph and operational hydrology. Water Resources Research 4(5):909–918.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change. Stationarity is dead: whither water management? Science 319(5863):573–4.

Mogollón, B., E. A. Frimpong, A. Hoegh, and P. L. Angermeier. 2014. What inland floods can be managed? Master's Thesis in Fish and Wildlife Conservation, Virginia Tech.

NCDENR. 2013. North Carolina Dam Inventory. http://portal.ncdenr.org/web/lr/dams.

Olden, J. D., and N. L. Poff. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications 19(2):101–121.

Ostroff, A., D. Wieferich, A. Cooper, and D. Infante. 2012. National Anthropogenic Barrier Dataset. https://www.sciencebase.gov/catalog/item/get/512cf142e4b0855fde669828.

Patterson, L. A., B. Lutz, and M. W. Doyle. 2012. Streamflow Changes in the South Atlantic, United States During the Mid- and Late 20th Century. Journal of the American Water Resources Association 48(6):1126–1138.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

Petrow, T., and B. Merz. 2009. Trends in flood magnitude, frequency and seasonality in Germany in the period 1951–2002. Journal of Hydrology 371(1-4):129–141.

Pielke, R. A., M. W. Downton, and J. Z. B. Miller. 2002. Flood damage in the United States, 1926 – 2000: A reanalysis of National Weather Service estimates. Pages 1–79. Boulder, Colorado.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime. Bioscience 47(11):769–784.

Poff, N. L., and J. . Ward. 1989. Implications of Streamflow Variability and Predictability for Lotic Community Structure: A regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805–1817.

Price, C., N. Nakagaki, K. Hitt, and R. Clawges. 2003. Mining GIRAS: Improving on a national treasure of land use data. ESRI 2003 International User Conference. San Diego, California.

PRISM Climate Group. 2013. Precipitation grids. http://prism.oregonstate.edu.

Rice, K. C., and R. M. Hirsch. 2012. Spatial and temporal trends in runoff at long-term streamgages within and near the Chesapeake Bay Watershed: U.S. Geological Survey Scientific Investigations Report 2012–5151. Page 56 p.

Rootzén, H., and R. W. Katz. 2013. Design life level: Quantifying risk in a changing climate. Water Resources Research 49(9):5964–5972.

Rosner, A., R. M. Vogel, and P. H. Kirshen. 2014. A risk-based approach to flood management decisions in a nonstationary world. Water Resources Research 50(3):1928–1942.

Smith, S. V., W. H. Renwick, J. D. Bartley, and R. W. Buddemeier. 2002. Distribution and significance of small, artificial water bodies across the United States landscape. The Science of the Total Environment 299(1-3):21–36.

Tollan, A. 2002. Land use change and floods: what do we need most, research or management? Water Science & Technology 45(8):183–190.

USACE. 2012. National Inventory of Dams. http://geo.usace.army.mil/pgis/f?p=397:1:0.

USGS. 2012. National Hydrography Dataset Waterbodies. http://nhd.usgs.gov/data.html.

USGS. 2013. National Water Information System. http://nwis.waterdata.usgs.gov/nwis.

Villamagna, A. M., B. Mogollón, and P. L. Angermeier. 2014. A multi-indicator framework for mapping cultural ecosystem services: The case of freshwater recreational fishing. Ecological Indicators 45:255–265.

Vogel, R. M., C. Yaindl, and M. Walter. 2011. Nonstationarity: Flood magnification and recurrence reduction factors in the United States. Journal of the American Water Resources Association 47(3):464–474.

Wang, D., and M. Hejazi. 2011. Quantifying the relative contribution of the climate and direct human impacts on mean annual streamflow in the contiguous United States. Water Resources Research 47(10):1–16.

Wardlow, B. D., and S. L. Egbert. 2003. A State-Level Comparative Analysis of the GAP and NLCD Land-Cover Data Sets. Photogrammetric Engineering and Remote Sensing 16(12):1387–1397.

Water Science and Tecnology Board. 2009. Urban stormwater management in the United States. National Academies Press, Washington, D. C.

Wilby, R. L., and R. Keenan. 2012. Adapting to flood risk under climate change. Progress in Physical Geography 36(3):348–378.

Ye, B., D. Yang, and D. L. Kane. 2003. Changes in Lena River streamflow hydrology: Human impacts versus natural variations. Water Resources Research 39(7):1–14.

Tables and Figures

Table 2.1. Selected statistics to assess the temporal and spatial correlation structure of precipitation and stream flow data. For the temporal correlation analysis, the Durbin-Watson (DW) statistic and P-value are used to assess data for the strength of autocorrelation. We tested a null hypothesis of no autocorrelation, where values near 2 suggest no autocorrelation. Three spatial correlation statistics are reported with the mean and 95% confidence interval (in parentheses); τ^2 is the variance and ϕ and σ^2 are parameters that control the covariance induced by the spatial structure. The parameters ϕ and σ^2 need to be considered jointly, as if ϕ is large relative to the distance between sites spatial autocorrelation is high; however, if σ^2 is small the contribution from the spatial structure is insignificant.

Metrics	**Temporal Correlation**		**Spatial Correlation**		
	DW Statistic	P-Value	τ^2	σ^2	ϕ
Count	2.27	0.75	0.043 (0.039, 0.048)	0.0001 (<0.0001, .0002)	23.7 (1.31, 48.4)
Duration	2.17	0.66	0.044 (0.04, 0.05)	0.0001 (<0.0001, 0.0002)	17.5 (0.2, 48.4)
Magnitude	1.92	0.42	0.51 (0.45, 0.56)	0.004 (<0.0001, 0.005)	25.0 (0.7, 48.9)
Flashiness Index	2.34	0.80	0.034 (0.031, 0.038)	0.0001 (<0.0001, 0.001)	17.0 (0.1, 48.1)
Precipitation	1.72	0.25	0.483 (0.435, 0.538)	0.00001 (<0.00001, .00002)	25.6 (2.05, 48.8)

Table 2.2. Reclassified land cover types based on the classification of the National Land Cover Database (NLCD) for 1992 and 2011.

Land Cover Category	NLCD 1992-Retrofit		NLCD 2011	
	ID	Cover Type	ID	Cover Type
Water	11	Open Water	11	Open Water
Forest	41	Deciduous Forest	41	Deciduous Forest
	42	Evergreen Forest	42	Evergreen Forest
	43	Mixed Forest	43	Mixed Forest
Wetlands	91	Woody Wetlands	90	Woody Wetlands
	92	Emergent Herbaceous Wetlands	95	Emergent Herbaceous Wetlands
			52	Shrub, Scrub
Urban	21	Low Intensity Residential	21	Open Space Developed
	22	High Intensity Residential	22	Low Intensity Developed
	23	Commercial, Industrial, Transportation	23	Medium Intensity Developed
	85	Urban, Recreational Grasses	24	High Intensity Developed
Grasslands	81	Pasture/Hay	71	Grassland/Herbaceous
			81	Pasture/Hay
Crops	82	Row Crops	82	Cultivated Crops
Barren	31	Bare Rocks, Sand, Clay	31	Barren (Rocks, Sand, Clay)
	32	Quarries, Strip Mines, Gravel Pits		
	33	Transitional Barren		

Table 2.3. Summary of generalized linear mixed models (mean ± one standard error) to explain variation in precipitation, flood count, flood magnitude, flood duration, and flashiness index across 31 watersheds. Gray font indicates the bounds around the mean include zero, and black font indicates the bounds around the mean do not include zero.

Response		Intercept	Year	Precipitation
Precipitation		19.03 ± 1.7	-0.006 ± 0.0009	
Flood Count				
	Small	0.2 ± 10.2	-0.002 ± 0.005	2.3 ± 0.3
	Large	-10.9 ± 39.8	-0.004 ± 0.02	5.5 ± 1.1
Flood Magnitude				
	Small	-0.6 ± 8.7	-0.002 ± 0.004	3.1 ± 0.4
	Large	0.5 ± 16.5	0.0004 ± 0.008	0.93 ± 0.6
Flood Duration				
	Small	15.2 ± 10.6	-0.01 ± 0.005	2.8 ± 0.5
	Large	2.5 ± 38.3	-0.006 ± 0.018	3.5 ± 1
Flashiness Index		-6.3 ± 2.5	0.0014 ± 0.001	0.9 ± 0.1

Table 2.4. Summary of the mean annual percentage changes in land cover and flow-regulating (Reg.) features for study watersheds (Total), by state, by watershed type and by stream flashiness trend that occurred from 1991 to 2013. Acronyms in the table include: North Carolina (NC), Virginia (VA), best management practices (BMP), and artificial water bodies (AWB).

	Land Cover							Flow-Reg. Features	
	Water	Urban	Barren	Forests	Grasslands	Crops	Wetlands	BMP	AWB
Total	-0.57	7.13	-0.06	-6.58	16.8	-21	4.20	0.124	0.461
State									
NC	0.01	4.86	0.10	-4.69	8.09	-10.7	2.34	0.110	0.269
VA	-0.58	2.27	-0.16	-1.89	8.75	-10.3	1.86	0.014	0.192
Watershed Type									
Forested	-0.01	0.28	-0.08	-1.36	0.34	-0.92	1.75	0.001	0.015
Semi-Forested	-0.11	0.81	-0.07	-0.90	3.74	-5.14	1.67	0.011	0.173
Rural	-0.28	1.15	0.00	-0.60	11.5	-12.2	0.45	0.003	0.179
Urban	-0.16	4.89	0.09	-3.72	1.31	-2.74	0.33	0.109	0.094
Stream Flashiness									
Decrease	0.00	0.30	0.01	-0.29	0.09	-0.22	0.11	0.012	0.005
No Change	-0.03	0.15	-0.01	-0.15	0.79	-0.92	0.17	0.001	0.019
Increase	0.00	0.46	0.01	-0.38	0.12	-0.23	0.03	0.006	0.012

Table 2.5. Summary of the generalized linear model testing the effect of the percentage change in urban cover, best management practices (BMPs) and artificial water bodies (AWBs) between 1991 and 2013 on Kendall's tau stream flashiness index. Gray font indicates the bounds around the mean include zero, and black font indicates the bounds around the mean do not include zero.

Parameter Estimate	**Mean ± SD**
Intercept	-0.02 ± 0.05
Urban	0.02 ± 0.01
BMPs	-0.53 ± 0.26
AWBs	-0.02 ± 0.09

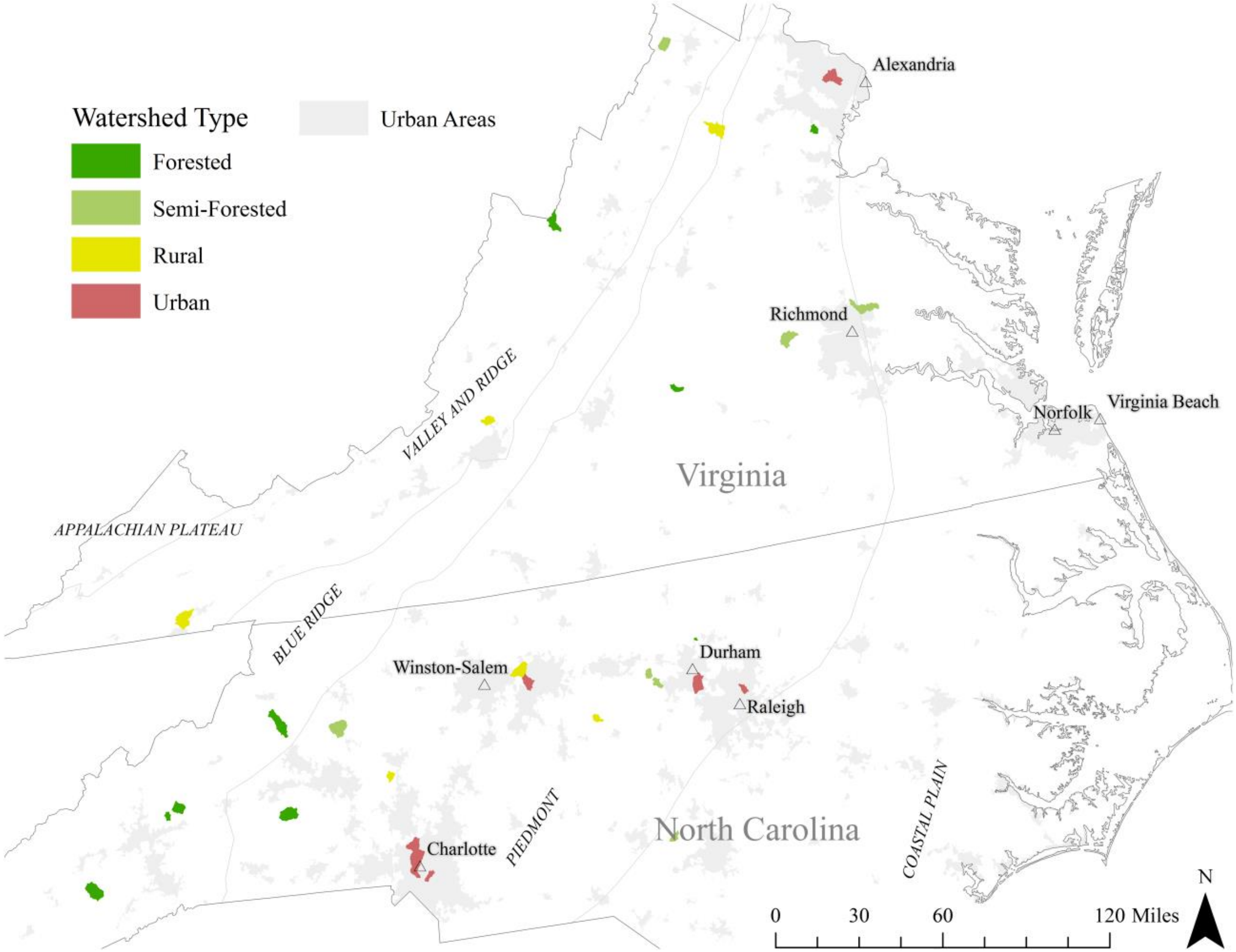

Figure 2.1. Map illustrating the type (i.e., forested, semi-forested, rural and urban) and location of the 31 study watersheds by physiographic province across North Carolina and Virginia.

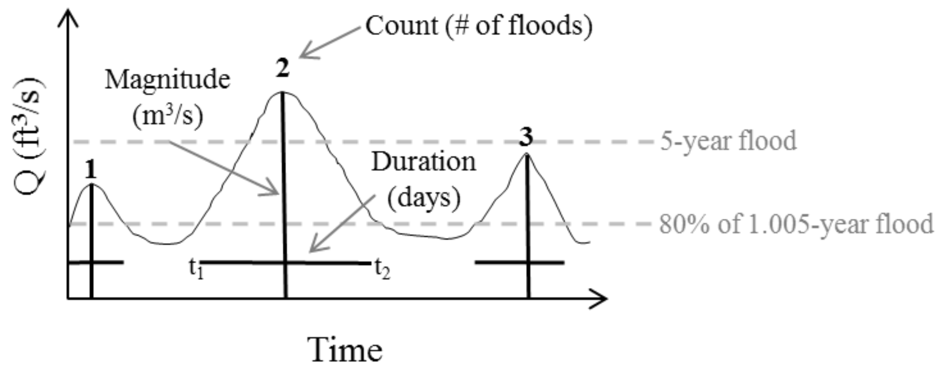

Figure 2.2. Schematic definition of flood magnitude, flood count and flood duration. A given discharge (Q) was considered a flood if it exceeded 80% of the discharge that recurs at a 1.005-year interval (henceforth referred to as 1-year flood). Small floods are those discharge events surpassing the 80% of a 1-year flood and less than the 5-year flood. Large floods are those discharge events greater than or equal to a 5-year flood.

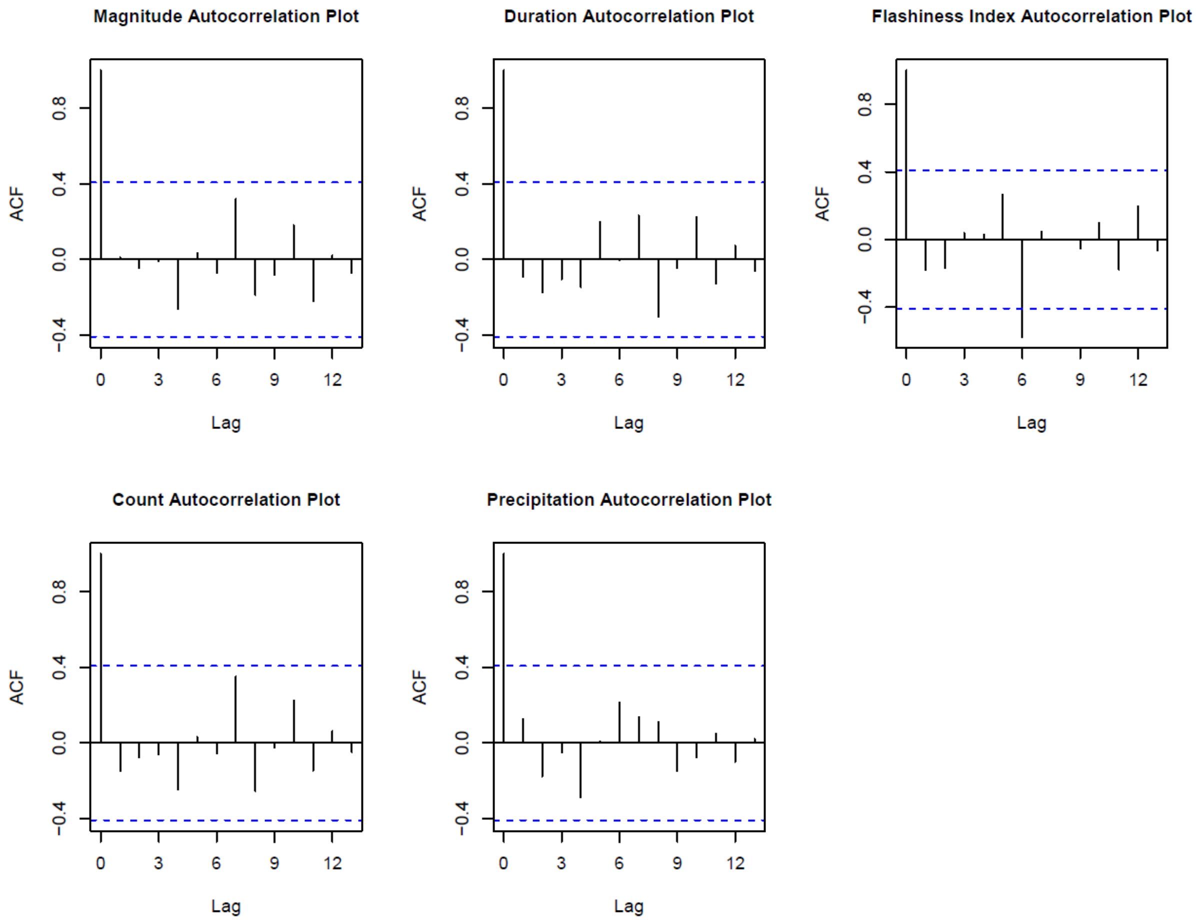

Figure 2.3. Temporal autocorrelation plots for precipitation, flood magnitude, flood count, flood duration and flashiness index. When autocorrelation is strong the autocorrelation covariance function (ACF) between previous time periods with lag, k, is large and consistently outside the dashed bands, which approximate confidence interval for the null hypothesis of no correlation. In our case the correlations attenuate quickly; information from one year provides little knowledge about the next year. Note that while the lag 6 correlation exceeds the bounds for the flashiness index, the DW statistic suggests no autocorrelation. Furthermore, as a lag 6 correlation is not physically meaningful (e.g., flashiness 6 years ago does not inform the current year); this pattern is largely a result of the ACF plot not inherently controlling for multiple comparisons.

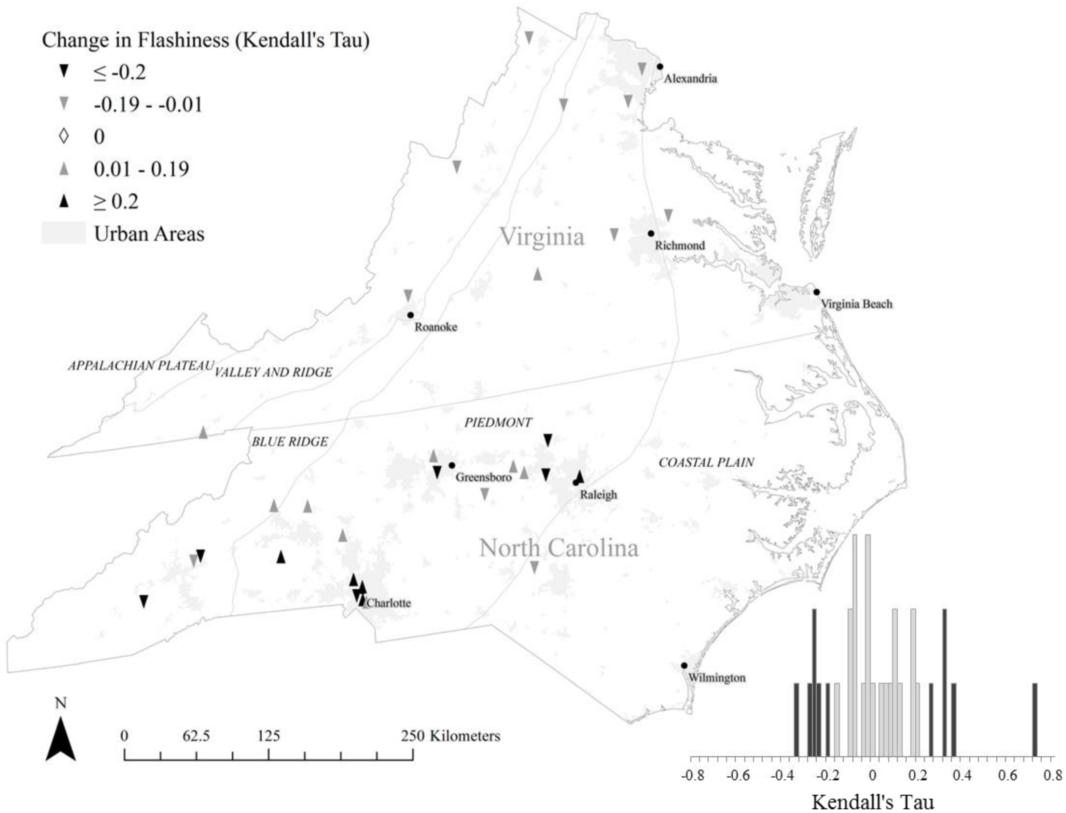

Figure 2.4. Map and histogram illustrating the direction of change in stream flashiness according to Kendall's tau tests for each watershed over the 23-year record (1991-2013) in North Carolina and Virginia. Downward arrows suggest a decreasing trend (tau < 0) and upward arrows suggest an increasing trend (tau > 0) in stream flashiness. Black arrows and histogram bars suggest an increase (tau ≥ 0.2) or decrease (tau ≤ -0.2) in stream flashiness, while gray arrows and histogram bars suggest little to no change in stream flashiness.

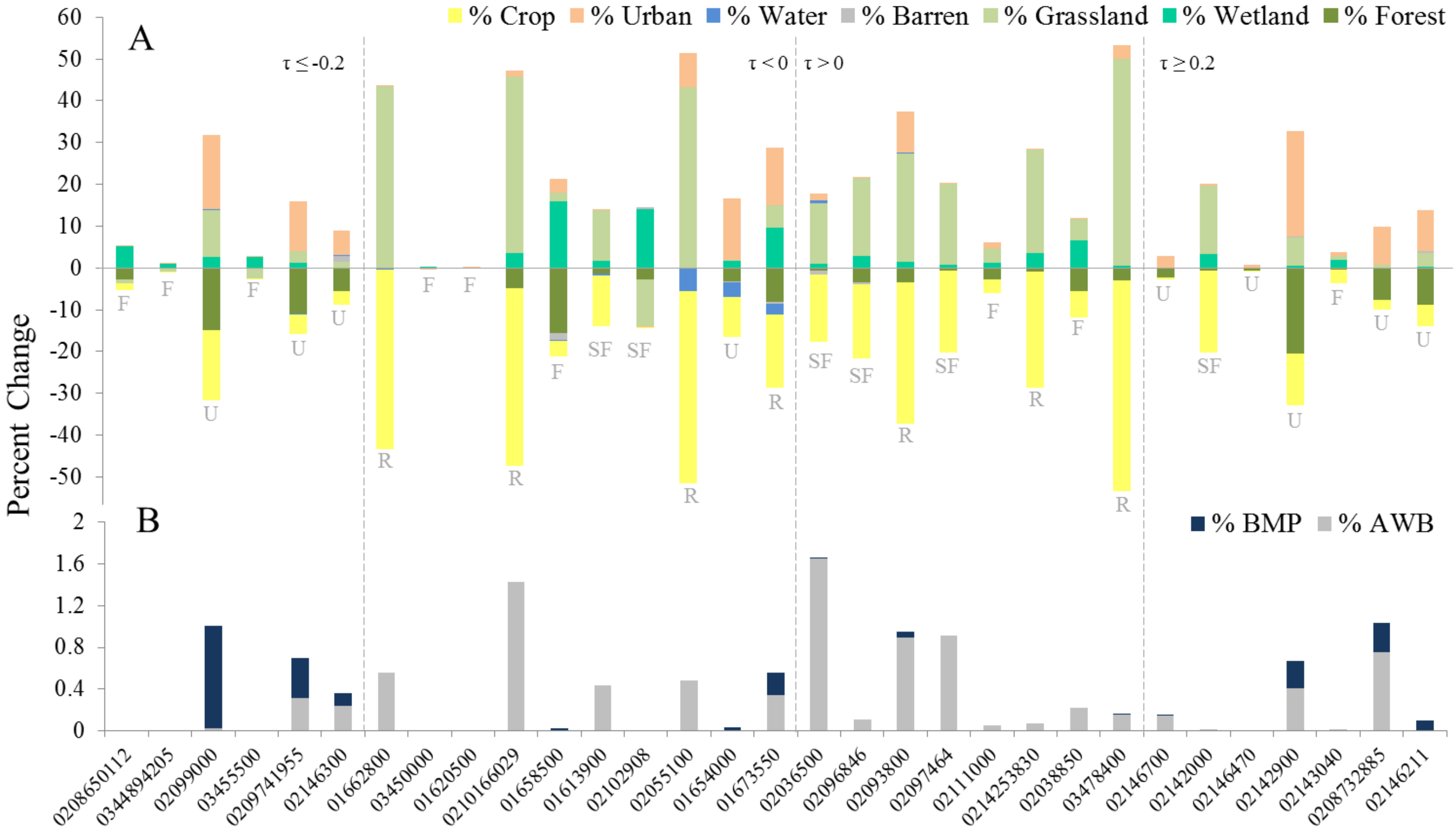

Figure 2.5. Percentage change in land cover (Panel A) and flow-regulating features (Panel B) from 1991 to 2013 for the 31 study watersheds identified across the bottom of the figure with their gage ID. The watersheds are ordered in increasing Kendall's tau value from left to right; the left-most watersheds had decreasing stream flashiness trends and the right-most watersheds had increasing stream flashiness trends. For land cover, a positive percentage change implies a gain of a land cover type and a negative percentage change implies a loss of a land cover type. The letter underneath each bar in Panel A refers to watershed type, where F = forested, SF = semi-forested, R = rural and U = urban. For the flow regulating features, BMP refers to best management practices and AWB refers to artificial water bodies.

Chapter 3: Mapping technological and biophysical capacities of watersheds to regulate floods

Abstract

Flood regulation is a widely valued and studied service provided by watersheds. Flood regulation benefits people directly by decreasing the socio-economic costs of flooding and indirectly by its positive impacts on cultural (e.g., fishing) and provisioning (e.g., water supply) ecosystem services. Like other regulating ecosystem services (e.g., pollination, water purification), flood regulation is often enhanced or replaced by technology, especially in urban landscapes, but the relative efficacy of natural versus technological features in controlling floods has scarcely been examined. In an effort to map flood regulation capacity for selected rivers in Virginia and North Carolina, we: (1) used long-term flood records to assess relative effects of technological and biophysical indicators that potentially regulate floods on flood magnitude and duration, (2) compared the widely used runoff curve number (RCN) approach to an alternative approach for assessing the biophysical capacity to regulate floods, and (3) mapped technological and biophysical flood regulation capacities based on indicator importance-values derived for flood magnitude and duration. We found that biophysical and technological capacities were negatively related to flood magnitude and positively related to flood duration. That is, watersheds with high capacities regulated floods. Further, the RCN approach yielded results opposite that expected, possibly because it confounds soil and land cover processes, while our alternative approach coherently separates these processes. Mapping biophysical and technological capacities revealed great differences among watersheds. Our study improves on previous mapping of flood regulation by (1) incorporating technological capacity, (2) providing high spatial resolution (i.e., 10-m pixel) maps of watershed capacities, and (3) deriving importance-values for selected landscape indicators. By accounting for technology that enhances or replaces natural flood regulation, our approach enables watershed managers to make more informed choices in their flood-control investments.

Introduction

Regulating ecosystem services are in global decline (Millennium Ecosystem Assessment 2005) but in many cases the services formerly provided by nature have been enhanced or replaced by technology (Fitter 2013). For example, services once provided by wild pollinators are now provided by commercial pollinator colonies (Sumner and Boriss 2006). In response to deteriorating water quality from intensive use and land cover change (Postel and Thompson 2005, Fiquepron et al. 2013), the water purification service previously provided by natural ecosystems has been replaced and enhanced by water treatment processes (Kraus-Elsin et al. 2010, Chowdhury et al. 2013). Unfortunately, studies quantifying and mapping regulating services rarely acknowledge the role of technology, despite its prevalence in enhancing or replacing diminished ecosystem services (Reyers et al. 2013). Excluding the technological enhancements of a service potentially omits important functions of the landscape and obscures the role of management in altering the provision and quality of services. Full understanding of the capacity of an ecosystem to provide a service requires integrating all natural and technological characteristics germane to that service.

The capacity of a watershed to regulate stream flow and floods is widely valued and studied (Posthumus et al. 2010, Eigenbrod et al. 2011, Schulp et al. 2012, Laterra et al. 2012, Jackson et al. 2013). Some studies assume natural ecosystems can reduce and moderate extreme floods, thereby reducing damage to people, property and infrastructure (Chan et al. 2006, Ennaanay et al. 2011, Nedkov and Burkhard 2012, Logsdon and Chaubey 2013). However, many studies show that landscapes play a negligible role in ameliorating extreme floods, which are usually driven by high precipitation (Sullivan et al. 2004, Chang and Franczyk 2008, Lecce and Kotecki 2008), but can regulate small floods (Findlay and Taylor 2006, Huang et al. 2008, Hawley and Bledsoe 2011, Mogollón et al. 2014). Regulation of recurrent small floods is important because they can facilitate stream bank erosion (Dutton 2012), impair water quality (Brabec et al. 2002), and incur substantial socioeconomic costs (Green and Penning-Rowsell 1989, Lantz et al. 2012). Small-flood regimes also influence the biotic health of streams (Paul and Meyer 2001), which, in turn influences cultural benefits such as fishing, wildlife watching, and esthetically pleasing environments (Villamagna et al. 2014).

As watersheds urbanize, the landscape characteristics that formerly provided flood regulation are often replaced or enhanced by technological features such as flood control dams, wet and dry ponds, bioretention areas, sand filters, and constructed wetlands. These technologies are usually intended to lower the peak flows and increase the duration of floods (Davis 2008, Burns et al. 2012). However, this approach to flood management has been questioned in light of high installation and maintenance costs (Thurston et al. 2003), legislative and institutional barriers (Roy et al. 2008), and continued impacts on aquatic environments (Booth et al. 2002). Despite these shortcomings, technological features are common in urban landscapes (Smith et al. 2002b, Downing et al. 2006, Ignatius and Jones 2014) and can significantly alter flows and regulate small floods (Goff and Gentry 2006, Su et al. 2010).

Flood regulation as an ecosystem service is commonly studied through mapping (Nedkov and Burkhard 2012, Radford and James 2012, Schulp et al. 2012, Laterra et al. 2012, Jackson et al. 2013, Koschke et al. 2013). Spatially assessing flood regulation is particularly useful because the benefits are spatially dependent (i.e., directly conveyed downstream). Maps can illustrate the spatial distribution of service capacity (i.e., where regulation occurs), which can be compared to the demand for the service (i.e., where regulation is needed) (Nedkov and Burkhard 2012, Villamagna et al. 2013). Most previous studies of flood regulation map only biophysical features (e.g., soils, vegetation, land use/cover) (Posthumus et al. 2010, Nedkov and Burkhard 2012, Schulp et al. 2012), but ignore common technological features germane to flood control (Smith et al. 2002b, Downing et al. 2006, Ignatius and Jones 2014). Omitting technological features significantly underestimates flood regulation capacity, which can impair the ability of watershed managers to make cost-effective choices regarding how to meet flood control objectives. Currently, FEMA's 1% flood (e.g., 100-year) floodplain maps guide managers to prevent and reduce the loss of lives and property, and maintain a functional floodplain (Tingle 1999). However, spatially explicit maps of biophysical and technological capacities provide a watershed-scale assessment, not limited to the 1-percent flood's floodplain, and can better inform flood-control investments.

The relative importance of biophysical and technological indicators in regulating floods varies among watersheds (Jencso et al. 2009, Eng et al. 2013a) but this variation is poorly understood. For example, watershed slope might be more influential in a mountainous area than in a flatter landscape where vegetation might play a bigger role (Barron et al. 2011). However, most studies of flood regulation have not differentially weighted mapped indicators to reflect their relative importance (Nedkov and Burkhard 2012, Villamagna et al. 2014). Analyzing long-term flow records by watershed provides insight into the relative importance of landscape indicators in regulating floods. Each indicator's relative importance is place-specific, reflecting topography, climate and land cover. Deriving importance-values is an objective methodology to rank indicators useful in mapping services. While this methodology fails to inform how much of the long-term discharge records are explained by each indicator, it provides a relative measure of importance among indicators, such that managers can focus efforts of flood-regulation on manageable, higher ranking indicators.

A landscape's ability to control surface runoff is an important part of its flood regulation capacity. A common method to map runoff potential in flood regulation studies (Ennaanay et al. 2011, Schulp et al. 2012, Laterra et al. 2012, Simonit and Perrings 2013, Koschke et al. 2013) is based on the Natural Resources Conservation Service's runoff curve number (RCN), a dimensionless estimate of runoff volume derived from data on rainfall infiltration, evapotranspiration, and surface storage by soil and vegetation (Rallison 1980). RCN is widely used because of its simplicity and general acceptance (Ponce and Hawkins 1996), but a major shortcoming in using RCN to map runoff potential is that it was not developed to predict runoff (Garen and Moore 2005, Ogden and Stallard 2013). Applying the RCN to each pixel in a GIS layer is a mis-use of the approach because landscape processes (e.g., infiltration, retention, evapotranspiration) governing water runoff are confounded, as is their variability through space and time (Garen and Moore 2005). Erroneously portraying the spatial distribution of water-related ecosystem services can mislead decision-making processes. Examining the validity of using RCN in assessments of flood regulation warrants a comparison to an alternative approach that distinguishes among the three main landscape processes that determine overland runoff: infiltration, evapotranspiration and retention.

The goal of this study is to use river flooding records to estimate the relative importance of selected landscape features in regulating floods, and then use those to map biophysical and technological capacities of watersheds to regulate floods. We focus on urban areas, as they have the most altered flooding patterns and the greatest extent of flow-regulating features (Poff et al. 2006). Our specific objectives are to (1) examine relationships among selected biophysical and technological indicators and flood metrics, (2) derive an importance-values for each biophysical and technological indicator based on selected flood metrics, (3) assess the RCN indicator and an alternative set of indicators to characterize biophysical capacity, and (4) map biophysical and technological flood regulation capacities for selected watersheds based on indicator importance-values derived from flood metrics. We conclude by discussing the landscape indicators that regulate floods, the methods and models to characterize flood-regulation capacity, the use of flood regulation capacity maps by watershed managers and planners, and the transferability and limitations of our methodology.

Methods

Study Area

We selected eight gaged urban watersheds in the piedmont of North Carolina (NC) based on size (≤ 80 km^2) and availability of long-term instantaneous discharge data (≥ 20 years; Table 3.1; Figure 3.1). Five watersheds were located in Charlotte, and the other three in Durham, Raleigh and Greensboro respectively. We limited the analysis to small watersheds to highlight the effect of land cover on flooding, as the influence of anthropogenic disturbance on stream flow strongly decreases with increasing watershed size (Tollan 2002, Bloschl et al. 2007, Chang and Franczyk 2008, Petrow and Merz 2009). The watersheds ranged from 60% to 100% urban land cover (includes open space, low, medium and high intensity development) and from 0% to 34% forested land cover (includes mixed, deciduous and evergreen forests, shrubs, and emergent herbaceous and woody wetlands). The watersheds share a similar mean annual precipitation (1018 - 1122 mm), but the percentage of sandy and loamy soils, which facilitate water infiltration, ranges widely (12% to 77%).

Flood metrics and landscape indicators

In the following section we describe how we selected the flood metrics and biophysical and technological indicators. We also describe how we use the two flood metrics and selected indicators to derive indicator-based importance-values for each flood metric, and then map biophysical and technological capacities to regulate floods.

Flood Metrics

We used long-term discharge records to evaluate the relative importance of the landscape in regulating floods. We selected magnitude and duration to characterize floods because these metrics are widely impacted by changes to the landscape (Moglen and Beighley 2002, Huang et al. 2008). Generally, studies in urban areas report an inverse relationship between magnitude and duration (i.e., the greater the magnitude, the shorter the duration of floods) (Hawley and Bledsoe 2011). Together, these metrics represent how flood regimes respond to the landscape.

We downloaded peak and instantaneous discharge records from the United States Geological Survey (USGS) Water for the Nation Database (USGS 2013). Using the peak discharge records, we used USGS's Peak FQ program to derive the discharge values for the 1.005-year flood (henceforth 1-year flood), and manually estimated 80% discharge of a 1-year flood. We defined the lowest flood as 80% of a 1-year flood, which is a flood that happens, on average, multiple times a year. This arbitrary threshold enabled us to assess small changes in the flood regime over time (Huang et al. 2008) at a flow below the bank-full discharge (Poff and Ward 1989; Bunn and Arthington 2002). Striving to represent trends in hydrology, as opposed to yearly variations in precipitation, we used at least 21 years of peak discharge records to derive return period in PeakFQ (Flynn et al. 2006).

We used instantaneous records to derive flood magnitude and duration from 1991 to 2013 as this period held a continuous record for most of the watersheds. We selected instantaneous discharge records to capture the peaks of floods that might be obscured in daily average data (Rice and Hirsch 2012). We defined a complete water year discharge record as one having at least 300 days of data. We compiled the magnitude and duration of independent floods (a flood could not last < 24 hours) per water year. Flood magnitude is the amount of discharge above 80% of a 1-year flood. Flood duration is the length of time when discharge exceeds the flood threshold per flood event. Since we were tabulating the flood metrics by water year, we had four events that occurred from one water year to the next, accounting for 0.2% of the records. We split these into independent floods by water year. For example, if a flood occurred from September 30th to October 1st, we counted these as two events, and summed the corresponding duration and magnitude for each event. We excluded flood events that surpassed the 5-year flood as events beyond this size have been shown to be driven by larger-scale drivers (i.e., climatic regimes) (Mogollón et al. *In Prep*; Hawley & Bledsoe 2011).

During the 23-year time period, six watersheds had from one to seven years of discharge data missing, for a total of 23 years missing or 14% of the entire record. One watershed had six consecutive years of missing data (1992-1997), two had four consecutive years, one had two sets of three consecutive years, and the other three had complete records. We interpolated the missing years for a given watershed, instead of discarding these watersheds, by taking the average of the two years preceding and succeeding the missing values. We verified that the interpolated records fell within the range of values for watersheds that had discharge records for each water year.

Biophysical and Technological Processes and Indicators

We identified biophysical (runoff, evapotranspiration, soil infiltration, water retention, surface flow, and natural water storage) and technological (flow-regulation) processes that regulate floods in urban environments, and derived a spatially-explicit indicator to represent each process from publicly available data (Leopold 1968, Smith et al. 2002b, Sun et al. 2005, Chang and Franczyk 2008) (Figure 3.2; Table 3.2). We first assessed the association between selected indicators and flood metrics using simple linear regression. Then we used the aforementioned annual flood metrics to derive importance values for each indicator, then applied the importance value to spatial indicators to map biophysical and technological flood regulation capacities. Our spatial analyses were carried out in ESRI's ArcGIS v. 10.1 (ESRI Inc., Redlands, CA) and our statistical analyses in SAS v. 9.3 (SAS Institute Inc. Cary, NC).

Biophysical processes and indicators

Runoff

Surface runoff is the excess water that is not infiltrated, evapotranspired or retained; instead, it flows over the land into streams (Lana-Renault et al. 2011). A common approach used to represent overland runoff is the runoff curve number (RCN), where a high value indicates high runoff potential (Villamagna and Angermeier 2014). Given equal precipitation conditions, we expect greater flood magnitude and shorter flood duration as runoff curve number (RCN) increases (Simonit and Perrings 2013).

We derived the RCN for our eight watersheds by intersecting SSURGO (Soil Survey Geographic Database) soil hydrologic groups (NRCS 2012) with land cover types from five time periods (1990, 1992, 2001, 2006 and 2011; Appendix H). We used the 1992-enhanced National Land Cover Database (NLCD) representing the 1990 time period (Nakagaki et al. 2007), retrofit 1992-2001 for 1992 and 2001 (Fry et al. 2009), 2006 (Fry et al. 2011) and 2011 (Jin et al. 2013) NLCD as data sources. For the years without land cover data, we interpolated the associated RCN value using the available dataset before and after the year of interest. We estimated the mean RCN for each year from 1991 to 2013 for each watershed.

Evapotranspiration

Evapotranspiration (ET) is the water entering the atmosphere from the landscape. To represent evapotranspiration, we used the relationship reported by Singh et al. (2013), where each land cover type from the NLCD is assigned an annual evapotranspiration rate (mm yr^{-1}). The estimates reported here are for Colorado which underestimate the real evapotranspiration rate of the southeastern US by around 200mm (Sanford and Selnick 2013), as the southeastern US sits at lower elevation and on average has longer days (Lu et al. 2003). However, we found no evidence that variation among land cover types differs between geographic regions. We expect flood magnitude and flood duration to decrease as ET rate increases (Sun et al. 2005). We used the same NLCD databases as for the RCN to derive an annual evapotranspiration rate for each of the five time periods. We estimated an ET rate for the years without land cover data by interpolating NLCD values using the most recent available dataset before and after the year of interest. We estimated the mean ET rate for each year from 1991 to 2013 for each watershed.

Soil Infiltration

During soil infiltration water moves through soils; sandy and clayey soils have high and low infiltration potential, respectively. We used SSURGO's saturated hydraulic conductivity (Ksat) as an indicator of soil infiltration (NRCS 2012). Ksat measures the amount of water flowing through the soil in a given time, ranging from 0 to 705 um s^{-1}. We expect Ksat to be positively related to flood duration and negatively related to flood magnitude (Ogden et al. 2000). Notably, we did not account for processes that occur once water infiltrates the soil. Groundwater flow paths vary in their lengths and residence times, and limited understanding of how these vary over space and time prohibits us from incorporating these processes in a spatially explicit way (Hester and Gooseff 2011). While groundwater could potentially influence flooding, we presume our study watersheds to behave similarly because they are predominantly urban and in the same physiographic province (Markewich et al. 1990, Rose and Peters 2001). We estimated the mean Ksat for each watershed and assumed the Ksat estimates remained constant through time.

Water Retention

Water retention reflects soil's capacity to retain water. Under unsaturated conditions, soil retention can play an important role in regulating floods in urban areas (Smith et al. 2002a). We used SSURGO's available water storage (AWS) indicator which measures the amount of water available for plants (in mm) that can be stored in the first 150cm of soil (NRCS 2012). We expect flood magnitude to decrease and flood duration to increase as AWS increases (Zhang et al. 2008). We estimated the mean AWS for each watershed and assumed AWS remained constant through time.

Surface Flow

Slope influences how quickly surface water runoff occurs. Runoff is faster and there is less water retention across steeper slopes compared to flatter terrain. We used a 10-m Digital Elevation Model (DEM) from the National Hydrography Database (NHD 2013) to derive the mean hill slope for each watershed using ESRI's ArcGIS Slope tool, and we used the same values through time. We expect flood magnitude to increase and flood duration to decrease with steeper slopes (Harden and Scruggs 2003).

Natural Water Storage

Natural water storage across landscapes occurs in streams or ponds that permanently or temporarily collect water. Land cover databases underestimate the true area of such water features, particularly streams (Di Sabatino et al. 2013). We used previous research to estimate the area of each watershed covered by streams. We derived a flow network from a 10-m flow accumulation and direction rasters (NHD 2013), identified stream segments using stream confluences, and then calculated the drainage area for each stream segment. We used this derived segment-level drainage area to estimate the width of the stream using the Piedmont physiographic province equation (Johnson and Fecko 2008) that relates watershed drainage area to stream width. We then used that estimated stream width to estimate the stream area of each segment, and calculated the percentage of stream (henceforth referred to as percent stream) area in each watershed. We expect flood magnitude to decrease and flood duration to increase as percent stream increases (Downing et al. 2006). We did not identify any natural ponds in our study watersheds.

Technological processes and indicators

Flow-regulating features are ubiquitous across the landscape and are widely used to control and store water (Smith et al. 2002b, Downing et al. 2006, Ignatius and Jones 2014). We used best management practices (BMPs) and artificial water bodies (AWBs) as indicators of flow-regulation. The primary distinction is that BMPs are intended to regulate stormwater, while AWBs store water but storage is not their main purpose. BMPs include flood control dams, wet and dry ponds, bioretention areas, stormwater wetlands, and sand filters; AWBs include farm ponds, golf course ponds, and water supply reservoirs among others. We include BMPs and AWBs under technological capacity because these are man-made features that replace or enhance ecosystem services (e.g., recreation, water supply or flood control).

We created a spatial database of BMPs and AWBs from the following sources: National Inventory of Dams (USACE 2012), Global Reservoirs and Dams Database (Lehner et al. 2011a), National Anthropogenic Barrier Dataset (Ostroff et al. 2012), Federal Emergency Management Agency (FEMA 2013b), National Hydrograph Database on Waterbodies (USGS 2012), North Carolina Database (NCDENR 2013), and BMP databases from selected counties. We used Google Earth aerial imagery to verify the existence of, classify, map the surface area of, and date BMPs and AWBs. We used surface area as a surrogate of volumetric capacity, as information on volumetric capacity was largely unavailable. We standardized BMPs and AWBs across watersheds by calculating the cumulative percentage of BMP and AWB (henceforth % BMP and % AWB, respectively) surface area, by watershed area per year, from 1991 to 2013 for each watershed. We expect flood magnitude to decrease and flood duration to increase as % BMP and %AWB increase (Zahran et al. 2008).

Deriving biophysical and technological indicator importance-values

After deriving the indicators, we assessed biophysical and technological capacities separately. We assessed biophysical capacity in two ways to isolate the effect of RCN: the first includes RCN, slope and streams (henceforth B1), and the second includes ET, Ksat, AWS, slope and streams (henceforth B2). We separated RCN from ET because they both were derived from the same land cover database, and because RCN represents runoff potential while ET, Ksat, and AWS represent the non-runoff potential (i.e., evapotranspiration, infiltration and retention).

We assessed the importance-values of biophysical and technological indicators via regressions and relations to flood metrics. Deriving importance-values provides an understanding of which indicators are most important in regulating the magnitude and duration of floods for selected watersheds. Flood magnitude and duration fit lognormal distributions (link=identity) with an over-dispersion parameter of

0.5 and 0.26, respectively (Anderson 2008). We used GLMMs to derive importance-values of B1, B2 and technological indicators, as these models handle non-normal data and incorporate random effects (Bolker et al. 2008). Variance in indicator importance-values depends on which other indicators are included in the model. We included watershed ID as a random effect, and B1, B2 and technological indicators, in addition to mean annual precipitation, as fixed effects. We ran an intercept-only model in addition to seven models for B1, 31 models for B2, and three models for technological capacity; these represent all possible combinations of indicators within each group for flood magnitude and duration. Mean annual precipitation for each year, from 1991-2012, was included in all models (PRISM Climate Group 2013), except for the intercept-only models. We used the model AICc values to derive AICc weights for the indicators within B1, B2 and technological models and derived the importance-value of each indicator by adding the AICc weights from each model where the indicator was present within B1, B2 and technological models (Anderson 2008). We conducted this process for both flood metrics, such that we derived four estimates of biophysical capacity (B1 and B2, for flood magnitude and duration) and two estimates of technological capacity (for flood magnitude and duration).

Mapping flood regulation capacity

Below we describe how we spatially standardized and applied the importance-value to each indicator within the B1, B2 and technological capacity models. We mapped the biophysical and technological flood regulation capacity for current condition (circa 2011-2013). We used ArcGIS v. 10.1 (ESRI Inc., Redlands, CA) to standardize the indicators and map capacities. We used 10-m-pixel resolution, which matched the resolution of most data; however, we had to resample the land cover databases (used in ET and RCN) from 30- to 10-m pixels. Many of the streams, BMPs and AWBs were less than $100m^2$, so we were likely to further underestimate these if mapped at a 30-m-pixel resolution.

All indicators had different ranges, which precluded meaningful comparisons if left unstandardized. We standardized the indicators from 0 to 1 with the equation,

$$\frac{(X - X_{min})}{(X_{max} - X_{min})}$$

where values near 1 mean relatively high flood regulation capacity, and values near 0 mean relatively low flood regulation capacity. These relative measures of capacity have been widely used in ecosystem service mapping (Nedkov and Burkhard 2012, Larondelle et al. 2014, Villamagna et al. 2014). We subtracted the standardized values from 1 for RCN and slope to portray flood regulation instead of flood potential (e.g., values near 1 mean relatively higher capacity to retain water). Without volumetric storage data, we relied on the surface area of streams, BMPs and AWBs as a surrogate. Data for these three indicators were standardized as binary values, where 1 represented presence and 0 represented absence. With this standardization, every 10-m pixel in each watershed had a value for each indicator.

After standardizing the indicators from 0 to 1, we scaled the importance-values within the B1, B2 and technological capacity models such that the indicator with the highest importance-value, within each model, equaled 1. Using Raster Calculator in ArcGIS, we added the indicators within the B1, B2 and capacity technological models, multiplying the standardized importance-value by the corresponding indicator. To compare B1, B2 and technological flood regulation capacities, we re-standardized each from 0 to 1 as each capacity had a different number of indicators. We derived standardized biophysical and technological flood regulation capacity maps using the importance-values derived for both flood magnitude and duration.

Results

Below, we summarize (1) relationships among selected biophysical and technological indicators and flood metrics, (2) importance-values for each indicator, (3) how the RCN and the alternative approach represented biophysical flood regulation, and (4) how we used importance-values derived from flood records to map biophysical and technological capacities to regulate floods.

Biophysical and technological indicators

Most biophysical indicators were not significantly related to flood metrics, yet some performed as expected (Table 3.3). In particular, the parameter estimates of RCN, slope, ET, AWS and Ksat were bounded by zero which precluded us from understanding their relation to flood magnitude. % stream was positively related to flood magnitude, although we expected a negative relation. For flood duration, the parameter estimate of slope was bounded by zero, while RCN, % stream, ET, AWS and Ksat were positively related to flood duration. We expected RCN and ET to negatively relate to flood duration.

The two technological indicators were not significantly related to flood magnitude and duration (Table 3.3). The majority (76%) of BMPs within our study watersheds were wet ponds, followed by flood control dams (11%) and dry ponds (10%). The majority (60%) of AWBs were reservoirs, followed by farm ponds (34%) and golf course ponds (6%).

Indicator importance-values

Biophysical indicator importance-values differed between flood magnitude and duration (Table 3.4). Percent stream exhibited the greatest importance-value for flood magnitude, while land cover indicators (RCN for B1 and ET for B2) had the greatest importance-values for flood duration. The unscaled indicator values for flood magnitude were similar to each other, ranging from 0.30 to 0.64, while the indicator values for flood duration were more varied, ranging from 0.28 to 0.98. There was no consistency in importance-values derived for flood magnitude versus duration (e.g., slope showed high importance for flood magnitude, but low importance for flood duration).

Technological indicator importance-values were similar between flood magnitude and duration (Table 3.4). Both indicators had similar importance-values for flood magnitude, while %BMP was not as important as %AWB for flood duration.

Approaches to characterizing biophysical capacity

Overall, we found RCN did not have the expected association with flood magnitude and duration (Table 3.3). An increase in RCN was associated with a decrease in flood magnitude and an increase in flood duration. In the B1 capacity model, the inverse of RCN was derived such that a greater runoff potential had a lower capacity. In comparing B1 capacity to flood metrics, we found that as B1 capacity increased, flood magnitude increased and flood duration decreased, which suggests that RCN is driving more of the relationship than the other two indicators in the model (i.e., percent stream and slope) (Table 3.4, Figure 3.3 and 3.4). In comparison, the alternative approach did yield the expected association with flood magnitude and duration. All indicators derived from soil and land cover in the B2 model had the expected relationship to flood metrics, except for AWS and slope (Table 3.3; see Biophysical and technological Indicators under Results). As expected, we found that as B2 increased, flood magnitude decreased and flood duration increased (Figure 3.3 and 3.4).

Exploring these patterns in individual watersheds provides insight into why the RCN approach provides the relationship opposite from what we expected. The watershed with the second highest B1 capacity for both flood magnitude- and duration-derived capacities (ID 02146470) had the highest flood magnitude and shortest duration (Figure 3.3 and 3.4). This watershed had the second greatest percentage of sandy and loamy soils (74%), and 100% urban cover (Table 3.1). In contrast, the watershed with the lowest B1 capacity for both flood metrics (ID 0209741955) had the lowest flood magnitude and longest flood duration. This watershed had the greatest forest cover percentage (34%), least urban cover (60%), and the smallest percentage of sandy and loamy soils (11.9%). These results suggest the RCN indicator gives greater importance to the soil hydrologic group, but masks the important effects of land cover. In separating the soil and land cover effects, we found flood magnitude decreased and duration increased with increasing B2 capacity (Figure 3.3 and 3.4), which is consistent with the notion of how watersheds regulate floods.

Maps of Biophysical and Technological Flood Regulation Capacity

Because B1 capacity results were confounded by the RCN, we focused further analysis on the spatial distribution of B2 and technological flood regulation capacities for derived importance-values. Overall, flood duration increases and flood magnitude decreases as B2 and technological capacities increase for capacity values derived from both flood metrics (Figure 3.3 and 3.4). Spatial differences in B2 between flood magnitude and duration are apparent (Figure 3.5 and 3.6). The low B2 capacity of watershed 02146470 and 02146700 in both maps contrasts with the high B2 capacity of watershed 0208732885. Magnitude-derived B2 shows distinct areas of low capacities which correspond to highly impervious surfaces (e.g., roads, airports, parking lots). These patterns are mainly driven by ET, Ksat and AWS because slope is relatively uniform throughout the watersheds. In contrast, the distribution of duration-derived B2 is driven mainly by ET, where patches of higher capacity are distributed throughout each watershed. Spatial patterns of biophysical capacity can help managers identify areas where enhancing evapotranspiration, Ksat and AWS processes (e.g., re-vegetation), leads to lower-peaked and longer duration floods.

Differences in the spatial distribution of technological capacity across watersheds are apparent (Figure 3.5 and 3.6). Two watersheds (IDs 02146470 and 02146700) have little technological flood regulation capacity, while three have high levels of technological capacity (IDs 02099000, 02974155, and 0208732885). BMPs and AWBs within the latter group were generally located along stream corridors, and uniformly distributed in the upper and lower parts of the watersheds. In watersheds with high percentages of BMPs and AWBs, spatial patterns of technological capacity can guide future development in the area, while watersheds with low percentages of technological features can help managers prioritize area where these technologies are needed most.

The relation between technological and biophysical capacities and flood metrics are illustrated by patterns in watersheds 02099000 and 02142900 through time. As non-urban (forested and agricultural) land cover decreased, BMPs and AWBs increased (Figure 3.7). From 1991 to 2013, AWBs increased 0.03% and 0.02%, and BMPs increased 0.26% and 0.98%, for watersheds 02142900 and 02099000, respectively.

Meanwhile, non-urban cover decreased 30% and 26%, respectively. That is, the loss of forest and agricultural lands, was met with building more flow-regulating features (AWBs and BMPs). Similar landscape characteristics between these two watersheds (Table 3.1), allow us to compare the effectiveness of non-urban land cover and flow-regulating features in regulating floods. We would expect watershed 02099000 to have lower and longer floods since it had between 0.5% and 1.3% more flow-regulating features than watershed 02142900. Instead, watershed 02099000 had higher floods (9.2 $m^3 s^{-1} km^{-1} yr^{-1}$ versus 5.5 $m^3 s^{-1} km^{-1} yr^{-1}$) with similar duration (1.13 days $km^{-1} yr^{-1}$ versus 0.98 days $km^{-1} yr^{-1}$) compared to watershed 02142900. These results suggest that a greater extent of BMPs in watershed 02099000 did not mitigate the impacts of impervious surface. Instead, non-urban land cover was perhaps more effective at regulating floods, as before 2007, watershed 02142900 had around 7% more non-urban cover than watershed 02099000.

Discussion

The results showed that (1) the RCN approach and B1 capacity did not have the expected relationship to flood metrics, and that the alternative approach provided greater coherence of soil and land cover processes to flood magnitude and duration, and (2) we were able to map the landscape's technological and biophysical flood regulation capacities where the relative-importance of the indicators used were based on long-term hydrologic records. Below, we discuss the landscape indicators that regulate floods, the methods and models to characterize flood-regulation capacity, the utility of flood regulation capacity maps to landscape-level managers and planners, and the transferability and limitations of our method.

Landscape indicators that regulate floods

In the flood magnitude models, % streams ranked highest in both the B1 and B2 models and the second highest for flood duration's B2 model. Previous studies do not consider stream area when mapping flood regulation, as in most cases they are available as polylines instead of polygons, and are generally underrepresented in land cover classifications because they make up a small percentage of watersheds (Di Sabatino et al. 2013). In integrating Johnson and Fecko's (2008) methodology, we derived the area for each watershed's stream network. In general, larger watersheds had greater area of streams because stream width increases exponentially with stream order. Our results confirm the importance of streams to flood regulation, as these are the drainage avenues of watersheds and where most of urban flood management takes place (Tingle 1999).

Surprisingly, slope was the second-ranked landscape indicator of the flood magnitude B1 and B2 models, yet the least ranked of the B2 flood duration model. We expected slope to rank lowest for both models as the watersheds are distributed in North Carolina's piedmont, with similar slopes ranging from 7° to 7.64°.

We consider this idiosyncratic result an artifact of our small sample, but consider it an important indicator to represent the rate of surface runoff.

As we expected, evapotranspiration and RCN ranked highest in the flood duration B1 and B2 models and third in the B1 and B2 flood magnitude models. Both indicators are derived with land cover data (in addition to land cover data, RCN uses soil hydrologic group data). The high ranking of these indicators is consistent with previous studies that found land cover to be important in regulating runoff and floods (Poff et al. 2006, Amini et al. 2011, Öztürk et al. 2013). These high ranked indicators in the flood duration models are consistent with previous studies that found significant differences in the duration of < 20-year floods between urban and non-urban watersheds, whereas differences in flood magnitude between urban and non-urban watersheds were different for < 5-year floods (Mogollón et al. 2014). In other words, these results suggest that flood duration is more sensitive to changes in land cover than flood magnitude as the flood return interval increases.

Both Ksat and AWS are a function of soil type and precipitation. In the flood magnitude model, AWS ranked lowest followed by Ksat, while in the flood duration model, AWS was the third highest followed by Ksat. Previous studies have found antecedent soil conditions important in determining the magnitude and duration of a flood, yet their importance decreases with urbanization, due to greater impervious surface (Smith et al. 2002a). For study watersheds having > 60% of urban coverage, the relatively low ranking of Ksat and AWS was expected.

Both technological indicators, %BMP and %AWB, overall ranked high in the magnitude and duration models. Contrary to expectations, %AWB ranked higher than %BMP in both models. BMPs included in this study were constructed for the purpose of flood regulation, and thus we expected these to be most important in both cases. We believe the capacity of BMPs might be underestimated by using surface area as a surrogate for capacity. On average, AWBs were three times larger than BMPs. Further work could use volumetric capacity per unit time to differentiate between those flow-regulating features that maintain certain volumes of water (e.g., AWBs), and those that are constantly changing their capacities based on rainfall intensity and frequency (e.g., BMPs) (Strecker et al. 2001).

Methods and models to characterize flood-regulation capacity

Using two different models to represent biophysical capacity allowed us to empirically show different responses between the widely-used RCN approach and an alternative approach. In assessing the GLMM parameter estimate results between RCN and flood metrics, we found RCN was positively related to flood duration, while its parameter estimate was bounded by zero for flood magnitude (Table 3.3). We expected a positive relation to flood magnitude and a negative relation to flood duration (Simonit and

Perrings 2013). None of the previous studies using RCN to portray runoff potential had used empirical flood records to understand if the processes depicted in the relation were as expected. In our study, the most urbanized watersheds also had the greatest extent of soils with high infiltration potential (A and B). These were also the watersheds that had the highest-peaked and shorter floods. We expected that the capacity of water to infiltrate diminished with increasing impervious surface (Smith et al. 2002a), yet we found that there was a lower runoff potential with increasing urbanization. An alternative, perhaps more sensitive metric in urban areas, is the degree of imperviousness to represent surface runoff potential (Schueler et al. 2009). We suspect imperviousness would be a better metric than RCN, as it is solely based on land cover and does not have the confounding soil effects seen with RCN. This is the first study to provide evidence in the ecosystem service mapping context of the contradictory results between flood records and RCN. Further research is needed to understand if the RCN approach is applicable to non-urban watersheds, yet as Garen and Moore (2005) argue, the RCN was never intended to be used on a per-pixel basis. The alternative approach, of separating evapotranspiration, infiltration and retention processes, mostly provided a coherent response to flood duration, yet parameter estimates were bounded by zero for flood magnitude.

Utility of mapping biophysical and technological flood regulation capacities

Unlike services such as food and energy that can be produced in one area and delivered to another, flood regulation is a spatially dependent local service as benefits are transferred downstream. Flood control management is divided among several actors. On the one side, stormwater managers focus mainly on controlling floods along floodplains as these have the highest risk of flooding and are where most of the property damages occur (Tingle 1999). At the same time, constructors are required to mitigate the impacts from additional impervious surfaces on water quality and quantity (EPA 2014). Besides their local knowledge of the area, planners may lack an understanding of the entire watershed's capacity to regulate floods. Thus, we consider the main utility of mapping biophysical and technological flood regulation capacities to be that it allows watershed coordinators and planners to identify the areas of relatively low and high capacity to implement and prioritize stormwater management programs because of its high spatial resolution (e.g., 10m pixels). Previous studies have shown that the type, design and location of BMPs and AWBs in the watershed impact the quantity and duration of water storage (Strecker et al. 2001). The spatial distribution of BMPs and AWBs embedded within the biophysical capacity context can provide a realistic spatial understanding of the watershed's overall capacity. However, determining what types of strategies are needed to reduce flooding would require an assessment of demand for the flood regulation service, in addition to an assessment of landscape patterns such that the relation between upstream to downstream capacities are accounted for (Nedkov and Burkhard 2012).

Methodological transferability and limitations

In developing this methodology, we wanted to improve on previous efforts to map flood regulation capacity while maintaining an accessible approach. One of our greatest limitations was our small sample size. Many parameter estimates between biophysical and technological indicators and flood metrics were not significant; and some of the observed significant relations were not as expected. We suspect results were sensitive to individual observation due to our small sample size, yet consider that we would find more definitive patterns when using larger samples. In addition to our small sample, we did not test for temporal or spatial autocorrelation but consider that not controlling for such correlations can reduce the effective number of samples (Douglas et al. 2000). Furthermore, one watershed (ID 02146211) was nested within the largest watershed (ID 02146300) in the area of Charlotte, NC. We kept both to increase our sample size but we recognize that the landscape attributes and discharge patterns are similar. Despite our small sample size, we believe the methodology developed herein improves previous mapping efforts of flood regulation and other ecosystem services.

The flood regulation capacity values derived in this study are relative to the watersheds assessed. In other words, if we were to re-run this analysis with seven instead of eight watersheds, we would get different capacity values but would expect the same general relationships between flood metrics and indicators. We consider that applying the same indicator importance-values to neighboring urban watersheds using the same databases as this study is appropriate; however, these importance-values do not apply to non-urban watersheds or watersheds with different topographies and climate regimes.

Integral to this study was the use of long-term hydrologic data to derive importance-values. While discharge data are widely available in the US, this might be a limitation in areas with no or limited stream gages. In addition, of the publicly available data used in this study, by far the most challenging to compile was the flow-regulating features for which no one database had accurate, up-to-date information on their distribution and size. Comparing the effectiveness between biophysical and technological capacities would require more detailed information on flow-regulating features, in addition to controlling for other landscape characteristics that influence flooding (Strecker et al. 2001). Herein, we used surface area as a surrogate for volumetric capacity of BMPs and AWBs, which means that a flood control dam and a constructed wetland with equal surface area, are weighed the same. We are aware that great variation exists among flow-regulating features, but the lack of available data on storage, infiltration and evapotranspiration capacities, leaves us unable to include this information. Compiling this information is not only important for watershed coordinators on a local-scale, but also important for researchers assessing the effectiveness of flow-regulating features and natural ecosystems across space and time.

Conclusion

Flood regulation is a service included in many studies of ecosystem service bundles (Posthumus et al. 2010), trade-offs and synergies (Latera et al. 2012), and scenario analyses (Eigenbrod et al. 2011; Jackson et al. 2013). Yet few studies have dissected the processes governing flood regulation (Ennaanay et al. 2011) which is an important first step if subsequent studies use it as one of many other services. In an attempt to improve previous flood regulation mapping efforts, we identified processes important to flood regulation, selected suitable technological and biophysical indicators from publicly available data, and used long-term flood records to assess the relative importance of indicators to regulate the magnitude and duration of <5-year floods. We subsequently used the ranking of indicators to map the biophysical and technological capacity to regulate floods. We found that biophysical and technological capacities were negatively related to flood magnitude and positively related to flood duration (with the exception of the B1 model containing RCN as discussed in *Methods and models to characterize flood-regulation capacity* under the Discussion), which suggests that the indicators used and the ranking process worked as expected.

A major advancement over previous studies was including technological features in mapping the flood regulation service. Particularly in urban environments, technological features are ubiquitous and in many cases intended to mimic the services provided by natural ecosystems (i.e., constructed wetlands, artificial waterbodies). In the conservation and ecosystem service literature, there is a sense that technology can enhance and replace certain ecosystem services, but they are often more expensive, less stable, and less reliable than the services conveyed naturally by ecosystems which frequently provide a wider array of social benefits than technological solutions (Bernhardt et al. 2006).

In our study, the majority of flow-regulating features were designed to retain and store surface runoff and have been shown to lower peak flows (Burns et al. 2012). Storage-based BMPs have, in some instances, decreased the socio-economic costs associated with inland flooding (Brody et al. 2011), but have been shown to only marginally decrease stream flashiness (Damodaram et al. 2010). Managers are confronted with decreasing flood related damage, while restoring the timing, variability and quantity of flows that aquatic environments need. Infiltration-and evapotranspiration-based technological features (e.g., rain gardens, green roofs, rainwater harvesting, permeable pavement) have been shown, on a small scale, to regulate floods by allowing surface runoff to infiltrate into the ground, evaporate into the atmosphere, or be used locally (Deutsch et al. 2005, Czemiel Berndtsson 2010, Damodaram et al. 2010). Besides reducing costs associated with recurrent flashy flows along properties near streams, these technological features could provide the conditions for healthy waterways (e.g., water quality, urban biodiversity and carbon sequestration). In this sense, technology in freshwater conservation might be redefined to benefit aquatic environments, as well as lower flood-related damages, in addition to acknowledge the services provided by free-flowing rivers (Auerbach et al. 2014). Excluding these features from ecosystem service assessments fails to provide a true assessment of the services provided. We believe our study will encourage others to acknowledge technological replacements or enhancements in mapping and quantifying ecosystem services. However, more research is needed to understand the true costs and effectiveness of replacing services provided by natural ecosystems with technological solutions (Fitter 2013).

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References

Amini, A., T. M. Ali, A. H. B. Ghazali, A. A. Aziz, and S. M. Akib. 2011. Impacts of Land-Use Change on Streamflows in the Damansara Watershed, Malaysia. Arabian Journal for Science and Engineering 36(5):713–720.

Anderson, D. R. 2008. Model Based Inference in the Life Sciences: A Primer on Evidence. Springer, New York City.

Auerbach, D. A., D. B. Deisenroth, R. R. McShane, K. E. McCluney, and N. LeRoy Poff. 2014. Beyond the concrete: Accounting for ecosystem services from free-flowing rivers. Ecosystem Services 10:1–5.

Barron, O. V., D. Pollock, and W. Dawes. 2011. Evaluation of catchment contributing areas and storm runoff in flat terrain subject to urbanisation. Hydrology and Earth System Sciences 15(2):547–559.

Bernhardt, E., S. E. Bunn, D. D. Hart, T. Muotka, R. J. Naiman, C. Pringle, M. Reuss, and B. Van Wilgen. 2006. Perspective : The challenge of ecologically sustainable water management. Water Policy 8:475–479.

Bloschl, G., S. Ardoin-Bardin, M. Bonell, M. Dorninger, D. Goodrich, D. Gutknecht, D. Matamoros, B. Merz, P. Shand, and J. Szolgay. 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes 1247(January):1241–1247.

Bolker, B. M., M. E. Brooks, C. J. Clark, S. W. Geange, J. R. Poulsen, M. H. H. Stevens, and J.-S. White. 2008. Generalized linear mixed models: a practical guide for ecology and evolution. Trends in Ecology & Evolution 24(3):127–135.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38(3):835–947.

Brabec, E., S. Schulte, and P. L. Richards. 2002. Impervious surfaces and water quality: A review of current literature and its implications for watershed planning. Journal of Planning 16(4):499–514.

Brody, S. D., W. E. Highfield, and J. E. Kang. 2011. Rising Waters: The Causes and Consequences of Flooding in the United States. Page 195. Cambridge University Press.

Bunn, S. E., and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30(4):492–507.

Burns, M. J., T. D. Fletcher, C. J. Walsh, A. R. Ladson, and B. E. Hatt. 2012. Hydrologic shortcomings of conventional urban stormwater management and opportunities for reform. Landscape and Urban Planning 105(3):230–240.

Chan, K. M. a, M. R. Shaw, D. R. Cameron, E. C. Underwood, and G. C. Daily. 2006. Conservation planning for ecosystem services. PLoS biology 4(11):e379.

Chang, H., and J. Franczyk. 2008. Climate change, land use change, and floods: Towards an integrated assessment. Geography Compass 2(5):1549–1579.

Chowdhury, F., C. Lant, and B. Dziegielewski. 2013. A century of water supply expansion for ten U.S. cities. Applied Geography 45:58–76.

Czemiel Berndtsson, J. 2010. Green roof performance towards management of runoff water quantity and quality: A review. Ecological Engineering 36(4):351–360.

Damodaram, C., M. H. Giacomoni, C. Prakash Khedun, H. Holmes, A. Ryan, W. Saour, and E. M. Zechman. 2010. Simulation of Combined Best Management Practices and Low Impact Development for Sustainable Stormwater Management. Journal of the American Water Resources Association 46(5):907–918.

Davis, A. P. 2008. Field Performance of Bioretention: Hydrology Impacts. Journal of Hydrologic Engineering 13(2):90–95.

Deutsch, B., H. Whitlow, M. Sullivan, and A. Savineau. 2005. A Green Roof Vision Based on Quantifying Storm Water and Air Quality Benefits. Pages 379–384 *in* The Cardinal Group Toronto, editor. Proceedings of the 3rd North American: Green Roof Conference on Greening Rooftops for Sustainable Communities. Washington, DC.

Douglas, E. M., R. M. Vogel, and C. N. Kroll. 2000. Trends in floods and low flows in the United States: impact of spatial correlation. Journal of Hydrology 240:90–105.

Downing, J. A., Y. T. Prairie, J. J. Cole, C. M. Duarte, L. J. Tranvik, R. G. Striegl, W. H. McDowell, P. Kortelainen, N. F. Caraco, and J. M. Melack. 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnology and Oceanography 51(5):2388–2397.

Dutton, P. D. 2012. Factors influencing roadside erosion and in-stream geomorphic stability at road-stream crossings for selected watersheds, North Shore, Minnesota, USA. Master's Thesis Natural Resources Science and Management, University of Minnesota.

Eigenbrod, F., V. A. Bell, H. N. Davies, A. Heinemeyer, P. R. Armsworth, and K. J. Gaston. 2011. The impact of projected increases in urbanization on ecosystem services. Proceedings of the Royal Society B: Biological Sciences 278(1722):3201–8.

Eng, K., D. M. Carlisle, D. M. Wolock, and J. A. Falcone. 2013. Predicting the likelihood of altered streamflows at ungaged rivers across the conterminous United States. River Research and Applications 29:781–791.

Ennaanay, D., M. Conte, K. Brooks, J. Nieber, M. Sharma, S. Wolny, and G. Mendoza. 2011. Valuing land cover impact on storm peak mitigation. Page Pages 73–88 *in* P. Kareiva, H. Tallis, T. H. Ricketts, G. C. Daily, and S. Polasky, editors. Natural Capital:Theory and Practice of Mapping Ecosystem Services. Oxford University Press.

EPA. 2014. Post-Construction Stormwater Management in New Development and Redevelopment. http://water.epa.gov/polwaste/npdes/swbmp/PostConstruction-Stormwater-Management-in-New-Development-and-Redevelopment.cfm.

FEMA. 2013. Map Service Center. https://msc.fema.gov/webapp/wcs/stores/servlet/info?storeId=10001&catalogId=10001&langId=-1&content=productDFIRM&title=DFIRM Databases&parent=productInfo&parentTitle=Product Information.

Findlay, S. J., and M. P. Taylor. 2006. Why rehabilitate urban river systems? Area 38(3):312–325.

Fiquepron, J., S. Garcia, and A. Stenger. 2013. Land use impact on water quality: Valuing forest services in terms of the water supply sector. Journal of Environmental Management 126C:113–121.

Fitter, A. H. 2013. Are Ecosystem Services Replaceable by Technology? Environmental and Resource Economics 55(4):513–524.

Flynn, K. M., W. H. Kirby, and P. R. Hummel. 2006. User's manual for program PeakFQ; annual flood-frequency analysis using Bulletin 17B guidelines. Reston, VA.

Fry, J. A., M. J. Coan, C. G. Homer, D. K. Meyer, and J. D. Wickham. 2009. Completion of the National Land Cover Database (NLCD) 1992 – 2001 land cover change retrofit product. Page 18.

Fry, J., G. Xian, S. Jin, J. Dewitz, C. Homer, L. Yang, C. Barnes, N. Herold, and J. Wickham. 2011. Completion of the 2006 National Land Cover Database for the Conterminous United States. Photogrammetric Engineering and Remote Sensing 77(9):858–864.

Garen, D. C., and D. S. Moore. 2005. Curve number hydrology in water quality modeling: uses, abuses and future directions. Journal of the American Water Resources Association 41(2):377–388.

Goff, K. M., and R. W. Gentry. 2006. The Influence of Watershed and Development Characteristics on the Cumulative Impacts of Stormwater Detention Ponds. Water Resources Management 20(6):829–860.

Green, C. H., and E. C. Penning-Rowsell. 1989. Flooding and the Quantification of "Intangibles". Water and Environment Journal (8):27–30.

Harden, C. P., and P. D. Scruggs. 2003. Infiltration on mountain slopes: a comparison of three environments. Geomorphology 55(1-4):5–24.

Hawley, R. J., and B. P. Bledsoe. 2011. How do flow peaks and durations change in suburbanizing semi-arid watersheds? A southern California case study. Journal of Hydrology 405(1-2):69–82.

Hester, E. T., and M. N. Gooseff. 2011. Hyporheic restoration in streams and rivers. Pages 167–187 *in* A. Simon, S. J. Bennett, and J. M. Castro, editors. Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses, and Tools. Geophysical Monograph Series, Washington, D. C.

Huang, S., S. Cheng, J. Wen, and J. Lee. 2008. Identifying peak-imperviousness-recurrence relationships on a growing-impervious watershed, Taiwan. Journal of Hydrology 362(3-4):320–336.

Ignatius, A., and J. Jones. 2014. Small reservoir distribution, rate of construction, and uses in the upper and middle Chattahoochee basins of the Georgia piedmont, USA, 1950–2010. International Journal of Geo-Information 3(2):460–480.

Jackson, B., T. Pagella, F. Sinclair, B. Orellana, A. Henshaw, B. Reynolds, N. Mcintyre, H. Wheater, and A. Eycott. 2013. Polyscape: A GIS mapping framework providing efficient and spatially explicit landscape-scale valuation of multiple ecosystem services. Landscape and Urban Planning 112:74–88.

Jencso, K. G., B. L. Mcglynn, M. N. Gooseff, S. M. Wondzell, K. E. Bencala, and L. A. Marshall. 2009. Hydrologic connectivity between landscapes and streams: Transferring reach- and plot-scale understanding to the catchment scale. Water Resources Research 45:1–16.

Jin, S., L. Yang, P. Danielson, C. Homer, J. Fry, and G. Xian. 2013. A comprehensive change detection method for updating the National Land Cover Database to circa 2011. Remote Sensing of Environment 132:159–175.

Johnson, P. A., and B. J. Fecko. 2008. Regional channel geometry equations: a statistical comparison for physiographic provinces in the Eastern US. River Research and Applications 834:823–834.

Koschke, L., C. Fürst, M. Lorenz, A. Witt, S. Frank, and F. Makeschin. 2013. The integration of crop rotation and tillage practices in the assessment of ecosystem services provision at the regional scale. Ecological Indicators 32:157–171.

Kraus-Elsin, Y., R. A. Kramer, and W. A. Jenkins. 2010. Valuing Drinking Water Provision as an Ecosystem Service in the Neuse River Basin. Journal of Water Resources Planning and Management (August):474–483.

Lana-Renault, N., J. Latron, D. Karssenberg, P. Serrano-Muela, D. Regüés, and M. F. P. Bierkens. 2011. Differences in stream flow in relation to changes in land cover: A comparative study in two sub-Mediterranean mountain catchments. Journal of Hydrology 411(3-4):366–378.

Lantz, V., R. Trenholm, J. Wilson, and W. Richards. 2012. Assessing market and non-market costs of freshwater flooding due to climate change in the community of Fredericton, Eastern Canada. Climatic Change 110(1-2):347–372.

Larondelle, N., D. Haase, and N. Kabisch. 2014. Mapping the diversity of regulating ecosystem services in European cities. Global Environmental Change 26:119–129.

Laterra, P., M. E. Orúe, and G. C. Booman. 2012. Spatial complexity and ecosystem services in rural landscapes. Agriculture, Ecosystems & Environment 154:56–67.

Lecce, S. A., and E. S. Kotecki. 2008. The 1999 Flood of the Century in Eastern North Carolina: Extraordinary Hydro-Meterological Event or Human-Induced Catastrophe? Physical Geography 29(2):101–120.

Lehner, B., C. R. Liermann, C. Revenga, C. Vorosmarty, B. Fekete, P. Crouzet, P. Doll, M. Endejan, K. Frenken, J. Magome, C. Nilsson, J. C. Robertson, R. Rodel, N. Sindorf, and D. Wisser. 2011. Global Reservoir and Dam Database, Version 1 (GRanDv1): Reservoirs, Revision 01. http://sedac.ciesin.columbia.edu/data/set/grand-v1-reservoirs-rev01.

Leopold, L. B. 1968. Hydrolog for Urban Land Planning - A guidebook on the hydrologic effects of urban land use. Pages 1–18. Washington, D. C.

Logsdon, R. a., and I. Chaubey. 2013. A quantitative approach to evaluating ecosystem services. Ecological Modelling 257:57–65.

Lu, J., G. Sun, S. G. McNully, and D. M. Amatya. 2003. Modeling actual evapotranspiration from forested watersheds across the southeastern United States. Journal of the American Water Resources Association 39(4):887–896.

Markewich, H. W., M. J. Pavich, and G. R. Buell. 1990. Contrasting soils and landscapes of the Piedmont and Coastal Plain, eastern United States. Geomorphology 3(3-4):417–447.

Millennium Ecosystem Assessment. 2005. Ecosystems and human well-being: current state and trends. World Resources Institute, Washington, DC.

Moglen, G. E., and R. E. Beighley. 2002. Spatially explicit hydrologic modeling of land use change. Journal of American Water Resources Association 38(1):241–253.

Mogollón, B., E. A. Frimpong, A. Hoegh, and P. L. Angermeier. 2014. What inland floods can be managed? Master’s Thesis in Fish and Wildlife Conservation, Virginia Tech.

Nakagaki, N., C. V. Price, J. A. Falcone, K. J. Hitt, and B. C. Ruddy. 2007. Enhanced National Land Cover Data 1992 (NLCDe 92). Sacramento, CA. http://water.usgs.gov/GIS/metadata/usgswrd/XML/nlcde92.xml.

NCDENR. 2013. North Carolina Dam Inventory. http://portal.ncdenr.org/web/lr/dams.

Nedkov, S., and B. Burkhard. 2012. Flood regulating ecosystem services — Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecological Indicators 21:67–79.

NHD. 2013. Digital elevation model, flow accumulation and direction rasters. http://nhd.usgs.gov/data.html.

NRCS. 2012. Soil Survey Geographic database (SSURGO). http://websoilsurvey.nrcs.usda.gov/.

Ogden, F. L., H. O. Sharif, S. U. S. Senarath, J. A. Smith, M. L. Baeck, and J. R. Richardson. 2000. Hydrologic analysis of the Fort Collins, Colorado, flash flood of 1997. Journal of Hydrology 228(1-2):82–100.

Ogden, F. L., and R. F. Stallard. 2013. Land use effects on ecosystem service provisioning in tropical watersheds, still an important unsolved problem. Proceedings of the National Academy of Sciences of the United States of America 110(52):E5037.

Ostroff, A., D. Wieferich, A. Cooper, and D. Infante. 2012. National Anthropogenic Barrier Dataset. https://www.sciencebase.gov/catalog/item/get/512cf142e4b0855fde669828.

Öztürk, M., N. K. Copty, and A. K. Saysel. 2013. Modeling the Impact of Land Use Change on the Hydrology of a Rural Watershed. Journal of Hydrology 497:97–109.

Paul, M. J., and J. L. Meyer. 2001. Streams in the urban landscape. Annual Review of Ecology and Systematics 32:333–365.

Petrow, T., and B. Merz. 2009. Trends in flood magnitude, frequency and seasonality in Germany in the period 1951–2002. Journal of Hydrology 371(1-4):129–141.

Poff, N. L., B. P. Bledsoe, and C. O. Cuhaciyan. 2006. Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. Geomorphology 79(3-4):264–285.

Poff, N. L., and J. . Ward. 1989. Implications of Streamflow Variability and Predictability for Lotic Community Structure: A regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805–1817.

Ponce, V. M., and R. H. Hawkins. 1996. Runoff curve number: Has it reached maturity? Journal of Hydrologic Engineering 1:11–19.

Postel, S. L., and B. H. Thompson. 2005. Watershed protection: Capturing the benefits of nature's water supply services. Natural Resources Forum 29(2):98–108.

Posthumus, H., J. R. Rouquette, J. Morris, D. J. G. Gowing, and T. M. Hess. 2010. A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. Ecological Economics 69(7):1510–1523.

PRISM Climate Group. 2013. Precipitation grids. http://prism.oregonstate.edu.

Radford, K. G., and P. James. 2012. Changes in the value of ecosystem services along a rural–urban gradient: A case study of Greater Manchester, UK. Landscape and Urban Planning 109(1):117–127.

Rallison, R. E. 1980. Origin and Evolution of the SCS Runoff Equation. Pages 912–924 Symposium on Watershed Management ASCE. ASCE, Volume II 912-924 New York NY.

Reyers, B., R. Biggs, G. S. Cumming, T. Elmqvist, A. P. Hejnowicz, and S. Polasky. 2013. Getting the measure of ecosystem services: a social–ecological approach. Frontiers in Ecology and the Environment 11(5):268–273.

Rice, K. C., and R. M. Hirsch. 2012. Spatial and temporal trends in runoff at long-term streamgages within and near the Chesapeake Bay Watershed: U.S. Geological Survey Scientific Investigations Report 2012–5151. Page 56 p.

Rose, S., and N. E. Peters. 2001. Effects of urbanization on streamflow in the Atlanta area (Georgia, USA): a comparative hydrological approach. Hydrological Processes 15:1441–1457.

Roy, A. H., S. J. Wenger, T. D. Fletcher, C. J. Walsh, A. R. Ladson, W. D. Shuster, H. W. Thurston, and R. R. Brown. 2008. Impediments and solutions to sustainable, watershed-scale urban stormwater management: lessons from Australia and the United States. Environmental Management 42(2):344–59.

Di Sabatino, A., L. Coscieme, P. Vignini, and B. Cicolani. 2013. Scale and ecological dependence of ecosystem services evaluation: Spatial extension and economic value of freshwater ecosystems in Italy. Ecological Indicators 32:259–263.

Sanford, W. E., and D. L. Selnick. 2013. Estimation of Evapotranspiration Across the Conterminous United States Using a Regression With Climate and Land-Cover Data. Journal of the American Water Resources Association 49(1):217–230.

Schueler, T. R., L. Fraley-Mcneal, and K. Cappiella. 2009. Is Impervious Cover Still Important? Review of Recent Research. Journal of Hydrologic Engineering 14(4):309–315.

Schulp, C. J. E. E., R. Alkemade, K. K. Goldewijk, K. Petz, and K. Klein Goldewijk. 2012. Mapping ecosystem functions and services in Eastern Europe using global-scale data sets. International Journal of Biodiversity Science, Ecosystem Services & Management 8(1-2):156–168.

Simonit, S., and C. Perrings. 2013. Bundling ecosystem services in the Panama Canal watershed. Proceedings of the National Academy of Sciences of the United States of America 110(23):9326–31.

Singh, R., G. Senay, N. Velpuri, S. Bohms, R. Scott, and J. Verdin. 2013. Actual Evapotranspiration (Water Use) Assessment of the Colorado River Basin at the Landsat Resolution Using the Operational Simplified Surface Energy Balance Model. Remote Sensing 6(1):233–256.

Smith, J. A., M. L. Baeck, J. E. Morrison, P. Sturdevant-Rees, D. F. Turner-Gillespie, and P. D. Bates. 2002a. The Regional Hydrology of Extreme Floods in an Urbanizing Drainage Basin. Journal of Hydrometeorology 3:267–282.

Smith, S. V., W. H. Renwick, J. D. Bartley, and R. W. Buddemeier. 2002b. Distribution and significance of small, artificial water bodies across the United States landscape. The Science of the Total Environment 299(1-3):21–36.

Strecker, E. W., M. M. Quigley, B. R. Urbonas, J. E. Jones, and J. K. Clary. 2001. Determining urban storm water BMP effectiveness. Journal of Water Resources Planning and Management 127(127):144–149.

Su, D., X. Fang, and Z. Fang. 2010. Effectiveness and Downstream Impacts of Stormwater Detention Ponds Required for Land Development. World Environmental and Water Resources Congress 2010:3071–3081.

Sullivan, A., J. . Ternan, and A. . Williams. 2004. Land use change and hydrological response in the Camel catchment, Cornwall. Applied Geography 24(2):119–137.

Sumner, D. A., and H. Boriss. 2006. Bee-conomics and the Leap in Pollination Fees. Agricultural and Resource Economics 9(3):9–11.

Sun, G., S. G. McNulty, J. Lu, D. M. Amatya, Y. Liang, and R. K. Kolka. 2005. Regional annual water yield from forest lands and its response to potential deforestation across the southeastern United States. Journal of Hydrology 308(1-4):258–268.

Thurston, H. W., H. C. Goddard, D. Szlag, and B. Lemberg. 2003. Controlling Storm-Water Runoff with Tradable Allowances for Impervious Surfaces. Journal of Water Resources Planning and Management (129):409–418.

Tingle, B. 1999. Executive Summary: Mecklenburg County Floodplain Mapping Summary Report. Pages 1–13 Floodplain Mapping. Charlotte, NC.

Tollan, A. 2002. Land use change and floods: what do we need most, research or management? Water Science & Technology 45(8):183–190.

USACE. 2012. National Inventory of Dams. http://geo.usace.army.mil/pgis/f?p=397:1:0.

USGS. 2012. National Hydrography Dataset Waterbodies. http://nhd.usgs.gov/data.html.

USGS. 2013. National Water Information System. http://nwis.waterdata.usgs.gov/nwis.

Villamagna, A. M., P. L. Angermeier, and E. M. Bennett. 2013. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity 15:114–121.

Villamagna, A. M., B. Mogollón, and P. L. Angermeier. 2014. A multi-indicator framework for mapping cultural ecosystem services: The case of freshwater recreational fishing. Ecological Indicators 45:255–265.

Zahran, S., S. D. Brody, W. G. Peacock, A. Vedlitz, and H. Grover. 2008. Social vulnerability and the natural and built environment: a model of flood casualties in Texas. Disasters 32(4):537–560.

Zhang, H., W. Ma, and X. Wang. 2008. Rapid Urbanization and Implications for Flood Risk Management in Hinterland of the Pearl River Delta, China: The Foshan Study. Sensors:2223–2239.

Tables and Figures

Table 3.1. Selected stream-flow gages in North Carolina (NC) along with gage identification number (ID), station name, drainage area, percentage forest cover from the 2011 National Land Cover Database (NLCD; includes deciduous, evergreen and mixed forests, woody and emergent herbaceous wetlands and scrub/shrubs), percentage urban cover from the 2011 NLCD (includes open space, low-, medium-, and high-intensity development), mean annual precipitation from 1991 to 2012 (PRISM Climate Group 2013), and percentages of sandy and loamy soils derived from estimating the A and B soil hydrologic groups from the Soil Survey Geographic Database (SSURGO).

Gage ID	Station Name	Drainage Area (km^2)	% Forested	% Urban	Precipitation (mm)	% Sandy and Loamy Soils
02146470	Little Hope Creek at Seneca Place in Charlotte	6.8	0%	100%	1018.2	73.5%
02146211	Irwin Creek at Statesville Avenue in Charlotte	15.5	19.55%	75.29%	1035.8	65.3%
0208732885	Marsh Creek near New Hope	17.7	4.49%	94.44%	1118.0	77.3%
02146700	McMullen Creek at Sharon View Road near Charlotte	18	3.07%	96.81%	1043.3	58.0%
02099000	East Fork Deep River near High Point	38.3	18.63%	67.76%	1047.9	49.6%
02142900	Long Creek near Paw Creek	42.5	22.75%	66.96%	1032.7	48.4%
0209741955	Northeast Creek at State Route 1100 near Genlee	54.6	34.16%	59.50%	1122.8	11.9%
02146300	Irwin Creek near Charlotte	79.5	8.74%	87.41%	1024.0	62.5%

Table 3.2. Indicators used to calculate the biophysical and technological flood regulation capacities. We standardized each indicator to take values 0-1, where values near 1 mean high capacity. Acronyms listed below refer to: National Land Cover Database (NLCD), Soil Survey Geographic Database (SSURGO), and digital elevation model (DEM). Dashes indicate that indicators do not change over time.

Category	Process	Indicator	Data Years	Standardization	Description
Biophysical	Evapotranspiration	Evapotranspiration Rate (ET)	1990, 1992, 2001, 2006, 2011	Standardize 0-1	Resample land cover 30-m to 10m-pixels, reclassify NLCD to ET rate (mm yr^{-1}) based on Table 3 in Singh et al. (2013)
	Water Retention	Available Water Storage (AWS)	-	Standardize 0-1	Available water volume stored for plant uptake in mm in the first 150cm of soil from SSURGO
	Soil Infiltration	Saturated Hydraulic Conductivity (Ksat)	-	Standardize 0-1	Saturated Hydraulic Conductivity in um s^{-1} from SSURGO
	Runoff	Runoff Curve Number (RCN)	1990, 1992, 2001, 2006, 2011	1 minus standardized 0-1 RCN	Intersection of SSURGO's soil hydrologic group (10-m resolution) and NLCD land cover (30-m resolution)
	Natural Water Storage	% Stream	-	Streams = 1, else = 0	Hydrography and Johnson and Fecko (2006) equation
	Surface flow	Slope	-	1 minus standardized 0-1 Slope	From 10-m resolution DEM, derived in degrees
Technological	Flow regulation	Best Management Practices (BMPs)	1965 - 2013	BMP = 1, else = 0	Wet and dry ponds, bioretention areas, stormwater wetlands, sand filters, and infiltration devices
		Artificial Water bodies (AWBs)	1944 - 2012	AWB = 1, else = 0	Farm ponds, golf ponds, and water supply reservoirs

Table 3.3. Parameter estimates (± one standard error) of generalized linear mixed models between selected biophysical and technological indicators and flood magnitude and duration (n=184). Acronyms listed below refer to: runoff curve number (RCN), evapotranspiration (ET), available water storage (AWS), saturated hydraulic conductivity (Ksat), artificial water bodies (AWB) and best management practices (BMP). Gray font indicates the bounds around the mean include zero, and black font indicates the bounds around the mean do not include zero.

Component	Indicator	Magnitude ($m^3/s/km^2$)	Duration (days)
Biophysical	RCN	0.04 ± 0.05	0.14 ± 0.04
	Slope	0.82 ± 0.99	0.01 ± 1.36
	% Stream	1.64 ± 1.01	1.94 ± 1.38
	ET	0.002 ± 0.002	0.01 ± 0.003
	AWS	0.06 ± 0.17	0.43 ± 0.17
	Ksat	-0.003 ± 0.03	0.06 ± 0.03
Technological	% AWB	-0.17 ± 0.74	-0.33 ± 1
	% BMP	-0.08 ± 0.27	-0.35 ± 0.38

Table 3.4. Unscaled and scaled importance-values for selected biophysical and technological indicators of flood magnitude and duration. Importance-values were scaled by dividing the indicators within each component by the largest importance-value, such that the maximum of the scaled importance-values is 1 for B1, B2 and technological components. Acronyms listed below refer to: runoff curve number (RCN), evapotranspiration (ET), available water storage (AWS), saturated hydraulic conductivity (Ksat), artificial water bodies (AWB) and best management practices (BMP).

Components		**Indicators**	**Magnitude**		**Duration**	
			Unscaled	**Scaled**	**Unscaled**	**Scaled**
Biophysical	B1	RCN	0.38	0.59	0.98	1.00
		Slope	0.45	0.70	0.46	0.47
		% Stream	0.64	1.00	0.28	0.29
	B2	ET	0.37	0.62	0.95	1.00
		Ksat	0.32	0.54	0.36	0.38
		AWS	0.30	0.50	0.51	0.54
		Slope	0.42	0.70	0.31	0.33
		% Stream	0.59	1.00	0.57	0.60
Technological		% BMP	0.56	0.99	0.49	0.76
		% AWB	0.57	1.00	0.64	1.00

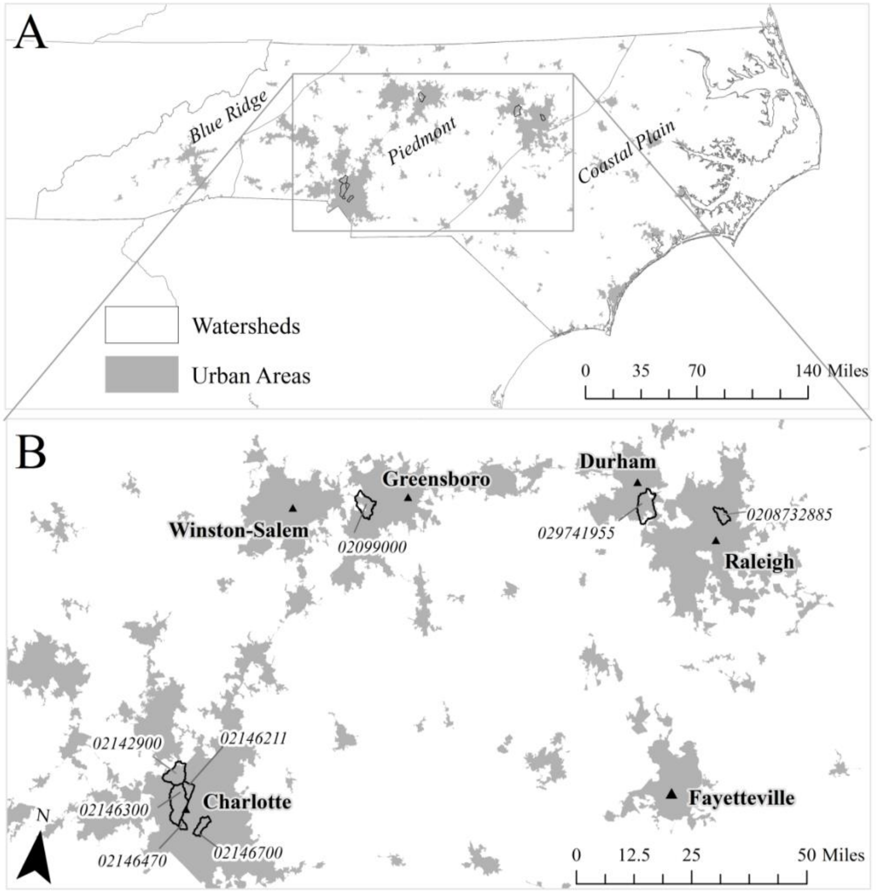


Figure 3.1. Map illustrating (A) the location of the eight gaged urban piedmont watersheds in North Carolina and (B) the proximity of the watersheds to major cities in bold. In panel B, the numbers in italics by each watershed represent the identification number of the stream gage.

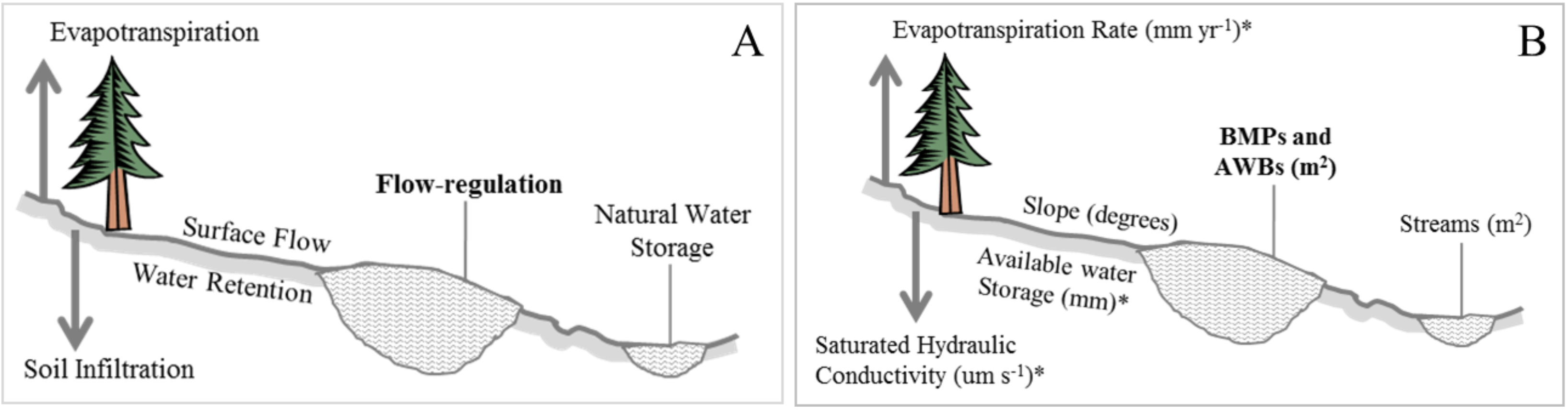


Figure 3.2. Landscape processes that regulate floods (A), and the indicators we derived from publicly available data to represent those processes (B). The bold labels in both panels represent technological processes and indicators; others represent biophysical processes and indicators. In panel B, the indicators with an asterisk are components of the runoff curve number (i.e., if water is not evapotranspired, retained or infiltrated, it runs off).

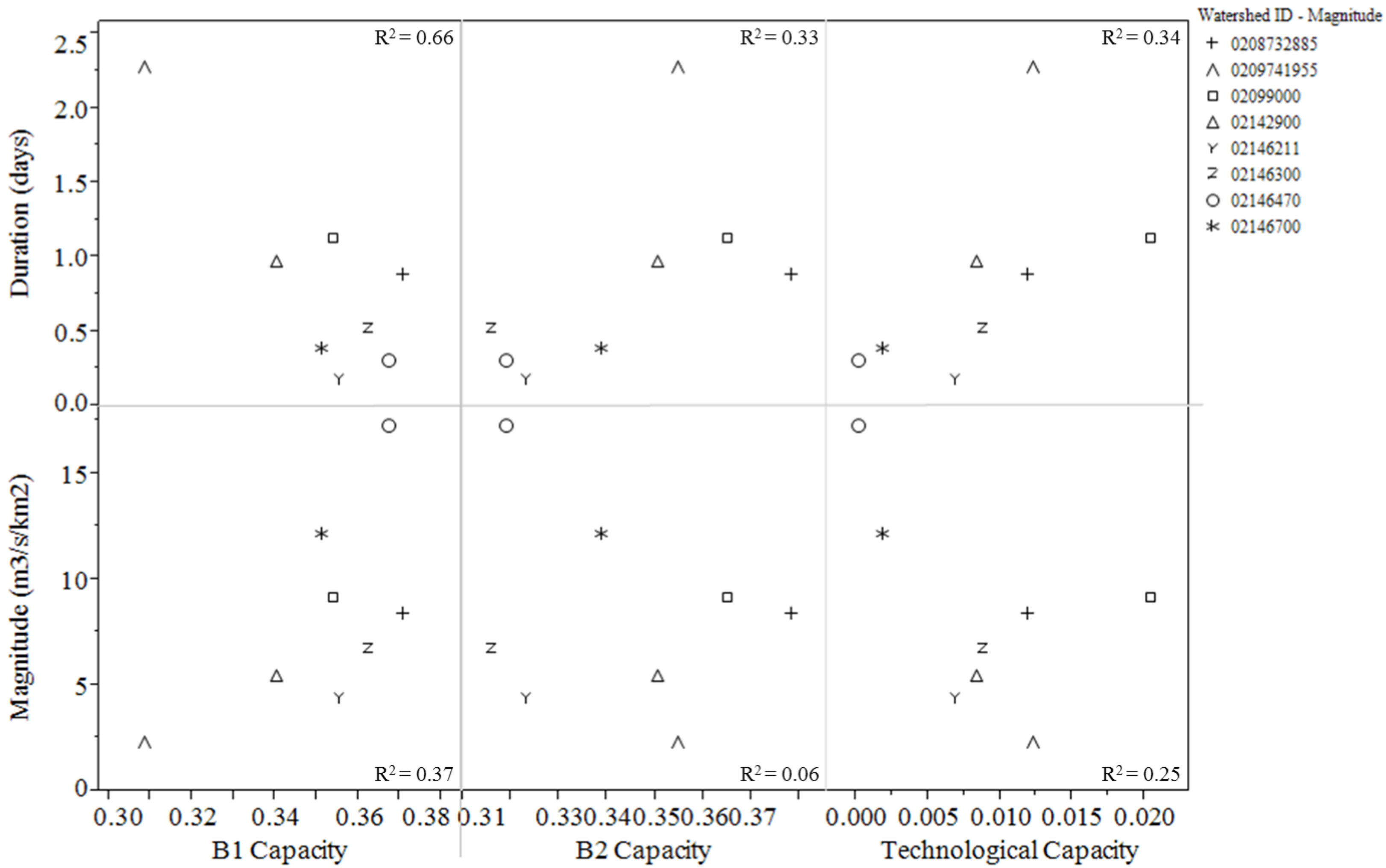


Figure 3.3. Scatterplots showing relations between flood response metrics (mean magnitude and duration from 1991 to 2013) and mean B1, B2 and technological capacity values based on 2011-2013 data for eight watersheds. Plots reflect flood magnitude-derived importance values.

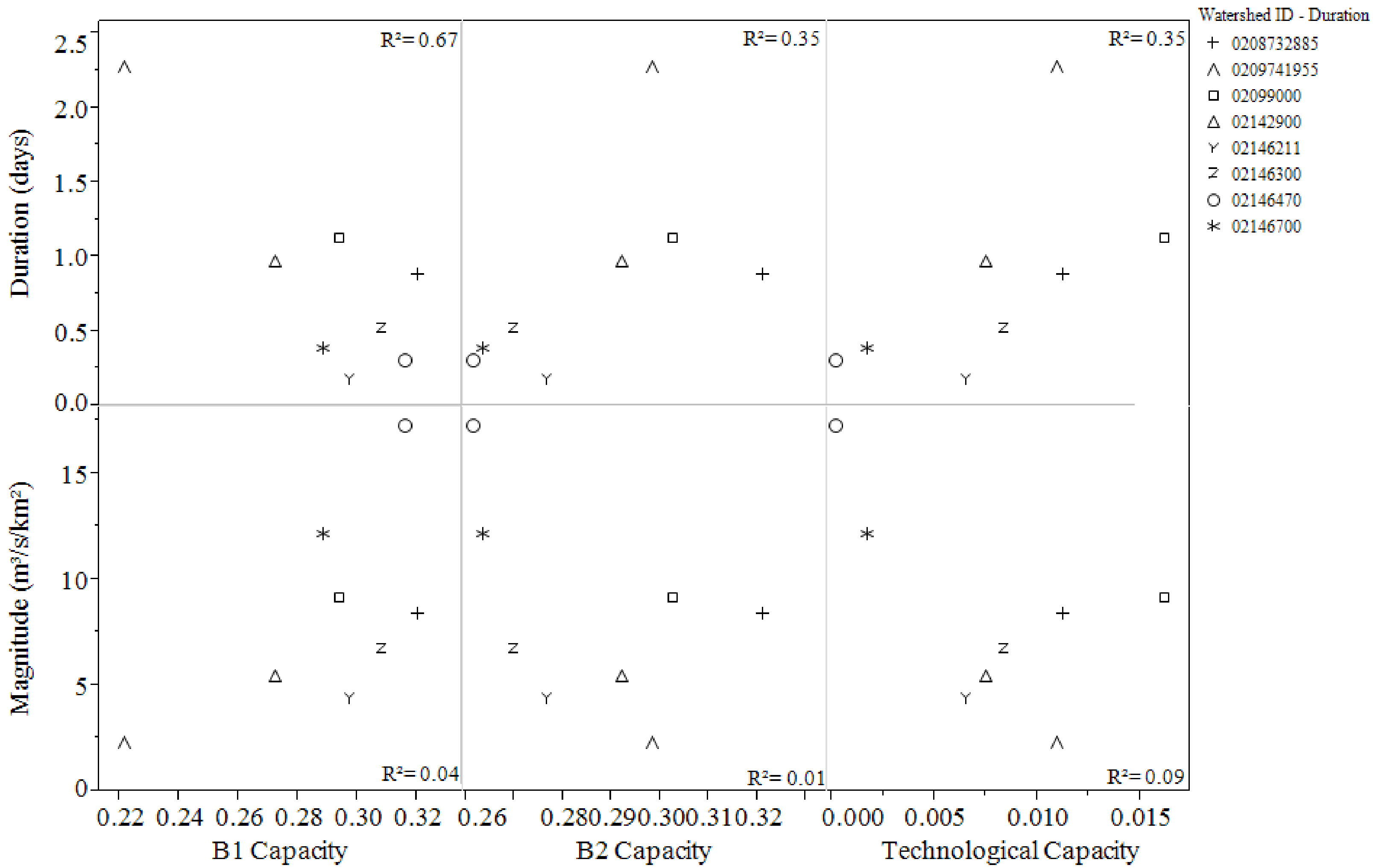


Figure 3.4. Scatterplot showing flood response metrics (mean magnitude and duration from 1991 to 2013) and mean B1, B2 and technological capacity values based on 2011-2013 data for each watershed based on the flood duration-derived importance values.

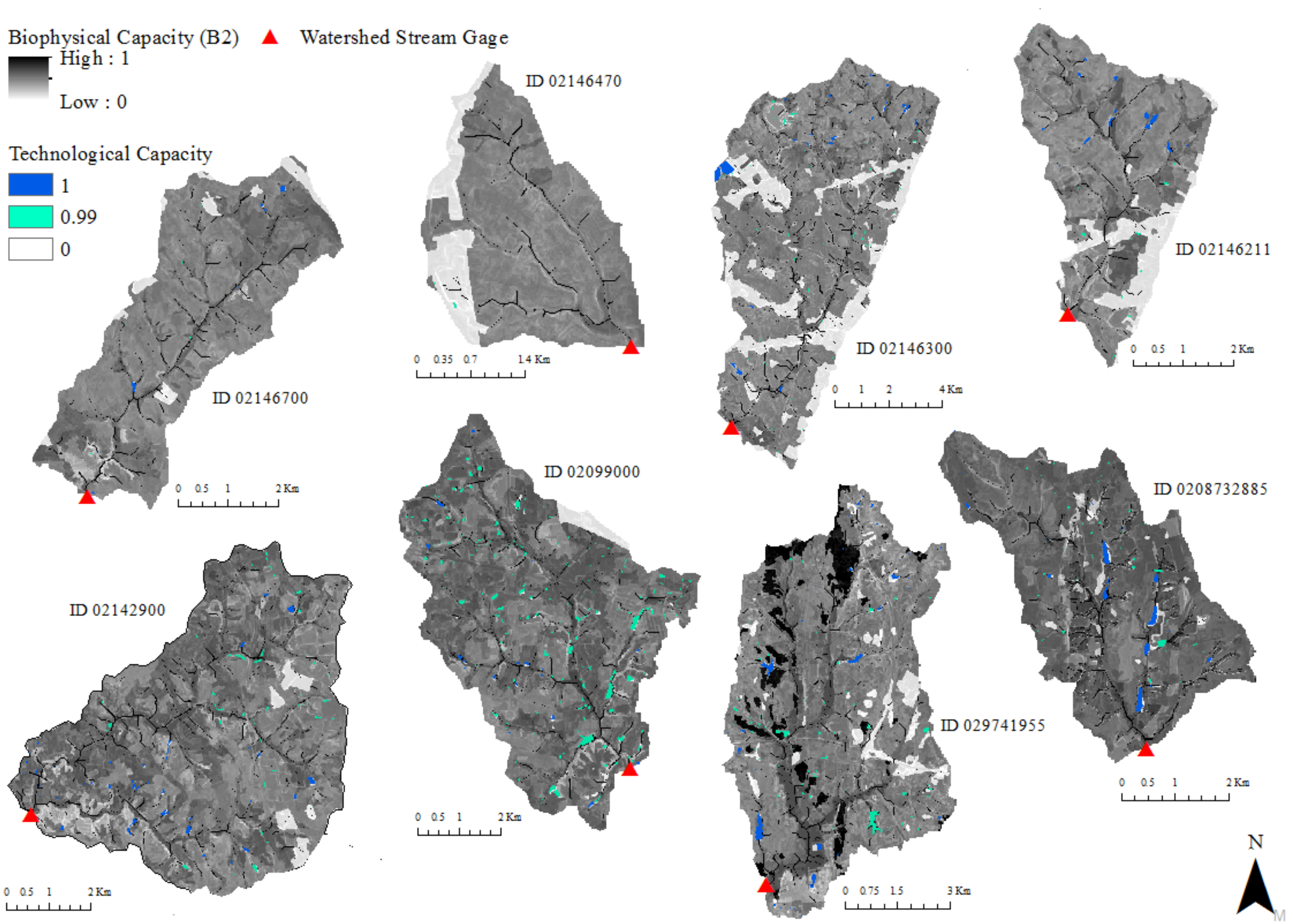


Figure 3.5. Maps showing the spatial distribution of flood magnitude-derived importance-values for B2 and technological capacity for eight urban watersheds in North Carolina, labeled with their stream gage identification number (ID).

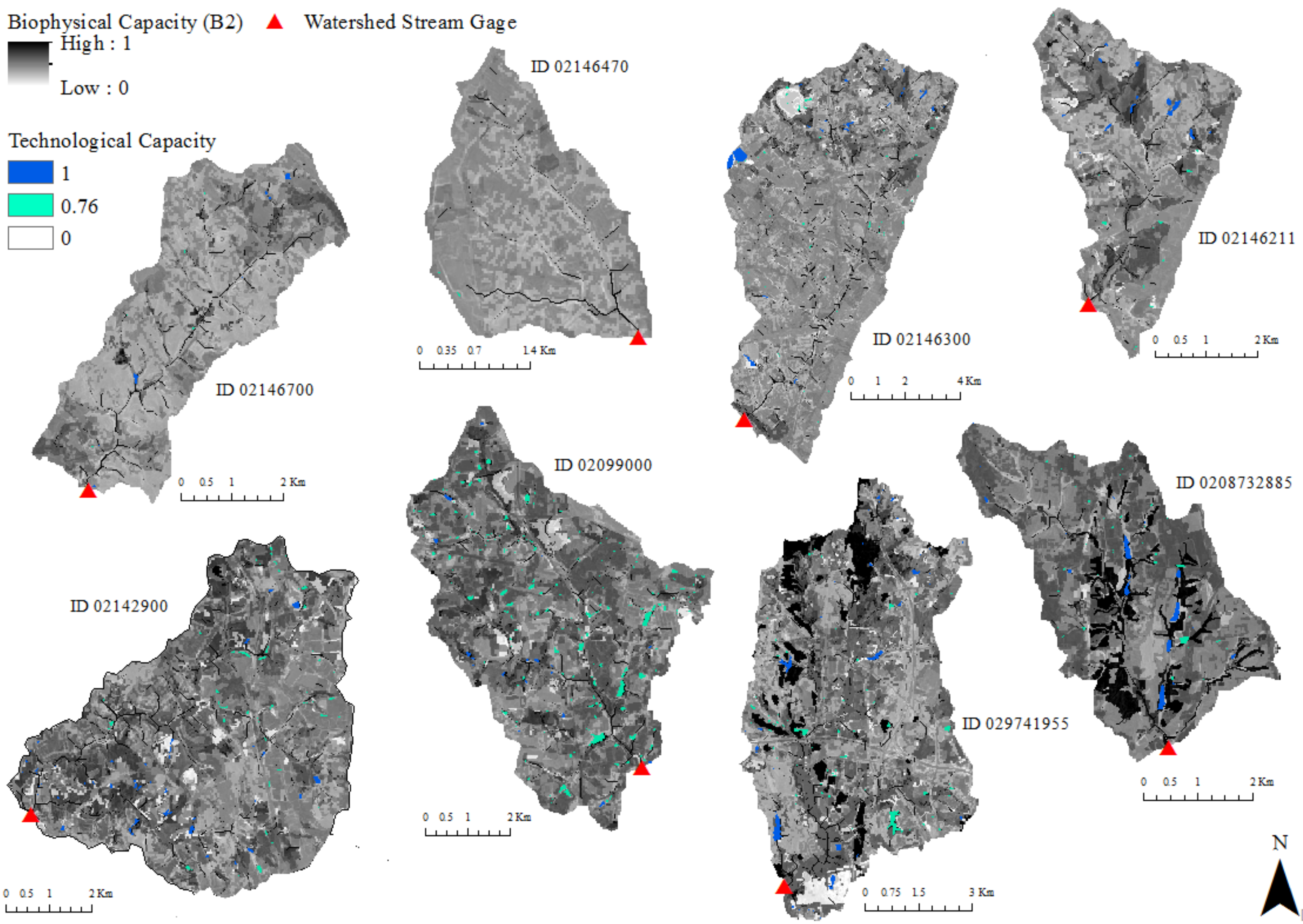


Figure 3.6. Maps showing the spatial distribution of flood duration-derived importance-values for B2 and technological capacity for eight urban watersheds in North Carolina, labeled with their stream gage identification number (ID).

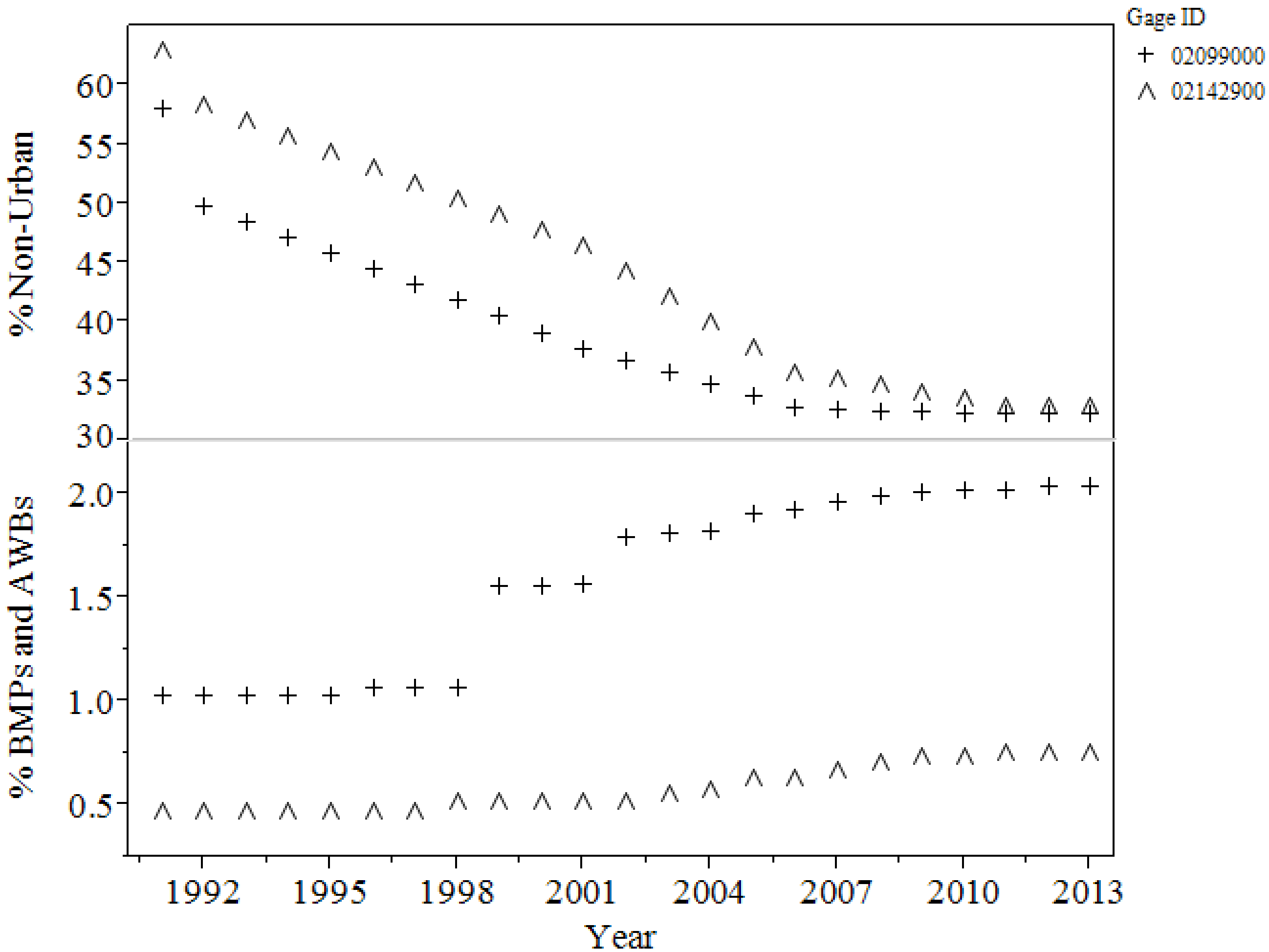


Figure 3.7. Plots illustrating the decrease in the percentage of non-urban land cover (forests and agriculture) and increase in the areal percentage of best management practices (BMPs) and artificial water bodies (AWBs) for two watersheds labeled with their gage identification number (ID) from 1991 to 2013.

General Conclusion

I selected this thesis project to better understand the importance of natural ecosystems in regulating floods, and the effectiveness of human interventions in curtailing flooding. The scope was sufficiently broad because findings are generalizable to other places and replicable with the appropriate data, but narrow enough that findings advanced the science and management of flood regulation. The overall goals of my master's thesis were to improve scientific understanding of (1) the types of floods that the natural landscape and human intervention can control, (2) how, where and why changes in the landscape have affected floods over time, and (3) how interactions among biophysical and technological features of watersheds, along with long-term hydrologic data, can be integrated to map the capacity of landscapes to regulate inland floods. Below, I expand on my main findings and discuss their broader implications.

The capacity of watersheds to regulate floods

Studies examining whether human development caused high-consequence floods, like the 1996 flood from Hurricane Fran or the 1999 North Carolina flood, found no convincing evidence (Sturdevant-Rees et al. 2001, Lecce and Kotecki 2008). In comparing flood return periods across watershed types, I found that urban watersheds had higher-peaked and shorter floods up to the 10-year flood compared to non-urban watersheds. Beyond the 10-year flood, all watersheds responded the same. I found no differences among watershed types for flood count, perhaps because this metric is driven by precipitation patterns. My results are consistent with other studies that found the greatest hydrologic response difference between natural and transformed landscapes is at small flood return periods (Leopold 1968; Hollis 1975; Smith et al. 2002; Wissmar et al. 2004). In addition, the greatest differences among watershed types have been in the magnitude and duration of floods, as we found no differences in flood count across watershed types.

A common management response to decrease the impact of high flows and floods in urban areas is to build flood control structures, yet mixed evidence on their effectiveness raise important questions about current flood control strategies (Booth et al. 2002, Thurston et al. 2003, Roy et al. 2008). In comparing forested watersheds to urban watersheds with low, medium and high percentages of flow-regulating features, I found that forested watersheds have lower-peaked and longer floods than urban watersheds, yet urban watersheds with a high percentage of flow-regulating features did have lower-peaked and longer floods than those with a low percentage of flow-regulating features. My results suggest that urban stormwater management to date has not completely counteracted the effects of expanding impervious surfaces on flood magnitude and duration.

Temporal trends in precipitation, floods and stream flashiness

Unlike national trends, I found decreasing trends in precipitation and flood magnitude, duration and count across selected watersheds, which is consistent with other regional studies (Patterson et al. 2012). In contrast, I found increasing trends in stream flashiness. In exploring each watershed's flashiness trend, I found that 20 showed stable stream flashiness, while five increased and six decreased in flashiness. Urban watersheds were among those that increased and decreased in flashiness. Watersheds that increased in stream flashiness gained more urban cover, lost more forested cover and had fewer best management practices (e.g., detention ponds, flood control reservoirs) installed than urban watersheds that decreased in stream flashiness. My results show that if land cover conversion is not met with greater effort to manage surface runoff, streams will become flashier.

Results contradict the flood frequency analysis assumption of stationarity, as I found that both precipitation and landscapes are dynamic, and change is spatially heterogeneous. This assumption has been questioned since the 1960s in light of population growth and climate change (Mandelbrot and W 1968, Milly et al. 2008, Rootzén and Katz 2013), which has implications for the design and risk of water-related infrastructure and policy. In relation to the stationarity assumption, I struggled with defining a flood. If climate and land cover have been changing, the rating curve to derive any flood return interval is bound to change. Subsequent measures of flood count, magnitude and duration are based on the return interval. Yet, the conventional protocol is to use the longest annual discharge record, excluding the unregulated years if it is regulated (i.e., the USGS denotes the years with a "known effect of regulation or urbanization"), to generate a rating curve and derive the return intervals of interest. If I had used the unregulated records in addition to the regulated records (i.e., violating the stationarity assumption), I suspect that differences between urban and non-urban watersheds would have been even greater. For managers, this implies that constructing a pre-development hydrograph would require a greater effort (e.g., implementing more flow-regulating features or low-impact development mechanisms to lower high-peaked and lengthen short floods), as the differences between urban and non-urban watersheds found herein are mediated by using a post-development rating curve in urban watersheds which reduces the differences among watersheds.

Mapping the capacity of watersheds to regulate floods

Developing a methodology to characterize flood regulation capacity based on technological and biophysical features in addition to long-term hydrologic records improves on previous efforts to map this service. I was able to derive importance-values for selected landscape indicators based on long-term hydrologic records, where the percentage of watersheds covered by streams ranked highest for flood magnitude and land cover-derived indicators (i.e., evapotranspiration rate and runoff curve number) ranked highest for flood duration. I found that biophysical and technological capacities were negatively related to flood magnitude and positively related to flood duration, where watersheds with high capacities regulated floods. The main utility of mapping biophysical and technological flood regulation capacities is that when done at high spatial resolution (e.g., 10-m pixels) it enables watershed coordinators and planners to identify the areas of relatively low and high capacity (Nedkov and Burkhard 2012) so they can more effectively implement and prioritize stormwater management programs.

The runoff curve number (RCN) approach yielded results opposite that expected, possibly because it confounds soil and land cover processes, while the alternative approach (e.g., having an indicator for evapotranspiration, retention and infiltration) coherently separates these processes. However, I struggled to understand if my results were anomalies associated with my study watersheds, or whether the same results would be expected in non-urban watersheds. In urban watersheds, where impervious surfaces are common, the soil plays a minor role in infiltrating and retaining surface runoff. However, in non-urban watersheds, both soil and land cover play important roles. Further studies exploring the utility of RCN methodology in non-urban watersheds is warranted to clarify the circumstances in which results of the RCN methodology are instructive.

Broader implications

The goal throughout my thesis is to broadly understand what controls water quantity. In the context of urban and suburban development, stormwater management has mainly focused on complying with the Clean Water Act by addressing water quality, and not necessarily quantity. Water quantity – and flooding – is driven mainly by precipitation which managers cannot control, while water quality is characterized by rules and regulations that managers can control, enforce and monitor. The majority of stormwater best management practices (e.g., bioretention, stormwater wetlands, wet detention basins, filter strips, grassed swales, restored riparian buffer) are designed to improve water quality, whereas water quantity control is perceived as a side benefit (NCDENR 2007). A search on the Web of Science of water quality and water quantity, where each is accompanied by "stormwater management" and "best management practices", yields 247 articles for water quality, and 46 for water quantity.

Yet, many water quality issues are directly related to and affected by water quantity issues. For example, in urban areas, surface runoff mobilizes pollutants and sediments to streams. Flashy flows erode stream banks and impede riparian areas to vegetate. By not decreasing the velocity and quantity of surface runoff, water quality issues might persist (Roni et al. 2002). So, one might ask "Is managing water quantity more effective than managing for water quality?" Increasing the infiltration of surface runoff has clear benefits to both water quality and quantity (Hester and Gooseff 2011). New trends in urban stormwater management focus on increasing infiltration through low-impact development (e.g., green roofs, permeable pavement, rain gardens) which preliminary studies have found to be effective (Czemiel Berndtsson 2010, Damodaram et al. 2010, EPA 2013). My thesis is timely in terms of these new approaches that can provide watershed coordinators, stormwater managers and the Environmental Protection Agency with insight into what type of floods landscape structure can manage, how changes in landscape structure (e.g., changes in land cover and flow-regulating features) have affected small floods and streamflow flashiness, which hydrologic metrics can be useful when assessing the efficacy of management strategies, and how both technological and biophysical features that regulate floods are distributed across the landscape.

References

Andreadis, K. M., and D. P. Lettenmaier. 2006. Trends in 20th century drought over the continental United States. Geophysical Research Letters 33(10):1–4.

Arrigoni, A. S., M. C. Greenwood, and J. N. Moore. 2010. Relative impact of anthropogenic modifications versus climate change on the natural flow regimes of rivers in the Northern Rocky Mountains, United States. Water Resources Research 46(12):1–16.

Black, P. E. 2012. The U.S. Flood Control Program at 75: Environmental Issues. Journal of the American Water Resources Association 48(2):244–255.

Bloschl, G., S. Ardoin-Bardin, M. Bonell, M. Dorninger, D. Goodrich, D. Gutknecht, D. Matamoros, B. Merz, P. Shand, and J. Szolgay. 2007. At what scales do climate variability and land cover change impact on flooding and low flows? Hydrological Processes 1247(January):1241–1247.

Booth, D. B., D. Hartley, and R. Jackson. 2002. Forest cover, impervious-surface area, and the mitigation of stormwater impacts. Journal of the American Water Resources Association 38(3):835–947.

Czemiel Berndtsson, J. 2010. Green roof performance towards management of runoff water quantity and quality: A review. Ecological Engineering 36(4):351–360.

Damodaram, C., M. H. Giacomoni, C. Prakash Khedun, H. Holmes, A. Ryan, W. Saour, and E. M. Zechman. 2010. Simulation of Combined Best Management Practices and Low Impact Development for

Sustainable Stormwater Management. Journal of the American Water Resources Association 46(5):907–918.

Deasy, C., A. Titman, and J. N. Quinton. 2014. Measurement of flood peak effects as a result of soil and land management, with focus on experimental issues and scale. Journal of Environmental Management 132:304–312.

Downing, J. A., Y. T. Prairie, J. J. Cole, C. M. Duarte, L. J. Tranvik, R. G. Striegl, W. H. McDowell, P. Kortelainen, N. F. Caraco, and J. M. Melack. 2006. The global abundance and size distribution of lakes, ponds, and impoundments. Limnology and Oceanography 51(5):2388–2397.

Eng, K., D. M. Wolock, and D. M. Carlisle. 2013. River flow changes related to land and water management practices across the conterminous United States. The Science of the Total Environment 463-464:414–22.

EPA. 2013. Proposed National Rulemaking to Strengthen the Stormwater Program. Alexandria, VA. http://cfpub.epa.gov/npdes/stormwater/rulemaking.cfm.

FEMA. 2014. Disaster Declarations. http://www.fema.gov/disasters.

Findlay, S. J., and M. P. Taylor. 2006. Why rehabilitate urban river systems? Area 38(3):312–325.

Hawley, R. J., and B. P. Bledsoe. 2011. How do flow peaks and durations change in suburbanizing semi-arid watersheds? A southern California case study. Journal of Hydrology 405(1-2):69–82.

Hester, E. T., and M. N. Gooseff. 2011. Hyporheic restoration in streams and rivers. Pages 167–187 *in* A. Simon, S. J. Bennett, and J. M. Castro, editors. Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses, and Tools. Geophysical Monograph Series, Washington, D. C.

Hollis, G. E. 1975. The effect of urbanization on floods of different recurrence interval. Water Resources Research 11(3):431–435.

Ignatius, A., and J. Jones. 2014. Small reservoir distribution, rate of construction, and uses in the upper and middle Chattahoochee basins of the Georgia piedmont, USA, 1950–2010. International Journal of Geo-Information 3(2):460–480.

Kundzewicz, Z. W. 1999. Flood protection — sustainability Issues. Hydrological Sciences Journal 44(4):559–571.

Lecce, S. A., and E. S. Kotecki. 2008. The 1999 Flood of the Century in Eastern North Carolina: Extraordinary Hydro-Meterological Event or Human-Induced Catastrophe? Physical Geography 29(2):101–120.

Magilligan, F. J., and M. L. Stamp. 1997. Historical land-cover changes and hydrogeomorphic adjustment in a small Georgia watershed. Annals of the Association of American Geographers 87(4):614–635.

Mandelbrot, B. B., and J. R. W. 1968. Noah, Joseph and operational hydrology. Water Resources Research 4(5):909–918.

Milly, P. C. D., J. Betancourt, M. Falkenmark, R. M. Hirsch, Z. W. Kundzewicz, D. P. Lettenmaier, and R. J. Stouffer. 2008. Climate change. Stationarity is dead: whither water management? Science 319(5863):573–4.

Milly, P. C. D., R. T. Wetherald, K. A. Dunne, and T. L. Delworth. 2002. Increasing risk of great floods in a changing climate. Nature 415(6871):514–7.

NCDENR. 2007. Stormwater Best Management Practices Manual. Raleigh, North Carolina.

Nedkov, S., and B. Burkhard. 2012. Flood regulating ecosystem services — Mapping supply and demand, in the Etropole municipality, Bulgaria. Ecological Indicators 21:67–79.

Patterson, L. A., B. Lutz, and M. W. Doyle. 2012. Streamflow Changes in the South Atlantic, United States During the Mid- and Late 20th Century. Journal of the American Water Resources Association 48(6):1126–1138.

Pielke, R. A., M. W. Downton, and J. Z. B. Miller. 2002. Flood damage in the United States, 1926 – 2000: A reanalysis of National Weather Service estimates. Pages 1–79. Boulder, Colorado.

Posthumus, H., J. R. Rouquette, J. Morris, D. J. G. Gowing, and T. M. Hess. 2010. A framework for the assessment of ecosystem goods and services; a case study on lowland floodplains in England. Ecological Economics 69(7):1510–1523.

Roni, P., T. J. Beechie, R. E. Bilby, F. E. Leonetti, M. M. Pollock, and G. R. Pess. 2002. A Review of Stream Restoration Techniques and a Hierarchical Strategy for Prioritizing Restoration in Pacific Northwest Watersheds. North American Journal of Fisheries Management 22(22):1–20.

Rootzén, H., and R. W. Katz. 2013. Design life level: Quantifying risk in a changing climate. Water Resources Research 49(9):5964–5972.

Roy, A. H., S. J. Wenger, T. D. Fletcher, C. J. Walsh, A. R. Ladson, W. D. Shuster, H. W. Thurston, and R. R. Brown. 2008. Impediments and solutions to sustainable, watershed-scale urban stormwater management: lessons from Australia and the United States. Environmental Management 42(2):344–59.

Schulp, C. J. E. E., R. Alkemade, K. K. Goldewijk, K. Petz, and K. Klein Goldewijk. 2012. Mapping ecosystem functions and services in Eastern Europe using global-scale data sets. International Journal of Biodiversity Science, Ecosystem Services & Management 8(1-2):156–168.

Smith, J. A., M. L. Baeck, J. E. Morrison, P. Sturdevant-Rees, D. F. Turner-Gillespie, and P. D. Bates. 2002a. The Regional Hydrology of Extreme Floods in an Urbanizing Drainage Basin. Journal of Hydrometeorology 3:267–282.

Smith, S. V., W. H. Renwick, J. D. Bartley, and R. W. Buddemeier. 2002b. Distribution and significance of small, artificial water bodies across the United States landscape. The Science of the Total Environment 299(1-3):21–36.

Sordo-Ward, A., L. Garrote, F. Martín-Carrasco, and M. Dolores Bejarano. 2012. Extreme flood abatement in large dams with fixed-crest spillways. Journal of Hydrology 466-467:60–72.

Sturdevant-Rees, P., J. A. Smith, J. Morrison, and M. L. Baeck. 2001. Tropical storms and the flood hydrology of the central Appalachians. Water Resources Research 37(8):2143–2168.

Sutherland, W. J., R. Aveling, T. M. Brooks, M. Clout, L. V. Dicks, L. Fellman, E. Fleishman, D. W. Gibbons, B. Keim, F. Lickorish, K. a. Monk, D. Mortimer, L. S. Peck, J. Pretty, J. Rockström, J. P. Rodríguez, R. K. Smith, M. D. Spalding, F. H. Tonneijck, and A. R. Watkinson. 2013. A horizon scan of global conservation issues for 2014. Trends in Ecology & Evolution (2013):1–8.

Thurston, H. W., H. C. Goddard, D. Szlag, and B. Lemberg. 2003. Controlling Storm-Water Runoff with Tradable Allowances for Impervious Surfaces. Journal of Water Resources Planning and Management (129):409–418.

Tollan, A. 2002. Land use change and floods: what do we need most, research or management? Water Science & Technology 45(8):183–190.

Tran, P., F. Marincioni, and R. Shaw. 2010. Catastrophic flood and forest cover change in the Huong river basin, central Viet Nam: a gap between common perceptions and facts. Journal of Environmental Management 91(11):2186–200.

Villamagna, A. M., P. L. Angermeier, and E. M. Bennett. 2013. Capacity, pressure, demand, and flow: A conceptual framework for analyzing ecosystem service provision and delivery. Ecological Complexity 15:114–121.

Wang, D., and M. Hejazi. 2011. Quantifying the relative contribution of the climate and direct human impacts on mean annual streamflow in the contiguous United States. Water Resources Research 47(10):1–16.

Ye, B., D. Yang, and D. L. Kane. 2003. Changes in Lena River streamflow hydrology: Human impacts versus natural variations. Water Resources Research 39(7):1–14.

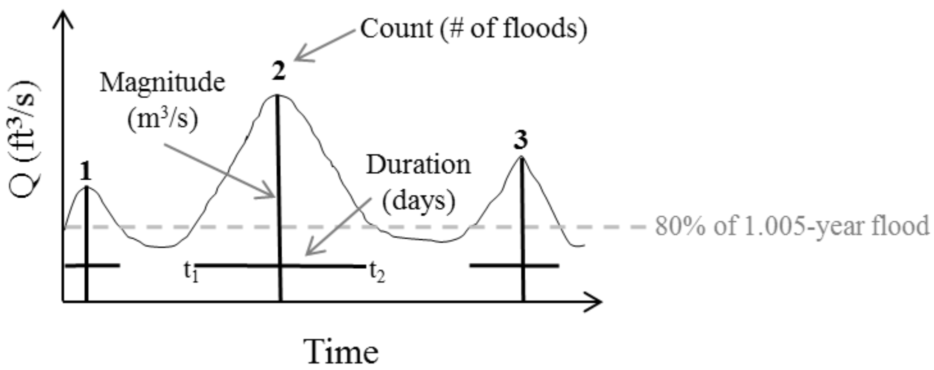
Appendices

Appendix A. Selected stream-flow gages in North Carolina (NC) and Virginia (VA), along with gage identification number (ID), station name, state, drainage area, and physiographic province.

Gage ID	Station Name	State	Drainage Area (km^2)	Province
0208650112	Flat River Tributary near Willardville	NC	3.0	Piedmont
02146470	Little Hope Creek at Seneca Place at Charlotte	NC	6.8	Piedmont
03450000	Beetree Creek near Swannanoa	NC	14.1	Blue Ridge
02146211	Irwin Creek at Statesville Avenue at Charlotte	NC	15.5	Piedmont
0208732885	Marsh Creek near New Hope	NC	17.7	Piedmont
02146700	McMullen Creek at Sharon View Road near Charlotte	NC	18.0	Piedmont
0214253830	Norwood Creek near Troutman	NC	18.6	Piedmont
0210166029	Rocky River at State Route 1300 near Crutchfield Crossroads	NC	19.2	Piedmont
02096846	Cane Creek near Orange Grove	NC	19.5	Piedmont
01658500	South Fork Quantico Creek near Independent Hill	VA	19.7	Piedmont
02102908	Flat Creek near Inverness	NC	19.8	Coastal Plain
02097464	Morgan Creek near White Cross	NC	21.6	Piedmont
02038850	Holiday Creek near Andersonville	VA	22.1	Piedmont
02055100	Tinker Creek near Daleville	VA	30.3	Valley and Ridge
0344894205	North Fork Swannanoa River near Walkertown	NC	37.6	Blue Ridge
02099000	East Fork Deep River near High Point	NC	38.3	Piedmont
01613900	Hogue Creek near Hayfield	VA	41.2	Valley and Ridge
02142900	Long Creek near Paw Creek	NC	42.5	Piedmont
01620500	North River near Stokesville	VA	44.8	Valley and Ridge
02093800	Reedy Fork near Oak Ridge	NC	53.4	Piedmont
0209741955	Northeast Creek at State Route 1100 near Genlee	NC	54.6	Piedmont
02036500	Fine Creek at Fine Creek Mills	VA	58.0	Piedmont
01654000	Accotink Creek near Annandale	VA	61.9	Piedmont
01673550	Totopotomoy Creek near Studley	VA	66.0	Coastal Plain
02143040	Jacob Fork at Ramsey	NC	66.6	Piedmont

01662800	Battle Run near Laurel Mills	VA	66.8	Blue Ridge
03478400	Beaver Creek at Bristol	VA	69.7	Valley and Ridge
03455500	West Fork Pigeon River above Lake Logan near Hazelwood	NC	71.5	Blue Ridge
02142000	Lower Little River near All Healing Springs	NC	73.0	Piedmont
02111000	Yadkin River at Patterson	NC	74.6	Blue Ridge
02146300	Irwin Creek near Charlotte	NC	79.5	Piedmont

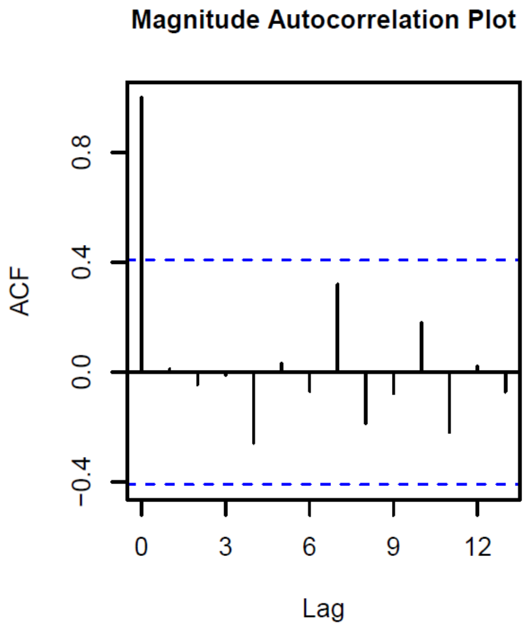
Appendix B. Schematic definition of flood magnitude, flood count and flood duration. A given discharge (Q) was considered a flood if it exceeded 80% of the discharge that recurs at a 1.005-year interval (referred to as 1-year flood). If $t_2 - t_1$ is less than 24 hours, peaks within that period were are collectively considered a single flood.

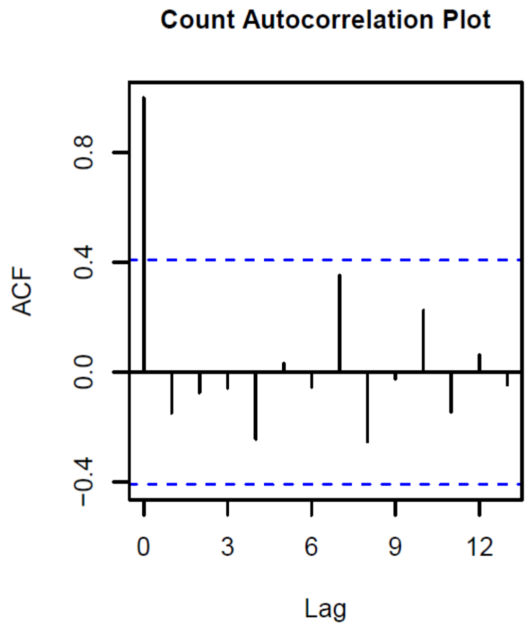


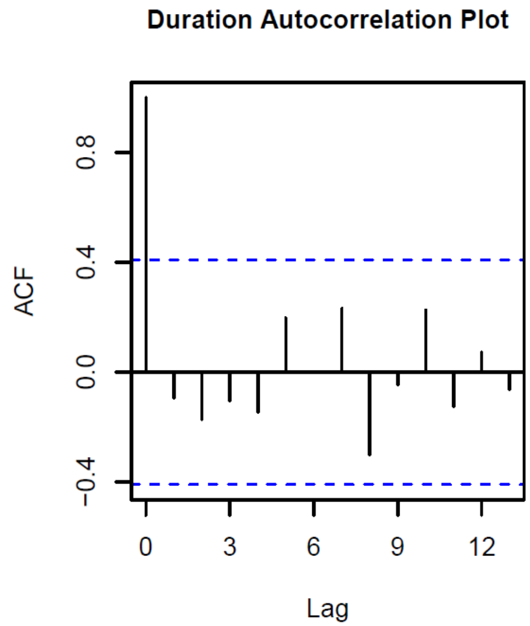
Appendix C. Reclassified land cover types based on the classification of five land cover datasets from four time periods. LULC is land use and land cover; NLCD is national land cover database.

Reclassified Land Cover	1992-Enhanced	1992-Retrofit	2001-Retrofit	2006 and 2011
Forest	Deciduous Forest Evergreen Forest Mixed Forest Woody Wetlands Emergent Herbaceous Wetlands	Deciduous Forest Evergreen Forest Mixed Forest Woody Wetlands Emergent Herbaceous Wetlands	Forest Wetland	Deciduous Forest Evergreen Forest Mixed Forest Woody Wetlands Emergent Herbaceous Wetlands Shrub/Scrub
Urban	Low Intensity Residential High Intensity Residential Commercial/Industrial/Transportation LULC Residential NLCD/LULC Forested Residential Urban/Recreational Grasses	Low Intensity Residential High Intensity Residential Commercial/Industrial/ Transportation Urban/Recreational Grasses	Urban	Developed, Open Space Developed, Low Intensity Developed, Medium Intensity Developed, High Intensity
Agriculture	Pasture/Hay Row Crops LULC Orchards/Vineyards/other	Pasture/Hay Row Crops	Agriculture Grass/Shrub	Grassland/Herbaceous Pasture/Hay Cultivated Crops

Appendix D. Temporal autocorrelation plots for flood magnitude, count, and duration. When strong autocorrelation is present the autocorrelation covariance function (ACF) between previous time periods with lag, k, is large and consistently outside the dashed bands, which approximate confidence interval for the null hypothesis of no correlation. In our case the correlations attenuate quickly; information from one year provides little knowledge about the next year.



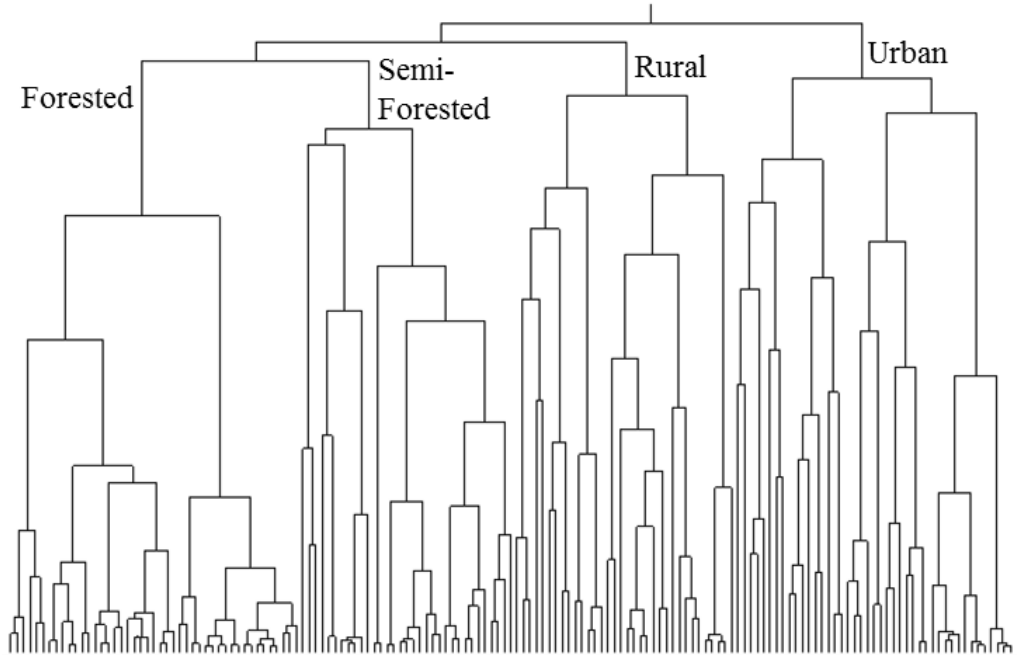




Appendix E. Selected distributions and their over-dispersion parameters (in parenthesis) for nine flood return periods for flood duration, count and magnitude. All but one of the over-dispersion parameters ($\hat{c}$) for the distributions were within the range ($\hat{c} \leq 1$) indicating that the flood metrics are a good fit for the given distribution.

Return Period	Duration	Count	Magnitude
80% Q1	Poisson (0.44)	Lognormal (0.28)	Negative Binomial (0.63)
Q1	Lognormal (0.77)	Lognormal (0.33)	Negative Binomial (0.79)
Q1.5	Lognormal (0.54)	Negative Binomial (0.95)	Negative Binomial (0.28)
Q2	Lognormal (0.60)	Lognormal (0.18)	Negative Binomial (0.27)
Q5	Negative Binomial (0.64)	Negative Binomial (0.99)	Negative Binomial (0.26)
Q10	Negative Binomial (0.87)	Poisson (0.98)	Negative Binomial (0.21)
Q20	Negative Binomial (0.40)	Negative Binomial (0.98)	Negative Binomial (0.22)
Q50	Negative Binomial (0.14)	Negative Binomial (0.33)	Negative Binomial (0.16)
≥Q100	Negative Binomial (0.09)	Poisson (1.0)	Poisson (5.35)

Appendix F. Hierarchical cluster identifying the four dominant watershed types (forested, semi-forested, rural and urban) derived from areal percentages of forested, urban and agricultural land cover for five time periods.



Appendix G. Selected stream-flow gages in North Carolina (NC) and Virginia (VA), along with gage identification number (ID), station name, state, drainage area, physiographic province and watershed type.

Gauge ID	Station Name	State	Drainage Area (km^2)	Province	Watershed Type
01613900	Hogue Creek near Hayfield	VA	41.2	Valley and Ridge	Semi-forested
01620500	North River near Stokesville	VA	44.8	Valley and Ridge	Forested
01654000	Accotink Creek near Annandale	VA	61.9	Piedmont	Urban
01658500	South Fork Quantico Creek near Independent Hill	VA	19.7	Piedmont	Forested
01662800	Battle Run near Laurel Mills	VA	66.8	Blue Ridge	Rural
01673550	Totopotomoy Creek near Studley	VA	66	Coastal Plain	Rural
02036500	Fine Creek at Fine Creek Mills	VA	58	Piedmont	Semi-forested
02038850	Holiday Creek near Andersonville	VA	22.1	Piedmont	Forested
02055100	Tinker Creek near Daleville	VA	30.3	Valley and Ridge	Rural
02093800	Reedy Fork near Oak Ridge	NC	53.4	Piedmont	Rural
02096846	Cane Creek near Orange Grove	NC	19.5	Piedmont	Semi-forested
02097464	Morgan Creek near White Cross	NC	21.6	Piedmont	Semi-forested
02099000	East Fork Deep River near High Point	NC	38.3	Piedmont	Urban
02102908	Flat Creek near Inverness	NC	19.8	Coastal Plain	Semi-forested
02111000	Yadkin River at Patterson	NC	74.6	Blue Ridge	Forested
02142000	Lower Little River near All Healing Springs	NC	73	Piedmont	Semi-forested
02142900	Long Creek near Paw Creek	NC	42.5	Piedmont	Urban
02143040	Jacob Fork at Ramsey	NC	66.6	Piedmont	Forested
02146211	Irwin Creek at Statesville Avenue at Charlotte	NC	15.5	Piedmont	Urban
02146300	Irwin Creek near Charlotte	NC	79.5	Piedmont	Urban
02146470	Little Hope Creek at Seneca Place at Charlotte	NC	6.8	Piedmont	Urban
02146700	McMullen Creek at Sharon View Road near Charlotte	NC	18	Piedmont	Urban
03450000	Beetree Creek near Swannanoa	NC	14.1	Blue Ridge	Forested

03455500	West Fork Pigeon River above Lake Logan near Hazelwood	NC	71.5	Blue Ridge	Forested
03478400	Beaver Creek at Bristol	VA	69.7	Valley and Ridge	Rural
0208650112	Flat River Tributary near Willardville	NC	3	Piedmont	Forested
0208732885	Marsh Creek near New Hope	NC	17.7	Piedmont	Urban
0209741955	Northeast Creek at State Route 1100 near Genlee	NC	54.6	Piedmont	Urban
0210166029	Rocky River at State Route 1300 near Crutchfield Crossroads	NC	19.2	Piedmont	Rural
0214253830	Norwood Creek near Troutman	NC	18.6	Piedmont	Rural
0344894205	North Fork Swannanoa River near Walkertown	NC	37.6	Blue Ridge	Forested

Appendix H. Runoff curve number (RCN) values used to map and derive a RCN estimate per watershed based on the Soil Survey Geographic Database's soil hydrologic groups and the National Land Cover Database's 2011 cover types. A and B soils have greater infiltration, and lower runoff potential, than C and D soils. For areas without a soil hydrologic group identity (labeled *None*), we used the mean of the A, B, C and D soil hydrologic groups for each particular land cover.

Land Cover Type	Runoff Curve Number							
	Soil Hydrologic Group							
	A	B	C	D	A/D	B/D	C/D	None
Water	92.0	92.0	92.0	92.0	92.0	92.0	92.0	92.0
Open Space Developed	44.4	66.8	77.2	82.8	63.6	74.8	80.0	67.8
Low Intensity Developed	44.4	66.8	77.2	82.8	63.6	74.8	80.0	67.8
Medium Intensity Developed	59.5	75.6	83.1	87.1	73.3	81.3	85.1	76.3
High Intensity Developed	90.0	93.3	94.9	95.7	92.8	94.5	95.3	93.5
Barren Land	39.0	61.0	74.0	80.0	59.5	70.5	77.0	63.5
Deciduous Forest	45.0	66.0	77.0	83.0	64.0	74.5	80.0	67.8
Evergreen Forest	25.0	55.0	70.0	77.0	51.0	66.0	73.5	56.8
Mixed Forest	36.0	60.0	73.0	79.0	57.5	69.5	76.0	62.0
Shrubland	45.0	66.0	77.0	83.0	64.0	74.5	80.0	67.8
Grassland/Herbaceous	49.0	69.0	79.0	84.0	66.5	76.5	81.5	70.3
Pasture/Hay	40.0	64.0	75.5	81.5	60.8	72.8	78.5	65.3
Cultivated Crops	56.0	72.0	81.0	86.0	71.0	79.0	83.5	73.8
Woody Wetlands	45.0	66.0	77.0	83.0	64.0	74.5	80.0	67.8
Emergent Herbaceous Wetlands	49.0	69.0	79.0	84.0	66.5	76.5	81.5	70.3