The Cumulative Impacts of Climate Change and Land Use Change on Water Quantity and Quality in the Narragansett Bay Watershed

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A Thesis Presented

by

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The Cumulative Impacts of Climate Change and Land Use Change on Hydrology and Water Quality in the Narragansett Bay Watershed, Rhode Island

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ABSTRACT

THE CUMULATIVE IMPACTS OF CLIMATE CHANGE AND LAND USE CHANGE ON WATER QUANTITY AND QUALITY IN THE NARRAGANSETT BAY WATERSHED

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Narragansett Bay, Rhode Island, is a valuable natural resource that suffers summer hypoxic events resulting from over a century of cultural eutrophication. Current efforts to reduce nitrogen loading from wastewater treatment facilities discharging into the Bay and its tributaries hold the promise of working towards ecological restoration. But, the efficacy of these efforts may be limited, or undone, if future changes in climate or land use increase nutrient and sediment loads to the Bay. This study developed a SWAT model of the upper Narragansett Bay watershed to simulate water quantity and quality. The baseline model was calibrated and validated to accurately reflect watershed behavior. I then used the model to simulate water quantity and quality under an altered climate, with an IPCC projected increase in temperature of 3°C and a 10% increase in precipitation by 2080. A second scenario incorporated projected 2080 land use in the absence of climate change. The third scenario combined the climate change and land use change alterations to examine cumulative impacts.

A comparison of scenario outputs against the baseline simulation highlighted the expected impacts climate change and land use change will have on the watershed. Both climate

change and land use change demonstrated impacts on surface runoff, water yield, PET and ET, streamflow, and loading of sediment, organic N, organic P and nitrate. Climate impacts were much greater than land use impacts, but land use impacts displayed greater regional variation. The results of the combined simulation indicate that future climate and land use change will likely negatively impact the Bay and undermine current efforts at restoration. However, the results also highlight the potential to utilize land use to mitigate some of the impacts of climate change.

CONTENTS

Page

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

NARRAGANSETT BAY: RESTORATION IN THE FACE OF CLIMATE AND LAND USE CHANGE

1.1 Introduction

1.1.1 The Challenge of Dynamic Land Use and Climate

The conservation and management of river systems and their receiving waters requires an understanding of the system's connections and responses to anthropogenic and natural conditions, specifically with regard to stream flow and variability, water quality, and ecosystem response. Land use is an important factor, as land cover influences the amount and timing of runoff, the export of sediment, and nutrient and chemical loadings to water bodies. The conversion of a watershed from its natural condition towards agricultural and urban development alters the system's hydrograph and contributes non-point source pollution (see de la Cretaz and Barten, 2007, for a review of impacts). As a result, watershed managers typically identify urban and agricultural areas as potential stressors to the system, and focus on mitigating their impacts to protect or restore a water body through the use of Best Management Practices (BMPs). Accurately implementing BMPs requires an understanding of the relative contribution of each land use to a water body's impairment, and often involves determining the relationship between the water body and land use empirically or through modeling. However, land use is dynamic, and changes as a result of demographic, social, economic and regulatory factors. Consequently, BMPs implemented today may not adequately protect water bodies in the future under different land use scenarios. To compensate, watershed managers need to plan BMPs that are effective over the long-term and in the context of changing land use.

While land use is a strong anthropogenic characteristic, climate is considered the dominant natural driver of river flow. Watershed managers typically operate under the assumption of stationarity, the concept that the climate, and therefore the river, fluctuates within "an unchanging envelope of variability" (Milly *et al.*, 2008, p. 573). Under this assumption, watershed models simulate climate using probability density functions developed from long-term empirical data. Although modeled climate can demonstrate substantial stochasticity, the variation is limited to what is likely under a historically stable climate. However, Milly et al. (2008) argue bluntly that "stationarity is dead" (p. 573). They assert that anthropogenic changes to earth's atmosphere have pushed hydroclimate beyond the range of natural variability, forcing water managers to abandon model assumptions of stationarity. As with land use, BMPs developed under current climate conditions may not be effective under ongoing climate change.

1.1.2 Climate Change in the Northeast

According to the Draft Third National Climate Assessment (NCADAC, 2013), the average temperature in the Northeastern United States increased 2°F between 1895 and 2011, a rate of 0.16°F per decade. Further, precipitation increased more than 10%, or by approximately five inches (0.4in per decade). The study also revealed that the region experienced a greater increase in extreme precipitation than any other part of the United States: a 74% increase in the amount of precipitation falling during very heavy events between 1958 and 2010. The authors expect these trends to continue into the future. The amount of warming will be dependent on the trajectory of greenhouse gas emissions, with warming projections of 4.5-10°F by the 2080s if emissions continue to increase, and of 3-6°F if global emissions are substantially reduced. Precipitation projections are more difficult, and present a greater degree of uncertainty. Models predict an increase in winter precipitation, with projections under a high emissions scenario

ranging from a 1-29% increase by the end of the century. Annual projections range from moderate decreases to large increases. However, researchers expect a continued increase in the frequency of heavy precipitation events and an increased risk of summer droughts. The report claims that by late this century, a heavy precipitation event that historically occurred once in 20 years will occur as frequently as every 5 to 15 years (NCADAC, 2013).

These changes will impact water resources as changes in temperature and precipitation alter watershed inputs and conditions. The National Climate Assessment (NCADAC, 2013) indicates that soil moisture in the Northeast increased over the last 20 years. Runoff and streamflow increased during the last century and are expected to continue to increase into the future. The authors note that annual peak flows increased at gauges in the Northeast over the past 85 years and expect more intense floods into the future. They acknowledge that more intense runoff generally increases river sediment, nitrogen and pollutant loads. Nearly everywhere in the continental United States is expected to experience more intense summer droughts. The combined impact of reduced low flows under drought and increased overland flow during floods has the potential to worsen water quality. The authors also note increased water temperatures, a trend expected to continue in tandem with increased surface temperatures, with increases ranging from 25-55% depending on the region and the land use type.

Alone, these impacts have the potential to stress aquatic systems. Yet, the report stresses that these impacts will occur within the context of existing stressors, such as wildfires, fertilizer use and land cover changes. The authors maintain that changing land cover, together with increasing flood frequencies and magnitudes, is expected to increase the mobilization of sediments in large river basins. Increased nutrient concentrations and residence times in streams, resulting from low water levels from the increasing frequency and duration of

droughts, may increase the likelihood of harmful algal blooms and low oxygen conditions (NCADAC, 2013). Thus, to adequately protect vulnerable watershed systems, watershed managers need to predict the combined impacts of these stressors.

1.1.3 Nutrient Pollution and Climate Change in Narragansett Bay

Predicting the cumulative impacts of climate change and land use change is important for Narragansett Bay, Rhode Island. The Bay is an important resource for the state, providing habitat for diverse plant and wildlife communities, generating tourism revenue, and supporting recreational and commercial fisheries. Tourism alone generates more than $2.31 billion for the state's economy, and Rhode Island's beaches play an important role (RIEDC, 2012). Yet, the Bay suffers from the impacts of a history of excess nutrient loading.

1.1.3.1 Past and Present Nitrogen Loading

Nitrogen loading increased dramatically from the end of the nineteenth century into the early twentieth century as expanding urban populations were connected to sewers that drained to the Bay and its tributaries. The Blackstone River experienced a 17-fold increase in dissolved inorganic nitrogen (DIN) in fewer than fifty years following 1870, and the Seekonk estuary was heavily enriched by 1913-14. Monitoring revealed ammonium increases in the Ten Mile and Taunton Rivers between 1900 and 1906. The Pawtuxet River maintained good water quality as a source of Providence drinking water, but nitrogen concentrations rose dramatically once the Scituate Reservoir was impounded and the protected status given to the Pawtuxet as a source of drinking water was removed. The small urban rivers in Providence (the Moshassuck and Woonasquatucket) were enriched by 1897 (Nixon et al., 2008).

Current nitrogen loading and recent trends are subjects of some debate. Nixon et al. (2008) maintain that nitrogen delivery to the Bay has not changed much from the early twentieth century, although improved sewage treatment transformed the form of nitrogen from

ammonium to nitrate. They further report that nitrogen concentrations in the Blackstone, Pawtuxet and Moshassuck Rivers during their 2003-2004 sampling period exhibited a decline from levels decades earlier. This led them to conclude that since the 1980s, nitrogen loading from the rivers remained stable or declined. Hamburg, Pryor and Vadeboncouer (2008) modeled results show a significant increase in nitrogen loading to the Bay from the Blackstone River during the twentieth century, roughly by a factor of two. They also state that DIN in the Taunton River between 1997 and 2002 was more than double the value from 1898-1915. Their model suggests an increase in nitrogen flux to the Bay of 73% from 1900 to 2000.

The high level of nitrogen loading to the Bay has two prominent ecological impacts: the loss of ecologically-sensitive eelgrass meadows and hypoxia. Both are the result of increased primary production from excessive nutrient loading. Many areas of the Bay exhibit a problematic degree of eutrophication. Deacutis (2008) assigned levels of eutrophication to regions of the bay based on chlorophyll a concentrations. The 90th percentile values for chlorophyll a in the Seekonk River exceeded the hypereutrophic threshold, while values for the Providence River fell within the “high” threshold and occasionally within the hypereutrophic threshold. The upper bay values fell just above the lower limit of the “high” threshold, while Greenwich Bay was within the upper level of the “medium” threshold or low end of “high”. The level of eutrophication declined with decreased latitude, demonstrating a clear north-south nutrient gradient. Elevated chlorophyll levels reduce light in the water column, a principle factor governing the growth of sea grass, and overgrowth of macroalgae can also shade out the vegetation (Deacutis, 2008). Deacutis (2008) maintains that the threshold for the loss of eelgrass meadows is 60 kg/ha/yr of nitrogen, while current loads to Greenwich Bay and the Providence River are > 130 kg/ha/yr and > 750 kg/ha/yr, respectively. As a result, no significant eelgrass beds exist north of the southern end of Prudence Island.

High phytoplankton production and, to a smaller extent, benthic macroalgae production are linked to intermittent hypoxia. Hypoxic events occur in the summer months and are associated with weak neap tides. The events typically last from hours to approximately a week. The Seekonk and upper Providence Rivers exhibit extreme hypoxia and occasional anoxia (0.0-1.3 mg/L DO), with the mid-Providence River ranging from 1.4-2.2 mg/L DO and the lower Providence River and upper bay ranging from 2.5-3.8 mg/L DO. Other studies observed even lower values (Deacutis, 2008). Deacutis (2008) details some of the impacts of Narragansett Bay hypoxia, including the loss of sensitive macrobenthic species and a loss of biogenic structure linked to the loss of blue mussel reefs.

1.1.3.2 Point Sources

Despite some disagreement over the amount of nitrogen loading and its relation to the aforementioned impacts, all sources agree that wastewater treatment facilities (WWTFs) are the dominant source of nitrogen loading to the Bay. The prominence of point sources of nitrogen in Narragansett Bay is typical of the urban northeast and contrasts with many of the estuaries of the mid-Atlantic, southeast and Gulf coast that are primarily affected by non-point source agricultural runoff. Boyer et al. (2002) constructed a nitrogen budget for the Blackstone River and found that the greatest nitrogen input to the river was nitrogen import in food that subsequently entered the river via sewage effluent. Vadeboncoeur et al.'s (2010) model shows that trends in nitrogen loading are dominated by the effect of sewage effluent, and attribute 51% of nitrogen loading to human waste delivered via sewers. Deacutis (2008) cites WWTFs as responsible for 67-73% of the nutrient load to the Bay, and for 66-68% of the nitrogen load to the Providence River. There are two types of WWTFs impacting the Bay: those that discharge directly into the Bay and those that discharge into the rivers in the watershed. The direct sewage discharge of nitrogen into the Bay is approximately equal to that discharged into the

rivers. Annually, the rivers provide most of the nitrogen to the Bay, although direct discharge facilities provide twice as much nitrogen as the rivers during the low flow months of summer and fall (Nixon et al., 2008). The two largest effluent discharges come from the Bucklin Point WWTF on the Seekonk River and Field's Point on the upper Providence River (Deacutis, 2008). The three facilities that discharge to the Seekonk and Providence River estuaries account for 82% of the sewage total nitrogen discharged directly into the Bay (Nixon et al., 2008). Since many rivers and WWTFs discharge in the northern portion of the Bay, they explain the clear north-south eutrophication gradient and related impacts.

Concerns by managers about summer hypoxia, and a notable fish kill in Greenwich Bay in 2003 that sparked public concern, motivated managers to pursue reductions in nitrogen loading to the Bay, primarily by accelerating plans to implement nutrient reduction technologies at Rhode Island WWTFs (Nixon et al., 2008; Oviatt, 2008). As a result, the state promulgated regulations that decreased nitrogen output in the effluent of all Rhode Island facilities to 5 mg/L (Oviatt, 2008). Estimates of projected loading reductions vary greatly. Deacutis (2008) claims total nitrogen loads to the Bay will decrease 30-35% once required tertiary treatment is fully operational. Nixon et al.'s (2008) calculations suggest that if all but the smallest WWTFs reduced their nitrogen, nitrogen reduction to the Bay during a median flow year would be 45-55% during June to September. Vadeboncoeur et al. (2010) state that if the established nitrogen targets are met by the eight large WWTFs, total land-based loading will be reduced 11% relative to business-as-usual, and if a similar target is applied to all Rhode Island facilities and the largest Massachusetts facilities(combined with reductions in atmospheric and fertilizer loading) there would be a 27% reduction.

Deacutis (2008) notes that even a 35% decrease in total nitrogen load would still exceed the 60 kg/ha/yr threshold for eelgrass meadows in the upper Bay. Oviatt (2008) further

investigated the effects of nutrient reductions and found that a response from a 20% decrease in total nitrogen would be difficult to detect. However, the study concluded that reductions in phytoplankton biomass would be detected at a 50% nitrogen reduction in the mid to upper Bay. South of Conimicut Point, a 50% reduction may result in an average decrease of 50% in primary production, chlorophyll, zooplankton and benthic animals. The author asserts that this would be sufficient to keep the water column from becoming hypoxic.

1.1.3.3 Non-Point Sources

Regardless of the specific reductions achieved, it is apparent that nitrogen loading to Narragansett Bay from point sources will be reduced considerably in the near future towards the goal of reducing summer hypoxic events. However, the efficacy of these planned reductions is dependent, in part, on other sources of nitrogen remaining static or declining. Already, other sources have an impact on the Bay. Nixon et al. (2008) found that while WWTF discharges accounted for nearly all nitrogen delivered to the Bay from the Blackstone and Ten Mile Rivers, there are other large sources contributing nitrogen to the Pawtuxet and Taunton Rivers. Of particular importance is the role of land use changes. The most prominent land use change in the Northeast has been the change in the amount and kind of forest cover due to urban expansion, although this expansion is projected to continue at a slow rate in the region due to the high level of existing development and low rates of population growth (NCADAC, 2013). Hamburg et al. (2008) report that the trend of increased forest cover in the Narragansett Bay watershed due to agricultural decline has recently reversed as secondary forest is cleared for low-density development. This residential and urban expansion could limit the effects of WWTF nitrogen reductions as cleared and developed land produces greater runoff and nitrogen exports than forested land (Hamburg et al., 2008). Boyer et al. (2002) reported a strong negative correlation between total nitrogen inputs and the fraction of forested land area. The authors

note the importance of urbanization as “a large human-derived source of Nitrogen to the [northeast]” (p. 163), and found a strong relationship between disturbed landscapes and total nitrogen loading. Vadeboncoeur et al. (2010) highlight the potential implications of increasing residential areas, noting that rates of fertilizer nitrogen application to suburban lawns in the northeast may be equal to or even double that of field crops. The authors further recognize that in the Narragansett Bay watershed, the greatest growth in nitrogen loading comes from suburban areas, including the Taunton watershed and the small watersheds surrounding Providence. This establishes two points. First, as point sources are controlled to the greatest technological and economical extent, the focus of mitigating nutrient pollution to the Bay may, and should, shift to controlling non-point source pollution from urban and residential areas. Second, if the trend of increased development at the expense of forested land continues, that change in land use may hamper current efforts to control nutrient pollution.

1.1.3 Climate Change

Narragansett Bay is already experiencing the effects of a changing climate. Air temperatures in Rhode Island rose 1.7°C from 1881 to 2006, faster than the global or New England regional averages, and Narragansett Bay water temperatures measured at Fox Island increased 0.91°C from 1959 to 2003, faster than the surface North Atlantic average. This may be influencing trophic dynamics in the mid and lower Bay. While the upper Bay continues to be enriched by nutrient loading, resulting in elevated chlorophyll levels and increased macroalgae growth, the mid to lower Bay experienced oligotrophication from the 1970s onward. There was a 70% decrease in chlorophyll biomass from the 1970s to 2006 and a decrease, or loss, of the winter-spring phytoplankton bloom. The current hypothesis is that warmer winter water temperatures is increasing zooplankton grazing pressure early in the spring bloom cycle, suppressing the bloom. Phytoplankton from the bloom typically sinks to the benthos and serves

as an important resource for detritivores. The increased grazing pressure limits that supply and replaces it with less nutritious zooplankton fecal pellets, stressing and altering the benthic community. This presents an unusual situation where the upper Bay is becoming increasingly eutrophic while the lower Bay is increasing oligotrophic, and should be considered when examining long-term management strategies. Warmer water temperatures also may be altering Bay fish communities. There has been an observed decrease in cold-water demersal species and in increase in warm-water pelagic species. Winter flounder populations decreased 90% from 1980 (Smith et al., 2010).

While temperature changes and sea level rise have been the focus of much research on climate change impacts to Narragansett Bay, less attention has been given to the impact of changing precipitation regimes. Yet, changes in precipitation will influence river flow to the Bay and the associated loading of nutrients, sediment and pollutants. Pilson (2008) cites a precipitation increase of 3 mm/yr from 1905 to 2006 in the Narragansett Bay watershed, a 30% increase over the 100-year time span. As noted above by the National Climate Assessment (NCADAC, 2013), surface runoff and river flows are expected to increase in the northeast as a result of climate change. Pilson (2008) calculated an increase in average river flow from the 1960s of 12 m^3/s, but notes that year-to-year variability creates a low level of confidence in that value.

Pilson (2008) also calculated residence time in the Bay, finding a range from 20 days in early spring to 35 days in late summer. Residence time is mostly controlled by inputs of freshwater; at the highest flow residence time is close to 12 days. From this, Pilson (2008) concludes that increased precipitation and the resulting increase in streamflow will reduce residence times in Narragansett Bay in the future. Residence time influences the occurrence of hypoxic events (Nixon, 2008; Oviatt, 2008; NCADAC, 2013), and decreases in residence times

may lower the risk of hypoxia. This prediction is supported by conditions in Greenwich Bay, which experiences severe hypoxia as a result of a long flushing time due, in part, to a lack of river inputs (Codiga et al., 2009). However, Codiga et al. (2009) found that river flow covaried positively with hypoxic events and was the most important independent variable in predicting hypoxia, indicating that hypoxia was more severe following higher river flows. The authors assert that the one-to-two-month response timescale for hypoxia suggests that June river flow can be used as an index for the severity and extent of July and August hypoxia. They attribute this relationship to the dual role of rivers in delivering nutrients that fuel primary production and freshwater that produces density stratification, which prevents the diffusion of oxygen to lower levels. Smith et al. (2010) also highlight the role of river flow in increasing stratification, turbidity and land-based nutrients.

1.1.4 The Need for Prediction and Purpose of this Study

Nitrogen reductions from WWTFs may not be enough to mitigate hypoxia and restore eelgrass meadows. Hamburg et al. (2008) claim that even after mandated denitrification is implemented, the rate of anthropogenic inputs to the upper Bay will remain higher than in any other sub-watershed. Further, Deacutis (2008) states that climate change could negate the benefits of nutrient reductions from WWTF controls. Hamburg et al. (2008) note that the goal moving forward is defining an acceptable level of nitrogen enrichment in an ecosystem increasingly stressed by climate change. Smith et al. (2010) acknowledge that impacts from increased precipitation will occur along with hydrograph changes caused by urban development and wetland losses. This highlights an important point: climate change and land use change do not affect water bodies independently, and their cumulative impacts are likely to present new challenges to watershed managers. Planning nitrogen reduction initiatives under the assumption of static climate and land use conditions will limit their efficacy. A successful

strategy to restore the Bay's health requires consideration of the long-term impacts of these changing factors and their interactions with each other. Thus, the long-term conservation and management of these systems requires a degree of predictive capability.

Dynamic process models can provide this prediction, and their application to specific water bodies and watersheds can inform watershed managers of potential impacts and drive the development of BMPs that can effectively mitigate these impacts. This study modeled the combined impacts of future land use change and climate change on the hydrology and water quality of rivers in the upper Narragansett Bay watershed. These results predicted how river flows and nutrient loading to Narragansett Bay may change over time, and inform a discussion of the likely impacts that will have on the ecology of the Bay and on the efficacy of nitrogen reduction efforts. Further, the modeling results can help develop BMP and policy scenarios that could mitigate the impacts of climate and land use changes. The National Climate Assessment (NCADAC, 2013) states that increasing resilience and adaptive capacity are useful strategies for water resource management and planning in the face of climate change. However, the authors argue that challenges to achieving this include a lack of scientific information and inadequate information useful for practical applications. The goal of this study was to provide scientific information that will be useful to managers and planners in informing their decisions to mitigate the future impacts of climate change and land use change on the health of the Bay and their restoration goals.

1.2 Literature Review of Combined Climate Change and Land use Change Modeling

Multiple studies have modeled the response of river systems to climate change or land use, but successful conservation and management requires consideration of long-term cumulative impacts. Studies modeling the combined impact of both land use and climate changes are needed but rare. This section reviews several studies that modeled the combined

impacts of climate change and land use change on river systems in the United States with the objectives to (1) identify the models used to simulate climate change, land use change and watershed response; (2) discuss the numbers and types of climate-land use scenarios simulated; (3) identify the types of responses simulated; (4) summarize the results of the studies to demonstrate expected responses and (5) discuss the limitations and challenges of modeling climate change and land use change impacts.

1.2.1 Model and Scenario Selection

Six watershed models were used to simulate hydrology and water quality (Table 1.1). Of these, SWAT and HSPF were the most common, both utilized by three studies. The array of models suggests that no singular watershed model is appropriate for assessing cumulative impacts, and other factors may drive model selection. Butcher et al. (2010) utilized both SWAT and HSPF, and selected them based on the requirements that a model be process-based, dynamic with daily time steps, able to simulate water quality, widely used and accepted, in the public domain and with an existing GIS interface. Thus, their rationale was in part mechanistic and in part based on ease of access. Praskievicz and Chang (2011) chose WinHSPF because it is one of the most commonly used public domain modeling systems with a large user community and abundant technical support, and because it is able to simulate water quality. They further specify that they chose WinHSPF over SWAT, another model that meets the prior criteria, because HSPF was intended for use in mixed land use basins, while SWAT was developed primarily for agricultural watersheds. Thus, their model selection was ultimately driven by the characteristics of their study area, the Tualatin River Basin, which maintains similar areas of forest, agriculture and urban land use. This indicates that model selection was driven by mechanistic, practical and situational factors. The use of various watershed models allows researchers to examine their relative strengths and weaknesses. Butcher et al. (2010) found that

HSPF was better able to replicate observations during calibration and validation, but SWAT simulations of watershed response to changing conditions were more robust.

All climate models were based on General Circulation Models (GCMs), although studies used varying GCMs from different sources. Nearly all studies relied on emissions scenarios developed in the United Nations Intergovernmental Panel on Climate Change Special Report on Emissions Scenarios (SRES) to drive the GCMs, but they differed in the number of emissions scenarios they used (Table 1.1). Emissions scenarios contain a series of assumptions regarding population growth, technology, global trade, the importance of industry and the proportional use of fossil fuel, and renewable energy sources. As such, these scenarios contain a great degree of uncertainty and selection of an emissions scenario was strictly subjective. Nelson et al. (2009) tested the A2 and B2 scenarios, representing mid-high and mid-low emissions, respectively. Both Viger et al. (2011) and Tu (2009) tested three scenarios, B1, A1B and A2, representing increasing emissions levels with B1 as the "best case" and A2 as the "worst case" in their analyses. Franczyk and Chang (2009) relied on the A1B scenario, and Wilson and Weng (2011) and Praskievicz and Chang (2011) on the A1B and the B1 scenarios. Butcher et al. (2010) relied only on the A2 scenario. Scenario selection reflects a choice by the researchers regarding the context of the results. Researchers using multiple scenarios sought to capture the variability implicit in the scenarios by including a range. Franczyk and Chang (2009) used the A1B scenario because it was the "middle of the road scenario" (p. 808), while Butcher et al. (2010) used the A2 scenario because it was the "pessimistic" (p. 1355) scenario. Only Tong et al. (2012) did not rely on a climate model or SRES scenarios, and instead developed four hypothetical scenarios. The use of these hypotheticals allowed the authors to examine a range of responses, but may have limited practical application as the results aren't predicted changes and cannot capture seasonal variability.

Table 1.1 Studies reviewed and their model and scenario selections (*excludes baseline)

Authors	Date	Location	Watershed Model	Climate Model	Emissions Scenarios	Land Use Change Model	Land Use Scenarios *	Combined Scenarios *
Hejazi & Moglen	2008	MD	McCuen continuous streamflow model	GCM	1	None	1	2
Franczyk & Chang	2009	OR	SWAT	GCM	1	None	3	3
Nelson *et al.*	2009	Chesapeake Bay Watershed	Continuous streamflow model	GCM	2	None	1	4
Tu	2009	MA	AVGWLF	GCM	3	Regression	2	6
Butcher *et al.*	2010	Nationwide	SWAT & HSPF	GCM	1	None	1	1
Praskievicz & Chang	2011	OR	WinHSPF	GCM	2	None	2	4
Viger *et al.*	2011	GA	PRMS	GCM	3	FORE-SCE	1	3
Wilson & Weng	2011	IL	SWAT	GCM	2	IDRISI	3	6
Tong *et al.*	2012	OH	HSPF	NONE	4	IDRISI	1	4

Land use was infrequently modeled. Models used included FORE-SCE (Viger et al., 2011), IDRISI (Wilson & Weng, 2011; Tong et al., 2012) and a linear regression extrapolating historical land use trends (Tu, 2009) (Table 1.1). Most land use scenarios were hypotheticals based on a prediction of increased impervious cover (Hejazi & Moglen, 2008, posited a 10% increase in impervious cover) or on scenarios developed by others representing a range of future development scenarios (Franczyk and Chang, 2009, used “Compact”, “Planned” and “Sprawl” scenarios representing increasing imperviousness and decreasing forest and wetland cover respectively, while Praskievicz and Chang, 2011, used a “development” and a “conservation” scenario). The paucity of land use modeling in these studies suggests difficulty in modeling

future land use that pushes researchers to rely on hypotheticals that reflect either a logical prediction or a range of possibilities.

1.2.2 Modeled Responses

Table 1.2 Modeled responses by study

Authors	Date	Streamflow Volume	Streamflow Variability	Water Quality	Ecosystem Response
Hejazi & Moglen	2008	X	X		
Franczyk & Chang	2009	X	X		
Nelson *et al.*	2009	X	X	X	X
Tu	2009	X	X	X	
Butcher *et al.*	2010	X	X	X	
Praskievicz & Chang	2011	X	X	X	
Viger *et al.*	2011	X			
Wilson & Weng	2011	X	X	X	
Tong *et al.*	2012	X		X	

1.2.2.1 Streamflow Volume

All studies investigated the impacts of climate and land use change on the volume of streamflow (Table 1.2). The influence of climate change and land use change on streamflow varied based on scenarios and location. Nelson et al. (2009) observed a 7% decrease in median discharge under urbanization as a result of lower infiltration supporting base flow, but a 40% increase under climate change due to higher annual precipitation that offset urbanization decreases. Franczyk and Chang (2009) found streamflow would increase 2.3-2.5% from 2001 to 2040 as a result of increasing urbanization. When both climate and land cover changes were considered together, the magnitude of the mean annual runoff volumes increased compared to the baseline and both the individual land use change and climate change scenarios. Wilson and Weng (2011) observed an increase in river discharge from the baseline for 2020 and 2030 of 46% and 204% respectively.

Viger et al. (2011) observed decreases in streamflow under two of three combined scenarios, although none were statistically significant. This study is useful in that it clearly highlights the complex interactions between climate change and land use change with regard to hydrology. Under increased urbanization, streamflow significantly increases as a result of increased surface runoff and reduced evapotranspiration (ET) resulting from higher impervious cover and less vegetation. Under climate change, streamflow decreases in all three scenarios, with variation driven by precipitation. Although streamflow decreases under both the climate-only and the combined scenarios, the addition of land use change to climate scenarios ameliorates some of the reduction in streamflow and makes it statistically insignificant. This results from a significant increase in surface runoff under land use change that offsets the decrease under climate change. This occurs despite decreased precipitation resulting from climate change, and is attributable to urbanization's effect on ET. Under climate change, higher temperatures equate to higher ET, and that works with decreased precipitation to substantially reduce surface runoff. However, because the expansion of impervious cover and loss of vegetation under land use change removes soil and plant storage, ET decreases under all three combined scenarios. Thus, while higher temperatures from climate change should increase ET, greater impervious cover under land use change effectively mitigates the effect of temperature and reverses the trend, allowing surface runoff to increase under the combined scenarios and mitigating some of the reduction in streamflow. This study demonstrates the need to assess both climate change and land use change simultaneously to obtain accurate predictions. Tu (2009) observed that, despite higher average precipitation, annual streamflow was expected to decrease in most watersheds under climate change. As in Viger et al. (2011), the addition of land use to Tu's (2009) model ameliorates the decrease in streamflow. For both land use scenarios (increasing land use at the current rate and at double the current rate), the combined model

produces no significant annual change in streamflow from the baseline. Butcher et al. (2010) and Tong et al. (2012) both found that different climate scenarios could produce annual increases or decreases in streamflow, ranging from a 26% increase to a 27% decrease for Butcher et al. (2010) and a 44% increase to a 53% decrease for Tong et al. (2012).

1.2.2.2 Streamflow Variability

Of greater importance to watershed managers and greater significance in the studies is streamflow variability. This involves both monthly variability and the frequency and magnitude of extreme events. Every study that investigated streamflow variability reported higher winter and fall flows and lower summer flows. Franczyk and Chang (2009), who observed annual streamflow increases, observed the largest increases in runoff depth in the fall (+1.83 cm, +23.5%) and winter (+1.36 cm, +5.1%), while the summer showed no significant departure from the baseline. This indicates that the timing of seasonal runoff will start earlier in the water year and end earlier. Praskievicz and Chang (2011) observed the same pattern, with winter flow increases ranging from 11% to 69% by the 2040s, and Butcher et al. (2010) observed a similar trend.

Tu (2009) also observed increases in the late fall and winter months with decreases in summer and early fall. Thus, while Tu (2009) found no significant departure from the baseline for annual streamflow, there were substantial impacts from climate change and land use on monthly streamflow. However, large decreases in summer flow offset large increases in fall and winter flow. This study highlights the importance of modeling response to climate on a sub-annual basis. Had the author reported only annual changes, the conclusion would be of no impact. This would mask substantial seasonal changes. Reduced summer flows may pose serious problems for water managers with regard to water supply and pollutant concentrations, while

higher winter flows could pose flood risk. Thus, studies that report annual changes exclusively may ignore important responses.

Praskievicz and Chang (2011) reported that flows for the highest 5% of flows were greater in magnitude (more than doubled), while the lowest 5% were lower in magnitude, under the combined model. Nelson et al. (2009) observed that minimum daily discharge did not vary, but that high flow days discharged 17% more water under urbanization, 27% more under climate change and 45% more under the combined scenario. Hejazi and Moglen's (2008) climate models indicated a fewer number of storms, but larger storm volumes than the observed record. The 10% increase in impervious cover under the land use scenario increased peak flows and decreased low flows, but this trend was not significant until impervious cover increased by 50%. Results varied between the two climate models, but the combined results showed that the addition of land use to the CCC climate model led to more significantly decreasing low flows and shifted the slightly decreasing trends in peak flows to slightly increasing for Q1 and Q5. The addition of land use to the Hadley climate model led to more significantly increasing peak flows and shifted the slightly increasing low flows to slightly decreasing. The overall results produced increases in peak flows (Q1, Q5, Q10) but small, if observable, changes to low flow (Q90, Q95, Q99) in the combined model compared to individual models. This suggests more variable streamflow when both factors are considered. Thus, while land use alone may not have a significant impact on extreme events, it did play a role in shifting the impact of climate change.

1.2.2.3 Water Quality

Fewer studies examined water quality (Table 1.2). Nelson et al. (2009) considered bed mobility and the supply of suspended sediment in the river system. They observed that bed mobility was greater under climate change and urbanization, resulting from increased storm size and increased flashiness, respectively. Extreme precipitation events under climate change and

erosion from construction under urbanization both increased suspended sediment, and the effect was additive when the two scenarios were combined. Maximum siltation varied between the stressors, with climate change decreasing siltation through greater scouring of the streambed and urbanization increasing siltation due to elevated inputs. Wilson and Weng (2011) investigated the concentration of total suspended solids (TSS), observing a marked seasonal difference. July exhibited a decrease in TSS from 2010 to 2030 across both climate scenarios and all three land use scenarios that they attributed to a reduction in precipitation. March exhibited an increase for all scenarios, with dramatic increases of up to 200% predicted for 2030, resulting from greater flow. Differences in land use, specifically the extent of agricultural and vacant lands, and slope drove differences between sub-watersheds, but precipitation was the major determinant of TSS concentrations. Authors noted that TSS responds to flushing more than dilution, accounting for higher concentrations despite higher flows. Praskievicz and Chang (2011) found that suspended sediment mimicked hydrological changes, with increases in winter and decreases in summer. Because sediment load depends on surface flow, land use scenario had a substantial impact; the "Development" scenario produced an 18% increase, while the "Conservation" scenario produced an 18% decrease. However, climate was still the primary determinant of sediment load.

Praskievicz and Chang (2011) observed that changes in phosphate loading closely track sediment loading, as expected since phosphorus binds to soil particles. Wilson and Weng (2011) found that by 2030, July phosphorus levels in the watershed were lower as a result of decreased precipitation reducing surface runoff. This was also driven by expected land use changes, specifically a decline in agricultural land, an important source of phosphorus. The authors generally found higher phosphorus concentrations in the winter/early spring resulting from greater precipitation, with differences between land use scenarios driven by the extent of urban

and residential land area, both important sources of phosphorus. However, they did find lower total phosphorus concentrations under the A1B scenario, despite increases of 365% and 200% for precipitation and river discharge respectively. This is because the load was smaller than the background level of the water, causing significant dilution. This reveals a potential problem with their representation of results, as the phosphorus load was higher but the concentration was lower.

Tu (2009) observed that nitrogen loading under climate change mimicked streamflow, with late fall and winter increases and summer decreases. However, the addition of land use to the model shifted the pattern, producing large increases in nitrogen loading for all months except December, with the greatest nitrogen increases occurring during late summer and early fall. The increases were even more pronounced under the doubled imperviousness rate scenario. While the annual load only increased slightly under the combined scenario, the results show that seasonal shifts are large and highlight the importance of including land use in the model. Tong et al. (2012) observed increases in nitrogen and phosphorus under all climate scenarios and combined scenarios, including those that produced decreases in streamflow. Of the combined scenarios, total nitrogen increases ranged from 3.97% to 11.55%. Total phosphorus increases under combined scenarios ranged from 6.73% to 21.35%. For both nutrients, the greatest increases occurred in the wettest climate scenarios and the smallest increases under the driest scenarios. However, even the scenario with the smallest increase in phosphorus, 2.63% under the driest climate change-only scenario, elevated phosphorus concentrations above the EPA water quality standard. Thus, climate change and land use change substantially impact water quality and have noticeable management implications. In sum, water quality impacts were variable across regions, climate scenarios and land use scenarios. However, water quality parameters typically emulated flow. This is supported by Butcher et al. (2010),

who observed increases in total suspended solids, total nitrogen and total phosphorus of 44, 10 and 24% as a result of flow increases.

1.2.2.4 Ecosystem Response

Only Nelson et al. (2009) modeled ecosystem response (Table 1.2). Their study investigated how climate change and urbanization-induced changes in hydrology, geomorphology and temperature affect fish assemblages. This involved developing indices to summarize the impacts of urbanization and climate change on fishes, specifically on spawning and juvenile growth. They examined changes in index values from the baseline to assess biotic response. Stress from urbanization took the form of siltation clogging interstitial pores that are important for insectivores, urban flashiness and a lack of leaf litter input. Stress from climate change resulted from changes in total precipitation and distribution affecting sediment loading, siltation and primary productivity in addition to temperature shifts. Their results asserted that urbanization alone stressed few species (eight showed reduced adult growth) while climate change affected most of the assessed species (22 to 29 species, depending on the scenario). Adding urbanization to the climate change model increased the number of stressed species. The two major stressors observed were on juvenile growth rates as a result of altered temperature and hydrology, and stress on adult growth rate resulting from altered temperature, siltation and food resources.

1.2.3 Limitations and Challenges

Every study has its own limitations with regard to the robustness of the model and the scope of inference of the results. Franczyk and Chang (2009) recognize that their study was limited both by the use of a single emissions scenario, and by the short time period of runoff data used for calibration and validation. Wilson and Weng (2011) acknowledge that their focus

on concentrations of total phosphorus makes it difficult to compare across studies that present loads, as dilution under heavy precipitation shifts results away from the linear relationship between flow and phosphorus levels. Tu (2009) cites the simplicity of the land use scenarios as a potential weakness. The scenarios were based on extrapolating the historical rate of change, with the assumption that the future trend will continue to be the conversion of forest to developed land at a linear rate. However, this assumption is weak as land use change is driven by a number of socioeconomic and biophysical factors that will force deviations from the historical trend. As stated earlier, studies that investigate changes in annual load only (Viger et al., 2011; Tong et al., 2012) are limited in their practical use as they ignore monthly or seasonal variability that may exhibit more significant impacts.

On a broader scale, modeling the cumulative impacts of climate and land use change has a number of challenges and limitations. Praskievicz and Chang (2009) identify four sources of uncertainty in hydroclimate impact assessment that limit the robustness of studies: choice of GCM, transferring large-scale climatology to regional-scale climatology (downscaling), the parameters of the hydrological model and the water resource impact models. They further define some of the challenges in choosing how to evaluate climate impacts. A study may use a "synthetic climate change scenario" (p. 651), altering the historical average climate parameters based on expected changes. This approach was used by Tong et al. (2012). While this avoids the uncertainty of GCMs and allows for sensitivity analysis, it also retains the historical range of variability, which is inconsistent with expected changes in climatic variability - as demonstrated in most of the reviewed studies above (Praskievicz and Chang, 2009). The alternative, used by the remainder of the reviewed studies, is to use emissions scenarios to drive a climate model. The authors assert that because each GCM models conditions and feedbacks differently, their projections can vary widely. Further, they require the use of regional climate models or

statistical downscaling to be applied on an appropriate scale. They maintain that the uncertainty associated with emissions scenarios is not problematic in the short term, as most show similar levels of emissions through the 2050s and the atmosphere takes time to respond (Praskievicz and Change, 2009). The studies reviewed above, however, do demonstrate differences, occasionally substantial, between the results of different emissions scenarios. Although there is uncertainty inherent in the hydrological model, the authors claim that results of climate impact studies are more sensitive to climate scenarios than to the choice of hydrologic model.

MacDonald (2000) further identifies some of the limitations and challenges of modeling cumulative impacts. These include an inability to predict secondary or indirect effects, often inclusive of synergistic effects, the difficulty in defining recovery rates, the uncertainty of future events, and the difficulty of validation. He also maintains that much of the confusion in recognizing and predicting cumulative impacts comes from poorly defining resources of concern and the spatial and temporal scales of analysis. However, with regard to the practical application of the science, MacDonald's most salient point involves the dissonance between the models developed by researchers and the needs of water managers. He cites Reid (in MacDonald, 2000) in stating that methods produced by management agencies are "simple, incomplete, theoretically unsound, unvalidated, implementable by field personnel and heavily used", while those developed by researchers are "complex, incomplete, theoretically sound, validated, require expert operators and unused" (p. 308). Thus, if the goal is the protection of river systems through mitigating or adapting to the impacts of climate and land use changes, then the greatest challenge for the research community may be in developing methods and producing information that is usable and applicable by the conservation and management community.

1.2.4 Summary of Literature Review

The large number of papers over a short span of time suggests that modeling the cumulative impacts of climate change and land use change is a rapidly developing field of research. Studies reviewed here utilized a variety of watershed models with SWAT and HSPF the most common. Model selection was dependent on the specifics of the research project, including watershed characteristics, available data and practical application. While nearly all studies relied on GCM climate models, few modeled land use, instead relying on hypothetical scenarios representing a range of possible development scenarios. This may be a product of the difficulty of modeling land use resulting from large uncertainty in demographic, socioeconomic and regulatory variables. If researchers are interested strictly in departure from baseline, then hypotheticals are useful in producing a range of responses and comparing alternative development pathways that can be used in land use planning. However, if the goal is prediction, land use modeling, such as IDRISI models used by Tong et al. (2012) and Wilson and Weng (2011), become more important.

Most studies used multiple emissions scenarios to capture the variability inherent in estimating future emissions. The differences between the results of different emissions scenarios were usually small, but occasionally produced very different results. This highlights the benefit of including multiple emissions scenarios to display the range of possible hydroclimatic effects. Most studies revealed relatively modest annual effects on streamflow from combined land use and climate changes. However, all studies showed increased variability. The common trend was greater flow in the late fall and winter and reduced flows in the summer. Studies also observed increases in the magnitude of extreme events. Thus, the major challenges faced by water managers and conservationists in the future may not be changes in annual flow, but

dramatic changes in hydrologic variability. This also suggests that models need to investigate hydroclimate on a seasonal or monthly scale to get an accurate picture of future challenges.

Water quality largely reflected changes in streamflow, suggesting that future watershed management will require responses to seasonal shifts in nutrient and sediment loading. The Nelson et al. (2009) study demonstrated potential loss of fish diversity as a result of climate change and land use changes. This was the only study of the nine reviewed to assess ecosystem response. This suggests that ecosystem response is complex and difficult to model. However, if long term conservation of aquatic systems is to be effective there needs to be some understanding of the response of the biotic ecosystem components to land use and climate changes

Despite varying model selections, scenarios and results, a constant across all studies was that climate had a greater impact on streamflow than land use. Typically, climate was the driver of any changes from baseline conditions, and land use played a role in either exacerbating the change (Nelson et al, 2009; Franczyk & Chang, 2009; Wilson & Weng, 2011) or ameliorating it (Viger et al., 2011; Tong et al., 2012; Tu 2009). This result has important implications. As noted by Nelson et al. (2009), the two processes of land use change and climate change operate over sharply contrasting spatial scales. Climate change is the result of global emissions and can only been controlled by international effort and agreement. Land use change happens on a local scale. Even though basins can be large and range across multiple jurisdictions, they usually span at most a few states. Often, land use is controlled at the level of the municipality. This contrast in the institutional scale of management suggests that land use change is an easier process to control, while climate is incredibly difficult. This is problematic, as these studies show that river systems are more sensitive to climate change than they are to land use change. Management that focuses solely on mitigating stress from land use change may, at best, blunt some of the

impacts of climate change. Yet, it is apparent that without action on carbon emissions river systems will be subject to substantial hydrologic changes in the future.

1.3 Research Objectives

- Develop a baseline watershed model for the Narragansett Bay watershed to simulate the hydrology and water quality of rivers under present conditions
- Simulate hydrology and water quality in the watershed under projected future climate conditions
- Simulate hydrology and water quality in the watershed under projected future land use conditions
- Simulate hydrology and water quality in the watershed under the combined effects of projected future changes in land use and climate
- Discuss management implications relative to the goal of maintaining the health of the Bay and the efficacy of current nutrient reduction efforts under conditions of changing climate and land use

CHAPTER 2

BASELINE MODELING OF THE NARRAGANSETT BAY WATERSHED

2.1 Study Area

The 142 mi^2 Narragansett Bay possesses a 1,820 mi^2 watershed comprised of most of Rhode Island and portions of Massachusetts. The watershed is 51% till and fine-grained stratified drift deposits that yield greater amounts of surface runoff during precipitation events. Coarse-grained stratified drift overlays another 35.2% of the watershed. The remaining 13.8% is covered by lakes, ponds, man-made reservoirs and wetlands. Mean annual temperature ranges from 46.8°F in Worcester to 50.3°F in Providence. Lowest monthly temperatures typically occur in January (23.3°F and 28.2°F for Worcester and Providence, respectively) and peak in July (69.9°F and 72.5°F). Precipitation averages 46 in (116.84 cm) annually with the greatest amounts in the northern and northeastern portions and the lowest in the central portion (Ries III, 1990). Pilson (2008) reports a slightly higher annual precipitation of 126 cm (49.61 in) in 2006, which may reflect differences in calculation, model errors or the influence of climatic change. Precipitation is fairly evenly distributed throughout the year, averaging 4in/month with greater precipitation in late fall/early winter than early to mid-summer (Ries III, 1990). Annual evapotranspiration from the watershed ranges from 40-55 cm/yr depending on rainfall, representing nearly half of annual precipitation (Pilson, 2008).

The three largest rivers in the basin are the Taunton, the Blackstone and the Pawtuxet. Together, these three rivers drain 75% of the watershed. The Taunton River is the largest, draining 562 mi^2 and the Blackstone is the second largest, draining 472 mi^2 (Ries III, 1990) from northern Rhode Island and central Massachusetts with a mean annual discharge of 24.4 m^3/s

(Nixon et al., 2008). The Pawtuxet River watershed is 232 mi^2 (Ries III, 1990) with a mean annual discharge of 11.7 m^3/s (Nixon et al., 2008), and drains central Rhode Island. The remainder of the watershed is drained by smaller rivers, including the Woonasquatucket, Moshassuck, Ten Mile, Palmer and Hunt Rivers. Rivers in the western half of the basin predominantly flow into the Providence River estuary, while those in the eastern half primarily flow into Mount Hope Bay.

Ries III (1990) estimates the annual mean runoff to Narragansett Bay during the 1987 water year at 137.62 m^3/s, ranging from an August low of 30.02m^3/s to an April high of 407.76m^3/s. The author further estimates long-term mean annual runoff at 104.49m^3/s, with long-term mean monthly runoffs ranging from 51.54m^3/s in July to 185.49m^3/s in March. Pilson (2008) calculates the average annual river flow between 1962 and 2004 at 90.9 m^3/s (the author used the watershed area given in Ries III (1990) in his calculations, but corrected it by subtracting the drainage area of the Sakonnet River). Vadeboncoeur et al. (2010) state that tributaries to Narragansett Bay have a mean combined flow of 100 m^3/s. Pilson (2008) also noted that the annual direct flow of sewage to the Bay is 5.1 m^3/s.

Due in large part to an industrial history and to public water supply, most rivers in the watershed possess dams, water diversions, or both. According to Ries III (1990), most damns have a negligible influence on flow because they are run-of-the-river dams and their reservoirs have little storage capacity. The author cites three reservoir systems that can affect short-term runoff to the Bay. The Scituate Reservoir is the largest and can substantially affect streamflow. The reservoir is located on the North Branch Pawtuxet River and maintains a storage capacity of 5,307,000,000 ft^3. The reservoir is the public water supply for Providence. The Flat River reservoir is located on the South Branch Pawtuxet River and has a capacity of 250,000,000 ft^3. Both reservoirs maintain large pools of water, with the effect that downstream flows are lower

than they would normally be during the higher-flow months of March and April and greater than they would during the summer months. Changes in storage can also affect annual mean runoff to the Pawtuxet River by up to 10% during some years. The West Hill Dam Reservoir has a storage capacity of 542,000,000 ft^3 on the West River in the Blackstone watershed. The reservoir retains flood waters that are released gradually. As a result, annual mean runoff is not typically affected and records from downstream gauging stations are adjusted to approximate natural flow. The most significant diversions occur in the Pawtuxet River, where the public water utility diverts an average of 101 ft^3/s. One-fourth of this flow is returned to the Pawtuxet above the Cranston gauge, while the remainder is returned below the gauge or directly to the Bay (Ries III, 1990).

Ries III's (1990) study cited 1986 U.S. Geological Survey data that showed that urban and suburban land areas comprised 23.3% of the watershed, with both land uses increasing. The major cities in the watershed are Worcester, Fall River, Taunton and Brockton, Massachusetts, and Providence, Woonsocket, Pawtucket, Cranston and Warwick, Rhode Island (Ries III, 1990). The most urbanized areas on the Bay are concentrated in the upper portion, reflecting the impaired condition of the upper Bay. Vadeboncoeur et al. (2010) cite the total population within the watershed in 2000 at 1,936,000. The USEPA (2006) reports that population in the 10 NOAA-designated coastal counties coincident with the Narragansett Bay watershed experienced a population growth rate of 24% from 1960-2000, and that the population density was 984 persons/mi^2. The watershed has an estimated 14% impervious cover (Watershed Counts, 2011).

2.2 Methods

2.2.1 Model Construction

This study utilized the Soil and Water Assessment Tool (SWAT) model to simulate hydrology and water quality for rivers within the Narragansett Bay watershed. The SWAT model was appropriate for this study as it is a dynamic process model, allowing for changes in climate and land use inputs over time to project future impacts. It is further useful for its ability to simulate water quantity, quality and sediment. Arnold et al. (1998) further highlight the benefits of using SWAT as it uses readily available inputs, is computationally efficient to operate on large basins (such as Narragansett Bay), and is capable of simulating long periods to compute the effects of management changes. Although SWAT was originally developed for use in agricultural watersheds, Franczyk and Chang (2009) successfully used it to simulate the hydrological effects of urbanization and climate change on the Rock Creek Basin in Oregon. Rock Creek Basin is a mixed use basin, with 13% agriculture, 15% forest and 60% developed. This is a greater level of urbanization than the Narragansett Bay watershed at 23.3% (Ries III, 1990), indicating that SWAT is appropriate for application to the Bay watershed.

I used the ArcSWAT2012 extension in ArcMap 10.1 to build the baseline model. The EPA BASINS program supplied requisite physical data, including elevation, stream centerlines and land use data [NLCD 2001]. Stream layers from MassGIS and RIGIS, merged with centerlines from BASINS, filled in gaps in stream continuity and expanded the stream network to include lower-order streams. That Natural Resources Conservation Service's STATSGO2 soils database provided soil layers for the region. I used seven weather stations to provide daily average precipitation and temperature data (Brockton, MA; East Milton, MA; Kingston, RI; New Bedford, MA; Providence, RI; Taunton, MA; Worcester, MA), retrieved from the National Climatic Data Center of the National Oceanic and Atmospheric Administration. When practical, I filled in

missing data with records from Weather Underground or interpolated from surrounding stations, though missing data points remain and SWAT attempts to fill them in with its weather generator from surrounding stations. The model ran at a daily time-step from 1/1/1949 to 12/30/2013, but analysis focused on monthly and annual averages for the period.

2.2.2 Model Calibration and Validation

I used three USGS stream gaging stations for monthly streamflow calibration and validation: Blackstone River at Woonsocket, RI (USGS01112500); Pawtuxet River at Cranston, RI (USGS01116500); Taunton River near Bridgewater, MA (USGS 01108000). These three stations represent stream gages on the three major tributaries to Narragansett Bay. I performed calibration only on the Blackstone River watershed, and then spatially validated the model using the Pawtuxet and Taunton watersheds. All calibration and validation ran for the full length of the model (65 years). Calibration began with manual calibration, adjusting appropriate hydrological parameters based on comparisons of the observed and simulated time series data. Because initial runs of the un-calibrated model revealed that the model underestimated baseflows, I focused calibration on parameters likely to influence baseflow, including soil and groundwater parameters. I then focused on matching peak flows by editing parameters that control surface runoff, including SCS curve number and those associated with evapotranspiration.

The final stage of calibration used Sequential Uncertainty Fitting (SUFI2) in the SWAT-CUP automatic calibration program, which attributes all model uncertainty to parameter uncertainties (Abbaspour, 2014). A description of the SUFI2 optimization algorithm is available in Abbaspour (2014). Table 2.1 displays calibrated parameters. In addition to altering parameters, I also altered the potential evapotranspiration equation from the default Penman-Monteith equation. Arnold et al. (1998) note that Penman-Monteith requires measured solar

radiation, wind speed and relative humidity, while Hargreaves' equation can provide realistic results without observed values. Because long-term data were not available for these parameters for the selected stations, I utilized Hargreaves.

I evaluated the model through the process articulated by ASCE (1993), first utilizing a visual comparison to provide a general overview of model performance before applying statistical criteria. I assessed goodness of fit during calibration and validation through three objective functions: coefficient of determination (R^2); Nash-Sutcliffe efficiency coefficient (NSE); and percent bias (PBIAS). While R^2, which describes the proportion of the variance explained by the model, is commonly used to evaluate models, it is over-sensitive to extreme values (Moriasi et al., 2007). Further, Krause et al. (2005) state that a major drawback of R^2 is that only the dispersion of the data is quantified, resulting in good R^2 values for models that systematically under- or over-predict all the time even when all predictions are wrong. Thus, Krause et al. (2005) maintain that multiple objective functions offer a more robust evaluation.

Nash-Sutcliffe efficiency is a normalized statistic that determines the relative magnitude of the residual variance compared to the measured data variance. The statistic ranges from $-\infty$ to 1.0, with 1.0 being optimal and values <0.0 indicating that the mean observed value is a better predictor than the simulated value (Moriasi et al., 2007). ASCE (1993) recommends using NSE for continuous hydrographs, and Moriasi et al. (2007) because it is commonly used, resulting in extensive information on reported values. However, Krause et al. (2005) does highlight a potential disadvantage of NSE in that it is not sensitive to quantification of systematic under-prediction and tends to be higher in catchments with higher dynamics. The final objective function, PBIAS, measures the tendency of simulated data to be larger or smaller than their observed counterparts (Moriasi et al., 2007). Positive values indicate model underestimation bias, whereas negative values indicate overestimation bias, with 0.0 the optimal value. Moriasi

et al. (2007) recommended PBIAS, in part, because it can clearly indicate poor model performance.

Table 2.1 Calibrated parameters

Parameter	Description
Groundwater Parameters	
GW_DELAY	Groundwater delay time (days): the lag between the time that water exits the soil profile and enters the shallow aquifer.
ALPHA_BF	Baseflow alpha factor: a direct index of groundwater flow response to changes in recharge
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur. Groundwater flow to the reach is allowed only if the depth of water in the shallow aquifer is equal to or greater than GWQMN
GW_REVAP	Groundwater revap coefficient: controls the movement of water from the shallow aquifer to the root zone in response to an overlying unsaturated zone.
REVAPMN	Threshold depth of water in the shallow aquifer for “revap” to occur.
Soil Parameters	
SOL_AWC	Available water capacity of the soil layer: plant available water calculated by subtracting the fraction of water present at permanent wilting point from that present at field capacity.
SOL_K	Saturated hydraulic conductivity: measure of the ease of water movement through the soil; reciprocal of the resistance of the soil matrix to water flow
ESCO	Soil evaporation compensation factor: determines the depth distribution used to meet the soil evaporative demand
Other Parameters	
EPCO	Plant uptake compensation factor: the amount of water uptake that occurs on a given day
IPET	Potential evapotranspiration method (Hargreaves method selected)
CN2	SCS curve number for moisture condition II
SURLAG	Surface runoff lag coefficient
CH_K(2)	Effective hydraulic conductivity in the main channel alluvium
CH_N(2)	Manning’s “n” value for the main channel

2.2.3 Uncertainties and Challenges

Modeling the Narragansett Bay watershed to predict future impacts to the Bay contains several uncertainties and challenges that require acknowledgement. As discussed above, flow restrictions and diversions can affect water quantity estimates. Although diversions do not change the total amount reaching the Bay, they can alter the flow of individual streams. Hamburg et al. (2008) report that the net effect of water drawn from the Scituate Reservoir and discharged from WWTFs is approximately 2.3 m^3/s of flow entering the Bay from plants on the Providence and Seekonk Rivers as opposed to the mouth of the Pawtuxet River. Nixon et al. (2008) report the mean annual discharge of the Pawtuxet River at 11.7 m^3/s, suggesting that 2.3 m^3/s is a substantial effect. Dams also can affect the quantity and timing of flow. As discussed above, damns on the North and South Branch Pawtuxet create large reservoirs that can substantially affect mean annual runoff and can noticeably impact seasonal flows. Ries III (1990) adjusted streamflow records to compute natural basin yields, and reported this resulted in an increase of annual mean runoff for the Blackstone of 20-25 ft^3/s and a decrease in the Taunton of 20-25 ft^3/s. Insufficient information regarding the amount and locations of diversions will create uncertainty in the final results, but Ries III (1990) reports that this uncertainty doesn't represent a large percentage of flow within the basin.

Nixon et al. (2008) suggest an uncertainty related to modeling nitrogen loading. They state that rivers are not pipes, and discuss how biogeochemical interactions at the water-atmosphere and water-sediment boundaries can lead to nitrogen storage or loss. They report that there is still substantial uncertainty about how factors such as distance residence time, depth and nitrogen concentrations influence these processes. The influence of WWTFs will also create uncertainty in nutrient loading estimates. Although the facilities have mandated nutrient reduction goals, these are subject to municipal action, and loading from plants is dependent on water consumption in the watershed. Consequently, the model will focus on non-point source

nitrogen simulated by the model, and examine relative differences in nitrogen loading to avoid projecting and incorporating WWTF effluent discharges. There is also uncertainty over future land use and climate change, as they are driven in part by economic, technological and political factors that are not within the scope of this project. This uncertainty is discussed in greater detail in the literature review.

2.3 Results

2.3.1 Calibration and Validation

Calibration of the Blackstone watershed yielded an R^2= 0.65 (Table 2.2). Krause et al. (2005) note that because R^2 values may not adequately capture systematic under- or over-prediction, it is important to consider the gradient *b* of the regression. In this instance, *b*= 0.71, reflecting 29% under-prediction. Visual comparison reveals this under-prediction occurring consistently at the peaks. The calibrated model produced an NSE= 0.64 and a PBIAS= -2.7. Through an extensive review of published literature on calibration and validation, Moriasi et al. (2007), sought to establish standardized guidelines for evaluating model performance. They determined a performance rating of 'satisfactory' for $0.50<NSE\leq0.65$ and 'good' for $0.65<NSE\leq0.75$ at a monthly time-step. The calibration results of the Blackstone fall close to the line between the two, indicating acceptable model performance. They label PBIAS $<\pm10$ as 'very good', suggesting accurate model simulation.

Validation on the Pawtuxet and Taunton River watersheds yielded acceptable objective function values, though expectedly lower than calibration values. Validation on the Pawtuxet River gave an R^2= 0.61, NSE= 0.52 and PBIAS= -8.9. Validation on the Taunton River produced R^2= 0.62, NSE= 0.61 and PBIAS= -4.4. These values indicate that the model adequately simulates the streamflow response of the Upper Narragansett Bay watershed.

Table 2.2 Calibration and validation values

River	R^2	NSE	PBIAS
Blackstone	0.65	0.64	-2.74
Pawtuxet	0.61	0.52	-8.91
Taunton	0.62	0.61	-4.36

2.3.2 Hydrology

Under the baseline simulation, the basin received and average of 1188.2mm of precipitation per year, 150.42mm as snowfall. This falls within the 116.84-126cm range reported by Ries III (1990) and Pilson (2008). Rainfall was greatest in spring and fall (Figure 2.1).

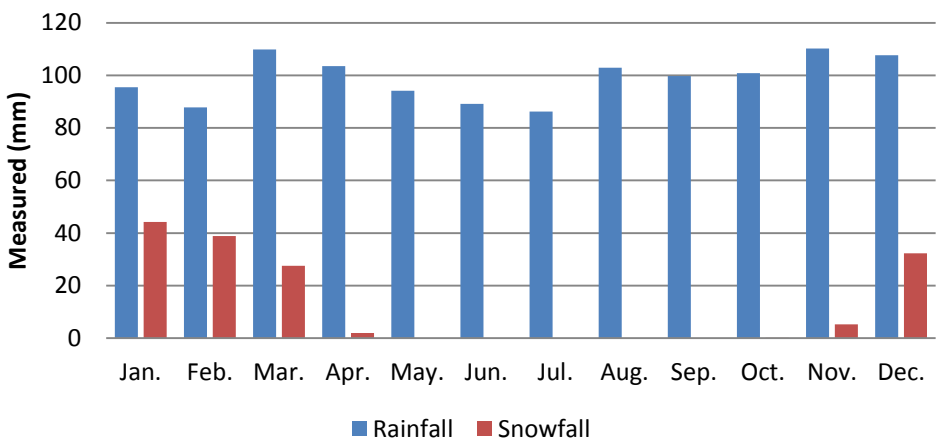

Figure 2.1 Average monthly precipitation in the Narragansett Bay Basin under baseline conditions

The average annual water yield (water yield = surface runoff + groundwater flow + lateral soil flow) was 716.86mm. Groundwater contributed 67% of average annual yield, with surface runoff contributing 30% and lateral soil flow contributing 3% (Figure 2.2). This indicates that groundwater flow has a greater impact on the basin's hydrology than surface runoff. A regression supports this claim, with the relationship between water yield and groundwater flow producing an R^2=0.85, as opposed to R^2=0.5 for surface runoff. However, there is substantial

seasonal variation (Figure 2.3), with surface runoff dominating during winter and groundwater during the remainder of the year. Surface runoff has the strongest relationship with water yield in January (R^2=0.72) and the weakest during November (R^2=0.14), while groundwater flow has the strongest relationship during June (R^2=0.83) and the weakest during February (R^2=0.10).

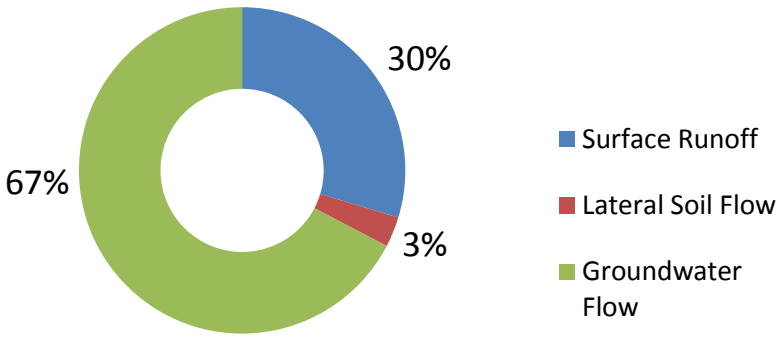

Figure 2.2 Contribution of surface runoff, groundwater flow and later soil flowto total water yield under baseline simulation

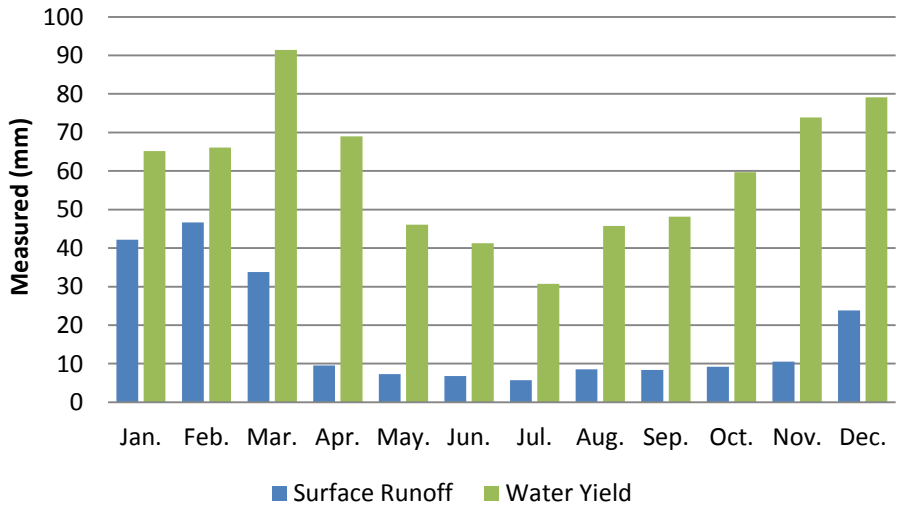

Figure 2.3 Average monthly surface runoff and total water yield in the Narragansett Bay Basin

The relative contributions of surface runoff and groundwater flow to water yield also vary spatially, with greater surface runoff in urbanized areas and less in forested areas. For

example, in heavily urbanized sub-basins in Worcester, MA, and south Providence, RI, the relationship between monthly average surface runoff and water yield in June produced an R^2=0.95 and R^2=0.9967, respectively. This indicated that in urbanized areas water yield is derived almost exclusively from surface runoff, even in June when the contribution of surface runoff basin-wide is very low (R^2=0.26). Conversely, in a mostly forested sub-basin in Foster, RI, the relationship between monthly average surface runoff and water yield in January was R^2=0.61, lower than the overall basin value of R^2=0.72. Thus, while the hydrology of the overall basin is driven by groundwater flow more than surface runoff, this varies temporally (seasonally) and spatially (urban vs. rural areas).

Average annual potential evapotranspiration (PET) was 943.5mm, and actual evapotranspiration (ET) was 390.7mm, in line with values reported by Pilson (2008). Both PET and ET also varied seasonally. Actual evapotranspiration was substantially lower than PET during summer, indicating a lack of sufficient soil moisture.

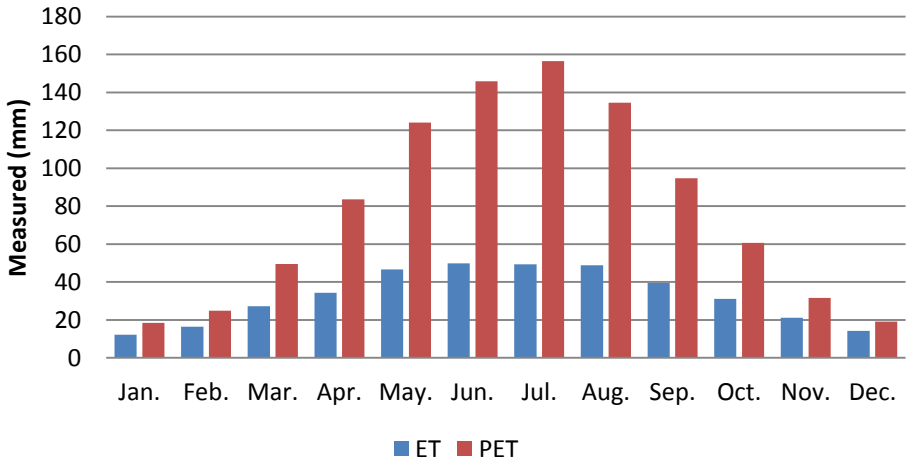

Figure 1.4 Average monthly evapotranspiration (ET) and potential evapotranspiration (PET) in the Narragansett Bay Basin

Because the study concerned the potential impacts changes in the hydrology and water quality of rivers in the Narragansett Bay watershed could have on restoration goals for the Bay, I considered streamflow at the outlet of the Bay's important tributaries: Pawtuxet River; Woonasquatucket-Moshassuck River; Blackstone River; Ten Mile River; Taunton River (Figure 2.5).

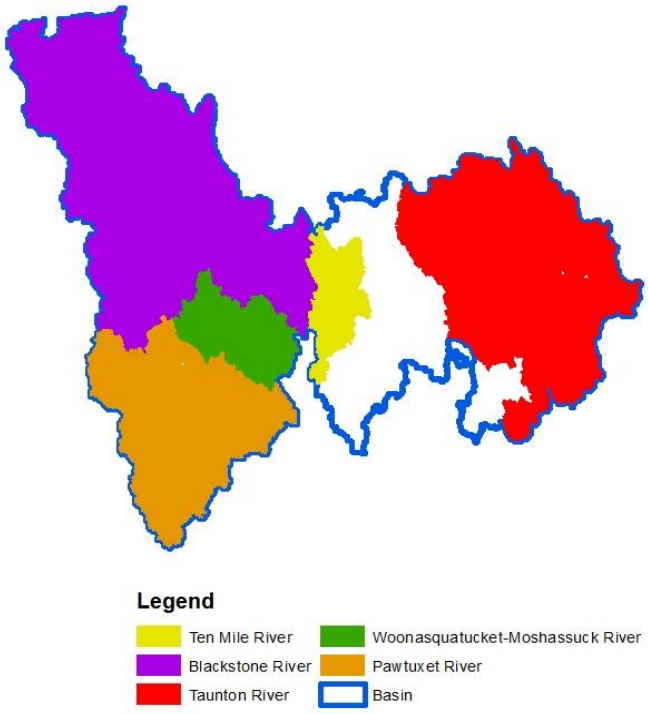

Figure 2.5 Fiver major watersheds studied: Ten Mile, Blackstone, Taunton, Woonasquatucket-Moshassuck and Pawtuxet Rivers

I did not consider streamflow at the outlets of smaller watersheds, such as Rocky Run, Cole River, Three Mile River and the Segreganset River. Flows were greatest from the Blackstone and Taunton Rivers and lowest from the Ten Mile and Woonasquatucket-Moshassuck Rivers. Streamflow for all five rivers was highest in March and lowest in July, consistent with simulated water yield patterns for the basin (Figure 2.6).

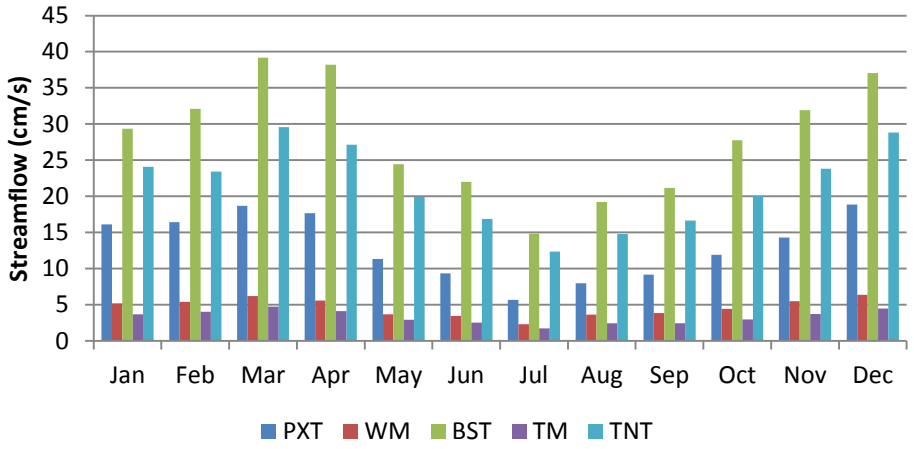

Figure 2.6 Average monthly streamflow for the five major rivers in the Narragansett Bay Basin (PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton)

2.3.3 Water Quality

I examined loadings of sediment, nitrogen and phosphorus from the watershed. Average annual sediment loading from the basin was 0.543 metric tons/ha, and loadings of organic nitrogen (N) and phosphorus (P) were 1.74 kg/ha and 0.25 kg/ha, respectively. Average annual nitrate (NO_3) loading in surface water was 1.26 kg/ha. Loadings were correlated with surface runoff (sediment R^2=0.63; organic N R^2=0.68; organic P R^2=0.72), but showed very weak correlation with overall water yield (sediment R^2=0.21; organic N R^2=0.22; organic P R^2=0.28), indicating that sediment and nutrient loading to streams in the watershed was driven predominantly by surface runoff. Further, N and P loadings were strongly correlated with sediment yields, R^2=0.83 and R^2=0.84, respectively. This is expected, as overland flow is likely to erode and carry sediment to streams, and nutrients are typically associated with sediment (Brooks et al., 2003). Although loadings correlated with surface runoff, and there was a strong relationship between the SCS runoff curve number of a sub-basin and surface runoff (R^2=0.87), there was no detectable relationship between curve number and loading. This suggests that additional watershed attributes determine loading volumes.

Looking at the five main rivers, average annual sediment, organic N and nitrate loading were greatest in the Taunton River and lowest in the Ten Mile River (Table 2.3). Average annual organic P loading was greatest in the Blackstone River and lowest in the Ten Mile River. Loading exhibited some positive correlation with streamflow in these rivers (sediment R^2=0.51; organic N R^2=0.56; organic P R^2=0.48, NO_3 R^2=0.87), which partially explains differences in loading volumes between rivers. However, the correlation values indicate that other watershed characteristics also likely played an important role.

Table 2.3 Average annual loading of sediment and nutrients from the five major rivers. PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

Average Annual Loading				
	Sediment (metric tons)	Organic N (kg)	Organic P (kg)	NO_3 (kg)
PXT	1461.785	1465.072	372.0589	74893.88
WM	710.689	661.0138	284.3944	39342.33
BST	3863.092	4687.651	858.3771	163724.5
TM	207.2862	617.4441	129.3531	34980.6
TNT	4028.054	5884.406	810.453	165321.6

Normalized loading data that removes the influence of streamflow demonstrates that some watersheds were more susceptible to sediment and nutrient loading (Figures 2.7 & 2.8). The Taunton both delivered the greatest volume of sediment and organic N, and had the greatest loading per unit streamflow. However, while the Woonasquatucket-Moshassuck River delivered a much lower volume of sediment than the Blackstone or Pawtuxet Rivers, it had greater sediment loads per unit streamflow. Although the Ten Mile River delivered the lowest

volume of organic N, it had the second greatest loading per unit streamflow. The Pawtuxet River delivered greater total volume of organic N than the Ten Mile or Woonasquatucket-Moshassuck Rivers, but had the lowest loading per unit streamflow of all five rivers. The Blackstone and Taunton Rivers delivered the greatest volume of organic P to the Bay, but the Woonasquatucket-Moshassuck and Ten Mile Rivers delivered the greatest loading per unit streamflow (respectively). A similar dynamic exists for nitrate, where the Ten Mile and Woonasquatucket-Moshassuck Rivers delivered the greatest volume per unit streamflow, despite small total contributions. Both the Ten Mile and Woonasquatucket-Moshassuck River watersheds are relatively small, but heavily urbanized. Their greater loading per unit streamflow has important management implications. Although they had a high rate of loading, their relatively low volume of streamflow results in relatively small contributions to the upper Bay compared to the three major tributaries. However, changes to the watershed that have the potential to increase streamflow subsequently have the potential to increase loading, and a streamflow increase from these two watersheds may have a greater impact on total loading to the Bay than similar increases in streamflow of some of the larger rivers.

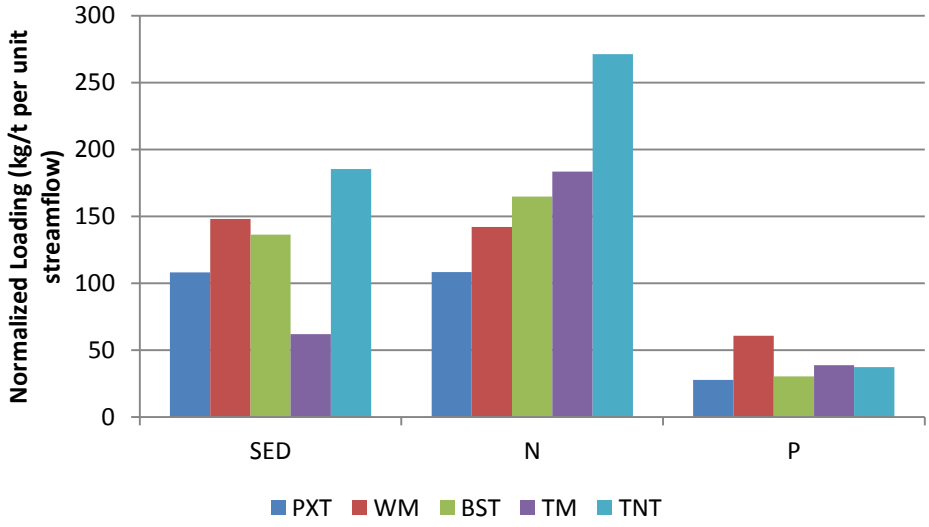

Figure 2.7 Normalized average annual loading of sediment, organic nitrogen (N) and organic phosphorus (P). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

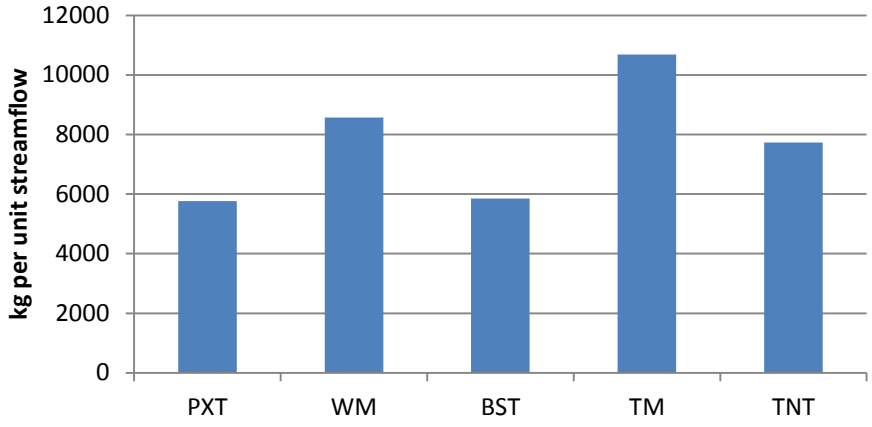

Figure 2.8 Normalized average annual loading of nitrate (NO_3). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

CHAPTER 3

CLIMATE CHANGE IMPACTS TO THE NARRAGANSETT BAY BASIN

3.1 Methods

To simulate the impacts of climate change on water quantity and quality, I added temperature and precipitation trends projected under climate change scenarios to the baseline model. Studies typically introduce climate changes projected by the IPCC SRES scenarios, which make assumptions of future greenhouse gas emissions based on narratives that incorporate socioeconomic, population and land use parameters (see Ch. 1.2.1). However, the Fifth Assessment Report (IPCC, 2013) abandoned the SRES scenarios in favor of Representative Concentration Pathways. As opposed to SRES, the RCP scenarios are projections of only radiative forcing that could result from any combination of socioeconomic and technological scenarios. The four RCPs are named according to their total radiative forcing in 2100: RCP3-PD2, RCP4.5, RCP6 and RCP8.5. RCP3-PD2 represents a best-case scenario, where radiative forcing peaks at ~3W/m^2 before 2100 then declines, while RCP8.5 is a worst-case scenario where a continually rising pathway leads to radiative forcing of 8.5W/m^2 in 2100. Scenarios RCP6 and RCP4.5 are the middle-of-the-road scenarios, with radiative forcing at 6 and 4.5 W/m^2 respectively at 2100 and stabilization after 2100 (WMO, n.d.).

I chose to utilize a middle-of-the-road scenario. Temperature and precipitation projections at 2080 were very similar for the northeast between RCP4.5 and RCP6. I selected the 50th percentile values for 2080-2100 from RCP4.5 for the climate trend to incorporate into the baseline model: +3°C and +10% precipitation. To add climate projections, I de-trended the temperature and precipitation data series from the baseline model and added the climate trend. A detailed description of this method can be found in Marshall and Randhir (2008). I then re-ran

the model with the altered climate trend and compared water quantity and quality outputs to the baseline to determine expected changes.

3.2 Results

3.2.1 Basin Impacts

Under the altered climate simulation, all hydrological parameters increased (Figure 3.1). Groundwater flow experienced the greatest average annual increase (24.06%), with surface runoff increasing by 9.47%. Total water yield increased 19.71%, with groundwater increasing its proportional contribution to 70% and surface runoff's contribution decreasing to 27%. Both PET and ET increased by 4.36% and 13.03%, respectively. The larger percent increase in ET relative to PET likely resulted from greater amounts of soil moisture available for evapotranspiration due to increased precipitation.

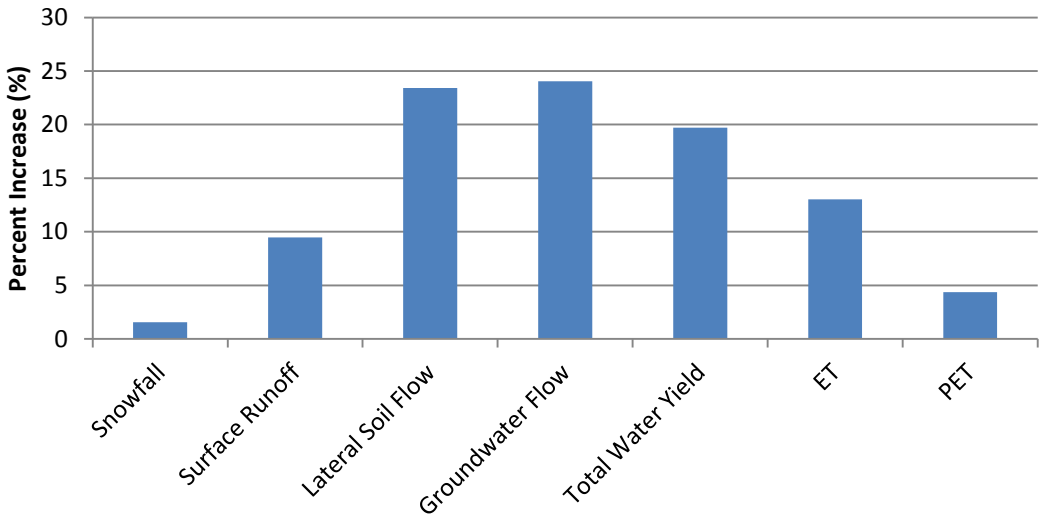

Figure 2.1 Percent increase in average annual hydrological parameters

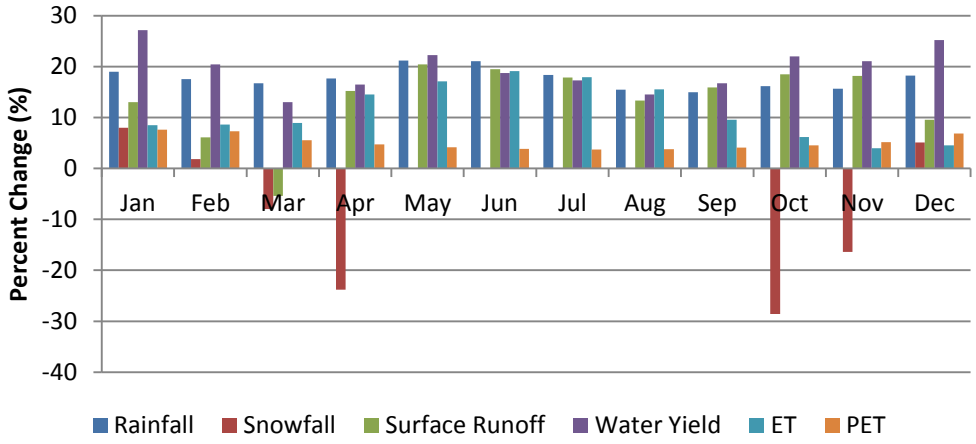

Figure 3.2 Percent change in average monthly hydrological parameters

There was substantial monthly variation, with the greatest increases in precipitation occurring in late spring and early summer (Figure 3.2). This contradicts many reports that the northeast will see greater winter and fall precipitation and less summer precipitation (e.g. Tu, 2009). However, the Fifth Assessment Report projects a 10% increase in April-September precipitation between 2081 and 2100, although with some uncertainty (IPCC, 2013, AISM-46). Snowfall increased under the altered climate simulation in December, January and February, but decreased in March, April, October and November. This is likely a result of increased temperatures that will shift early spring and late fall precipitation from snow to rain. However, despite less snowfall and more rain in March, surface runoff also decreased even while water yield increased. This is an unusual result, and may be related to the change in snowfall during March. Potential evapotranspiration increased most during the winter, resulting from higher temperatures. Actual evapotranspiration increased most during the summer, reflecting greater water availability for ET to occur. The altered climate simulation produced dramatic increases in sediment and nutrient loading (Figure 3.3). Basin-wide, sediment yield increased 43.65%,

organic N and P increased 26.91% and 25.6%, respectively, and nitrate loading increased 10.54%.

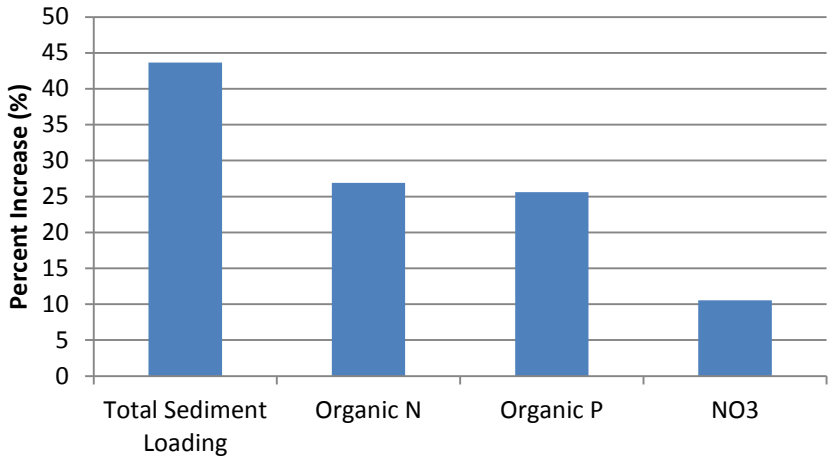

Figure 3.3 Percent increase in average annual water quality parameters (total sediment loading, organic N and P, nitrate [NO_3])

Average annual and average monthly changes only tell part of the story, as they average parameter values over the full simulation length. Yet, climatic changes occur over time, with impacts increasing over the course of the century. Thus, the magnitude of the change between baseline and climate scenarios will increase over time. To highlight this pattern, I examined differences in impacts between the first and last decade of the simulation (Figure 3.4). Impacts were relatively minor over the first decade. For example, during the first decade of the simulation, there was a statistically significant difference between the data series for surface runoff ($p=0.03$) and water yield ($p<0.001$), but not for sediment yield ($p=0.18$), organic N ($p=0.47$) or organic P ($p=0.38$). However, during the last decade there was a significant difference ($p<<0.001$) between baseline and altered data for all parameters. Thus, while average annual data show increases for all studied parameters, time series analysis reveal that impacts start very small and magnify considerably over time.

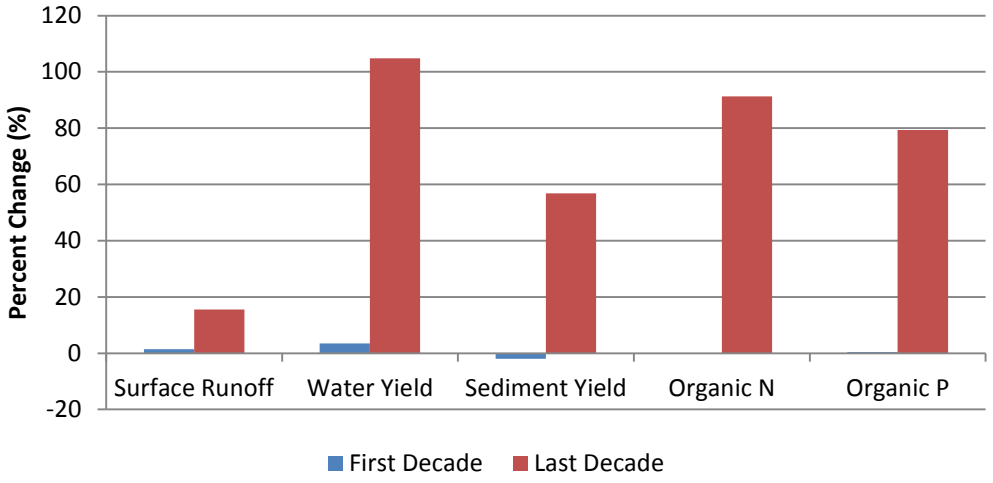

Figure 3.3 Percent change in water quantity and quality parameters in the first and last decade of the simulation.

3.2.2 River Impacts

All five main tributaries experienced similar increases in average annual streamflow, ranging from +18.16% for the Taunton River to +23.05% for the Blackstone River (Figure 3.5). The difference between average annual streamflow between baseline and climate simulations was statistically significant for all rivers ($p<<0.001$). The greatest increases occurred in January for all rivers, but there was more seasonal variation for the smallest increases: August for the Pawtuxet and Woonasquatucket-Moshassuck Rivers; September for the Blackstone River; March for the Ten Mile River; April for the Taunton River (Figure 3.6a). However, these values are averaged over the 65-year time period. Yet, as previously stated, climatic changes (temperature and precipitation increases) amplify over time. Figure 3.7a demonstrates that the percent change between average annual streamflow increases over time, with minimal initial increases versus larger increases towards the end of the model run. These later percent increases can be much larger than the overall mean; in year 64, flow in the Pawtuxet is 58.15% higher than under the baseline simulation.

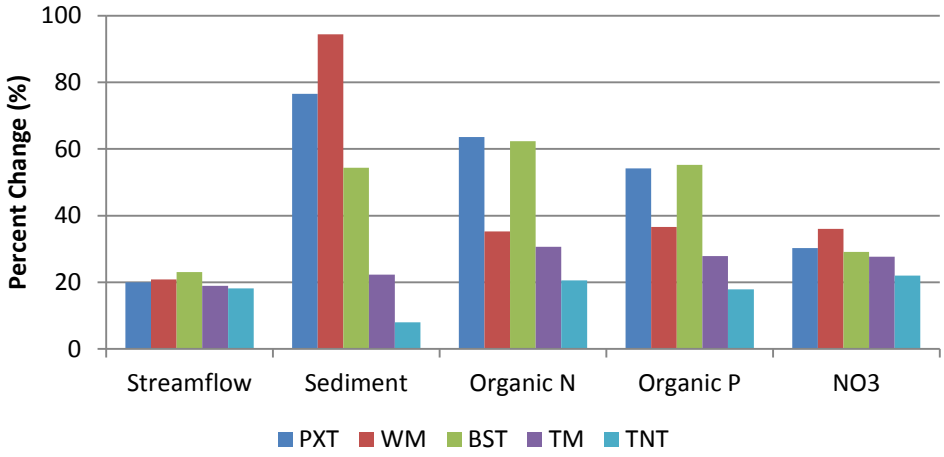

Figure 3.4 Percent change in average annual streamflow and water quality parameters (PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton).

All rivers experienced increased average annual sediment loading, but there was greater variation than exhibited by streamflow (Figure 3.6b). Percent change in sediment loading ranged from + 7.95% for the Taunton River to +94.36% for the Woonasquatucket-Moshassuck River. The change was statistically significant for all rivers, though much weaker in the Taunton ($p=0.017$ for the Taunton River, compared to $p<<0.001$ for all others). There was substantial monthly variation, with the greatest increases for the Pawtuxet and Woonasquatucket-Moshassuck Rivers in January, the greatest increases for the Blackstone and Ten Mile Rivers in December, and the greatest increase for the Taunton River in October. Increases were lowest for all rivers during the summer except for the Taunton River, which experienced a 6.80% decrease in sediment loads in March.

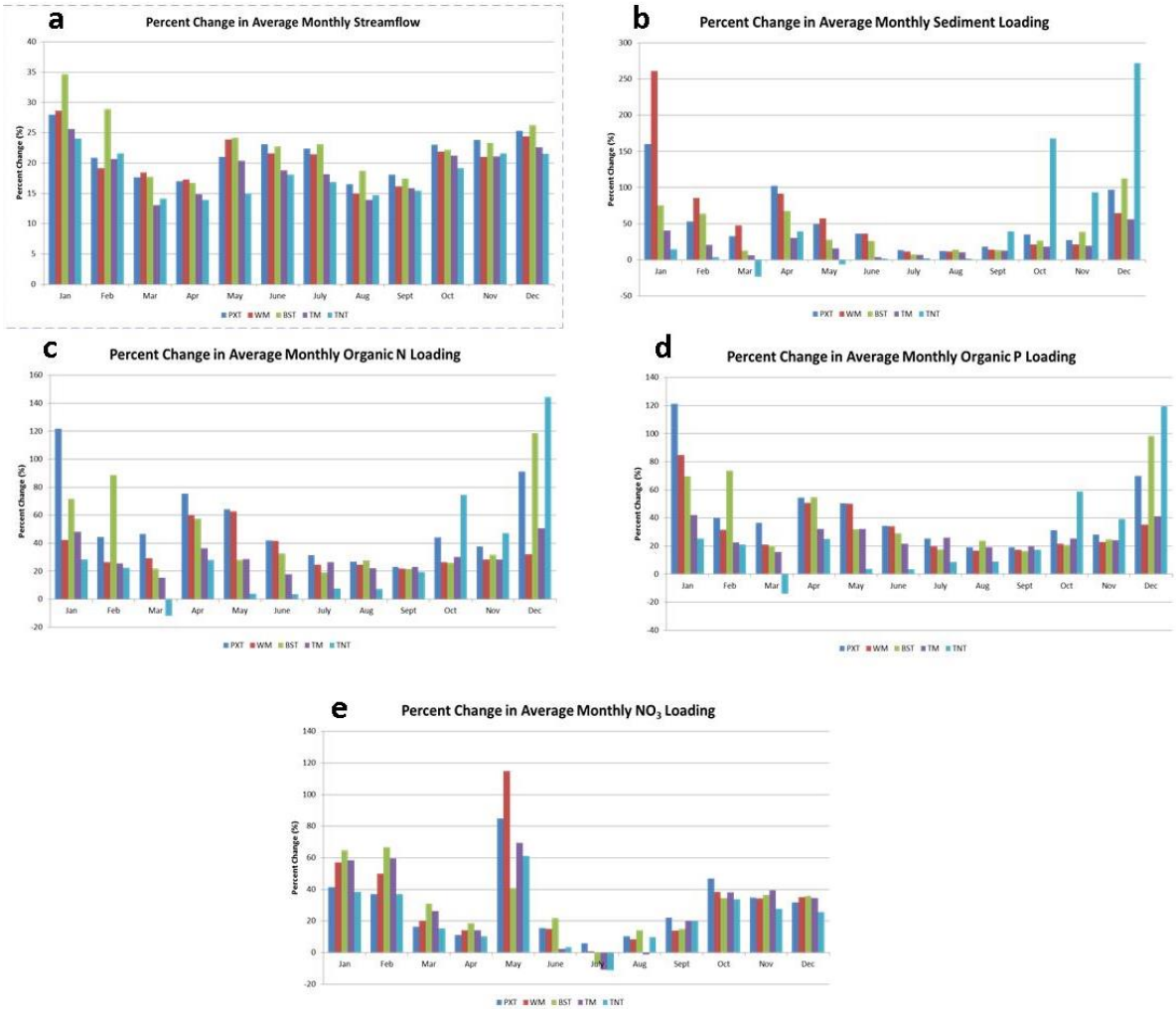

Figure 3.5 Percent change in monthly averages for streamflow (a), total sediment loading (b), organic N loading (c), organic P loading (d) and nitrate loading (e). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

Figure 3.7b displays the considerable volatility of percent change in average annual sediment loading over time. While there is a discernible positive trend over time, there is considerable variation around the trend for each river. Further, while all rivers display a similar streamflow pattern over time, the rivers display very different behavior over time with regard to sediment loading. Figure 3.7b also highlights the relatively small positive trends for the Taunton and Ten Mile Rivers (*b*=0.78 and *b*=0.95, respectively) relative to the stronger trends for the Pawtuxet and Woonasquatucket-Moshassuck Rivers (*b*=2.74 and *b*=3.07, respectively).

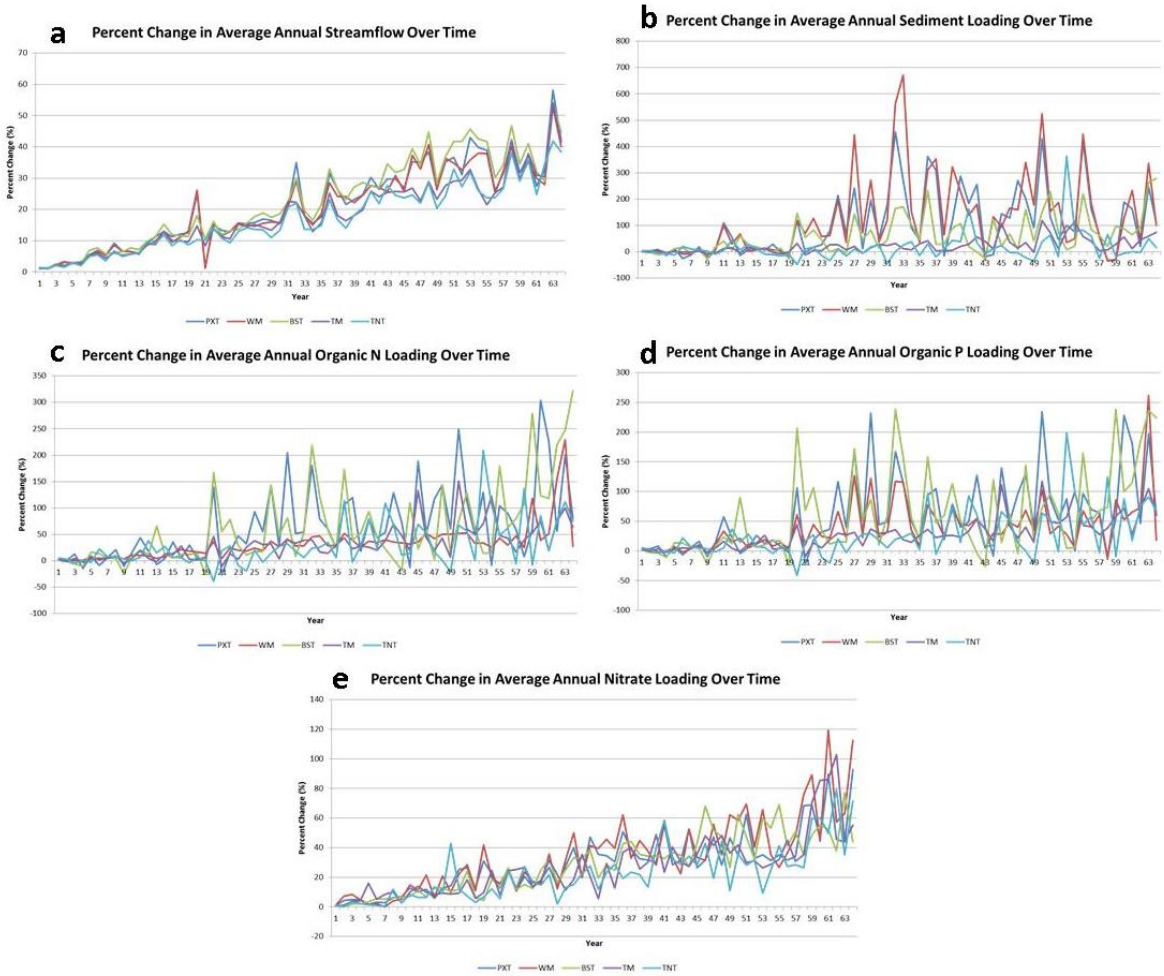

Figure 3.6 Percent change over time in annual averages for streamflow (a), total sediment loading (b), organic N loading (c), organic P loading (d) and nitrate loading (e). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

Nutrient loading displayed less variation between rivers and over time than sediment loading (Figure 3.6). The change between the baseline and climate simulations was statistically significant for all nutrient parameters for all rivers ($p<<0.001$). Percent change in organic N and P loading was greatest in the Pawtuxet and Blackstone Rivers and lowest in the Taunton River. Percent change for both nutrients exhibited seasonal variation, with the greatest increases in winter and spring and smallest during summer, with the exception of March for the Ten Mile and Taunton Rivers (the latter saw negative percent change in march for both nutrients).

Percent change in average annual nitrate loading was greatest in the Woonasquatucket-Moshassuck River and smallest in the Taunton. Nitrate displayed a similar monthly pattern to organic N and P with the greatest increases occurring in winter and fall, but with a large spike for all rivers in May. Figure 3.7 displays the positive trend over time in organic N and P, and nitrate loading, albeit with considerable noise around the trend.

The Taunton River represents a perplexing result. Under the baseline simulation, the Taunton River delivered the greatest volume of sediment and nitrogen to the Bay, and the second-greatest volume of phosphorus. Further, it also had the greatest sediment and organic N load per unit streamflow. Yet, it maintained the smallest percent change for sediment and nutrients, despite a similar streamflow increase to the other rivers. It is possible that present-day sediment loads are already approaching the maximum loading rate, so additional streamflow increases will have a minimal impact. In contrast, the Woonasquatucket-Moshassuck River demonstrates an expected result. As stated in Ch. 2.2.3, the high loading rate per unit streamflow indicates that streamflow increases could have a large impact on sediment loads. Figure 3.6 highlights this result, with the Woonasquatucket-Moshassuck River displaying the largest percent increases in annual sediment and nitrate loading. A similar result, however, did not occur for phosphorus loading, of which the Woonasquatucket-Moshassuck River also had the greatest loading rate. Despite the Pawtuxet having relatively low loading rates of sediment and nutrients, it demonstrated very high percent increases. These examples reinforce the idea that while streamflow likely has some influence on sediment and nutrient loading, other watershed factors also exert considerable influence.

CHAPTER 4

LAND USE CHANGE IMPACTS TO THE NARRAGANSETT BAY BASIN

4.1 Methods

To simulate the impacts of expected land use change on water quantity and quality in the basin, I replaced the NLCD2001 land use with raster data for projected 2080 land use. I retrieved the projected land use raster from the Human-Environment Modeling & Analysis Laboratory's National Land Transformation Model. This project utilized the Land Transformation Model (LTM) coupled with drivers, including population, urban density, and nearest-neighbor distance to existing urban areas, highways and roads, to project urban expansion across the United States over the twenty-first century. Greater detail regarding the methods behind projecting 2080 land use is available in Piourde, Doucette and Braun (2013). I resampled the data to match the resolution and extent of the original land use raster and ran the simulation under baseline conditions with the new land use data.

4.2 Results

4.2.1 Projected Land Use Changes

Basin-wide changes in land use between present and 2080 were relatively small (Figure 4.1; Table 4.1). The proportion of the watershed covered by urban development decreased by 0.23%. However, reductions in developed open space (-0.22%) and high-density development (-0.11%) drove this decrease. Low-density and medium density urban development increased their share of the watershed by 0.08% and 0.02%, respectively. The overall decrease in urban space and increase in forested areas contradicts the current trend of second-growth forests converted to suburban residential developments cited by Hamburg et al. (2008). However, the LTM model is sensitive to population, and US Census (2014) data highlights Rhode Island's decreasing population trend. Further, the loss of high-density urban to forest supports the

current trend of suburbanization. While the land cover changes were small across the basin, they could have significant local impacts, especially in small regional watersheds.

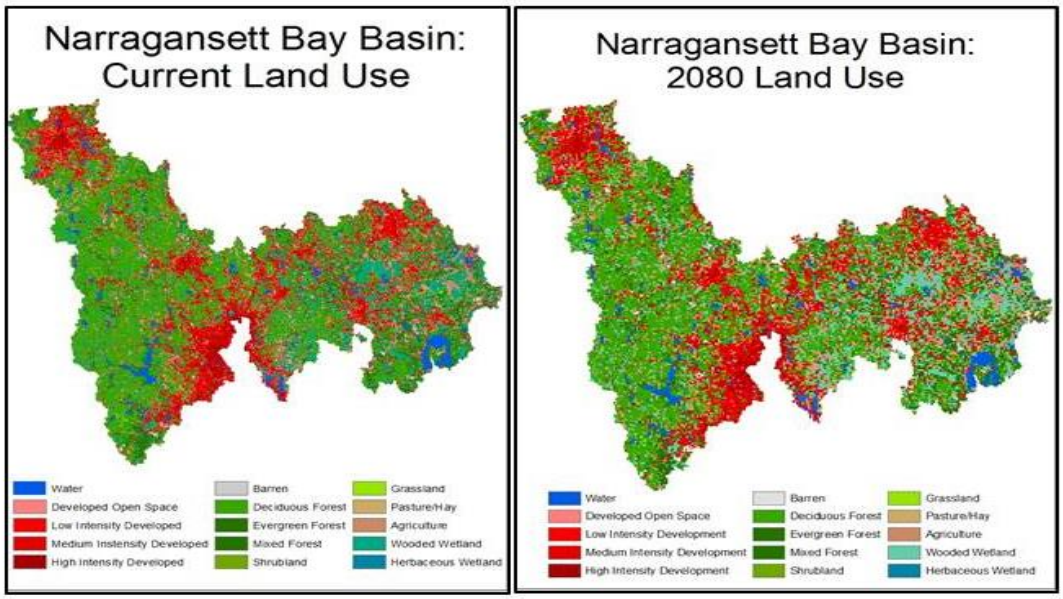

Figure 4.1 Land cover in the Narragansett Bay Basin at current (left) and projected 2080 (right).

Table 4.1 Change in percent coverage of land use type in the Narragansett Bay Basin between present and 2080

Land Use	Change in Coverage
Developed-Open	-0.22%
Developed Low	0.08%
Developed Med	0.02%
Developed High	-0.11%
Forest	0.16%
Grassland/Shrubland	0.06%
Pasture/Agriculture	-0.02%
Wetland	-0.09%

There was substantial variation between the five major river basins (Table 4.2). Urban development increased in the Pawtuxet River watershed, driven by developed open space, low-

density and high-density development (+0.47%, +0.27% and +0.29%, respectively), while medium density development declined by 0.1%. Forest cover decreased by 0.5%, while wetland area decreased by 0.21%. The Pawtuxet exhibited the greatest increase in developed area and the greatest decrease in undeveloped area. Conversely, urban area decreased 2.5% in the Woonasquatucket-Moshassuck watershed, while undeveloped areas increase by 1.9%. Developed open-space increased slightly (0.47%), but all other urban development decreased, while forestland and grassland/shrubland both increased. The Ten Mile watershed displayed an increase in undeveloped area and a decrease in developed area, while the Taunton River watershed demonstrated the opposite. Both developed and undeveloped areas increased in the Blackstone River watershed, with those increases offset by decreased agricultural area. Thus, while the Basin experienced small percent changes, at the watershed level there were substantial differences that result in differential impacts.

Table 4.2 Change in percent coverage of land use type in the Narragansett Bay Basin between present and 2080. PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

Watershed	Developed Open	Developed Low	Developed Medium	Developed High	Forest	Grassland/ Shrubland	Pasture/ Agr.	Wetland
PXT	0.468	0.272	-0.099	0.291	-0.512	-0.007	-0.115	-0.209
WM	0.723	-1.328	-1.371	-0.532	1.668	0.246	0.220	-0.011
BST	-0.254	0.245	0.092	-0.034	0.303	0.214	-0.264	-0.291
TM	-1.566	0.115	-0.311	0.326	0.697	0.127	-0.199	0.002
TNT	0.023	0.010	0.466	-0.216	-0.282	-0.096	0.081	-0.073

4.2.2 Basin Impacts

Under the projected land use scenario, total annual water yield decreased 3.17%, surface runoff decreased 1.18% and groundwater flow decreased 3.78%. Looking at monthly values (Figure 4.2), evapotranspiration exhibited large increases under 2080 land use, while surface runoff and water yield showed consistent reductions. These can both likely be attributed

to increased forest cover. Greater forest cover alters the curve number to produce less surface runoff as more water is retained in the soil. This storage, and the additional vegetation, can then also be used to meet potential evapotranspiration demand, resulting in greater evapotranspiration from the basin and a lower water yield. There was substantial monthly variation, with ET highest in spring and late fall, and reductions in surface runoff and water yield greatest in late spring and early summer. Water quality impacts were more varied. Average annual total sediment load from the basin increased 16.39%, while organic N and P increased only slightly at 3.22% and 3.20%, respectively. Nitrate loading decreased by 0.63%.

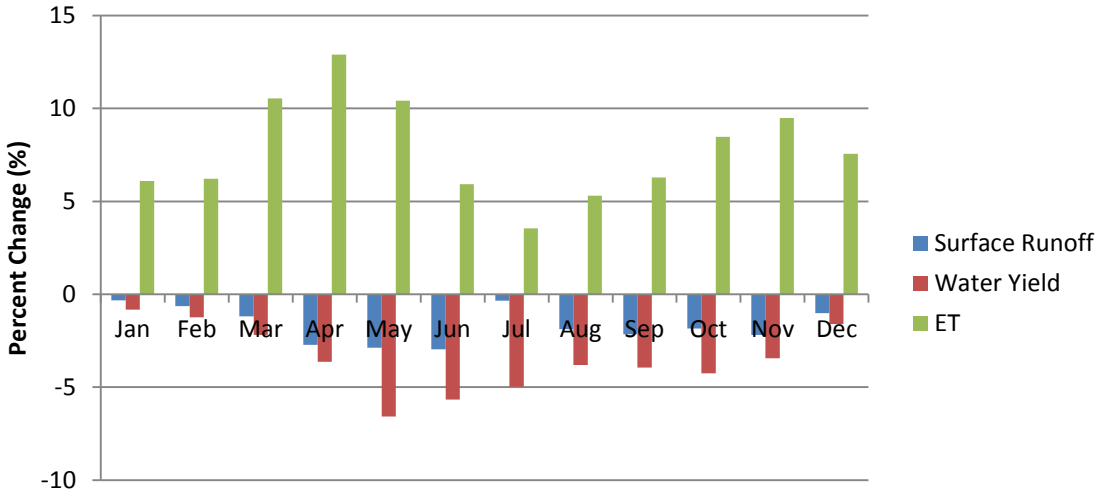

Figure 4.2 Percent change in average monthly values for Narragansett Bay Basin

4.2.3 River Impacts

Impacts at the individual watershed were greater and more diverse (Table 4.3). Average annual streamflow decreased in the Pawtuxet and Woonasquatucket-Moshassuck River watershed, but increased in the Blackstone, Ten Mile and Taunton River watersheds. There was substantial monthly variation, with only the Woonasquatucket-Moshassuck experiencing average decreases every month and only the Ten Mile displaying average increases every month. There was no discernible trend across all five watersheds. The Woonasquatucket-

Moshassuck experienced the most substantial impacts from land use change, which reflects the greater amount of change in that watershed relative to others (see Table 4.2). It is likely that the streamflow reduction is a result of the 2.51% decrease in urban area, which translates to reduced surface runoff, and the 1.90% increase in forest area, which decreases streamflow via soil storage and transpiration.

However, the Ten Mile River watershed also experienced decreased urban development and increased forest cover, but exhibited monthly streamflow increases. Further, the Pawtuxet River displayed reductions in streamflow (second only to the Woonasquatucket-Moshassuck), but experienced the largest increase in developed area and the greatest decrease in forest cover. It should be noted that the decrease in urban area in the Ten Mile river was largely driven by the loss of developed open land, which is likely to behave more similarly to natural conditions than urban conditions, while experiencing increases in low-density and high-density urban development, which could increase surface runoff due to increased impervious cover to a greater degree than increased forest cover would reduce it. In contrast, the reduction in developed areas in the Woonasquatucket-Moshassuck was the result of decreases in low- medium- and high-density urban development, while developed open space increased. Thus, while both watersheds experienced increased forest cover at the expense of development, the type of development likely had a significant impact on resulting streamflow. In the Pawtuxet watershed, most of the increased development was the result of increased developed open space, which could mitigate some of the impact of the increases in low- and high-density urban development. The Blackstone and Taunton Rivers showed little impact on streamflow, reflecting the minimal changes in land use.

Table 4.3 Percent change in average monthly values relative to baseline conditions. PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Streamflow	PXT	0.25	-0.46	-0.03	-0.23	-1.27	-1.62	-2.69	-3.86	-3.30	-2.13	-1.02	0.17	-2.19
	WM	-1.00	-0.92	0.48	-0.58	-2.35	-2.60	-6.20	-5.85	-5.45	-2.63	-1.86	-0.01	-1.86
	BST	0.39	0.49	0.15	0.04	-0.14	0.13	0.32	-0.38	-0.71	0.19	0.34	0.21	0.20
	TM	0.39	0.07	0.66	0.46	0.39	0.82	1.14	0.79	0.30	0.72	0.09	0.81	0.51
	TNT	0.94	0.49	0.80	0.66	0.21	-0.03	0.25	0.16	0.06	0.09	0.21	0.52	0.48
Sediment	PXT	112.36	66.52	61.66	12.56	-7.43	-11.03	-14.24	-13.55	-12.34	21.48	13.36	22.72	63.21
	WM	-32.85	-32.11	-33.07	-24.66	-27.55	-26.17	-26.75	-26.26	-24.97	-26.44	-25.80	-31.72	-32.02
	BST	17.00	14.67	13.86	12.07	-1.38	5.48	3.74	4.19	4.19	2.75	2.20	-5.45	12.58
	TM	53.97	49.54	38.42	1.11	-3.84	-3.87	-5.54	-4.97	-5.41	-2.21	-3.04	3.58	26.27
	TNT	26.03	24.54	25.22	8.25	3.95	-3.68	-7.12	-4.17	-3.59	0.03	-1.29	4.76	24.12
Organic N	PXT	19.86	9.77	5.88	-6.28	-10.92	-11.38	-12.21	-13.63	-12.09	-2.95	-8.16	-6.68	3.74
	WM	-18.46	-17.05	-20.49	-22.82	-23.43	-23.72	-24.26	-24.53	-24.17	-23.18	-23.68	-21.62	-21.95
	BST	-5.09	-4.99	-6.60	7.31	2.61	8.28	8.00	7.65	7.69	6.42	8.66	-0.31	-3.73
	TM	-6.39	-9.68	-4.29	1.64	-1.29	2.02	2.52	1.00	2.24	1.45	1.55	-0.97	-3.28
	TNT	4.08	2.88	5.17	13.16	-7.85	-3.43	-2.95	-2.77	-0.17	-0.37	-0.03	-4.32	4.06
Organic P	PXT	12.49	4.73	1.75	-4.44	-7.30	-6.95	-7.83	-9.62	-7.25	-3.65	-5.70	-6.09	0.87
	WM	-17.83	-17.04	-17.98	-18.28	-19.58	-20.13	-21.25	-21.92	-20.99	-19.29	-19.12	-17.02	-18.66
	BST	-5.66	-5.27	-4.51	6.78	3.80	6.19	6.30	5.80	5.36	4.56	6.13	-1.06	-3.32
	TM	-3.95	-6.94	-2.54	1.73	-1.80	0.57	0.86	-0.75	0.67	1.08	1.19	0.00	-1.88
	TNT	4.56	3.47	5.86	6.62	-5.59	1.70	4.18	3.87	7.25	6.16	7.16	-1.25	5.03
NO_3	PXT	-3.44	-3.23	-3.32	-3.55	-2.33	1.22	-2.28	-3.75	-4.90	-2.38	-1.68	-1.84	-2.02
	WM	2.56	2.95	2.37	3.03	7.70	9.28	0.76	2.94	-2.21	-5.95	-0.83	-0.72	4.49
	BST	4.99	12.86	8.22	10.82	12.97	13.65	14.07	4.36	2.54	1.45	1.68	1.75	9.72
	TM	-7.36	-8.54	-8.92	-12.50	-10.15	-8.51	-9.53	-1.39	-5.45	-3.20	-3.94	-4.73	-8.84
	TNT	0.34	0.85	3.46	4.22	5.59	6.58	7.38	6.39	7.98	2.51	0.90	0.29	5.02

With regard to water quality, the Woonasquatucket-Moshassuck acted as the outlier result. While the other four rivers exhibited increased average annual sediment loading under 2080 land use, the Woonasquatucket-Moshassuck displayed a 32.02% decrease, evenly spread across all months. As demonstrated in Ch. 2.3.3, sediment loading was correlated with surface runoff, so decreased loading is expected as surface runoff decreases as urban areas contract. The other four rivers all displayed winter increases in sediment loading, with the Pawtuxet showcasing a dramatic 112.36% increase in average January sediment loading (Figure 4.3a). Yet,

the Pawtuxet, Ten Mile and Taunton Rivers also displayed consistent summer reductions, with the greatest from the Pawtuxet. Winter increases were expected, as increased urban areas increase surface runoff, and dormant vegetation cannot mitigate through evapotranspiration. However, I was surprised by summer reductions in sediment loading from the Pawtuxet. Of the four water quality parameters, sediment was most sensitive to changes in land use.

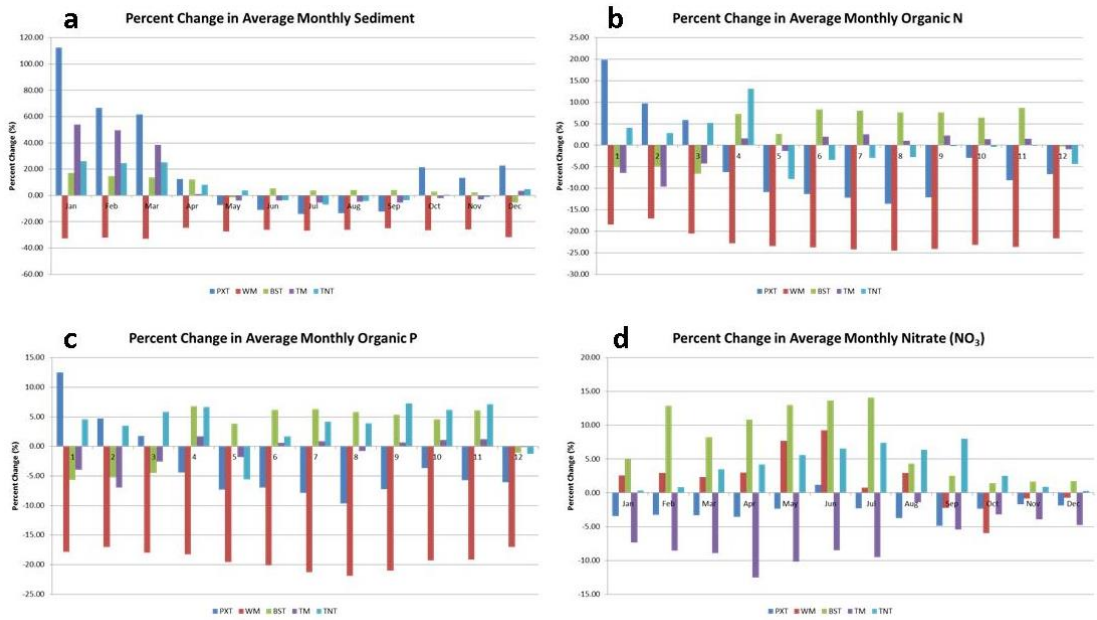

Figure 4.3 Percent change in monthly averages for total sediment loading (a), organic N loading (b), organic P loading (c) and nitrate loading (d). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

Changes in organic N and P were generally mild, with the exception of the Woonasquatucket-Moshassuck, which experienced average annual reductions of 21.95% and 18.66% for organic N and P, respectively. For both parameters, the Pawtuxet and Taunton Rivers showed average annual increases, while the Blackstone and Ten Mile Rivers showed average annual decreases. For nitrogen, the two pairs of rivers also displayed opposite monthly trends, with the former exhibiting winter increases and summer and fall reductions, and the latter two exhibiting the inverse (Figure 4.3b). However, the Taunton broke this trend for

phosphorus, experiencing average monthly increases in every month except for May and December. Nitrogen was slightly more sensitive to land use changes, which is important to the nitrogen-limited Bay.

Nitrate loading behaved very differently from the other water quality parameters. Whereas the Woonasquatucket-Moshassuck was most sensitive to land use changes with regard to sediment, organic N and organic P, it showed a milder trend for nitrate, with an annual increase of 4.49%, winter, spring and summer increases, and fall reductions (Figure 4.3d). Yet, the Blackstone and Ten Mile Rivers experienced substantial changes in nitrate loading in opposite directions in response to altered land use. Average annual loading increased 9.72% in the Blackstone, with the greatest increases occurring in late spring and early summer. Nitrate loading decreased 8.84% in the Ten Mile River, with the greatest reductions in late spring an early summer. The Pawtuxet River had the weakest response with a loading decrease of 2.02%.

4.2.4 Comparison with Climate Impacts

For nearly all measured parameters, climate change had a substantially greater impact than land use change. In many cases (such as average annual streamflow), the impacts of climate change were ten times greater than the impact of land use change. This reinforces the results of numerous studies that found climate to be a greater driver of watershed processes than land use change, which has important management implications (see Ch. 1.2.4). The two exceptions were sediment loading in the Ten Mile and Taunton Rivers, where loading increases under land use change were greater than under climate change. While climate change nearly always had a greater impact on water quantity and quality, the impacts of land use change were more diverse and localized. Climate change operates on a large-scale, with little variation in the relative changes in temperature and precipitation inputs between watersheds. Thus, the rivers generally respond similarly to climate change. In contrast, land use change happens on a more

regional level and varies between watersheds, producing starkly different results in different parts of the basin.

CHAPTER 5

CUMULATIVE IMPACTS OF LAND USE CHANGE AND CLIMATE CHANGE TO THE NARRAGANSETT BAY BASIN

5.1 Methods

To simulate the cumulative impacts of projected climate change and land use change, I ran the calibrated model using both the altered climate data to reflect increasing temperatures and precipitation and the raster data with projected 2080 land use. This simulation, thus, was a combination of the two individual climate change and land use change scenarios.

5.2 Results

5.2.1 Basin Impacts

Under projected climate and land use change, surface runoff increased 9.88%, groundwater increased 22.95% and total water yield increased 18.79% (Table 5.1). The addition of land use change increased surface runoff 0.37% over the climate change scenario, but reduced groundwater flow and total water yield by 0.89% and 0.77%, respectively. Conversely, basin-wide increases in ET and PET of 14.61% and 4.38% were both greater than the climate change scenario by 1.40% and 0.03%, respectively. Water quality parameters increased substantially over the baseline, but slightly less than under the climate scenario. The exception was nitrate, which increased 11.25% over the baseline and 0.65% over the climate change scenario. Thus, in most cases projected land use change served to buffer some of the impacts of climate change, but with small increases in nitrate and surface runoff. Monthly values closely mirrored those of the climate change scenario (Figure 5.1, and Figure 3.2 for comparison). Similar to average annual values, monthly ET and PET were higher with the addition of land use, while total water yield was consistently slightly lower. Surface runoff showed a more variable

response, increasing over the climate-only scenario in most months, but slightly lower in April and November.

Table 5.1 Percent change in average annual basin values for hydrology and water quality parameters relative to the baseline scenario and the climate change scenario.

Percent Change (%)		
	Baseline	Climate Change
Surface Runoff	9.88	0.37
Lateral Soil Flow	13.64	-7.92
Groundwater Flow	22.95	-0.89
Total Water Yield	18.79	-0.77
ET	14.61	1.40
PET	4.39	0.03
Total Sediment Loading	42.73	-0.64
Organic N	26.28	-0.50
Organic P	25.20	-0.32
Nitrate (NO_3)	11.25	0.65

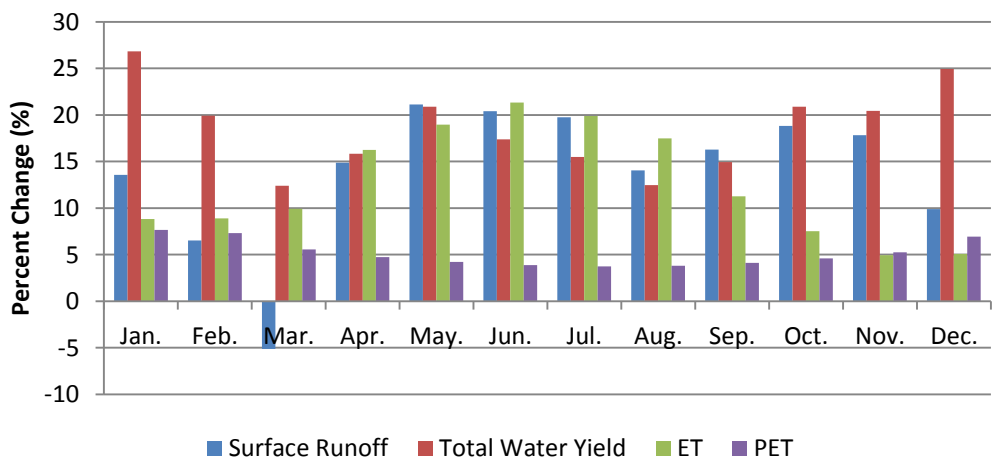

Figure 5.1 Percent change in average monthly values relative to baseline simulation.

5.2.2 River Impacts

Land use change and climate change produced substantial changes in the measured parameters of the five major river systems (Table 5.2). Streamflow changes largely reflected

those observed under the climate scenario, with average annual increases in all watersheds over the baseline period.

The Blackstone River experienced the greatest streamflow increase, 22.39%, while the Taunton River experienced the smallest increase, 17.68%. In all river systems, the addition of land use negated some of the increase associate with climate change. This was true across all months. However, the degree to which the addition of land use change reduced the impact of climate change varied substantially by basin and month. Average annual reductions in the Blackstone, Ten Mile and Taunton ranged from 2.20-2.88%. But the Pawtuxet and Woonasquatucket-Moshassuck Rivers displayed reductions from the climate scenario of 7.86% and 12.18%, respectively. At the monthly scale, summer months exhibited the greatest reductions from the climate scenario. In August, average monthly streamflow in the Pawtuxet and Woonasquatucket were 11.96% and 8.55% greater than the baseline, but 27.57% and 43.14% lower than the climate scenario. This reflects the large relative impact of land use change on streamflow in the Pawtuxet and Woonasquatucket-Moshassuck Rivers described in Ch. 4.2.3.

Table 5.2 Percent change in average monthly and annual streamflow and water quality parameters relativeto baseline. PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton

		Percent Change Relative to Baseline												
		Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Annual
Streamflow	PXT	27.31	20.35	17.08	16.25	19.00	20.27	17.36	11.96	14.20	20.68	22.41	24.62	18.43
	WM	27.29	18.72	17.89	16.42	20.36	16.98	12.87	8.55	10.18	18.88	18.98	23.60	18.35
	BST	34.04	28.75	17.59	16.31	23.14	22.06	21.94	17.12	16.08	21.24	22.92	25.57	22.38
	TM	24.97	20.32	13.03	14.66	19.79	18.54	17.98	13.87	15.47	21.17	20.61	22.45	18.55
	TNT	23.69	21.45	14.33	13.74	14.56	17.28	15.80	13.46	14.22	18.29	21.07	21.12	17.68
Sediment	PXT	250.26	102.49	71.99	108.34	36.97	21.82	-0.04	-2.13	11.60	62.02	46.89	156.07	127.92
	WM	148.31	31.36	4.17	39.83	13.19	-1.73	-17.49	-16.84	-14.05	-8.76	-10.76	14.98	36.96
	BST	83.84	67.48	20.73	69.90	26.30	28.97	14.23	17.44	16.06	33.49	41.63	113.43	60.19
	TM	93.65	66.95	35.91	28.35	7.26	-2.65	0.08	3.78	7.36	16.88	18.85	76.02	46.45
	TNT	40.03	27.74	-7.06	46.65	-6.33	-2.93	-5.61	-1.49	43.36	205.87	107.24	333.52	30.71
Organic N	PXT	137.30	47.89	47.47	63.56	47.46	27.74	16.72	10.19	8.46	40.20	28.45	100.12	64.33
	WM	16.38	4.84	1.91	22.70	25.51	6.13	-5.64	-5.96	-8.13	-2.83	-2.24	2.61	5.47
	BST	62.13	79.36	18.12	66.36	36.24	42.48	32.64	37.30	33.74	35.28	42.57	109.10	57.07
	TM	40.75	17.09	13.09	35.40	24.14	17.08	27.50	21.79	25.04	30.48	28.45	50.84	26.70
	TNT	30.41	24.38	-8.32	30.95	-4.61	3.15	5.60	6.23	19.31	73.77	47.89	135.45	22.295
Organic P	PXT	125.84	41.37	35.96	47.37	40.86	26.49	17.17	8.55	9.80	27.06	21.25	71.62	91.02
	WM	54.30	12.94	1.27	21.57	20.82	5.84	-4.86	-8.47	-7.51	-1.53	-0.68	11.95	22.48
	BST	58.71	64.90	15.28	59.97	37.57	36.23	26.95	29.79	25.51	26.87	31.41	87.74	107.86
	TM	37.08	16.47	14.53	32.05	27.80	21.06	26.07	18.34	20.51	25.83	24.16	41.96	64.33
	TNT	28.48	23.81	-9.44	33.08	-1.92	8.56	14.90	15.32	25.68	66.35	47.88	115.76	88.61
NO_3	PXT	37.42	33.06	12.69	7.90	86.16	15.37	5.59	2.82	15.29	43.52	32.40	29.35	27.95
	WM	60.82	54.05	22.46	15.34	129.52	24.64	-1.11	4.85	5.74	37.88	36.00	33.92	40.92
	BST	73.34	85.31	42.65	31.37	58.60	38.12	4.37	18.71	16.48	36.97	39.96	38.13	41.58
	TM	45.91	45.88	15.37	1.47	57.37	-7.57	-20.70	-8.15	13.80	36.23	35.38	29.62	17.06
	TNT	39.06	38.92	18.93	14.51	70.37	9.69	-8.35	14.43	24.94	35.59	28.10	25.96	26.93

Organic N and P loading increased significantly in all major rivers, with the greatest increase in the Pawtuxet for organic N and the Blackstone for organic P. The addition of land use increased organic N loading over the climate scenario in the Pawtuxet and Taunton Rivers, while reducing it in the others. With regard to organic P, land use changes increased phosphorus loading in all rivers except the Woonasquatucket-Moshassuck. Nitrate loading increased in all watersheds, with the greatest increases in the Blackstone and Woonasquatucket-Moshassuck Rivers (41.59% and 40.92%, respectively) and the smallest in the Ten Mile (17.06%).

In general, the average annual percent increase over baseline for the combined land use change and climate change scenario was similar to the sum of the changes from the two individual scenarios. However, this rule did not apply to all rivers and parameters. For example, in the Woonasquatucket-Moshassuck River sediment loading increased under the climate scenario by 94.36% and decreased under the land use scenario by 32.02%. Yet, in the combined scenario the increase over the baseline was only 36.96%, perhaps related to the decrease in streamflow from the climate scenario that occurs when land use is added. This is likely a result of increased forest cover combining with higher temperatures for greater ET. In the opposite direction, organic P loading from the Blackstone River increased 55.26% under the climate scenario and decreased 3.32% under the land use scenario. Yet the combined scenario produced a substantial 107.86% increase. The Taunton saw an 88.61% increase in the combined scenario, despite individual increases of 17.87% and 5.03% in the separate climate and land use scenarios. This suggests that on a local level, interactions between the impacts of climate change and land use change can have cumulative impacts that diverge from a simple additive effect. As with the climate change scenario (Ch. 3), the magnitude of the impact of climate change and land use change increases over time as temperature and precipitation changes magnify (Figure 5.2). The results are fairly minimal initial impacts, especially with regard to sediment, nitrogen and phosphorus, but large cumulative impacts over time.

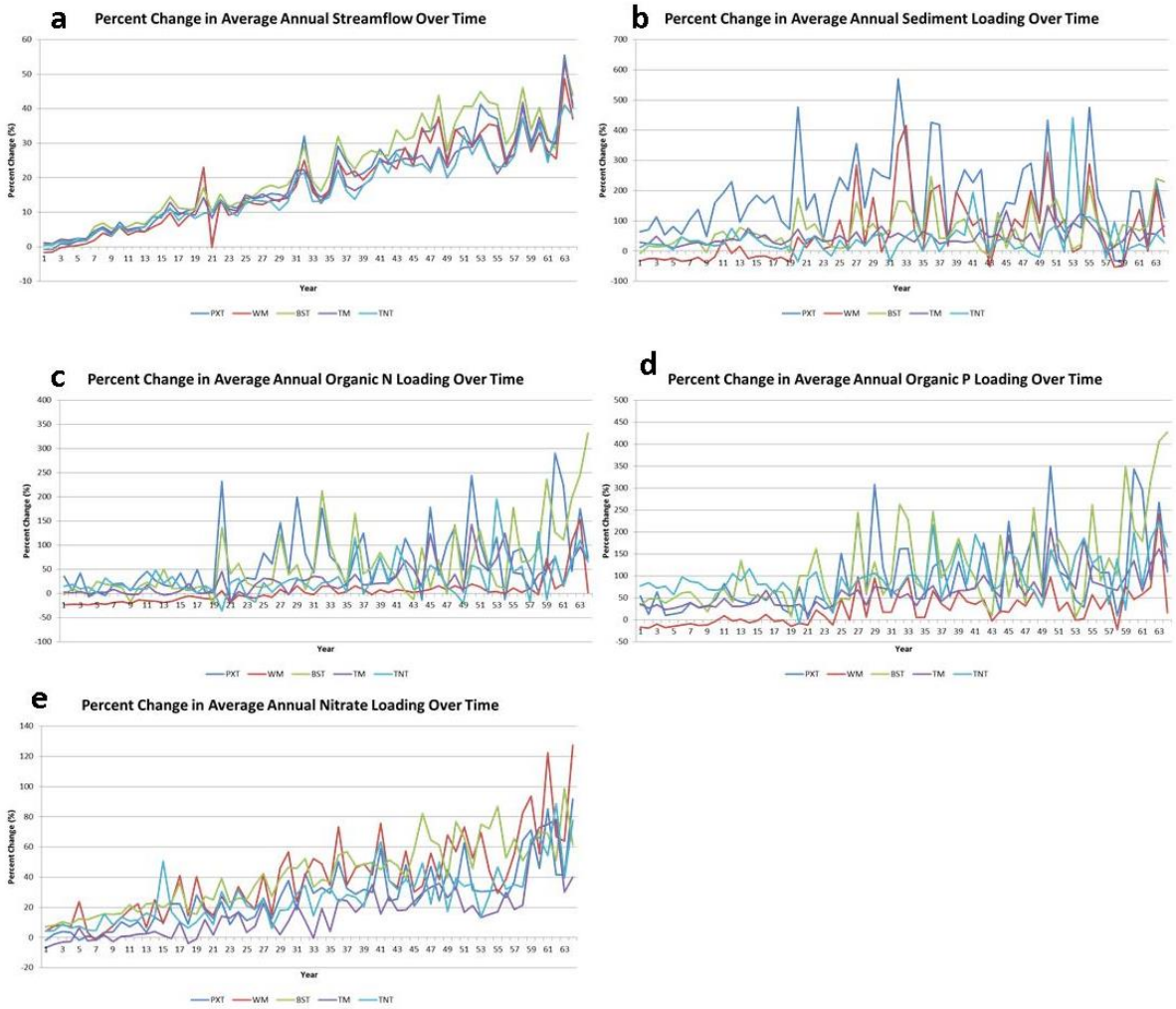

Figure 5.2 Percent change relative to baseline over time in annual averages for streamflow (a), total sediment loading (b), organic N loading (c), organic P loading (d) and nitrate loading (e). PXT=Pawtuxet, WM= Woonasquatucket-Moshassuck, BST= Blackstone, TM=Ten Mile, TNT=Taunton.

CHAPTER 6

CONCLUSIONS AND MANAGEMENT IMPLICATIONS

6.1 Impacts on Nutrient Management Goals

After more than a century of eutrophication that threatened the ecology of Narragansett Bay, watershed managers are working towards ambitious goals to reduce nitrogen pollution from the wastewater treatment facilities that discharge into the Bay and its tributaries. While the magnitude of the expected reductions is subject to debate (Nixon et al., 2008; Vadeboncoueur et al., 2010; Deacutis, 2008), these efforts, in tandem with current efforts to manage stormwater and close combined-sewer overflows, will make important headway towards restoring ecologically-sensitive and important eelgrass beds and avoiding summer hypoxia and consequent fish kills. Yet, the water quality of the Bay is driven, in part, by the quantity and quality of the water delivered by its tributaries, which is driven by climate and land use. Land use is an innately dynamic variable, changing in response to socioeconomic, demographic and regulatory changes. Although climate historically operated under the assumption of stationarity, varying within an established probability distribution (Milly et al., 2008), anthropogenic climate change is driving temperature and precipitation outside historical envelopes. Both climate and land use change maintain important implications for the Bay's restoration goals, as any efforts planned under the assumption of stationary climate and land use are likely to be limited in their efficacy

The purpose of this study was to investigate the potential impacts coincident climate change and land use change could have on the Bay's watershed to inform future management decisions aiming to restore the ecology of the Bay. The results indicate that from present to 2080, surface runoff, streamflow and nutrient loading will increase dramatically across the basin. Climate change, especially the projected 10% annual increase in precipitation, will be the

dominant driver of these increases. The impacts of climate change are applied similarly across the basin, with some regional variation but similar broader trends. Changes in land use behave very differently. The impacts of land use are considerably smaller than of climate change, especially on annual streamflow (Table 4.3), but more regionally diverse. This is especially true for sediment loading, with a range in percent change relative to baseline from -32.02% in the Woonasquatucket-Moshassuck watershed to +63.21% in the Pawtuxet watershed. Thus, while climate change will have a significantly greater impact on the watershed, localized land use change will produce substantial variation across the basin.

These increases will likely have significant impacts on the ecology of the Bay and on the restoration goals. Codiga et al. (2009) found that hypoxia was more severe after higher river flows, and that the magnitude of June river flows could be used as an index for the severity and extent of July and August hypoxia. Under the combined scenario, average annual June streamflow from the five major tributaries increased 16.98-22.06%, indicating the potential for more hypoxic events in the future. Streamflow increases also deliver greater amount of nutrients to the Bay (Brooks et al., 2003; Smith et al., 2010). Predicted increases of organic N, organic P and nitrate loading highlight this problem.

Nitrate is the more problematic nutrient, as it is readily absorbed by plants and can stimulate the growth of algae in aquatic systems. The combined model predicts increased nitrate loading from all tributaries. For example, under the baseline simulation, the Blackstone discharged and average 163,724.5kg of nitrate annually into the Bay, and the model projects an average annual increase of 41.59%. This is likely to increase primary productivity in the Bay and, consequently, the likelihood of fish kills. Organic N is important in that it will break down into ammonia, and then nitrate (Brooks et al., 2003). Average annual loading increases of organic N would increase available nitrogen and fuel algae growth. While average annual increases are

large, the greatest increases occur in winter, when algae growth is minimal and hypoxia is unproblematic. Summer increases are smaller, albeit likely to exacerbate nutrient problems. Nitrogen is typically the limiting nutrient in marine and estuarine systems, while phosphorus is limiting in freshwater. Because the focus of this study was how changes in the watershed may impact the Bay, phosphorus was of little concern. Phosphorus becomes important in estuarine systems, like the Bay, when nitrogen is in such abundant supply that phosphorus becomes limiting. The N:P ratio can provide guidance as to the likelihood of phosphorus becoming limiting, with an N:P ratio above 7.2:1 indicating that phosphorus is limiting (Chapra, cited in Marshall & Randhir, 2008). The average annual N:P ratios for the water discharging at the outlet of the five major tributaries ranged from 2:1 to 4.7:1, suggesting that the climate and land use change impacts on the watershed will not significantly alter the N:P ratio of the Bay and it will remain nitrogen limited. Monthly values vary, with the ratio in the Taunton River during winter months about 7.2:1, but that is when algae growth is dormant and will have minimal impact.

6.2 Adaptation Potential

6.2.1 Adaptation Through Land Use

As stated in Ch. 1.2.4, the dominance of climate change over land use in driving hydrologic changes in the basin has a significant, and unfortunate, management implication. Land use change is a regional process that can be addressed at a local level, but climate change is a global process that must be addressed on an international level. This disparity in institutional scale means that watershed managers in the Narragansett Bay region do not have the capacity to prevent the most dramatic changes projected for the basin. However, the results of the combined model suggest that land use can be used to mitigate the severity of the impacts.

Although the combined model produced increased values for streamflow and pollutant loading, the addition of the altered land use scenario to the climate change scenario had

variable impacts that may highlight potential management options. The addition of the land use scenario increased sediment loading in all rivers except the Woonasquatucket-Moshassuck, where adding the land use scenario reduced sediment loading under climate change by 60.83%. This impact was even greater on monthly values, to the extent that the combined model produced percent decreases in sediment loading from June through November, despite increases greater than 20% during those months under the climate change scenario. A potential factor behind this is that the Woonasquatucket-Moshassuck watershed saw a reduction in developed land of 2.50% and an expansion of forest of 1.90% under the 2080 scenario. Further, the loss of low- medium- and high-density urban development drove this reduction. Since forests yield lower amounts of sediment than urban and mixed use basins (Brooks et al., 2003), the greater forest cover at the expense of urban areas likely helped mitigate the increased sediment loading associated with streamflow.

A similar pattern occurred for organic phosphorus. For organic nitrogen, reductions from the climate change scenario also occurred in the Blackstone and Ten Mile River watersheds. These are the three watersheds that experienced an increase in forest area. Nitrate behaved differently, with reductions relative to the climate change scenario exhibited in the Pawtuxet and Ten Mile River watersheds, where the greatest increases in high-density urban development occurred. Nitrate loading from forested watersheds is usually from nitrification in soils, while nitrate in urban and suburban watersheds is usually a result of suburban lawn fertilizer, leaking septic systems or pet waste (Burns et al., 2009). However, there is no observable pattern between the land use changes in the five watersheds and the resultant changes in nitrate loading, though it may be a product of interactions between forest, agriculture and streamflow. While the relationship with nitrate is unclear, the other water quality results suggest that projected increases in forested area will blunt some of the impact of

climate change. This could present an opportunity for watershed managers. Policies that restore or expand forest cover, such as buying and restoring properties in floodplains, have the potential to reduce sediment and nutrient loading to the Bay. In urban areas, where expanding forest land is more difficult, best management practices that mimic the forest processes that help reduce surface runoff and nutrient loading (soil storage, surface roughness, plant uptake and transpiration) could achieve similar goals.

6.2.2 The Pawtuxet Watershed: Case Study in Application

The Pawtuxet River stands out as a watershed where the lessons gleaned from the land use scenario can be applied to adapt to the impacts of projected land use and climate change. Of the five major watersheds that feed into the upper Narragansett Bay, the Pawtuxet demonstrated the greatest increase in developed area over the simulation period, driven by developed open space, high-density and low-density urban development. The watershed also experienced the greatest reduction in forest land and the second-greatest reduction in wetland area. The result was increased sediment and organic nutrient loading with the addition of land use to the climate model, with the watershed exhibiting the greatest percent increase over the baseline for sediment and organic N loading, and the second greatest increase for organic P.

While this study did not focus on flooding, the Pawtuxet's history of flooding and the potential for future flooding merit mention here. Flood stage at the Cranston, RI, USGS gage is 9ft (48.14 m^3/s), and the Action stage is 8ft (39.64 m^3/s). Under the combined model, the number of months where the average monthly mean streamflow was above flood stage increased from 2 under baseline conditions to 5 under projected land use and climate change, while the number of months above the Action stage increased from 8 to 11. These projections are, further, likely underestimates for two reasons. First, validation in the Pawtuxet showed that the model consistently under-predicted peak flows. Second, monthly averages may not

accurately reflect extreme events, and this study did not simulate a daily time scale. The infamous 2010 flood displays the potential for monthly averages to obscure the scale of individual events. The monthly mean for March 2010 was 61.31 m^3/s, yet on March 30th average discharge at the USGS gage was 369.46 m^3/s, breaking the previous record set 1.5 weeks prior (NOAA, n.d.). Thus, under altered climate and land use the Pawtuxet River watershed is likely to experience substantially more flooding than at present.

Increased flooding has significant implications. The lower Pawtuxet River watershed encompasses the densely populated cities of Cranston, Warwick and West Warwick, and there is a great deal of floodplain development. The impact of flooding in the watershed on residential and commercial properties was evinced during the 2010 flood, when many businesses and houses went under water, most iconically the Warwick Mall. Soucy et al. (2011) identified those populations most impacted by the flood to highlight the differential impact of flooding. They note that many of the residential properties inundated were rental properties. Flood-prone properties are cheaper, and buyers are likely to capitalize on this low price and then rent out the unit. This means that renters were disproportionately impacted by the flooding. Populations most likely to be renters are low-income and elderly. These are also the populations less likely to have the disposable income to react to a flood event, less able to navigate the insurance claims process, and more likely to be subject to insurance fraud. This highlights the inequities in who will be impacted by changes in water quantity as a result of future land use and climate change.

The scale of the projected water quantity and quality changes, the transition from forest land to urban development, the history of flooding and increased occurrence of future flooding, and the inequitable distribution of impacts all position the Pawtuxet River watershed as a target for BMPs. Looking at the watershed at the finer sub-basin scale, one sub-basin stands out as both exhibiting the greatest increase in developed area at the expense of forest between

present and 2080, and the greatest increase in surface runoff over the baseline under the land use change scenario (ranging from 42% to 155%). Increased urbanization in this watershed is likely driven by its location at the intersection of two highways (I-95 and RI-37), and its proximity to the high-traffic US Route 1. Current development is predominantly suburban east of Jefferson Blvd and commercial/industrial to the west. The commercial/industrial space includes larger areas of pavement for parking lots, and a large amount of roof space for warehouses and commercial buildings.

BMPs that reduce this impervious cover, or intercept runoff from it, and mimic the aforementioned forest processes can reduce the amount of surface runoff reaching the river and help limit the impact of future land use and climate changes. These practices include, but are not limited to, rain gardens, green roofs and vegetated swales. Because this sub-basin contains a large portion of Pawtuxet River floodplain, efforts by the state to purchase and restore un-used or under-used commercial/industrial properties to natural floodplain forests could help reduce surface runoff to the river while absorbing and detaining flood waters to alleviate downstream flooding. There may be some political support for this effort, as evidenced by language in the recently-passed Resilient Rhode Island Act.

6.3 Limitations and Future Research

While the results of this study can be used to drive future planning of climate adaptation, the structure of the study placed several limitations on the data that could limit that usability. SWAT runs at a daily time-step, but I analyzed only average annual and average monthly values. This choice focused on highlighting cumulative changes and evincing seasonal trends, but prevents a more thorough analysis of extreme events. As mentioned previously, the baseline monthly mean streamflow for March 2010 was 61.31 m^3/s, yet on March 30th average discharge at the USGS gage was 369.46 m^3/s (NOAA, n.d.). Because the monthly mean averages

this extreme value with lower flows during the month, examining only the monthly mean does not reveal the magnitude of the March 30th flood event. Because extreme events are projected to increase (NCADAC, 2013), and because extreme events can have significant impacts on the watershed system through greater flashiness, higher transport of sediment and nutrients through surface runoff, and greater instream erosional force leading to scouring and consequent increased suspended sediment and bed load, these extreme events pose an important challenge for watershed managers. Thus, future research should examine daily means to provide greater resolution of watershed response.

The results are also limited by the use of a single climate and land use change scenario. As shown in the literature review (Ch. 1.2), many studies chose to capture the range of uncertainties inherent in climate models and land use projections by including a number of scenarios. Running the model under different RCP scenarios would reveal the range of possible future climate impacts and allow watershed managers the discretion to prepare for worst-case scenarios. Using the land use projection based on the LTM gives land use planners a specific prediction for which to plan. However, using a range of land use scenarios could showcase the magnitude of the differences in impact from different development trajectories that could better inform future land use planning. It could also help test thresholds to determine at what levels of land use change significant impacts emerge. Further, the LTM used operates through a series of spatial rules that are built on assumptions predicated on historical trends. Yet, these rules may not apply going forward, as the dynamic nature of land use may upend these trends. For example, the dominant trajectory of the recent past has been the conversion of secondary forest into low-density urban areas to meet growing demand from residents leaving cities and settling in the suburbs and exurbs. This trend may not hold going forward, and land use

scenarios that test alternative assumptions (such as the revitalization of urban centers and marginalization of suburbs) would help display a fuller array of possible impacts.

Thus, future research should attempt to examine daily impacts, and include multiple climate and land use scenarios. Future research could also work to use the model to test specific BMPs. Multiple BMP simulations could be run by targeting specific sub-basins and altering hru representation to reduce effective imperviousness and expand forest to test the efficacy of expanding forest (or green infrastructure that mimics forest processes) cover on impacts. This process could reveal where land use changes and BMPs would be most effective or efficient.

6.4 Conclusion

The model results do not indicate that changes to land use can completely counteract the annual impacts of climate change (although they do suggest that they could substantially alter or reverse seasonal impacts), highlighting the need for watershed managers to plan their restoration efforts and goals within the context of future climate change. But, the results do highlight the potential for managers to limit those impacts through best management practices to, at least, minimize to the greatest extent possible the impact that climate change will have on the Bay's nutrient reduction goals. Further analysis at a sub-basin level could help managers identify specific sub-basins where efforts to implement BMPs can have the greatest success and prioritize adaptation efforts.

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