Assessing the Impact of the Urban Tree Canopy on Streamflow Response: An Extension of Physically Based Hydrologic Modeling to the Suburban Landscape

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ABSTRACT

TAMARA MITTMAN: Assessing the Impact of the Urban Tree Canopy on Streamflow Response: An Extension of Physically Based Hydrologic Modeling to the Suburban Landscape

(Under the direction of Lawrence E. Band)

This work examines the impact of land cover composition and pattern on catchment hydrologic response in an ungauged suburban catchment in Baltimore, MD. Field data collected by the Baltimore Ecosystem Study (BES) are integrated with the Regional Hydro-Ecologic Simulation System (RHESSys) to develop models of the study catchment and a nearby reference catchment. A proxy-catchment calibration method is applied to calibrate model parameters, and the Generalized Likelihood Uncertainty Estimation (GLUE) method is applied to assess model uncertainty. To examine the impact of urban tree canopy on catchment hydrologic response, four vegetation management scenarios are simulated. Results suggest that parameter transfer from a forested reference catchment to an ungauged suburban catchment is viable for lightly urbanized catchments, and indicate that the extent of the urban tree canopy is a key determinant of streamflow response. Results also demonstrate the importance of preserving upland as well as riparian forest in maintaining ecosystem function.

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1. Introduction

This thesis examines the impact of land cover composition and pattern on urban hydrologic response. Though planning practice assumes a relationship between urban pattern and aquatic ecosystem function, scientific knowledge of this relationship is limited (Alberti, 2005). Extensive research has documented the impacts of land cover composition on hydrologic response and examined the mechanisms through which these impacts are generated, but the interaction of these mechanisms with land cover pattern remains poorly understood. To advance our understanding of hydrologic processes and pathways in the urban environment, this study explores the hydrologic response of a suburban catchment in Baltimore, Maryland to different patterns of vegetation. We integrate field data collected by the Baltimore Ecosystem Study, part of the Long Term Ecological Research (LTER) network established by the National Science Foundation, with a distributed ecohydrologic simulation system to develop models of the study catchment and a nearby reference catchment.

This research consists of two components. The first involves the calibration and validation of the Regional Hydro-Ecologic Simulation System (RHESSys) for an ungauged suburban catchment in Baltimore, Maryland. We apply a proxy-catchment method to calibrate the model, calibrating model parameters for a nearby forested reference catchment and transferring the calibrated parameters to the study catchment.

The second component involves the application of RHESSys to a set of vegetation management scenarios. The impact of vegetation management on aggregated and distributed hydrologic response is discussed. For the aggregated response, the impact of land cover manipulation is compared to the uncertainty associated with parameter estimation. Finally, the implications for watershed planning are discussed.

2. Background

a. Significance

Research has demonstrated that urban development dramatically alters catchment hydrologic response. Urban development alters the hydrologic cycle by armoring the landscape with pavement and rooftops and altering soils and vegetation (Walsh 2005a, Walsh 2005b, Endreny 2005, Riley 1998). These changes reduce infiltration and evapotranspiration and increase runoff volumes. Urban drainage systems amplify these changes by conveying all runoff to the nearest water body, producing flashy flow regimes with high peak flows.

Changes in catchment hydrologic response have generated a host of unanticipated consequences. Among the consequences that most directly affect humans are increased flooding, stream channel erosion, and damage to infrastructure along streams. Other consequences include reduced groundwater recharge, increased transport of pollutants (including sediment, nutrients, pathogens and heavy metals), and degradation of riparian and aquatic habitat. In the early 1970s, detention and retention basins were introduced into the urban drainage system in an attempt to minimize damage caused by peak flows (Endreny 2005). Studies suggest, however, that 1) basin designs often fail to provide the intended control of peak flow, 2) even when basins mitigate peak flows at the site scale, they may fail to mitigate them at the regional scale, and 3) because basins address only the runoff rate and not the runoff volume, they cannot prevent stream channel erosion and the associated economic and ecological impacts (Booth and Jackson 1997, Emerson 2005, Endreny 2005, Walsh 2005b).

As the urbanization of the American landscape continues, damage to aquatic ecosystems will likely follow. According to the USDA's Natural Resource Inventory, the area of developed land in the United States (defined as "large urban and built-up areas, small built-up areas, and rural transportation land") increased by ~48% between 1982 and 2003 – an area approximately equal to that of the state of New York. Most of this development occurred on land that was previously forested or farmed, and much of it created new suburbs (Alig 2004, Brown 2005, Theobald 2005). The pace at which we are transforming the landscape exceeds the rate of population growth. Whereas the area of urban, suburban, or exurban land uses increased by an average of 1.6% per year between 1980 and 2000, the population increased by an average of only 1.18% per year (Theobald 2005).

To minimize the hydrologic and ecological impacts of urbanization, federal and state governments have adopted new stormwater legislation, while planners and engineers have developed new approaches to stormwater management. At the federal level, the 1987 amendments to the Clean Water Act expanded the National Pollutant Discharge Elimination System (NPDES) program to include discharges from municipal stormwater systems. At the state level, states from coast to coast have adopted legislation regulating the quantity and quality of urban stormwater. The Maryland Stormwater Act of 2007, for instance, states as its goals the reduction of local flooding, the reduction of stream channel erosion, the maintenance of predevelopment hydrology (including groundwater recharge and baseflows), the reduction of pollution, and the reduction of siltation and sedimentation. To achieve stormwater management goals, state legislation generally includes an associated set of standards informed by science. Because research has suggested that a large proportion of stream channel erosion occurs at an "effective discharge" approximately equal to the bankfull flow, most state legislation requires the control of runoff *rates* to maintain the frequency of a design flow (Doyle et al. 2002). Because more recent research has suggested that frequent, smaller events may be more important causes of channel incision than infrequent, larger events, recent legislation often requires the control of runoff *volumes* as well as rates (Walsh 2005b). Other stormwater standards vary significantly from state to state, but often include constraints on pollutant loads and annual recharge volumes.

A gradual transition in the principles and practice of stormwater management has accompanied the changes in state and federal standards. Whereas stormwater management was once the domain of engineers who developed centralized, "end-of-the-pipe" facilities to control property damage from large infrequent storms, in recent years the objectives of stormwater management have evolved to address a more diverse set of impacts across a broader range of spatial and temporal scales (British Columbia Ministry of Water, Land, and Air Protection, 2002). In the United States, many planning professionals refer to this comprehensive approach as Low Impact Development (LID). LID seeks to minimize the hydrologic and ecological impacts of urban development by addressing impacts at the regional and catchment scales as well as the site scale, and by addressing flow regimes as well as peak flows. To address impacts at larger spatial scales, LID identifies and preserves sensitive areas, confining all development to a "development envelop" (Prince George's County, 2000). Sensitive areas include variable source areas, riparian areas, wetlands, areas with steep slopes, and areas with high permeability soils. To mimic predevelopment hydrology across a range of precipitation events and soil moisture conditions, LID applies distributed as well as centralized practices that increase infiltration and evapotranspiration as well as storage. These practices include the minimization of impervious cover, the management of urban vegetation, and the installation of "soft-engineering" facilities such as rain gardens, grassed swales, and green roofs.

Though science and policy have converged on the objective of mitigating the hydrologic impact of urban development, scientific knowledge at the scale and resolution demanded by urban planning remains poorly developed. Planners operate across large scales, developing plans for entire towns, cities and counties. While much research has examined the impacts of development at the catchment scale, research on the effectiveness of LID techniques is largely confined to the site scale (Dietz and Clausen, 2008). Within the boundaries of the town, city, or county, planners are expected to develop spatially explicit plans. In contrast, scientific research examining the impacts of conventional development and the effectiveness of LID in mitigating these impacts has generally neglected the role of spatial position and urban pattern. We select one urban development intensity (suburban development) and one suggested management technique (vegetation management) to begin to provide insight into the mechanisms through which urban pattern affects catchment hydrologic response.

b. Review of empirical studies of urbanization impacts on hydrology

An extensive review of the literature indicated that empirical studies of the impact of land cover change on urban hydrologic response generally examine land cover composition, rather than land cover pattern, and the significance of impervious cover, rather than the significance of vegetation type. Decades of empirical research have documented relationships between the extent of impervious cover within catchments and various measures of stream health. Studies have noted dramatic changes in flow regime (Konrad 2001, Burns 2005, Chang 2007, Changnon 1996, Dow 2007, Jennings 2002, Rose 2001), channel geomorphology (Hammer 1972, Doll et al 2002), pollutant loading and timing (Griffin 1980, Shields 2006, 2008), habitat quality (Cianfrani 2006), and biological assemblages (Klein 1979, Moore and Palmer 2005, Morley 2002, Strayer 2003, Snyder 2003) as impervious cover within catchments increases. Though earlier research noted a minimum threshold below which ecosystem degradation was negligible (Booth and Jackson 1997, Arnold and Gibbons 1996, Klein 1979), more recent research attributes this threshold to measurement imprecision and demonstrates a continuous decline in measures of biological integrity as % imperviousness exceeds zero (Walsh et al 2005a, Booth et al. 2004, Booth et al 2002, Moore and Palmer 2005, Karr and Chu 2000, May and Horner 2000, Booth et al., 2001).

While several empirical studies have documented improvements in stream health as percent forest cover within urbanized catchments increases, we found no studies examining the impact of vegetation type on catchment hydrologic response. In his classic analysis of the relationships between stream channel enlargement and land cover in urbanized watersheds, Hammer found land in forest to have a negative relationship to channel enlargement (1972). Research relating forest and impervious cover to measures of stream biotic integrity has consistently demonstrated that both land covers are important predictors of stream health, observing measures of biotic integrity to increase with forest cover and decrease with impervious cover (Goetz and Fiske 2008, Carlisle and Meador 2007, Strayer 2003, Steedman 1988). In his study of 10 catchments in southern Ontario, Steedman not only found basin Index of Biotic Integrity (IBI) scores to be directly related to forest cover and inversely related to urban land cover, but noted a greater impact on biotic integrity per increment change in forest cover. It is hoped that the present study will elaborate upon this research to provide insight into the role of urban grasses as well as trees in shaping catchment hydrologic response.

Previous research indicates that the mechanisms through which changes in land cover degrade stream health are largely driven by changes in catchment hydrology. Removal of upland and riparian vegetation and addition of impervious cover and drainage systems transform land-water linkages, reducing interception, evapotranspiration, infiltration, and groundwater recharge, and increasing volumes and rates of surface flow. These shifts lead to less stable flow regimes and enhanced delivery of pollutants; simplification of stream channels and reduction in water quality; and, ultimately, reduced biologic integrity (Moore Palmer 2005, Allan 2004, Snyder 2003). Again while several empirical studies have addressed the impact of urban forests on catchment hydrologic processes, none have explicitly addressed the relative impact of different types of vegetation.

In recent years, interest has increased in the impact of land cover pattern, as well as extent, on hydrologic and ecosystem response (Alberti 2005, King 2005). Recent research into the effects of urbanization on aquatic ecosystem function has examined several components of landscape pattern, including: 1) the connectivity of impervious cover to stream channels, 2) the composition of land cover within the riparian corridor, 3) the distance of land covers from the stream channel, and 4) the size of land cover patches. The most extensively studied components to date are the connectivity of impervious cover and the composition of riparian land cover. Empirical studies of the relationships between catchment physical characteristics and various measures of ecosystem function have consistently found that ecosystem function is better predicted by the extent of connected impervious cover than by the extent of all impervious cover (Newall and Walsh 2005, Taylor et al 2004, Hatt et al 2004, Hammer 1972). Laboratory simulations of rainfall on various arrangements of pervious and impervious surfaces have also shown impervious connectivity to have a significant impact on runoff volume (Pappas 2008, Shuster 2008). In the experiments of Pappas and Shuster (2008), upslope impervious cover initially produced less runoff than downslope impervious cover, but this difference was observed to narrow or even reverse with continued rainfall. Despite these and other recent advances, however, scientific understanding of the mechanisms through which spatial arrangement shapes the impact of land cover change remains limited.

Extensive scientific research has also accumulated on the impact of riparian forests on ecosystem function. In constructing empirical models of stream biotic integrity, many researchers have examined the relative predictive power of catchment-wide versus riparian land covers. Their conclusions are inconsistent. Several authors have found that riparian land cover is a significant predictor of in-stream habitat but not fish biological assemblages, suggesting that alterations in flow regime and reductions in water quality overwhelm the capacity of riparian vegetation to maintain biological integrity (Snyder 2003, Strayer 2003). In contrast, other authors have found that riparian forests protect invertebrate diversity even in catchments with substantial urbanization (Carlisle and Meador 2007, Moore and Palmer 2005, Steedman 1988).

Research has only recently become available on the impact of landscape position and land cover aggregation on aquatic ecosystem function. Perhaps the earliest study to assess the impact of landscape position was Hammer's classic analysis of channel enlargement (1972). Based on his analysis of 72 small catchments near Philadelphia, PA, Hammer found significant interactions between the impact of impervious development on channel size, topographic characteristics of the catchment, and the location of impervious development within the catchment. Hammer observed the distance of impervious cover from the stream channel to have a significant influence on channel enlargement, and found this influence to be highly dependent on slope. Subsequent studies based on empirical models have stated their conclusions in less certain terms. Both King (2005) and Goetz and Fiske (2008) included distance-weighted variables in their assessments of land cover variables as predictors of stream biotic integrity. King et al found that weighting of developed land by distance from the sampling station provided better predictions of biotic integrity than land cover percentages alone. Goetz and Fiske found that weighting of land covers by distance from the stream increased model performance, but noted that the distance weighting scheme that was most effective integrated tree cover density and distance from the stream. Alberti et al (2007) applied landscape ecology metrics to examine the impact of land cover pattern on stream biotic integrity. Their research found that mean patch size of impervious areas and mean patch size of forested areas explained much of the variability in stream biotic integrity, but were so highly correlated with the amount of impervious area that no conclusions could be drawn. Some empirical studies have found that the explanatory power of land cover composition variables declines in smaller catchments, suggesting that the spatial arrangement of land covers becomes more important at smaller scales (King 2005, Strayer 2003). Whereas significant research has examined the mechanisms through which the connectivity of impervious cover and the composition of riparian land cover impact stream health, little is known about the mechanisms through which landscape position of different land covers impact stream health.

c. Review of modeling studies of urbanization impacts on hydrology

Modeling studies of the impact of urbanization on catchment hydrologic response extend knowledge acquired through empirical research by: 1) predicting the impacts of future urban development on hydrologic response, and 2) providing insight into the mechanisms through which development impacts hydrologic response. In the first case, empirical research is difficult if not impossible because measurements (either of the pre-developed past or developed future) are often unavailable, while in the second case empirical research is possible, but so many measurements would be required to properly account for spatial and temporal heterogeneity in catchment characteristics and climate variables that empirical research becomes prohibitively costly and complex (Cuo et al 2008).

The structure of the hydrologic models most commonly applied to the simulation of urban catchments has confined most research to the analysis of land cover composition (rather than pattern) and impervious land cover (rather than vegetation). Refsgaard (1996) identified three model structures commonly applied in hydrologic simulation: 1) empirical black box, 2) lumped conceptual, and 3) distributed physically based. The vast majority of the modeling systems applied to the simulation of land cover change in urban catchments belongs to the second class. Lumped conceptual models partition catchments into hydrologically similar areas and attempt to represent hydrological processes by calculating fluxes of water and mass to and from these areas. Though the entire constellation of urban hydrologic models characterized by this structure cannot possibly be examined here, two representative examples will be discussed to illustrate the constraints associated with this structure. Many lumped conceptual models applied to the prediction of development impacts on catchment hydrology are based on the Soil Conservation Service (SCS) curve number method

(McColl 2007, Girling and Kellet 2002, Bhaduri 2000, Choi 2003, Miller 2002, Tang 2005, Wu 2007). These models generally partition a catchment into areas with similar land covers, assign a set of soil moisture-dependent curve numbers to each land cover, and apply the SCS equation to each land cover to estimate overland flow at each time step. Among the many shortcomings associated with this approach (see Garen and Moore, 2005) is the difficulty of assigning any physical meaning to the empirically-derived "curve numbers" (Beven 1989). Because the curve number lacks physical meaning, it is difficult to select a curve number that reflects patterns of land cover or vegetation processes.

Another model commonly applied to the prediction of the hydrologic impacts of urbanization is the federally-supported HSPF simulation system (Booth et al 2002, Brun and Band 2000). Though HSPF is more process-based than curve number models, its structure still cannot support analysis of the impact of land cover patterns or vegetation processes. In HSPF, segments (or sub-catchments) may be assigned pervious and impervious percentages, but the model cannot account for the arrangement of pervious and impervious areas within sub-catchments and their interaction (such as the re-infiltration of run-off, for example). One study has attempted to analyze the impact of urban vegetation in HSPF, finding that the conversion of forest to lawn was more significant than impervious cover in determining peak discharge increases from exurban catchments (Booth et al 2002). Other researchers, however, suggest that the representation of interception and evapotranspiration processes in HSPF is too crude to support the analysis of vegetation effects (Wang et al 2008).

In recent years, researchers have begun to apply the third model structure identified by Refsgaard – distributed physically based models – to the analysis of urban hydrology to better characterize the variety and distribution of hydrologic processes in urban catchments (Easton 2007). Initial research indicates that distributed physically based models that include representations of impervious cover are able to reproduce stream flow from partially urbanized catchments very well (Cuo 2008, Easton 2007). Research has also demonstrated the potential of distributed models of urbanized catchments to examine the hydrologic impacts of land cover pattern and drainage network configuration. Easton et al (2007) found that the pattern of impervious cover in an urban catchment in upstate New York shaped the distribution of soil moisture and runoff production. Tague and Pohl-Costello (2008) found that drainage network configuration may interact with antecedent soil moisture condition in semi-arid urban catchments to determine streamflow response to precipitation events. To date, only one research effort has explicitly addressed the role of vegetation in determining urban hydrologic response. Wang et al (2008) have developed a semi-distributed physically based model to examine the impact of urban trees on urban hydrologic response. Preliminary research suggests that the model performs well, and that interception and evapotranspiration play significant roles in the urban water balance.

While empirical research on urbanized catchments suggests that both land cover composition and land cover pattern are significant determinants of aquatic ecosystem function, it has not identified the mechanisms through which urban pattern shapes urban hydrologic process. Lumped conceptual models of urban catchments have also provided little insight into the role of urban pattern. Research suggests, however, that distributed physically based models of urbanized catchments have great potential to advance our understanding of the effects of land cover pattern on hydrologic response. The following sections discuss three obstacles that limit the use of hydrologic simulation models to assess the impacts of land cover change in urban catchments.

d. Model calibration in ungauged catchments

Simulation models of urban catchments are often limited by lack of a continuous streamflow record. In the absence of measured streamflow, poorly constrained model parameters cannot be calibrated to reproduce observed streamflow from the study catchment. Several approaches have been suggested for estimating model parameters in the absence of data. The first is the transfer of parameters from a similar catchment for which data is available to the catchment of interest. As Tague and Pohl-Costello (2008) noted, this is the basis of empirical runoff-coefficient models such as the SCS Curve Number method. Researchers have cautioned, however, that storm runoff processes vary significantly from one catchment to another, particularly with changes in climate or catchment physical characteristics (Pilgrim 1983). The similarity of storm runoff processes should therefore be ascertained before parameters are transferred from one catchment to another. A second approach is to simulate catchment response across the range of plausible parameter values (Tague and Pohl-Costello, 2008). Model findings can then be assessed in the context of the sensitivity of the results to model parameters. A third approach is to collect a limited number of streamflow measurements to calibrate the catchment model. Studies conducted as part of the Prediction in Ungauged Basins initiative (PUB) indicate that as few as 6 measurements can be effective in constraining prediction uncertainties (Seibert and Beven 2009).

e. Model validation for studies of land cover change

Even when streamflow records are available for urban catchments, simulation of the effects of future change is limited by the problem of model validation. Refsgaard and Henriksen (2004) describe model calibration as the adjustment of parameter values to reproduce observations, and model validation as the demonstration that the calibrated model performs well *in a context consistent with its intended application*. In recent years, many hydrologists have discussed the importance of selecting validation tests that demonstrate a model's fitness for its intended purpose (Refsgaard 2004, Ewen 1996, Klemes 1986). According to this view of model validation, when the intended application of a model is the prediction of the effects of land cover change, model validation must show that the model can accurately predict hydrologic response for different land covers. Klemes (1986) proposed that the most appropriate test for this model application is the differential split-sample test. Differential split-sample tests involve the calibration of a model for data collected before a catchment change occurred, adjustment of model parameters to reflect that change, and the validation of the model based on data collected after the change occurred. Because measurements are seldom available before and after land cover change occurs, this test is often impossible to implement. Ewen et al (1996) suggested that another appropriate test for a model intended to predict the effects of land cover change is the proxy-catchment test. Proxy-catchment tests involve calibration of a model for one catchment, adjustment of model parameters to reflect a second catchment, and validation of the model for the second catchment.

f. Model uncertainty

All model predictions are limited by uncertainty derived from many sources including: model structural error, errors in model input data, errors in output-variable measurements, uncertainty in parameter values, and uncertainty in initial conditions. One technique for quantifying model predictive uncertainty frequently employed in environmental simulation modeling is the generalized likelihood uncertainty estimation methodology (GLUE) developed by Beven and Binley (1992). In the GLUE methodology, parameters sets are randomly sampled from a prior distribution of parameter values (often a uniform distribution) and used to run the model. Model output for each parameter set is assessed using a likelihood measure (sometimes called a goodness-of-fit measure) that quantifies the correspondence between model predictions and available observations. Parameter sets that result in likelihood measures below a certain threshold are designated “non-behavioral,” and the predictions of the remaining parameter sets are weighted according to the associated likelihood measure. Beven (2001) has suggested that the GLUE technique addresses uncertainty derived from most data and model errors as well as parameter uncertainty. Recent criticism, in contrast, has questioned the ability of the GLUE methodology to address model uncertainty derived from model errors, input-data errors, or output-variable measurement errors (Stedinger et al, 2008). Even this criticism, however, concedes that the GLUE methodology provides insight into model sensitivity to parameter values. Though the GLUE methodology is now widely recognized to be a subjective technique that generates qualitative uncertainty bounds, it is also widely applied as a simple approach to uncertainty estimation in nonlinear systems and is generally acknowledged to describe model sensitivity to parameter uncertainty.

3. Statement of Problem

Though much research has examined the impacts of urban and suburban development on water resources, and though many planners and policy makers are eager to mitigate these impacts, scientific knowledge that might advance policy or practice is lacking (Alberti et al 2007, Wolosoff and Endreny 2002).

One approach to mitigation that has attracted great interest is the management of vegetation in urban and suburban areas to increase interception, evapotranspiration, and infiltration. Though some research has addressed the impact of expanded tree canopies on hydrologic response, none has addressed the impact of the spatial distribution of vegetation.

Distributed, physically-based hydrologic models offer significant advantages over empirical and lumped-conceptual models in understanding the effects of land cover change on catchment hydrologic response (Beven 2001). To date, however, few studies have applied such models to urban catchments.

This thesis examines the use of an existing eco-hydrologic simulation model to investigate the impact of land cover composition and pattern on urban hydrologic response. This thesis focuses on the impact of vegetation patterns. The themes and questions addressed by this study are presented below:

Theme A: Application of a distributed, physically based model to an ungauged urban catchment

- Question A1: Can calibrated soil and groundwater parameters from a forested reference catchment be transferred to an ungauged suburban catchment?
- Question A2: Can a distributed, physically based model accurately reproduce streamflow from a suburban catchment?

Theme B: Prediction of the impact of land cover composition and pattern on catchment hydrologic response.

- Question B1: What is the impact of different extents of tree cover in a suburban catchment on aggregate catchment response? Does this impact exceed the uncertainty generated by parameter uncertainty?
- Question B2: What is the impact of different patterns of tree cover in a suburban catchment on aggregate catchment response? Does this impact exceed the uncertainty generated by parameter uncertainty?
- Question B3: What is the impact of different patterns of tree cover in a suburban catchment on distributed catchment response?

4. Study Area Description

a. Topography

Pond Branch (PB) and Baisman Run 3 (BR3) are sub-catchments of the extensively studied Baisman Run catchment in the Piedmont region of Maryland (Figure 4.1). Pond Branch is a 0.31 km^2 catchment with elevations ranging from 190 m along the northwestern crest to 130 m at the outlet to the south (Figure 4.2). Gentle upland slopes and steep side slopes drain to a broad riparian area containing a perennial headwater stream. The stream channel is relatively narrow, and is confined in places by bedrock. Baisman Run 3 is a 0.69 km^2 catchment with elevations ranging from 200 m along the southwestern crest to 130 m at the outlet to the northeast (Figure 4.2). Side slopes are gentler than those in Pond Branch but also drain to a broad riparian area containing a perennial headwater stream. Towards the outlet the stream channel becomes incised and widened.

b. Soils

Both catchments are underlain by micaceous schist, with occasional bedrock outcrops occurring along the stream channels and on steeper slopes (Cleaves 1970). The NRCS SSURGO database assigns the soils in the Pond Branch and Baisman Run 3 catchments to 5 soil series: Baile silt loam, Codorus silt loam, Elioak silt loam, Glenelg loam, and Manor loam. According to SSURGO soils are very deep and saturated hydraulic conductivity is moderate to very high. Field surveys demonstrate, however, that the low resolution SSURGO data masks significant variability. Upland soils are deep and underlain by thick saprolite; midslope soils are extremely shallow; and bottomland soils are deep with a substantial organic layer (Tague and Band 2004, Wolman 1987, Cleaves 1970).

c. Vegetation and Land Cover

Land cover in Pond Branch consists almost entirely of forest, except for 3 acres of grasses along a gas pipeline. The forest is composed mostly of hardwoods including tulip poplar (*Liriodendron tulipifera*), chestnut oak (*Quercus prinus*), blackjack oak (*Quercus marilandica*), white oak (*Quercus alba*), red oak (*Quercus rubra*), pin oak (*Quercus palustris*), red maple (*Acer rubrum*), box elder, (*Acer negundo*), American beech (*Fagus grandifolia*), dogwood (*Cornus florida*), and others (personal communication, Oregon Ridge State Park, Wolman 1987, Brush et al 1980). Land cover in Baisman Run 3 was obtained from a 5 m land cover classification map generated by Zhou and Troy (2006). Based on GIS analysis described further below, land cover was determined to consist of 45 ha of forest in the eastern portion of the catchment (65.3% of the catchment area), 5 ha of impervious cover in the western portion of the catchment (7.3% of the catchment area), and 19 ha of lawn distributed throughout the catchment (27.3% of the catchment area)(Figure 4.3).

d. Climate and Precipitation

Climate in the Baltimore region is characteristic of mid-latitude, continental climates. Temperature varies markedly with the season, with a mean annual temperature of 14°C. Precipitation is distributed uniformly throughout the year and averages 50 - 100 mm per month, except for a late spring and summer maximum of 100 – 140 mm. Mean annual precipitation is ~1066 mm (data collected at Baltimore Washington International Airport (BWI) from 1971 – 2000) (Maryland State Climate Office). For water years 2000 through 2007, mean annual streamflow from PB and BR3 accounted for less than 40% of mean annual precipitation at BWI. Mean annual precipitation recorded at BWI was 1104 mm, while the observed annual discharge was 417 mm from PB, and the estimated annual discharge was 438 mm from BR3.

5. The Regional Hydro-Ecological Simulation System (RHESSys)

The Regional Hydro-Ecological Simulation System (RHESSys) is a spatially distributed, GIS based model that represents both hydrologic and ecologic processes to simulate the fluxes of water, carbon, and nutrients within a catchment (a detailed description is provided by Tague and Band, 2004). According to data availability and computing constraints, the model may be run on hourly to daily time steps. Model inputs consist of climate time series characterizing the vertical fluxes of water and energy, and GIS layers characterizing the catchment physical characteristics that determine catchment processing of mass and energy, including topography, soils, vegetation, and impervious cover. Because RHESSys simulates both hydrologic and vegetation processes within a spatial context, it is well suited to the modeling of suburban catchments with a mix of natural and engineered drainage components. Below is a brief review of the model processes relevant to land cover change in suburban catchments:

a. Interception

Canopy interception (CI) is calculated as a function of rainfall depth (RT), gap fraction (GF), plant area index (PAI), specific rain capacity (cp_{rain}) , and current interception storage (θ_I) as follows:

$$CI = \max\{0.0, \min[(1 – GF)RT, PAIcp_{rain} - \theta_I]\} \quad (1)$$

When precipitation occurs, interception by the vegetation canopy may be limited by either the depth of precipitation ((1-GF)RT), or the remaining canopy storage capacity ($PAIcp_{rain} - \theta_I$). Note that in modeling interception, RHESSys considers both the spatial and temporal variability of gap fraction and canopy storage capacity. In the spatial domain, these parameters vary with plant assemblage across the catchment, while in the temporal domain these parameters vary with the season.

b. Evapotranspiration

Evapotranspiration rates (ET) are computed using the standard Penman-Monteith equation with a Jarvis-based model of canopy stomatal conductance (Jarvis 1976). The Penman-Monteith method is a “big leaf” model that estimates evapotranspiration based on the available energy, the vapor pressure deficit at some reference height, and two conductance coefficients: the canopy conductance, and the boundary layer conductance. The canopy conductance is the product of the leaf area and the stomatal conductance. To account for the environmental and physiological controls on conductance, Jarvis-type models estimate stomatal conductance as the product of a theoretical maximum conductance and a series of functions of environmental factors ranging from 0 – 1. In RHESSys, the environmental factors included in the model of stomatal conductance are light, atmospheric CO_2, leaf area index, vapor pressure deficit, and leaf water potential (which is itself a function of rooting zone percent saturation). The canopy conductance term thus accounts for the dependence of actual ET on vegetation type, season, and soil moisture conditions. Through this term, RHESSys can account for the spatial variability in ET as vegetation and soil moisture characteristics vary across the catchment and the temporal variability in ET as environmental conditions vary across the seasons.

c. Vegetation Growth

RHESSys may be run in either a dynamic growth mode or a static mode. In the dynamic growth mode, allocation of net photosynthesis among the various vegetation components is explicitly simulated, and vegetation structure changes from year to year in response to the availability of carbon and nitrogen as well as environmental conditions. In the static growth mode, in contrast, vegetation structure is prescribed by the modeler and does not change from year to year. To describe the vegetation structure, the modeler must define the maximum leaf area index (LAI) and rooting depth. In both "static" and "dynamic growth" mode RHESSys simulates the seasonal growth and senescence of vegetation. Leaf on and leaf off are simulated according to the timing defined by the modeler. Thus both growth modes can account for the temporal variability in vegetation processes. Because the present research was more interested in the hydrologic impact of vegetation than the associated biogeochemical fluxes of carbon and nutrients, we implemented the static mode.

d. Infiltration

RHESSys considers the effect of antecedent soil moisture, rainfall intensity, and impervious cover on rates of infiltration. Infiltration is computed using the widely-applied Philip equation. This equation determines infiltration as a function of rainfall intensity, time to ponding, sorptivity, and saturated hydraulic conductivity at the wetting front. A key feature of RHESSys in the urban context is its ability to represent the restriction of infiltration by impervious cover. Wherever impervious cover occurs, the catchment surface is assigned a vertical hydraulic conductivity of zero.

e. Surface and subsurface flows

As discussed in Tague and Band (2004), two algorithms are provided for the simulation of lateral fluxes of water: a TOPMODEL algorithm adapted from Beven and Kirkby (1979) and an explicit routing algorithm adapted from DHSVM (Wigmosta et al. 1994). The TOPMODEL algorithm calculates a topographic index for each landscape patch, and assumes that all patches with the same value of the topographic index behave in a hydrologically similar way. Based on this assumption, soil moisture is calculated for each value of the topographic index, rather than each patch within the catchment, and the results mapped onto the catchment. TOPMODEL assumptions also allow the calculation of subsurface flows based on the average saturation deficit, and the calculation of surface flows based on the extent of saturated source areas.

The explicit routing algorithm, in contrast, attempts to represent the flowpaths of water as well as the distribution of hydrologic response. Surface and subsurface flows are calculated from each patch to all of its downslope neighbors. Subsurface flows are calculated based on the local hydraulic gradient, hydraulic transmissivity, and flow width:

$$q(t)_{a,b} = Tr(t)_{a,b} \tan\beta_{a,b}\omega_{a,b}, \qquad (2)$$

where $q(t)_{a,b}$ is the saturated throughflow from patch a to patch b, $Tr(t)_{a,b}$ is the transmissivity from patch a to patch b, $\tan\beta_{a,b}$ is the local slope, and $\omega_{a,b}$ is the flow width between patches a and b. For non-road patches, surface flows follow the same patch topology as subsurface flows, and are assumed to exit the catchment within a single time step unless re-infiltrated in a downslope patch. The routing of surface flows from road patches is intended to represent the presence of road drainage and storm drain networks. If no storm sewer network is defined, surface flow from road patches is routed to the nearest downslope stream patches as described above. In contrast, if a storm sewer network is defined, surface flow from road patches is routed to the appropriate storm sewer outlet.

In the urban context, the explicit routing algorithm offers several advantages over the TOPMODEL algorithm. First, soil moisture patterns reflect the distribution of vegetation and evapotranspiration, as well as topographic position. Second, overland flow may be re-infiltrated in downslope patches. And third, surface flowpaths reflect the presence of road drainage and storm drain networks. This research therefore implemented the explicit routing algorithm to compute lateral fluxes of water.

It should be noted that two algorithms are also provided for the representation of soil hydraulic conductivity profiles: one in which soil depth is infinite and saturated hydraulic conductivity declines exponentially with depth, and one in which soil depth is finite and saturated hydraulic conductivity is constant with depth. Because field measurements in the study catchments suggested constant saturated hydraulic conductivity with depth, this research implemented the second representation of soil structure (Tague personal communication).

f. Deep groundwater flows

The most recent version of RHESSys also includes a simple representation of deep groundwater flows. Inflows to the deep groundwater store are calculated as a constant fraction of precipitation, while outflows from the deep groundwater store are calculated as a linear function of the volume of water stored.

6. Datasets

Data required to conduct the catchment simulations included climate time series of minimum and maximum temperature and precipitation; GIS layers describing catchment topography, land use, land cover, vegetation characteristics, impervious surface, and soils; default files describing soil and vegetation properties; and streamflow time series for calibration and validation. As stated in the introduction, much of the data for the present study was collected in the last decade as part of the Baltimore Ecosystem Study (BES) – one of twenty four Long Term Ecological Research (LTER) projects funded by the National Science Foundation (NSF).

a. Climate time series

Climate data were obtained from the National Climatic Data Center (NCDC) and the LTER Climate and Hydrology Database. Daily temperatures recorded at Baltimore Washington International airport (~35 km south of the study catchments) were obtained from the NCDC, while daily precipitation depths recorded at McDonough School (~10 km south of the study catchments) were obtained from the LTER. Precipitation depths at McDonough School were measured with a tipping bucket gauge, and were available from 5/2000 to 12/2001 and from 1/2003 to the present. Though precipitation was available at resolutions as fine as 15 min, data was aggregated to the daily scale for this research. Although a tipping bucket gauge was installed nearer to the study catchments in Oregon

Ridge State Park (within 1 km of Baisman Run), the gauge was infrequently maintained and data from the gauge was deemed unreliable during the time domain of the simulations. It is expected that the distance between the McDonough gauge and the study catchments may introduce some error into the model, particularly for summer convective storms.

b. GIS datasets

GIS layers describing catchment physical characteristics were derived from 2 datasets provided by the BES. Layers describing catchment topography were derived from a 1 m LIDAR dataset provided by the BES, while layers describing catchment land use, land cover, and vegetation characteristics were derived from a 5 m land cover classification map generated by Zhou and Troy (2006). Zhou and Troy conducted object-oriented analysis of digital aerial imagery and LIDAR data to classify land cover in the study catchments into 4 distinct classes: building, pavement, fine textured vegetation, and coarse textured vegetation. The generation of RHESSys input maps from these GIS layers is described further below.

c. Default files

RHESSys requires the modeler to provide a series of files defining the physical characteristics associated with each land use type, land cover type, and soil type. For this research, parameter values for these files were adapted from the existing library of default files available through the RHESSys 5.8 online manual. Default files for urban and undeveloped land uses and for impervious, grass, and deciduous forest land covers were applied without modification. Soil default files were adapted from the library file for sandy loams and modified to reflect the variability in soil properties with topographic position. Based on field observations, catchment soils were classified into 3 classes: riparian, midslope, and upland. Riparian soils were assigned a soil depth of 8 m, soil porosity of 0.485, and pore size index of 0.589; midslope soils were assigned a soil depth of 1 m, soil porosity of 0.485, and pore size index of 0.189; and upland soils were assigned a soil depth of 15 m, soil porosity of 0.435, and pore size index of 0.204 following Law (2004).

d. Streamflow time series

Time series of mean daily discharge were obtained from the US Geological Survey (USGS). Daily discharge from PB was obtained from USGS gauge number 01583570, located at the outlet of Pond Branch. Though no stream gauge was available at the outlet of BR3, USGS gauge number 01583580 was available at the outlet of Baisman Run. Because land cover in BR3 resembles land cover in Baisman Run more closely than land cover in Pond Branch, daily discharge from BR3 was estimated based on the USGS gauge located at the outlet of Baisman Run. Mean daily discharge from both gauges was available from November 1999 to 2008. Estimates of discharge from BR3 were based on synoptic samples of instantaneous volumetric discharge collected at the outlet of BR3. Figure 4.4 shows the location of the USGS stream gauges and synoptic sample site.

Daily discharge from BR3 was estimated based on the regression of synoptic samples taken at the outlet against USGS data for BR. Twenty four synoptic samples of discharge from BR3 were collected between July 2001 – January 2003 and August 2006 – October 2007. Corresponding flows from BR were obtained from the USGS database, and discharges from the two locations were plotted against one another (Figure 4.5). Though the measurements at larger discharges suggest that the data might best be characterized by a power function, the number of samples was considered inadequate to determine the power function coefficients, and a linear relationship was assumed. Linear regression was performed on the thirteen synoptic samples collected between 7/2001 and 1/2003 measuring discharges of less than 555 m^3/day (or 0.80 mm/day). The slope of the best fit line between volumetric discharge from BR3 and BR was 0.21, a figure quite close to the ratio of catchment areas (0.18). It is therefore probable that estimates of discharge from BR3 are more accurate for low and moderate flows than for high flows, and may underestimate high flows.

Hydrologic simulation models are often limited by the accuracy of available measurements of output variables. In the present study, two limitations on the accuracy of the streamflow time series should be noted. The first is the accuracy of the Pond Branch gauge. Because of the tendency for low flows at Pond Branch to bypass the weir (soil pipes are periodically excavated in the stream bank on one side, allowing portions of flows to circumvent the weir) and for very high flows to overtop or bypass the weir, the Pond Branch gauge produces records of limited accuracy and is rated “fair” by the USGS. The second is the first order approximation of discharge from BR3. Because so few measurements at high flows were available, the linear relationship observed at low flows was assumed to characterize high flows as well. Several synoptic samples at somewhat higher flows, however, suggest that the relationship between discharge from BR3 and discharge from BR might be better approximated by a power function. It is because of this severe limitation to the accuracy of our estimates of streamflow from BR3 that a proxy-catchment approach is applied to calibrate the model for BR3. Though measurements of very high flows from PB are also of limited accuracy, the error in the PB record is believed to be less than the error we would generate by approximating a power function with a best-fit-line.

7. Methods

a. Spatial Data Processing

Extensive processing was conducted to translate the GIS datasets described above into the landscape and flowpath representations required by RHESSys. Spatial data processing used 3 programs: the Terrain Analysis System (TAS), ESRI ArcMap, and the RHESSys utility CREATE_FLOWPATHS. Prior to all processing described below, the source datasets were resampled to 10 m resolution. This aggregation was required to reduce the quantity of computational units to a number that would not exceed dedicated computational resources.

i. Catchment Delineation

For both catchments, we first coarsened the 1 m LIDAR dataset described above to a 10 m digital elevation model (DEM), then used TAS to derive the catchment boundaries, stream channels, and hillslope boundaries. Though a detailed map of stream channels was available from Baltimore County, we derived the channel network from the DEM for model consistency. Stream channels were derived using the O'Callaghan and Mark (1984) method, a global sequential algorithm which classifies cells as belonging to the stream network if their specific contributing area exceeds a certain threshold and if the stream segment of which they are a part exceeds a certain length. Pairs of thresholds were systematically tested to produce channel networks that most resembled the channels mapped by the county. The best approximations were derived with a specific contributing area threshold of 450 m^2, and a stream length threshold of 180 m. Figure 4.6 shows derived catchment boundaries and stream networks, along with the stream channels mapped by the county. TAS was also used to derive hillslope boundaries based on the DEM and derived channel networks. Delineation generated three hillslopes for PB, and twenty hillslopes for BR3. Derived hillslope boundaries are shown in Figure 4.7.

ii. Catchment Topography and Soils

To generate a landscape representation for each catchment, RHESSys required maps of catchment slope, aspect, and wetness index, as well as elevation. Functions provided by TAS were used to generate each of these layers from the 10 m DEM. As noted in the descriptions of the study area and default files, soils in both catchments are observed to vary significantly with topography. Because the soil coverages provided by SSURGO were of insufficiently fine resolution to represent the variation of catchment soils with topography, catchment topographic layers were processed to produce a layer describing the distribution of soil types. A simple conceptual model was constructed to classify catchment soils into 3 classes: riparian, midslope, and upslope. Riparian soils were predicted to occur in areas of low slope within a small distance of the stream channel, while upland soils were predicted to occur in areas of low slope beyond a small distance from the stream channel, and midslope soils were predicted to occur in all other areas. To translate this conceptual model into a representation of catchment soil distributions, cells with slope less than 8% occurring within 20 m of the stream channel were classified as riparian, cells with slope less than 5% occurring beyond 20 m of the stream channel were classified as upland, and all other cells were classified as midslope (Figure 4.8). Though approximate, the resulting classifications were in accord with expert knowledge of the catchments.

iii. Catchment Land Use and Land Cover Layers

Coverages of land use, land cover, and vegetation characteristics were derived from the 5 m land cover classification map provided by Zhou and Troy (2006). To generate a coverage of land use, we reclassified building and pavement as urban and all vegetation as undeveloped. Similarly, to generate a coverage of land cover we reclassified building and pavement as impervious, fine textured vegetation as grass, and coarse textured vegetation as forest. Figure 4.3 shows the resulting map of land cover for BR3. According to this map, land cover in BR3 consists of ~65.3% (or 45 ha) forest, ~7.3% (or 5 ha) impervious surface, and ~ 27.3% (or 18.7 ha) lawn.

To describe vegetation characteristics, RHESSys requires maps of rooting depth and leaf area index (LAI). To generate a coverage of rooting depth, we assigned building and pavement land covers a rooting depth of 0, fine-textured vegetation a rooting depth of 8 cm, and coarse-textured vegetation a rooting depth of 1 m. Because no field measurements were available, we applied order of magnitude estimates based on a review of the literature. Studies report a mean rooting depth for temperate deciduous forests of 2.9 ± 0.2 m (Canadell et al.,1996), while turfgrass scientists report a typical rooting depth for cool-season turfgrasses of 5 – 15 cm (Landschoot 2007, Lilly personal communication). A sensitivity analysis (described further below) was performed to assess the sensitivity of model results to the rooting depth parameters. To generate a coverage of LAI, we initially assigned building and pavement a maximum LAI of 0, fine-textured vegetation a maximum LAI of 0.5, and coarse-textured vegetation a maximum LAI of 5. Field measurements of leaf litter in the BES permanent plots suggested an all-sided LAI of 10, and therefore a one-sided LAI of 5. Again, because no field measurements were available for grass LAI, we applied an order of magnitude estimate based on values published in the literature (Lazzaroto et al. 2009). Initial calibration results for PB and BR3 (described further below) suggested that LAI values for PB forest canopy might be lower than those for BR3. These results corresponded with field observations of the tree canopy in the 2 catchments. In Pond Branch, greater damage to the tree canopy was observed following Hurricane Isabel, and the overstory along the riparian corridor was observed to be poorly developed relative to the overstory in BR3. PB LAI values for coarse textured vegetation were therefore modified to 4.5 for midslope and upland locations, and 2.5 for riparian locations. For urban catchments, RHESSys also requires a coverage defining the extent of impervious surface. To generate a coverage of impervious surface, we reclassified building and pavement as impervious and all other land covers as pervious.

iv. Catchment Flowpaths

For simulations that use the explicit-routing algorithm to represent lateral fluxes of water, RHESSys requires a flow table describing the topology of the flow network. A utility to produce flow tables with the appropriate format is provided on the RHESSys website. We used this utility to generate the flow tables required by RHESSys.

b. Calibration and Validation

Calibration and validation of RHESSys to simulate land cover change in BR3 was complicated by all three obstacles discussed in Sections 2d, 2e, and 2f. First, limited streamflow data were available to calibrate the model. As discussed above, the availability of streamflow data for BR as well as a set of instantaneous streamflow measurements for BR3 permitted the estimation of daily discharge (Q) from BR3, but the linear relationship developed was observed to perform poorly at higher flows. We therefore applied the first approach to model calibration in the absence of data presented in Section 2d, and calibrated RHESSys for data from PB. The model was calibrated for data from October 1, 2004 to September 30, 2005 and October 1, 2006 to September 30, 2007 (analysis of seasonal precipitation and discharge data suggested that precipitation data for water year 2006 was inaccurate). Five parameters were calibrated: a parameter describing the exponential decay of hydraulic conductivity with depth (m), a multiplier for saturated hydraulic conductivity in the horizontal dimension (K_{sat0}), a multiplier for saturated hydraulic conductivity in the vertical direction ($K_{sat0,v}$), a parameter describing the flux of water into the groundwater store (gw1), and a parameter describing the flux of water from the groundwater store to the stream (gw2). Feasible ranges for each parameter were defined based on previous modeling experience, and parameter values within these ranges were randomly sampled to generate 4,000 parameter sets. Model performance was quantified by calculating the Nash Sutcliffe efficiency for discharge at the catchment outlet (Nash and Sutcliffe 1970). Because the goals of stormwater management address both peak flows and baseflows, we determined that it was important for the model to accurately predict both peak flows and baseflows, and calculated two Nash Sutcliffe measures for each parameter set. We calculated the Nash Sutcliffe efficiency of Q to measure the accuracy of peak flow predictions, and the Nash Sutcliffe efficiency of log(Q) to measure the accuracy of baseflow predictions. All parameter sets for which the Nash Sutcliffe efficiency of both Q and log(Q) were greater than 0.5 were designated behavioral.

The second obstacle complicating model calibration and validation was the absence of streamflow data collected across a change in land cover. As discussed above, model predictions may be regarded as reliable only when model validation has demonstrated the model's fitness for its intended application. For this research, model validation must demonstrate the model's ability to accurately predict streamflows for different land covers. Though insufficient data was available for a differential split-sample test, sufficient data was available for a limited proxy-basin test. At low flows, the linear regression of streamflow from BR3 against streamflow from BR was observed to produce accurate estimates of streamflow from BR3. Low flows from BR3 were therefore deemed sufficiently accurate for use in model validation. Calibrated parameter sets were transferred to BR3, and the model was run for October 1, 2004 to September 30, 2005 and October 1, 2006 to September 30, 2007. Because taking the log of Q diminishes the weight of higher discharges and enhances the weight of lower discharges, model performance was quantified by calculating the Nash Sutcliffe efficiency of log(Q). All parameter sets for which the Nash Sutcliffe efficiency of log(Q) was greater than 0.5 were designated behavioral. By calibrating model parameters with streamflow from the forested Pond Branch catchment and validating model parameters with *low flows* from the suburban Baisman Run 3 catchment, we achieved a limited proxy-basin test.

The third obstacle complicating model calibration and validation was the uncertainty derived from model errors, input data errors, output variable errors, and parameter uncertainty. This research uses the GLUE methodology to generate uncertainty bounds for model predictions. As discussed above, the GLUE methodology is a widely applied, though qualitative, technique for describing model sensitivity to parameter uncertainty. Beven (2001) notes that all subjective decisions made in applying the GLUE methodology should be made explicit so that the analysis can be discussed, disputed, or repeated with alternative assumptions. We therefore briefly review the decisions made about the ranges for each parameter value, the sampling strategy for the parameter sets, and the likelihood measure developed to weight the model predictions. For this research, feasible parameter ranges were determined based on previous model experience. The m parameter was allowed to range from 0.1 to 20, the lateral Ksat multiplier was allowed to range from 1 to 1000, the vertical Ksat multiplier was allowed to range from 1 to 100, the gw1 parameter was allowed to range from 0.01 to 0.45, and the gw2 parameter was allowed to range from 0.001 to 0.1. Prior distributions of all parameter values were assumed to be uniform, and values were randomly sampled to prepare 4,000 Monte Carlo simulations. The likelihood measures were computed based on the Nash Sutcliffe efficiencies for streamflow from PB and BR3. According to Beven, "the choice of a likelihood measure should clearly be determined by the nature of the prediction problem" (Beven 2001). Because this research is interested in peak flows and runoff volume, for Pond Branch the Nash Sutcliffe efficiencies of Q were selected for inclusion in the likelihood measure. Because estimates of high flows from BR3 were of limited accuracy, for BR3 the NS efficiencies of log(Q) were selected for inclusion in the likelihood measure. Beven (2001) identifies summation and multiplication as appropriate operations to combine likelihood measures. For this research, the likelihood measure was calculated as the product of NS(Q) for Pond Branch and NS(logQ) for BR3, normalized so that the sum of all measures for all behavioral parameter sets was 1.

c. Simulation of Vegetation Management Practices

Three vegetation management practices were simulated in Baisman Run 3: conversion of all lawn to forest, conversion of downslope lawn to forest, and conversion of upslope lawn to forest. To generate the first scenario, all 18.65 ha of lawn were converted to forest. To generate the second and third scenarios, the lawn was partitioned into equal areas based on upslope contributing area. The value of the upslope contributing area for each patch was obtained from the flowtable generated by the CREATE_FLOWPATHS utility. In the second scenario, 9.25 ha of lawn with upslope contributing area greater than 620 m^2 was converted to forest. In the third scenario, 9.25 ha of lawn with upslope contributing area less than 600 m^2 was converted to forest. Figure 4.9 shows the lawn area converted to forest for each scenario. In this and subsequent figures, FA denotes the conversion of all lawn to forest, FD denotes the conversion of downslope lawn to forest, and FU denotes the conversion of upslope lawn to forest. For each scenario the model was run with all behavioral parameter sets.

To assess the impact of estimated rooting depth on model results, a limited sensitivity analysis was performed. Grass rooting depth was defined as 30 cm (rather than 8 cm) and simulations were repeated with the parameter set associated with the highest likelihood measure.

8. Calibration and Validation Results

a. Calibration

As discussed above, model parameters were calibrated for Pond Branch by comparing the simulated streamflows produced by randomly-generated parameter sets to the observed streamflow recorded at Pond Branch. Initial calibration results assuming a forest LAI of 5.0 yielded no behavioral parameter sets (defined as having NS>0.5 for both Q and log(Q)). Initial calibration achieved a maximum Nash Sutcliffe efficiency for Q of 0.47, and a maximum Nash Sutcliffe efficiency for log(Q) of 0.53. The range of predicted streamflows for all parameter sets with NS efficiencies > 0.4 indicated that the model consistently underpredicted streamflows (Figure 5.1). The consistent underprediction of streamflow suggested an overprediction of ET and LAI. Forest LAI values were therefore adjusted to 4.5 in upland areas and 2.5 in riparian areas, and the calibration simulations were repeated.

The adjustment of Pond Branch LAI values significantly improved streamflow predictions. Calibration results yielded 193 behavioral parameter sets (defined as having NS>0.5 for both Q and log(Q)). Calibration achieved a maximum Nash Sutcliffe efficiency for Q of 0.56, and a maximum Nash Sutcliffe efficiency for log(Q) of 0.60. Figures 5.4a, b, and c compare the range of predicted streamflows to the observed streamflow. Though streamflows are still often underpredicted, model predictions bound observations for a much greater proportion of the simulation period. Field measurements of LAI should be collected to better constrain model values.

b. Validation

Model validation transferred the parameter sets calibrated for Pond Branch to Baisman Run 3. For the performance criteria described above, model validation yielded 92 behavioral parameter sets. Interestingly, goodness of fit results for validation often exceeded those for calibration (Figure 5.2). Validation achieved a maximum Nash Sutcliffe efficiency for Q of 0.71, and a maximum Nash Sutcliffe efficiency for log(Q) of 0.68. Prior and posterior distributions of the parameters to which model performance was most sensitive are shown in Figure 5.3.

Comparisons of the range of predicted discharges to the observed (for PB) and estimated (for BR3) discharges are shown in Figures 5.4a, b, and c and 5.5a, b, and c. Though the observed/estimated discharge is not consistently bounded by the predicted range for either PB or BR3, the model generally reproduces the trends in discharge very well. The most significant deviation between the model predictions and the observed/estimated discharges occurs in July and August of 2007, when the observed/estimated discharge falls precipitously in both PB and BR3, and the predicted discharge does not. This apparent error in model predictions may derive from either errors in the input data, or errors in the model structure. Because the Pond Branch gauge performs poorly at low flows and rating curves are known to be less accurate at low flows, it may be that our observed/estimated values of streamflow are themselves underestimates of the actual values. Because July and August of 2007 were very dry months, it may also be that the study streams experience transmission losses in extremely dry conditions which the model algorithms cannot reproduce. Interestingly, the model appears to perform better for the suburban validation catchment than for the forested calibration catchment.

Bias and mean absolute error (MAE) were calculated for the expected value of daily discharge from PB and BR3 (obtained by taking the weighted average of simulated discharge for each behavioral parameter set). While PB simulated discharge exhibited a downward bias of 0.2 mm/day (or ~14% of the mean daily discharge of 1.47 mm), BR3 simulated discharge exhibited an upward bias of 0.17 mm/day (or ~14% of the mean daily discharge of 1.24 mm). Mean absolute errors were comparable for PB and BR3, with a MAE of 0.32 for PB and 0.35 for BR3.

Figure 5.6 presents the difference between simulated expected discharge and observed daily streamflow for each catchment. For both catchments, the model underpredicts peak flows. The difference in the direction of model bias is observed to derive from simulated baseflows. For PB, the model underpredicts baseflows, while for BR3 the model overpredicts baseflows.

Five limitations that affected the calibration/validation process should be noted: 1) the limited accuracy of the PB stream gauge, 2) the lack of discharge data from the outlet of BR3, 3) the lack of measured LAI data for either catchment, and 4) the inaccurate precipitation data for WY 2006, which likely produced an upward bias in simulated discharge during the first months of WY 2007, and 5) the distance of the precipitation gauge from the study catchments.

9. Vegetation Management Results

a. Impact of Vegetation Management on Streamflow Regime

Both runoff volumes and peak flows declined dramatically with the conversion of some or all of the lawn area in BR3 to forest. Annual runoff volumes fell ~100 mm (or 20%) when all lawn was converted to forest and ~50 mm (or 11%) when half the lawn area was converted to forest (Figure 5.7). To examine the impact of vegetation management on peak flows, we estimated the 2, 5, and 10 year flows given the actual land cover. We fitted a Log Pearson Type III distribution to the annual peak flows for water years 2000-2008. To estimate the flows that occur with recurrence intervals of 2, 5, and 10 years, we applied the following equation:

$$\log(Q) = \overline{X} + K\sigma_{\log Q}, \tag{3}$$

where Q is the flow magnitude at the selected recurrence interval, $\overline{X}$ is the average of the logarithms of the available peak flows, K is a frequency factor that is a function of the skewness coefficient and recurrence interval, and σ is the standard deviation of the logarithms of the available peak flows. The estimated flow magnitudes were 9.0 mm/day for the 2 year flow, 13.4 mm/day for the 5 year flow, and 15.5 mm/day for the 10 year flow. (Note that these values are expected to be of limited accuracy, given the short times series from which they were obtained, and were calculated only to provide reference values for comparison with the management scenarios.) The number of exceedances expected for each land cover scenario was determined by weighting the number of exceedances for each behavioral parameter set by the corresponding likelihood measure. Exceedances of all examined flows fell most when all lawn was converted to forest, less when the downslope lawn area was converted to forest, and even less when the upslope lawn area was converted to forest (Table 5.1).

Daily and seasonal analyses indicated a seasonal pattern in the change in catchment hydrology associated with land cover change. Prediction bounds for daily discharge for the actual and entirely forested scenarios are shown in Figure 5.8. While the predicted change in daily streamflow exceeds the uncertainty associated with parameter estimation for the months of June through March, negligible change is observed in the months of April and May. Seasonal discharge and rainfall depths are shown in Figure 5.9a. For all three reforestation scenarios, the predicted change in seasonal discharge (as compared to the actual scenario) is greatest in the fall, declines through the winter and spring, and increases again in the summer. As expected, the runoff ratio follows an inverse pattern. The ratio of streamflow to precipitation increases in the winter and early spring, when vegetation is dormant, and decreases in the summer, when vegetation is growing. Changes in seasonal discharge were compared to seasonal patterns in the *distribution* of precipitation as well as seasonal patterns in the amount of precipitation. For each season, the range of daily precipitation depths was divided into fourteen 5 mm bins, and the number of events within each bin and cumulative depth of events within each bin was calculated. Figure 5.9b shows the frequency of events of different magnitude and the cumulative depth provided by events of different magnitude for each season. For all seasons, storms of 0 to 10 mm per day are most frequent, while storms of 10 - 65 mm per day account for most of the cumulative depth of precipitation. Interestingly, the frequency of the largest storms (greater than 50 mm per day) is comparable for all seasons. Though precipitation in the Baltimore region is generally delivered by less intense frontal storms in the winter and more intense convective storms in the summer, seasonal differences in storm intensity are not perceived at the daily scale. These patterns in seasonal streamflow, runoff ratio, and precipitation indicate that evapotranspiration is a key mechanism shaping catchment hydrologic response.

Analysis of daily and annual streamflows also indicated that varying the topographic position of re-forested areas produced a slight but unexpected change in streamflow response. Note that annual discharge for the forested-downslope scenario is slightly greater than annual discharge for the forested-upslope scenario (Figure 5.7). Table 5.2 presents the change in annual discharge (from the actual scenario) per unit change in catchment LAI. The greatest reduction per unit change in LAI is observed in the forested-upslope scenario. At the seasonal scale as well, discharge for the forested-downslope scenario exceeds discharge for the forested-upslope scenario for 3 of 4 seasons (Figure 5.9). Figure 5.10 shows that the difference in catchment hydrologic response for the forested-downslope and forested-upslope scenarios varies for baseflows and peak flows and follows a seasonal pattern. While peak flows from the forested upslope scenario generally exceed peak flows from the forested downslope scenario, the reverse is often observed for baseflows. Baseflows from the forested-downslope scenario generally exceed baseflows from the forested-upslope scenario during the fall, winter, and spring, while baseflows from the forested-upslope scenario are greater during the summer.

b. Sensitivity of Predicted Streamflow to Estimated Root Depth

Limited analysis of the sensitivity of streamflow results to the definition of grass rooting depth indicated that the above conclusions are robust across a range of rooting depths. Figure 5.11 compares simulated monthly discharge for the entirely forested scenario, the actual scenario with shallow grass roots (8 cm), and the actual scenario with deep grass roots (30 cm). For both deep and shallow grass root depths, monthly discharge is predicted to be significantly lower for the entirely forested scenario than for the actual scenario, with the difference greatest in the summer, fall, and winter and least in the spring. The sensitivity analysis also suggested that the unexpected dependence of streamflow response on the topographic position of catchment vegetation is robust across the range of rooting depths examined. Figure 5.12 shows the difference between monthly discharge from the forested upslope scenario and monthly discharge from the forested downslope scenario for both shallow and deep grass roots (compare to Figure 5.10). For deep grass roots as well as shallow grass roots, streamflows from the forested downslope scenario are greater in the fall and winter, while streamflows from the forested upslope scenario are greater in the summer.

c. Impact of Vegetation Management on Evapotranspiration

Analyses of evapotranspiration for each of the four vegetation management scenarios demonstrated that differences in ET account for much, but not all, of the differences in discharge. ET increases dramatically with the conversion of some or all lawn area to forest. Annual ET for the 4 vegetation management scenarios is shown in Figure 5.13. Note that the change in annual ET when lawn is converted to forest exceeds the change in annual discharge. ET increases by ~150 mm when all lawn area is converted to forest, and by ~75 mm when half the lawn area is converted to forest. The topographic position of re-forested areas does not appear to have a significant impact on annual ET.

As with streamflow, seasonal trends are apparent in the response of ET to the various vegetation management scenarios. Differences in monthly ET among the 4 scenarios are greatest in the growing season and negligible in the winter and fall, suggesting that transpiration accounts for most of the difference in ET (Figure 5.14). Comparison of changes in monthly evaporation to changes in monthly transpiration when lawn area is converted to forest confirms the dominance of transpiration (Figure 5.15a, b). Note that while evaporation is enhanced throughout the year, transpiration is enhanced only between the months of May and October. At the monthly scale as well, topographic position of re-forested areas does not appear to have a significant impact on ET.

Maps of annual ET were generated to show the spatial distribution of changes in ET for the 3 re-forested scenarios (Figure 5.16). For each scenario the distribution of increases in ET mirrors the distribution of the lawn area converted to forest. The distribution of decreases in ET is also significant. By comparing Figure 5.16 to Figure

4.8, it is observed that decreases in ET are greatest in the riparian areas. Note that the extent of areas experiencing decreased ET is greater for the forested-downslope scenario than for the forested-upslope scenario.

d. Impact of Vegetation Management on Soil Moisture

As expected, simulation results indicated that catchment-average saturation deficit is significantly altered by vegetation management. Figure 5.17 shows daily average saturation deficit for the entirely forested and actual scenarios. The increase in saturation deficit associated with re-forestation is more persistent in time than the increase in ET. While ET from the entirely forested scenario exceeds ET from the actual scenario only in the spring and summer, saturation deficit for the forested scenario exceeds saturation deficit for the actual scenario throughout the year. During the months in which ET from the forested scenario exceeds ET from the actual scenario, the difference in saturation deficit between the two scenarios is observed to increase. Conversely, during the months in which ET from the forested scenario is equivalent to ET from the actual scenario, the difference in saturation deficit between the two scenarios is observed to decline.

Unexpected differences are also observed in the temporal patterns of soil moisture for the forested-downslope and forested-upslope scenarios. Figure 5.18 shows the difference between forested-upslope and forested-downslope percent saturated area (middle pane), and forested-upslope and forested-downslope saturation deficit (bottom pane). In the fall, winter, and spring the forested-upslope scenario has a smaller saturated area than the forested-downslope scenario, while in the summer the forested-upslope scenario often has a larger saturated area. In all 4 seasons the forested upslope scenario has a greater saturation deficit than the forested-downslope scenario, but the difference declines throughout the fall, winter, and early spring, and rises in the late spring and summer. The timing of these differences appears to correspond to the timing of the differences in daily streamflow for the two scenarios (5.14 top pane). During the months in which the forested-upslope scenario produces less baseflow than the forested-downslope scenario, the forested-upslope scenario also has a smaller saturated area. Conversely, during the months in which the forested-upslope scenario produces more baseflow than the forested-downslope scenario, the forested-upslope scenario also has a greater saturated area. A similar correspondence is observed for the differences in daily average saturation deficit. During the months in which the saturation deficit of the two scenarios is converging, the forested-upslope scenario produces less baseflow than the forested-downslope scenario. Conversely, during the months in which the saturation deficit of the two scenarios is diverging, the forested-upslope scenario produces more baseflow than the forested-downslope scenario.

Figures 5.19 and 5.20 present maps of saturation deficit for two dates on which storms occurred: December 15, 2005 (when the catchment received ~45 mm of precipitation) and July 8, 2005 (when the catchment received ~60 mm of precipitation). In these figures ACT denotes actual land cover, FA denotes the conversion of all forest to lawn, FD denotes the conversion of downslope forest to lawn, and FU denotes the conversion of upslope forest to lawn. During the summer storm, the runoff generating areas (areas where the saturation deficit approaches zero) are located in downslope positions where flowpaths converge. These source areas occupy a greater proportion of the catchment in the actual and forested upslope scenarios, and a significantly smaller proportion of the catchment in the entirely forested and forested downslope scenarios. During the winter storm, in contrast, the runoff generating areas are located throughout the catchment and occupy a smaller proportion of the catchment in the entirely forested and forested upslope scenarios.

10. Discussion

This section begins with a discussion of the results and their relevance to land use planning in the context of the research questions, and concludes with a review of the research limitations.

a. Research Questions

Question A1: Can calibrated soil and groundwater parameters from a forested reference catchment be transferred to an ungauged suburban catchment?

Goodness-of-fit results for BR3 suggest that the transfer of parameters from a forested to a lightly urbanized catchment is viable, though neglecting the dependence of soil parameters on land cover may degrade model performance. While Nash Sutcliffe results suggest that the accuracy of model predictions for BR3 exceeds the accuracy of model predictions for PB, daily bias results indicate that the transfer of soil and groundwater parameters from a forested to an urbanized catchment may introduce error into the urban model. Potential sources of model bias include errors in the model structure and errors in the model parameters. We note that much of the model bias may be explained by errors in the LAI, hydraulic conductivity, and groundwater bypass parameters. In Pond Branch, the overestimation of LAI would explain the underprediction of streamflow, while in BR3 we predict that the overestimation of the hydraulic conductivity and groundwater bypass parameters might explain the underprediciton of peak flows and overprediction of baseflows. Indeed, previous research suggests that we likely overestimated the hydraulic conductivity and groundwater bypass parameters in BR3 by assuming these parameters to be independent of land cover. Field studies of infiltration rates in urban areas have found that urban soils are generally more compacted than undisturbed soils and tend to infiltrate water at lower rates (Gregory et al. 2006, Pitt et al. 2001, Hamilton and Waddington 1999). By transferring soil and groundwater parameters from a forested to an urbanized catchment without modifying parameter values to reflect the change in land cover, we likely overestimated the values of the hydraulic conductivity and groundwater bypass parameters. Though we cannot provide conclusive evidence that errors in these parameters are the source of model bias in BR3, our preliminary assessment indicates that this explanation is consistent with both field studies and model results.

Our research elaborates upon previous assessments of parameter transfer techniques. Whereas previous studies have demonstrated the viability of parameter transfer techniques among undeveloped catchments with similar climatic, topographic, and land cover characteristics (Gan 2006, Wagener 2006, Van der Linden and Woo 2003), we demonstrate the viability of parameter transfer from an undeveloped catchment to a suburban catchment. We note, however, that our result may not be robust for highly urbanized catchments. BR3 is a lightly urbanized catchment with only 7.3% of its area occupied by impervious cover and 27.3% occupied by lawn. Moreover, lawns in BR3 are large and well-established, and support unusually high rates of infiltration (Lipscomb, personal communication). We therefore propose that the viability of parameter transfer from forested to urbanized catchments may be dependent on the extent of urbanization, and recommend that this technique be examined further in more highly urbanized catchments.

Question A2: Can a distributed, physically based model accurately reproduce streamflow from a suburban catchment?

The goodness-of-fit results for BR3 suggest that distributed, physically based models are not only capable of reproducing streamflow from suburban catchments, but may perform better in suburban catchments than in forested catchments. This result is consistent with previous studies of the application of distributed, physically based models to urbanized catchments, which determined that model performance in urbanized catchments compared favorably to model performance in undeveloped catchments (Im et al. 2009, Jia et al. 2001, Cuo et al. 2008, Easton et al. 2007). In the present study, we suggest that the increase in model performance in BR3 relative to PB results from the simplification of hydrologic processes in urbanized catchments. Whereas subsurface flow processes are notoriously difficult to model, the representation of direct runoff from impervious areas is much more tractable. The greater prevalence of this process in urbanized catchments may explain the greater accuracy of our model predictions in the suburban study catchment.

This result and our parameter transfer results above offer a promising approach to the problem of predicting the impacts of land cover change in data-sparse urban areas. While parameter transfer schemes among catchments with similar physical characteristics can compensate for the lack of calibration data in urban areas, distributed, physically based models can provide distributed predictions of the impacts of land cover change and greater insights into the mechanisms producing those impacts. Further research should be conducted to develop and demonstrate this promising methodology.

Question B1: What is the impact of different extents of tree cover in a suburban catchment on aggregate catchment response? Does this impact exceed the uncertainty generated by parameter uncertainty?

Our study supports previous findings that the extent of forest and lawn in suburban catchments is a significant determinant of catchment hydrologic response, with increased tree canopy reducing peak and annual flows (Booth et al. 2002, Wang et al.2008). Our study also expands upon the findings of the only previous study explicitly designed to examine the impacts of urban vegetation on catchment hydrologic response (Wang et al. 2008). Wang et al. (2008) examined the impact of changes in interception associated with changes in LAI on streamflow response. Their research found that doubling the canopy LAI produced a significant increase in interception, but only a modest decline in annual runoff (1.3%). Our research, in contrast, examines the effect of changes in forest extent on both interception and transpiration. We find that transpiration is the dominant process determining the impact of vegetation on catchment hydrologic response, and that modest increases in catchment LAI produce significant decreases in annual runoff (~20% per unit change in catchment LAI). Moreover, by comparing the changes in streamflow response to the prediction bounds of the behavioral parameter sets, we demonstrate that the change in streamflow response associated with different extents of vegetation cover exceeds the uncertainty associated with parameter estimation.

This result has significant implications for land use planning. We demonstrate that expanding the urban tree canopy is an effective approach to reducing runoff volumes and peak flows from suburban catchments. Given the well established connection between flow regimes and stream channel erosion, pollutant delivery, and habitat degradation, we interpret this result to suggest that the expansion of the urban tree canopy is an effective approach to mitigating the symptoms of urban stream syndrome. This interpretation agrees with the results of previous empirical studies demonstrating the importance of tree cover as a predictor of stream biotic integrity (Hammer 1972, Goetz and Fiske 2008, Carlisle and Meador 2007, Strayer 2003, Steedman 1988).To attain water quantity and quality goals, land use planners should preserve or plant as much tree cover in urban areas as is consistent with other community goals.

Question B2: What is the impact of different patterns of tree cover in a suburban catchment on aggregate catchment response? Does this impact exceed the uncertainty generated by parameter uncertainty?

Our analysis of the impact of the topographic position of tree cover on streamflow response produced unexpected results with complex implications for land use planning. Though the planning literature generally recommends the planting or preservation of riparian forests to minimize the ecological impacts of urbanization, we found that riparian forests may not provide greater mitigation of the hydrologic impacts of urbanization than upslope forests. At the annual scale, the conversion of upslope lawn to forest actually reduced streamflow more than the conversion of downslope lawn to forest, while at the seasonal scale the conversion of upslope lawn to forest produced greater reductions in streamflow in 3 of 4 seasons. At the daily scale, however, the interpretation of our results becomes more complex. Though the conversion of upslope lawn to forest produces lower baseflows than the conversion of downslope lawn to forest, it consistently produces higher peak flows. Because both the reduction of peak flows and the reduction of runoff volumes are goals of stormwater management, this result requires a tradeoff among management goals. We propose that the management strategy most protective of ecosystem function may depend on the relative sensitivity of channel morphology and stream biota to erosive peak flows versus amplified baseflows, and the pollutant loads of each.

To place our results in the context of previous research on the impact of urban pattern on stream structure and function, we present a table reviewing previous studies (Table 5.3). Our research differs from all reviewed studies in its methodology: applying a modeling approach rather than empirical analysis. Our research also differs from all but two of the reviewed studies in its analysis of deliberately designed scenarios. Most of the research reviewed applied statistical regression techniques to analyze existing urban patterns over which the researchers had no control. Because we were able to confine the expansion of the urban forest to upslope or downslope positions, we were able to examine an aspect of land cover pattern previously unaddressed in urbanized catchments. Previous studies have examined the impact of the landscape position of vegetation in *agricultural* catchments (Crosbie et al. 2008, George et al.1999), but none have examined this aspect of land cover pattern in *urbanized* catchments. Our results are consistent with the finding of George et al. (2008) that (where salinity is not a constraint) the water table response to tree planting increases as trees are located further upslope. We recommend, however, that further research be conducted to corroborate or dispute our results. In our research, the impact of different patterns of vegetation on aggregate hydrologic response does not greatly exceed the uncertainty associated with parameter estimation. Our findings should therefore be regarded as hypotheses to guide further research, rather than conclusive results.

Finally, we note that even if riparian forests did not provide greater mitigation of the hydrologic impacts of urbanization than upslope forests, riparian forests are known to provide many other important functions. Riparian forests reduce the delivery of nutrients to the stream, enhance instream habitat for aquatic species, provide shade to the stream, and serve as habitat corridors for terrestrial species. In deciding upon the distribution of forest conservation areas, land use planners should consider the range of functions served by riparian forests.

Question B3: What is the impact of different patterns of tree cover in a suburban catchment on distributed catchment response?

Distributed model results allow us to visualize the patterns of catchment hydrologic response and develop more detailed explanations of the mechanism through which the topographic position of vegetation affects streamflow response. Figure 5.21 shows a flow chart illustrating our interpretation of distributed model results. Model results suggest that by reducing recharge to upslope areas during the growing season, the reforestation of upslope areas reduces the lateral subsidy to riparian areas during the following seasons. By the time summer arrives, however, the extent of catchment saturation is similar in both the forested upslope and forested downslope scenarios, and the lateral subsidy to riparian areas is approximately equivalent for both scenarios. By the time summer arrives, the downslope transfer of soil moisture is also less important in determining streamflow response. Figures 5.19 and 5.20 may be interpreted as showing the location of runoff generating areas within the catchment. In the fall, winter, and spring runoff generating areas are observed to be distributed throughout the catchment, while in the summer runoff generating areas are observed to be concentrated in riparian areas. Thus in the summer, when riparian processes are dominant in determining streamflow response, the planting of forest in downslope positions produces slightly less streamflow.

This research demonstrates the immense potential of distributed, physically based models in advancing both the understanding of hydrologic processes in urban areas and the development of land use policies protective of stream form and function. By providing insights into distributed responses, distributed, physically based models allow us to develop hypotheses about the mechanisms through which urban pattern informs streamflow response. These insights allow us to design land use policies that minimize the impacts of urban development on stream ecosystems.

b. Model and methodology shortcomings

The following is a brief review of study features that may limit the accuracy and application of our results. Model accuracy is limited by errors in the input climate data as well as errors in the streamflow data available for calibration. Model accuracy is also limited by uncertainty in model parameters. Some of this uncertainty arises because effective grid-scale parameters often cannot be estimated from measured point variables. Some of this uncertainty in model parameters also derives from lack of measured data. For this research, for instance, model parameterizations could be significantly improved if measurements of LAI and hydraulic conductivity were available. Measurements of LAI would allow us to better characterize the differences in LAI between PB and BR3, while measurements of hydraulic conductivity would allow us to better characterize the difference in hydraulic conductivity between soils beneath forest and lawn.

Further limitations derive from our choice of model time step. Because we represent all processes at a daily time step, our model cannot differentiate between long low-intensity storms and short high-intensity events. In the summer, we may therefore overestimate infiltration.

Finally, our results may be limited in their geographic application. Because hydrologic processes depend on local climate, geologic formations, topographic patterns, and dominant forms of vegetation, the conclusions of this research may not apply to catchments in very different regions. Also, because hydrologic processes depend on the spatial scale of analysis, the conclusions of this research may be limited to catchments of comparable size.

11. Conclusion

a. Implications for land use planning

Land use planning intended to maintain aquatic ecosystem function could be much improved by the application of distributed, process-based models in place of lumped conceptual models. Though data constraints complicate the calibration of distributed, process-based models in urban areas, our results demonstrate that parameter transfer from forested reference catchments to lightly urbanized catchments can produce accurate models. The potential gains in hydrologic understanding appear to warrant the greater complexity of model calibration. With lumped conceptual models we can predict little more than the effect of land cover extent on aggregate catchment response. With distributed, process-based models, however, we can expand our understanding of the hydrologic impact of land cover change to include both the impact of the *pattern* of land cover change, and the impact upon *distributed* hydrologic response. The spatial distribution of runoff generation is important in predicting sediment production and pollutant delivery as well as stream hydrographs (Dunne 1983). With distributed, physically based models we can therefore anticipate the impact of urban development on water quality as well as quantity. Because of these features, distributed, physically based models offer immense promise in assessing land use policies and guiding low impact design.

The present study begins to generate results that may inform land use policy. Our results suggest that, though riparian forests may provide greater mitigation of peak flows than upslope forests, upslope forests may provide greater mitigation of runoff volumes. This result raises questions about a common strategy in watershed planning. Many municipalities require the preservation of riparian forests to maintain ecosystem function (Chapel Hill, for instance, requires a riparian setback of 150 feet from all perennial streams). Our research suggests, however, that if downstream water impairment is sensitive to annual runoff volume as well as peak flows, the preservation of upslope forests is also important in protecting ecosystem function. This result reminds us that different elements of the landscape serve different functions, and that we cannot secure ecosystem function by preserving only one element of the landscape. Land use management should therefore be prepared to preserve different parts of the landscape depending on their particular function in their particular context.

b. Future research

To advance our understanding of the impact of urban pattern on catchment hydrologic response, this research should be integrated with field studies. Dunne (1983) recommends the joint development of field and modeling studies to define the kind and rigor of field measurements and to increase efficiency in the use of field data. Conversely, field data can corroborate or falsify model results and identify model conceptual errors. It was suggested above that the findings of this thesis regarding the impact of vegetation pattern on distributed hydrologic response should be regarded as a guide to further research, rather than conclusive results. Results from this research might be used to guide a field sampling campaign to determine whether the impacts of increased transpiration on soil moisture and the extent of variable source areas are indeed as significant as this research has suggested.

Further research should be conducted to extent this methodology to the prediction of water quality as well as quantity, to the assessment of additional LID practices, and to the analysis of land cover change impacts in more highly urbanized catchments. As stated above, the spatial heterogeneity of runoff generation is important not only in the prediction of stream hydrographs, but in the prediction of runoff quality as well. In many aquatic ecosystems, sediment and nutrient delivery is known to be an important cause of ecosystem degradation. Distributed, physically based models that include the spatial distribution of erodibility and nutrient loads would advance our understanding of the impact of land cover pattern on these important water quality parameters. In the background section, we introduced many approaches to low impact design. This research demonstrated the application of a distributed, physically based model to assess only one of these practices. Further studies should address the hydrologic impacts of other practices such as green roofs, rain gardens, and grassed swales. Finally, this research applied a distributed, physically based model to a lightly urbanized catchment in which man-made drainage infrastructure was not prevalent. The lack of a hydraulic component to model the movement of water through man-made conduits was therefore not a significant limitation for this study. Future studies should address the application of distributed, physically based hydrologic models to more densely urbanized catchments.

To accurately model the function of drainage infrastructure, future research should address the integration of distributed, process-based hydrologic models with the hydraulic models generally applied to manmade flow networks in densely urbanized catchments.

Land cover change models that represent ecosystem and hydrologic processes as well as hydraulic processes and that predict distributed as well as aggregate response offer immense promise in developing scientific knowledge that can inform ecologically sensitive land use policy and urban design.

Appendix A: Tables

Return Period (years)		Q (mm/day)	Simulated Exceedances, WY 2005 and 2007: Actual Land Cover	All Lawn to Forest	Downslope Lawn to Forest	Upslope Lawn to Forest
	2	9.0	1.23	0.85	0.94	1.00
	5	13.4	0.60	0.32	0.41	0.43
	10	15.5	0.35	0.12	0.21	0.21

Table 5. 1: Comparison of expected exceedances of the 2, 5, and 10 year peak flows for each of the vegetation management scenarios. The log Pearson type III technique was applied to estimate the flows that occur with recurrence intervals of 2, 5, and 10 years, and the number of simulated flows that exceeded these peak flows was calculated for each scenario. Exceedances of all examined flows fell most when all lawn was converted to forest, less when the downslope lawn area was converted to forest, and even less when the upslope lawn area was converted to forest.

	All Lawn to Forest	Downslope Lawn to Forest	Upslope Lawn to Forest
delta Q (mm/year) Min / Max	-106.3 / -105.7	-52.4 / -52	-54.3 / -58.6
% Lawn Area Converted to Forest	100%	50%	50%
delta LAI	1.23	0.61	0.61
delta Q / delta LAI Min / Max	-86.4 / -86.0	-85.9 / -85.3	-89.1 / -96.1

Table 5. 2: Expected change in annual discharge per unit change in catchment LAI for all vegetation management scenarios. Interestingly, the greatest reduction per unit change in LAI is observed in the forested-upslope scenario.

Table 5. 3: Previous studies of the impact of urban pattern on catchment hydrologic response.

Study	Study Type	Pattern Variables Assessed	Response Variables Assessed	Findings
Pappas et al. 2008	Laboratory simulation	Impervious area connectivity	Runoff rate and cumulative runoff for 96 min – duration storm	Downslope impervious cover initially produced more runoff, but the difference between impervious treatments declined as soil saturation increased
Shuster et al. 2008	Laboratory simulation	Impervious area connectivity	Runoff rate and cumulative runoff for 5 year recurrence interval storm event	Downslope impervious cover generally produced more runoff than upslope impervious cover for dry initial conditions, but often produced more runoff for wet initial conditions.
Newall and Walsh 2005	Empirical analysis	Impervious area connectivity	Water quality and diatom-based indices	Impervious area connection was the strongest explanatory variable.
Taylor et al. 2004	Empirical analysis	Impervious area connectivity	Benthic algal biomass	Impervious area connection was a stronger explanatory variable than impervious area extent.
Hatt et al. 2004	Empirical analysis	Impervious area connectivity	Pollutant concentrations and loads	Several response variables were more strongly correlated with impervious area connection than impervious area extent.
Snyder et al. 2003	Empirical analysis	Riparian land cover	Fish assemblage structure and instream habitat	Though indices of fish assemblage structure were more strongly related to catchment-wide

				land cover than riparian land cover, several measures of instream habitat were more strongly related to riparian land cover.
Strayer et al. 2003	Empirical analysis	Riparian land cover	Nitrate loads; species richness of fish, benthic macroinvertebrates, and aquatic plants	Though most response variables were not better predicted by riparian land cover than by catchment-wide land cover, macroinvertebrate species richness was more strongly related to riparian land cover.
Carlisle and Meador 2007	Empirical analysis	Riparian land cover	Benthic macroinvertebrates	Degraded macro-invertebrate condition in urban settings was associated with reduced riparian forests.
Moore and Palmer 2005	Empirical analysis	Riparian land cover	Macroinvertebrate richness	Macroinvertebrate biodiversity was highly correlated with the extent of riparian forest.
Steedman 1988	Empirical analysis	Riparian land cover	Fish IBI	Fish IBI was more strongly correlated with riparian forest than with catchment-wide forest.
Alberti et al. 2007	Empirical analysis	Land cover aggregation	Benthic IBI	Benthic IBI was highly correlated with mean patch size of impervious areas and mean patch size of forested areas, but these variables were also correlated with the extent of impervious cover.
Hammer 1972	Empirical analysis	Land cover position	Channel enlargement	Channel enlargement was

				highly correlated with the distance of impervious cover from the stream channel.
King et al. 2005	Empirical analysis	Land cover position	Nitrate nitrogen and macroinvertebrate assemblages	Degraded macro-invertebrate condition was better explained by distance-weighted developed land than by catchment-wide developed land.
Goetz and Fiske 2005	Empirical analysis	Land cover position	Benthic IBI	Macro-invertebrate condition was best explained by the distance weighting scheme accounting for the distance of tree cover from the stream channel.
Crosbie et al. 2008	Empirical analysis	Landscape position of vegetation	Vegetation water use	Tree belts in discharge zones used significantly more water than tree belts in recharge zones, pasture in discharge zones, and pasture in recharge zones.
George et al. 1999	Empirical analysis	Landscape position of vegetation	Water table response	In low salinity recharge zones the magnitude of the water table response to tree planting increases as the trees are located further upslope.
Present study	Distributed, process-based model	Landscape position of vegetation	Streamflow response	Annual scale: conversion of upslope lawn to forest produces slightly lower streamflow than conversion of downslope lawn to forest. Seasonal scale: conversion of

				upslope lawn to forest produces significantly lower streamflow in the fall, winter, and spring. Daily scale: conversion of upslope lawn to forest produces lower baseflows in the fall, winter, and spring, and higher peak flows in all seasons.

Appendix B: Figures

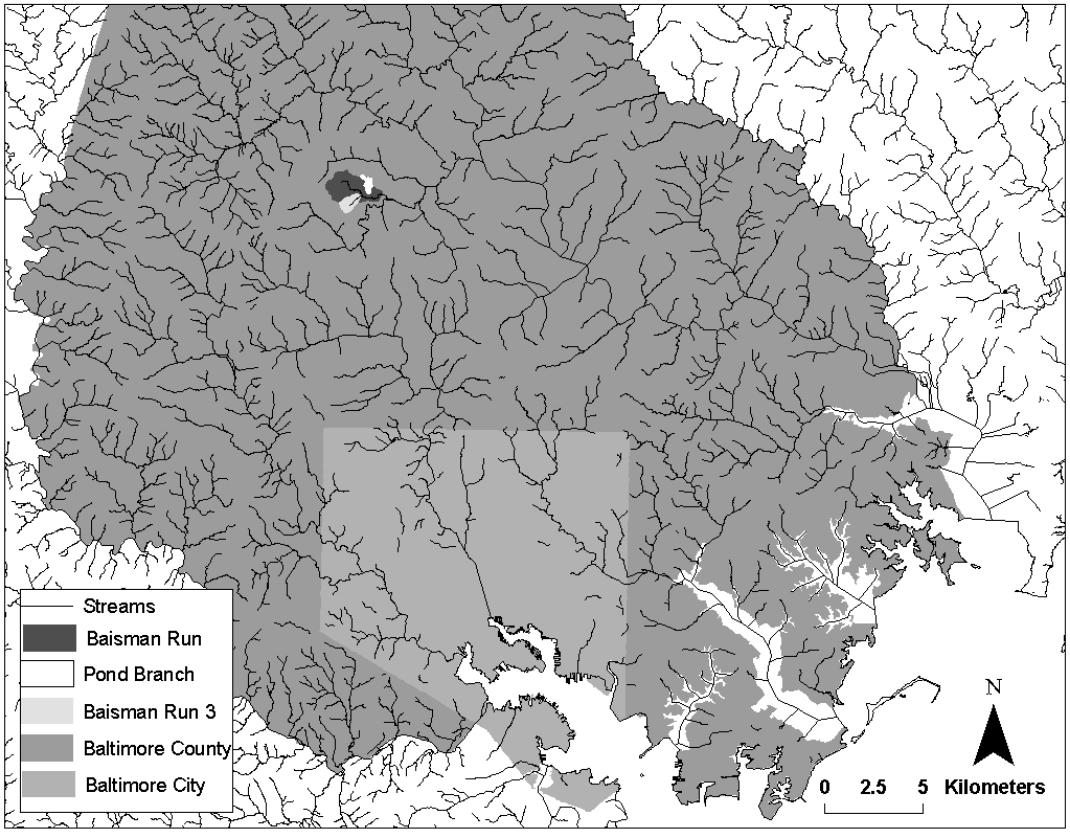


Figure 4. 1: Location of the study catchments within Baltimore County, Maryland.

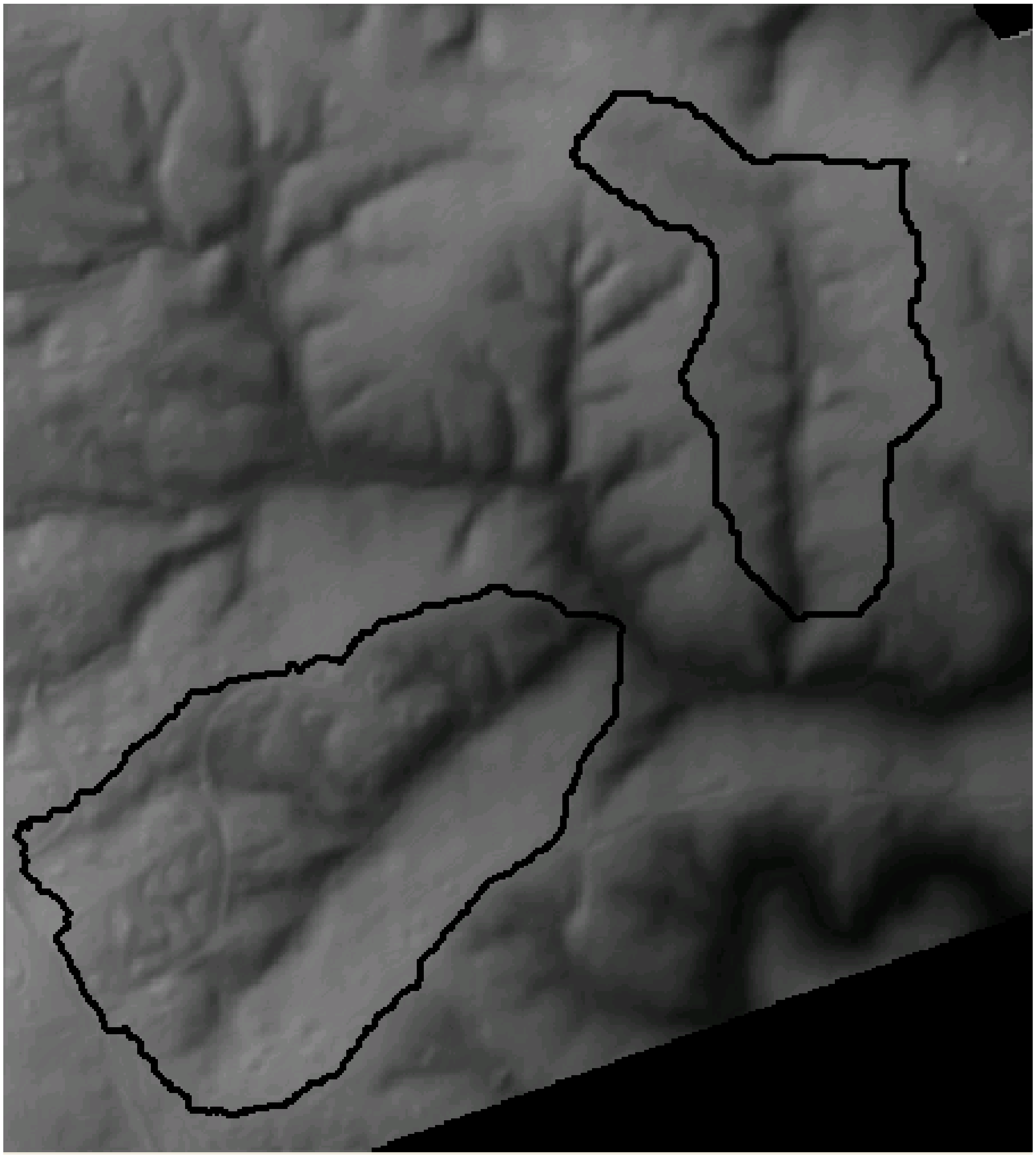


Figure 4. 2: 10 m DEM of the study catchments.

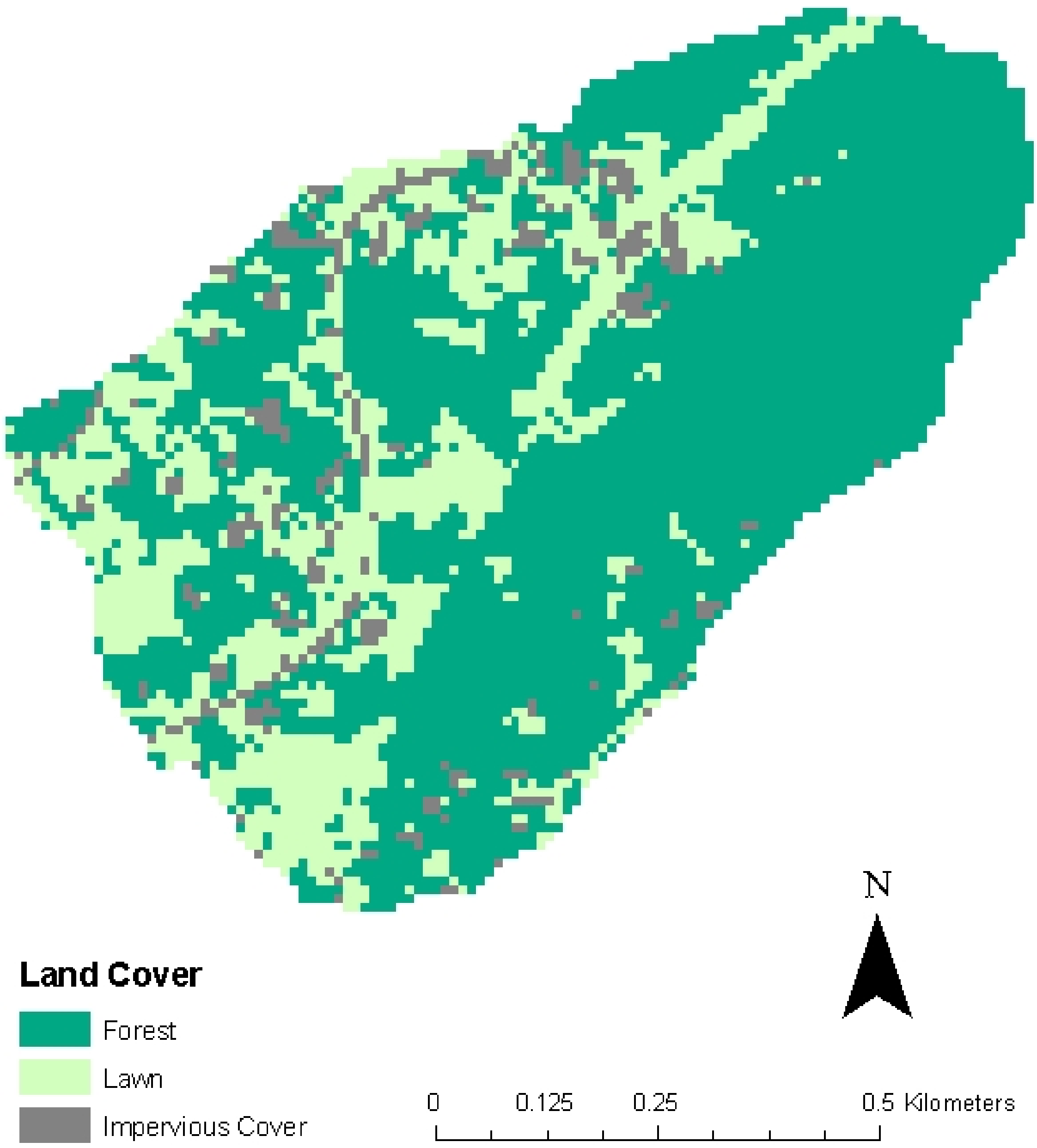


Figure 4. 3: 10 m land cover classification for BR3.

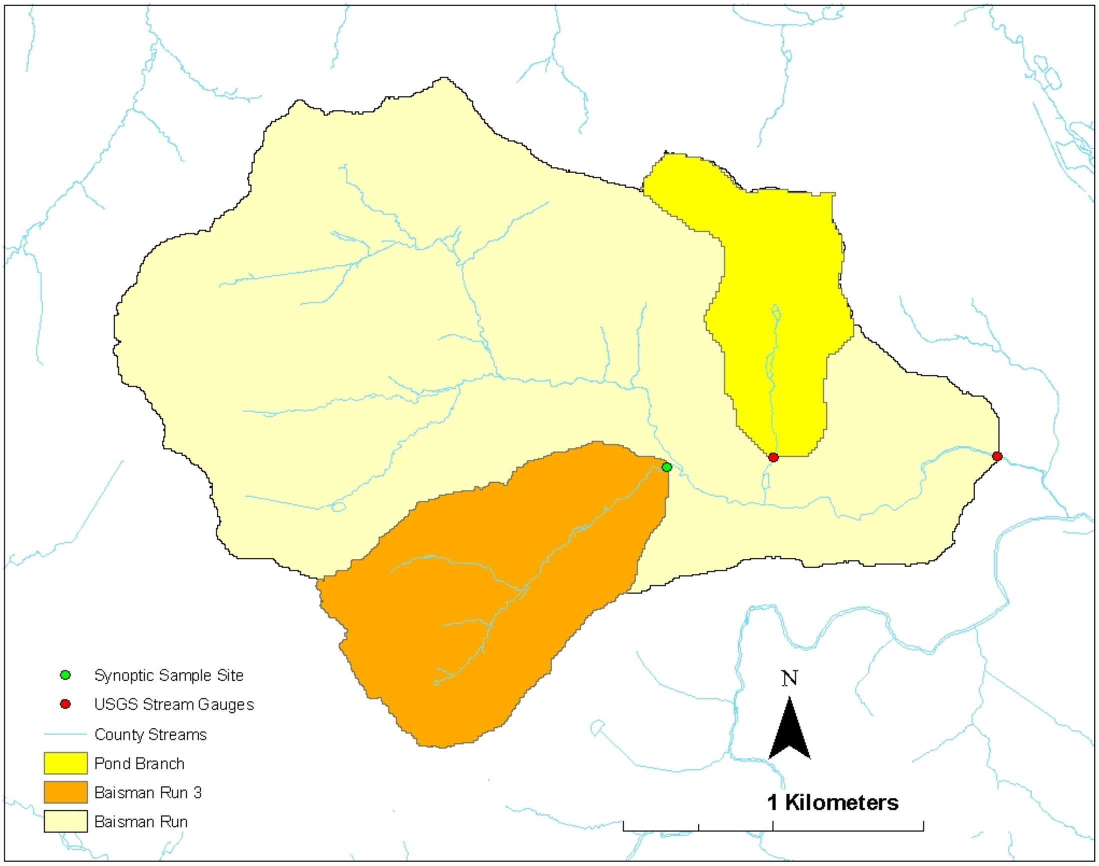


Figure 4. 4: Location of USGS stream gauges and synoptic sample sites.

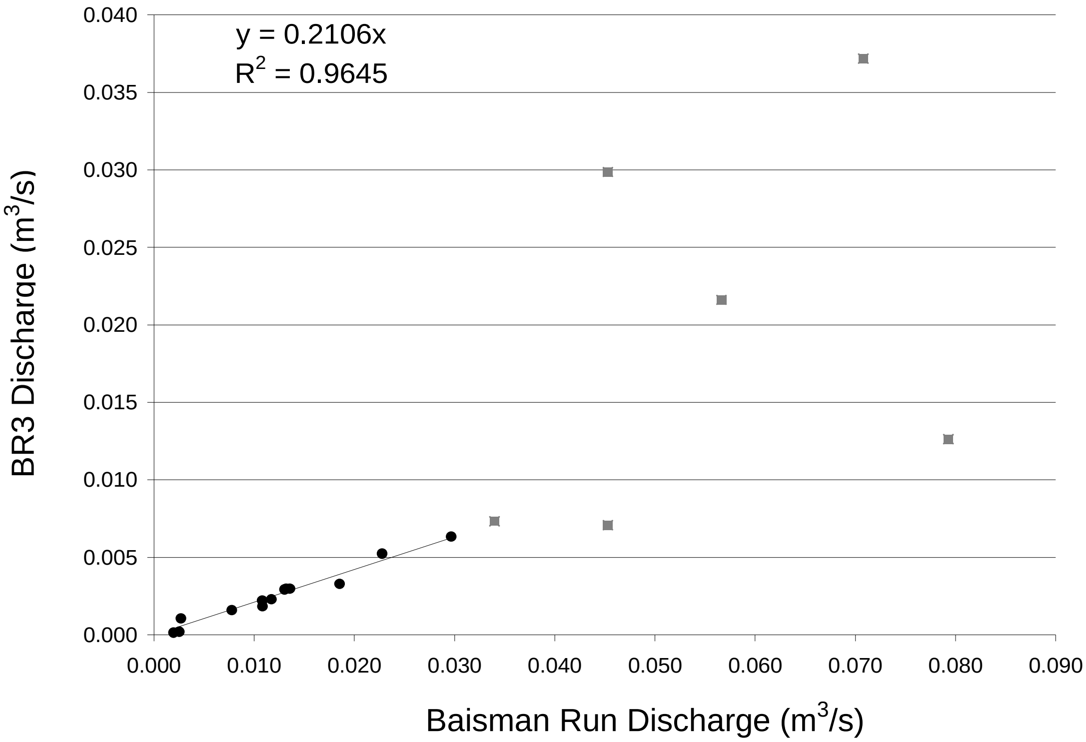


Figure 4. 5: Comparison of the synoptic samples of streamflow collected at the outlet of BR3 to the corresponding discharge recorded by the USGS gauge at the outlet of BR. Black squares represent samples included in the linear regression and gray squares represent samples excluded from the linear regression. While the relationship at lower flows is well approximated by a linear function, the relationship at higher flows may be better characterized by a power function.

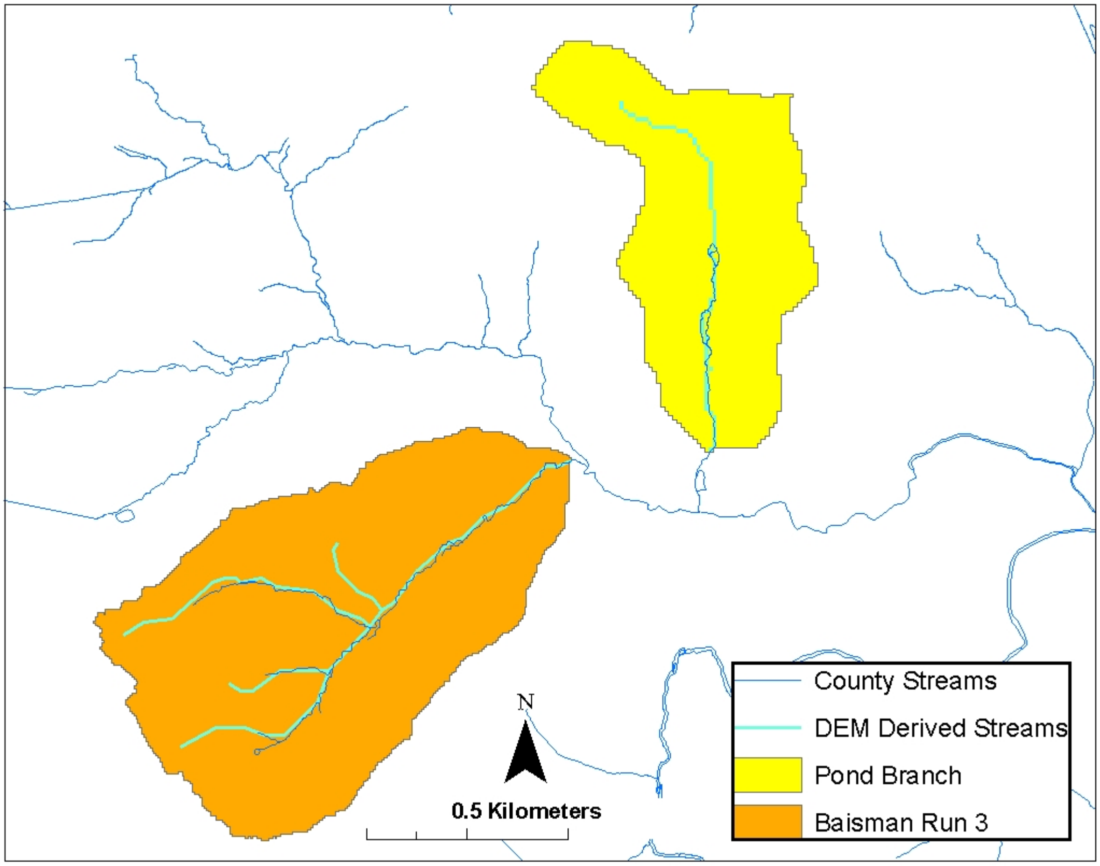


Figure 4. 6: Catchment boundaries and stream network derived in the Terrain Analysis System (TAS). Stream channels were derived using the O'Callaghan and Mark (1984) method with a specific contributing area threshold of 450 m^2, and a stream length threshold of 180 m. The stream channels derived in TAS (light blue) correspond very closely to the stream channels mapped by Baltimore County (dark blue).

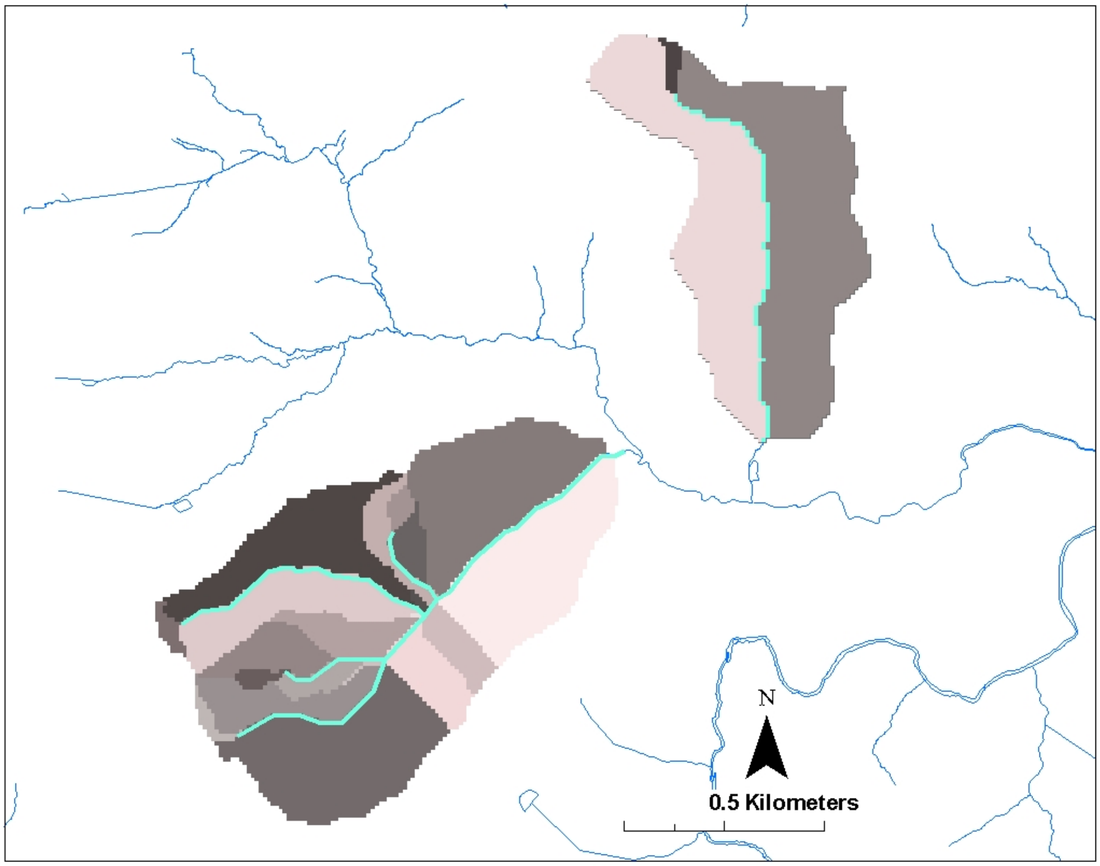


Figure 4. 7: Hillslope boundaries derived in .the Terrain Analysis System.

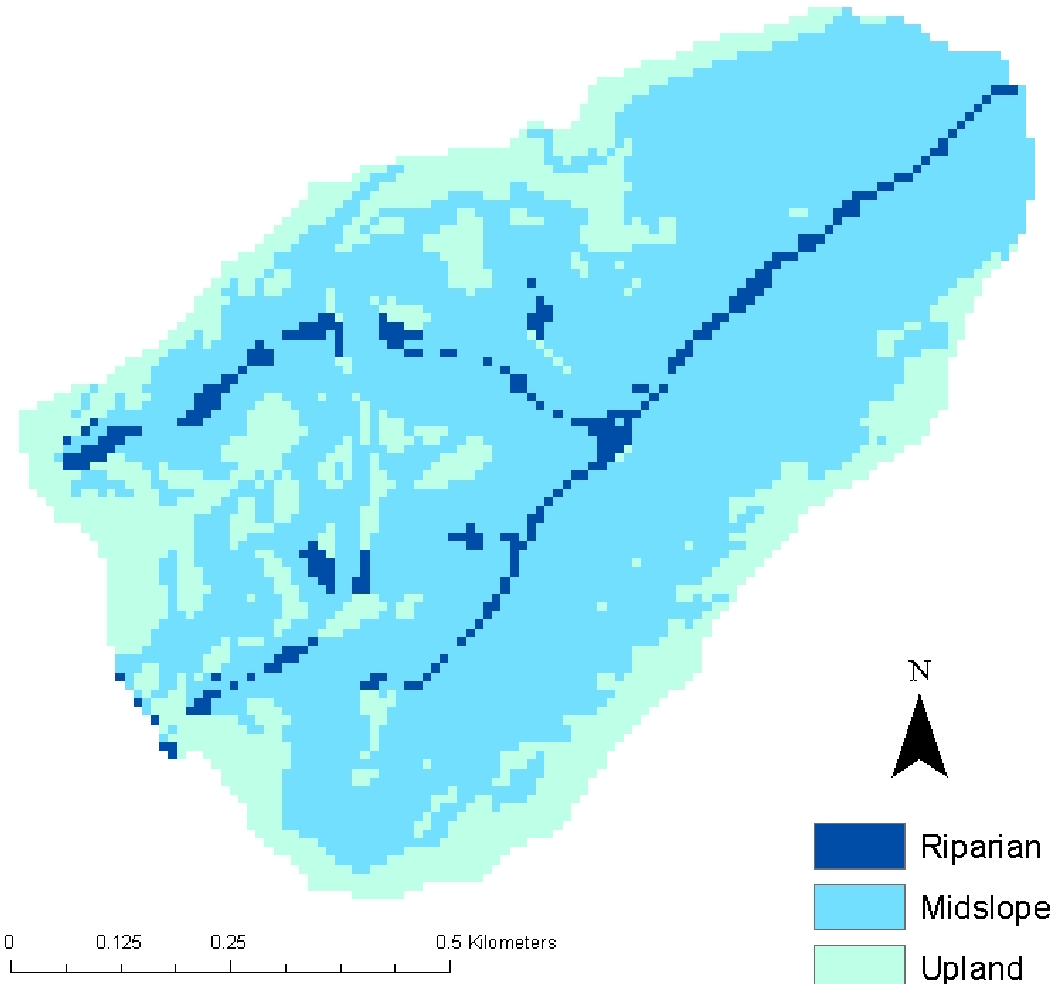


Figure 4. 8: Estimated soil distribution in BR3.

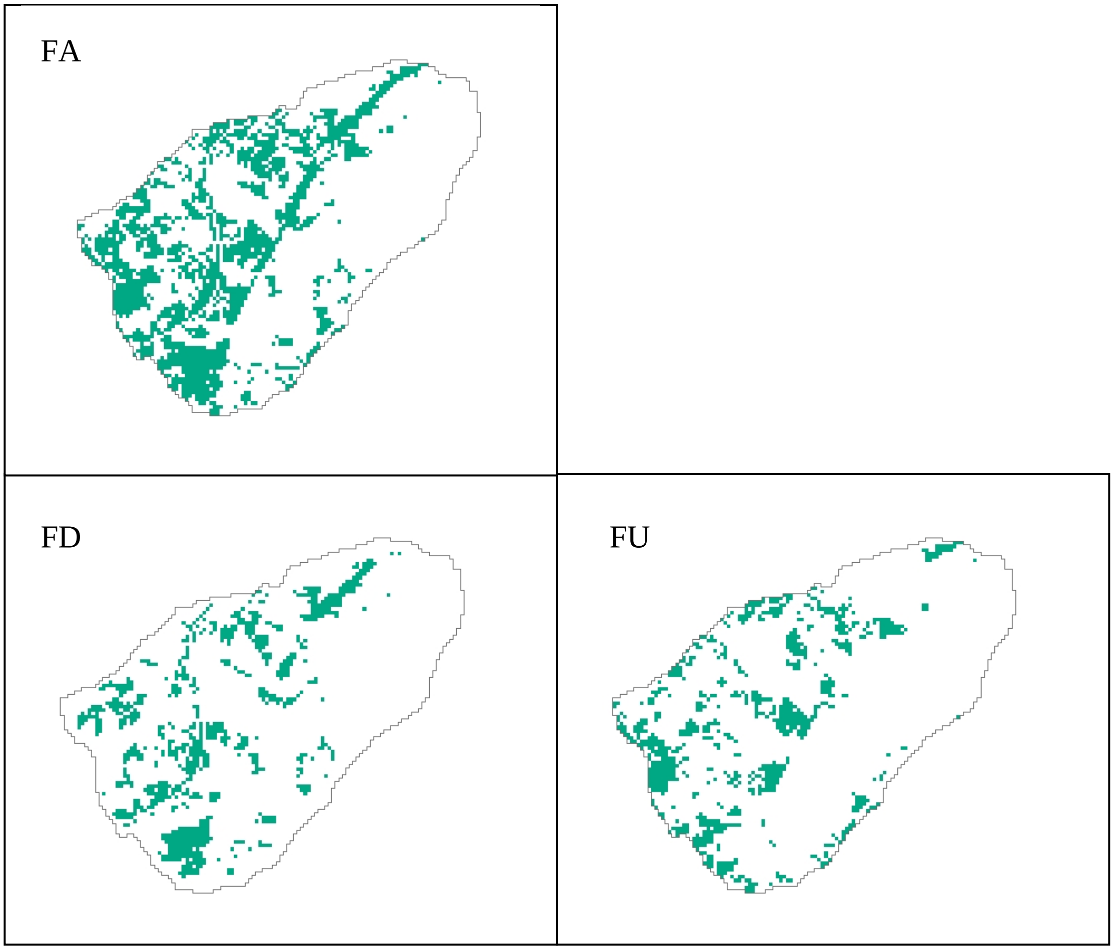


Figure 4. 9: Location of areas converted from lawn to forest for the entirely forested scenario (FA), forested downslope scenario (FD), and forested upslope scenario (FU).

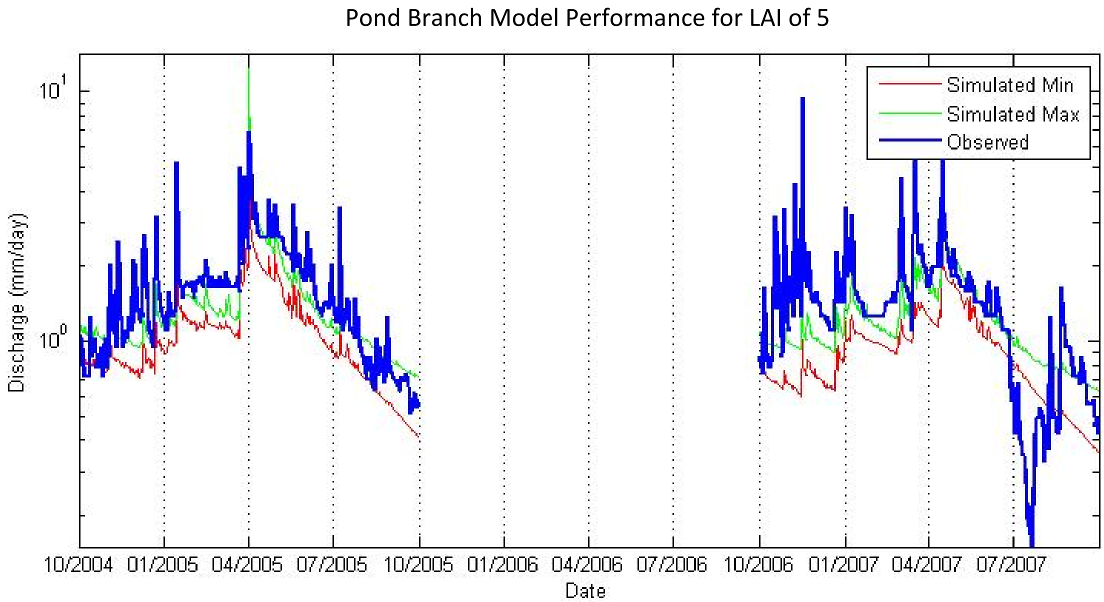


Figure 5. 1: Observed and predicted discharge from Pond Branch for the simulations in which LAI was assigned a value of 5. The range of predicted discharges includes only the predictions of those parameter sets yielding Nash Sutcliffe efficiencies greater than 0.4. The observed daily discharge is shown in blue, the lower bound of the predicted daily discharge in red, and the upper bound of the predicted daily discharge in green. When LAI is assigned a value of 5, the model consistently underpredicts daily discharge.

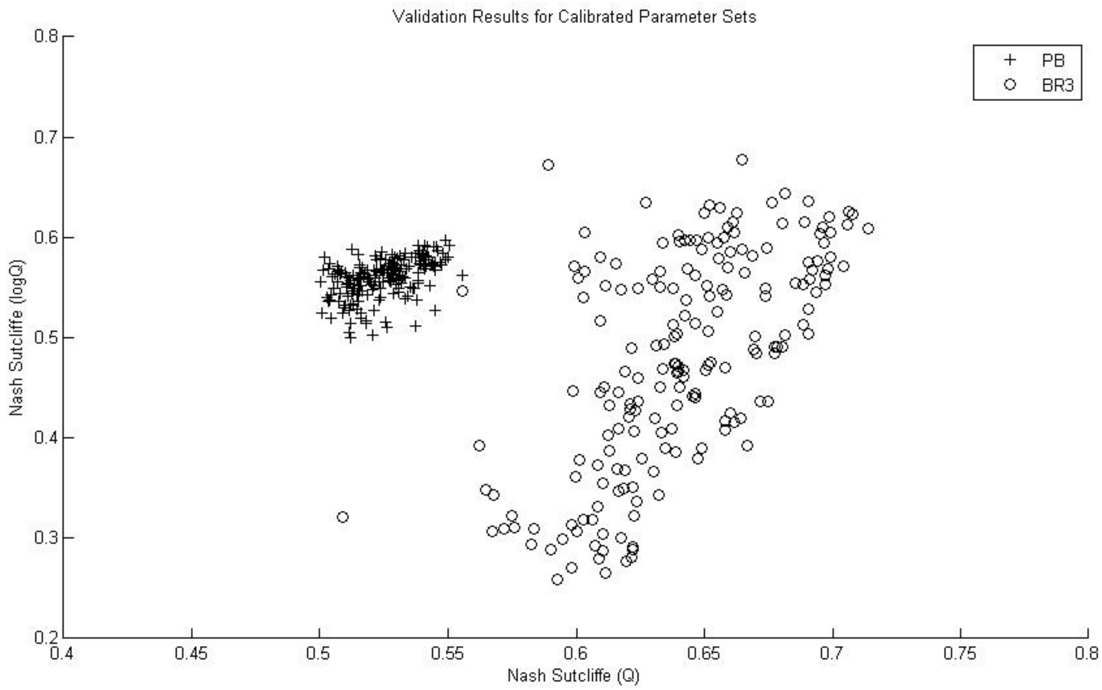


Figure 5. 2: Nash Sutcliffe efficiencies of the parameter sets meeting the performance criteria for Pond Branch. Behavioral parameter sets often predict streamflow more accurately for BR3 (circles) than for PB (crosses).

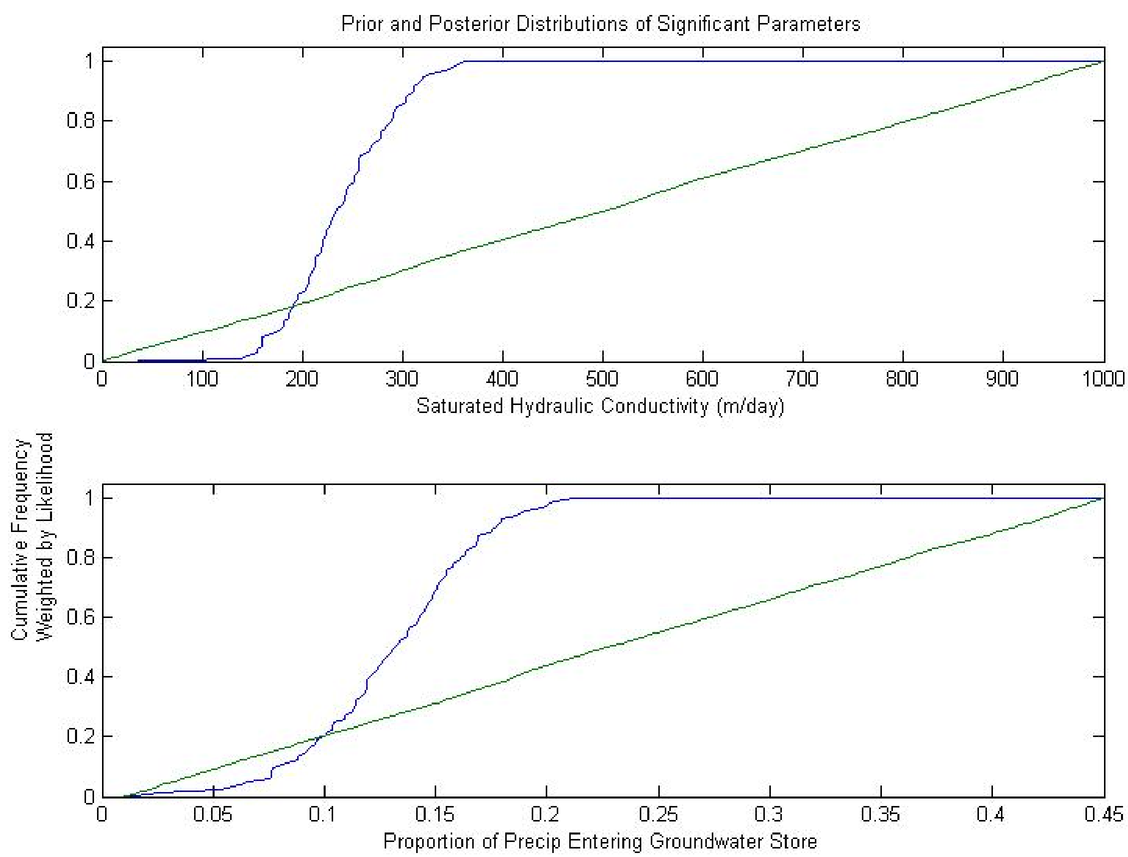


Figure 5. 3: Prior and posterior distributions of the soil and groundwater parameters most sensitive to calibration. Prior and posterior distributions were generated with the GLUE methodology.

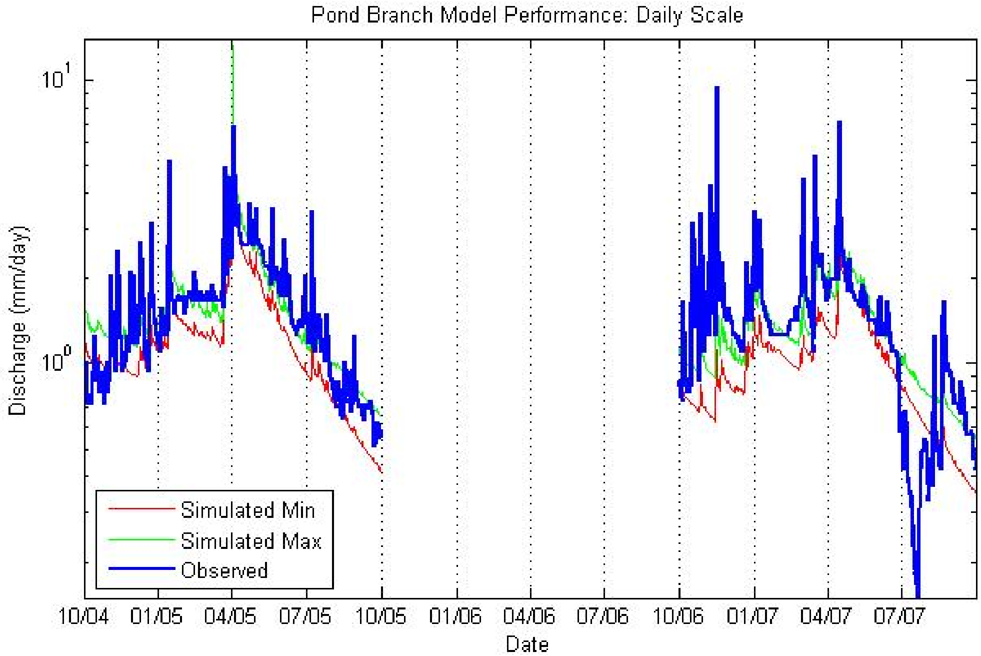


Figure 5. 4a: Observed and predicted daily discharges from Pond Branch for the simulations in which LAI was assigned a value of 4.5 in upland areas and 2.5 in riparian areas. The range of predicted discharges includes only the predictions of behavioral parameter sets. The observed daily discharge is shown in blue, the lower bound of the predicted daily discharge in red, and the upper bound of the predicted daily discharge in green. When LAI is assigned values of 4.5 and 2.5, the model predictions bound observations for a significant portion of the simulation period. Transmission losses may account for the significant overprediction of streamflow in July and August of 2007.

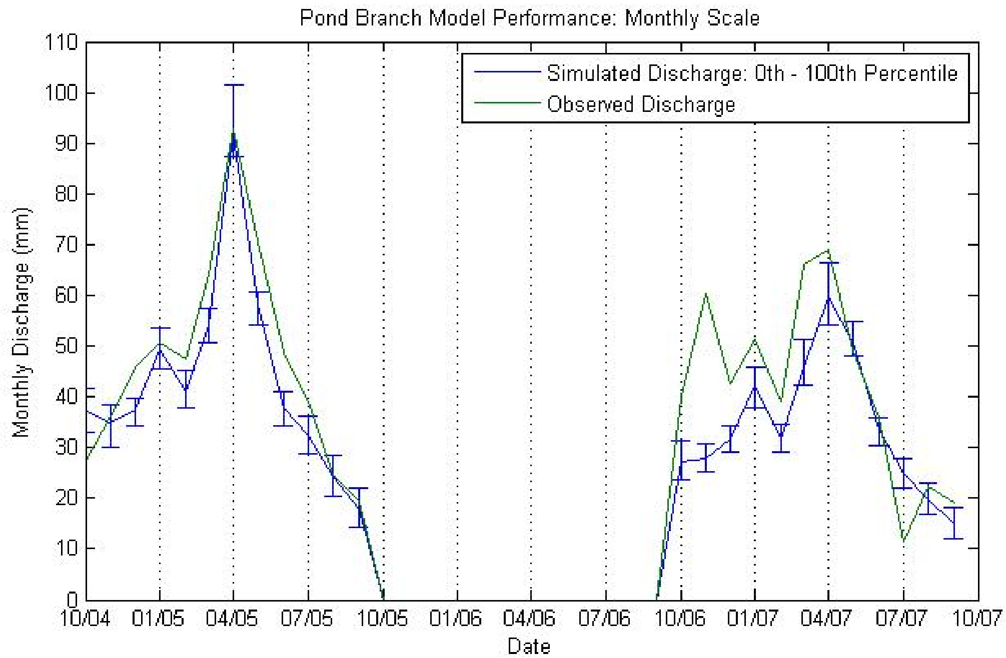


Figure 5.4b: Observed and predicted monthly discharges from Pond Branch for all behavioral parameter sets. The observed monthly discharge is shown in green and the simulated monthly discharge is shown in blue, with the curve indicating the expected values, and the bars indicating the range of predicted values. Though the model tends to underpredict monthly discharge, it reproduces the trends in discharge quite well.

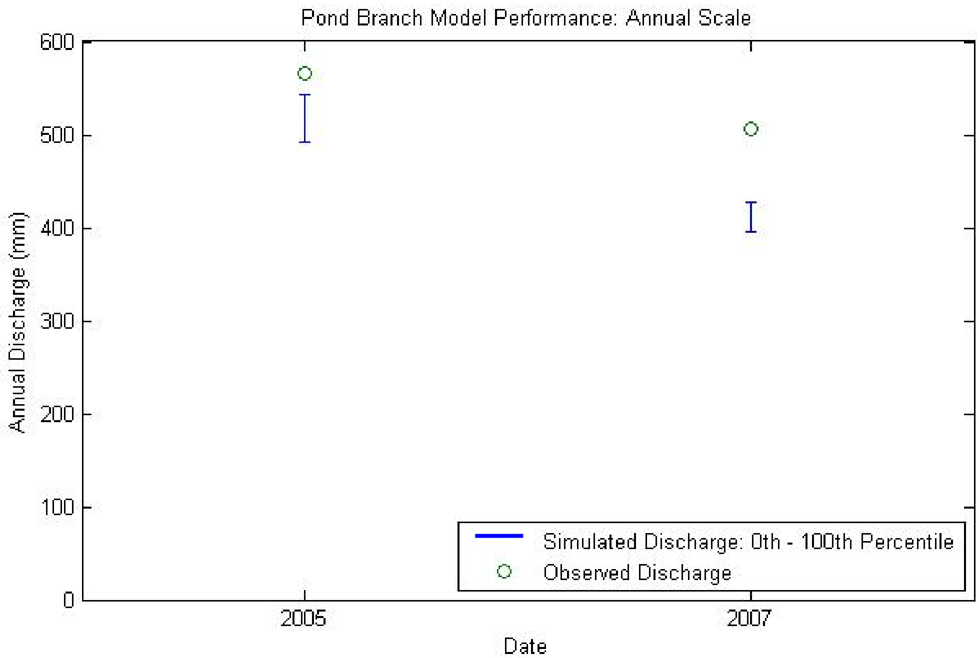


Figure 5.4c: Observed and predicted annual discharges from Pond Branch for all behavioral parameter sets. The observed annual discharge is shown in green and the simulated annual discharge is shown in blue, with the bars indicating the range of predicted values. The model tends to underpredict annual discharge.

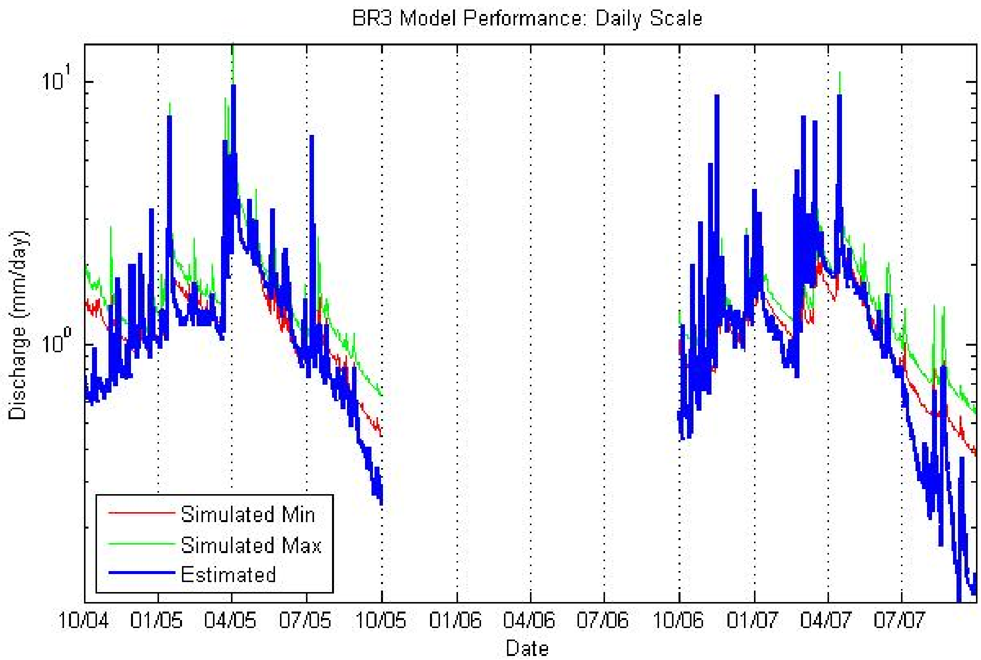


Figure 5. 5a: Estimated and predicted daily discharges from Baisman Run 3 for all behavioral parameter sets. The estimated daily discharge is shown in blue, the lower bound of the predicted daily discharge in red, and the upper bound of the predicted daily discharge in green Though the model occasionally overpredicts discharge, it appears to perform better than the model of PB.

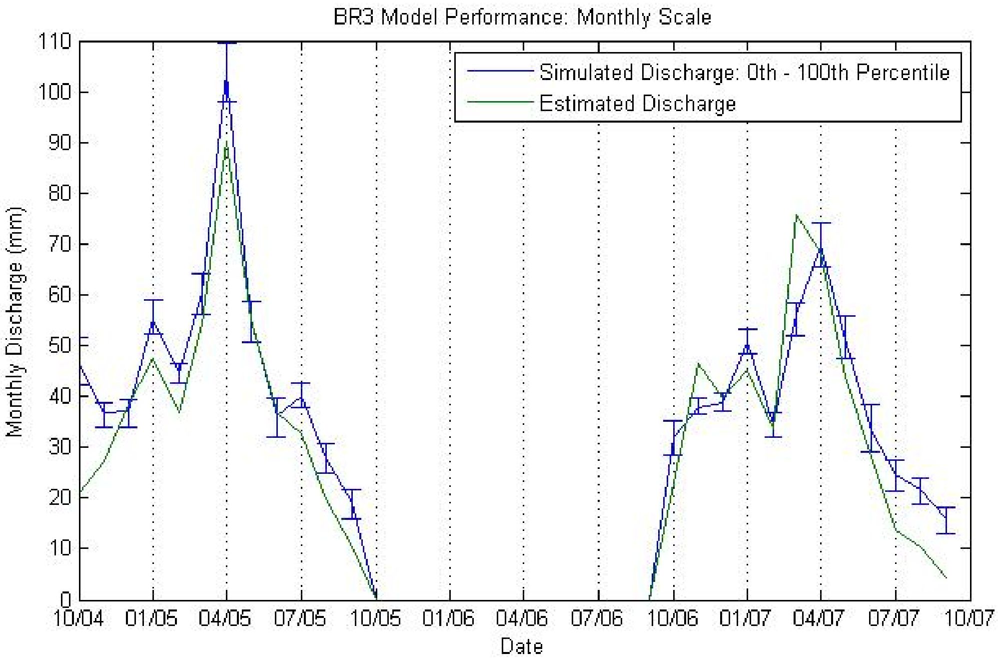


Figure 5.5b: Estimated and predicted monthly discharges from Baisman Run 3 for all behavioral parameter sets. The estimated monthly discharge is shown in green and the simulated monthly discharge is shown in blue, with the curve indicating the expected values, and the bars indicating the range of predicted values. Though the model tends to overpredict monthly discharge, it reproduces the trends in discharge quite well.

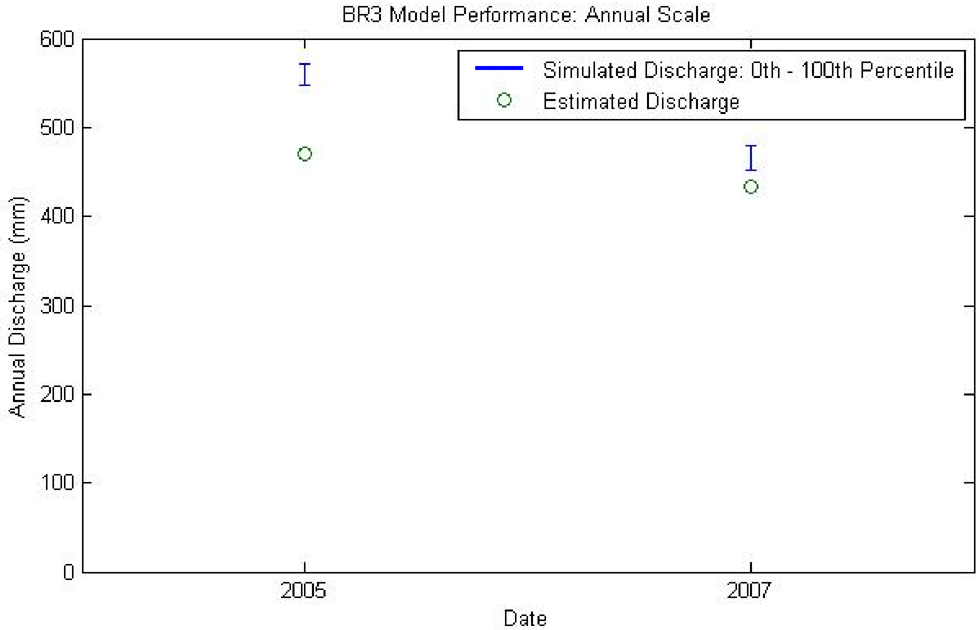


Figure 5.5c: Estimated and predicted annual discharges from Baisman Run 3 for all behavioral parameter sets. The estimated annual discharge is shown in green and the simulated annual discharge is shown in blue, with the bars indicating the range of predicted values. The model tends to overpredict annual discharge.

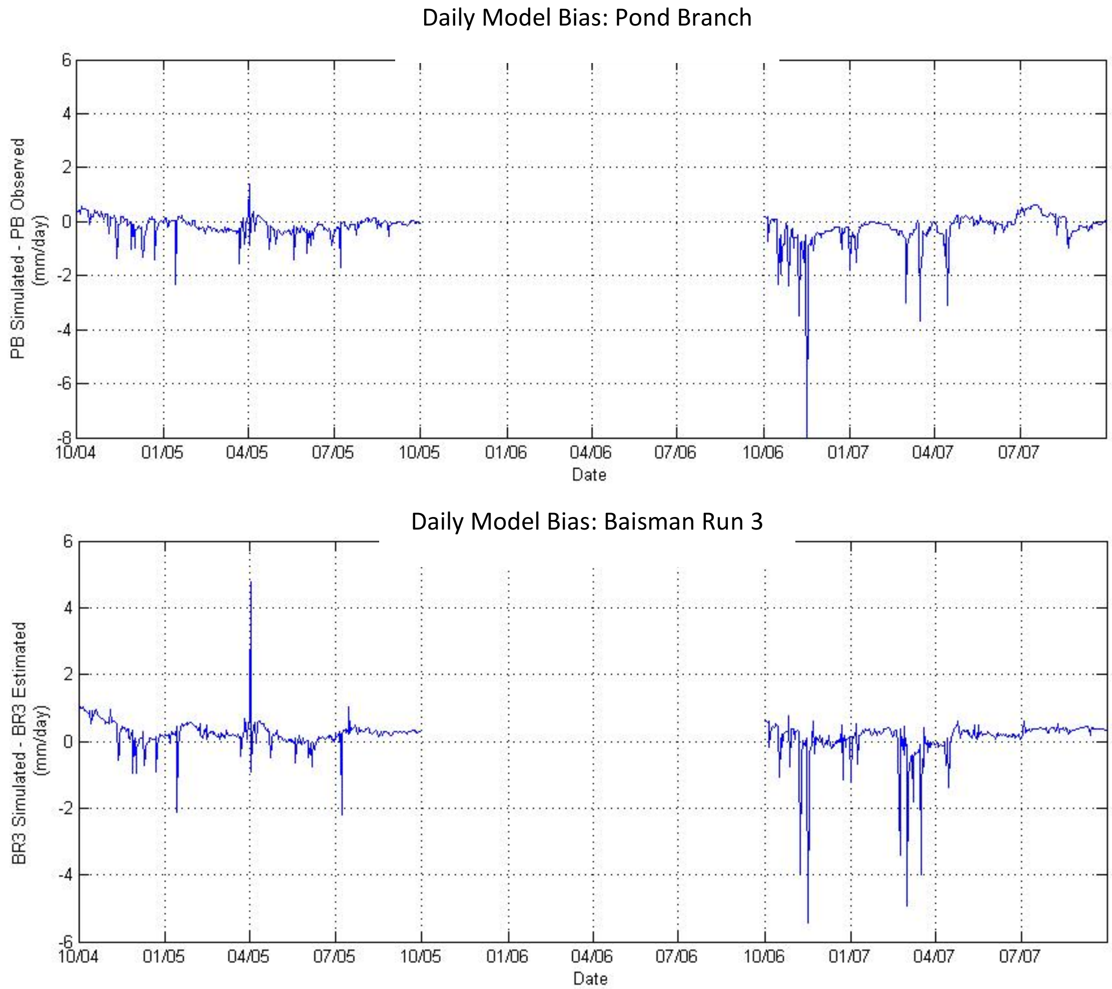


Figure 5. 6: Difference between simulated expected discharge and observed discharge for PB (above) and BR3 (below). While peak flows are underpredicted for both catchments, base flows are underpredicted for PB and overpredicted for BR3.

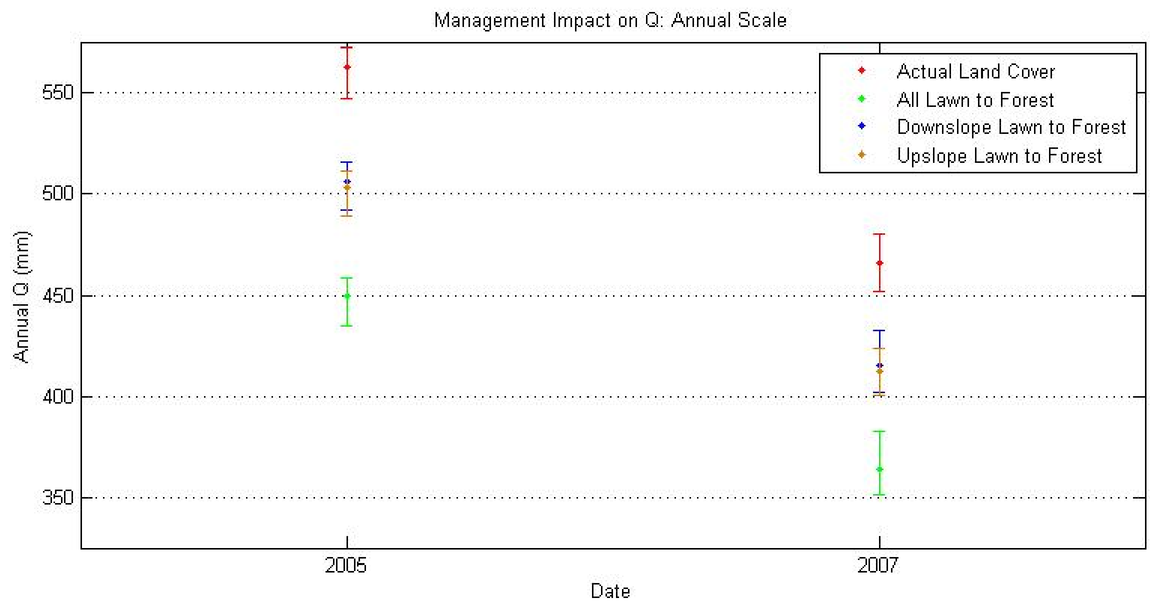


Figure 5. 7: Simulated range of annual runoff volumes for all vegetation management scenarios. The circles show the expected value, and the bars show the range of predicted values for all behavioral parameter sets. Annual streamflow declines dramatically when half or all of the lawn area is converted to forest. When all lawn is converted to forest, annual streamflow declines by ~20%.

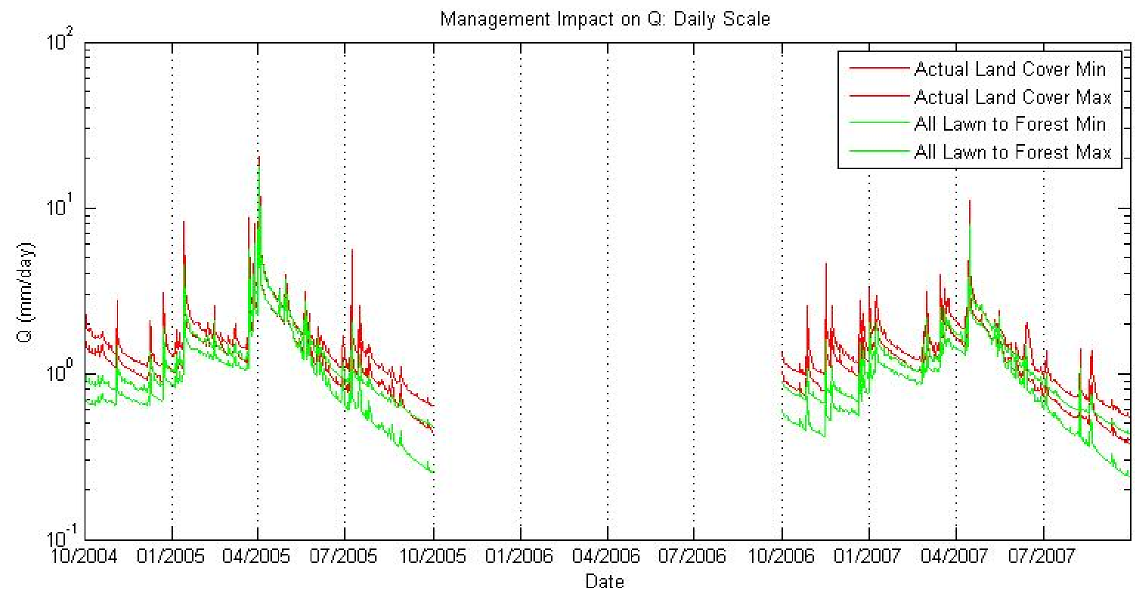


Figure 5. 8: Simulated range of daily discharge for the actual (red) and entirely forested (green) scenarios. For each scenario, the upper and lower bounds show the range of predicted discharges for all behavioral parameter sets. While the difference in simulated streamflow exceeds the uncertainty associated with parameter estimation in the summer, fall, and winter, the difference is negligible in the spring.

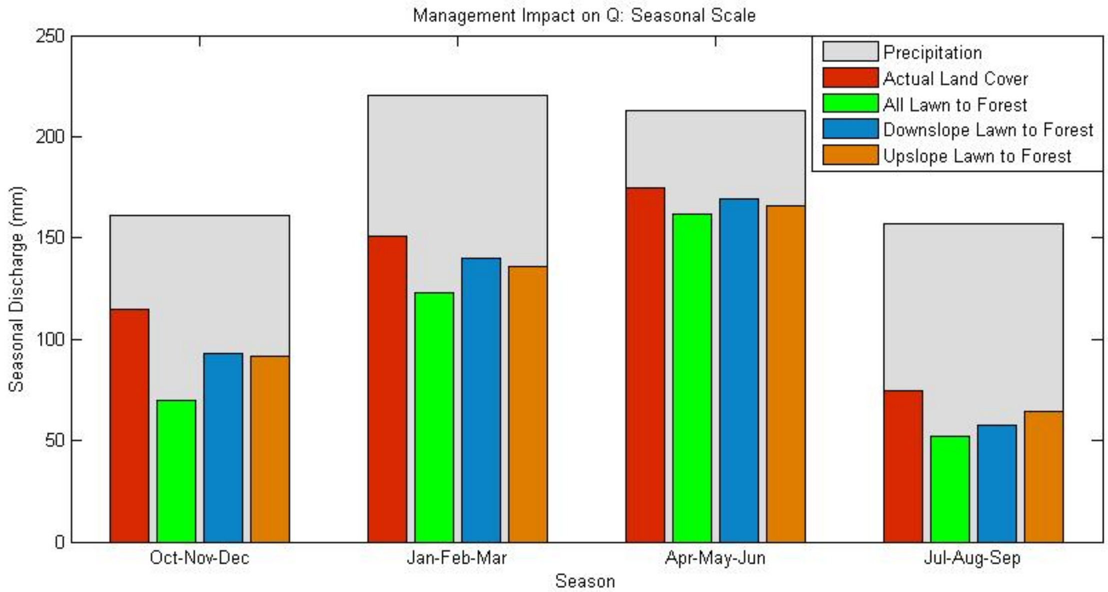


Figure 5. 9a: Comparison of seasonal runoff depths and seasonal precipitation depths for all vegetation management scenarios. For all three reforested scenarios, the predicted reduction in seasonal discharge is greatest in the fall, declines through the winter and spring, and increases again in the summer. Also, for all seasons the predicted reduction in runoff is greatest for the entirely forested scenario.

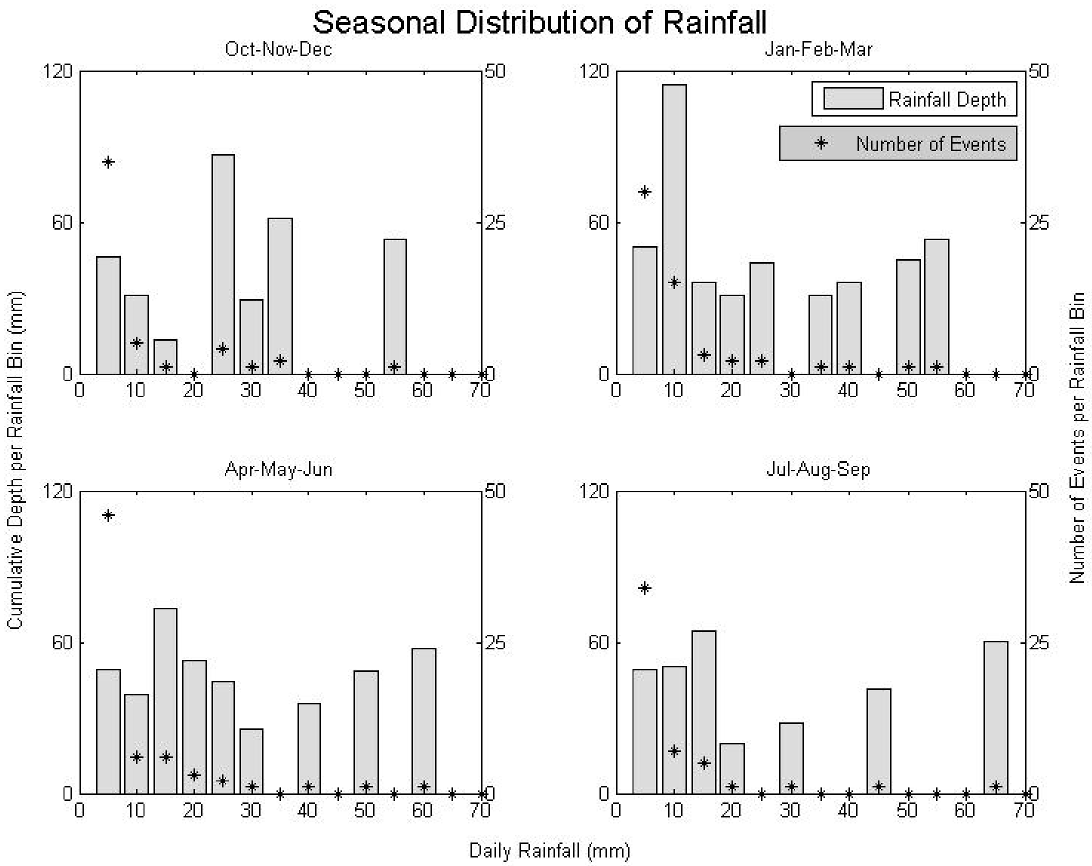


Figure 5.9b: Distribution of precipitation in each season. The stars show the frequency of events of different magnitude, while the bars show the cumulative depth provided by events of different magnitude. No correspondence is observed between the seasonal distribution of precipitation (this figure) and the seasonal reduction in discharge (Figure 5.9a).

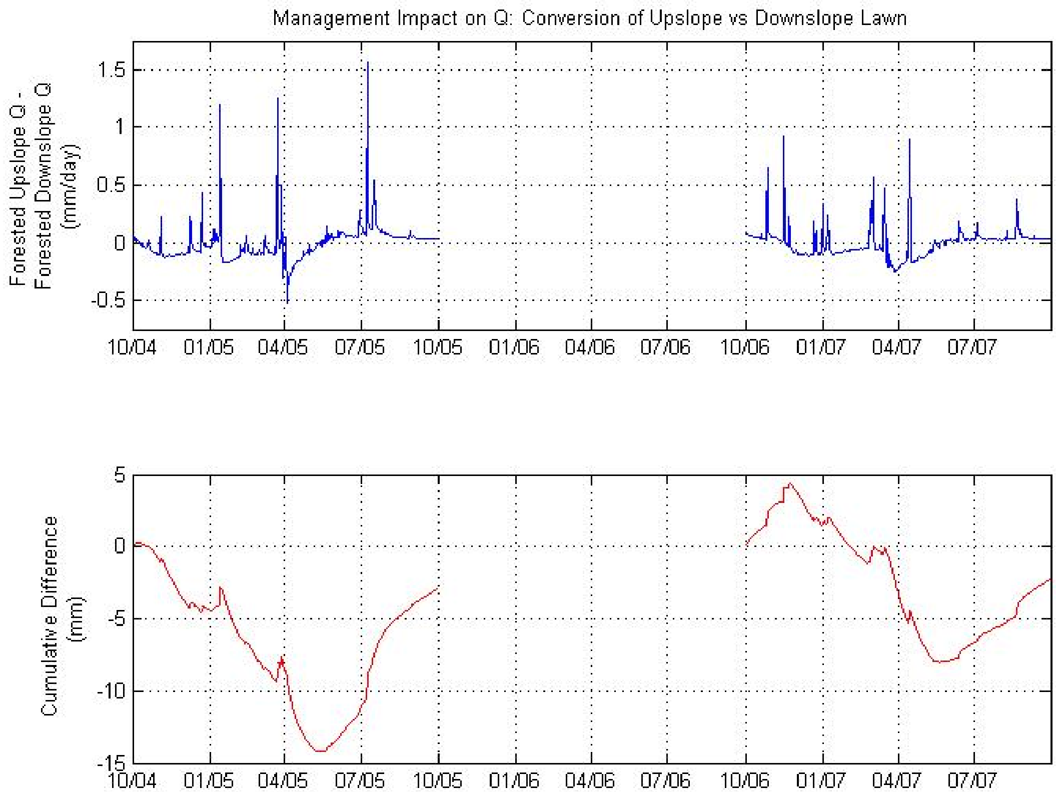


Figure 5. 10: This figure plot the difference in simulated daily discharge between the forested upslope and forested downslope scenarios (top pane) and the cumulative difference in simulated discharge (bottom pane). While baseflows from the forested downslope scenario are higher in the fall, winter, and spring, baseflows from the forested upslope scenario are higher in the summer. The cumulative difference in streamflow increases through the fall, winter, and spring, and decreases in the summer.

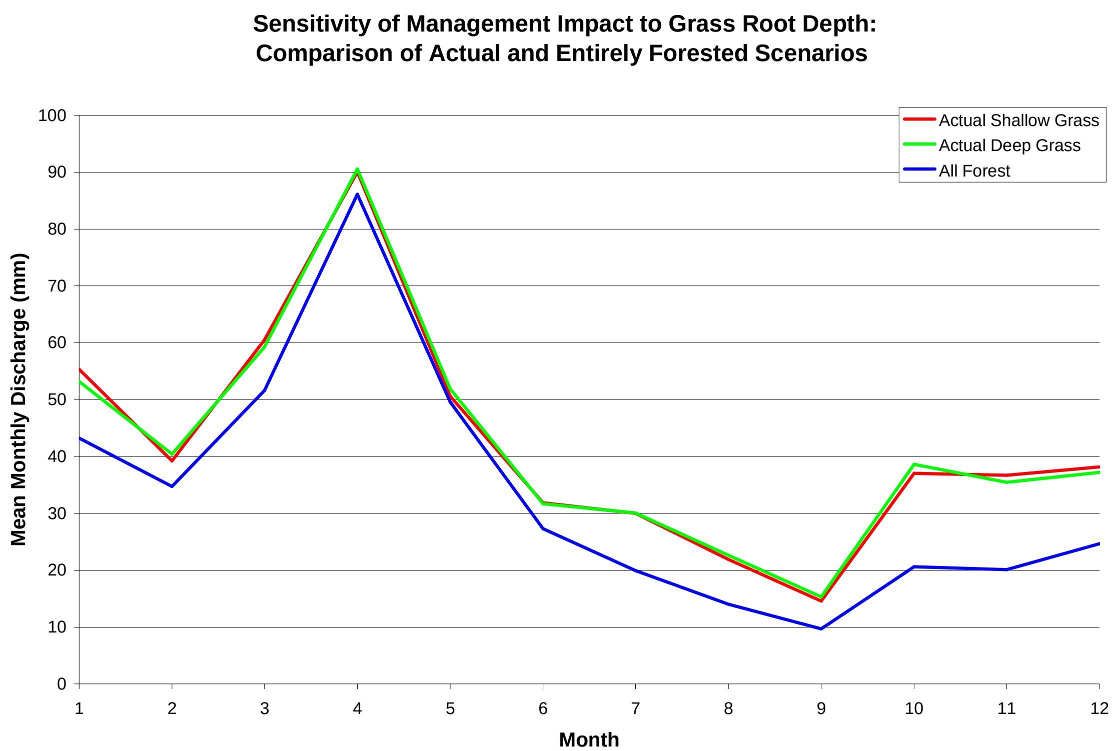


Figure 5. 11: Comparison of simulated monthly discharge for the entirely forested scenario (blue), the actual land use scenario with shallow grass roots (red), and the actual land use scenario with deep grass roots (green). The impact of forest extent on streamflow response is robust across the rooting depths examined.

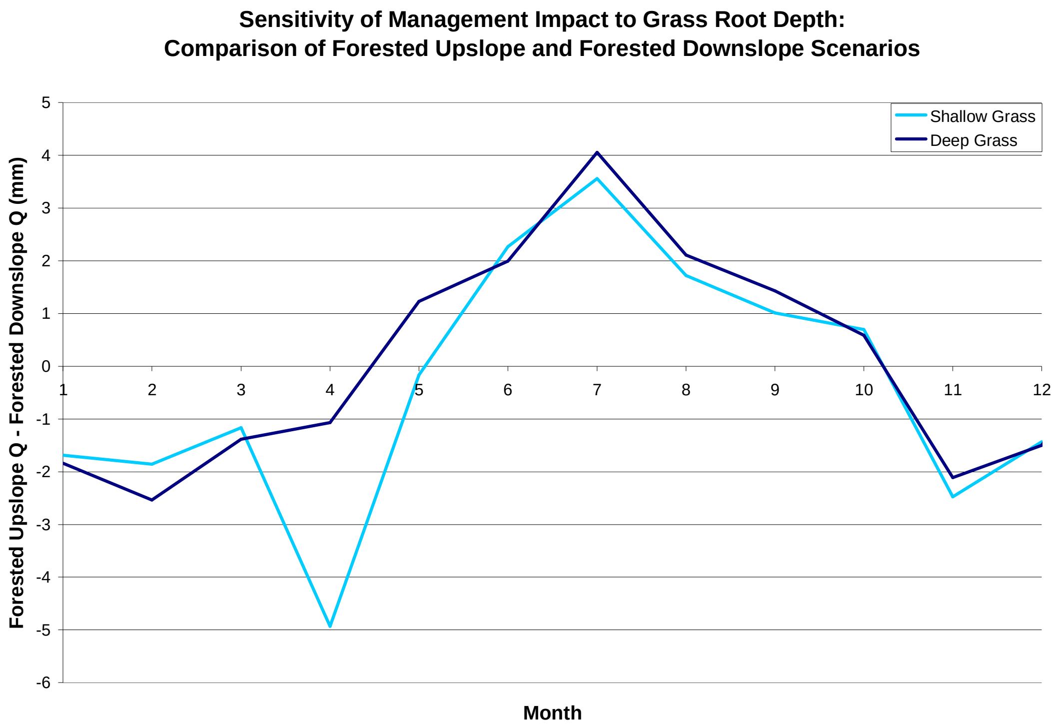


Figure 5. 12: This figure plots the difference in monthly discharge between the forested upslope and forested downslope scenarios for the simulation with shallow grass roots (light blue) and the simulation with deep grass roots (dark blue). The impact of the topographic position of reforested areas on streamflow response is also observed to be robust across the rooting depths examined.

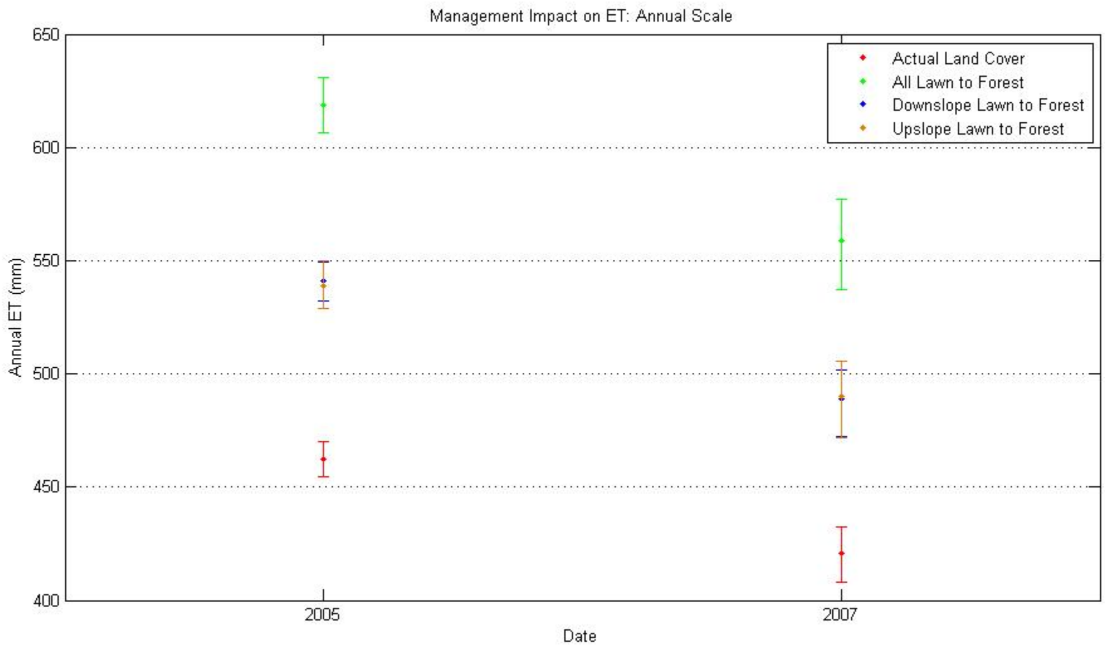


Figure 5. 13: Simulated range of annual evapotranspiration volumes for all vegetation management scenarios. The circles show the expected values, and the bars show the range of predicted values for all behavioral parameter sets. Annual ET increases dramatically when half or all of the lawn area is converted to forest. ET increases by ~150 mm when all lawn area is converted to forest, and by ~75 mm when half the lawn area is converted to forest. The topographic position of re-forested areas does not appear to have a significant impact on annual ET.

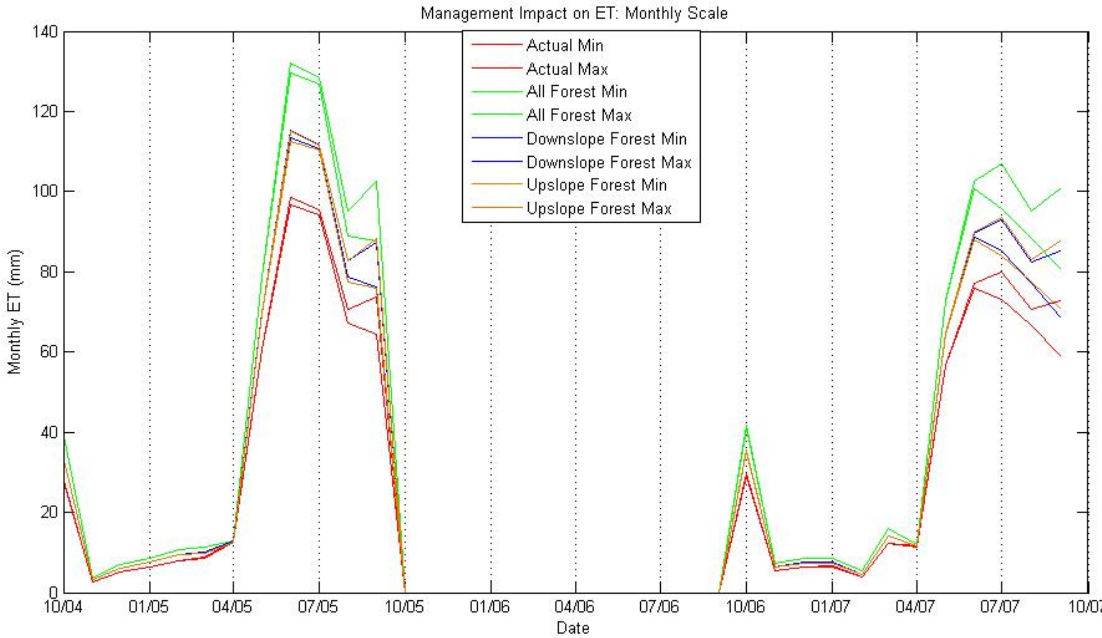
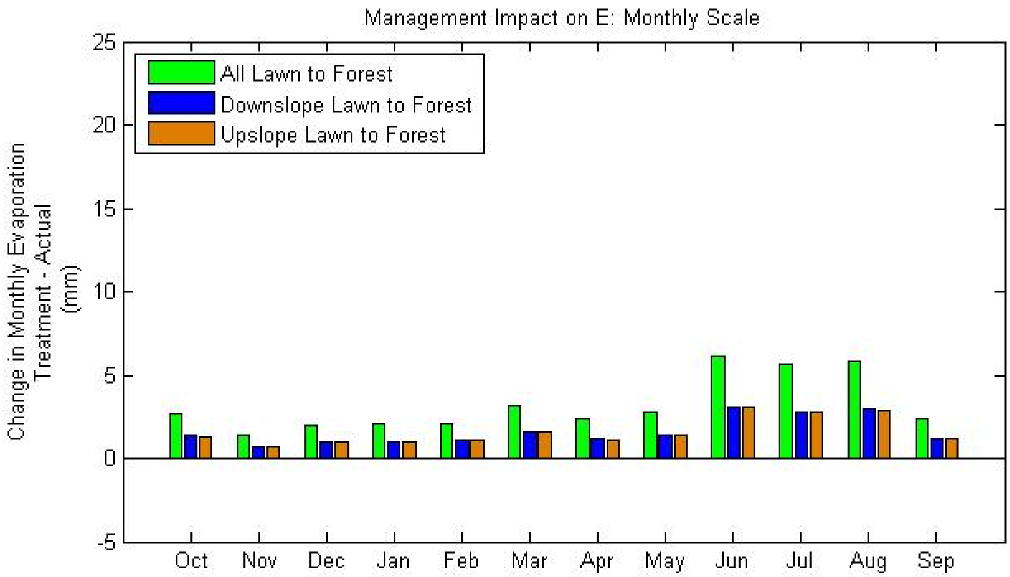


Figure 5. 14: Simulated range of monthly ET for all vegetation management scenarios. For each scenario, the upper and lower bounds show the range of predicted discharges for all behavioral parameter sets. Differences in monthly ET among the 4 scenarios are greatest in the growing season and negligible in the winter and fall.

a)



b)

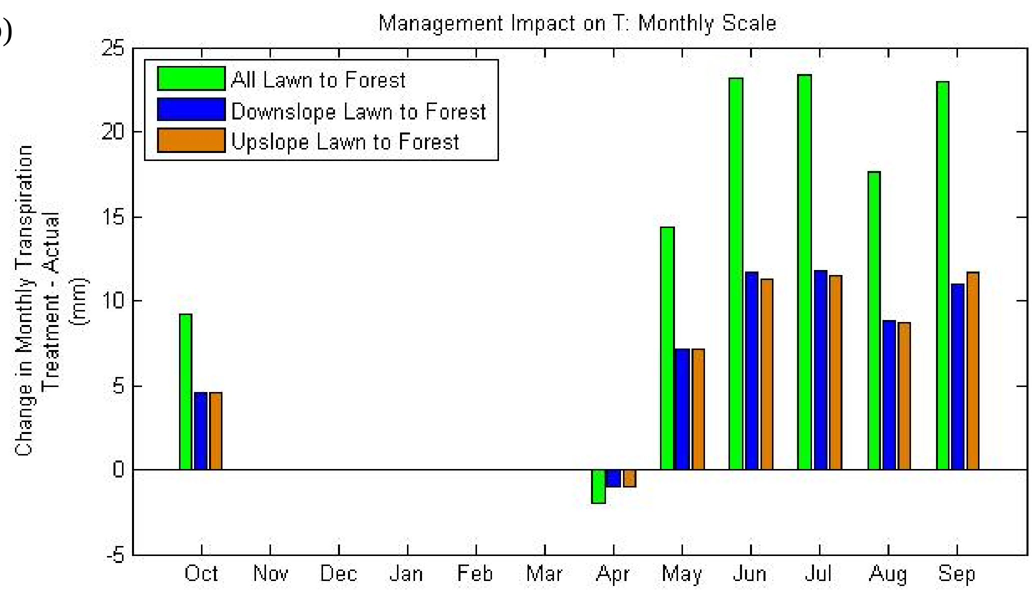


Figure 5. 15: Expected change in monthly evaporation (a) and transpiration (b) for the three reforested scenarios. Transpiration is observed to be the dominant mechanism determining the impact of vegetation extent on catchment hydrologic response.

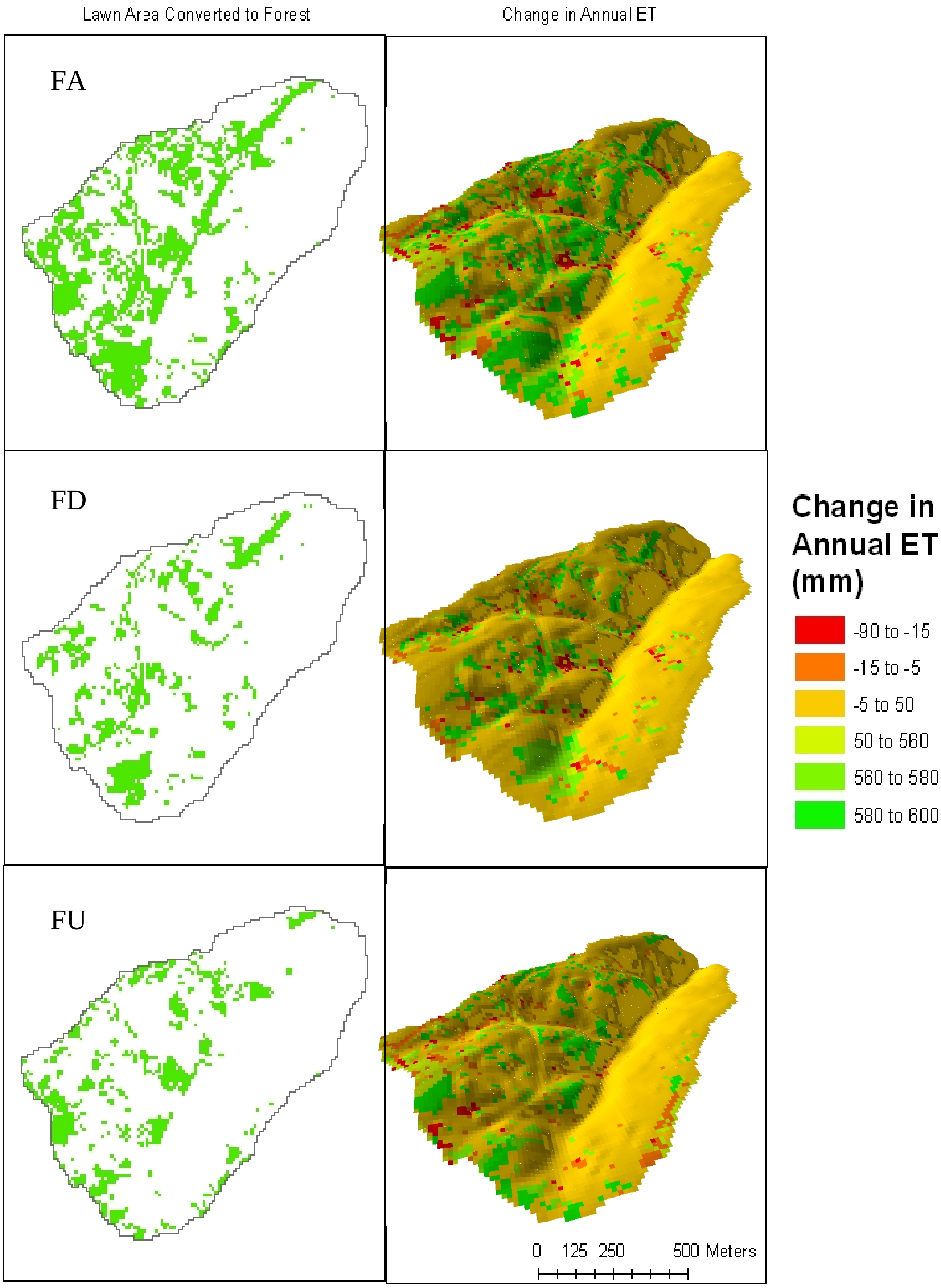


Figure 5. 16: Estimated spatial distribution of the expected change in annual ET for the three reforested scenarios. For each scenario the distribution of increases in ET mirrors the distribution of the lawn area converted to forest.

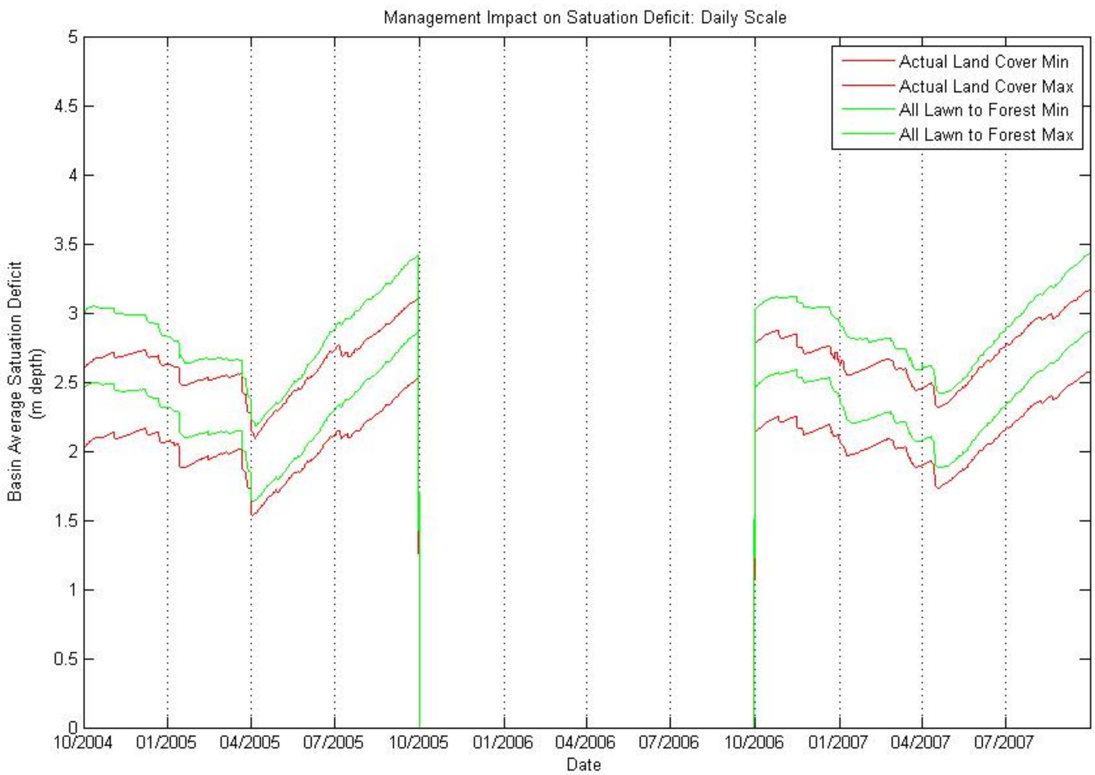


Figure 5. 17: Simulated range of daily saturation deficit for the actual and entirely forested scenarios. For each scenario, the upper and lower bounds show the range of predicted discharges for all behavioral parameter sets. Average saturation deficit for the entirely forested scenario exceeds average saturation deficit for the actual scenario throughout the year.

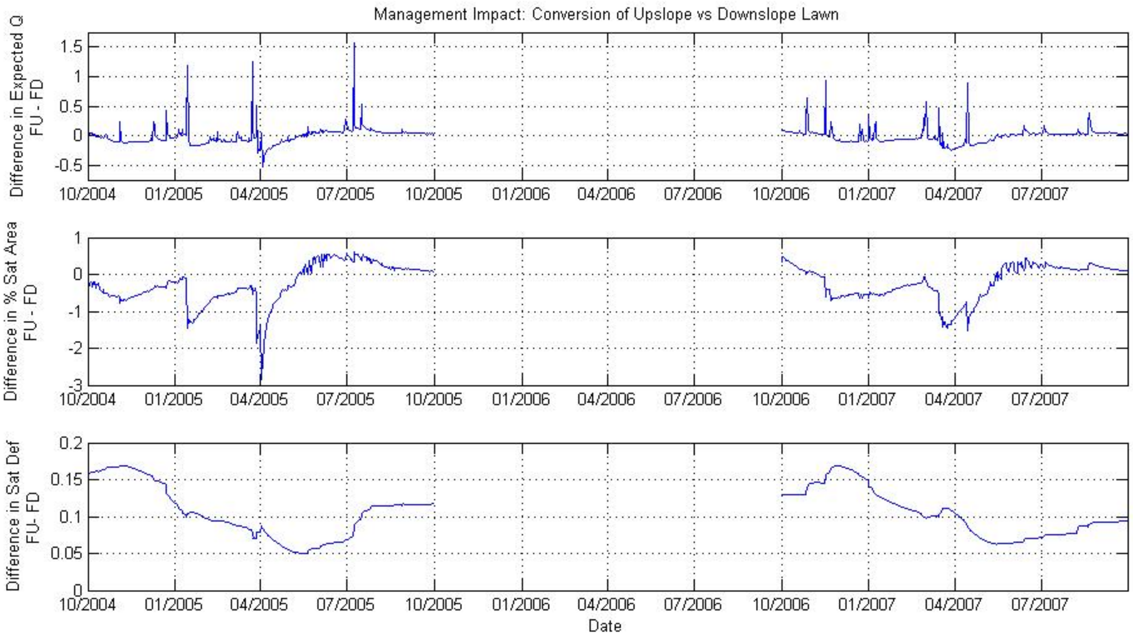


Figure 5. 18: This figure shows the impact of land cover pattern on streamflow (top pane), percent saturated area (middle pane), and catchment average saturation deficit (bottom pane). The middle pane plots the difference in daily % saturated area between the forested upslope and forested downslope scenarios, while the bottom pane plots the difference in average saturation deficit between the two scenarios. The impact of vegetation pattern on the extent of catchment saturation is seen to be related to the impact of vegetation pattern on streamflow response.

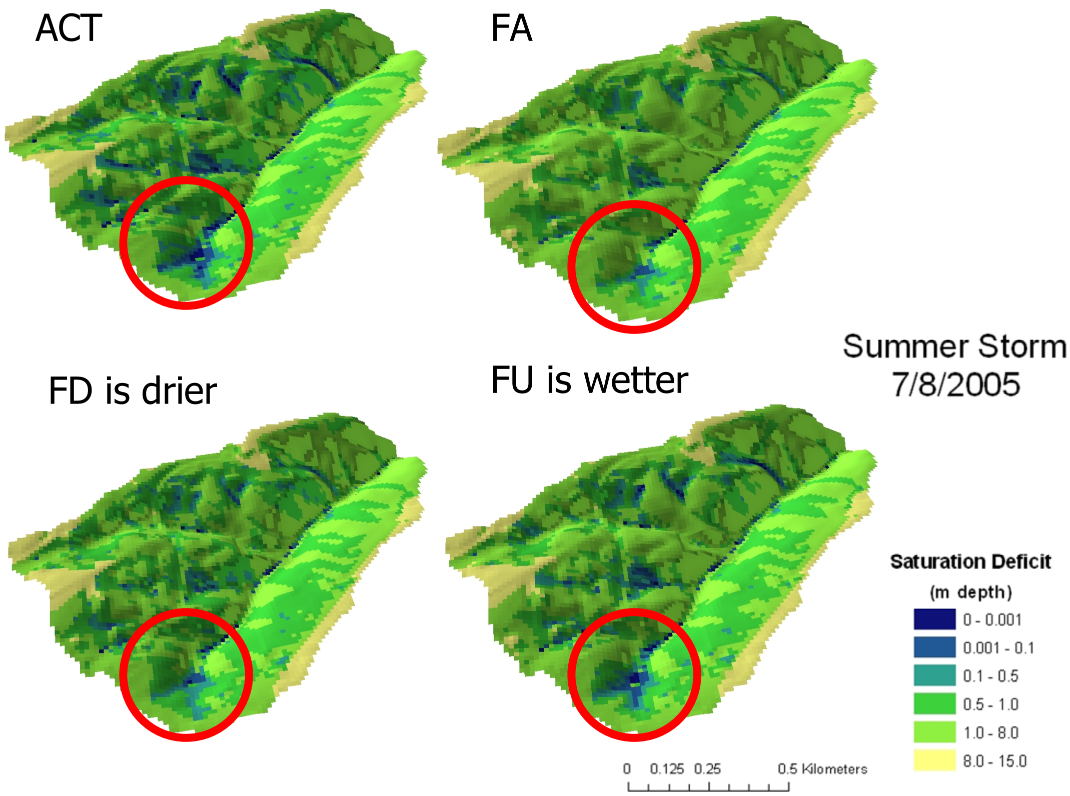


Figure 5. 19: Estimated spatial distribution of saturation deficit for all vegetation management scenarios following a precipitation event on July 8, 2005. The catchment is drier for the entirely forested and forested downslope scenarios and wetter for the actual and forested upslope scenarios. Differences in saturation extent are particularly evident within the red circled areas.

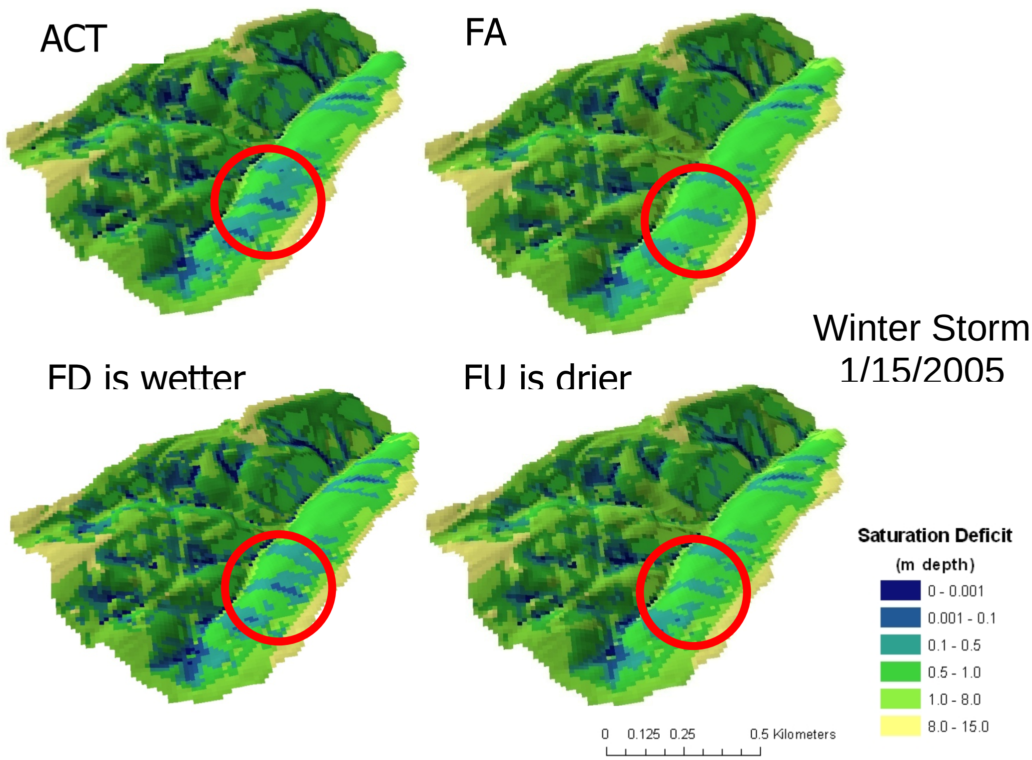


Figure 5. 20: Estimated spatial distribution of saturation deficit for all vegetation management scenarios following a precipitation event on January 15, 2005. During the dormant season, the catchment is drier for the entirely forested and forested upslope scenarios and wetter for the actual and forested downslope scenarios. Differences in saturation extent are particularly evident within the red circled areas.

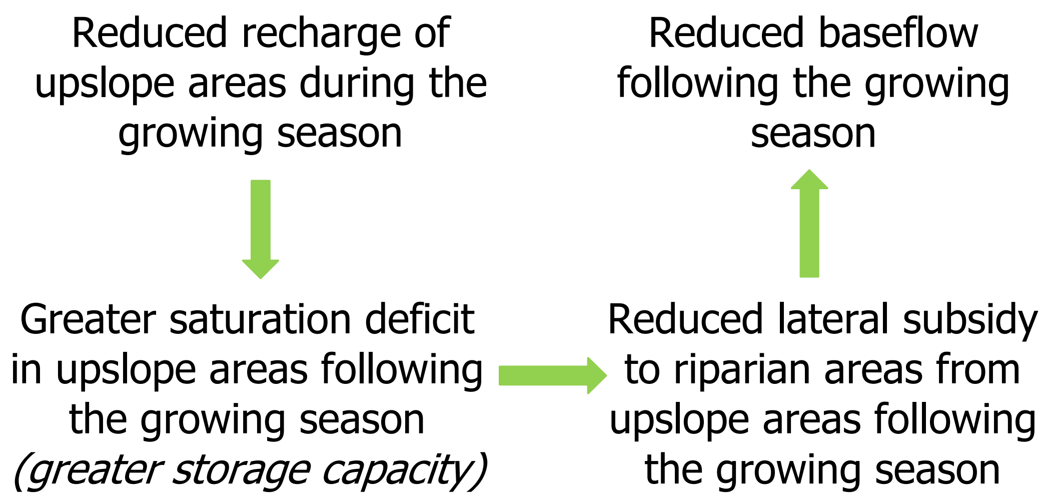


Figure 5. 21: Flow chart illustrating the mechanism through which the topographic position of reforested areas is believed to affect streamflow response.

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