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Benefit Indicators for Flood Regulation Services of Wetlands:

A Modeling Approach

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Office of Research and Development
National Health and Environmental Effects Research Laboratory, Atlantic Ecology Division

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Benefit Indicators for Flood Regulation Services of Wetlands: A Modeling Approach

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Foreword

This report builds on research reported in Bousquin et al. (2014). Whereas the previous report focused on developing an event-based flood modeling process that would explicitly account for flood water retention in freshwater wetlands, this report improves the modeling and details how that modeling process was used to develop indicators to assess increases in flood protection benefits from potential wetlands restoration. The assessment of flood protection benefits follows a non-monetary benefit indicators framework outlined in Mazzotta and Wainger (in preparation). The intensive modeling performed for the watershed case study presented in this report may be too extensive for some watersheds and flood benefits assessments. In light of this, we used the results to develop a set of indicators that could be collected without the modeling. In this report we outline a three-tiered approach, present indicators for Tiers II and III, and describe the process one would follow to apply such indicator sets. Future work will further investigate the transferability of these flood benefit indicators and the modeling process to other watersheds, as well as ways of making the assessment and tools more accessible to a wider audience of users.

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Executive Summary

This report describes a method for developing indicators of the benefits of flood regulation services of freshwater wetlands and presents a companion case study. The critical role of wetlands in flood protection, the difficulty in modeling that role, the high value that communities place on flood protection, and the fact that many ecosystem services assessment tools do not address flooding led to our focus on indicators of the benefits of flood regulation services provided by wetlands. We demonstrate our approach through an application to the Woonasquatucket River watershed in northern Rhode Island.

The benefits indicators approach developed in this report attempts to provide decision makers with a more accessible alternative to monetary valuation, with an approach that explicitly links functions to benefits. We propose criteria to facilitate comparison of freshwater wetland restoration scenarios based not only on production of ecosystem services, but also on how those services reach and benefit people. These benefit indicators are intended to augment existing assessment methods, to which they can add critical information about benefits to functional and service assessments already in use. With additional effort, our method can be extended to incorporate location-specific dollar values or can be used to improve benefit transfer of dollar values by calibrating those transfers.

The purpose of the indicators we present is to provide metrics of the factors that influence the spatial flow of services from production to benefits, by assessing how flood waters flow across the geographic area between wetlands that retain or slow storm flows and the important structures and other resources that might be protected from flood risks. We based indicators on sophisticated flood modeling results to ensure the rigor of results, but we recognize that in some contexts a rapid assessment that can be readily applied to inform a decision may be more valuable than a complex approach that is not feasible to apply. In an attempt to satisfy the requirements of different decision contexts, we created a 3-tiered approach. Based on the appropriate level of rigor for the decision context, each tier becomes more resource intensive to implement and requires more site-specific data. Certainty and robustness increase with each indicator tier.

Our modeling approach includes several linked models, applied in sequence. First, we used hydrologic models to estimate baseline hydrology in each basin and translate restoration scenarios—or changes in wetland coverage—to changes in hydrology and resulting stream hydrographs. Second, we incorporated these hydrographs into hydraulic models, which produced flood depth profiles. Third, we applied the flood depth profiles, using flood impact models, to map flood depth and extent and estimate the impacts of these floods on potential beneficiaries. We identified potential beneficiaries using counts of structures in the flood-prone area and estimated benefits as reductions in flooding depth, based on the flood maps produced from simulations.

In order to run scenarios for our case study watershed using these models, we first modeled flow in the Woonasquatucket watershed under current conditions. We then calibrated that model using observed flows at a USGS gage in the watershed from two separate storms. These two calibrated models were validated against flows observed for two additional storms, and we then chose the calibrated model that performed best based on this validation. We then conducted sensitivity analysis to test the effects of varying important model parameters.

We developed 12 scenarios to investigate the influence of different levels of wetland restoration across a range of storm events on stream flow and resulting flood extent. The range of storm events were based on three synthetic storms of known recurrence intervals (1-year, 5-year, and 25-year) that resulted in modeled flows of known recurrence intervals (10-year, 100-year, and 500-year). We chose different levels of wetland restoration based on the range of sizes of potential restoration sites previously identified in the watershed (Golet et al. 2003), and modeled the different levels of wetland restoration in each individual subbasin by converting 1%, 5%, 10%, or 25% of subbasin non-wetland area to wetlands in our calibrated and validated hydrologic model. We then simulated flow hydrographs for each subbasin, wetland restoration level and synthetic storm event using these updated models. We extracted peak flows from these hydrographs to estimate the extent of flooding under different conditions using the hydraulic model. Based on the changes in flow and flooding under each of these simulations, we developed indicators of wetland flood regulation services and benefits to people.

These ecosystem service benefit indicators are based on a framework developed by Mazzotta and Wainger (in preparation). The framework uses four questions to guide the process of indicator selection and measurement: (1) Is an ecosystem service supplied?; (2) How likely is it that the service will continue to be provided over the long run?; (3) How many people benefit?; and (4) By how much do people benefit? These questions are ordered so that answering each question in turn contributes additional information to an ecosystem service benefits assessment. Each of these questions may in turn be answered in lesser or greater detail.

Our Tier III indicators are based on direct results of the watershed-specific models. We developed Tier II indicators by generalizing the model outputs, through finding those factors that are the most critical determinants of who will benefit and where. Tier I indicators are based on results in the literature. We do not present Tier I indicators in this report, which focuses on describing the modeling to develop Tier II and III indicators and the process for applying the Tier II and III approaches. The Tier I indicators will be included in our forthcoming guide to applying this approach (Mazzotta et al. in preparation).

The Tier III indicators are based on modeled peak flows and flood maps for the Woonasquatucket Watershed. Using these model results, we present an example of how one would compare wetland restoration scenarios using our set of benefit indicators. Using our models, a user could compare scenarios for other subbasins within the Woonasquatucket Watershed. To apply the Tier III approach to another watershed, a user would need to perform extensive modeling similar to that described here. For those who want to apply this approach to a different watershed, we describe the models and process involved in our application, including caveats and factors that may differ for other locations.

We developed the Tier II indicators from trends and sensitivity observed in our modeling results, where we generalized the model results to determine the factors that relate differences in potential restoration sites to differences in flood reduction services and the benefits of subsequent flood damage reduction. The primary consideration for generalizing to Tier II indicators is the need to determine the relevant benefits area for flood regulation services. Flood regulation occurs at the site of a wetland, but benefits people and structures downstream of the wetland site. Therefore, our primary aim in generalizing from our detailed modeling was to determine the "benefits area" and the factors that may influence how this varies. Based on our analysis, a reasonable and conservative distance for delineating the area where people could benefit is 4 km (2.5 mi) downstream of the restoration, based on the average over all scenarios. A reasonable upper estimate, based on the average distance for a 25% restoration during a 1-year storm event, includes beneficiaries within 7.4 km (4.6 miles) of the restoration site.

Although Tier II benefit indicators are expected to be less rigorous than the Tier III indicators, they should be able to inform decisions without intensive modeling, based instead on existing datasets. Because the Tier II results are based on the model outputs rather than simple judgment, we expect that they will be applicable to watersheds similar in hydrological characteristics to our case study watershed. While Tier II indicators are more easily applied, in many cases they still require some analysis, often requiring knowledge of Geographic Information Systems (GIS). We demonstrate how the Tier II indicators could be applied in other locations.

What is most useful about the approach presented here is that it directly incorporates people and the benefits they receive from ecosystem restoration. Further, it provides a framework that can be used to compare potential wetland restoration scenarios based on these benefits without the need for estimating dollar values. Using an approach to assessing non-dollar benefits that is grounded in economic principles allows for more robust discussion of alternatives through a disaggregated and transparent presentation of the various factors that are likely to affect the level of benefits to people. This can inform many decision contexts where a strict benefit-cost framework is either not appropriate or not necessary. This indicators approach can also easily be extended to incorporate conceptions of value beyond the economic definition of value, and can be transferred to other decision settings and benefit types. We demonstrate how it can be used to complement an existing functional assessment approach, and propose that it can also be used to inform benefit transfers to add more insight into variations in restoration benefits across locations, and who might receive those benefits.

CHAPTER 1: INTRODUCTION

This report describes a method for developing indicators of the benefits of flood regulation services of freshwater wetlands and presents a companion case study. We demonstrate our approach through an application to the Woonasquatucket River watershed in northern Rhode Island. The critical role of wetlands in flood protection, the difficulty in modeling that role, the high value that communities place on flood protection, and the fact that many ecosystem services assessment tools do not address flooding led to our focus on evaluating the benefits of flood regulation services provided by wetlands.

We have chosen to focus on non-monetary benefit indicators rather than dollar values for several reasons. Because of the spatial variations in provision of wetland services and the people who benefit, dollar values are highly context-dependent, making benefit transfer difficult; our aim is to develop an approach that can be applied in different locations with minimal effort and resources. For local decision makers trying to choose rapidly where to invest resources, applying primary valuation studies will usually take too long, require too much expertise, and in many cases exceed their needs. This often leads to local decisions based solely on either supply side functional assessments or benefit transfer of somewhat generic wetland values. Although both functional assessments and benefit transfers can be rigorous and provide useful information, often these assessments rely on wetland area, with little consideration of location-specific aspects and spatial flows of services and their benefits.

This work is intended to address an important component of the assessment of ecosystem services: the development of metrics that clearly show how ecosystem functioning benefits people. This approach contributes in three ways to the assessment of flood regulation services provided by wetlands. First, it goes beyond standard ecological assessments of wetland functioning by linking functioning with how, where, and how many people benefit from wetlands. Second, it provides a means of estimating defensible metrics using a tiered approach ranging from metrics that are more easily estimated but with greater uncertainty, to metrics that require detailed modeling and provide reduced uncertainty. Third, it works as an add-on to existing functional assessment tools, in order to extend their applicability to assessing ecosystem services, or augments economic benefit transfer approaches by providing metrics that can be used to determine the “extent of the market” for flood reduction benefits or to adjust values to better reflect local conditions.[1]

Our approach is consistent with the recently released *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Olander et al. 2015). A primary recommendation of that guide is to “extend assessments beyond purely ecological measures that are not explicitly tied to people’s values to measures of ecosystem services that are directly relevant to people” (Olander et al. 2015, p. 2). To address that need, we adopt a general conceptual framework

[1] Benefit transfer is an economic valuation method that applies existing values or value functions, from studies conducted at particular locations, to estimate values at a different location (Richardson et al. 2015).

(Figure 1-1) based on the ecosystem service cascade (Potschin and Haines-Young 2011; also see Turner et al. 2000 for a similar model and discussion of integration across disciplines). Figure 1-1 shows how supply and demand interact to produce a valued ecosystem service and includes the types of assessments that are relevant to each part of the cascade. Our work focuses on the assessment of ecosystem services and indicators of their benefits.

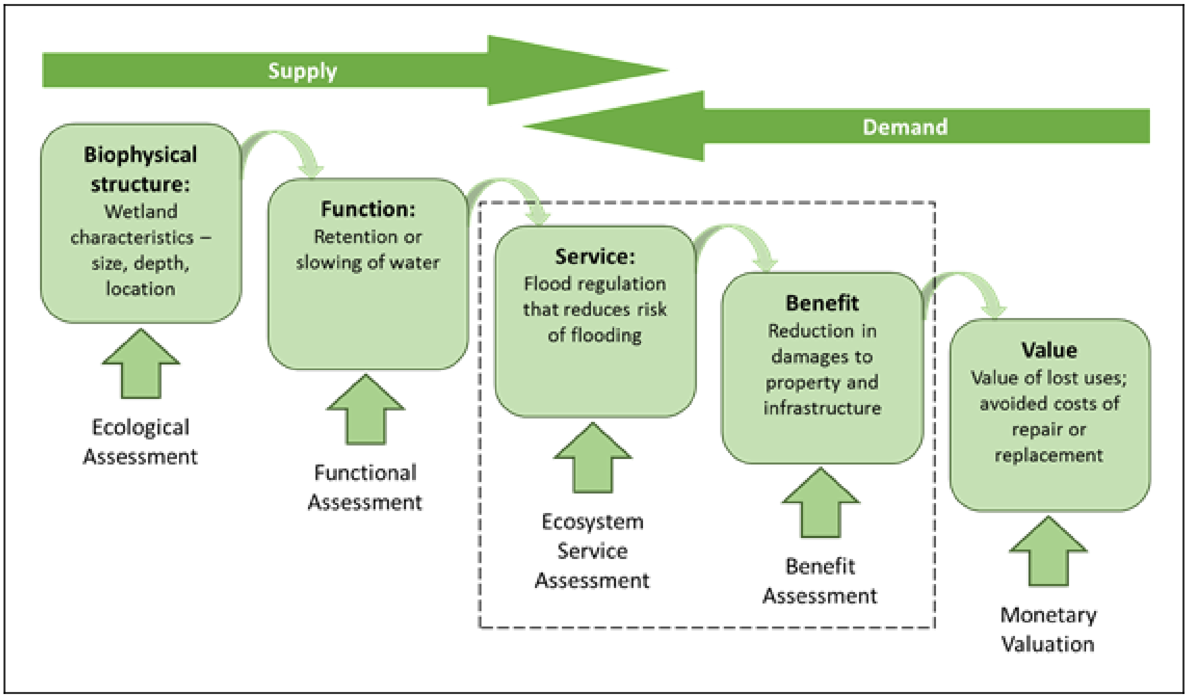

Figure 1-1: The ecosystem service cascade (adapted from Potschin and Haines-Young 2011). The cascade shows how supply of and demand for ecosystem services are related, and illustrates how the ecosystem's structure and processes affect its functioning, and lead to the provision of ecosystem services to people who benefit from and value those services. Different types of assessments focus on different parts of the cascade. Our analysis focuses on ecosystem service assessment and benefit assessment.

Despite the importance of the ecosystem services wetlands provide, the extent of wetlands in North America has declined substantially since colonization, particularly in urban areas (Dahl and Allord 1996). Wetland restoration is one way to try to recover some of the benefits that have been lost. However, with many potential restoration sites and limited funding for restoration, managers need to prioritize sites that have the highest chances for success and the highest potential benefits. To make use of available funds, these decisions often must be made rapidly and opportunistically, and few metrics exist to easily compare the ecosystem services and benefits to people from wetland restoration projects under such circumstances. Other decision contexts important to wetlands include choices about whether to re-construct a wetland that has been filled rather than restoring another type of ecosystem; how to prioritize conservation efforts to protect existing wetlands; and whether to invest in wetland restoration or construction as an alternative to gray infrastructure for stormwater or waste water management.

There are many existing methods for evaluating aspects of the ecosystem service cascade for fresh water wetlands, ranging from relatively easily applied functional assessments (e.g., and Golet 2001) to complex and data-intensive spatial models of the full ecosystem service cascade such as the ARIES model (Villa et al. 2014). The majority of existing wetland assessment methods focus on wetland functions, with cursory attention to benefits, typically in the form of a judgment regarding the "social significance" of each function (see King et al. 2000 and King and Price 2004 for a review and listing of many of these approaches). At the other end of the ecosystem service cascade, many economic valuation studies address dollar values of wetlands, some of which value specific services of wetlands (see Brander et al. 2006 and Ghermandi et al. 2010 for summaries). These studies are context and location-specific and therefore require additional information to be useful for benefit transfer and to be able to distinguish among sites. Our work is notable in its linking of functions to benefits and values for flood regulating services of fresh water wetlands and providing an approach that may be rapidly applied, in light of existing methods that largely address either functions or dollar values or require data-intensive modeling to apply.

In contrast to some other ecosystem services, the service of flood regulation is particularly dependent on the spatial and hydrological characteristics of the landscape because sites providing flood regulation typically benefit people at a distance downstream. Also, the value of the flood regulation service is highly dependent on the number of people who benefit and the value and level of vulnerability of structures protected from flooding. Thus, our analysis focuses strongly on estimating the spatial flow of the flood regulation service, defining the area where people are likely to benefit, and identifying the assets protected.

The Woonasquatucket River Watershed

The Woonasquatucket watershed is a 132 km^2 (82 $miles^2$) basin in northern Rhode Island (Figure 1-2). The basin contributes to the Woonasquatucket River, a river with a long history of cultural and industrial development. The Woonasquatucket joins the Moshassuck River to become the Providence River, which flows through Rhode Island's capital, Providence, and into upper Narragansett Bay. As was typical in the early industrialization of New England, the Woonasquatucket River was used to generate power and transport goods, and development occurred immediately adjacent to and even over the Woonasquatucket, often destroying wetlands and filling floodplains (Hardmeyer and Spencer 2007). Currently, urbanization in the watershed follows a gradient of increasing urbanization from north to south, where the northern portion is less urban than the city of Providence near the southern river outlet. Growth projections suggest the watershed will continue to urbanize in years to come (Rhode Island Statewide Planning Program 2006).

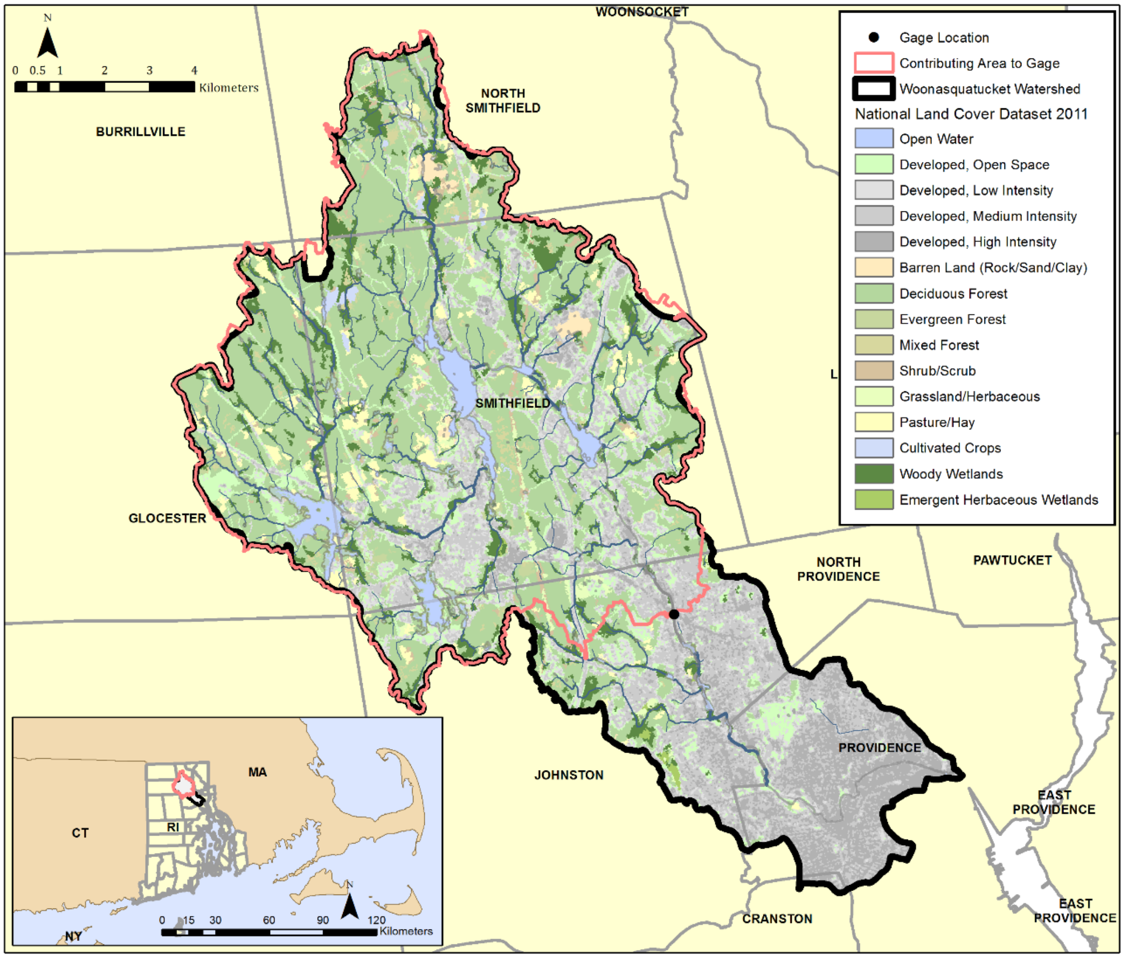

Figure 1-2. The Woonasquatucket River watershed within Rhode Island, USA showing land cover and the location of the stream gage and its contributing area.

As in many urbanizing watersheds in the northeast (Collins 2009; Hodgkins 2010; Villarini and Smith 2010; Smith et al. 2010; Hirsch and Ryberg 2012; Peterson et al. 2014), flood magnitudes are increasing in the Woonasquatucket. This is driven both by increased runoff due to loss of permeability associated with urbanization and increasing duration and intensity of precipitation events. In recent years, flood frequency curves have shifted up (Figure 1-3). Eight of the ten largest recorded flood events have occurred since 1970, and the largest recorded flood event occurred in March of 2010 and was greater than a 200-year event (Corcoran 2007).

In addition to increasing flow magnitudes due to urbanization, rainfall events causing these flows are expected to continue to increase. Long-term data suggest that annual average rainfall has been increasing by 1" each decade for the last 80 years (Figure 1-4). Depending on the conditions, increased rainfall does not necessarily correlate with increased stream flow. However, the annual peak stream flows observed have shown a trend of about 46 ft^3/sec (cfs) more each decade (Appendix C-1).

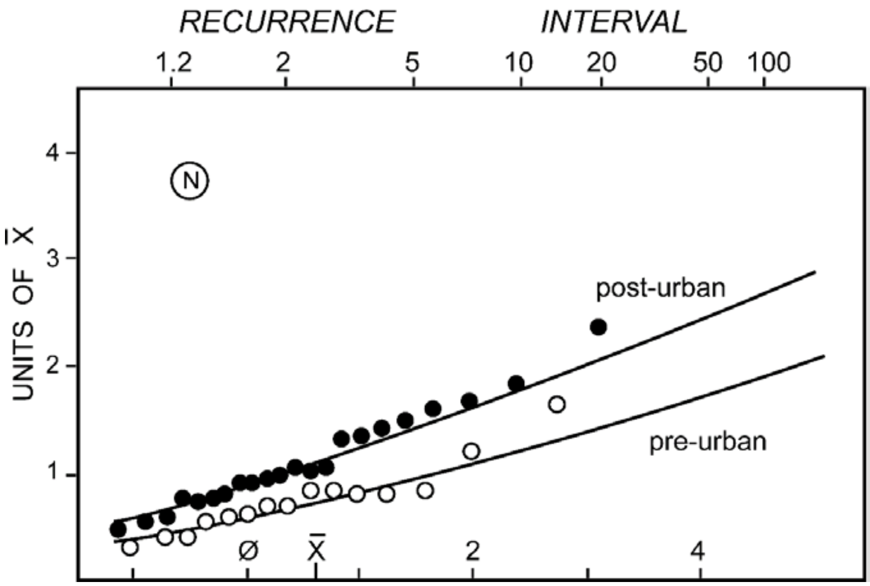

Figure 1-3. Flood frequency curves for pre- and post-urban development (circa 1952- and 1972, respectively) in the Woonasquatucket River (from Doehring and Smith 1978). The vertical scale is flow normalized to the mean annual flood (CFS) and the horizontal scale is the recurrence interval in years.

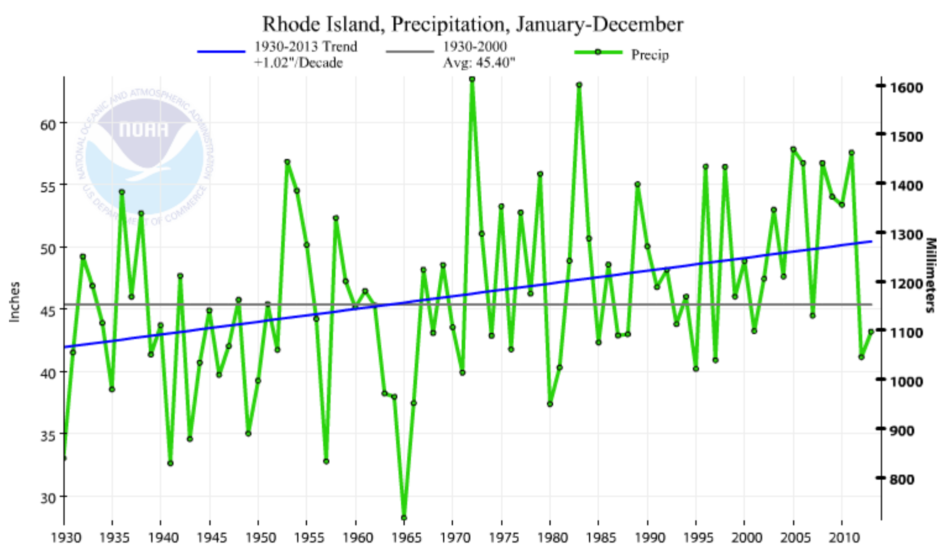

Figure 1-4. Average annual precipitation for Rhode Island for the period 1930 to 2013 (Valle and Giuliano 2014)

With more people and infrastructure located near the river, future floods will likely result in increased damages. Therefore there is a growing need to understand how wetlands and other green infrastructure may alleviate flooding, and to help facilitate land use decision making around these critical resources in order to maintain or increase flood protection to urbanizing areas.

Indicator Development

Use of indicators, measurable metrics that represent more complex phenomena, is a viable way to assess ecosystem services and their benefits while potentially decreasing the necessary time and expertise required for implementation. In developing the indicators presented here, we had two major objectives: (1) providing a way to extend existing functional assessments and valuation methods, and (2) developing a defensible approach to the use of indicators, based on sound theory and science. This requires that our indicators be sensitive to and able to connect with measurable information in other parts of the ecosystem service cascade. For flood regulation, this means that flood modeling must incorporate parameters of wetland function, and must also have a strong spatial component that accounts for important aspects of the flow of services from wetlands (production of a service) to people who benefit. And, to be compatible with economic theory and methods of valuation, the models must address factors that are important in determining differences in value across people and locations. To this end, the set of indicators chosen fit into a benefits assessment framework structured around the economic theory of value (Mazzotta and Wainger in preparation).

Using indicators in place of directly measuring the phenomena of interest may result in poor decisions if the indicators do not provide valid measures of the phenomena of interest. As a result, a major criticism of indicators approaches is that correlations between indicators and the actual supply and demand of services are often assumed rather than demonstrated quantitatively (Anderson et al. 2009; Duelli and Obrist 2003). Where correlations cannot be demonstrated it becomes hard to discern whether indicators are rigorous enough for a given decision context. Although empirical demonstration of indicator robustness exceeds the scope of this report, our aim is that our indicators be defensible metrics for the types of decision contexts often encountered when investing in wetland restoration.

The purpose of the indicators we present here is to provide metrics of the factors that influence the spatial flow of services from production to benefits, by assessing how flood waters flow across the geographic area between wetlands that retain or slow storm water flows and the important structures and other resources that might be protected from flood risks. We based indicators on sophisticated flood modeling results to ensure the rigor of results, but we recognize that in some contexts a rapid assessment that can be readily applied to inform a decision may be more valuable than a complex approach that is not feasible to apply. In an attempt to satisfy the requirements of different decision contexts, we created a tiered approach. Based on the appropriate level of rigor for the decision context, each tier becomes more resource intensive to implement and requires more specific data. Certainty and robustness increase with each indicator tier (from I to III; Figure 1-5).

To apply the Tier III approach, an end user would need to perform extensive modeling similar to that described here, which is often too resource intensive and technical for most decision makers and many decision contexts. We developed the Tier II indicators from trends and sensitivity observed in our modeling results. Because they are based on the model outputs rather than simple judgment, we expect that they will be applicable to watersheds similar in hydrological characteristics to our case study watershed. We intend to test this in future work. We expect the Tier I indicators to be the least robust but most accessible and more broadly applicable. We did not derive these indicators from model outputs; they are based on results in the literature. We do not present Tier I indicators in this report, but they are included in our guide for decision makers, which presents the overall benefit indicators approach and provides a spreadsheet tool for applying Tier I indicators (Mazzotta et al. in preparation).

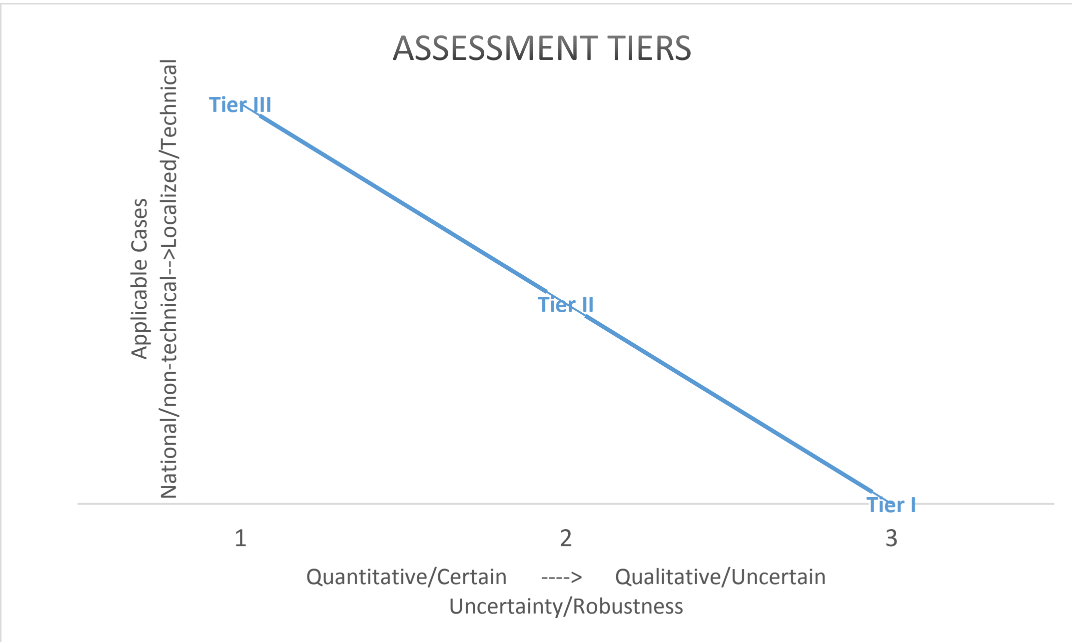

Figure 1-5. Required resources and results uncertainty across the three tiers of indicators.

Reading this Report

In Chapter 2, we describe the modeling process, assumptions, parameter values, model validation and model sensitivity. Our intention is that users will be able to replicate this modeling process to conduct a similar analysis in other watersheds. Results for both observed and synthetic storms with known recurrence intervals are given. Chapter 3 shows how to fit results from the modeling into the Tier III benefits indicators framework, and how these results, along with the model sensitivity analysis from Chapter 2, are used to develop Tier II benefits indicators. Chapter 3 also demonstrates how the indicators might be used, by presenting an example implementation using Tier III indicators developed here to make a hypothetical decision between two restoration sites. Chapter 4 provides a summary and ideas for future research. The modeling details and other specifics, including data sources, are included in appendices.

CHAPTER 2: FLOOD MODELING

Introduction

This chapter describes the detailed modeling that we conducted in order to run scenarios using simulated wetland restorations to evaluate their influence on downstream flooding and flow. Ultimately, we developed indicators for flood regulation services and benefits from wetlands based on the downstream influence of these simulated wetland restorations. Our modeling approach includes several linked models, applied in sequence (Figure 2-1). First, we used hydrologic models to estimate baseline hydrology in each basin and translate restoration scenarios—or changes in wetland coverage—to changes in hydrology and resulting stream hydrographs. Second, we incorporated these hydrographs into hydraulic models, which produced flood depth profiles. Third, we applied the flood depth profiles, using flood impact models, to map flood depth and extent and estimate the impacts of these floods on potential beneficiaries.

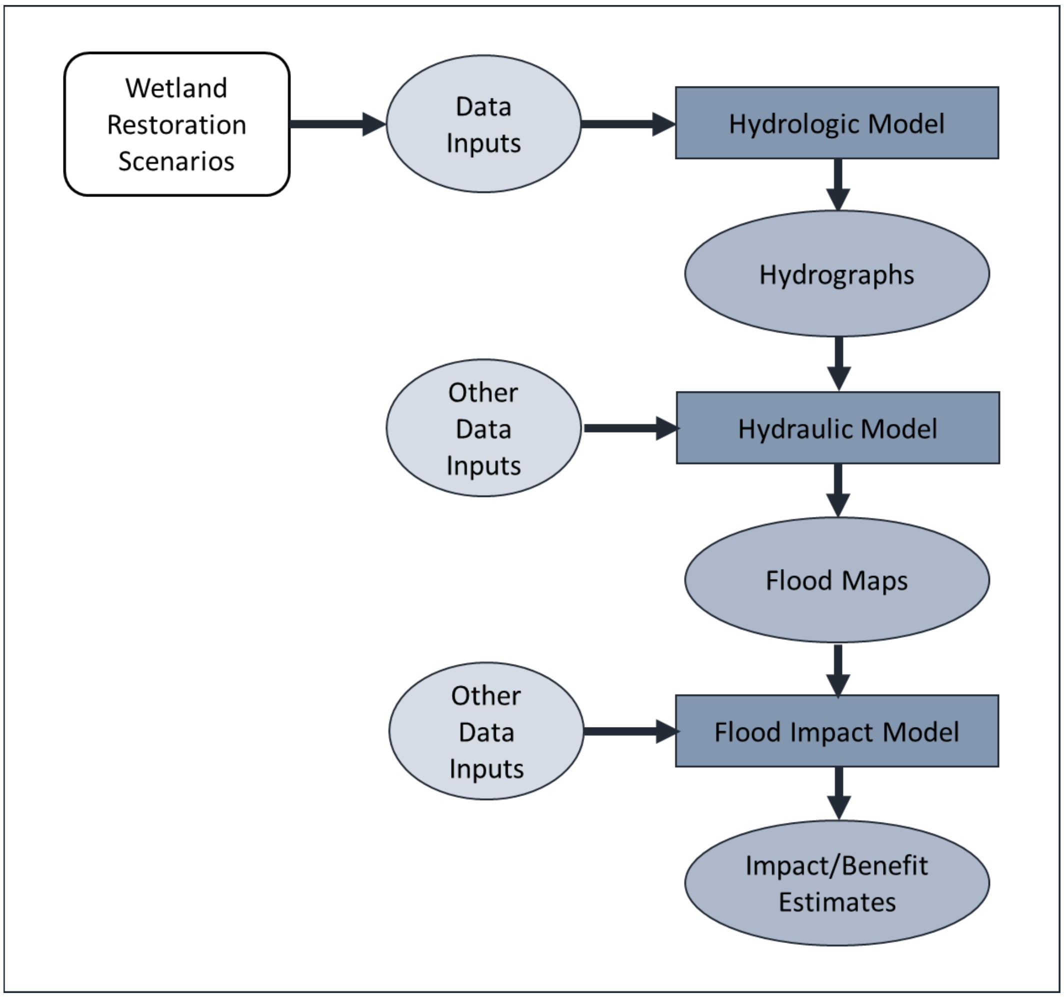

Figure 2-1. Modeling Approach

In order to explore scenarios using these models, we first modeled flow in the Woonasquatucket watershed under current conditions, and then calibrated and validated the models using observed flows at a USGS gage in the watershed. We applied these models to a 99.2 km^2 (61.6 mi^2) basin area of the watershed (Figure 1-2 area outlined in red). We did not model the entire watershed because the area below the USGS gage used for model calibration is highly urbanized and lacks information on stormwater infrastructure, which is likely to lead to errors in predicting flood extent. The methods and assumptions for incorporating wetlands into the models are critical to predicting how wetland restorations can potentially provide flood regulation and the resulting potential to reduce flood risks to people and valued assets (structures and infrastructure). To provide transparency regarding these critical topics we emphasize them in Chapter 2 in our discussion of how we parameterized the hydrologic model.

To evaluate the impact of wetland restoration on flooding and to develop indicators of the benefits of flood regulation, we modeled four wetland restoration scenarios. These scenarios increased wetland surface area in each subbasin by converting 1%, 5%, 10%, or 25% of non-wetland area to wetlands. We ran these wetland scenarios with three different synthetic storms (24-hour precipitation events with recurrence intervals of 1-year, 5-year, and 25-year; Appendix G). The resulting peak flows (approximately 10-year, 100-year, and 500-year corresponding flow events) were each the result of a different synthetic storm. Based on the changes in flow and flooding under these scenarios, we developed indicators of wetland flood regulation services and benefits to people.

Our Tier III indicators are based on direct results of these watershed-specific models drawing heavily on the sensitivity analysis to understand which variables most strongly influence results, which is detailed in the Sensitivity Analysis section. We developed Tier II indicators by generalizing the model outputs, through finding those factors that are the most critical determinants of who will benefit and where. Defensible Tier II indicators depend on model selection, assumptions, parameterization and validation.

An additional objective of our modeling approach was that both Tier II and Tier III indicators be compatible with existing functional assessments. Such compatibility allows the indicators to build on those assessments, adding information regarding benefits to people and how those benefits differ across locations. To support the compatibility of our approach with a functional assessment, we selected model inputs and developed benefit indicators to be consistent with the Miller and Golet (2001) functional indicators assessment (Appendix A).

Model Selection and Inclusion of Wetlands

Model Selection

We considered a number of existing flood models before selecting the models we deemed most appropriate for our purposes. Available models differ in terms of the characteristics and processes that they parameterize and their data requirements. In choosing models, we needed to balance competing objectives: obtaining realistic results to produce the most defensible indicators and choosing an approach that other practitioners can reproduce in other watersheds. Therefore, we opted to forgo the most sophisticated models with the most specific and detailed data in favor of more flexible models.

In selecting models, we consulted previous studies where wetlands have been integrated into modeling to estimate their downstream influence, and evaluated model advantages, disadvantages, and standards for parameterization and optimization (Bengtson and Padmanabhan 1999; Yuan and Qaiser 2011; Ogawa et al. 1986). In addition, some modeling and analysis had been performed previously in the Woonasquatucket watershed, and we selected models that are compatible with this previous work (Zarriello et al. 2013; Zarriello et al. 2010; ACOE 2000).

The Hydrologic Model

The first criterion in selecting a hydrologic model was that the model reasonably account for retention of storm water in wetlands. More specifically, the model needed to account for current wetlands under baseline conditions, but also accommodate scenarios with added or removed wetlands in different parts of the watershed. This requirement is not met by some models that integrate wetlands. For example, wetlands are a land cover type used in the Curve Number (CN) method (Williams and LaSeur 1976) that is used in many hydrologic models to parameterize runoff characteristics of a surface. The CN method models wetland runoff loss as a function of permeability, not accounting for surface water detention, and thus does not meet our criterion. Additionally, we also selected a model that would be compatible with the functional indicators used for flood retention by Miller and Golet (2001) (Appendix A), which is discussed in greater detail in the Treatment of Wetlands section of this chapter.

The second criterion for the hydrologic model was the need to accommodate rainfall from storms of various sizes, while differentiating between a baseline scenario and scenarios with the same moisture and stream conditions but with a change in wetlands. The model needed to reflect how a change in wetlands under the restoration scenario would lead to changes in where runoff is stored (loss) and resulting changes in river flow. Hydrologic models with this capability vary in complexity; for example, they may be spatially distributed, semi-distributed, or lumped, so that input parameters are either characterized at a fine-scale resolution, within sub-regions, or generalized to one number for the entire study area, respectively.

Based on these criteria, we selected the U.S. Army Corps of Engineers (US ACOE) Hydrologic Engineering Center's Hydrologic Modeling System (HEC-HMS) for the hydrologic modeling phase (Flemming 2013). HEC-HMS has the ability to model event-based (single-storm) simulations based either on observed rainfall and stream flow or, once calibrated to observations, simulations based on synthetic storms with known probabilities. Both observed and synthetic rainfall data can be associated with a single gauged point or distributed across the watershed (grid format). HEC-HMS is a semi-distributed model, meaning it aggregates distributed data up to the subbasin. Although we might prefer a fully distributed model such as the Hydrological Simulation Model-Fortran (HSPF) (Bicknell et al. 1997), we were able to increase the distribution of the model by dividing the basin into many small subbasins. HEC-HMS and its earlier version, HEC-1, have been used in previous hydrologic assessments of wetlands on flood hydrographs (Bengston and Padmanabhan 1999; Qaiser et al. 2012). Although HEC-HMS is not explicitly designed to parameterize flood water retention in wetlands, these previous studies were able to account for retention in wetlands by treating them as diversions.

The compatibility of HEC-HMS with other models was another critical factor. The Geospatial Hydrologic Modeling extension (HEC-GeoHMS; Flemming and Doan 2013) ArcGIS toolkit produced by the same US ACOE center allows for streamlined generation of model basin and subbasin parameters from existing spatial data. The data management system (HEC-DSS) used to store output hydrographs from HEC-HMS is used throughout HEC software, so HEC-HMS can easily interface with HEC hydraulic software.

The Hydraulic Model

The second phase of the modeling approach uses hydraulic models for flow of water and how water acts within the stream channel and floodplain, describing either how rapidly and with how much loss water is conveyed or where it will overflow the channel and cause inundation. Hydraulic models can use simple linear interpolation, can be one or two-dimensional, and can simulate steady or unsteady flow, all with a range of detail and assumptions regarding the input parameters. The main criteria for selection of a hydraulic model were compatibility with the hydrologic model, Geographic Information Systems (GIS), and previous studies. Based on these criteria, we selected the Hydrologic Engineering Center's River Analysis System (HEC-RAS; Brunner 2010).

HEC-RAS performs one-dimensional steady or unsteady flow analysis. This analysis uses geometric data that can be collected in an ArcGIS toolkit, HEC-GeoRAS (Ackerman 2009). The geometric data account for elevations and roughness along stream profiles and cross sections; any structures along the stream such as constricting bridges, culverts, levees, or dams; and storage areas for use in unsteady flow analysis. Like HEC-HMS, HEC-RAS is also widely used. Qaiser et al. (2012) used HEC-RAS effectively in their evaluation of wetlands for flood mitigation. In the Woonasquatucket watershed, the U.S. Environmental Protection Agency (EPA) used HEC-RAS for steady flow analysis to evaluate downstream flood impacts of removing the dam at Centredale (US ACOE 2000). As part of FEMA's process of updating Flood Insurance Rate Maps (FIRMs), U.S. Geological Survey (USGS) was contracted to create and validate a HEC-RAS model for the entire Woonasquatucket River (Zarriello et al. 2013). We integrated the geometric data used for this USGS study into our HEC-RAS model.

Flood Impact Modeling

Flood risk has two components: hazard and vulnerability. Flood hazard measures the exposure or severity of the storm independent of its impact on people. To model flood hazard, we overlaid flood depth profiles from the HEC-RAS model onto elevations in the watershed to determine where flooding occurred for each scenario, and how deep that flooding was.

Flood vulnerability measures who is impacted and the severity of that impact. For our assessment we were interested in the impacts to the built environment and severity of those impacts. We investigated several types of data to account for where vulnerable structures exist in our projected floodplain, and found address information to be most useful at the resolution of our analysis (Bousquin et al. 2014). We measured the severity of impacts on these addresses in terms of areal extent and depth of flooding based on the flood maps produced from simulations.

Treatment of Wetlands in the Models

Wetlands perform several hydrologic functions that support flood regulation. They can desynchronize storm flows by providing short-term surface water storage, which can reduce downstream flood peaks (Hubbard and Linder 1986; Hey and Philippi 1995) and they can provide long-term storage and increase infiltration, which recharges groundwater and maintains river and stream base flows (Winter 1999). The amount of flood regulation provided varies with wetland type and landscape position (Brinson 1993). Therefore the provision of flood regulation by restored wetlands is a function of both the type of restoration and the location of the site such that it can receive flood water and is upstream of otherwise flood-prone areas (Miller and Golet 2001). Below we discuss how wetlands are handled in the hydrologic model and how we related this to the Miller and Golet (2001) wetland flood protection functional assessment.

Wetland short-term surface water retention is represented in HEC-HMS as diversions—or subbasin flow that is diverted from entering the main channel. The amount diverted from a given subbasin in the model is a factor of (1) the rate at which subbasin outflow is diverted, and (2) the maximum cumulative volume of water that can be diverted. Parameterizing these diversion factors based on wetlands in a given subbasin required assumptions about the function of wetlands.

Estimating Flow Diverted by Wetlands

To apply HEC-HMS, we divided the modeled basin (99.2 km^2) into subbasins (n=139, mean area=0.7 km^2). The model cannot account for the location of a wetland within a subbasin. Effectively, this means that the model calculates the water collected in a subbasin and then diverts some portion of water based on all of the wetlands inside the subbasin before passing the flow downstream (Figure 2-2A). In reality, each wetland captures some of the water from its individual catchment area within the subbasin (Figure 2-2B). Aggregating wetlands and the diversion of water to wetlands in this way could easily overestimate the percent of water available for diversion to wetlands since all water from the subbasin, not just the wetland's catchment, is subject to diversion. To alleviate this issue, we assumed that 100% of water entering the wetland would be retained but that the catchment area of each wetland equaled the wetland's areal extent, effectively assuming all wetlands were on the perimeter of their subbasin where there was no conveyance of water from surrounding areas (Figure 2-2C). The result, when wetlands' retention (Figure 2-2C) is considered on the subbasin scale (Figure 2-2A), is that a subbasin's percent diverted equals the percent of the subbasin area in wetlands.

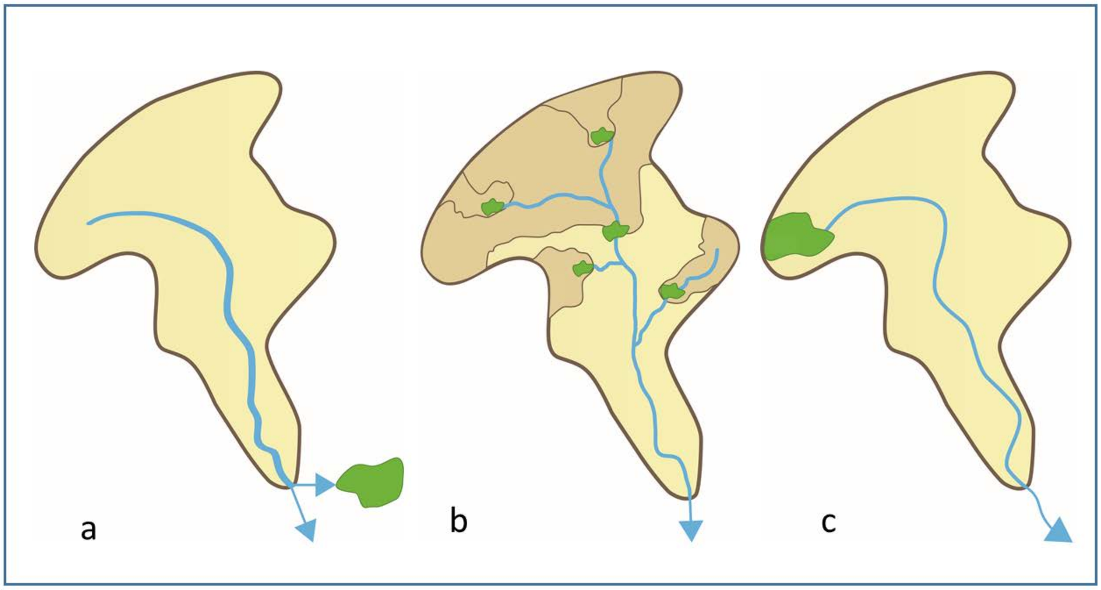

Figure 2-2- Figure showing wetlands in subbasin based on different assumptions: (a) how HMS implements diversions, (b) bio-physical reality, and (c) the wetland contributing area assumption we used.

This approach is more conservative than that of past studies. Wetlands in the watershed used in Bengston and Padmanabhan (1999) occupied a surface area ranging from 0.58 to 5.14% of the total watershed area, based on areas in the National Wetlands Inventory and their estimated drained wetlands available for restoration. Divergence rates assumed for this wetlands area were 25% to 50% (Bengston and Padmanabhan 1999). In Yuan and Qaiser (2011), wetlands covered a maximum of 10% of the watershed area, and 25% of subbasin peak runoff was set as the diversion rate.

Estimating Maximum Outflow Diverted

We used a Python toolbox in ArcMap 10.2 to estimate potential maximum volume retention, following Lane and D'Amico (2010), where we used surface elevations from the Digital Elevation Model (DEM; RIGIS 2013) as bottom contour and the minimum or average elevation from perimeter vertices as height. The DEM was developed using LiDAR, a remote sensing technology that uses pulses of light to detect variable distances to the ground to measure ground elevations during a flyover, and was provided at a 1 m resolution with a vertical precision of 0.01 m. We based the perimeter of wetlands on the same spatial dataset used to define wetlands in the Miller and Golet (2001) functional assessment (RIGIS 1993). Details of this process, including treatment of wetland types, adjacent wetlands and residual water storage in wetlands before a precipitation event, are provided in Appendix B.

Water diverted by wetlands is not released back into the modeled flow for a given storm event. This is not an issue for our analysis since we are interested in the peak flows of water, which cause the extent of flooding, rather than the total volume of water throughout a storm. We assume that water causing peak flows occurs early in the storm, before wetlands are fully saturated.

Wetland Flood Protection Functional Assessment

As mentioned above, we related wetland function in the model to the Miller and Golet (2001) approach. The Miller and Golet (2001) functional assessment indicators are categorized as opportunity, effectiveness, or social significance. In this section, we discuss how we incorporated these criteria into our modeling effort.

Opportunity

Opportunity for a wetland to perform flood regulation functions relates to how much water flows into or through the wetland. In the model, this is captured by the amount of water within the subbasin available for divergence to wetlands. Some of the functional assessment indicators for opportunity are landscape characteristics such as nearby impervious surfaces and steep slopes, which are accounted for in a lumped fashion as parameters for the subbasin in the hydrologic model. Although these characteristics were accounted for and increased water available to be diverted by wetlands, more influential nearby characteristics were aggregated with subbasin characteristics.

The model does not account for point-source inflow or flow into wetlands from streams. Most point-source inflows in the Woonasquatucket are likely stormwater infrastructure, and basin-wide spatial data for such infrastructure were not available. Excluding stormwater structures and flows will influence model results more in urbanized areas. Calibrations of the hydrologic model should correct for the role of stormwater infrastructure; however downstream of the USGS gage is heavily urban and is uncalibrated, so we did not model that portion of the basin.

Effectiveness

The effectiveness of wetlands to perform flood regulation functions is related to how much water can be retained in those wetlands. The Miller and Golet (2001) assessment tool includes three factors to measure effectiveness: whether the wetland is a basin wetland, whether the outlet is constricted, and whether the dominant vegetation is dense and persistent. Our model accounts for the wetland type (basin or not) and whether the outlet is constricted in the method used to calculate maximum volume diverted. If a wetland is not a basin, the maximum volume will be zero. Our model does not account for the type of vegetation, because all water diverted in wetlands was effectively removed from the system, making slowing by vegetation irrelevant.

Social Significance

The functional assessment includes an indicator of social significance: whether there are developed flood-prone areas within 5 miles (8 km) downstream of the site or to the nearest dam, with a connection by stream or floodway. One of the main goals of our model is to go beyond this simplistic type of social significance measure in order to better measure how much people benefit from flood regulation services of wetlands. To this end, we directly modeled flood extents and impacts. This is discussed in more detail in Chapter 3.

Hydrologic Modeling with HEC-HMS

HEC-HMS is a physically-based, semi-distributed model that can be used to simulate rainfall-runoff for a single event (Bedient et al. 2007). We derived variables for the HEC-HMS model parameters in several ways (Table 2-1). Many of the parameter variables were generated from spatial datasets (Appendix D) using the HEC-GeoHMS toolbox (Flemming and Doan 2013) in ArcGIS 10.1 (ESRI 2012). Other model parameters, including initial abstraction, were initially set to standard defaults (e.g., 0.2 initial abstraction ratio from Soil Conservation Service 1985). Once the model was fully parameterized, we calibrated it to observed precipitation and observed stream flow. After calibration, we validated the HEC-HMS model against other observed storms. After validation, we conducted sensitivity analysis on the calibrated model to determine sensitivity of the results to changes in selected parameters and assumptions. Once the hydrologic model validation and sensitivity analysis were complete, we assembled synthetic storms of known recurrence intervals to use for evaluating wetland restoration scenarios with the model.

Table 2-1. HEC-HMS parameters

Parameter-Variable	Element Type	Processing Method	Appendix
Routing – X	Reach parameter	Variable default, calibrated	Table D-3
Routing – K	Reach parameter	Calculated from basin lag, calibrated	Table D-3
Routing –SubReaches	Reach parameter	Calculated	Equation D-3
Loss – CN	Basin parameter	GeoHMS, Calibrated	Table D-2
Loss - % Impervious	Basin parameter	GeoHMS	Table L-1
Loss – Initial Abstraction	Basin parameter	Calculated using abstraction ratio	Equation
Transform - Lag Time	Basin parameter	GeoHMS, TR-55	not provided
Diversion Rate	Diversion parameter	Calculated, Inflow-Diversion paired data	Table B-1
Max Volume	Diversion parameter	Calculated, Python Toolbox	Table B-1
Storage-Discharge	Reservoir parameter	Storage-Discharge paired data	E
Initial Discharge	Reservoir parameter	0.5 m^3/s from Appendix E	E
Precipitation (in/hr)	Time series data	NOAA	C
Stream Flow (m^3/s)	Time series data	USGS gage	C

Parameterization

The HEC-HMS model allows for the use of several loss, routing, and transform methods. Of the available methods, we chose methods that could be parameterized using available geospatial data. Initial parameterization of the HEC-HMS model used here matches that used in Bousquin et al. (2014). Here we briefly summarize the methods used and draw attention to any revisions. A list of geospatial data used, a summary of parameterization, and values for those parameters are provided in Appendix D.

The watershed—or basin in HEC-HMS—includes the upper Woonasquatucket watershed with the USGS gage at its outflow. We delineated the basin into 139 subbasins using a subbasin contributing area threshold of 0.5 km^2 and adjustment for reservoirs. This threshold is less than the default, which is 1% of total watershed area or 1 km^2 in our basin, and therefore results in smaller subbasins. This produces a more distributed model and therefore comes closer to being able to differentiate single-site restorations. Because our models are differentiated at the subbasin level, to compare two sites within a single subbasin, a decision maker could evaluate sites based on their size and specific spatial characteristics such as slope. This is discussed further in Chapter 3. The HEC-HMS basin is divided into elements for subbasins, stream reaches, junctions, and other special features, including reservoirs and the subbasin diversions we used to represent wetlands (Appendix D).

Subbasin Elements

We used the loss method in HEC-HMS to generate runoff volume based on the Soil Conservation Service (SCS) CN approach (Williams and LaSeur 1976) and the percent impervious surface in each subbasin. The transform method, the rate runoff moves through subbasins to a junction with a downstream stream reach, used the SCS Unit Hydrograph with the lag time calculated using the TR-55 method (McCuen 1982).

Reach Elements

Stream reaches carry water from one junction to the next. We used the Muskingum routing method (McCarthy 1938) to parameterize the rate of this conveyance.

Reservoir Elements

We used Dams with a maximum storage of 12 acre-feet or more to identify reservoirs (Appendix D). In the model, reservoirs store and release water based on a storage-discharge curve and have an initial storage and discharge at the start of an observed storm event. The storage-discharge curves were based on a linear relationship between normal and maximum storage in Bousquin et al. (2014). We have since updated the tables used for the storage-discharge curves to add data from fitted exponential equations for each reservoir (see example in Appendix E). Modeled storm peaks improved slightly using these updated equations, despite a slight decrease in the overall fit of the model (Appendix E). As in Bousquin et al. (2014), if the maximum storage was exceeded, the discharge increased to approximately infinity to simulate unrestricted streamflow (see example in Appendix Figure E-3).

Reservoir discharges likely help sustain observed baseflow in the watershed. Without direct measurements of water storage or discharge before a storm event there is no way to know the actual initial reservoir discharge. Bousquin et al. (2014) used a default initial discharge of inflow = outflow, but we updated this to 0.5 m^3/s for the model presented here. This better represents reality and helps “pre-wet” the system, putting water into the hydrologic system before observed storm events (see Appendix E for comparisons).

Diversions

Diversions divert water from subbasins at a rate based on curves from inflow-diversion tables and the diversion's maximum volume. We used percent diverted, described in the Treatment of Wetlands section, to parameterize the inflow-diversion tables, where the diversion rate for each subbasin was a percentage of inflow proportionate to the percent of the subbasin that is wetland (Appendix B). Once a diversion reaches its maximum—the maximum outflow diverted, described in the Treatment of Wetlands section—it stops diverting water. No initial storage of water in wetlands was assumed beyond that accounted for in the volume calculation assumptions.

Baseflow

Baseflow observed in the system is thought to predominately stem from reservoirs, but the model requires parameterized baseflow to be distributed throughout the subbasins. Therefore we computed and accounted for total baseflow outside the model. To remove baseflow from the model, we assumed the minimum flow from the modeled time period to be baseflow and removed for the entire time period, following the constant discharge method (Hall 1968). Baseflow removed from each of the calibration and validation storms is provided in Appendix F.

Precipitation and Runoff Data

HEC-HMS uses rainfall event precipitation as an input and predicts runoff, in the form of a hydrograph. The model does this by parameterizing basin characteristics in terms of the amount of water, where it travels, and how much time it takes. The model is refined through calibration to observed rainfall and runoff.

Our observed time series data for both precipitation and runoff came from stationary gages. We used hourly precipitation data from the Providence Airport Station (COOP: 376698; NOAA 2014), and stream flow data from the USGS gage station at Centerdale (USGS# 01114500; USGS 2014). The precipitation station is outside the modeled watershed, 15.9 km (9.9 mi) south of the USGS gage station.

Storm Selection for Calibration and Validation

We calibrated the model to observed storms with the largest flood magnitudes to ensure that our models would capture flooding impacts. We considered this a reasonable choice, as these storms are becoming more common. Since some of the increase in flooding in recent years may be caused by changes in land cover, we chose modeled storms that occurred between 2000 and 2010 for calibration, since spatial data used to set initial parameter values were gathered during this time frame. The ten largest stream flow peaks within this time period were identified for initial consideration (Table 2-2).

The storm of record occurred in March of 2010; however, much of the data around the peak of the event were not available. We chose to calibrate to the second largest event (October 15, 2005; Figure 2-3) and the third largest event (March 22, 2001; Figure 2-4). We selected these storms to calibrate to both spring conditions with very wet antecedent conditions, and fall conditions where there was more time between events. We used the fourth and sixth largest storms, both spring storms, for validation. We could not use the fifth largest storm because it dropped below freezing temperatures during the duration of the storm and our model was not set up to handle snow. Details on the four storm event start and stop times, durations, and baseflow precipitation and hydrographs are provided in Appendix F. Probabilities and estimated recurrence intervals from analysis in PeakFQ (Flynn et al. 2006) for the storm event discharges are provided in Appendix C.

Table 2-2. Top 10 Streamflow peaks for Woonasquatucket River gage for 2000-2010 in order of peak flow

Rank	Date	Height (m)	Flow (m^3/s)	Use
1	Mar. 30, 2010	2.80	51.25	Not used: missing data
2	Oct. 15, 2005	2.52	43.32	Fall calibration storm
3	Mar. 22, 2001	2.00	30.30	Spring calibration storm
4	Apr. 03, 2005	1.86	26.70	Validation storm 1
5	Dec. 12, 2008	1.85	26.56	Not used: below freezing
6	Apr. 16, 2007	1.74	24.10	Validation storm 2
7	Feb. 13, 2008	1.67	22.46	Not used
8	Dec. 13, 2010	1.59	20.70	Not used
9	Apr. 14, 2004	1.55	19.91	Not used
10	Apr. 22, 2000	1.28	14.07	Not used

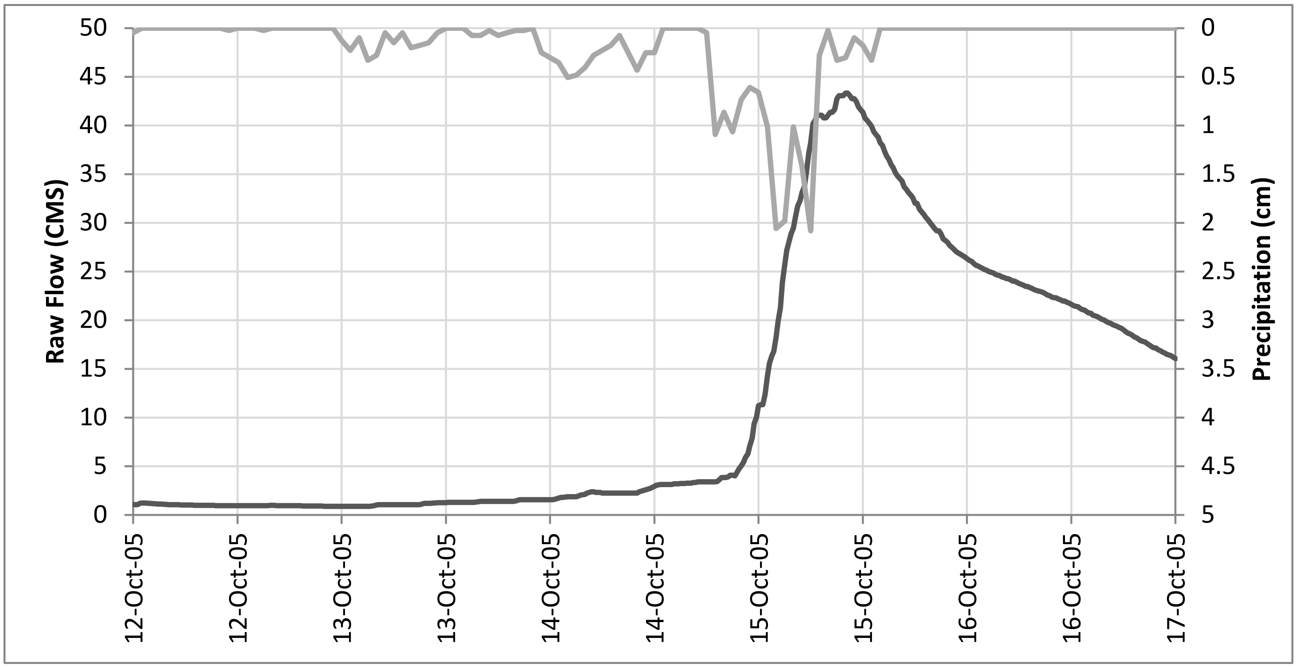

Figure 2-3. Hydrograph of 15 min interval flow (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the "Fall '05" calibration storm (October 15, 2005). Raw flows are shown without baseflow removed.

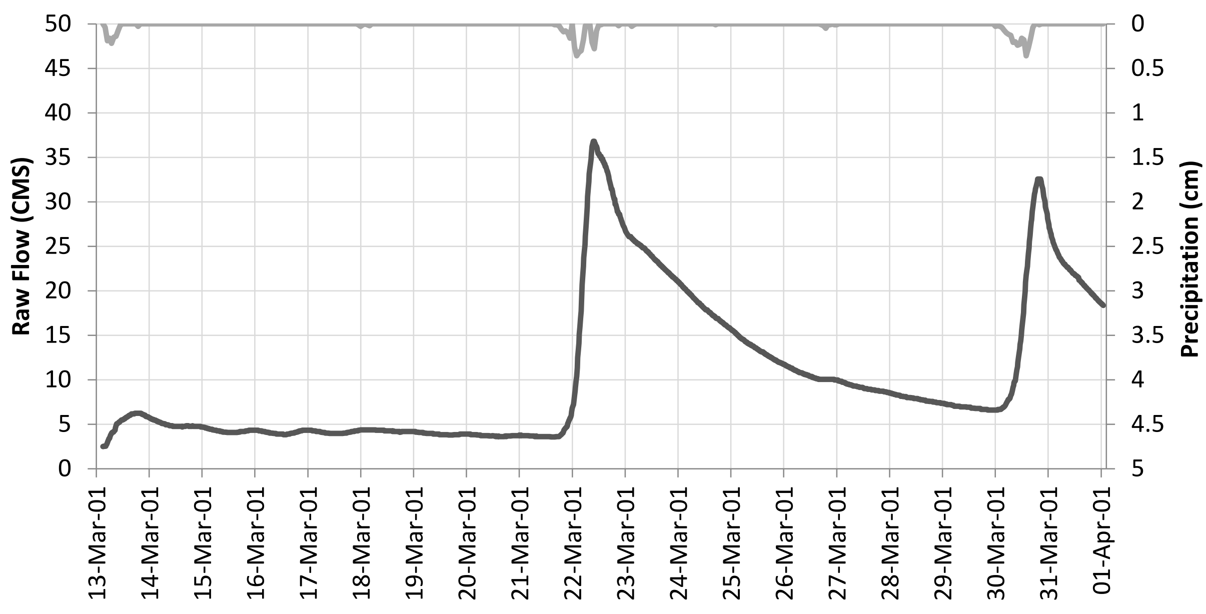

Figure 2-4. Hydrograph of 15 min interval flow (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the "Spring '01" calibration storm (March 22, 2001). Raw flows are shown without baseflow removed.

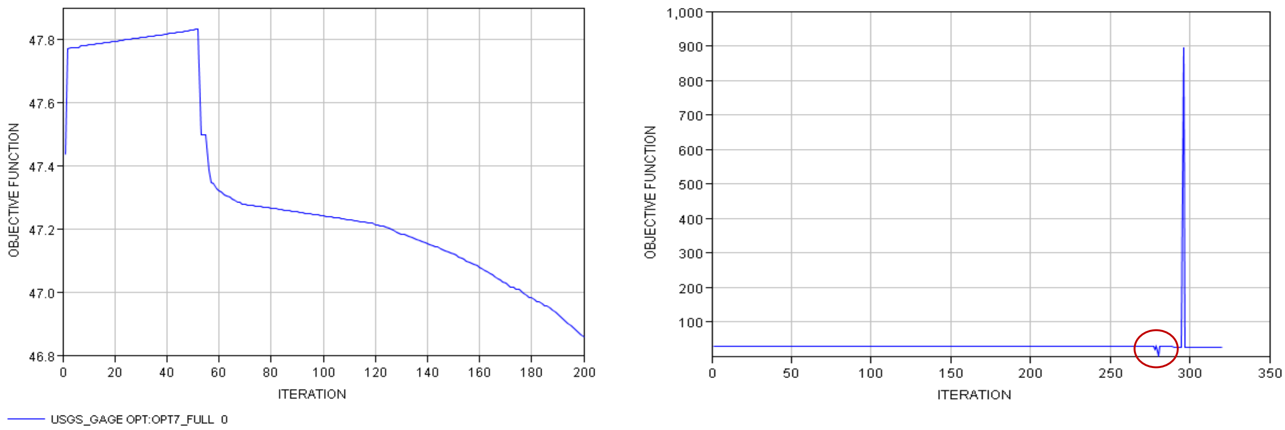

Figure 2-5. Two sets of optimizations were performed in sequence for the Spring '01 storm. The first (a) had 200 iterations the second (b) had 320. The objective function is defined by how close the model matches the observed peak. The area circled in red had the optimal parameters used by the model (Calibration 2) resulting in lowest objective function.

Calibration

We began by machine calibrating to the Fall '05 storm (Oct '05; 1530 cfs) and Spring '01 storm (Mar '01; 1070 cfs). Precipitation occurred both before and after the Spring '01 storm. The model run included some minor preceding precipitation to pre-wet the model, but ended before precipitation for a second peak to ensure calibration and metrics for model fit were for the correct peak flow.

The pre-calibrated model peak flows were overestimated for the Fall '05 storm and underestimated for the Spring '01 storm (Figures 2-6 and 2-7). Machine calibrations varied three parameters: Basin Loss CN, Routing Muskingum X, and Routing Muskingum K. The machine calibration procedure followed for the Fall '05 storm is detailed in Bousquin et al. (2014). We ran the calibration for the Spring '01 storm as two consecutive optimizations. The first Spring '01 optimization included 200 iterations (Figure 2-5a); the second included another 320 iterations (Figure 2-5b), using the results of the first optimization. The later iterations in the second optimization were suboptimal, so we used the best calibration from earlier iterations (Figure 2-5b iterations circled in red). Appendix D presents all pre- and post-calibration parameter values.

In these initial calibrations, Basin Loss CN calibrations diverged for the two models, but both routing parameters were calibrated in the same direction (Appendix D). This is understandable considering the differences between the types of storms that take place in the fall compared to the spring, in terms of antecedent moisture conditions, intensity, and other factors. The conditions for the Spring ’01 storm were generally wetter, with larger preceding baseflow (2.5 m^3/s compared to 0.88 m^3/s) and more rainfall observed in the time period leading up to the storm. To address this, we developed two models, with the same optimized routing parameters but different loss parameters, depending on whether the storm occurred during wet or dry conditions.

Subbasin loss is expected to be most influenced by wet and dry conditions. The HEC-HMS model uses a default initial abstraction ratio (which is the ratio of the amount of water before runoff to the potential maximum soil moisture retention after runoff begins) of 0.2 in loss calculations, based on the National Engineering Handbook (SCS 1985, Figure 10.2). Other studies have found that most basins have a lower abstraction ratio, closer to 0.05 (Woodward et al. 2003). We maintained the 0.2 ratio for the dry condition storm, but reduced the initial abstraction ratio to 0.02 for the wet condition storm to emphasize the wet conditions. CN can also be adjusted to better represent antecedent soil moisture conditions. Antecedent Soil Moisture Condition II (AMC II) definitions of CN used for pre-calibration were converted to AMC I for dry conditions (Fall ’05) and to AMC III for wet conditions (Spring ’01) following Ward and Trimble (2003) and NRCS (1984). This method for calibrating CN was preferable since it weighted increases and decreases in CN based on original CN, where the machine calibration altered all CNs uniformly. Without further calibration, these changes to initial abstraction and CN improved performance of both models (Table D-4; Appendix D).

Although both machine calibrations increased routing parameters, increases were not the same for both calibrations. Having altered the loss calculations for both models after calibration may have also altered the optimal routing parameters. To address this, we calculated a blended routing parameter value for each reach based on the average increase for all reaches (Appendix D-3). The three routing parameter values (Fall ’05 calibrated, Spring ’01 calibrated, and blended calibration) showed the least peak error across both storms using the wet, Spring ’01 calibrated values.

Validation

The two calibrated models, represented by blue lines in Figures 2-6 and 2-7, outperformed the uncalibrated models, represented by red lines, and showed peak flows very close to those observed (Table 2-3). Modeled volume of both calibrated models was less than the observed volume. Water diverted to wetlands may account for some of this missing volume.

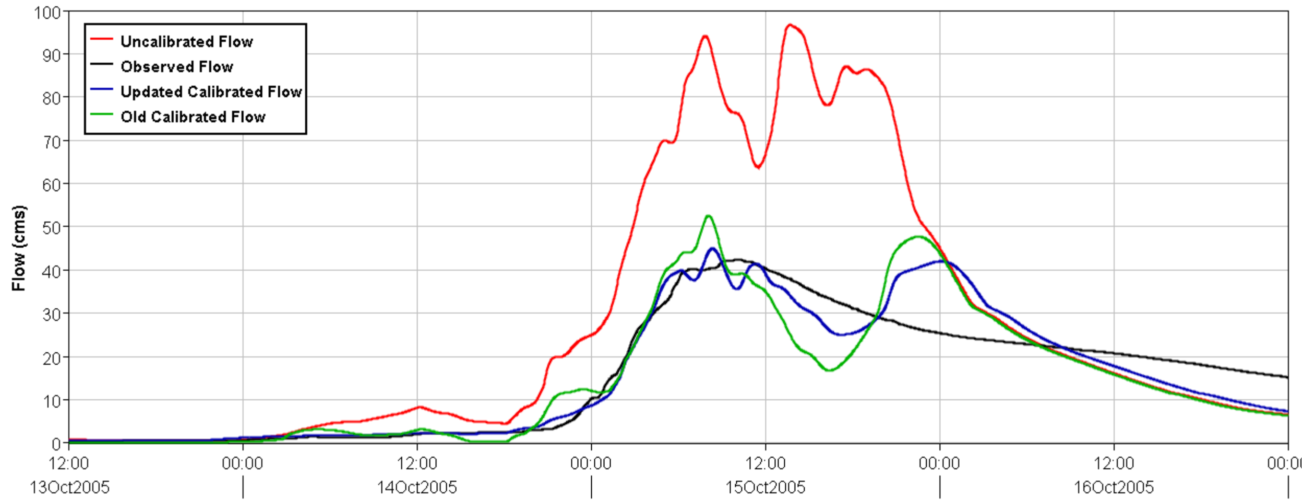

Figure 2-6. Fall '05 storm (Oct '05) hydrographs for Observed Flow (Black), Calibrated Flow (Blue), Calibrated Flow from Bousquin et al. (2014) (Light Blue), and Un-calibrated Flow (Red).

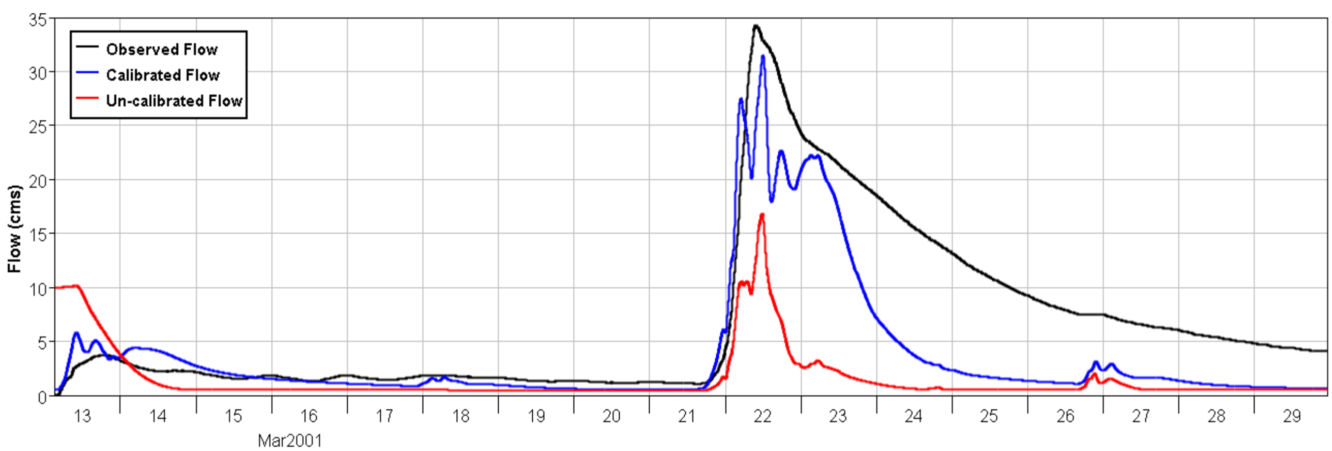

Figure 2-7. Spring '01 storm (Mar '01) hydrographs for Observed Flow (Black), Calibrated Flow (Blue), and Un-calibrated Flow (Red).

Table 2-3. Validation Results for Spring '01 and Fall '05 models validated and fit across four observed storms

	Observed Storms											
	Mar '01 (Spring Calibration)			Apr '05 (Validation 1)			Oct '05 (Fall Calibration)			Apr '07 (Validation 2)		
	Peak	SD		Peak	SD		Peak	SD		Peak	SD	
Observed (m^3/s)	34.3	7.6		26.1	6.2		42.4	13.5		20.0	5.4	
	Peak	RMSE	N-S	Peak	RMSE	N-S	Peak	RMSE	N-S	Peak	RMSE	N-S
Fall model ('05)	5.0	9.4	-0.54	4.9	8.6	-0.94	45.2	4.2	0.91	3.9	7.2	-0.69
Spring model ('01)	31.5	5.0	0.56	14.0	4.2	0.52	168.7	42.3	-8.77	30.9	5.5	-0.01

We evaluated model fit using Nash-Sutcliffe (N-S) efficiency and Root Mean Square Error (RMSE) statistics (Legates and McCabe 1999; Table 2-3). The N-S efficiency (Nash and Sutcliffe 1970) measures the relative difference between the model residuals (difference between modeled values and observed values) and the data's variance (difference between observed values and the observed mean). The index can range from -∞ to 1, with 1 representing a perfect fit between the model and observation and 0 indicating that the model is as good as using the mean of the observed data. N-S values between 0 and 1 are generally considered acceptable values; however for stream flow model evaluation, an N-S efficiency greater than 0.5 is considered satisfactory (Moriasi et al. 2007). We consider the 0.5 N-S standard for satisfactory model fit to be overly restrictive for our model application, because N-S is based on all discharges and in our model the volume of water diverted to wetlands was removed from the system rather than rereleased, artificially reducing discharges after the peak flow. Therefore, we judged an N-S value greater than 0 as acceptable for our evaluation. The Spring '01 calibration met the > 0.5 criterion for the April 2005 storm, and came very close to the > 0 criterion for the April 2007 storm. The Fall '05 calibration did not meet the > 0.5 criterion for either of the validation storms (which were both spring storms). This reinforces the importance of antecedent conditions and using a wetter spring-calibrated model for spring storms.

RMSE is the un-standardized sum of differences between observations and the model output; therefore this index is in the units of the modeled parameter (in this case flow, or m^3/s). A zero RMSE indicates a perfect fit between the modeled and observed data, with lower RMSE indicating a better model fit. An RMSE less than 70% of the standard deviation is considered satisfactory (Moriasi et al. 2007). The RMSE results were similar to those of the N-S efficiency tests. The Spring '01 model RMSE was < 70% of the standard deviation observed during the March 2001 calibration storm (7.6 m^3/s; RMSE < 5.3) and the April 2005 validation storm (6.2 m^3/s; RMSE < 4.33). The Spring '01 model RMSE was > 70% of the standard deviation observed for the April 2007 storm (13.5 m^3/s; RMSE < 9.5) and the October 2005 storm (5.4 m^3/s; RMSE < 3.8). While the Fall '05 model RMSE met the RMSE validation criterion for the storm used to calibrate it, it did not meet the criterion for any of the validation storms.

Sensitivity Analysis

We conducted sensitivity analysis using the Spring '01 calibrated model. We excluded parameters from the sensitivity analysis based on 3 criteria: those that were used during calibration (routing Muskingum X and K, basin loss CN, and initial abstraction ratio), those that were calculated based on those calibrated parameters (Muskingum subreaches), and those that were used to parameterize other HEC-HMS parameters externally from the model (percent imperviousness and slope). The following sections describe each of the sensitivity analyses.

Lag Method

The basin lag method can be calculated using either CN or the TR-55 method. We chose the TR-55 method because it integrates a greater number of basin characteristics. Comparing results from the two methods suggested that the model was not sensitive to the choice of loss method (Figure 2-8).

Precipitation

We examined five scenarios by increasing or decreasing all rainfall during the modeled duration by an equal factor (Figure 2-9). As expected, the model was very sensitive to changes in precipitation (Figure 2-10). The results showed a shift in hydrograph shape and peak flow when precipitation was increased by 50% or more. Small increases or decreases in precipitation (+/- 10%) resulted in proportional changes in peak flow (+/- 10%). However, more extreme increases or decreases in precipitation (+/- 50%) resulted in disproportionately larger changes in peak flow (+100% and -30%).

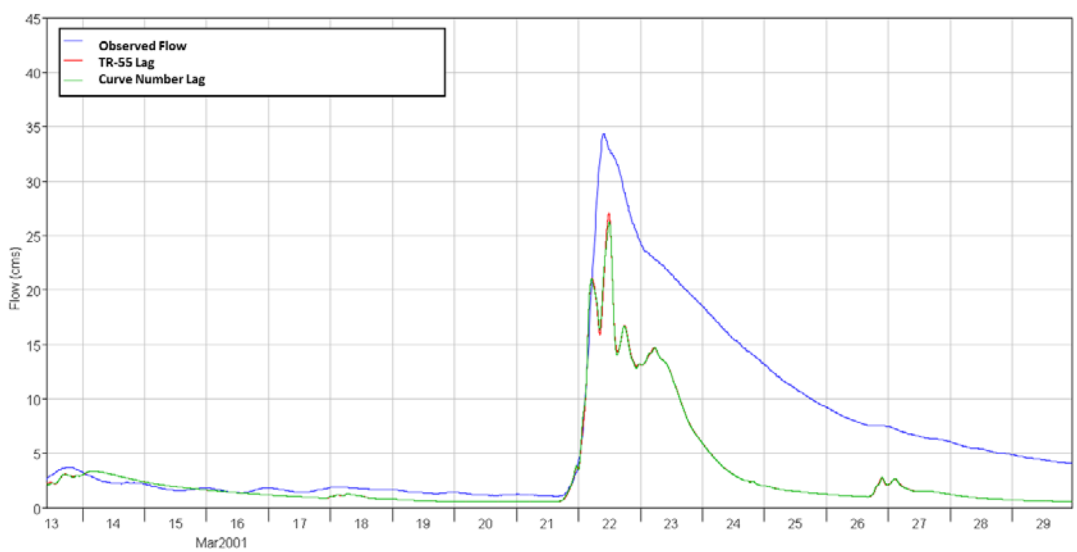

Figure 2-8. Results of altering lag method in comparison to observed flow (Blue). The TR-55 method (Green) was used in the model, where the Curve Number method (Red) was not.

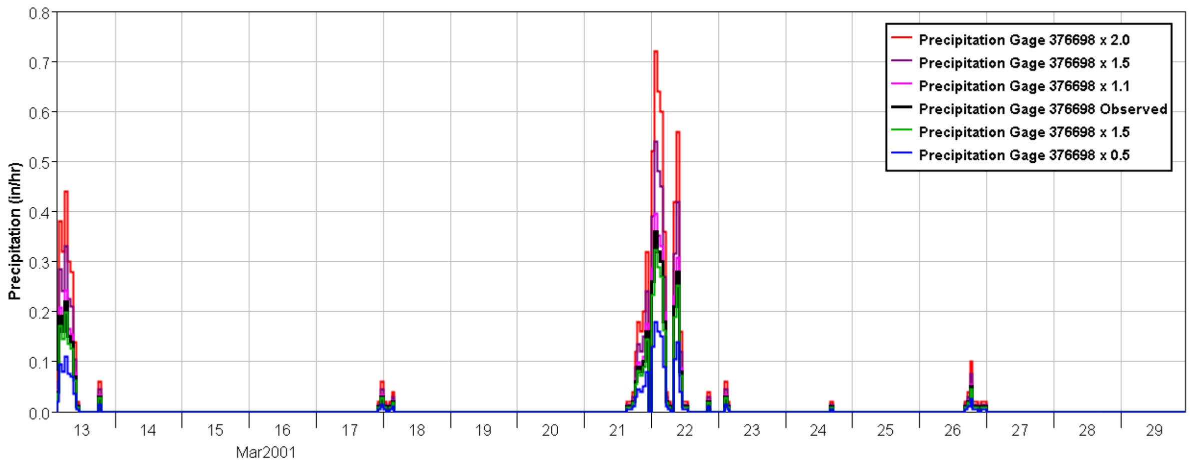

Figure 2-9. Precipitation for the different sensitivity analysis runs.

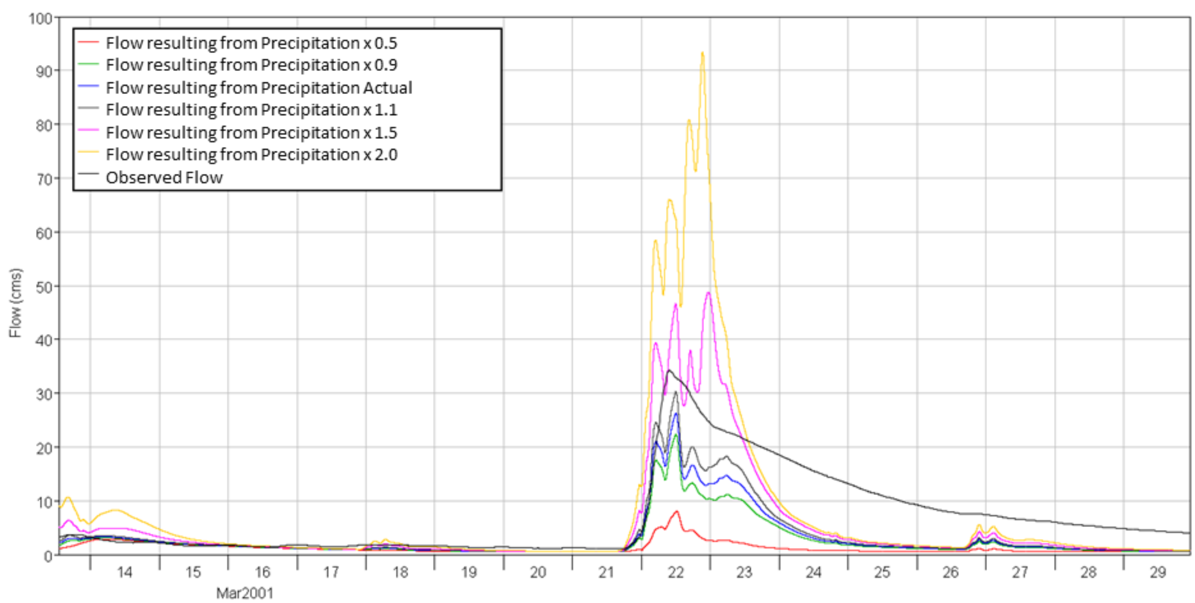

Figure 2-10. Sensitivity analysis hydrographs based on varied precipitation.

Wetland Percent Diverted

Important assumptions and uncertainties surrounding parameters dictating the percent of subbasin outflow diverted to wetlands have already been detailed in the Treatment of Wetlands section of this chapter. Results of the modeling effort rely on those assumptions, so we performed sensitivity analysis on those parameters to test our assumptions. Using the most conservative assumption—where percent diverted is equal to the percent of the subbasin surface area in wetlands—as a baseline, we increased the area contributing to wetlands in the remaining catchment area by 10%, 25%, 50%, and 75%. For example, for the 50% run, represented as a black line (Figure 2-11), half of the non-wetland area was assumed to contribute flow into the wetlands. Although this may seem like an extreme increase in contributing area, a conservative 100-m buffer around wetlands adds an area comparable to the 50% run used in sensitivity analysis (Appendix B). The results showed a strong relation between peak flow and assumptions about the subbasin area that contributes flow to wetlands, which we used in making generalizations to develop Tier II indicators (as described in Chapter 3).

Wetland Max Volume Method

During initial parameterization of wetlands, we calculated wetland volume following Lane and D'Amico (2010). This method derives volume from the area of a wetland times its height—the average elevation of wetland perimeter vertices. A more conservative estimate of height was based on the minimum elevation of wetland perimeter vertices. We tested model sensitivity to this assumption, and found a small difference (Figure 2-12). During this storm only one subbasin divergence (D_W2980) reached its average-based maximum volume. In a scenario where more of the subbasins reach maximum volume, such as when there is a simultaneous increase in percent diverted to wetlands (Figure 2-11) or precipitation is increased (Figure 2-10), we suspect the model sensitivity to wetland maximum volume would increase.

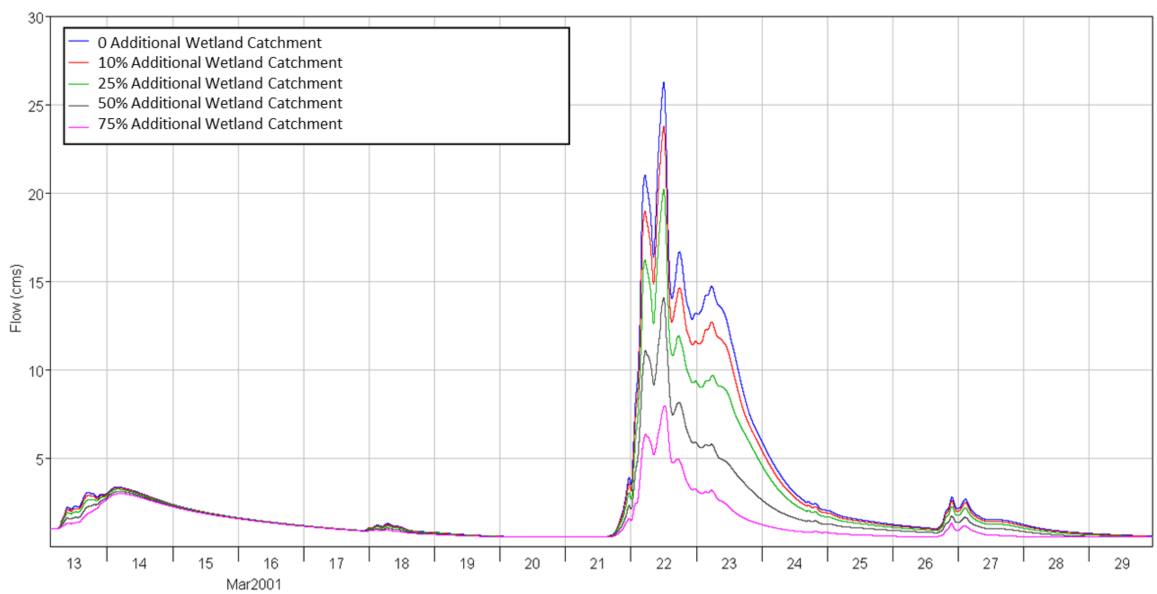

Figure 2-11. Resulting hydrographs from sensitivity analysis model runs with different wetland percent diverted assumptions ranging from the model assumption (Blue) to a 75% increase in wetland catchment area contributing to diversion (Pink).

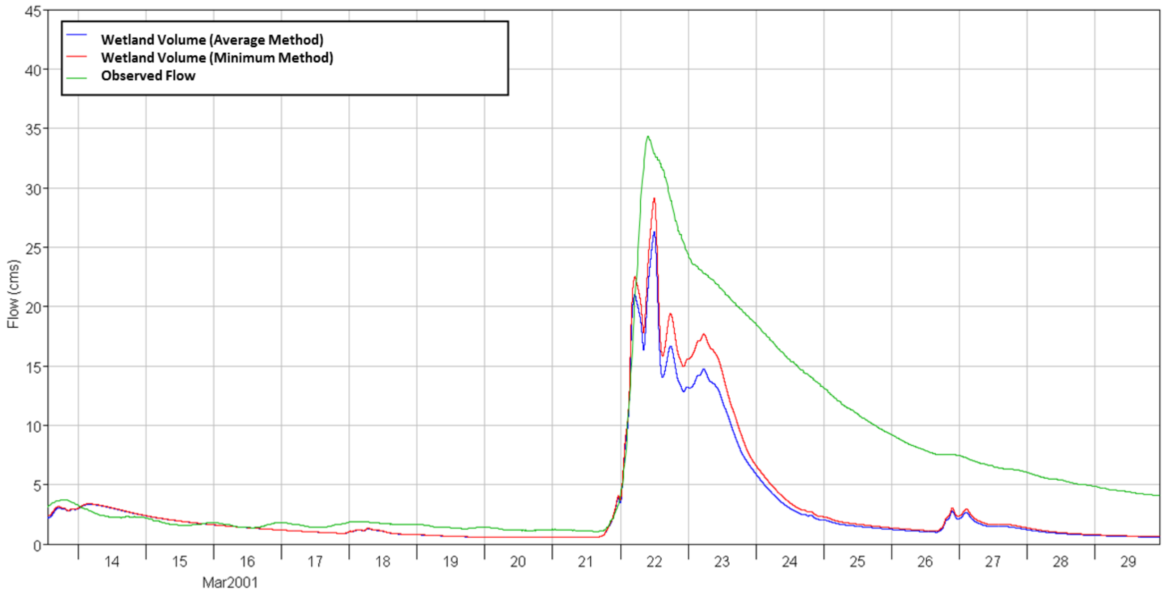

Figure 2-12. Sensitivity analysis results hydrographs with different wetland max volume assumptions: water retention height derived from average (blue) or minimum (red) elevation of wetland perimeter vertices.

Hydraulic Modeling with HEC-RAS

We estimated the extent of flooding using the hydraulic model Hydrologic Engineering Center River Analysis System (HEC-RAS) of the Army Corps of Engineers (Brunner 2010). HEC-RAS generates water surface elevations at profiles along a channel for steady, gradually varying flow using a standard step method for finding solutions to the 1-dimensional energy equation (Bedient et al. 2007).

Surface profiles for HEC-RAS use cross sections along the river reaches and are part of the HEC-RAS geometry dataset. HEC-RAS geometry includes river reaches (the synthetic streams generated from HEC-HMS, not including first order stream reaches), cross sections, and other constructed elements along the river such as bridges, culverts, levees, and dams. We used the geometry dataset from Zarriello et al. (2013), supplemented by data generated by Bousquin et al. (2014). The added HEC-RAS geometry was characterized with elevations, from the DEM, and roughness, from landuse data (Appendix D). We developed an R script (Appendix J) to extract peak flows from HEC-HMS to use as inputs into HEC-RAS to perform steady flow analysis.

USGS validated their hydraulic model against observed storm elevations (Zarriello et al. 2013). The additional geometric features from Bousquin et al. (2014) were not as intensively validated, but were compared to Digital Flood Insurance Rate Maps (DFIRMs) based on flood extent and input from the Town Engineer of Smithfield, RI.

The final modeling step was flood impact modeling, where we exported flood depth profiles from HEC-RAS into ArcGIS and compared to other data to estimate the impact of the flood.

Wetland Restoration Scenarios

We developed 12 scenarios to investigate the influence of different levels of wetland restoration, across a range of storm events, on stream flow and resulting flood extent. We used the resulting peak flows and flood maps to demonstrate the development of Tier III benefit indicators to compare wetland restoration scenarios (see Chapter 3 Tier III Assessment Section). We also relied on the wetland restoration scenario results to develop Tier II benefit indicators, generalizing the results to develop indicators of differences in potential restoration sites that correlate with differences in flood reduction services and subsequent benefits from flood damage reduction.

Size of Restoration in Scenarios

To simulate various wetland restoration scenarios we needed to vary the volume and/or the area of wetlands by subbasin. The sensitivity analysis suggested the model was not sensitive to changes in wetland volume because most subbasins did not exceed their maximum volume–when we ran the model for the Spring ’01 storm (wet conditions), only HEC-HMS diversions in 2 subbasins reached their maximum volume. Also, altering the wetland volume calculation method showed only a minor response in flow (Figure 2-12). However, sensitivity analysis of wetland percent divergence, which is equivalent in the model to increasing wetland area in a subbasin, showed that model results were very sensitive to this parameter, where a 25% increase in wetland divergence resulted in greater than 25% decrease in peak flow (Figure 2-11). Based on these findings, we simulated wetland restoration using changes in wetland surface area.

The subbasins differ in size and in proportion of land that is currently wetlands. Representing wetland restoration scenarios in the hydrologic model with across-the-board increases by a given area (e.g., adding 1000 m^2 of wetland area to each subbasin) would result in unequal changes in subbasin percent diverted, which would make it difficult to make generalizations for Tier II indicators. Alternatively, representing wetland restoration scenarios as across-the-board increases by a given percentage (e.g., adding 10% to wetland area in each subbasin) would increase the proportion of wetlands in some subbasins to greater than 100%. Instead, we reduced the non-wetland area of the subbasin by a set percent (e.g., for a 10% conversion of non-wetland area, a subbasin with 20% wetlands would increase to 28% wetlands). This method ensured a consistent change in subbasin outflow while never exceeding 100% wetlands in a subbasin.

We selected a range of wetland change scenarios that were realistic enough to inform Tier III indicators, but also large enough to show detectable changes in flow downstream for developing Tier II indicators. To determine what might constitute realistic wetland scenarios, we used the average and range in surface area of restorable wetland fill sites identified in the watershed by Golet et al. (2003). The average area was 6,026 m^2, which is approximately the area that would be added by a 1% conversion of non-wetland area (5,922 m^2). The maximum area restorable to wetlands identified by Golet et al. (2003) was 88,383 m^2, which is comparable to the area increased with a 10% conversion (59,216 m^2). Based on our sensitivity analysis, where contributing area was defined by the 100-m buffer around a wetland and was comparable to a 50% conversion, we also modeled a 25% restoration scenario to account for larger restorations; and, to represent restoration across a range of potential scenarios, we added a 5% scenario. So, the four restoration scenarios we ran were 1%, 5%, 10% and 25% conversions of non-wetland area to wetlands.

Location of Restoration in Scenarios

Because the hydrologic model is spatially lumped to the subbasin, the location of simulated restorations can only be varied by subbasin. However, we expected the wetland scenarios applied across different subbasins to have differing effects on flow and flooding. For example, a subbasin that is directly upstream of a dam might alter downstream flow differently than one with no dam downstream. Since there were many subbasin characteristics (e.g., distance to outlet, downstream dams, and basin area) that might correlate with changes in flow, we ran all four restoration size scenarios for each of the 139 subbasins individually, resulting in 556 "basin scenarios."

Storm Events

We developed synthetic storms based on recently updated gridded storm recurrence period data (DeGaetano and Zarrow 2011). We interpolated the gridded data over the entire watershed rather than measuring at a single point such as the meteorological station data used for calibration. To integrate these data, we aggregated the data to subbasins using their centroids. We imported three storms, with 100%, 20% and 4% probabilities (1-year, 5-year, and 25-year recurrence intervals) into HEC-HMS. Each of these three storms had 6 duration intervals (1-hour, 2-hour, 3-hour, 6-hour, 12-hour, and 24-hour) used to create the synthetic storm and define peak rainfall intensity (DeGaetano and Zarrow 2011). We excluded shorter duration intervals because they exceeded the resolution of the hourly precipitation data used for calibration and, because the model was run on a 1-minute time interval, shorter duration intervals resulted in rainfall with unrealistically high peaks (Appendix G).

The recurrence intervals for precipitation-based synthetic storms (1-year, 5-year, and 25-year) did not result in a peak flow or flood with the same recurrence interval. This is primarily because the evapotranspiration and antecedent moisture conditions at the time of the event greatly influence how much precipitation actually ends up as runoff. We demonstrated this by estimating the peak flow recurrence intervals for the three synthetic storms using both the spring and fall calibrated models in peakFQ analysis (Table 2-4).

For the basin scenarios, we modeled the range of synthetic storms with the model calibrated to wet conditions (Spring '01), because it validated better than the Fall '05 model and because wet conditions typically lead to more flooding from the same size precipitation event. We ran the four restoration sizes and three synthetic storms for each subbasin. A total of 1668 simulations were run using the hydrologic and hydraulic models, plus a baseline for each synthetic storm

(Table 2-5). These hydrologic and hydraulic simulations are the root of the process used to derive Tier II and Tier III indicators presented in Chapter 3.

Table 2-4. Approximate recurrence intervals for synthetic storms and their peak flows using different models

Recurrence	Spring			Fall		
Precipitation	1 Year	5 Year	25 Year	1 Year	5 Year	25 Year
Flow	10 Year	100 Year	500+ Year	1 Year	3 Year	5 Year

Table 2-5. Number of simulations generated based on three synthetic storms, four restoration sizes and 139 individual basins

	1-Year	5-Year	25-Year
Restoration			
1%	139	139	139
5%	139	139	139
10%	139	139	139
25%	139	139	139
Baseline	1	1	1

CHAPTER 3: INDICATORS

In Chapter 2 we introduced and presented models that we used to generate subbasin-specific scenarios for four different wetland restoration sizes and three storm magnitudes (Table 2-5). In this chapter, we demonstrate how we derived Tier II and Tier III indicators from these scenario results. First, we demonstrate how the scenario results can be used to produce Tier III indicators, and illustrate their application to a comparison of wetland restoration in two subbasins in the Woonasquatucket Watershed. We then present the development of Tier II indicators using the scenario results combined with sensitivity analysis of the models presented in Chapter 2.

The Indicators Framework

We compiled the ecosystem service benefit indicators using a framework developed by Mazzotta and Wainger (in preparation). We summarize the framework here. It is based on four questions that guide the process of indicator selection and measurement. It is not necessary to fully answer all of the questions; they are ordered so that answering each question in turn adds additional information to an ecosystem service benefits assessment. Each of these questions may in turn be answered in lesser or greater detail. Thus, question 1 alone can provide some information that adds value to a functional assessment in moving towards Benefit Relevant Indicators (BRIs), and answering the first 3 questions provides basic BRIs. Adding information that addresses question 4 will move the assessment closer to value assessment. The questions are:

1. Is an ecosystem service supplied?

By definition (Munns et al. 2015), ecosystem services require use or appreciation by people and thus are distinguished based on the interaction of supplied ecological outputs and demand for those outputs by people. This step determines whether an ecosystem service exists by assessing potential supply and demand of an ecosystem service by evaluating three things. First, a service can only be supplied if wetland functioning meets thresholds required to provide benefits to people. Second, people must care about, or demand, the service. For flood regulation, this means that the wetland regulates flood waters at a level that will protect structures and infrastructure that people care about. Third, in many cases other necessary conditions, such as access points for recreational services, must be present for people to benefit. These necessary conditions may include both physical supports to enjoying the ecosystem service, such as required infrastructure and other conditions that allow physical access, and institutional supports or constraints, such as regulatory limits to harvest (Olander et al. 2015). In the case of flood regulation, this is typically not a relevant concern.

2. How likely is it that the service will continue to be provided over the long run?

This step assesses temporal reliability of ecosystem services by considering factors that affect the probability that the wetland will continue to function at a sufficient level to provide services over time. This is important to consider when comparing projects because two sites may provide identical benefits in the short run, but if one of the sites is threatened by stressors, it may not continue to provide services into the future, resulting in a lower overall stream of benefits for that site. The results of this step provide an indicator of whether and how long a flow of benefits is likely to continue into the future.

3. How many people benefit?

This step assesses the number of people who benefit, which is the most basic measure of the magnitude of benefits. If an ecosystem service exists (as determined through Question 1), the number of people who benefit will be a strong indicator of its overall value. The total benefits of an ecosystem change depend, to a large extent, on how many people stand to benefit. While it is important to understand how much each individual values a change, the aggregate social value of a change can be more sensitive to the size of the beneficiary pool than the magnitude of change to an individual (Bateman et al. 2006), so as long as the average increase to an individual is positive, the number of beneficiaries will be an important benefit metric. Therefore, the number of beneficiaries can be used as a primary indicator in the most basic of benefit indicator approaches.

4. By how much do people benefit?

This step assesses the magnitude of benefits to affected individuals or households, using indicators of magnitude of the change in quality or quantity of the ecosystem service relative to the pre-change baseline, availability and quality of substitutes, availability and quality of any necessary complementary inputs, and strength of preferences for the ecosystem service. This is a more difficult question to address using indicators than the first three questions.

This step may also include measures related to environmental justice regarding who is likely to benefit in different locations. An example is the Social Vulnerability Index (Cutter et al. 2003).

Tier III Assessment

This section describes how we developed Tier III indicators based on the models presented in Chapter 2, and presents an example of a hypothetical comparison of wetland restoration in two subbasins (Figure 3-1). Because our model compares restorations at the subbasin scale, the model cannot compare sites within a given subbasin (mean area=0.7 km^2; 170 acres). Flows from a subbasin depend on total wetland area within that subbasin, so a comparison of two sites within a subbasin would result in the larger site ranked as superior. Therefore, a user wanting to compare sites within a subbasin might make that decision either by simply selecting the larger site, or by considering additional location-specific factors that might favor one site over the other (e.g., slope), using best professional judgment or alternative functional assessment tools. In addition, the decision maker might consider other ecosystem services provided by each site.

Flood reduction results for the two restorations in the subbasins presented are from the scenario results modeled (Chapter 2, Wetland Restoration Scenarios Section). Of the four wetland restorations (1%, 5%, 10% and 25%) the largest, 25%, is shown in the hypothetical comparison to illustrate the largest predicted decrease in flooding. Of the three synthetic storm conditions (1-year, 5-year, 25-year), we chose the 1-year storm because it produced a baseline peak flow (1116 cfs, a 10-year flow) closest to those used to calibrate the hydrologic model (1070 cfs). Although larger storms are used in flood mapping such as the DFIRM, regional stormwater regulations primarily target small events. Stormwater regulations in Rhode Island require stormwater conveyances to handle at least the 10-year, 24-hour precipitation storm event (RI DEM 2010), and Massachusetts requires that development peak discharge rates are equal to or less than the predevelopment for the 2-year and 10-year 24-hour precipitation events (MassDEP 2008). However, both states require onsite attenuation or no offsite impacts for the 100-year 24-hour precipitation events. We would expect the same comparison based on a larger storm to show more beneficiaries in the flood zone who would potentially benefit from the restored wetlands.

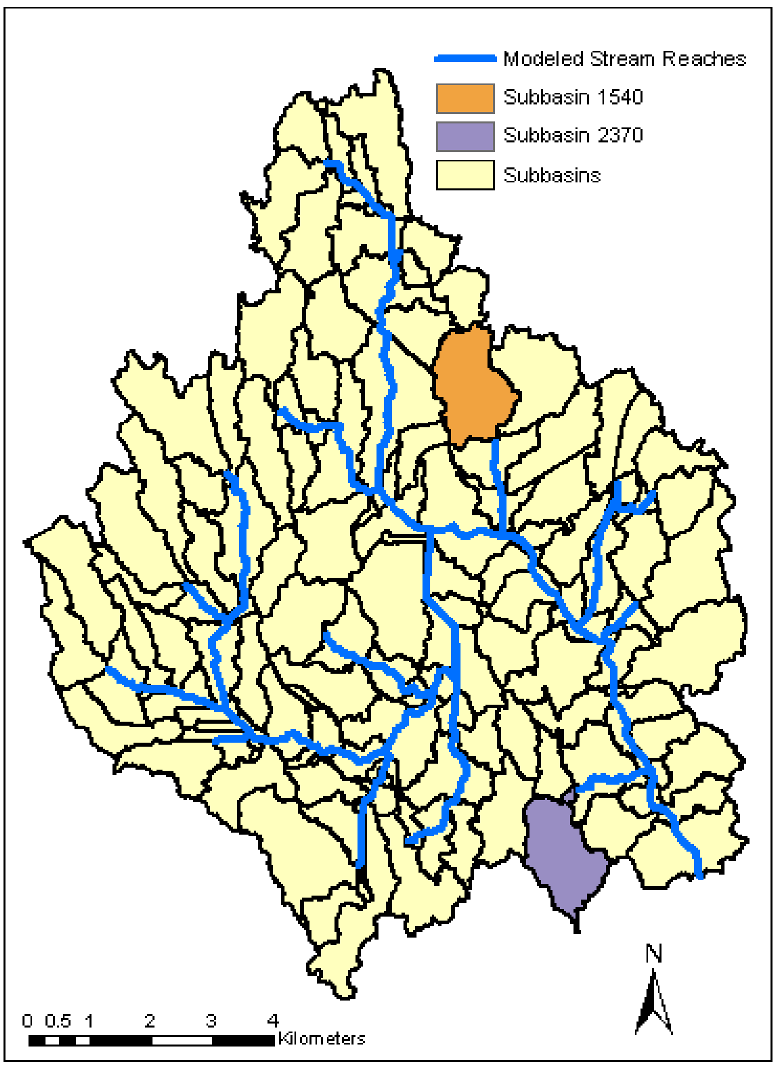

Figure 3-1. Subbasins selected for the example application

The remainder of this section describes how we translated our model results into Tier III indicators, and presents our example application of these indicators. Figure 3-2 illustrates the process of applying the indicators. Throughout, we have made various assumptions that a user of this approach might change, depending on the needs and context of the analysis. Wherever possible we have specified our assumptions, and discuss how a user might modify the analysis to address different concerns.

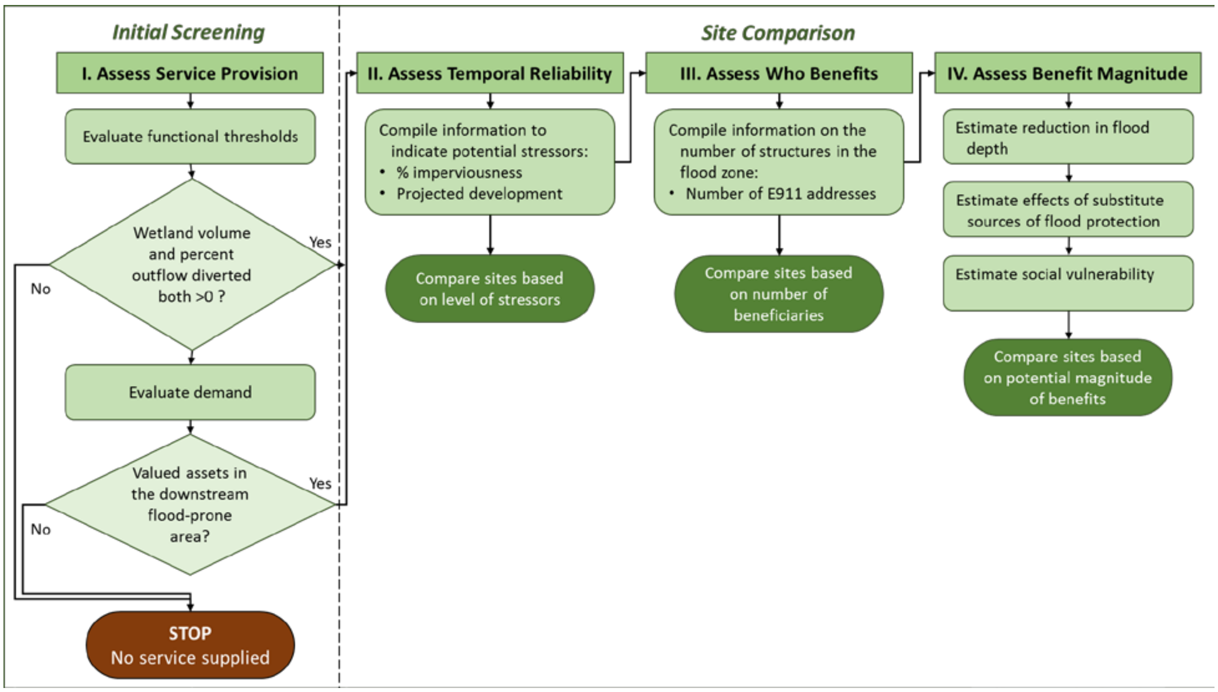

Figure 3-2. Indicator application process

I. Assess existence of an ecosystem service

We have applied this step as an initial screening, using yes/no questions to determine whether functioning meets the necessary threshold and potential beneficiaries exist.

Are functional thresholds required to supply the service met?

To provide flood regulation services the wetland needs to retain enough water to reduce downstream peak flows. To perform this retention function, a wetland must provide adequate retention volume and have a large enough source of water available for redirection into retention. In our model, we had two metrics for the retention functioning of wetlands, wetland volume and percent outflow diverted, both quantified using GIS calculations (see Chapter 2 Treatment of Wetlands in the Models Section and Appendix B). Based on observations of modeled flows we have assumed that, provided both metrics are greater than 0, the service threshold is met. This assumption can be adjusted within our approach if a stricter definition is desired, but it was adequate for the smallest restoration scenarios we analyzed. Some of the existing functional assessments could also be used to answer this question. The Miller and Golet (2001) functional assessment requires wetlands be basin wetlands to perform flood regulation functions (see Chapter 2 Wetland Flood Protection Functional Assessment Section for a more in depth comparison).

Is there evidence of demand for the service within the relevant provision area?

For people to benefit from flood regulation there must be structures, infrastructure, or other valued assets that are vulnerable to a flood hazard within the downstream flood-prone area. We evaluated this by determining, through GIS analysis, whether there are RI E911 addresses (RIGIS 2014) for buildings within the baseline scenario flood zone (Appendix H-1). The delineation of the flood zone will vary with the size and type of storm evaluated. Therefore, users of this approach in other locations may want to consider the most relevant storms to model for their purposes. In addition, we have chosen to evaluate only the existence of buildings, as indicated by E911 addresses, in the modeled area. The analysis could be expanded to also include other valued assets such as important roads or other infrastructure, or could be refined to only include those valued assets exposed to floods of a certain magnitude, such as floods of 1 foot or more.

For our two comparison subbasins, we found that both conditions are met (Table 3-1). Percent Flow Diverted and Wetlands Volume in both subbasins exceeded the threshold required to perform flood regulation services (Table 3-2). There were also structures in the 10-year flow baseline flood zone produced with the 1-year storm (Appendix H-1).

Table 3-1. Assessment of supply and demand

Indicator:	Criteria	Subbasin 1540	Subbasin 2370
Thresholds met?	Wetland area >0 m^2?	Yes	Yes
	Wetland volume >0 m^3?	Yes	Yes
Evidence of demand?	Structures in flood zone?	Yes	Yes

Table 3-2: Subbasin and restoration characteristics

	Subbasin 1540	Subbasin 2370
Total area	1.88 km^2	1.76 km^2
Original wetlands area	0.28 km^2	0.23 km^2
Restoration area	0.40 km^2	0.38 km^2
Percent diverted	36%	35%
Wetlands volume	272,702.8 m^3	244,336.2 m^3

II. Assess temporal reliability of the service

This step addresses the question: What is the probability that the site will continue to provide a service into the future? Our modeling did not estimate this probability. However, some of the information used to parameterize the models might be used, combined with judgments by the decision maker, to assess whether one site is likely to provide services more reliably over time. CN (Appendix D) and imperviousness (Appendix D) both indicate level of urbanization. A restoration site in an urbanized subbasin is likely more exposed to stressors and future development, meaning it has lower reliability. Therefore, either lower CN or lower imperviousness should indicate higher reliability. An additional indicator available for Rhode Island is Rhode Island Statewide Planning's projected development for 2025 (Appendix H-2). From the data on development projections for 2025, we used information on the percent of the subbasin area which is protected or otherwise expected to have limited development in 2025 (Figure 3-3). A higher value for % limited development thus indicates higher reliability.

In the case of our simulated restorations, we summarized all three reliability indicators for each subbasin. In cases where specific sites are being evaluated, a more detailed stressor assessment based on the local conditions in the surrounding area may be possible and warranted. While it is not practical to account for all possible future scenarios, modeling and expert judgment of potential threats to persistence of benefits can strongly influence site prioritization and therefore are important to consider. Based on the indicators % imperviousness, CN, and projected 2025 land area protected from development for the subbasins we conclude that Subbasin 2370 has a higher probability of reliably providing flood regulation services over time (Table 3-3).

Table 3-3. Assessment of reliability

Indicator	Subbasin 1540	Subbasin 2370
% Imperviousness	13.92 %	6.83 %
Curve number (CN)†	70.99	65.83
% Limited development 2025*	13.76	23.53

*Limited Development includes: "Conservation/Limited", "Major Parks & Open Space", "Reserve" and "Water Bodies", and a higher number indicates higher reliability.
†A lower Curve Number or percent imperviousness indicates higher reliability.

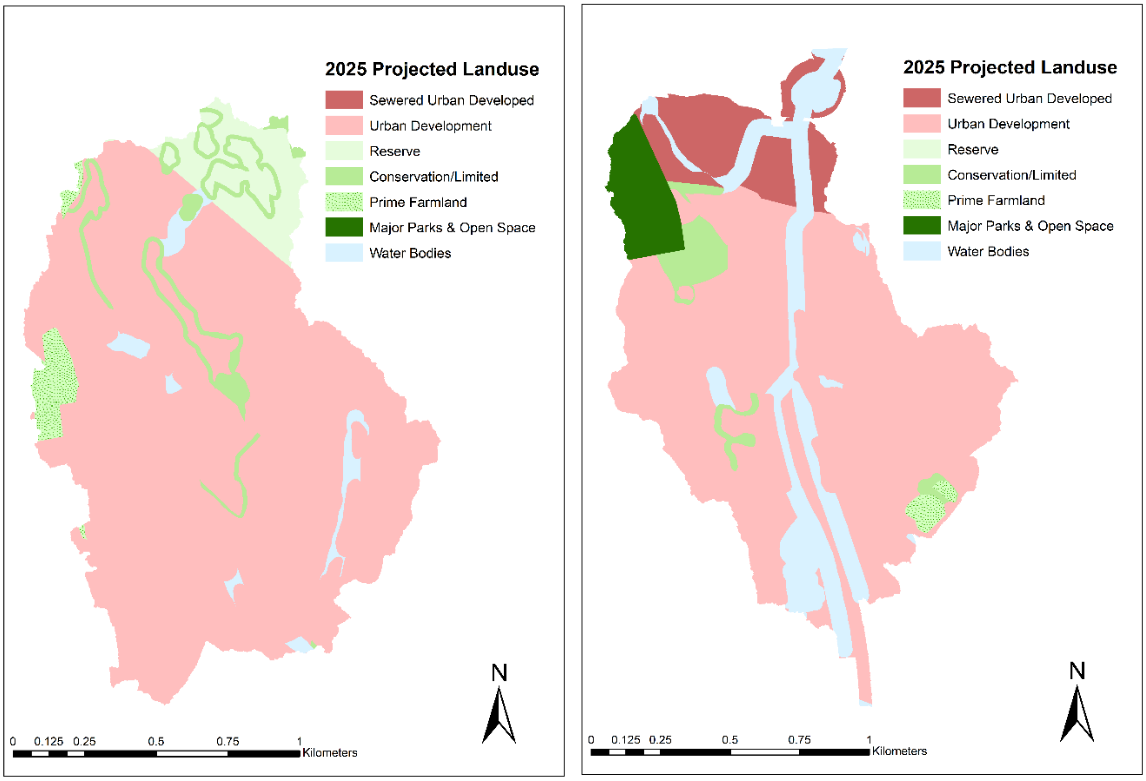

Figure 3-3. 2025 Projected Landuse for Subbasin 1540 (left) and Subbasin 2370 (right)

III. Assess who benefits

This step addresses the question: How many people benefit from the service? In our approach, we quantify this using the number of addresses in the E911 database that fall within the predicted flood zone, for the modeled floods. As discussed above, this analysis could be extended to include critical roads and other infrastructure. To determine the number of beneficiaries in each restoration scenario, we examined the addresses within the baseline scenario flood zone (Appendix H-1) and used that as a count of the number of buildings that were predicted by the model to experience a reduction of 0.04 ft or more in flood depth after the restoration. We chose a depth of 0.04 ft because of the vertical accuracy of the Digital Elevation Model (DEM) used to create and compare flood profiles. The DEM is provided with a 0.01 m vertical precision, and had a Root Mean Square Error of 0.067 m based on raw LiDAR calibration control points. Although our hydraulic model showed changes in flooding of less than 0.04 ft, we used this as a "detection threshold," since lesser changes could be attributed to error.

For our example, the restoration in Subbasin 1540 reduced flooding for one building, whereas the restoration in Subbasin 2370 protected five buildings (Table 3-4 and Figure 3-4). These numbers are low mainly because of the limited number of downstream buildings vulnerable to flooding during a 10-year peak flow flood. Based on these indicators, we concluded that Subbasin 2370 has a greater number of beneficiaries.

Table 3-4. Assessment of quality of service and beneficiaries' vulnerability

Subbasin 1540 Beneficiaries	Flood Depth Reduction	Substitutes	Social Vulnerability Index	DFIRM
Address #6	0.13 ft	0	Medium Low	Minimal
Subbasin 2370 Beneficiaries				
Address #1	0.19 ft	0	Medium	Outside DFRIM
Address #2	0.11 ft	0	Medium	Outside DFRIM
Address #3	0.11 ft	0	Medium	Outside DFRIM
Address #4	0.06 ft	0	Medium	Outside DFRIM
Address #5	0.06 ft	0	Medium	Outside DFRIM

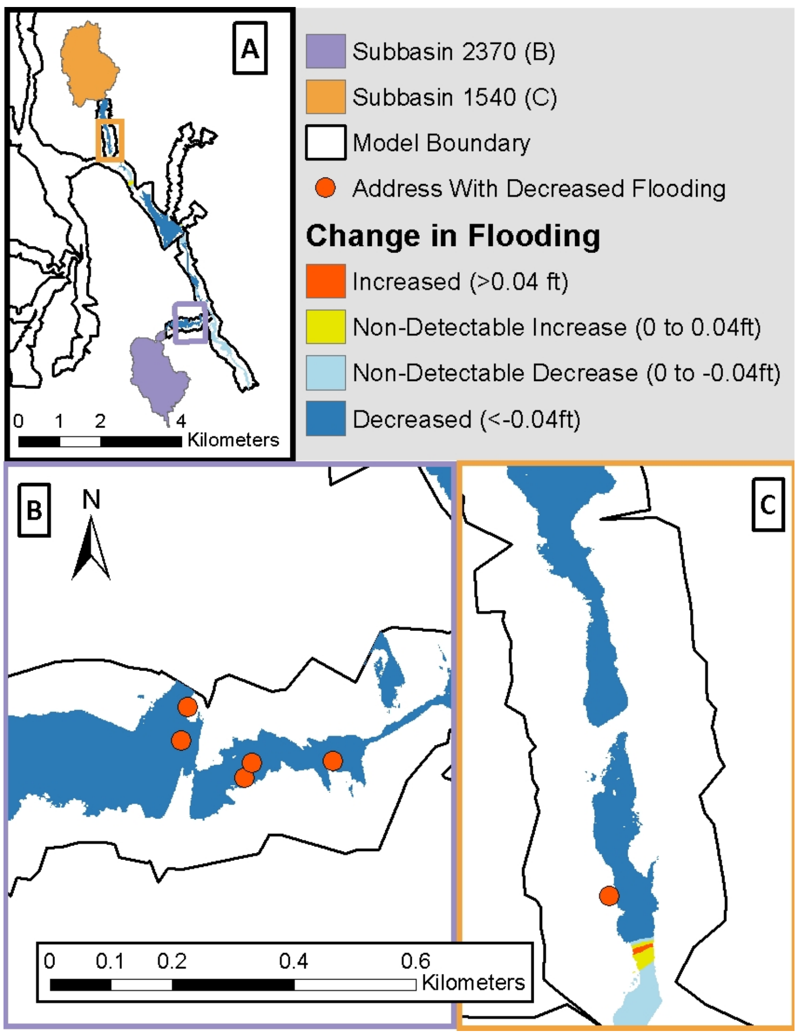

Figure 3-4. Building locations in the altered flood zone. Subbasin 2370 is shown in purple on map a, with its flood zone magnified in map b. Subbasin 1540 is shown in orange on map a, with its flood zone magnified in map c.

IV. Assess the magnitude of benefits to individuals or households

This step addresses the question: How much do people benefit? It may incorporate a number of measures, including the magnitude of change in the service relative to the baseline (i.e., the quality of the service), the availability and quality of substitutes, any necessary complementary inputs, and strength of preferences for the service.

We quantified the quality of flood regulation using an estimate of the reduction in flood depth experienced at the protected buildings. For evaluation of quality of flood reduction to infrastructure that is a two dimensional area, e.g., a road, park, or athletic field, the quality of benefits could be quantified by flood extent (area), or the peak volume of water flooding that area.

Other wetlands, dams or levees may provide substitute sources of flood regulation. However, the way in which these substitutes provide flood regulation services differ and those differences can determine the quality of services provided by substitutes. Assuming levee design conditions are intended to completely block the flow of water to assets they protect, upstream restoration will not benefit people protected by the levee unless water overtops the levee. If water overtops the levee, additional flood reduction benefits from upstream restoration could be provided. Similarly, if a dam detains water downstream of a restoration site, the restoration may not add to flood regulation benefits for beneficiaries downstream of that dam. Miller and Golet (2001) assumed this to be the case (Appendix A). However, the effect of a dam on an upstream restoration's benefits depends on the flood regulation benefits provided by the dam and the flood being considered. If there is flooding downstream of the dam and the upstream restoration would reduce water flowing over the dam, the benefits of the restoration may be decreased, but they are not completely substituted. In contrast to levees and dams, existing wetlands already provide flood reduction benefits downstream but do not prevent restored wetlands from providing additional benefits. Wetland flood reduction benefits are cumulative and, once the maximum flood reduction benefits are received (i.e., there is no flooding), it is hard to discern which wetland provided the benefits.

Our hydrologic models and hydraulic models include dams, levees and existing wetlands, so the effects of these in terms of substituting for restored wetlands is already incorporated in the predictions of flood extent. It would be possible to remove dams and levees from the models, to compare wetland benefits to benefits from substitutes. However, since the intent is to compare restoration across locations, there is little to be gained from doing this. People may also choose to adapt to flood risk by taking actions on their own property, for example by raising the elevation of a vulnerable structure. We did not consider such potential adaptation actions, but users of this approach could develop indicators of adaptation.

Strength of preferences and quality of complementary inputs are the last set of factors that influence benefits. Flood benefits do not require complementary inputs, and strength of preferences is difficult to quantify without directed data collection from beneficiaries. However, there are a few ways to characterize beneficiaries based on their ability to recover after sustaining flood damage. A beneficiary with fewer resources, such as income, flood insurance, or ability to find temporary shelter, would be expected to have a harder time recovering from flooding. One existing indicator for this is the Social Vulnerability Index, which synthesizes multiple census socioeconomic variables (Cutter et al. 2003). Another simple way to indicate resources for recovery is to overlay the beneficiaries with FEMA's DFIRMs to analyze whether the building is within the 100-year (1%) or 500-year (0.2%) flood zone, which is the basis for National Flood Insurance Program's (NFIP) regulations and flood insurance requirements (although homes without a mortgage are not required to have flood insurance, so one cannot assume all homes within the DFIRM are insured) (Appendix H-5).

We present the results of our example analyzed in terms of marginal benefits, defined as the decrease in flood depth, quantified for each beneficiary from Figure 3-4 (Table 3-4). Although substitutes are already incorporated into the model, we counted any possible substitutes between beneficiaries and the restoration subbasins (Appendix H-4; Table 3-4). We indicate preferences based on the Social Vulnerability Index (Figure 3-5; Appendix H-3) and the DFIRM (Figure 3-6; Appendix H-5).

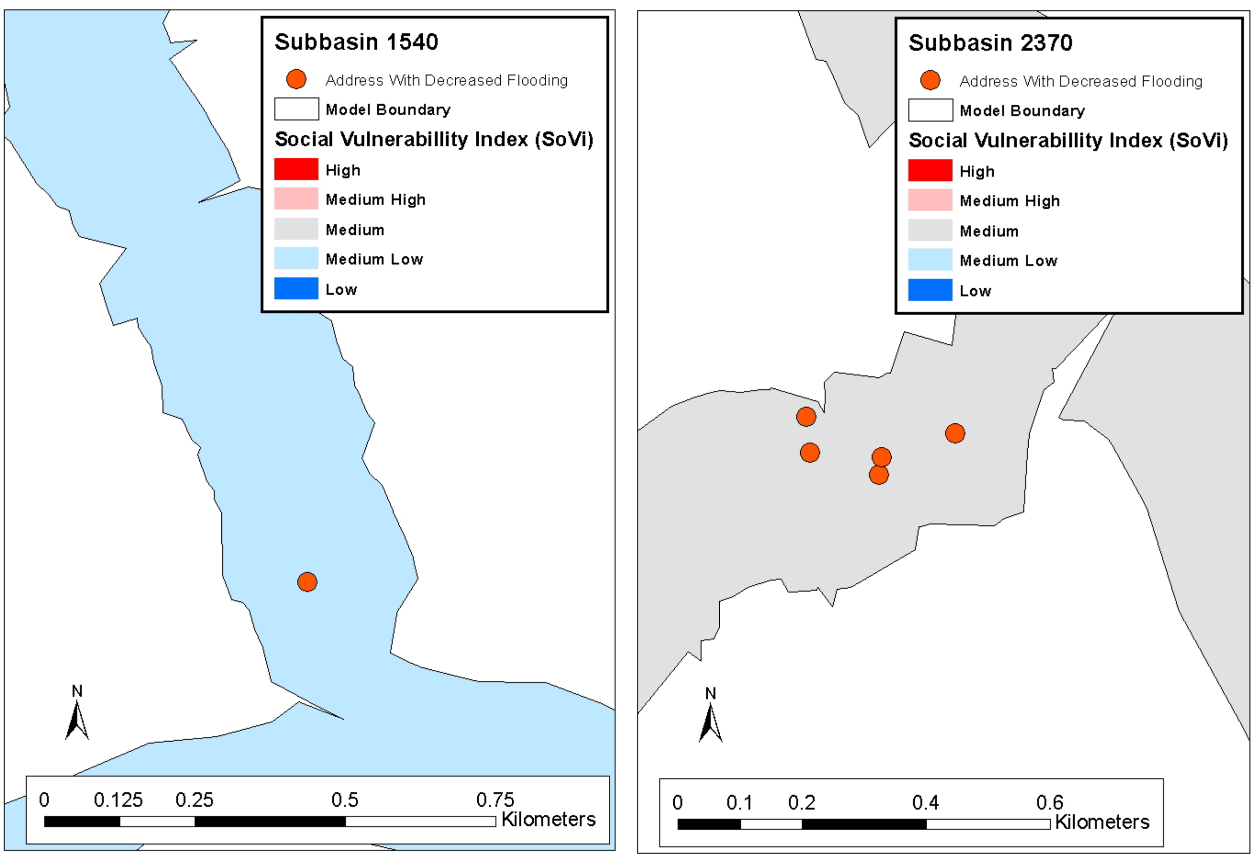

Figure 3-5. Census tract Social Vulnerability Index for Subbasin 1540 (left) and Subbasin 2370 (right)

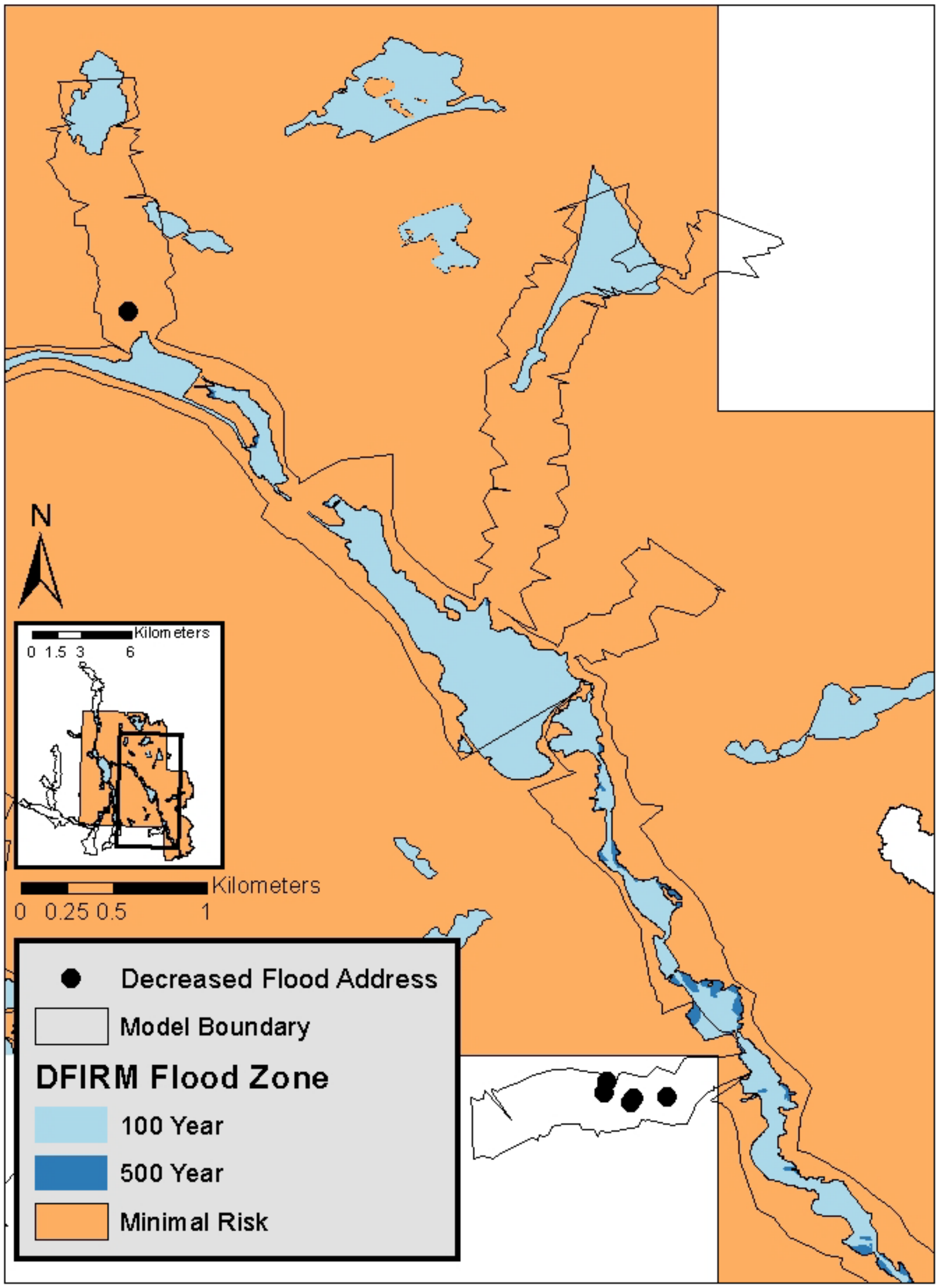

Figure 3-6. Buildings where flooding is decreased by restorations in Subbasin 2370 and 1540 (black points) compared to DFIRM flood zones (100-year and 500-year) in the FEMA modeled area (orange) and outside the FEMA modeled area (white). Our model boundary (black outline) is provided as a reference for building locations.

Based on these indicators, we concluded that Subbasin 2370 provides greater benefits to one of its beneficiaries than Subbasin 1540 provides to its sole beneficiary. None of the beneficiaries of either subbasin had dams (i.e., potential substitutes) between the restoration and their location. The beneficiaries of wetland restoration in Subbasin 2370 had a higher vulnerability index than those of restoration in Subbasin 1540. None of the buildings fall within the DFIRM flood zones, so would not be required to have flood insurance, despite being vulnerable to flooding based on our model, which included smaller streams and smaller floods.

Tier II Indicators

Tier III indicators provide a rigorous, quantitative, and defensible evaluation of the benefits of potential restorations. However, the need for extensive watershed-specific modeling makes this type of assessment inaccessible to many decision makers. Although Tier II benefit indicators are expected to be less rigorous than the Tier III indicators, they should be able to inform decisions without intensive modeling, based instead on existing datasets. While Tier II indicators are more easily applied, in many cases they still require some analysis, often requiring knowledge of Geographic Information Systems (GIS).

Like the Tier III indicators, the Tier II benefit indicators address the four questions of the indicators framework, although they do not answer all of the questions at the same level of detail, and some of the optional sub-questions are not addressed. Rather than demonstrating a simulated example application as we did for Tier III, for the Tier II indicators we show how we generalized model results and translated them into indicators that can be used by others. The next sections describe this process.

Generalizing from Tier III Model Results

The primary consideration for generalizing to Tier II indicators is the need to determine the relevant benefits area for flood regulation services. Flood regulation occurs at the site of a wetland, but benefits people and structures downstream of the wetland site. Therefore, our primary aim in generalizing from our detailed modeling was to determine the “benefits area” and the factors that may influence how this varies.

Methods

Using the hydrologic and hydraulic models, we generated hydrologic and floodplain change maps for 1,668 subbasin scenarios (139 subbasins*4 restoration scenarios*3 storm events) (Table 2-5), plus a baseline, or no restoration scenario, for each synthetic storm (3 basin scenarios). There were several options for how to evaluate this extensive data set to assess where benefits are provided.

Option 1 was to perform the same comparison as Tier III for all wetland restoration simulations: comparing the change in flood depth or flood extent from the baseline at all protected building locations for all the basin scenarios. However, the actual location of the buildings or assets at risk is unique to and characteristic of the development patterns and physical geography of the study area, a limitation when developing indicators to be used beyond the study area.

Option 2 was to compare changes in flooded area from the baseline for all restoration scenarios without consideration of where assets are located. Although this option would assess the effect of restorations on the entire floodplain, there were drawbacks to this option as well. The concern was that stream and floodplain morphology play an important role in determining the extent of flooding. Two simulated wetland restorations may result in an equal reduction of flows or flood-water volume, but will have very different effects on the extent of flooding based on the downstream morphology. A narrow-deep channel will show little change in flood extent but a greater change in flood depth in comparison to a wide-shallow channel. This can be seen in the Tier III indicator comparison maps (Figure 3-4).

Option 3 was to simply examine flood-water volume directly, as percent change from baseline peak flows from the hydrologic model. These peak flows were available for all subbasin simulations at all element locations along the stream network: basins, junctions, reaches, reservoirs and the outflow point. Because the model is spatially lumped at the subbasin, the location of a simulated wetland within the subbasins is unspecified, so the distance from the simulated wetland to the outlet of the subbasin was assigned the longest flow path distance.

Results

Figure 3-7 shows plots of the change in flow from the baseline against the distance downstream from the simulated restoration for each subbasin scenario, showing the distance decay in the reductions in flow from the restorations (Figure 3-7; the same figure is provided with the two subbasins from the Tier III application highlighted in Appendix I). Closer to the restoration, most scenarios show the full change in flow from the restoration (-1%, -5%, -10%, or -25%), with a more or less rapid decline in the percent change in flow with distance from the outlet of the scenario subbasin (note that the y-axis scale changes with increased restoration size).

In order to understand how far downstream these changes in wetlands influence downstream flow, we examined the changes in flow to estimate how far downstream the change in flow became effectively zero. In many instances the percent change fluctuates and approaches but never actually reaches zero. This suggests sustained low change in flow could be due to rounding errors and not due to significant changes in flow as a result of the upstream restoration.

We first defined a change in flow as effectively zero using a conservative cutoff—any change less than -0.2%—by estimating where the subbasin scenario curves began to flatten. This allowed us to estimate the distance downstream to the -0.2% threshold. We summarized these results using box plots (Figure 3-8) by storm size (Figure 3-8a), by wetland change (Figure 3-8b), and for all scenarios (Figure 3-8c).

Across storms (Figure 3-8a), the 1-year storm has the greatest variability and reaches no change in flow slightly farther downstream, indicating that the simulated wetland restoration scenarios had the strongest influence during small events. This makes sense given that each wetland scenario holds a fixed volume of flood-water and that volume is a smaller proportion of a larger event. Across wetland change scenarios (Figure 3-8b), the mean and variability in the distance downstream to no change in flow increases as wetland scenario size increases indicating the larger change in flow associated with a larger increase in wetland extent results in changes in flow that are sustained farther downstream. Looking across both changes in wetland extent

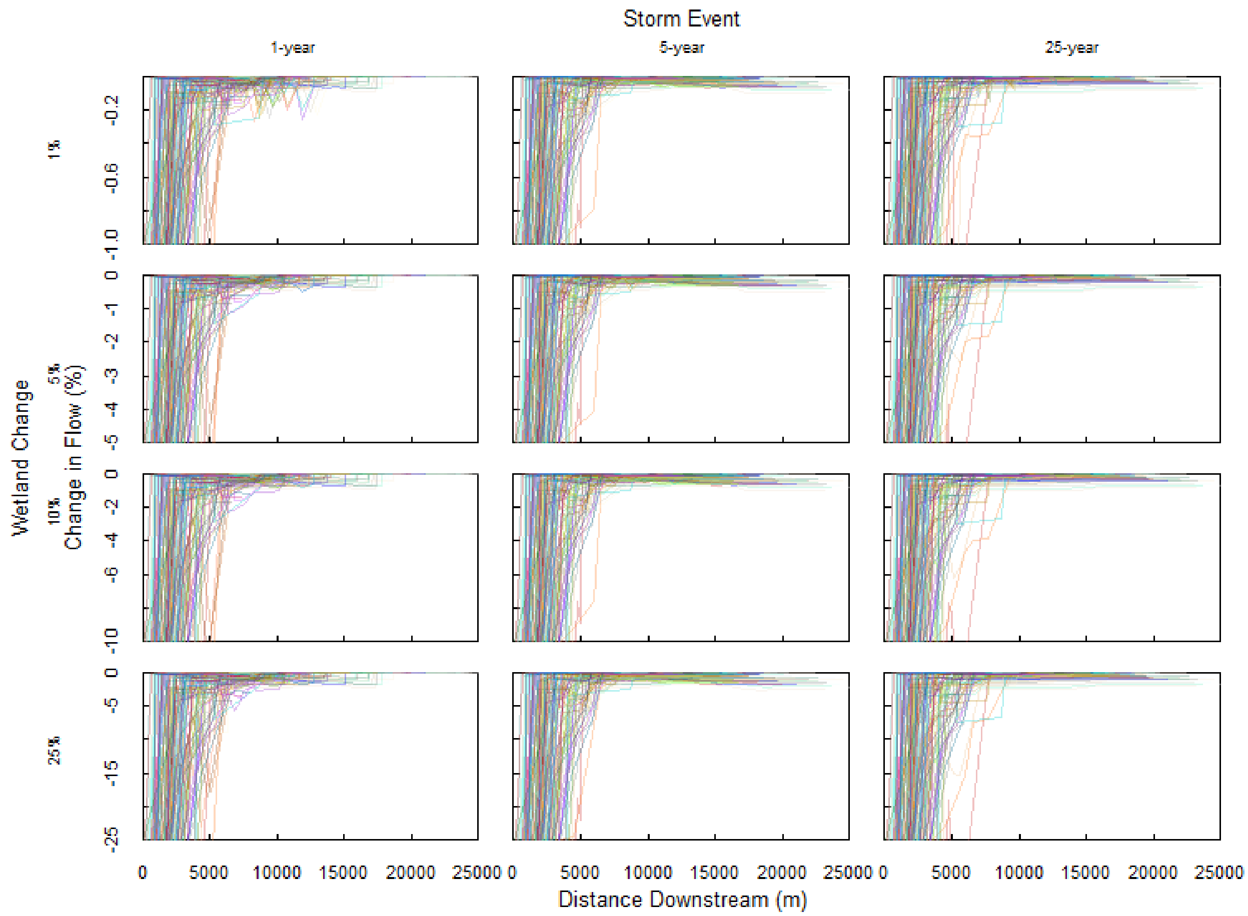

Figure 3-7. The change in flow (%) with distance downstream (m) from the simulated restoration for all subbasins (n=139) for each restoration scenario (wetland change scenarios 1%, 5%, 10%, and 25%) and synthetic storm event (1-year, 5-year, and 25-year). Note that the y-axis scale varies between sub-plots for each of the wetland change scenarios.

and storm sizes (Figure 3-8c; Table 3-5), the longest downstream influences (mean of 7359.3 m) were seen with the largest restoration scenarios (25%) in the smallest storm event (1-year), again, because the largest wetland scenarios retain the largest volumes of flow which makes up a larger proportion of a smaller event. The shortest downstream influences (mean of 3260.1 m) were seen with the smallest restoration scenarios (1%) in the largest storm event (25-year).

The mean distance downstream where change in flow dropped below -0.2% for wetland change scenarios, was 4311.7 m (2.7 miles), with individual simulations that ranged from 506.9 m (.31 mi) to 19464.2 m (12.1 mi) (Table 3-5).

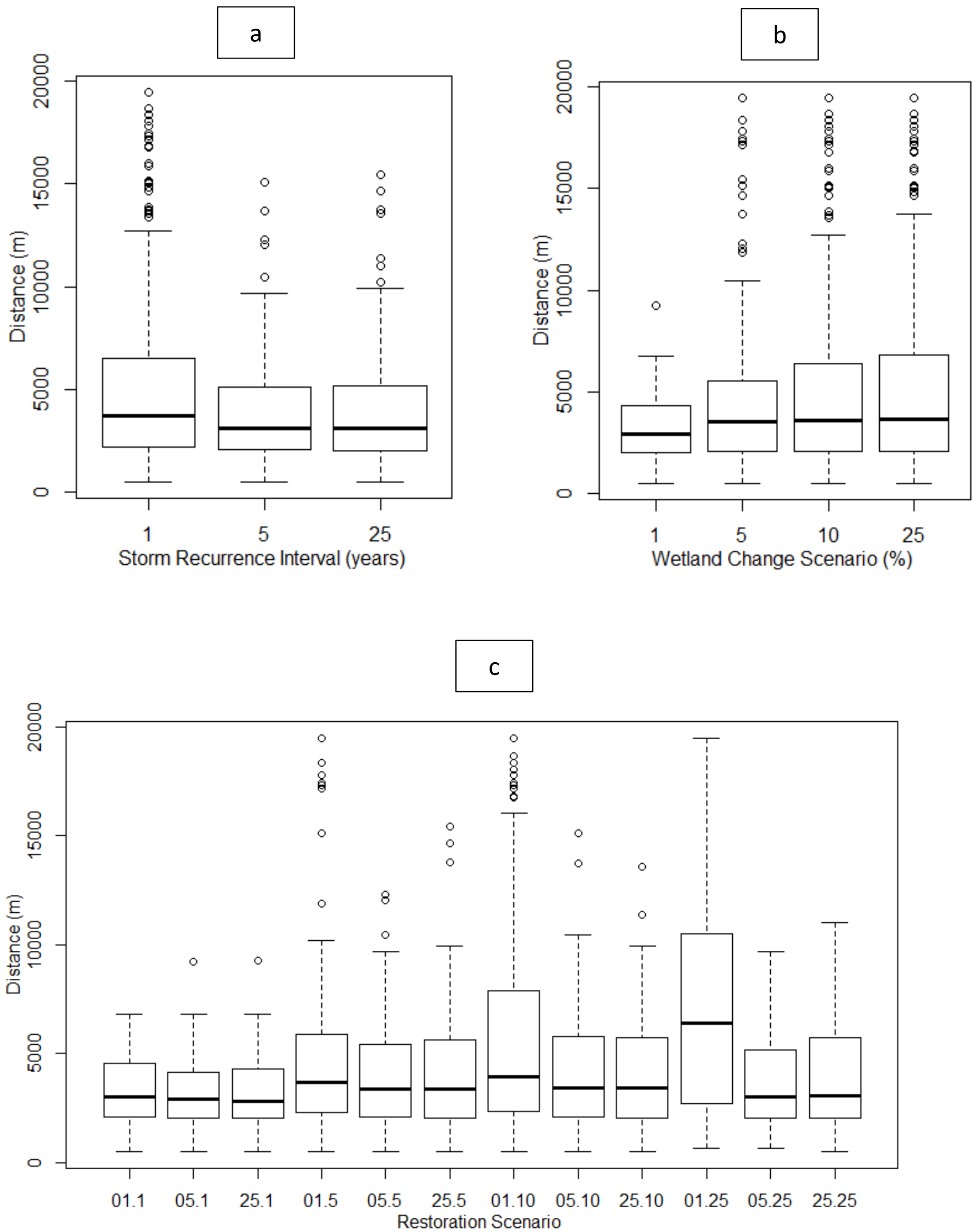

Figure 3-8. Boxplots of downstream distance (m) to no change in flow (<-0.2% change) by restoration scenarios: (a) storm recurrence interval, (b) wetland change, and (c) both wetland and storm recurrence interval.

Table 3-5. Downstream distance (m) where change in flow became negligible (-0.2%).

Wetland Restoration		Storm Event 1 Year	5 Year	25 Year
1%	Mean	3302.5	3255.8	3260.1
	Max	6779.7	9234.6	9252.9
	Min	506.9	506.9	506.9
	SD	1632.6	1658.1	1701.7
5%	Mean	4696.3	3949.0	4137.5
	Max	19464.2	12287.9	15411.5
	Min	506.9	506.9	506.9
	SD	3842.7	2438.6	2814.9
10%	Mean	6046.6	4093.4	4051.1
	Max	19464.2	15099.1	13547.7
	Min	506.9	506.9	506.9
	SD	5110.4	2728.8	2593.4
25%	Mean	7359.3	3629.5	3959.9
	Max	19464.2	9662.3	10997.6
	Min	627.5	627.5	506.9
	SD	5312.4	2182.3	2605.8

Next, we increased the threshold to a -1% change in flow as the cutoff to be considered effectively no change in flow. We selected this new threshold based on inspection of flood maps where changes in flow less than 1% frequently did not result in changes in flood depth greater than the detection criterion of 0.04 ft. Figure 3-9 shows the box plots for this analysis.

Examination of results across storms (Figure 3-9a) showed that the mean and variability were again greatest for the 1-year storm. Examination of results across wetland change scenarios (Figure 3-9b) identified an issue with the 1% restoration scenarios when a 1% threshold is used for defining change in flow as effectively no change in flow. Because the maximum change in flow for the 1% restoration scenarios is already 1%, the threshold will be met as soon as subbasin discharge joins other downstream flows. This results in distances to no change for the 1% scenarios equaling the distance of flow within the subbasin itself or the subbasin's longest flow path distance. Excluding the 1% restoration scenario, the updated threshold showed the same trend in variability and mean distance as with the smaller threshold, with mean and variability in downstream influence increasing with restoration size.

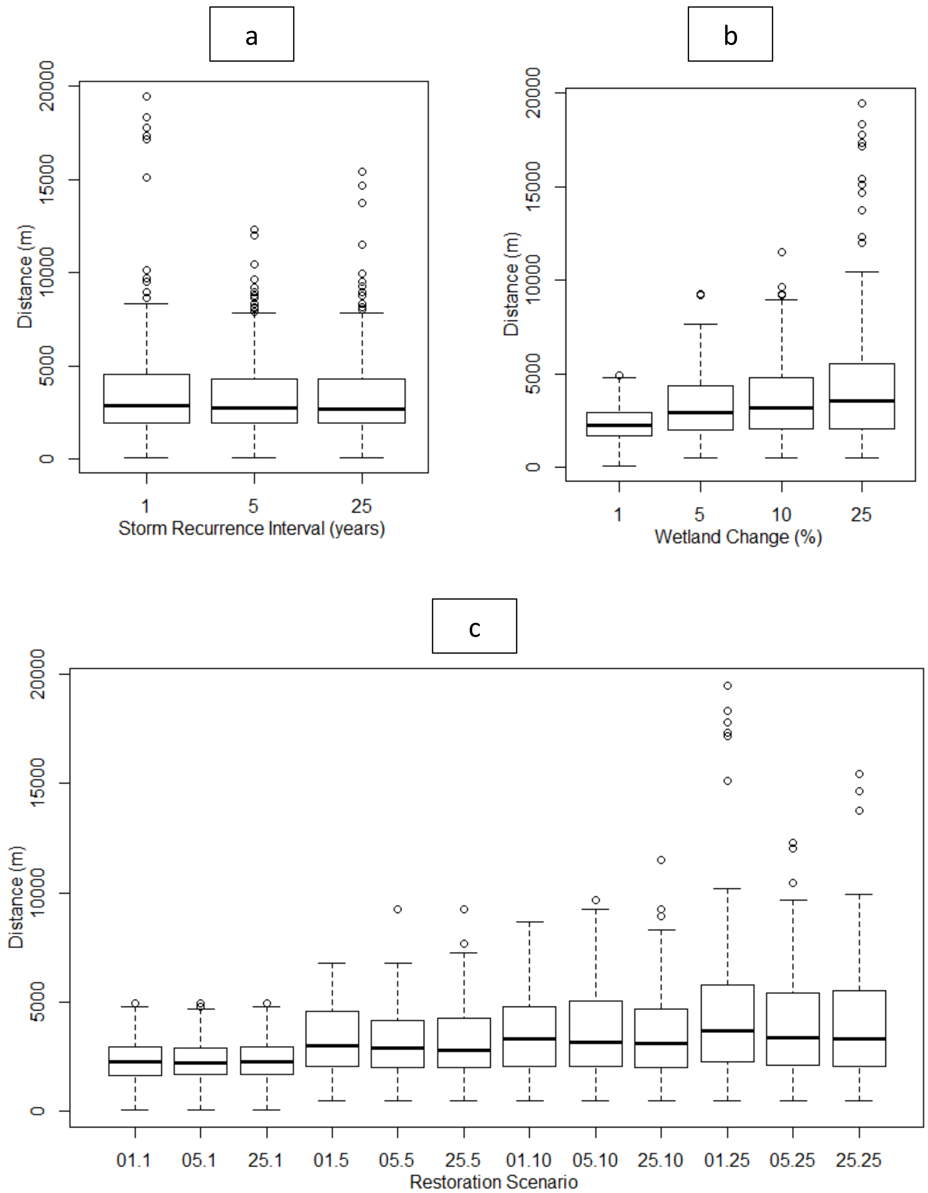

Figure 3-9. Boxplots of downstream distance (m) to no change in flow (<1% change) by restoration scenarios: (a) storm recurrence interval, (b) wetland change, and (c) both wetland and storm recurrence interval.

The mean distance downstream where change in flow dropped below -1% for wetland change scenarios above, excluding the 1% restoration, was 3688.7 m (2.3 mi). Mean distances for scenarios ranged from 3255.8 m (5% restoration and 5-year storm) to 4545.7 m (25% restoration and 1-year storm). Individual simulations ranged from 506.9 m (.31 mi) to 19464.2 m (12.1 mi) (Table 3-6).

Table 3-6. Downstream distance (m) where change in flow became negligible (-1%).

Wetland Restoration		Storm Event 1 Year	5 Year	25 Year
5%	Mean	3302.5	3255.8	3299.4
	Max	6779.7	9234.6	9252.9
	Min	506.9	506.9	506.9
	SD	1632.6	1658.1	1766.5
10%	Mean	3560.2	3634.2	3556.1
	Max	8649.3	9662.3	11512.2
	Min	506.9	506.9	506.9
	SD	1804.1	2023.1	2029.8
25%	Mean	4545.7	3943.1	4101.8
	Max	19464.2	12287.9	15411.5
	Min	506.9	506.9	506.9
	SD	3619.0	2438.0	2809.2

In general, for both cutoff points, the distance to zero change in flow stayed roughly the same across storm recurrence intervals, but went up with the percent in wetland change. And as described above, when combined, the longest distances were seen with the largest change in wetland area and the smallest storm events.

Next, we examined how results vary with different restoration basin characteristics to see if variability in downstream distance to the same thresholds (-0.2 and -1%) followed any general trends (Figure 3-10; Appendix I-2). These restoration basin characteristics included: the distance to the gauged basin outflow, restoration basin area, initial percent wetlands, basin slope, basin percent imperviousness, and the longest flowpath distance.

Scatter plot results based on the <-0.2 and <-1% thresholds were very similar, so we present the -0.2% threshold here. Because some characteristics appeared to show slight trends, we generated box plots based on splits at the mean value for each characteristic (Figure 3-11 for <-0.2%; Appendix I-3 for <-1% threshold).

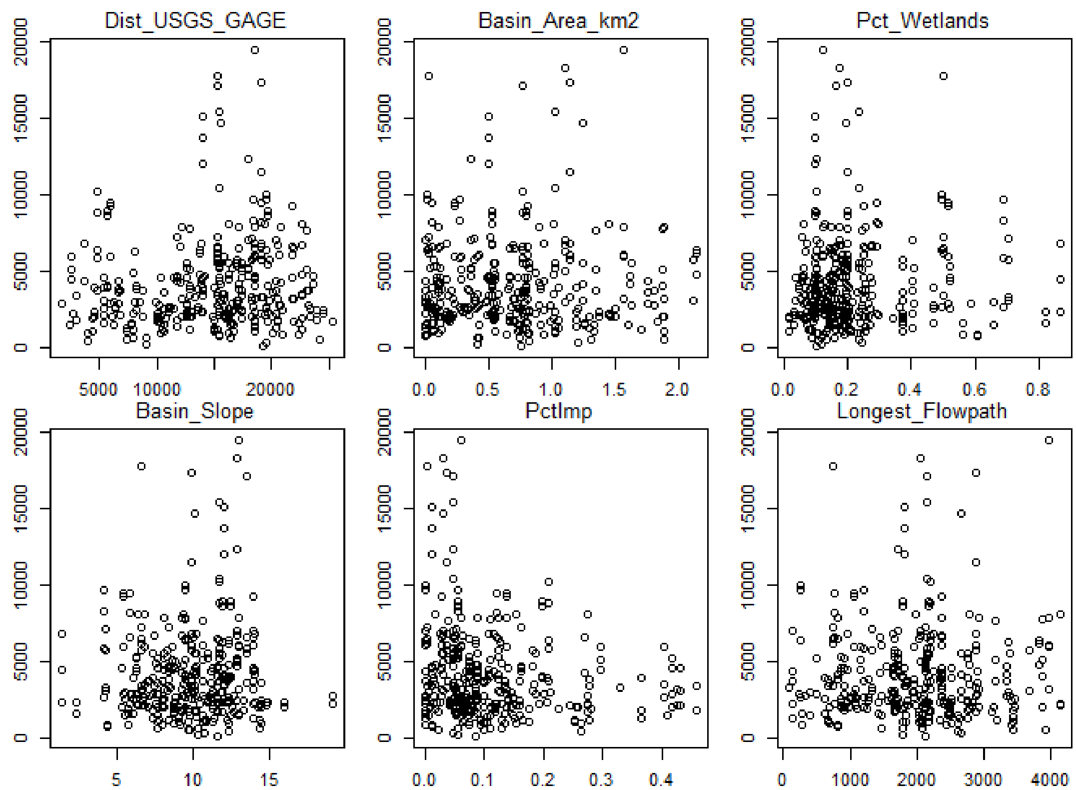

Figure 3-10. Downstream distance to no change in flow (<-0.2% change) plotted against subbasin characteristics: distance (m) to the outlet (Dist_USGS_GAGE), subbasin area (Basin_Area_km^2), % wetlands (Pct_Wetlands), mean basin slope (Basin_Slope), % impervious cover (PctImp), and the longest flowpath distance (m) (Longest_Flowpath).

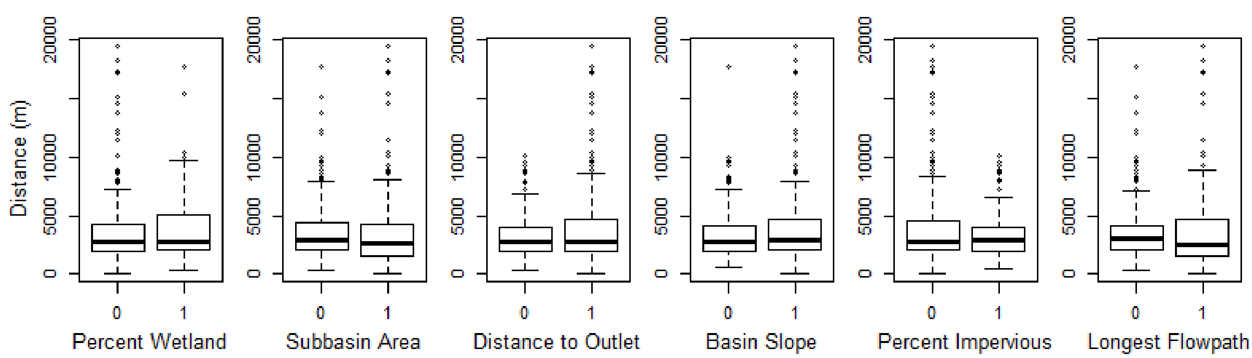

Figure 3-11. Boxplots of downstream distance to no change in flow (<-0.2% change) plotted by subbasin characteristics (distance to the outlet, subbasin area, % wetlands, mean basin slope, % impervious cover, and longest flowpath) split into below (0) and above (1) the mean values for the basin characteristics (Table 3-7).

Though there were slight differences between the mean distance to zero change in flow above and below the means for the subbasin characteristics, for the purposes of informing Level II indicators these differences were very small (Table 3-7). The assumption that all wetland restorations were the same distance from the subbasin outflow as the longest flow path for that subbasin could have biased downstream distances, increasing the minimum distance to the longest flowpath distance. However, the lack of correlation between longest flowpath distance and mean distance to no change in flow (0.2%) suggests distance is not biased by the longest flowpath assumption. Also, none of the means across subbasin characteristics varied significantly, so this supports the selection of a threshold for downstream influence based on the average downstream distances before changes in flow reach the no change threshold (-0.2 or -1%). For the -0.2% threshold the mean distance was 4311.7 m and for the -1% threshold this distance was 3688.7 m when the 1% restoration scenario was excluded. The average of the two thresholds is 4000.2 m or 2.50 miles. There was considerable variation about the mean for all subbasin characteristics, which indicates that there may be some geographic settings in which downstream flow reduction extends over a larger area due to variations in these characteristics. In an area where a particular characteristic has a strong influence, restoration might optimize for that characteristic.

Interpretation of Results for Indicators

Based on our analysis, a reasonable distance for delineating the area where people could benefit is 4 km (2.5 miles) downstream of the restoration, based on the mean distances to no change in flow (both 0.2% and 1% thresholds) across restoration scenarios and storms. In some cases there may be a larger distance where people could benefit—the mean distance to no change in flow for the largest restoration (25%) and the smallest storm event (1-year, 10-year flow) was higher, 7.4 km (4.6 miles). But based on our analysis, the 4 km (2.5 mile) distance indicator was robust across most scenarios.

In our example results using the Tier III indicators, we found that the beneficiaries identified were within 4 km (2.5 miles) of the subbasins with restoration (Appendix H-7). This is consistent with our mean estimate for the Tier II indicators. Based on our modeling we found that, at least for the watershed and conditions modeled, the downstream benefits area may not extend as far as 5 miles (8 km), as assumed by Miller and Golet (2001).

Applying Tier II Indicators

I. Assess existence of an ecosystem service

In keeping with our goal of providing a relatively rapid assessment we have applied this step using simple yes/no criteria. The indicators for service production are the same as those used for the Tier III assessment. The user can either follow the GIS based process for calculating wetland area and volume (see Chapter 2, Treatment of Wetlands in the Models), or can use a functional assessment like Miller and Golet (2001) to define the thresholds for service production. Flood reduction services do not require any complementary inputs so that, as long as flood reduction service thresholds are met, a flood reduction service is produced.

The quick check for demand is that there is evidence of demand for the service within the relevant provision area being considered. This requires an initial determination of the relevant benefits area. In the Miller and Golet (2001) functional assessment (Appendix A), the social significance indicator determines beneficiaries based on their downstream distance from the restored wetland: "D*eveloped flood-prone areas downstream within 5 miles or to the nearest dam (connection by stream or floodway required)."* Through our modeling, presented above, we found a 2.5 mile (4 km) downstream area to be the relevant area.

This initial question does not address how many people benefit but simply determines whether there are potentially people who benefit. Therefore, it would simply answer the question: Are there valued assets within the benefits area?

Tier III used the baseline map of modeled flooding as evidence of demand, meaning vulnerable assets need to be identified some other way for Tier II. If available, FEMA DFIRMs (Appendix H-5) can be used to represent the area expected to experience flooding. If a valued asset is inside the 100-year or 500-year flood zone in the DFIRM map it is potentially vulnerable, meaning there is potential demand for flood reduction services. In the Woonasquatucket watershed, the DFIRM does not extend into the upper watershed, making it difficult to use as an indicator if a proposed wetland restoration is in that part of the watershed. Also, in the Tier III assessment we found that some locations outside the bounds of the DFIRM may experience flooding.

There are several alternatives for defining and identifying valued assets. Some options include using address points to locate protected buildings, as was done for Tier III (Appendix H-1), using town parcel data, using other critical infrastructure including roads or emergency response assets, using population data, or using imagery like that available through Google Earth to visually search for assets exposed in the DFIRMs (Appendix H-6). An alternative to using DFIRMs would be to confirm that flooding occurs in the area being studied through other non-spatial datasets, such as repetitive loss areas (available through FEMA as part of the Community Rating System; FEMA 2014), other insurance claims data, or interviewing emergency response personnel or others familiar with flooding in the area.

II. Assess temporal reliability of services

The Tier III indicator assessment of temporal reliability of services did not rely on any model outputs and can be applied in the same way for Tier II.

III. Assess who benefits

This step addresses the question: How many people benefit from the service? This step would further refine the yes/no assessment of demand to estimate the number of people who benefit. The Tier III indicators quantified how many people benefit from the service using the number of addresses to count protected buildings in the modeled flood zone. For Tier II indicators, people can be quantified the same way, using number of addresses, but determining the relevant flood zone requires further analysis. We list several methods for doing this under question I above.

IV. Assess the magnitude of benefits to individuals or households

This step addresses the question: How much do people benefit? It may incorporate a number of measures, including the magnitude of change in the service relative to the baseline (i.e., the quality of the service), the availability and quality of substitutes, any necessary complementary inputs, and strength of preferences for the service. Since each measure included increases the complexity of the assessment, it will be difficult to find Tier II indicators that assess the magnitude of benefits, and the user may have to rely on the number of beneficiaries combined with best professional judgment of the factors listed here that indicate magnitude (as in our Tier I indicator approach).

Our modeling results provide some useful information that can assist in applying professional judgment to this question. In terms of the magnitude of change in the flood regulation service relative to the baseline, we did not find strong evidence for specific factors that lead to variations in the benefits decay function for downstream distance. However, the mean and range for the maximum downstream distance where benefits are delivered suggests that flood regulation services decrease quickly beyond a certain distance. One potential way to approach this with Tier II indicators would be to qualitatively rank beneficiaries that fall within multiple downstream distance buffers. For example, beneficiaries 0 km to 4 km “likely receive benefits” whereas beneficiaries 4 km to 7.4 km “may receive low benefits.”

Tier II indicators for substitutes could be analyzed in the same way they were for Tier III, quantifying the number of dams and levees within the downstream buffer being considered. Although Miller and Golet (2001) identified downstream dams as playing a strong role in eliminating downstream flood regulation benefits from wetlands, we were not able to observe dams eliminating wetland flood regulation benefits in this way. Percent wetlands already in the subbasin being restored did not show a strong negative influence on downstream benefits of additional restored wetlands as might have been expected.

The Tier II benefit indicators successfully address the four question in the indicators although they do not answer all of the questions at the same level of detail as Tier III indicators. While Tier II indicators are more easily applied, in most cases they still require some analysis, often requiring knowledge of Geographic Information Systems (GIS). We generalized model results and translated them into indicators for the downstream distance within which flood regulation benefits are able to be received. Instead of a quantitative decay function for the quantity of benefits delivered, a qualitative downstream boundary can be used where beneficiaries 0 km to 4 km “likely receive benefits” whereas beneficiaries 4 km to 7.4 km “may receive low benefits.” The observed trends in this downstream boundary based on storm and restoration size also provide useful information when adapting such boundaries to a more specific decision context.

CHAPTER 4: DISCUSSION AND NEXT STEPS

In this report, we have presented benefit indicators for the flood regulation service provided by wetlands, and an approach for developing those indicators. The indicators presented in Chapter 3 follow a framework that is grounded in economic theory (Mazzotta and Wainger in preparation) and a quantitatively defensible modeling process (outlined in Chapter 2). While the indicators themselves are useful for others evaluating flood regulation benefits, equally important is the general approach to developing benefit indicators that we present, and its compatibility with typical functional assessment tools and potential usefulness for improving benefit transfer.

In this chapter, we summarize some caveats and present important considerations regarding the broader applicability of our specific Tier II and Tier III indicators. Applicability beyond our case study for the Woonasquatucket Watershed hinges on the applicability of the models used to develop the indicators, as well as data availability for and characteristics of other watersheds. For the Tier III indicators, we chose our models with future applications in mind, and we provide suggestions for improving such applications to other geographic locations. While some of the Tier II indicators developed are broadly applicable across locations, not all are applicable to all decision contexts or all watersheds. We summarize limitations on Tier II indicator applicability and what might be required to expand this applicability.

Modeling Summary

Although we chose the models we considered to be most appropriate, based on the selection criteria detailed in Chapter 2, some aspects of modeling might be improved with additional data and effort, or enhanced models. It is important to note that the primary reason for developing benefit indicators is to inform decisions by facilitating comparisons across sites, and not to provide the most precise predictions of flood impacts. Thus, we balanced precision with the ability to develop useful benefit indicators that allow for comparison across wetland restoration scenarios.

We point out some of the model complications and caveats here, to inform future modeling efforts of this kind. These include:

- Past studies used HEC-HMS to effectively show the role of wetlands in the watershed, but if trying to develop indicators for individual wetlands (rather than by subbasin) a spatially distributed model has some clear advantages. HEC-HMS is a spatially lumped model, and trying to make it function in a more distributed way by using more subbasins increased computational requirements and strained time interval limitations, making machine optimization arduous. A more distributed model might reduce optimization time, at the expense of increased model complexity and data requirements. A spatially distributed model would better account for where wetlands are located within the subbasin and how much water is available to them. A spatially distributed model may not be the only solution; it may also be possible to model wetland catchments and then parameterize that in HEC-HMS. Such a method still would not necessarily account for water available to wetlands from adjacent streams however.

- With additional modeling it might be possible to improve upon the assumptions inherent to how wetlands were integrated into these models, for example the size of their catchment area or the effect of restorations on infiltration parameters. Our assumptions about the runoff available to wetlands were purposely conservative. Sensitivity analysis suggests that including even a 100m buffer catchment around wetlands would significantly increase their impact on downstream flows. Better accounting for the actual water available to wetlands would be a major improvement.
- Although model fit improved by using a spring storm, which is the time when flooding is most likely, the variability in antecedent conditions for reservoir release and abstraction made it very difficult to generate a single model that generalizes all the potentially relevant conditions.
- Calibrating two models based on distinctly different storms showed how greatly antecedent conditions can impact flood model results. A model that could better account for these, and better data quality could improve model results.
- Sensitivity analysis of precipitation in the models showed small increases in total precipitation cause large increases in flow relative to total flow, suggesting precipitation data quality is very important to model accuracy. HEC-HMS is able to model gridded precipitation data and this may be worth investigating in future modeling efforts, especially when stationary precipitation gages may not be representative of rainfall in the watershed
- In a watershed where more data were available on characteristics of dams and their actual management, it would be possible to better model the actual role dams play as substitutes for wetland flood reduction benefits.
- Baseflow was accounted for outside of the model and then removed from flow. Although reservoirs were parameterized with an initial discharge, a model better equipped for baseflow from small diffuse reservoirs might perform better. Better data on these reservoirs, their actual storage-discharge relationships, and how they are managed, such as drawdown before storms, could also improve models for watersheds like the Woonasquatucket where there are numerous small dams.
- HEC-RAS performed well for hydraulic modeling but additional information, such as better characterization of dams and more information about infrastructure in the headwaters in general, might improve the flood model and better represent the role of wetlands in flood retention. We used steady flow analysis, but non-steady flow analysis may be able to better capture storage in the watershed. This becomes more important in

a watershed with many diffuse reservoirs and could also help account for storage in buffer wetlands.

Tier III Indicators

The Tier III application was based on model results and, although the modeling process is transferable, the modeled results are specific to the modeled area within the Woonasquatucket watershed. Where indicators did not require model results and instead used more widely available datasets such as the Social Vulnerability Index, the indicators presented are applicable to any location where the required data are available (NOAA 2015). Further, the specific models used may not transfer well to another setting, but the process of summarizing model results to inform our indicators framework is transferable.

The Tier III indicator development process also showed how the models could quantitatively relate functional assessments, such as the Miller and Golet (2001) assessment, to benefit indicators. Our Tier III indicators might also replace or augment similar indicators in other ecosystem service models. For example, ARIES uses FEMA 100- and 500-year flood zones. Where developed, the modeled flood area from our Tier III modeling process could be used in place of these FEMA flood maps. This would allow for a better quantification of the magnitude of flooding that potential beneficiaries are projected to experience. Our overall indicator framework can be applied to link the benefits indicators to values, and may allow for synthesis with valuation methods. For example, the SolVES tool (Sherrouse and Semmens 2010) can be used to transfer social-value models to physically and socially similar areas, and benefits indicators such as those presented here could be one way to evaluate similar areas.

Tier II Indicators

Tier II indicators are designed to be more transferable than the Tier III indicators. Many of the Tier II indicators were developed from existing datasets, but required some geospatial analysis to characterize specific wetland restorations. In addition to these existing datasets, we used our models to simulate restorations that could be generalized to a downstream distance for transfer of flood regulation benefits. These are more transferable to watersheds with similar hydrological and land use characteristics.

Though the trends in our estimated downstream distance for benefits are somewhat uncertain, they give a valid range to consider in identifying potential beneficiaries. This range of downstream distance also provides a quantitative model-based test of assessments that are based on expert judgment, such as the 5 miles (8 km) suggested in Miller and Golet (2001). We were unable to generalize the rate that benefits decay as they are transported downstream, meaning the change in the level of benefits received with greater distance from a restoration site cannot be quantified directly from our results, but will necessarily involve some expert judgment regarding how benefits may decline with distance.

We were also unable to unambiguously quantify the role of dams in decreasing benefits from wetlands. Although dams clearly play a role in the Woonasquatucket watershed, based on the data available and how dams and reservoirs were modeled, the presence of a dam downstream from a wetland does not necessarily prevent the flood regulation benefits of the wetland from reaching beneficiaries downstream of the dam. This may be explained by the fact that many of the dams in the Woonasquatucket watershed are small and become run of the river during larger storms. Larger dams that are managed more directly for flood abatement may prevent additional downstream flood benefits from upstream wetlands, and our models did show that some dams reduce wetlands benefits downstream. Therefore, our results regarding dams are mixed, and could simply be because of the variations in conditions across dams, which may need to be considered by decision makers on a case by case basis, based on local knowledge. Our indicator framework guidebook will allow for this consideration.

Until similar modeling is conducted in other watersheds, it is difficult to say how transferable Tier II indicators are. We expect that these indicators could be used in similar watersheds in the Northeast. Downstream distances for benefits delivery did not appear to vary greatly within the range of subbasin areas and imperviousness explored. Other factors such as stream morphology and subbasin slope may play a role in this as well. Although we were unable to develop generalized indicators quantifying the role of substitutes, such as dams, in decreasing wetlands benefits, it is possible the number of dams in the Woonasquatucket influenced our results for the downstream distance of benefits.

Future Directions

The overall process for developing tiered flood reduction benefits indicators from quantitative modeling was successful. There are several areas for future research to further develop this approach. Since wetlands provide many benefits, one area for future exploration is whether the process we followed can be used for other wetlands benefits.

Although we suspect that Tier III indicators will provide more benefit relevant information than Tier II indicators, it will be important to examine how the information provided by each relates to results from valuation studies and the implications for decisions that are based on each level of indicator. If there is little difference between decisions made based on these different levels of information, then Tier II indicators could be used in most cases, because they require less time and expertise. Tier II indicators will be more rapid than most of the methods previously available and require much less analysis than the Tier III indicators. However, Tier II indicators should be tested and compared to models for other watersheds before assuming transferability.

Now that Tier II indicators have been developed, the resource requirements to implement these indicators elsewhere using a similar approach could be reduced through tools we have developed. For example, the geospatial tool used to calculate wetland volumes based on characterizing their perimeters and depths from Digital Elevation Models will be available for future use. Tools could also be created to characterize beneficiaries more easily or determine which are within the downstream distance required to benefit from wetland sites. These tools could also be provided outside of proprietary geographic information systems through web based services or as standalone programs.

Summary

What is most useful about the approach presented here is that it directly incorporates people and the benefits they receive from ecosystem restoration. Further, it provides a framework that can be used to compare potential wetland restoration scenarios based on these benefits without the need for estimating dollar values. Using an approach to assessing non-dollar benefits that is grounded in economic principles allows for more robust discussion of alternatives through a disaggregated and transparent presentation of the various factors that are likely to affect the level of benefits to people. This can inform many decision contexts where a strict benefit-cost framework is either not appropriate or not necessary. This indicators approach can also easily be extended to incorporate conceptions of value beyond the economic definition of value, and can be transferred to other decision settings and benefit types. We demonstrate how it can be used to complement an existing functional assessment approach, and propose that it can also be used to inform benefit transfers to add more insight into variations in restoration benefits across locations, and who might receive those benefits.

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Appendix A – Wetland Flood Protection Functional Assessment

Functional indicators from Miller and Golet (2001) were incorporated as model inputs, to demonstrate how Tier III benefits indicators derived from the model results can be used as an extension to existing functional assessments.

Function	*	Criteria (highlighted criteria are necessary to the function)	O,E,S†	Source‡	Notes
Flood Abatement		Impervious surfaces cover > 20% of land within 500 feet	O	L	
		Slopes within 500 feet of wetland are > 15%	O	L	
		Point-source inflow	O	L,F	
		Bordering or containing a lower perennial stream	O	L,F	
		Basin wetland	E	F	
		Dominant vegetation is dense and persistent (EM, SS, or FO)	E	L,F	1,2
		Constricted outlet	E	L,F	
		Developed flood-prone areas downstream within 5 miles or to nearest dam (connection by stream or floodway required)	S	L	

*Mark each box as Y, N, D, or NA (i.e., yes, no, don't know, or not applicable)
†O = opportunity; E = effectiveness; S = social significance
‡L = lab data; F = field data
[1]Not applicable if entire wetland unit has been destroyed.
[2]Not applicable if the wetland types of the existing unit and the destroyed portion are different.

Appendix B – Wetland Parameters

Wetland short-term surface water retention was represented in HEC-HMS as diversions—or subbasin flow that is diverted from entering the main channel. Model flow diverted from a given subbasin is a factor of (1) the rate at which subbasin outflow is diverted and (2) the maximum cumulative volume of water that can be diverted. This appendix details the process used to quantify these two diversion parameters from spatial wetland data (Figure B-1).

Parameters for wetlands in the HEC-HMS model included the maximum volume that could be diverted to the wetland (Table B-1, Avg. Vol.) and percent of subbasin flow diverted (Table B-1, % Wetland). Percent of subbasin flow diverted was based on the surface area of wetlands (Table B-1, Area of wetlands) as a percent of total basin area (Table B-1, % Wetland). This percentage informed the diversion's Paired data table (Table B-2 demonstrates the diversion D_W1570 for subbasin W1570), which HEC-HMS used to generate a curve for the Inflow-Diversion function. When total volume diverted to a wetland reaches the maximum volume, flow is no longer diverted from subbasin outflow (Figure B-2).

A Python toolbox in ArcMap 10.2 was used to estimate potential maximum volume retention. The tool calculates volume based on surface elevations as bottom contour, the minimum or average elevation from perimeter vertices as the water surface elevation for height, and areal extent based on wetland polygons. Surface elevations were taken from the LiDAR-derived Digital Elevation Model (DEM). LiDAR returns bounce off standing water, meaning the DEM identifies the bottom of the wetland as the elevation of surface water in the wetland at the time of survey (April 22-May 6, 2011). Any water detected by LiDAR was considered the residual storage in wetlands before flood events, meaning wetland storage was not adjusted in any of the scenarios for more or less residual water.

The water surface elevation for each wetland was determined using the Lane and D'Amico (2010) method where water surface elevation in the wetland is estimated based on the average elevation at vertices around the wetland polygon perimeter. Adjacent wetland polygons were combined and treated as one continuous wetland. This eliminated doughnut or stepped water surfaces formed where interior or upstream wetlands would otherwise have a lower water surface elevation than exterior or downstream wetlands. Although the toolbox allows for volumes to be separated based on wetland type, we did not use this information. Each basin may still contain multiple wetlands (Table B-1, No. provides the original count of wetlands in that basin). Surface elevations were attributed to each wetland and then wetlands were divided into the subbasins before volume was calculated. This ensured water surface elevations were consistent for each wetland but volumes were calculated only for the area of the wetland inside the subbasin where that volume was assigned. Wetland volumes (m^3) were calculated for each subbasin based on both the minimum (Table B-1, Min. Vol.) and average (Table B-1, Avg. Vol.) water surface elevation.

Sensitivity analysis of wetland parameters included the comparison of methods used to calculate wetland maximum diversion volume and percent subbasin flow diverted. To compare methods, the model was run using volumes calculated based on both minimum water surface elevations and average water surface elevations. Sensitivity analysis of percent flow diverted to increased wetland catchment area was modeled by shifting non-wetland area to wetland area in each subbasin by 10%, 25%, 50%, and 75%.

To better understand our conservative assumption of wetland catchment size (only the areal extent of wetlands), the areal extent of wetlands plus a 100-m buffer was also calculated (Figure B-1). Following this assumption, a catchment defined by areal extent plus a 100-m buffer would increase the average percentage diverted by 51% (Table B-1, % Wetland or Buffer).

Similarly, restoration scenarios converted a percentage of non-wetland area in each subbasin to wetland. An example of the percent subbasin flow diverted from the 25% scenario is given in Table B-1, 25 % Restoration Scenario % Wetland, along with the surface area of wetlands that would be added (Table B-1, Increase in Wetlands Area).

Table B-1: **Subbasin wetland parameters and characteristics related to their sensitivity analysis** (subbasin names; number of wetlands in each subbasin; minimum volume (m^3); average volume (m^3); surface area of wetlands (m^2); percent area of subbasin in wetlands; increase in wetland area in 25% Restoration Scenario (m^2); percent area of subbasin in wetlands in 25% Restoration Scenario; and percent of subbasin area in wetlands or 100-m buffer around wetlands).

						25% Restoration Scenario		
Subbasin	**No. of**	**Min. Vol.**	**Avg. Vol.**	**Wetland Area**	**%**	**Increase in**		**%**
W1260	21	235	57644	193663	12%	342450	34%	70%
W1270	8	914	70937	97075	14%	144361	36%	69%
W1280	11	4	19809	38937	11%	82532	33%	76%
W1290	13	1154	126771	294886	24%	227590	43%	77%
W1300	11	3289	381545	225733	20%	229077	40%	70%
W1320	12	513	84013	128685	17%	162337	37%	75%
W1330	7	0	49754	53526	10%	125513	32%	59%
W1340	3	51	43557	47238	52%	11004	64%	100%
W1350	3	11	3170	2041	61%	332	70%	100%
W1360	3	96	17508	17023	50%	4220	63%	100%
W1370	9	119	211908	246137	24%	197529	43%	72%
W1380	10	1404	289029	156578	19%	167333	39%	87%
W1390	13	2966	711624	191622	17%	229178	38%	80%
W1400	9	683	148042	127818	17%	161110	37%	80%
W1410	1	0	154	130	11%	258	33%	100%
W1420	10	1150	140516	365574	25%	274747	44%	76%
W1430	18	3170	158881	314003	15%	457938	36%	60%
W1440	16	4158	353654	402552	25%	306405	44%	80%
W1450	7	1842	152580	121413.82	15%	168712	36%	86%
W1460	4	31	18098	44694.76	13%	73768	35%	66%

						25% Restoration Scenario		
Subbasin	No. of	Min. Vol.	Avg. Vol.	Wetland Area	%	Increase in		%
W1470	11	573	747595	243987.75	19%	252717	40%	88%
W1480	11	3902	642571	334141.92	18%	376178	39%	75%
W1490	8	67	25634	72026.12	9%	176692	32%	56%
W1500	16	4282	305844	98652.97	7%	318029	30%	61%
W1510	2	6	22354	5240.58	3%	41856	27%	49%
W1520	11	774	28637	67077.94	7%	211334	31%	70%
W1530	7	33	7533	50521.56	10%	114884	32%	66%
W1540	15	1239	272703	281186.94	15%	400508	36%	65%
W1550	6	0	654556	195225.45	24%	155950	43%	66%
W1560	14	1843	155684	230047.00	19%	242986	39%	86%
W1570	6	0	179516	123019.13	21%	113302	41%	84%
W1580	1	0	23399	10990.35	56%	2172	67%	100%
W1590	3	0	89714	95334.85	17%	113859	38%	71%
W1600	4	0	145882	199681.28	27%	136066	45%	66%
W1610	12	396	552908	332883.06	25%	253966	44%	81%
W1620	1	0	91312	74880.77	37%	31441	53%	81%
W1640	2	0	92534	11997.66	15%	16898	36%	70%
W1650	3	4	54294	72873.22	34%	34758	51%	84%
W1660	2	0	2917	28589.77	6%	122138	29%	41%
W1670	14	4381	629244	182537.18	15%	268474	36%	74%
W1680	6	113	292082	143675.38	18%	160362	39%	72%
W1690	8	10132	554300	309140.92	24%	251226	43%	89%
W1700	5	0	225748	274304.30	37%	114993	53%	87%
W1710	7	38	48143	63010.56	8%	183345	31%	53%
W1720	5	0	304838	55524.59	26%	40157	44%	95%
W1750	6	0	149209	249967.08	36%	108944	52%	78%
W1770	14	3458	351952	375347.15	20%	376688	40%	55%
W1780	2	0	66699	200640.02	27%	135027	45%	61%
W1790	12	1380	781163	349811.83	19%	364828	40%	67%
W1800	17	19167	320454	228021.39	15%	320136	36%	80%
W1810	8	1060	554893	245656.43	15%	343687	36%	55%
W1820	3	0	169474	66012.48	9%	171945	32%	38%
W1830	3	0	468	4181.93	1%	198018	25%	20%
W1840	13	728	157789	201449.01	25%	152306	44%	94%
W1850	1	0	19853	1474.22	40%	551	55%	100%
W1860	3	0	130707	147106.76	27%	97173	46%	87%
W1870	10	260	90006	112974.04	16%	148549	37%	80%
W1880	7	108	640710	79077.85	19%	86093	39%	90%
W1890	11	175	239144	147579.03	17%	182883	38%	75%
W1900	1	0	14850	24095.52	23%	20296	42%	80%

						25% Restoration Scenario		
Subbasin	No. of	Min. Vol.	Avg. Vol.	Wetland Area	%	Increase in		%
W1910	9	2288	17060	52259.91	15%	75369	36%	88%
W1920	1	0	3043	2913.63	66%	377	74%	100%
W1930	9	20	50590	41680.26	5%	218165	28%	51%
W1940	1	0	41420	64608.43	21%	62376	40%	54%
W1950	1	0	12398	9604.70	49%	2458	62%	100%
W1960	4	41	200861	158143.22	20%	160850	40%	78%
W1970	1	0	124231	123065.50	51%	29227	63%	85%
W2000	11	773	38571	62813.93	6%	239331	30%	53%
W2010	7	9860	432051	251801.56	13%	410642	35%	57%
W2020	1	0	26544	24396.66	47%	6929	60%	87%
W2030	6	29	37534	181523.45	14%	279230	35%	43%
W2040	4	614	59862	51956.44	10%	120197	32%	54%
W2050	9	2548	470700	124138.52	18%	144305	38%	81%
W2060	10	3493	94479	107295.19	10%	232401	33%	63%
W2070	6	69	83776	122890.47	19%	130319	39%	80%
W2090	12	3932	235553	157307.53	17%	187053	38%	79%
W2100	1	0	0	1054.41	1%	17795	26%	41%
W2110	2	0	17375	16093.27	29%	9750	47%	93%
W2130	12	5022	128521	188634.27	13%	315597	35%	49%
W2140	6	489	105906	155634.15	29%	93281	47%	79%
W2150	7	2755	98471	102372.58	15%	143018	36%	77%
W2160	11	479	32317	80443.34	11%	171264	33%	71%
W2170	3	2	18826	79644.14	10%	173033	33%	46%
W2180	1	0	171989	189351.28	69%	21255	77%	97%
W2190	1	0	142670	145424.19	87%	5633	90%	100%
W2200	3	387	339594	215897.11	20%	210022	40%	53%
W2210	4	0	15209	60515.13	11%	117141	34%	55%
W2220	7	141	71316	117015.17	12%	207448	34%	59%
W2230	1	0	195137	274283.28	70%	28831	78%	99%
W2240	2	9	12486	26674.51	25%	19524	44%	100%
W2250	7	134	157678	265799.66	26%	191836	44%	78%
W2260	10	6	204971	238941.99	26%	174472	44%	89%
W2290	5	428	11695	9762.75	2%	137027	26%	38%
W2300	1	0	20962	12497.82	7%	43861	30%	67%
W2310	12	314	62571	104856.52	9%	273492	32%	60%
W2320	8	85	133593	66027.31	10%	149844	32%	75%
W2330	10	12	36466	76387.15	7%	257761	30%	61%
W2340	5	13	85093	69026.59	7%	216849	31%	47%
W2350	1	0	97976	46119.13	16%	60070	37%	62%
W2360	2	0	5022	17454.86	4%	118811	28%	35%

						25% Restoration Scenario		
Subbasin	**No. of**	**Min. Vol.**	**Avg. Vol.**	**Wetland Area**	**%**	**Increase in**		**%**
W2370	11	39	244336	225732.24	13%	382694	35%	58%
W2380	3	1295	70701	76911.96	13%	130673	35%	60%
W2390	1	11	24842	9700.12	47%	2768	60%	100%
W2400	1	0	65559	100393.48	82%	5496	87%	100%
W2410	9	428	227016	391316.74	27%	264586	45%	75%
W2430	4	60	95257	24699.08	4%	131646	28%	43%
W2440	1	0	9257	4500.30	59%	794	69%	100%
W2450	11	1535	109888	88025.92	11%	183230	33%	68%
W2470	3	11	26671	144754.12	26%	101132	45%	69%
W2480	1	0	9174	23838.05	37%	10130	53%	79%
W2490	15	7241	406071	394654.75	19%	431940	39%	63%
W2500	3	2	260337	105841.56	19%	110475	39%	78%
W2520	4	111	8053	11134.11	8%	30910	31%	83%
W2530	1	10	68243	96882.10	19%	101589	39%	75%
W2570	15	882	19439	56768.31	7%	183290	30%	78%
W2580	19	1416	178303	248633.32	19%	273789	39%	84%
W2630	1	51	38474	11000.56	5%	52017	29%	42%
W2670	4	185	15421	20729.86	15%	29980	36%	71%
W2680	1	0	8909	14271.67	21%	13266	41%	77%
W2720	5	97	93044	40297.99	10%	93781	32%	67%
W2770	2	0	279262	68439.70	16%	91035	37%	58%
W2780	6	7	52577	111963.67	14%	170537	36%	66%
W2830	11	386	363106	349482.82	27%	238317	45%	76%
W2870	5	14	273007	114541.26	20%	117037	40%	76%
W2880	2	0	14944	20832.01	10%	46984	32%	44%
W2920	4	712	27457	69361.67	10%	163515	32%	37%
W2930	2	0	10591	61538.51	21%	59622	40%	70%
W2970	2	39	34263	34170.66	27%	22554	46%	100%
W2980	2	0	124	7684.74	6%	29006	30%	59%
W3030	1	0	8414	18645.33	41%	6822	55%	97%
W3070	4	126	14948	24423.18	39%	9512	54%	96%
W3080	5	0	8064	47338.45	12%	88460	34%	70%
W3120	7	1	79033	48285.28	3%	390897	27%	25%
W3130	7	21	58498	74560.22	15%	109924	36%	69%
W3180	17	918	162826	303998.12	18%	351670	38%	70%
W3220	9	21	111732	64923.78	13%	104917	35%	78%
W3230	9	46	56079	50818.03	12%	92895	34%	77%
W3240	8	1558	95405	90992.84	9%	224448	32%	60%
W3250	3	52	84289	44922.39	16%	60700	37%	53%

Table B-2: Diversion D_W2980 paired data table

Inflow (m^3/s)	Diversion (m^3/s)
0	0
1	0.062
100	6.212
100000	6212

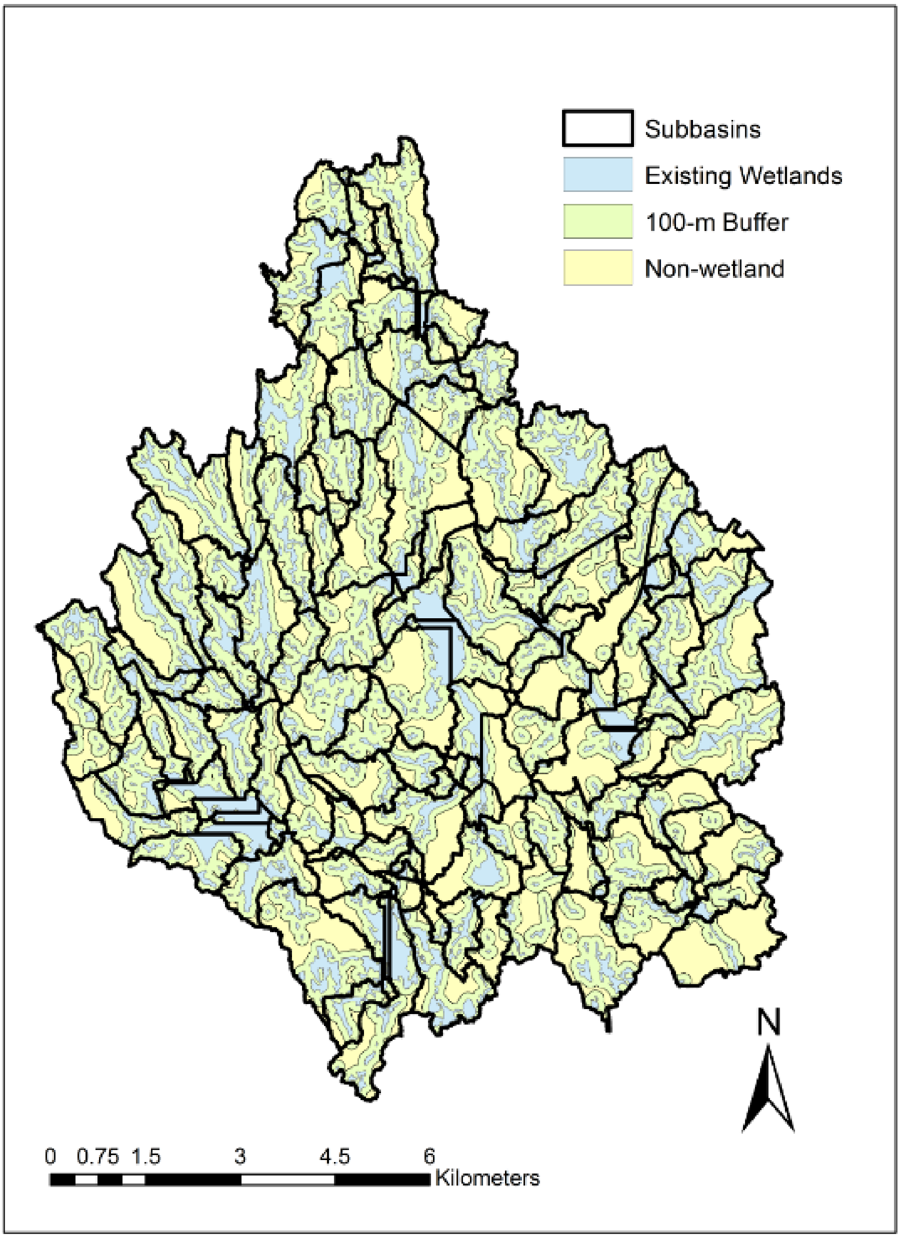


Figure B-1: Modeled area showing wetlands area (blue) a 100m buffer around wetlands (green) and the remaining non-wetlands area (yellow). Black lines show the borders of individual subbasins.

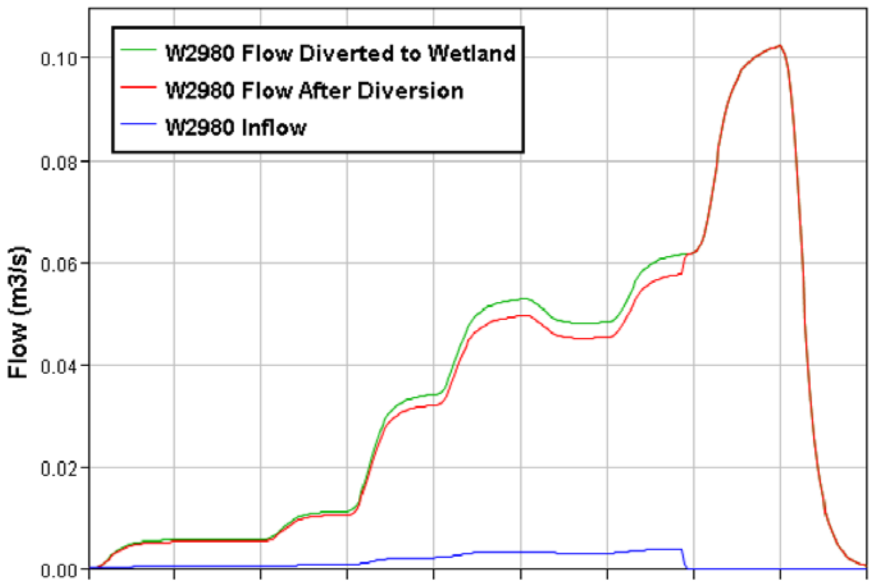


Figure B-2: Inflow, flow diverted to wetlands, and flow after diversion (m^3/s) for subbasin W2980. When the total diverted to Diversion D_W2980 reaches the maximum, flow diversion (blue) goes to zero and subbasin Inflow (green) equals subbasin outflow (red).

Appendix C – PeakFQ Analysis

The PeakFQ program (Flynn et al. 2006) was used to analyze observed peak flows for the Woonasquatucket watershed (Figure C-1 systematic peaks), estimate recurrence intervals for storms modeled (Table C-1), and relate flows from synthetic storms to flows of standard probabilities (Table C-2). All available annual peak stream flows for the Woonasquatucket USGS gage (USGS #01114500) from 1936 to 2014 were used for a total of 73 data points (Figure C-1). Results from PeakFQ include a frequency fitted to the observed peaks and upper and lower uncertainty bounds for those peaks (Figure C-2). A linear regression of the observed peak flows (Figure C-1) showed an average increase of 46 cfs each decade, though the trend is weak due to high annual variability (R^2= 0.0958).

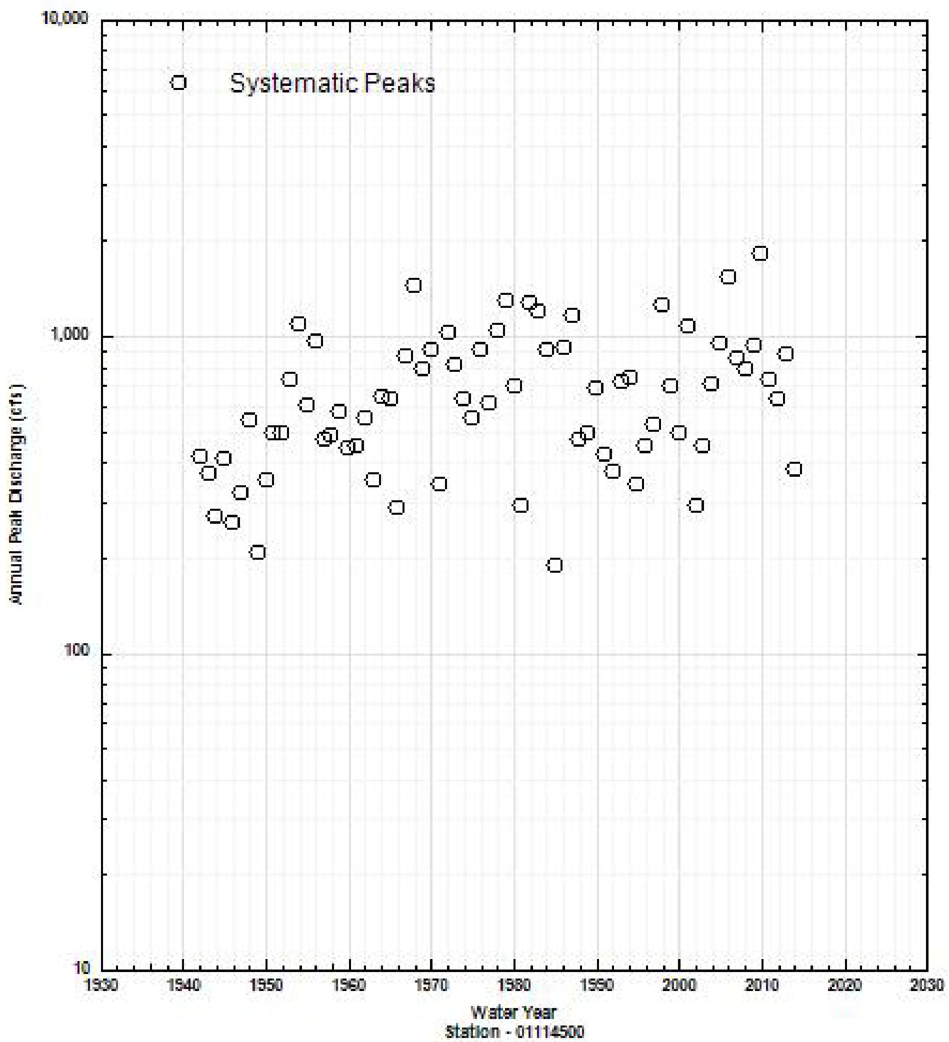


Figure C-1: All annual peak discharge values (cfs) for water years at USGS station #011145000. The general trend in annual peak flows over time increased an average of 46 cfs each decade (y = 4.6588x -8538.9, R^2 = 0.0958).

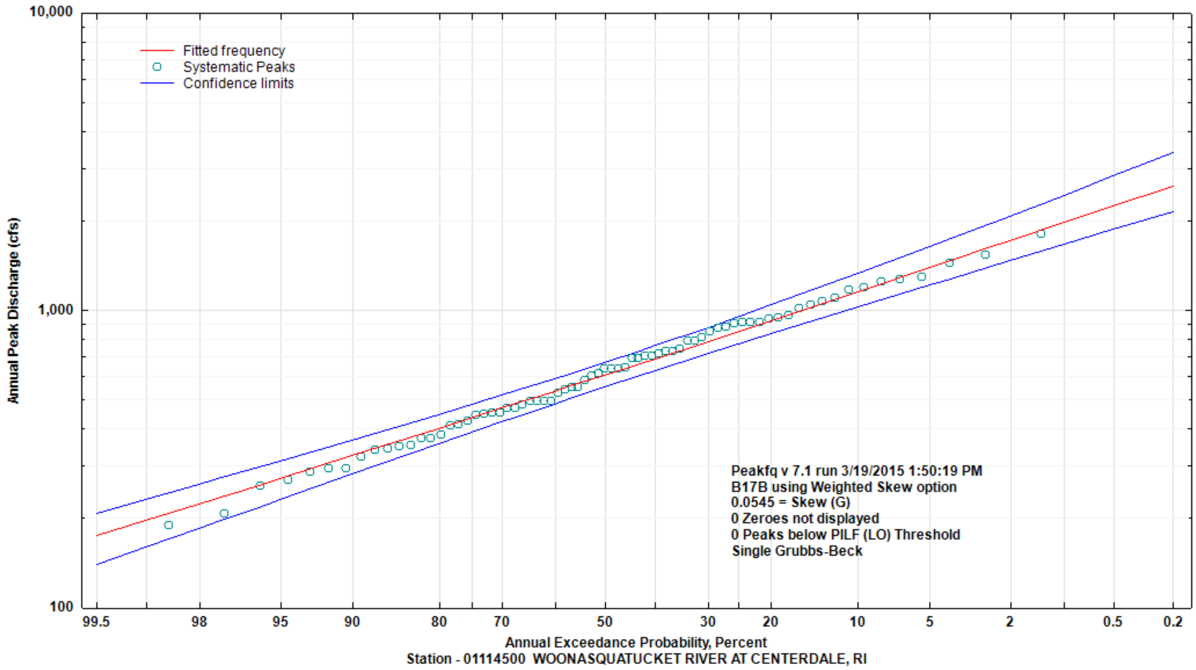


Figure C-2: Annual exceedance probabilities for all observed (systematic) peaks, a curve fit to those probabilities (red), and its upper and lower confidence intervals (blue).

Table C-1: Probability and recurrence intervals for the storm of record and the four storms considered for use in hydrologic modeling

Rank	Date	Use	Discharge (cfs)	Probability (B17B)	Recurrence (Years)
1	Mar. 30, 2010	Not Used: Missing data	1810	0.0135	74.1
2	Oct. 15, 2005	Calibration Storm 1	1530	0.0270	37.0
3	Mar. 22, 2001	Calibration Storm 2	1070	0.1351	7.4
4	Apr. 03, 2005	Validation Storm 1	943	0.1892	5.3
6	Apr. 16, 2007	Validation Storm 2	851	0.2973	3.4

Table C-2: Expected discharge (cfs) required for peak flows of standard probabilities

Probability	Recurrence	B17B	Record	Lower	Upper
0.002	500	2621.0	2371.0	2151.0	3377.0
0.005	200	2239.0	2072.0	1868.0	2821.0
0.010	100	1968.0	1851.0	1664.0	2434.0
0.020	50	1710.0	1635.0	1466.0	2074.0
0.040	25	1463.0	1422.0	1273.0	1739.0
0.100	10	1152.0	1142.0	1023.0	1330.0
0.200	5	922.4	926.0	830.8	1041.0
0.500	2	605.8	614.2	550.0	667.0
0.995	1.005	174.0	161.0	139.1	207.4

Appendix D –HMS Parameters

Table D-1: **Spatial data used to produce variables and parameterize HMS model**

Name	Type	Creator	Date	Resolution	Availability
Dams	Dams Inventory	RI DEM Division of Compliance & Inspection's Dam Safety Program	2000	NA	Statewide
LiDAR DEM (1m): Statewide Spring 2011	Digital Elevation Model	URI Environmental Data Center	2011	1 m	Statewide
Impervious Surfaces 2003-2004	Impervious Surfaces	Sanborn	2003-2004	2ft	Statewide
Land Use- 2003/2004	Land Use/Land Cover	Sanborn Map Company, State of RI	2007	minimum mapping unit of 0.5 acre	Statewide
National Hydrography Dataset Plus (Version 2) Flowlines	Streams	USGS and Horizon Systems	2012	1:100,000	Most of the continental U.S.
5K Streams (1:5000)	Streams	RIDOT	1997	1:5,000	Statewide
Wetlands	Wetlands	IEP inc	1993	1km^2 (0.25 acre)	Statewide

Curve Number Loss Method Parameters

At the subbasin scale, we used the loss method in HEC-HMS to generate runoff volume based on the Soil Conservation Service Curve (SCS) Curve Number (CN) approach and the percent impervious surface in the subbasins (Table D-2). The SCS method requires a grid of CN values, based on soil type and land cover (Table D-1), where land cover types are assigned CN values (USDA 1986). The average basin CN value was 67 and the average subbasin percent imperviousness was 12% (averages not weighted by basin area). The transform method uses the SCS Unit Hydrograph with the lag time calculated using TR-55 method (McCuen 1982). We estimated parameters for the TR-55 method, including impervious surface area and Manning's N for sheet flow. Parameters left as default values were: cross-sectional flow area (2 m^2), wetted perimeter (6 m), and Manning's N for channel flow (0.03). Basin slopes for the TR-55 method were determined using the original DEM rather than the filled DEM, meaning occasionally channel slope (28/139 instances) and watercourse slope (4/139 instances) were ≤0. For many of these subbasins slope was minimal because the channel passed through a reservoir. The slopes for these channels and watercourses were adjusted to 0.0001 to reflect a flat slope while still allowing for TR-55 calculations.

Table D-2: Subbasin parameters; % impervious, Curve Number (CN) values directly from HEC-GeoHMS based on AMC II, after machine calibration to the 2005 Fall storm, after machine calibration to the 2001 Spring storm, based on the Dry AMC I method, and based on the Wet AMC III method

Subbasin Name	Impervious (%)	Initial CN (AMC II)	’05 CN Calibration	’01 CN Calibration	Dry CN (AMC I)	Wet CN (AMC III)
W1260	6.1%	59.4	40.0	69.0	39.8	77.3
W1270	7.6%	64.1	43.1	74.4	42.9	83.3
W1280	4.6%	60.6	40.7	70.3	40.6	78.7
W1290	4.8%	60.4	40.6	70.1	40.5	78.5
W1300	3.7%	57.2	38.4	66.3	38.3	74.3
W1320	4.8%	62.3	41.9	72.3	41.7	81.0
W1330	9.0%	64.2	43.2	74.5	43.0	83.4
W1340	7.4%	68.7	46.2	79.7	50.1	83.1
W1350	1.2%	71.1	47.8	82.6	51.9	86.1
W1360	0.4%	69.3	46.6	80.4	50.6	83.8
W1370	4.8%	70.3	47.3	81.6	51.3	85.0
W1380	5.6%	61.3	41.2	71.2	41.1	79.7
W1390	3.1%	58.8	39.5	68.2	39.4	76.4
W1400	6.7%	67.9	45.7	78.9	49.6	82.2
W1410	32.9%	82.2	55.3	95.4	64.9	93.7
W1420	0.8%	56.9	38.3	66.1	38.1	74.0
W1430	8.5%	62.9	42.3	73.0	42.1	81.8
W1440	2.9%	67.9	45.6	78.8	49.5	82.1
W1450	1.6%	60.4	40.6	70.2	40.5	78.6
W1460	6.8%	63.0	42.4	73.1	42.2	81.9
W1470	3.0%	63.9	43.0	74.2	42.8	83.0
W1480	2.2%	63.5	42.7	73.7	42.6	82.6
W1490	4.2%	60.7	40.8	70.5	40.7	78.9
W1500	6.4%	61.0	41.0	70.9	40.9	79.3
W1510	4.3%	56.8	38.2	66.0	38.1	73.9
W1520	16.0%	68.7	46.2	79.8	50.2	83.2
W1530	1.1%	59.6	40.1	69.1	39.9	77.4
W1540	13.9%	71.0	47.7	82.4	51.8	85.9
W1550	13.7%	71.3	48.0	82.8	52.1	86.3
W1560	8.4%	66.1	44.5	76.7	48.3	80.0
W1570	21.2%	76.0	51.1	88.2	60.0	86.6
W1580	0.0%	55.9	37.6	64.9	37.5	72.7
W1590	0.0%	65.0	43.7	75.5	47.5	78.7
W1600	14.9%	76.9	51.7	89.3	60.7	87.7
W1610	4.1%	62.6	42.1	72.6	41.9	81.3
W1620	11.9%	73.8	49.6	85.7	53.9	89.3
W1640	6.2%	56.3	37.9	65.4	37.7	73.2
W1650	22.7%	77.4	52.1	89.9	61.2	88.2
W1660	16.4%	64.3	43.2	74.6	43.1	83.5
W1670	5.2%	65.8	44.3	76.4	48.0	79.6
W1680	8.2%	75.6	50.8	87.8	59.7	86.2
W1690	2.3%	63.3	42.6	73.5	42.4	82.3
W1700	7.5%	71.4	48.0	82.9	52.1	86.4
W1710	16.7%	67.0	45.0	77.7	48.9	81.0
W1720	8.1%	66.7	44.9	77.5	48.7	80.7

Subbasin Name	Impervious (%)	Initial CN (AMC II)	'05 CN Calibration	'01 CN Calibration	Dry CN (AMC I)	Wet CN (AMC III)
W1750	7.3%	72.9	49.0	84.6	53.2	88.2
W1770	11.0%	67.1	45.1	77.9	49.0	81.1
W1780	14.0%	67.6	45.4	78.4	49.3	81.7
W1790	9.9%	66.9	45.0	77.7	48.9	81.0
W1800	4.8%	67.3	45.2	78.1	49.1	81.4
W1810	1.2%	58.5	39.3	67.9	39.2	76.1
W1820	13.0%	64.6	43.5	75.0	43.3	84.0
W1830	15.4%	70.3	47.3	81.7	51.3	85.1
W1840	2.9%	64.0	43.1	74.4	42.9	83.3
W1850	0.0%	70.9	47.7	82.3	51.8	85.8
W1860	5.2%	63.7	42.8	73.9	42.7	82.8
W1870	4.9%	67.1	45.1	77.9	49.0	81.2
W1880	0.2%	60.6	40.8	70.4	40.6	78.8
W1890	13.1%	68.5	46.1	79.5	50.0	82.9
W1900	9.8%	63.0	42.4	73.1	42.2	81.9

Muskingum Routing Method Parameters

At the reach-scale, the hydrologic routing of flows followed the Muskingum approach. In this approach, the travel time in the reach (K) is estimated using lag time. Reach K (hrs) values were calculated using K = L/3600V, where L = reach length (m) and V = reach velocity (m/s) (Olivera and Maidment 2000). Reach length was already available, and channel average velocity from the TR-55 method was used for reach velocity. The average Reach K value of reaches included in the HEC-HMS model was 0.446 hours. Attenuation within the reach is accounted for through a constant (X) which ranges from 0 to 0.5, with 0 meaning storage is controlled by outflow (full attenuation), and 0.5 giving equal weighting to inflow and outflow (no attenuation). Reach X values were set to a default of 0.2, except where reaches overlapped large bodies of open water, where Reach X was set to 0.4 to represent increased storage. We estimated the minimum and maximum number of reaches based on: the length of the river, the reach velocity (from the TR-55 method), and the one minute time step of the model (Olivera and Maidment 2000). This method allows for the time it takes for water to move through a reach to be less than the time step of the model. We also assumed no baseflow from each subbasin, instead adjusting our gage data within HEC-HMS to reflect baseflow.

Table D-3: Reach parameters; Initial Muskingum K and X values, Muskingum values after machine calibration to the 2005 storm, Muskingum values after machine calibration to the 2001 storm

Reach	Initial		'05 Calibration		'01 Calibration	
	K	X	K	X	K	X
R100	0.13	0.2	0.16	0.23	0.20	0.25
R1010	0.27	0.2	0.30	0.23	0.38	0.25
R1040	0.21	0.2	0.24	0.23	0.32	0.25
R1050	0.30	0.2	0.33	0.23	0.37	0.25
R1060	0.14	0.2	0.17	0.23	0.22	0.25

Reach	Initial		’05 Calibration		’01 Calibration	
	K	X	K	X	K	X
R1070	0.11	0.2	0.14	0.23	0.20	0.25
R1090	0.02	0.4	0.05	0.43	0.12	0.46
R110	0.21	0.2	0.24	0.23	0.32	0.25
R1100	0.97	0.2	1.00	0.23	1.06	0.25
R1130	1.09	0.2	1.12	0.23	1.14	0.26
R1150	0.03	0.2	0.06	0.23	0.12	0.25
R1160	0.44	0.4	0.47	0.43	0.53	0.46
R1200	0.72	0.2	0.75	0.23	0.81	0.25
R1220	0.37	0.2	0.40	0.23	0.43	0.25
R1240	0.33	0.2	0.36	0.23	0.43	0.25
R130	0.65	0.2	0.68	0.23	0.73	0.25
R160	0.02	0.2	0.05	0.23	0.12	0.25
R190	0.08	0.2	0.11	0.23	0.18	0.25
R20	0.23	0.2	0.26	0.23	0.34	0.25
R200	0.05	0.2	0.08	0.23	0.15	0.25
R2550	1.97	0.4	2.00	0.43	2.06	0.46
R260	0.19	0.2	0.22	0.23	0.29	0.25
R2640	0.30	0.4	0.33	0.43	0.39	0.46
R2690	0.11	0.4	0.14	0.43	0.21	0.46
R2750	0.17	0.2	0.20	0.23	0.28	0.25
R2800	0.17	0.2	0.20	0.23	0.28	0.25
R2890	0.22	0.2	0.25	0.23	0.34	0.25
R2940	1.08	0.2	1.11	0.23	1.16	0.25
R2990	0.11	0.2	0.14	0.23	0.20	0.25
R300	0.13	0.2	0.16	0.23	0.24	0.25
R3040	0.37	0.2	0.40	0.23	0.44	0.25
R3090	0.35	0.2	0.38	0.23	0.42	0.25
R3140	0.04	0.2	0.08	0.23	0.13	0.25
R3200	1.45	0.4	1.48	0.43	1.52	0.45
R330	0.76	0.4	0.79	0.43	0.84	0.46
R340	0.23	0.2	0.26	0.23	0.34	0.25
R350	0.06	0.4	0.09	0.43	0.15	0.46
R380	0.52	0.4	0.55	0.43	0.59	0.46
R400	0.10	0.2	0.13	0.23	0.21	0.25
R410	0.15	0.2	0.18	0.23	0.27	0.25
R420	1.51	0.4	1.57	0.43	1.61	0.46
R430	1.53	0.4	0.20	0.43	0.29	0.46
R460	0.30	0.4	0.33	0.43	0.39	0.46
R470	0.11	0.4	0.14	0.43	0.19	0.46
R480	0.57	0.2	0.60	0.23	0.67	0.25

Reach	Initial		’05 Calibration		’01 Calibration	
	K	X	K	X	K	X
R510	0.02	0.2	0.05	0.23	0.12	0.25
R530	0.55	0.2	0.58	0.23	0.64	0.25
R560	0.09	0.4	0.12	0.43	0.21	0.46
R580	0.02	0.3	0.05	0.30	0.14	0.36
R610	0.12	0.2	0.15	0.23	0.21	0.25
R620	0.18	0.2	0.21	0.23	0.29	0.25
R630	0.04	0.2	0.07	0.23	0.14	0.25
R640	0.93	0.4	0.96	0.43	1.05	0.46
R650	0.28	0.2	0.31	0.23	0.40	0.25
R660	2.90	0.28	2.93	0.28	2.98	0.33
R690	0.82	0.4	0.86	0.43	0.96	0.46
R70	0.42	0.2	0.45	0.23	0.49	0.25
R700	0.09	0.2	0.13	0.23	0.21	0.25
R710	0.02	0.2	0.05	0.23	0.15	0.25
R720	0.38	0.4	0.41	0.43	0.45	0.46
R770	0.21	0.2	0.24	0.23	0.35	0.25
R790	0.20	0.4	0.23	0.43	0.31	0.46
R80	0.02	0.2	0.05	0.23	0.22	0.25
R810	0.77	0.4	0.80	0.43	0.86	0.46
R820	0.28	0.2	0.31	0.23	0.39	0.25
R840	0.11	0.2	0.14	0.23	0.20	0.25
R850	0.21	0.2	0.24	0.23	0.34	0.25
R870	1.66	0.4	1.69	0.43	1.74	0.46
R880	0.55	0.4	0.58	0.43	0.63	0.46
R890	0.57	0.4	0.60	0.43	0.64	0.46
R910	1.39	0.4	1.42	0.43	1.47	0.46
R920	0.44	0.4	0.47	0.43	0.53	0.46
R940	0.92	0.2	0.95	0.23	1.01	0.25
R960	1.23	0.2	1.26	0.23	1.30	0.25
R980	0.62	0.2	0.65	0.23	0.70	0.25
R990	0.34	0.2	0.37	0.23	0.44	0.25

Parameter Calibration

Every reach in the HEC-HMS model was optimized for both its Muskingum X and K values. In confirmed water storage areas where the initial Muskingum X value was 0.4, the X value was restricted between 0.2 and 0.5 so the values for X would not be less than the overall default or greater than the theoretical maximum. For all other reaches where Muskingum X values were initially the default (0.2), the value was restricted to between 0.1 and 0.4, since these reaches should have an X value greater than the default and less than that of open water. Reach Muskingum K values were restricted to an order of magnitude above (10K) or below (K/10) the original value (Table D4). Since less emphasis was placed on accurately assessing initial values for other parameters, this restriction helped to ensure that the calibration process didn't skew K values in place of other parameters, at least for the initial calibrations. There were two exceptions where reach Muskingum X values were not optimized because increased X would have resulted in instability that sub-reaches could not correct. These reaches were R580 (X=0.3) and R660 (X=0.28).

Table D-4: **Example of restrictions on reach component optimization for parameters Muskingum X and K**

Reach	Initial X	Min X	Max X	Initial K	Min K	Max K
R100	0.2	0.1	0.4	0.13	0.013	1.3
R920	0.4	0.2	0.5	0.44	0.044	4.4

Basin CNs were uniformly calibrated by a scale factor. A scale factor works the same way as our restrictions on Muskingum K, where all values are increased by that factor. CNs were held to a factor of 0.01 to 100 (HEC-HMS default).

As an alternative to the machine calibrations, CNs were adjusted for both the Fall 2005 model using AMC 1 methods (D-1 AMC I Dry) and the Spring 2001 model using AMC III methods (D-1 AMC III Wet). Without further calibration these methods yielded similar or improved fits for both models and were chosen over the machine calibrated basin CNs.

Both the Spring 2001 and Fall 2005 models were used to compare the three sets of routing parameter values (those from the '05 calibration, the '01 calibration, and an average of the two "blended"). Both model peaks performed better with the wet parameters; whereas Root Mean Square (RMS) Error and Nash-Sutcliff were slightly better for both models using the blended parameters (Table D-5). Model peak fit is more important for our application than general model fit, so the routing parameters from the Spring '01 calibration were used for both the spring and fall models.

Table D-5: Comparison of both model results using three sets of routing parameters

Routing Parameters	Storm	Peak Error (m^3/s)	Peak Error (Hours)	RMS Error	Nash-Sutcliff
Dry Parameters	Oct '05	7.5	~2 early	4.6	0.885
Blended Parameters	Oct '05	6	~1 early	3.8	0.921
Wet Parameters	Oct '05	2.8	~1.5 early	4.2	0.904
Dry Parameters	Mar '01	-3.7	~2 late	5.1	0.547
Blended Parameters	Mar '01	-3.6	~3 late	4.9	0.580
Wet Parameters	Mar '01	-2.8	~2.5 late	5.0	0.564

Appendix E – Reservoirs

Reservoirs were parameterized in the models by identifying reservoirs associated with dams with a maximum storage of 12 acre-feet or more from a statewide spatial data layer of dams (Table D-2). Each reservoir was assigned an outflow curve (e.g. Figure E-1 for Georgiaville Reservoir) based on storage-discharge (e.g. Table E-1 for Georgiaville Reservoir) with initial conditions as inflow equal to outflow meaning any baseflow actually exiting at the time was ignored. Once the reservoir volume exceeded storage the discharge increased with no corresponding increase in storage.

Table E-1: Georgiaville Reservoir storage-discharge table

	Storage	Flow
Normal	1603.5	0
Maximum	2405.3	106.84

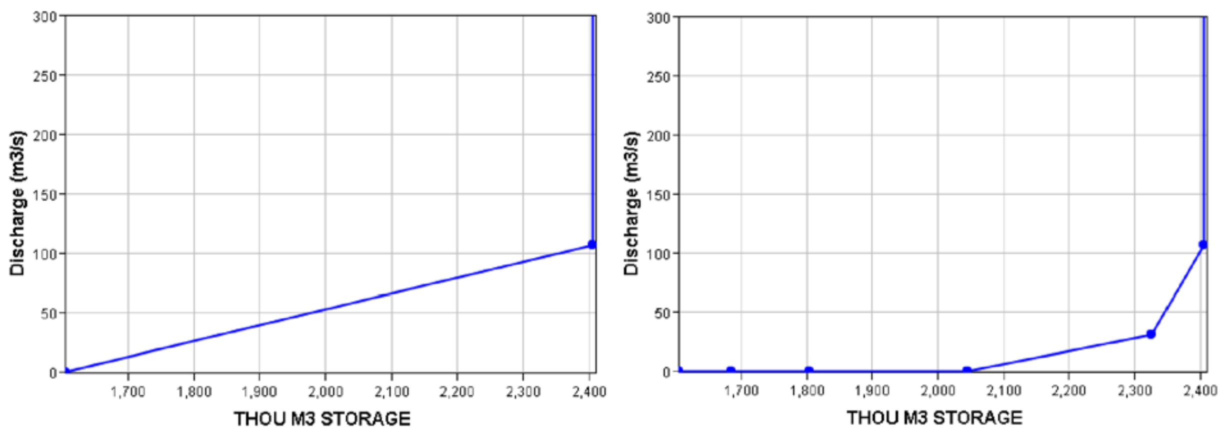


Figure E-1: Georgiaville Reservoir Storage-Discharge Function: original (from Table E-1, left) and with an exponential function (from Table E-2, right).

After initial calibration model reservoirs were updated in two ways. First, four data points were added to the storage-discharge table based on an exponential relationship between the two original data points. This change allowed reservoirs to fill up faster and drain more slowly. The impact of changing individual reservoirs in this way was variable, but overall peak fit for the Spring ’01 model increased by 0.5 m^3/s. The second update was to increase the initial reservoir discharge to better represent conditions before the Spring ’01 storm. Modeled flows were calculated using initial discharges for all reservoirs of 0, 1, 2, 5, and 10 m^3/s (Figure E-3). The results suggested the impact of initial discharge varies over the duration of the model. Initially modeled flows were very sensitive to the discharge, but modeled flows converged again after about 3 days. However, modeled flow where initial reservoir discharge was equal to inflow did not converge with the other modeled flows. An initial reservoir discharge of 0.5 m^3/s was chosen because this value was large enough to increase peak flow but small enough not to exceed baseflow.

Table E-2: Updated Georgiaville Reservoir storage-discharge table based on an exponential relationship between normal and maximum discharge

	Storage	Flow
Normal	1603.5	0
10%	1683.7	0.003
25%	1804.0	0.017
75%	2044.5	0.55
90%	2325.1	31.27
Maximum	2405.3	106.84

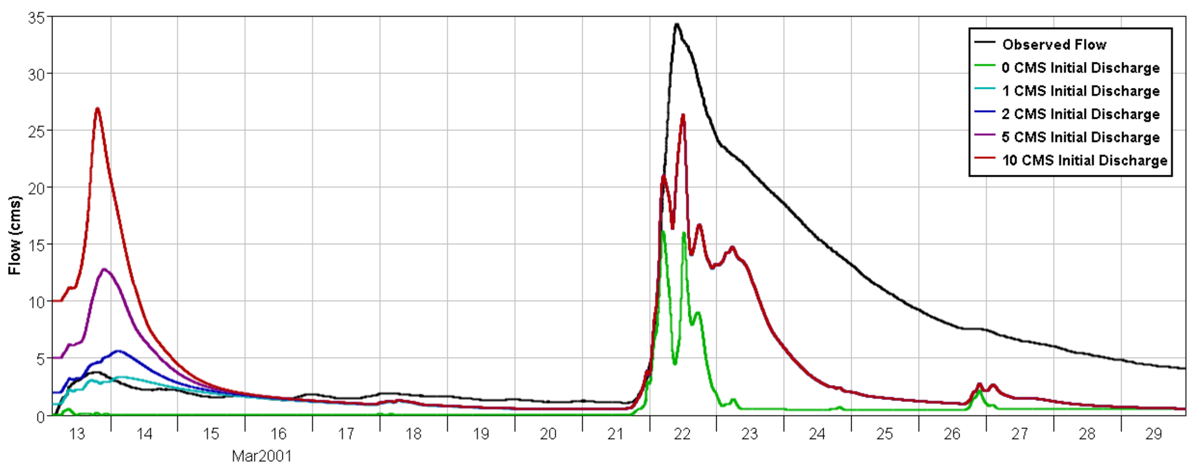


Figure E-3: Results of altering initial reservoir discharge compared to observed flow (black). Initial reservoir discharges tested ranged from 0 CMS (green) to 10 CMS (red).

Appendix F – HMS Calibration and Validation Data

The storm of record occurred in March of 2010, however much of the data around the peak of the event were not available. Therefore we chose to calibrate to the second largest event (October 15, 2005; Figure F-1). The model was started at 00:00 on October 12, 2005 and stopped at 00:00 on October 17th, 2005, for a total duration of 121 hours. Baseflow, the minimum flow during this time period, was 0.878 m^3/s.

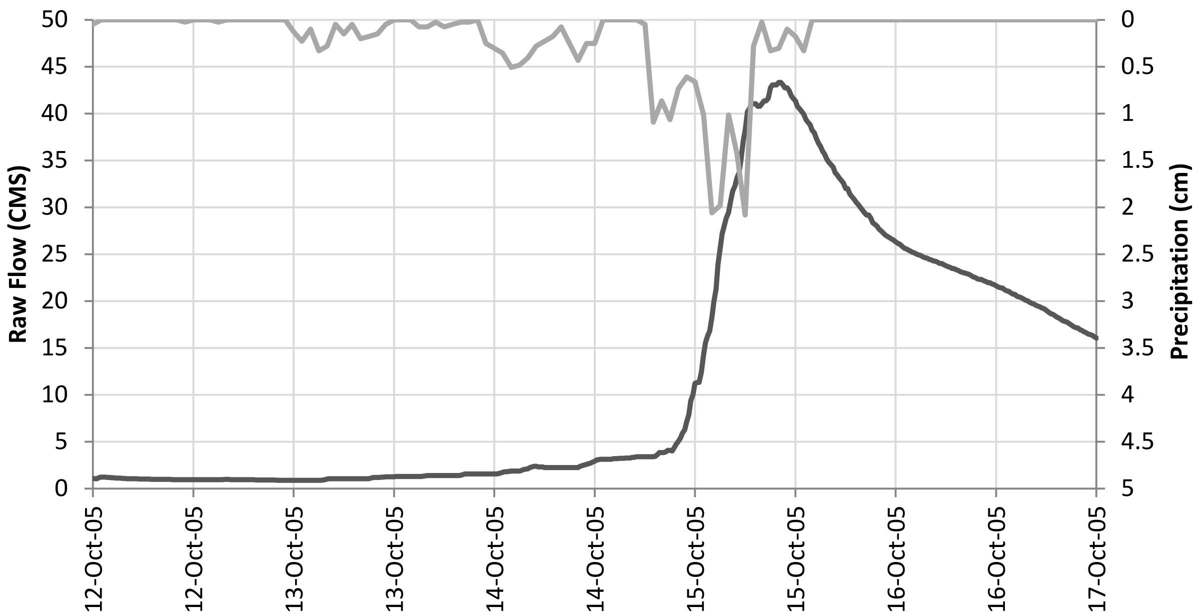


Figure F-1: Hydrograph (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the calibration storm (October 15, 2005). Raw flows are shown without baseflow removed.

The third largest flow (March 22, 2001) was used to calibrate the second flow model (Figure 13). Calibration data started at 03:00 the morning of March 13, 2001 and ran until 23:00 March 29, 2001, for a total of 404 hours. Baseflow, the minimum flow observed during the run time, was 2.5 m^3/s.

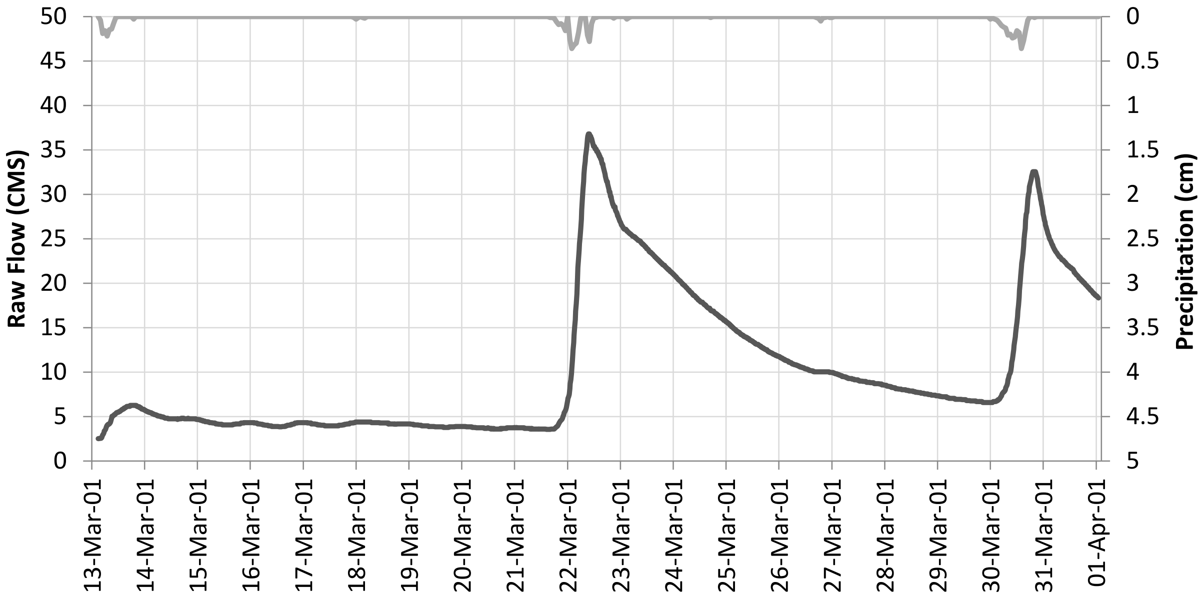


Figure F-2: Hydrograph (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the calibration storm (March 22, 2001). Raw flows are shown without baseflow removed.

The fourth largest flow (April 03, 2005) actually occurred in the spring of the same year as the storm used to calibrate the model. Precipitation and flow data surrounding this flow were used as the second validation storm (Figure F-3). Validation started at 4:00 on April 2, 2005 and ran until 23:00 April 7, 2005. Daylight savings was observed during this time period, at 02:00 on April 3rd, resulting in a duration of 138 hours. Baseflow, the minimum flow observed during the run time, was 5.15 m^3/s.

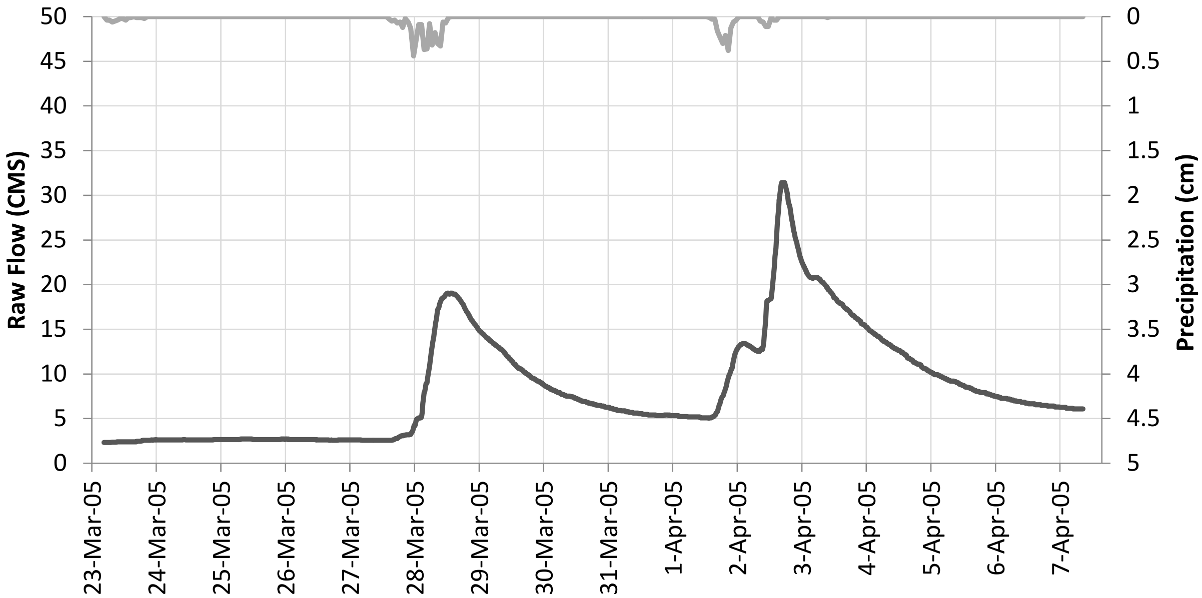


Figure F-3: Hydrograph (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the validation storm (April 03, 2005). Raw flows are shown without baseflow removed.

The fifth largest flow occurred in December, when average temperatures drop below freezing (Figure 7). Although HEC-HMS has parameters to be able to accommodate sub-freezing temperatures, we had not collected the necessary data or calibrated the model for these conditions. Skipping the fifth largest flow, the sixth largest flow (Table 5; April 16, 2007) became the third and last validation storm. The third validation started at 00:00 on April 13, 2007 and ran until 15:00 April 23, 2007 for a total duration of 255 hours (Figure F-4). Baseflow, the minimum flow observed during the run time, was 4.13 m^3/s.

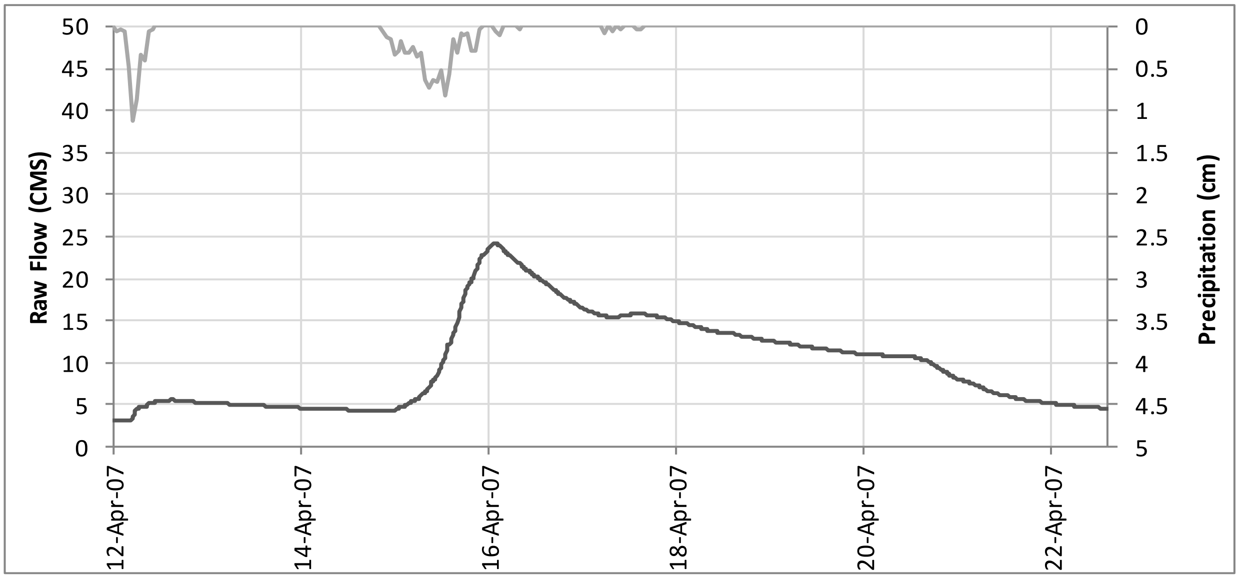


Figure F-4: Hydrograph (USGS Station 01114500; black line in m^3/s (CMS) on left axis) and hourly precipitation (NOAA Coop 376698; gray line in cm on right axis) for the calibration storm (April 16, 2007). Raw flows are shown without baseflow removed.

Table F-1: Summary of storm baseflows.

Storm	Estimated Baseflow (m^3/s)
October '05	0.88
March '01	2.52
April '05	5.15
April '07	4.13

Appendix G – Synthetic Storms

Synthetic storms of known probability were used as precipitation data to generate simulated flows with the calibrated model. The duration of the synthetic storm was 24 hours based on observed storm events used to calibrate and validate the model. Peak precipitation intensity was positioned in the center of that 24-hour duration. Duration intervals are used in HEC-HMS to create precipitation curves for the synthetic storm, where higher intensity precipitation is received during shorter intervals, and longer intervals are move evenly distributed across time. Duration intervals of as little as 5 minutes were available. Based on comparisons of 5-minute (blue), 1-hour (red), and 2-hour (green) duration intervals (Figure G-1) peak precipitation during the 1-hour interval still exceeded peak precipitation observed in calibration and validation storms, but 5-minute duration intervals were off by degrees of magnitude. Ultimately, 1-hour duration interval was used for the synthetic storms. This interval also corresponded to the observed hourly precipitation used to calibrate and validate the models.

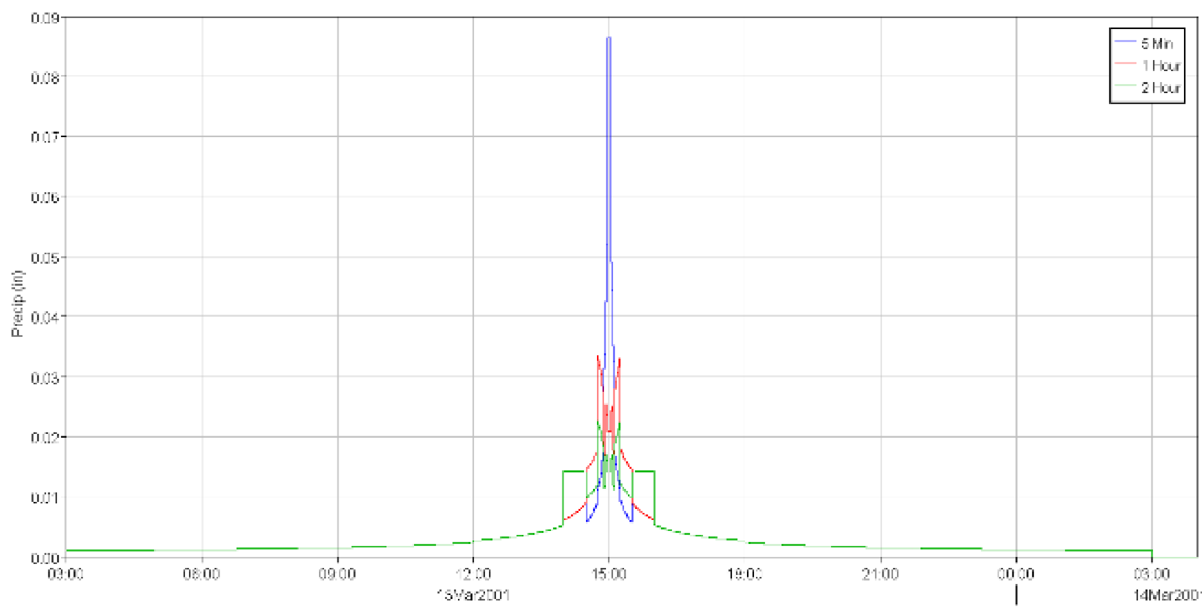


Figure G-1: Peak precipitation values were compared across three duration intervals: 5-minute (blue), 1-hour (red), and 2-hour (green).

Appendix H – Maps

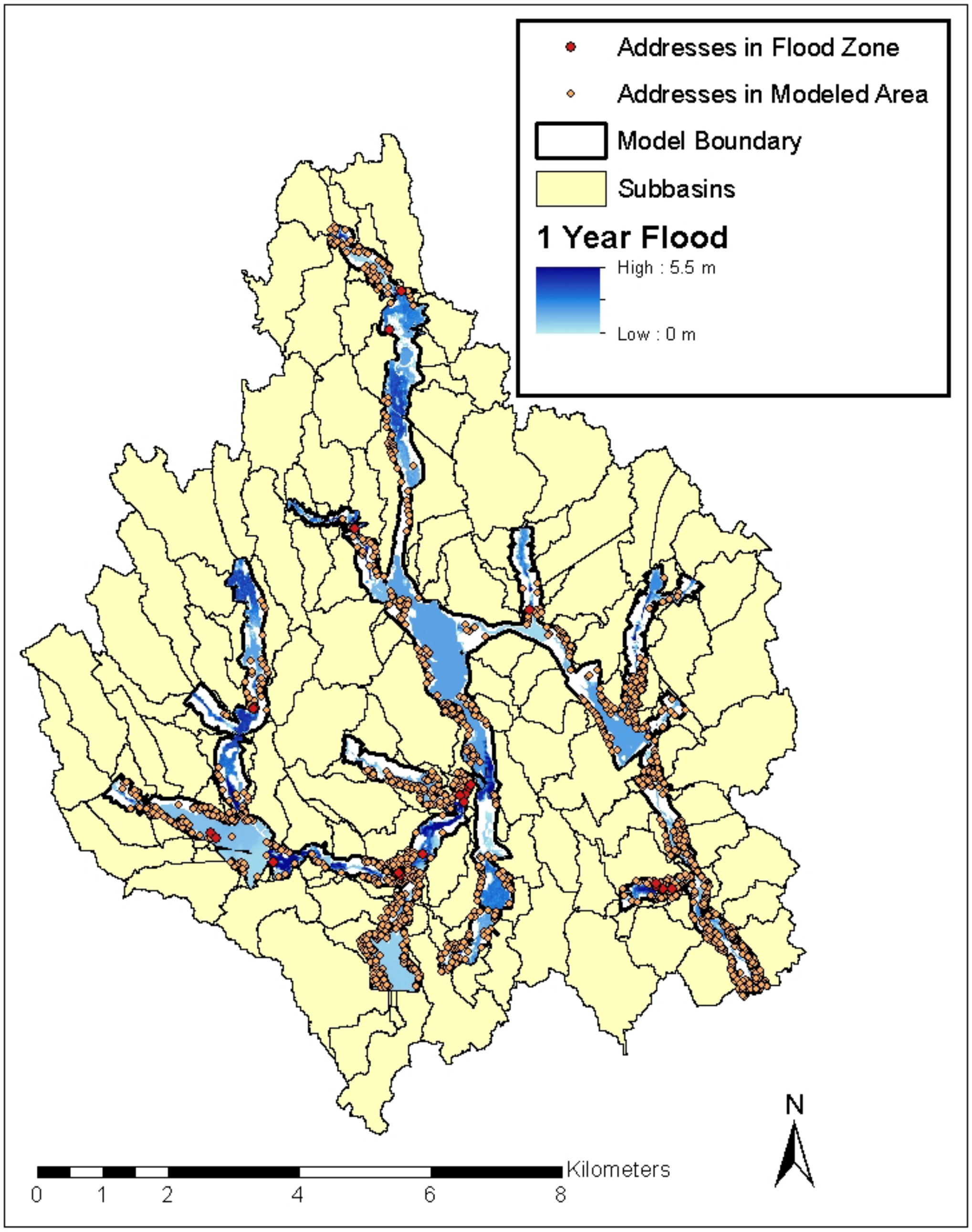
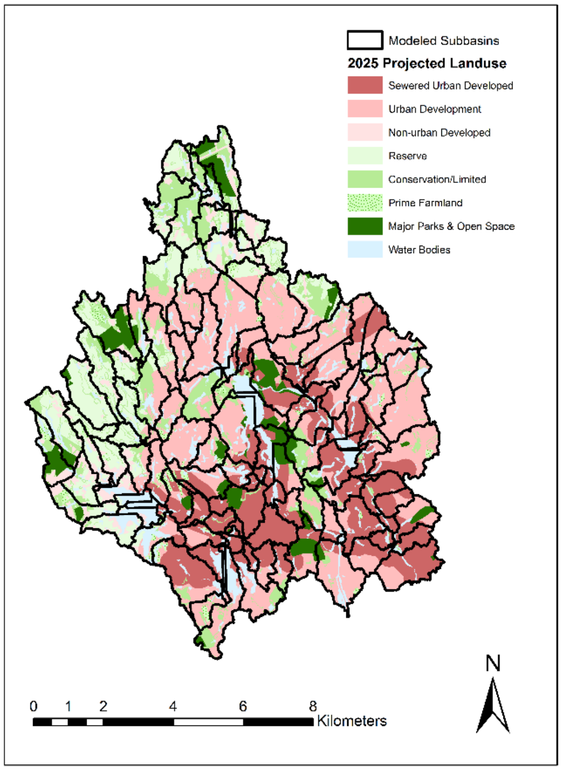


Figure H-1: Addresses (orange) included in the area where flood modeling was performed (black border) and the subset of those addresses (red) that experienced flooding in the 1-year Baseline Scenario (blue).



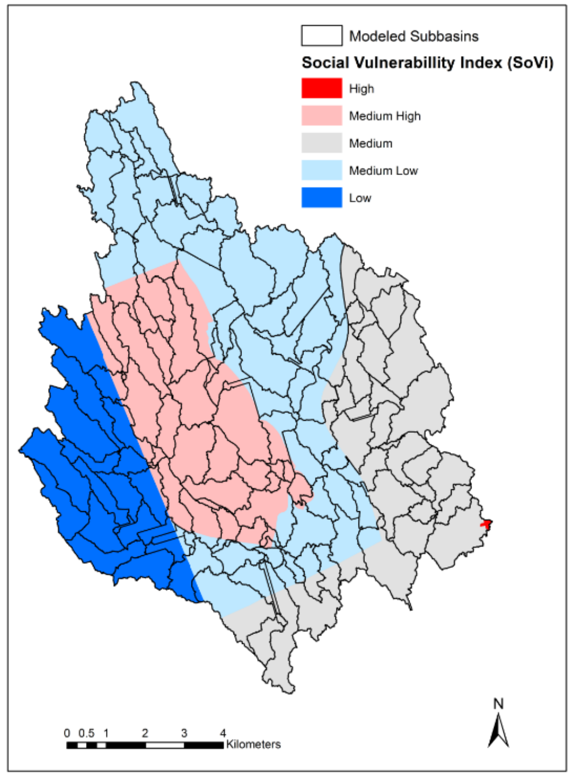


Figure H-2: 2025 Projected landuse across modeled subbasins.

Figure H-3: Social Vulnerability Index across modeled subbasins and flood modeled area.

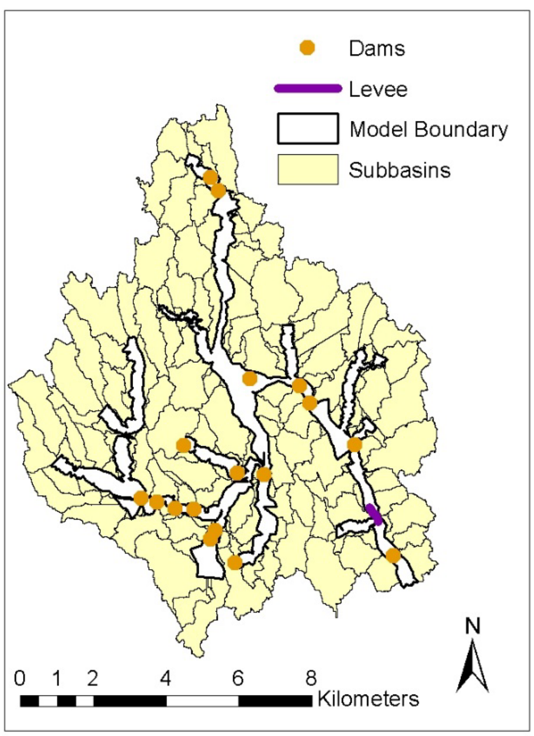


Figure H-4: Dams (brown) and levees (purple) in the flood modeled area (black border).

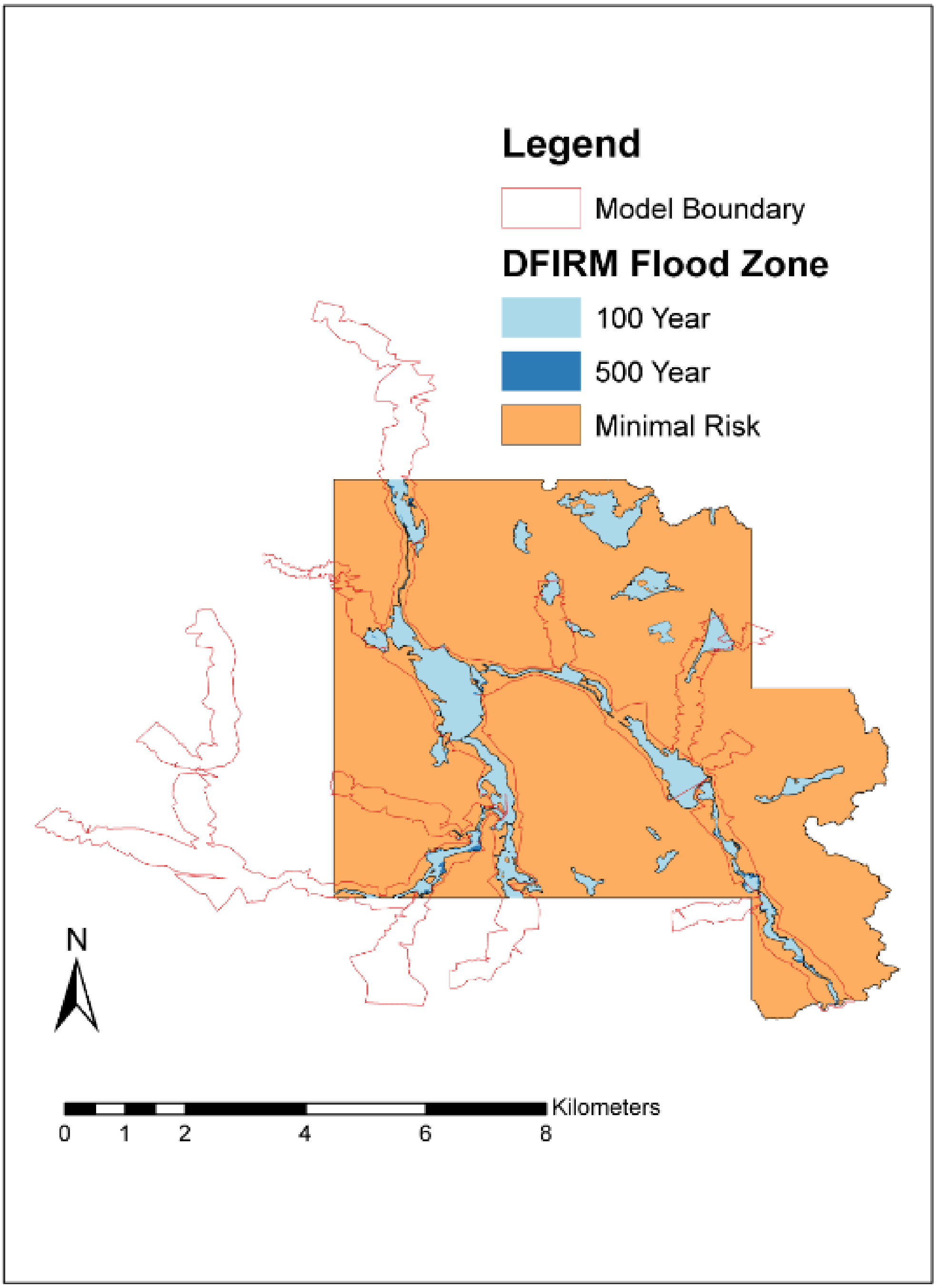


Figure H-5: Flood model boundary (red boundary) compared to FEMA DFIRM mapped flood zones. Minimal risk areas are outside of the 500 year flood zone.

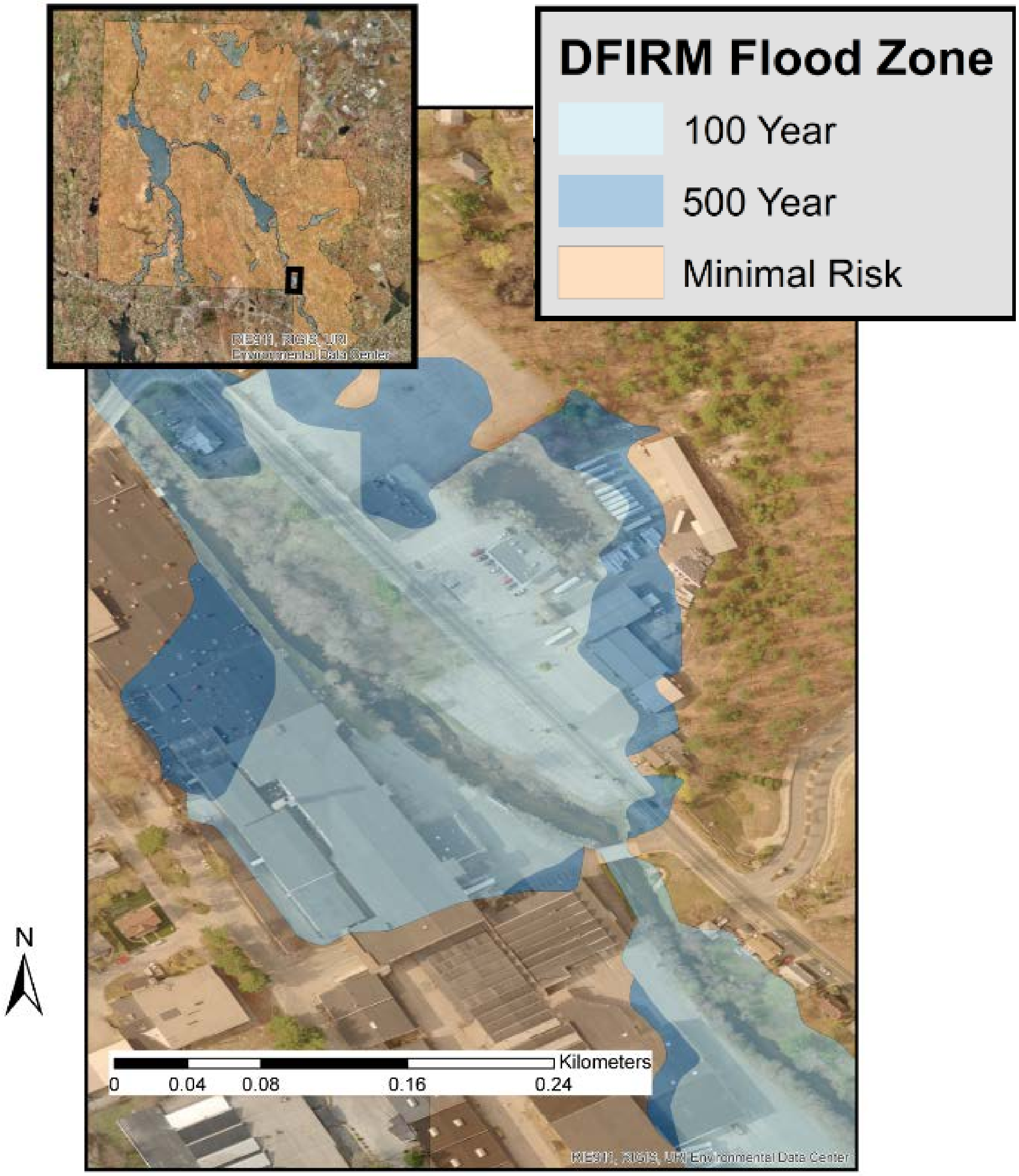


Figure H-6: RIGIS imagery with DFIRM to identify assets in the 100- and 500-year flood zones as defined by FEMA.

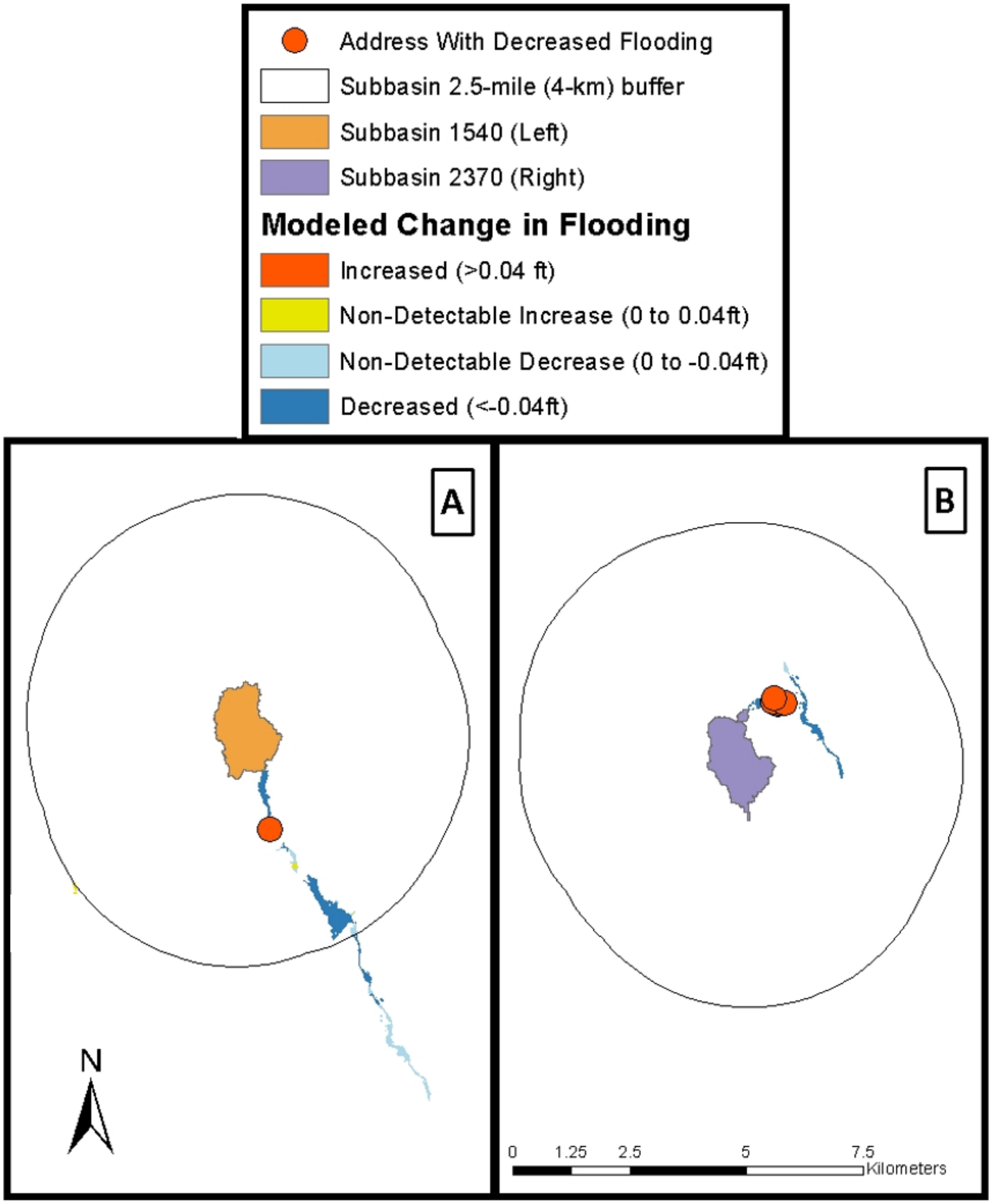


Figure H-7: A 2-mile downstream buffer identifies the same beneficiaries that were counted in the Tier III example assessment.

Appendix I – Tier II Indicator Development

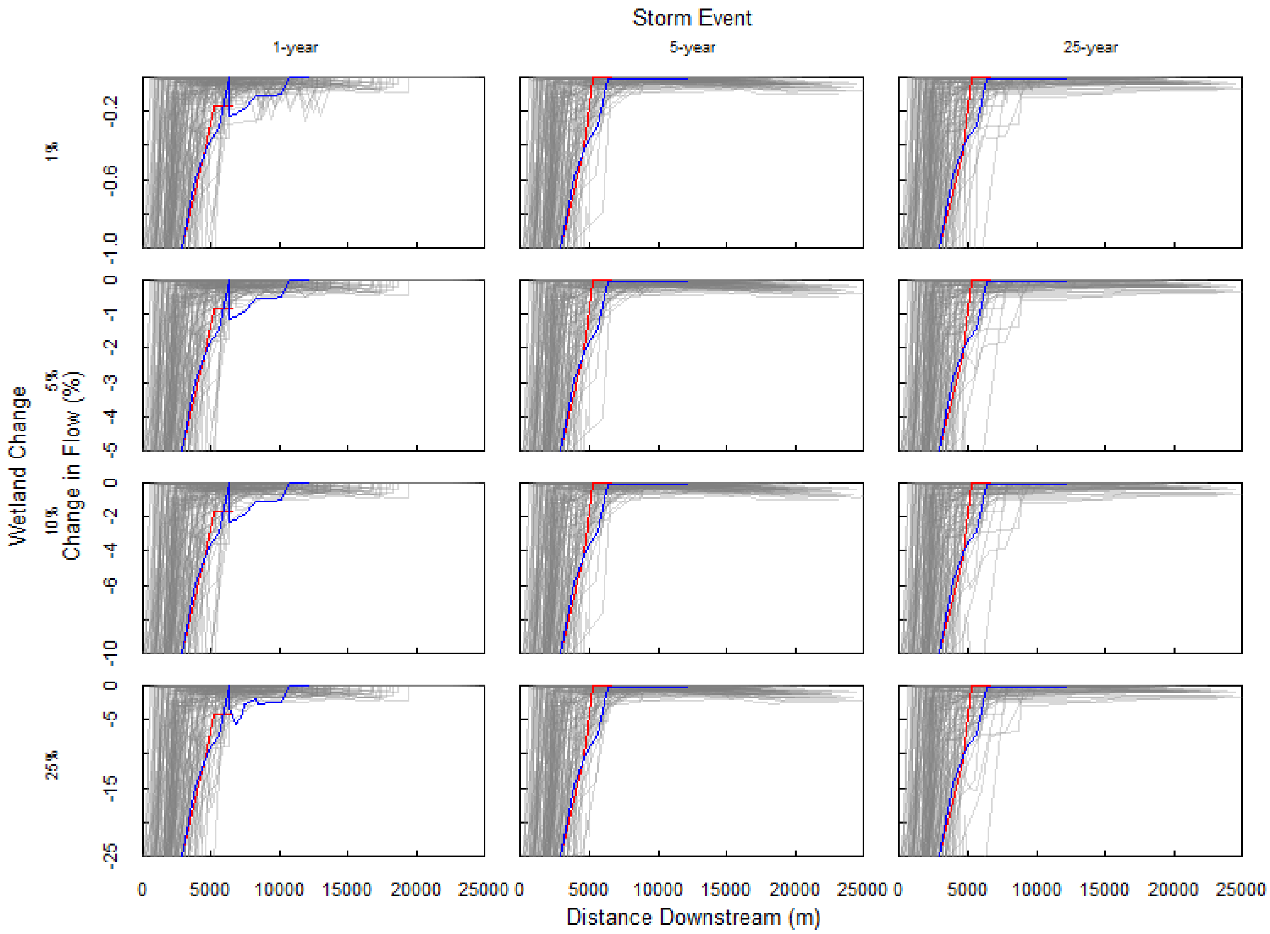


Figure I-1: The change in flow (%) as the distance downstream (m) from the simulated restoration for all subbasins (n=135) for each restoration scenario (wetland change scenarios (1%, 5%, 10%, and 25%) and synthetic storm event (1-year, 5-year, and 25-year)). Two subbasins are shown in red (W2370) and blue (W1540). Note that y-axis vary between sub-plots with the changes in wetland change scenarios.

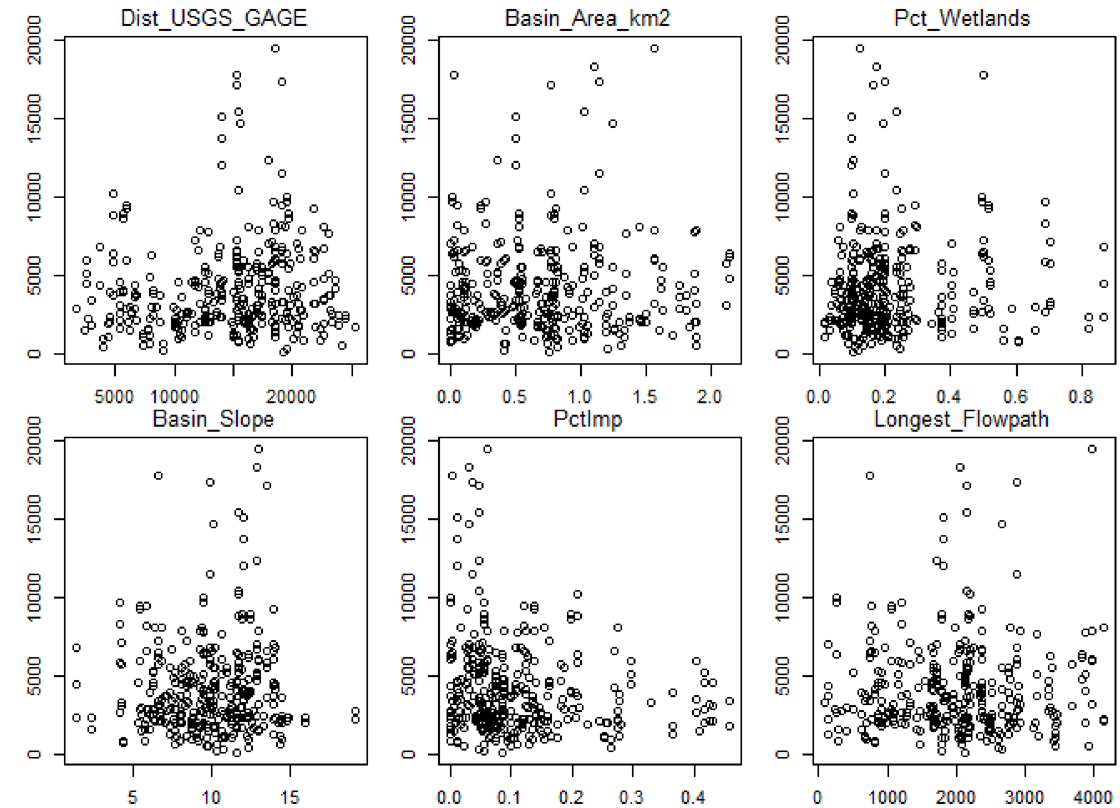


Figure I-2: Downstream distance (m) to no change in flow (<1% change) plotted against subbasin characteristics: (distance to the outlet (m), subbasin area (km^2), % wetlands, mean basin slope (degrees), % impervious cover, and longest flowpath (m)).

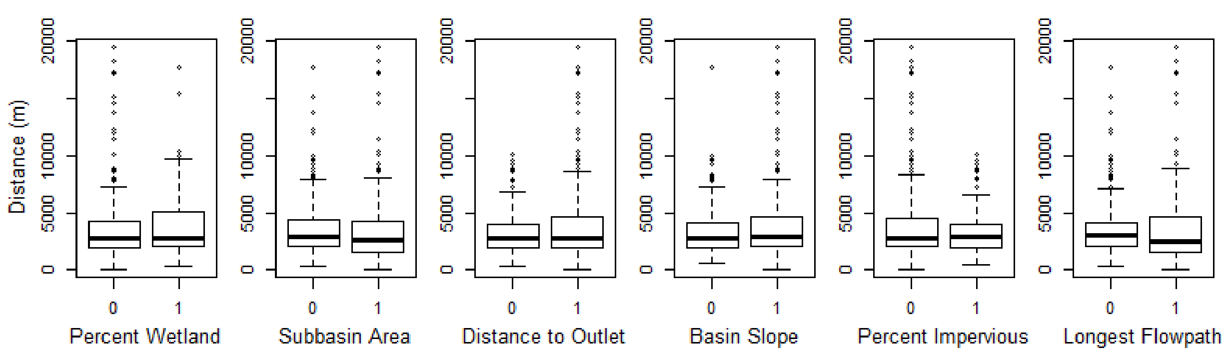


Figure I-3: Boxplots of downstream distance (m) to no change in flow (<0.2% change) plotted by subbasin characteristics (% wetlands, subbasin area (km^2), distance to the outlet (m), mean basin slope (degrees), % impervious cover, and longest flowpath (m)) split into below (0) and above (1) the mean values for the basin characteristics.

Appendix J – R Script to read in DSS Peak Flows and save as csv

```
#DESCRIPTION: This script pulls in all dss files in a folder, finds the peak flows for specified points and creates
#a table of these peak values. That table can be used to find differences between scenarios.
#########################PACKAGES#########################
install.packages("rJava")
library(rJava)
install.packages("devtools")
dss-rip package
devtools::install_github("eheisman/DSS-Rip",args="--no-multiarch")
library(dssrip)
########################FUNCTIONS#######################
reachTOrun <- function(element, run, type){ #ASSEMBLES THE PATH NAME
  data= paste0("//", element, "/", type, "//1MIN/", run, "/")
  return(data)
}
maxFROMxts <- function(path, file){ #GETS THE PEAK FLOW FROM THE XTS FILE
  XTS = getFullTSC(file, path)
  return (max(XTS))
}
getPeak <- function(elements, run, file, type){ #THIS ONE DOES ALL THE WORK
  file<-opendss(file)
  elements.paths <- sapply(elements, reachTOrun, run, type)
  elements.peaks <- sapply(elements.paths, maxFROMxts, file)
  file$close()
  return(elements.peaks.dt <-data.frame(elements.peaks, row.names = elements))
  #return(names(elements.peaks.dt)<-run)
}
singleMaxFlow <- function(HMS_location, file, run, flows, type){ #BASICALLY JUST CALLS getPeak
  myFile = paste0(HMS_location, file)
  table_ALL=getPeak(flows, run, myFile, type)
  names(table_ALL)<-run
  return(table_ALL)
}
lstMaxFlow <- function(tableName, myRunLst, myFileLst, flows, type){ #BASICALLY JUST CALLS getPeak
across a list
  tableName <- data.frame(row.names = flows)
  i=0
  if (length(myRunLst)==length(myFileLst)){
    for (file in myFileLst){
      i=i+1
      #find peaks for all HMS Elements in flows
      table = getPeak(flows, myRunLst[i], file, type)
      names(table) <- myRunLst[i]
      tableName <- (cbind(tableName, table))
    }
  }
  return(tableName)
}
##########ELEMENT LISTS##########
#all HMS elements in hydrologic order
###COPIED FROM HMS, DSS REQURIES ALL CAPS###
FlowsALL_hydro <-c('W1480', 'D_W1480', 'W1590', 'D_W1590', 'J597', 'R260', 'W1490', 'D_W1490',
'W1580', 'D_W1580', 'J600', 'R530', 'W1790', 'D_W1790', 'W1610', 'D_W1610', 'W1870', 'D_W1870', 'J539',
'R610', 'W1840', 'D_W1840', 'W1900', 'D_W1900', 'J531', 'R630', 'W1810', 'D_W1810', 'W1710', 'D_W1710',
'W1860', 'D_W1860', 'J542', 'R510', 'W1690', 'D_W1690', 'W1850', 'D_W1850', 'J545', 'R620', 'W1880',
```

‘D_W1880', 'W1950’, ‘D_W1950’, ‘J518’, ‘R880’, ‘W1800’, ‘D_W1800’, ‘W2000’, ‘D_W2000’, ‘J506’, ‘R790’,
‘W2090’, ‘D_W2090’, ‘W2070’, ‘D_W2070', 'J483’, ‘R820’, ‘W2050’, ‘D_W2050’, ‘W2110’, ‘D_W2110’, ‘J478’,
‘R870’, ‘W1960’, ‘D_W1960’, ‘W2140’, ‘D_W2140’, ‘J470’, ‘R890’, ‘W2160', 'D_W2160’, ‘W2180’,
‘D_W2180’, ‘J460’, ‘R920’, ‘W2220’, ‘D_W2220’, ‘W2260’, ‘D_W2260’, ‘W2190’, ‘D_W2190’, ‘J440’, ‘R960’,
‘W2230', 'D_W2230',"RES_WATERMAN",'J448’, ‘R940’, ‘W2250’, ‘D_W2250’, ‘W2240’, ‘D_W2240',
"RES_GREENVILLEMILLPOND", 'J443’, ‘R2800’, ‘W2780', 'D_W2780', "RES_KNIGHTMILLPOND”,
“RESKNIGHTMILLPOND",'R2750’, ‘W2770’, ‘D_W2770', "RES_STILLWATERMILLPOND”,
“RESSTILLWATERMILLPND", 'R1010’, ‘W2490’, ‘D_W2490’, ‘J380’, ‘R1200’, ‘W2470’, ‘D_W2470’,
‘W2500’, ‘D_W2500’, ‘W2480’, ‘D_W2480’, ‘J385’, ‘R1160’, ‘W2410', 'D_W2410’, ‘W2400’,
‘D_W2400',"RES_SLACK",'J402’, ‘R1090’, ‘W2380’, ‘D_W2380’, ‘W2390’, ‘D_W2390’, ‘J407’, ‘R2640’,
‘W2630’, ‘D_W2630', "RES_HOPKINSPOND”, “RESHOPKINSPOND",'R2690’, ‘W2680’,
‘D_W2680',"RES_MOWRYPOND”, “RESMOWRYPOND",'R1070’, ‘W2720’, ‘D_W2720', 'W2670’,
‘D_W2670’, ‘J435’, ‘R980’, ‘W2580’, ‘D_W2580',"RES_UPPERSPRAGUE”, “RESUPPERSPRAGUE",'R720’,
‘W2060’, ‘D_W2060’, ‘W2570', 'D_W2570’, ‘J491’, ‘R2550’, ‘W2530’, ‘D_W2530',"RES_LOWSPRAGUE”,
“RESLOWSPRAGUE",'R840’, ‘W2130’, ‘D_W2130’, ‘W2520’, ‘D_W2520', 'J473’, ‘R2990',
"RES_GRANITEMILLPOND", 'W2980’, ‘D_W2980',"RESGRANITEMILLPOND",'R810’, ‘W2830’,
‘D_W2830',"RES_HAWKINSPOND", "RESHAWKINSPOND",'R2890’, ‘W2880’, ‘D_W2880',
"RES_REAPERPOND”, “RESREAPERPOND",'R1240’, ‘W2870’, ‘D_W2870’, ‘W2430’, ‘D_W2430', 'J397’,
‘R1100’, ‘W2200’, ‘D_W2200’, ‘W2210’, ‘D_W2210’, ‘J453’, ‘R2940’, ‘W2930’, ‘D_W2930',
"RES_MOUNTAINDALE”, “RESMOUNTAINDALE", 'R910’, ‘W2920’, ‘D_W2920’, ‘W2970’, ‘D_W2970’,
‘J486’, ‘R660’, ‘W1770’, ‘D_W1770’, ‘W1780’, ‘D_W1780’, ‘J561’, ‘R460’, ‘W1290’, ‘D_W1290', 'W1270’,
‘D_W1270’, ‘J672’, ‘R20’, ‘W1300’, ‘D_W1300’, ‘W1280’, ‘D_W1280’, ‘J669’, ‘R3040’, ‘W3030’,
‘D_W3030',"RES_CESARIOPOND”, “RESCESARIOPOND", 'R3090’, ‘W3080’, ‘D_W3080',
"RES_PRIMROSEPONDLOWER”, “RESPRIMROSEPONDLOWER",'R70’, ‘W3070’, ‘D_W3070’,‘J664', 'R80’,
‘W1260’, ‘D_W1260’, ‘W1330’, ‘D_W1330’, ‘W1340’, ‘D_W1340’, ‘W1350’, ‘D_W1350’, ‘J657’, ‘R100’,
‘W1380’, ‘D_W1380’, ‘J649’, ‘R110', 'W1360’, ‘D_W1360’, ‘J654’, ‘R130’, ‘W1370’, ‘D_W1370’, ‘W1320’,
‘D_W1320’, ‘W1400’, ‘D_W1400’, ‘J644’, ‘R160’, ‘W1390’, ‘D_W1390', 'W1410’, ‘D_W1410’, ‘J641’, ‘R330’,
‘W1420’, ‘D_W1420’, ‘W1470’, ‘D_W1470’, ‘J628’, ‘R190’, ‘W1530’, ‘D_W1530’, ‘W1460’, ‘D_W1460', 'J614’,
‘R200’, ‘W1450’, ‘D_W1450’, ‘W1510’, ‘D_W1510’, ‘J619’, ‘R340’, ‘W1500’, ‘D_W1500’, ‘J580’, ‘R350’,
‘W1430’, ‘D_W1430’, ‘W1640', 'D_W1640’, ‘J588’, ‘R380’, ‘W1660’, ‘D_W1660’, ‘W1650’, ‘D_W1650’, ‘J583’,
‘R420’, ‘W1700’, ‘D_W1700’, ‘W1750’, ‘D_W1750',"RESSTUMPPOND", 'R3200’, ‘W3180’, ‘D_W3180',
"RES_STILLWATERPOND”, “STILLWATERPOND",'R470’, ‘W1540’, ‘D_W1540’, ‘W1440’, ‘D_W1440’,
‘J611’, ‘R300', 'W1560’, ‘D_W1560’, ‘W1550’, ‘D_W1550’, ‘J606’, ‘R430’, ‘W3240’, ‘D_W3240’, ‘W3230’,
‘D_W3230’, ‘J345’, ‘R480’, ‘W3220’, ‘D_W3220', 'W3250’, ‘D_W3250',"RES_CAPRON",'J342’, ‘R560’,
‘W1830’, ‘D_W1830’, ‘W1820’, ‘D_W1820’, ‘J550’, ‘R580’, ‘W1930’, ‘D_W1930’, ‘W1920', 'D_W1920’, ‘J524’,
‘R640’, ‘W1680’, ‘D_W1680’, ‘W1600’, ‘D_W1600’, ‘J577’, ‘R410’, ‘W1520’, ‘D_W1520’, ‘W1570’,
‘D_W1570’, ‘J603', 'R400’, ‘W1720’, ‘D_W1720’, ‘W1620’, ‘D_W1620’, ‘J568’, ‘R650’, ‘W1670’, ‘D_W1670’,
‘W1940’, ‘D_W1940’, ‘J521’, ‘R690’, ‘W1890’, ‘D_W1890', 'J534’, ‘R700’, ‘W1910’, ‘D_W1910’, ‘W1970’,
‘D_W1970’, ‘J513’, ‘R710’, ‘W2030’, ‘D_W2030’, ‘W2020’, ‘D_W2020',"RES_GEORGIAVILLE", 'J499’,
‘R770’, ‘W2010’, ‘D_W2010’, ‘W2040’, ‘D_W2040’, ‘J496’, ‘R850’, ‘W2150’, ‘D_W2150’, ‘W2100’,
‘D_W2100’, ‘J467’, ‘R990’, ‘W2170', 'D_W2170’, ‘W2290’, ‘D_W2290’, ‘J432’, ‘R1040’, ‘W2330’, ‘D_W2330’,
‘W2320’, ‘D_W2320’, ‘J422’, ‘R1060’, ‘W2370’, ‘D_W2370’, ‘W2360', 'D_W2360’, ‘J412’, ‘R1050’, ‘W2310’,
‘D_W2310’, ‘W2300’, ‘D_W2300’, ‘J427’, ‘R1130’, ‘W2340’, ‘D_W2340’, ‘W2350’, ‘D_W2350’, ‘J417',
'R1150’, ‘W2450’, ‘D_W2450’, ‘W2440’, ‘D_W2440’, ‘J392’, ‘R3140’, ‘W3130’, ‘D_W3130',
"RES_GREYSTONEDAM”, “RESGREYSTONEDAM", 'R1220', 'W3120’, ‘D_W3120’, ‘USGS_GAGE')

#List with the correct order to import directly into RAS both FLOW and COMBINE data
Full_RAS <- c('J577', 'R2890', 'R1240', 'J397', 'J453', 'R910', 'R720', 'J491', 'R840', 'J380', 'J385', 'R1090', 'J407',
'R2690', 'R1070', 'J440', 'J506', 'J483', 'J478', 'J542', 'J545', 'J412', 'J534', 'J603', 'J568', 'J611', 'J606', 'J597', 'J600',
'J539', 'J531', 'J518', 'J470', 'J460', 'R940', 'R2800', 'R2750', 'R1010', 'J435', 'J473', 'R810', 'J486', 'J628', 'J614', 'J619',
'J580', 'J649', 'J672', 'J669', 'R3090', 'R1220', 'J657', 'J654', 'J644', 'J641', 'J588', 'J583', 'R3200', 'R470', 'J345', 'R560',
'J524', 'J521', 'J513', 'R770', 'J496', 'J467', 'J432', 'J427', 'J392', 'R70', 'USGS_GAGE')

#list of Flows to subset
RAS_FLOW <-c('J577’, ‘J397’, ‘J453’, ‘J491’, ‘J380’, ‘J385’, ‘J407’, ‘J440’, ‘J506’, ‘J483’, ‘J478’, ‘J542’, ‘J545’,
‘J412’, ‘J534’, ‘J603’, ‘J568’, ‘J611', 'J606’, ‘J597’, ‘J600’, ‘J539’, ‘J531’, ‘J518’, ‘J470’, ‘J460’, ‘J435’, ‘J473’,

```
‘J486’, ‘J628’, ‘J614’, ‘J619’, ‘J580’, ‘J649’, ‘J672’, ‘J669', 'J657’, ‘J654’, ‘J644’, ‘J641’, ‘J588’, ‘J583’, ‘J345’,
‘J524’, ‘J521’, ‘J513’, ‘J496’, ‘J467’, ‘J432’, ‘J427’, ‘J392’, ‘USGS_GAGE')
#list of reach combines to subset
RAS_Combine <-c('R2890’, ‘R1240’, ‘R910’, ‘R720’, ‘R840’, ‘R1090’, ‘R2690’, ‘R1070’, ‘R940’, ‘R2800’,
‘R2750’, ‘R1010’, ‘R810’, ‘R3090’, ‘R1220’, ‘R3200’, ‘R470', 'R560’, ‘R770’, ‘R70')

####PARAMETERS AND IMPLEMENTATION#####
HMS_location = "C:\\HMS\\Outputs\\FinalScenario\\"
####single file####
run = "RUN:MARCH_2001_HEAD_MW_MIDCALIB_01_WET"
file = "ResultsAll.dss"
flows = FlowsALL_hydro
#Run baseline analysis
baselineTable_ALL<-singleMaxFlow(HMS_location, file, run, flows, "FLOW")
baselineTable_Combine<-singleMaxFlow(HMS_location, file, run, RAS_Combine, "FLOW-COMBINE")
####file list####
runConvention = "RUN:MARCH_2001_HEAD_MIDCALIB_01_SCENARIO_"
runNumber = 1:139
#1% scenario
dssFileLst1 = 1:139
myFileLst1 <- paste0(HMS_location, "Opt_Results_", dssFileLst1, ".dss")
runConvention1 = paste0(runConvention, "1", "_")
#10% scenario
dssFileLst10 = 279:417
myFileLst10 <- paste0(HMS_location, "Opt_Results_", dssFileLst10, ".dss")
runConvention10 = paste0(runConvention, "10", "_")
#25% scenario
dssFileLst25 = 418:556
myFileLst25 <- paste0(HMS_location, "Opt_Results_", dssFileLst25, ".dss")
runConvention25 = paste0(runConvention, "25", "_")
#5% scenario
dssFileLst5 = 140:278
myFileLst5 <- paste0(HMS_location, "Opt_Results_", dssFileLst5, ".dss")
runConvention5 = paste0(runConvention, "5", "_")
myRunLst10 <- paste0(runConvention10, runNumber)
myRunLst25 <- paste0(runConvention25, runNumber)
myRunLst5  <- paste0(runConvention5, runNumber)
myRunLst1  <- paste0(runConvention1, runNumber)
#Run analysis
scenario10_table_All <- lstMaxFlow(scenario10_table_All, myRunLst10, myFileLst10, FlowsALL_hydro,
"FLOW")
scenario25_table_All <- lstMaxFlow(scenario25_table_All, myRunLst25, myFileLst25, FlowsALL_hydro,
"FLOW")
scenario5_table_All <- lstMaxFlow(scenario5_table_All, myRunLst5, myFileLst5, FlowsALL_hydro, "FLOW")
scenario1_table_All <- lstMaxFlow(scenario1_table_All, myRunLst1, myFileLst1, FlowsALL_hydro, "FLOW")
#Run FLOW-Combine
scenario10_table_combined <- lstMaxFlow(scenario10_table_All, myRunLst10, myFileLst10, RAS_Combine,
"FLOW-COMBINE")
scenario25_table_combined <- lstMaxFlow(scenario25_table_All, myRunLst25, myFileLst25, RAS_Combine,
"FLOW-COMBINE")
scenario5_table_combined <- lstMaxFlow(scenario5_table_All, myRunLst5, myFileLst5, RAS_Combine, "FLOW-
COMBINE")
scenario1_table_combined <- lstMaxFlow(scenario1_table_All, myRunLst1, myFileLst1, RAS_Combine, "FLOW-
COMBINE")
#subset RAS flows
scenario10_RAS_FLOW <-scenario10_table_All[RAS_FLOW,]
```

```
scenario25_RAS_FLOW <-scenario25_table_All[RAS_FLOW,]
scenario5_RAS_FLOW <-scenario5_table_All[RAS_FLOW,]
scenario1_RAS_FLOW <-scenario1_table_All[RAS_FLOW,]
#Join the two preppared for RAS
scenario10_RAS <- rbind(scenario10_table_combined, scenario10_RAS_FLOW)
scenario10_RAS<- scenario10_RAS[match(Full_RAS, row.names(scenario10_RAS)),]
scenario25_RAS <- rbind(scenario25_table_combined, scenario25_RAS_FLOW)
scenario25_RAS<- scenario25_RAS[match(Full_RAS, row.names(scenario25_RAS)),]
scenario5_RAS <- rbind(scenario5_table_combined, scenario5_RAS_FLOW)
scenario5_RAS<- scenario5_RAS[match(Full_RAS, row.names(scenario5_RAS)),]
scenario1_RAS <- rbind(scenario1_table_combined, scenario1_RAS_FLOW)
scenario1_RAS<- scenario1_RAS[match(Full_RAS, row.names(scenario1_RAS)),]
####Write to csv####
stormRun="spring01"
#raw flows
write.csv(scenario10_table_All, file=paste0(HMS_location, "Flows_scenario10_All_", stormRun,".csv"))
write.csv(scenario25_table_All, file=paste0(HMS_location, "Flows_scenario25_All_", stormRun,".csv"))
write.csv(scenario5_table_All, file=paste0(HMS_location, "Flows_scenario5_All_", stormRun,".csv"))
write.csv(scenario1_table_All, file=paste0(HMS_location, "Flows_scenario1_All_", stormRun,".csv"))
######baseline to CSV#####
baselineFLOW <- data.frame(baselineTable_ALL[RAS_FLOW,], row.names = RAS_FLOW)
colnames(baselineFLOW)<-"RUN:MARCH_2001_HEAD_MW_MIDCALIB_01_WET"
scenarioBaseline <- rbind(baselineTable_Combine, baselineFLOW)
write.csv(scenarioBaseline[Full_RAS,], file=paste0(HMS_location, "Baseline_", stormRun,".csv"))
#pct change from baseline
baseline = baselineTable_ALL$"RUN:MARCH_2001_HEAD_MW_MIDCALIB_01_WET"
pctChange_scenario10_all <- ((scenario10_table_All-baseline)/baseline)*100
write.csv(pctChange_scenario10_all, file=paste0
        (HMS_location, "pctChange_scenario10_All_", stormRun,".csv"))
pctChange_scenario25_all <- ((scenario25_table_All-baseline)/baseline)*100
write.csv(pctChange_scenario25_all, file=paste0
        (HMS_location, "pctChange_scenario25_All_", stormRun,".csv"))
pctChange_scenario5_all <- ((scenario5_table_All-baseline)/baseline)*100
write.csv(pctChange_scenario5_all, file=paste0
        (HMS_location, "pctChange_scenario5_All_", stormRun,".csv"))
pctChange_scenario1_all <- ((scenario1_table_All-baseline)/baseline)*100
write.csv(pctChange_scenario1_all, file=paste0
        (HMS_location, "pctChange_scenario1_All_", stormRun,".csv"))
#raw RAS tables
write.csv(scenario10_RAS, file=paste0(HMS_location, "Flows_scenario10_RAS_", stormRun,".csv"))
write.csv(scenario25_RAS, file=paste0(HMS_location, "Flows_scenario25_RAS_", stormRun,".csv"))
write.csv(scenario5_RAS, file=paste0(HMS_location, "Flows_scenario5_RAS_", stormRun,".csv"))
write.csv(scenario1_RAS, file=paste0(HMS_location, "Flows_scenario1_RAS_", stormRun,".csv"))
```

Appendix K – Data Quality and Limitations

Information on data quality, as well as limitations on the use of model and indicator results, have been incorporated into the main text where appropriate. Here, we compiled all the relevant data quality and limitation information in one place.

Spatial datasets were selected to optimize spatial resolution, temporal relevance, and accuracy (Bousquin et al. 2014). Metadata for all spatial datasets are available from RIGIS. Of particular significance to results are accuracy of the Digital Elevation Model (DEM) and E911 address points. The DEM had a 0.01-m vertical precision and an RMSE of 0.067 m based on raw LiDAR calibration control points. Based on this vertical precision, 0.012 m was used as a "detection threshold" for our hydraulic model, since lesser changes in flooding could be attributed to error. E911 addresses have a horizontal accuracy of 5-10 m and may be as old as 2001. This dataset is updated frequently as structures change and better data become available. The most recent update (December 31, 2014) available at the time of analysis was used to demonstrate indicators. This analysis is intended to demonstrate the indicator development approach and to develop general indicators (Tier II) of flood protection by freshwater wetlands. This analysis is not a formal flood zone delineation or an actual assessment of structures currently at risk of flooding.

All hydrologic and meteorologic time series data were obtained from USGS (station 01114500) and NOAA (COOP station 376698) and were subject to their quality assurance standards. Any provisional or incomplete flow data, such as those for the March 2010 flood, were rejected for use in calibration or validation of the hydrologic model. Flow data during prolonged cold periods that could be erroneous due to ice effects, such as those for the December 2008 flood, were rejected for use in calibration or validation of the hydrologic model. We did not model the entire watershed, because the area below the USGS gage used for model calibration is highly urbanized and lacks information on stormwater infrastructure, which is likely to lead to errors in predicting flood extent.

We evaluated hydrologic model fit using comparison to gauged storm peak flows, Nash-Sutcliffe (N-S) efficiency, and Root Mean Square Error (RMSE) statistics. Two models were calibrated, one for dry conditions (October 2005) and one for wet conditions (March 2001). These two models were not expected to cross validate well, representing opposing antecedent moisture conditions. The March 2001 calibrated model had an N-S of >0.5 for the storm calibrated and the April 2005 validation storm, an N-S of approximately 0 for the 2007 Validation storm, and an N-S <0 for the October 2005 storm. This suggests the calibrated model performs well with wetter antecedent moisture conditions and larger storms, is less satisfactory for smaller storms (April 2007 had a flow recurrence interval of 3.4 years), and unfit for dry antecedent moisture conditions (October 2005).

Sensitivity Analysis showed the hydrologic model was particularly responsive to precipitation and assumptions of divergence of runoff to wetlands based on wetland area. The precipitation station (NOAA COOP #376698) is outside the modeled basin and 15.9 km south of the USGS gage station. The assumptions behind treatment of runoff diverted to wetlands received particular attention in our methods and analysis. These assumptions should be considered when using the model or Tier II indicators developed here.

Simulated water levels from the hydraulic model obtained from USGS were previously compared to 2010 flood high-water marks in Zarriello et al. 2013. Based on those comparisons, the model was considered valid, provided structures and stream channels remained clear of debris. However, that hydraulic model was adapted for use in this study. The adapted hydraulic model included additional cross section and stream reach geometry and flows from hydrographs from the hydrologic model. The hydraulic model and derived flood maps were used for Tier III indicator demonstration purposes only and should not be used as an actual assessment of actual flood risk. Given this use of the adapted hydraulic model, no further validation was performed.

Tier II indicators of downstream distances for flow of flood benefits were based on changes in flow in simulated restoration scenarios. The number of simulations necessitated automation, and results were checked for quality. Of the 139 subbasins, flows from four subbasins did not correspond to the restoration scenario in one or more simulations (W2980, W1830, W1660, and W2360). To be consistent, those four subbasins were removed from the dataset, reducing the number of restoration simulations to 1,620.

There were two further quality restrictions on individual data points within simulations. First, no percent change in flow should show an increased flow and, second, no percent change in flow should exceed the percent restoration, based on the way that restorations were implemented in the model (i.e. 5% scenario results in a 5% reduction of flow). The number of instances where these errors (1-Increased or 2- Exceed) were greater than 0.1% are shown in Table K-1. These errors are extremely low, considering there are 2,484 data points in each of the 12 basin scenarios. Closer inspection of these errors showed they typically originated from flows leaving dams. These errors were assumed to be non-systemic given their rarity (e.g. in the scenarios with the most errors, 25% restoration and 25-year storm scenarios, only <0.006% of data points had increased flow), and only individual data points were removed from consideration.

Table K-1: Single reach data points where percent change was outside of the expected bounds.

	1-Year		**5-Year**		**25-Year**	
Restoration	**Increased**	**Exceed**	**Increased**	**Exceed**	**Increased**	**Exceed**
1%	0	3	0	0	8	5
5%	0	4	0	2	6	5
10%	0	5	0	2	9	4
25%	0	3	0	1	11	5

We expect that Tier II indicators developed in this study could be used in similar watersheds in the Northeast, but until similar modeling is conducted in other watersheds, it is difficult to say how transferable the indicators are. Downstream distances for benefits delivery did not appear to vary greatly within the range of subbasin areas and imperviousness explored. Other factors such as stream morphology and subbasin slope may play a role in this as well. Although we were unable to develop generalized indicators quantifying the role of substitutes, such as dams, in decreasing wetland benefits, it is possible the number of dams in the Woonasquatucket influenced our results for the downstream distance of benefits.