

AN ABSTRACT OF THE DISSERTATION OF

Kellie B. Vache for the degree of Doctor of Philosophy in Bioresource Engineering presented on February 11, 2003. Title: Model Assessment of the Effects of Land Use Change on Hydrologic Response.

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Abstract approved: _____


John P. Bolte

The effect of landuse change on the hydrologic, biogeochemical and ecological response of watersheds is a concern throughout the world. To help characterize the potential magnitude of such changes, and of the potential to remediate or avoid undesirable features, studies focused on the cumulative watershed effects of site level change are necessary. The current state of the art model for water quality in agricultural lands, Soil Water Assessment Tool (SWAT), was used to estimate the effects of a set of future landscape scenarios on water quality in the Corn Belt region of the United States. These results indicated that changes to the current water quality management strategies will be necessary to significantly improve water quality in the Corn Belt region. In addition, the experience of implementing SWAT suggested a variety of changes to the model structure and study design with potential to improve the quality of the results. These changes include improved treatment of hydrologic process, full integration

of input data and model code, different methods of distributing data across space, the use of fewer parameters, more sophisticated numerical techniques, and improved methods for generating potential landscape scenarios.

A new model structure (WET_Hydro) was developed to address these issues. The hydrologic components of the model focus on a conceptual physically based characterization of the movement of water in soils, as overland flow, and in channels. Tests using a variety of input data sets, including both synthetic inflows and real watershed data were developed to verify the hydrologic components of the model. Additional model analyses evaluate how model scale interacts with parameters and with measurements. These analyses point toward additional criteria that may prove useful to the determination of correct model scales and to the utility of the flexible model structure which provides automatic changes to model scale. In addition to the scale analysis, a method of estimating the average new water contribution to storm discharge was developed. This additional model criterion was shown to provide further understanding of model utility under different hydrologic regimes.

The hydrologic model was extended to produce estimates of erosion and sediment export. Sensitivity to various restoration options were developed focusing on simple descriptions of remediation potential, and a minimum of parameters. In addition, the water quality model was coupled with a Decision Support System (DSS). Example applications demonstrate the potential of the combination to improve the process of restoration planning at the watershed scale.

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Model Assessment of the Effects of Land Use Change on Hydrologic Response

by
Kellie B. Vache

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Kellie B. Vache, Author

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Many student colleagues also provided significant support throughout my experience. Members of the Biosystem Analysis Group spent time reading and critiquing papers and their effort is greatly appreciated. David Rupp contributed significantly through conversations regarding hydrologic processes, and also was

instrumental in the development of portions of the data on which this work is based. I've spent considerable time with Matt Cox, and I would venture to guess that our understanding of models, code and how they ought to work has improved significantly, and in parallel. And lastly, I owe a debt of gratitude to those who truly made this work possible - my family, Mindy and John Crandall and Ziggy. Your support has been tremendous.

CONTRIBUTIONS OF AUTHORS

Mary Santelmann and Joe Eilers were instrumental in the development the paper comprising Chapter 2. Dr. Santelmann also contributed to a variety of the ideas making up Chapter 5. John Bolte provided extensive editing and suggestions for the content in Chapters 3, 4, and 5. He also participated in the initial development of many components making up the model outlined in these chapters. Jeff McDonnell and John Selker both provided input into the content of Chapter 4.

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Model Assessment of the Effects of Land Use Change on Hydrologic Response

Chapter 1.0: Introduction

1.1 Background

Spatially distributed models of hydrologic response provide landuse planners and managers the opportunity to study the effects of proposed landuse changes on water quantity and quality metrics. The development of the concepts underpinning these models began in the early 1960s with the development of the Stanford Watershed model, and has progressed considerably since that time. Despite this progress, there is still no generally accepted procedure for developing and applying hydrologic models. The most compelling reason for the lack of consensus is the wide variety of problems to which hydrologic models are applied. Issues commonly approached through the application of hydrologic models include agricultural non-point source pollution, flood control and water availability, climate change, landuse effects on peak discharges in forested regions, as well as procedures to test and verify conceptual descriptions of runoff pathways. Not only do state variables themselves vary, but the scales of interest also vary widely.

Reductionist theory suggests that detailed understanding and simulation of point scale processes can be combined with routing mechanisms to develop

catchment scale simulations of hydrologic response. These concepts form the basis for a suite of physically-based watershed models and are well represented in the literature. One of the reasons for their popularity appears to be the conceptual clarity of taking measurable, well-founded point-scale concepts, applying them at many points, and making statements about watershed-scale catchment behavior. Two general groups of models, both relying heavily on reductionist theory have evolved. The first group includes the fully distributed physics based models, represented by Mike SHE [Abbott et al., 1986].

The second group of models freely combine empirical models, such as the SCS curve number method (see Dunne and Leopold, [1978] for a useful description of this method) with physics based models and a variety of routing mechanisms. These models extend the empirical equations in time and space to produce continuous simulations of a wide variety of water quality and quantity metrics. The distributed, semi-empirical models have generally been developed specifically to evaluate landuse change at watershed scales, especially in agricultural areas. A variety of studies have utilized these models to produce useful statements regarding landuse change and the effects on a variety of hydrologic metrics.

Despite widespread acceptance, an alternative view has emerged which suggests that issues related to non-linear dynamics, uncertainty and parameterization significantly reduce the utility of the fully distributed physically/empirically based models [Beven, 1989]. From this perspective, the

response of a watershed is not simply the sum of the responses of a distributed network of points, and must be quantified in an alternative fashion. Most commonly the solutions rely on a heuristic description of watershed processes and are more conceptual than their physically-based counterparts.

These conceptual models have significant potential to predict anthropogenic landuse change effects on hydrologic and water quality responses across large areas. We have identified three characteristics of these models that contribute to this potential:

- The small number of parameters allow for improved characterization of parameter space, and therefore uncertainty of model results.
- The relative simplicity of the model structure allows for reasonably fast simulations.
- The spatial dimension provides an opportunity to evaluate the effects of landuse on response.

The conceptual modeling approach is well-represented in a rainfall-runoff and water chemistry context [Beven and Kirkby, 1979; Leavesley et al., 2002 ; Seibert and McDonnell, 2002; Uhlenbrook and Leibundgut, 2002] but application to watershed scale landuse change is unusual. One of the reasons may be the focus on parsimonious parameterization and uncertainty reduction. Introducing unquantifiable increases in uncertainty through the simulation of unmeasurable future scenarios might be seen as a step backwards. But the coupling of a

distributed box model with simple models of landuse change represents a useful alternative to simulations where even hydrologic uncertainty is unbounded.

An additional complication is the development of watershed-scale landuse/landcover restoration plans (or future scenarios). These hypothetical datasets are necessary input to a model designed to evaluate landuse effects on hydrology and water quality, but methods to develop them are not generally a component in the hydrologist's toolkit. In fact, the use of future alternative scenarios as tools to guide environmental decision-making is, in and of itself, a relatively new approach to watershed management [Vos and Opdam, 1993; Schoonenboom, 1995; Harms et al., 1996]. Mitsch et al., [2000] pursued the watershed-scale future scenario approach through lists of a set of potential improvements needed to address NPS pollution for agricultural regions at a watershed level. The work failed though to provide spatially-specific watershed plans to 1) illustrate how such restorations might look on a landscape and 2) allow for the potential evaluation of the watershed scale impacts of such plans. A variety of other applications to the development and use of future scenarios are documented in Dale and Hauber, [2001]. Of these, only Santelmann et al., [2001] applied a spatially-specific water quality model to a set of alternative futures in an effort to evaluate quantitatively the effect of changes in land use and management on water quality and quantity. A more detailed review of the modeling outlined in Santelmann et al., [2001] is presented in Vache et al., [2001] – Chapter 2 of this thesis. This type of application, focused the quantitative evaluation of spatially

explicit landuse change data, has potential to significantly improve our ability to manage watersheds at the watershed scale

1.2 Objectives

The overall goal of this study is the application of the current generation of conceptual hydrologic modeling ideas to the evaluation of spatially-distributed landuse change. Along the way, the work attempts to make useful contributions of interest to GIS scientists and hydrologists. Specifically, the objectives of the thesis are:

- Document the current state of the art in landuse change modeling through application of standard modeling package.
- Design and implement a model of hydrology and landuse change that works in conjunction with an existing decision support system. The model focuses on watershed or meso-scale analysis (1 to 1000 square kilometers)
- Develop efficient code to evaluate parameter space. An evaluation of parameter space provides information on the appropriate parameter choices and additionally, on parameter uncertainty.

The thesis is organized into five chapters, each of which describes a well-defined portion of the research results. Chapter 1 provides a general introduction. The bulk of the thesis begins in chapter 2 with an application paper focused on the

simulation of the effects of landuse change on water quality in two agricultural watersheds. The calculations make use of the SWAT model, and the paper is representative of the current types of analysis available to landuse managers. The paper was recently published as an article in the Journal of the American Water Resources Association [Vache et al., 2002].

The third chapter provides a description and analysis of the stream routing model implemented as part of the thesis. The focus of this chapter is on the development of data structures and algorithms designed to facilitate distributed modeling of hydrology. The paper was developed for submission to Computers and Geosciences and presents a novel approach to network development and analysis.

Chapter four provides a detailed analysis of the hydrologic model, using results from five separate watersheds. The watersheds presented in the study were chosen to represent a broad range of hydrologic regimes, basin sizes and available data. Specifically, the objectives of this chapter are to:

- Demonstrate the capability of the model to simulate the rainfall runoff relationship across a broad range of hydrologic regimes. We elected to use a wide variety of input datasets in an effort to fully characterize model components. Evaluating the model at a variety of sites does represent a tradeoff. On one hand it provides a useful indication of how well the model functions across a spectrum of hydrologic processes. At the same time this choice limits the amount of detailed analysis at each of the sites.

Somewhat less detail across a wider range of sites is compatible with the goal of developing and applying a watershed scale model.

- Present results from a variable time step solution procedure and demonstrate how it allows for the quantification and management of numerical dispersion.
- Outline the unique vector-based description of space and how this is used to quickly vary the scale over which calculations occur. Unlike most other distributed models, the simulations are not dependent on an *a priori* definition of grid cell size. An analysis of the effect of model scale on parameter identification is a corollary to this objective.
- Introduce methods to estimate a new/old water ratio and the temporal overland flow frequency, as additional watershed scale criteria for model analysis.

In chapter 5, the hydrologic model is extended to produce estimates of change in the discharge of sediments as a result of watershed scale landuse plans. This chapter provides an extension of the work in Chapter 2, where the focus is on a set of static endpoint landscapes and the SWAT model. This chapter differs in two major categories from Chapter 2. First, the work introduces a Decision Support System (DSS) called RESTORE and outlines how this system provides a more dynamic analysis of the potential for future landuse and landcover changes. Second, we continue to focus the conceptual distributed modeling approach described in Chapters 3 and 4. We present a set of models that incorporate

directly (and with a minimum of parameters) the modeler's conceptualization of the effects of site-scale landuse change on erosion and sedimentation. The hydrology model relies on a distributed set of these site-scale models to provide an understanding of the cumulative effects of watershed-scale landuse changes. The maintenance of the parsimonious modeling approach leaves open the opportunity to evaluate the effects of parameter uncertainty on model predictions.

2.0 Water Quality Modeling Of Alternative Agricultural Scenarios In The
U.S. Corn Belt

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2.1 Abstract

Simulated water quality resulting from three alternative future land-use scenarios for two agricultural watersheds in central Iowa was compared to water quality under current and historic land use/land cover to explore both the potential water quality impact of perpetuating current trends and potential benefits of major changes in agricultural practices in the U.S. Corn Belt. The Soil Water Assessment Tool (SWAT) was applied to evaluate the effect of management practices on surface water discharge and annual loads of sediment and nitrate in these watersheds. The agricultural practices comprising Scenario 1, which assumes perpetuation of current trends (conversion to conservation tillage, increase in farm size and land in production, use of currently-employed Best Management Practices (BMPs)) result in simulated increased export of nitrate and decreased export of sediment relative to the present. However, simulations indicate that the substantial changes in agricultural practices envisioned in Scenarios 2 and 3 (conversion to conservation tillage, strip intercropping, rotational grazing, conservation set-asides and greatly extended use of BMPs such as riparian buffers, engineered wetlands, grassed waterways, filter strips and field borders) could potentially reduce current loadings of sediment by 37-67% and nutrients by 54-75%. Results from the study indicate that major improvements in water quality in these agricultural watersheds could be achieved if such

environmentally-targeted agricultural practices were employed. Traditional approaches to water quality improvement through application of traditional BMPs will result in little or no change in nutrient export and minor decreases in sediment export from Corn Belt watersheds.

2.2 Introduction

Despite advances in our knowledge and understanding of the sources of agriculturally-derived nonpoint source pollution (NPS), and the development of best management practices intended to reduce NPS pollution, aquatic ecosystems linked to agricultural regions in the United States continue to receive high loadings of agricultural pollutants [Runge, 1996; Goolsby and Battaglin, 1997; Becker et al., 2000; Schilling and Thompson, 2001]. In addition, owing to hydrologic modifications of agricultural landscapes (loss of wetlands, tile drainage, channelization of streams and channel incision), streamflow increases following precipitation and snowmelt in agricultural watersheds are rapid and annual discharge elevated relative to historic conditions. In 1995, the U.S. Corn Belt was identified by the Office of Technology Assessment as the number one priority region for water quality concern in the U.S. [OTA, 1995].

Here we report results of research on the potential influence of changes in land use and management on water quality based on comparisons of simulations for three alternative future scenarios and current land use. The future scenarios

contrasted landscapes that could evolve over the next 25 years from perpetuation of current trends with two alternatives. In Scenario 1, the projection of current trends, priority is given to maximizing agricultural production and profit on a 1 to 5 year time scale. In Scenario 2, both public support and agricultural policy place highest priority on achieving substantial improvement in water quality; and in Scenario 3, highest priority is given to restoration of native biodiversity as well as improvement in water quality.

Current water quality conditions in Walnut Creek have been characterized through analysis of water quality data collected monthly since 1990 for Walnut Creek, Story County watershed [Cambardella et al., 1999; Hatfield et al., 1999; Jaynes et al., 1999]. Water quality conditions in Buck Creek watershed were characterized as part of the current project. The Soil and Water Assessment Tool (SWAT) [Arnold et al., 1995] was calibrated to current conditions in each watershed. This model was then applied to evaluate water quality for the three alternative future scenarios. Additionally, to provide another benchmark for comparison of simulated water quality in these watersheds, the model was applied to historic land cover to estimate historical discharge and surface water chemistry. The purpose of the comparison with historic landscapes is not to promote a return to historical conditions in these watersheds, but rather to provide an estimate of the upper bound for potential water quality improvements attainable in the region. The objective of this paper is to evaluate the effects of agriculture on water quality at the watershed scale and provide quantitative estimates of how landscape

and management changes might affect water quality. We present two steps towards accomplishing the objective. The first is an analysis of the water quality conditions in Buck Creek watershed, as compared to conditions characterized by Cambardella et al., [1999] and Jaynes et al., [1999] in Walnut Creek watershed. The second is an evaluation of alternative future scenarios with respect to modeled water quality endpoints and a discussion of the potential of land use and management practices to improve water quality. Our research adds to a body of literature [Wolf, 1995; Becker et al., 2000; Mitsch et al., 2001; Schilling and Thompson, 2001] indicating that significant water quality improvement, on the order of 50% loading reductions, will require land managers to develop fully integrated watershed plans including innovative land use and management practices. Because such practices will entail a higher degree of economic uncertainty, agricultural policy and public support for such practices must be a part of these plans, as assumed in our scenarios.

2.3 Study Area

Walnut Creek (Story County) and Buck Creek (Poweshiek County) are located in central Iowa (Figure 2.1). Both watersheds were selected as representative of their respective physiographic regions in central Iowa as part of the Midwest Agrichemical Surface-subsurface Transport and Effects Research (MASTER) program [Freemark and Smith, 1995]. Landcover data from 1994

summarizing current conditions were available in a GIS database as a product of that research [Freemark and Smith, 1995].

Like many watersheds in the Des Moines Lobe physiographic region, Walnut Creek watershed has relatively little topographic relief and poorly drained soils [Prior]. Much of Walnut Creek watershed has been extensively tile-drained. Land cover in 1994 consisted primarily of row crops, with 83% of the watershed area in corn/soybeans. Few livestock operations occur in the watershed, with pasture, hay, and grassland comprising approximately 5% of the land cover. The Walnut Creek watershed has an area of 51.3 km² with an elevation range from 267 m to 320 m, and has a stream channel density 50% less than Buck Creek watershed. In contrast, Buck Creek is located in the Southern Iowa Drift Plain physiographic region. The watershed has a rolling topography, moderate topographic relief, and varied agricultural land cover. The area of Buck Creek watershed is 88.2 km² with elevations ranging from 236 m to 305 m. Land cover in Buck Creek in 1994 was 45% corn/soybean rotation, 15% Conservation Reserve Program (CRP), 14% pasture and 26% in other land cover types. The watershed has a highly dendritic stream network with a total channel length of 115 km. Land cover and other watershed attributes for Walnut and Buck Creeks are summarized for current conditions (1994) in Figure 2.2, and for the three scenarios in Tables 2.1a and 2.1b.

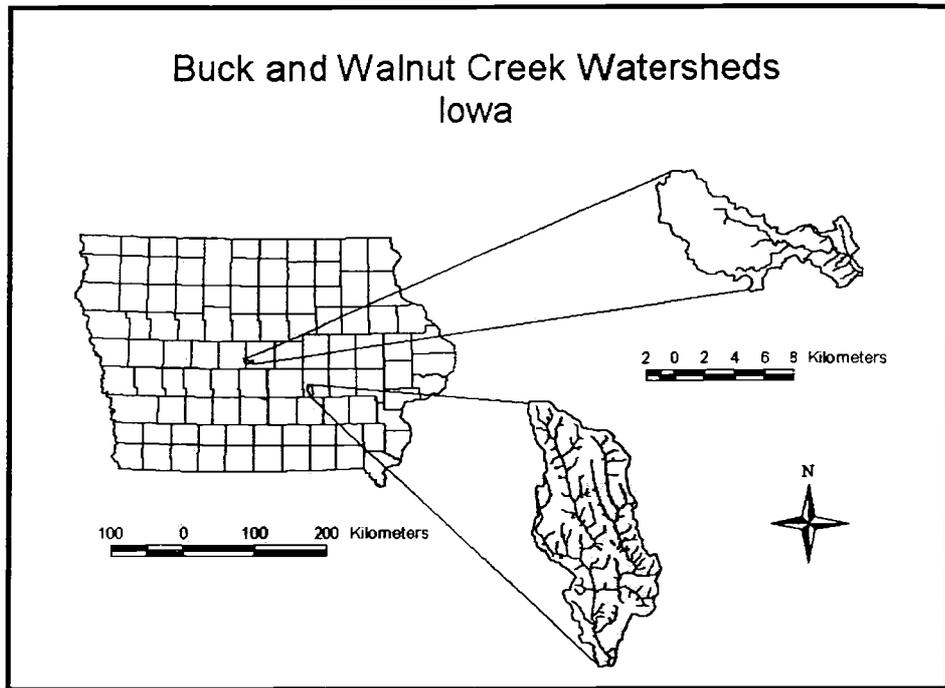


Figure 2.1. Buck and Walnut Creek Watershed site map

Table 2.1a. Summary of landuse in the Walnut Creek Watershed under the current conditions and also the three future scenarios.

| | Present Area(ha) | Scenario 1 | | Scenario 2 | | Scenario 3 | |
|-----------------|------------------|------------|----------------|------------|----------------|------------|----------------|
| | | Area (ha) | Percent Change | Area (ha) | Percent Change | Area (ha) | Percent Change |
| Woodland Closed | 66.4 | 61.9 | (-6.9) | 21.4 | (-67.7) | 126.4 | (90.2) |
| Woodland Open | 114.2 | 69.4 | (-39.2) | 82.8 | (-27.5) | 109.1 | (-4.5) |
| Savannah | 41.3 | 24.3 | (-41.1) | 39.8 | (-3.5) | 34.4 | (-16.6) |
| Corn/Soybeans | 4190.5 | 4510.4 | (7.6) | 2882.3 | (-31.2) | 1725.1 | (-58.8) |
| Crp | 0 | 128.2 | (100.0) | 302.1 | (100.0) | 0.0 | (-100.0) |
| Alfalfa | 89.7 | 0.0 | (-100.0) | 400.9 | (346.5) | 0.0 | (-100.0) |
| Pasture | 118.6 | 0.0 | (-100.0) | 826.7 | (596.9) | 0.9 | (-100.0) |
| Pond/Lake | 3.7 | 3.6 | (-4.3) | 6.1 | (61.9) | 3.7 | (-0.3) |
| Fencerow | 67.4 | 0.2 | (-99.7) | 58.5 | (-13.2) | 0.0 | (-99.9) |
| Riparian Areas | 25.1 | 46.7 | (85.8) | 72.9 | (189.6) | 263.2 | (945.1) |
| Intercropping | 0.0 | 0.0 | (0.0) | 0.0 | (0.0) | 1931.7 | (100.0) |
| Biodiversity G. | 0.0 | 0.0 | (0.0) | 84.1 | (0.0) | 91.5 | (100.0) |
| Organic Crops | 0.0 | 0.0 | (0.0) | 0.0 | (0.0) | 153.8 | (100.0) |
| Set aside | 0.0 | 0.0 | (0.0) | 0.0 | (0.0) | 91.4 | (100.0) |
| Wetland | 0.0 | 0.0 | (0.0) | 0.0 | (0.0) | 197.9 | (100.0) |
| Other | 343.1 | 285.3 | (-16.8) | 352.4 | (2.6) | 400.9 | (14.4) |

Table 2.1b. Summary of landuse in the Buck Creek Watershed under the current conditions and also the three future scenarios

| | Present Area (ha) | <i>Scenario 1</i> | | <i>Scenario 2</i> | | <i>Scenario 3</i> | |
|--------------------|----------------------|-------------------|-------------------|-------------------|-------------------|-------------------|-------------------|
| | | Area (ha) | Percent Change | Area (ha) | Percent Change | Area (ha) | Percent Change |
| Woodland Closed | 406.1 | 210.7 | -48.1 | 60.6 | -85.1 | 582.4 | 43.4 |
| Woodland Open | 240.4 | 98.0 | -59.2 | 171.4 | -28.7 | 235.2 | -2.2 |
| Savannah | 126.3 | 59.4 | -52.9 | 123.1 | -2.5 | 155.6 | 23.2 |
| Corn/Soybeans | 3823.3 | 5277.1 | 38.0 | 1063.1 | -72.2 | 88.2 | -97.7 |
| Crp | 1392.2 | 1919.2 | 37.8 | 371.6 | -73.3 | 0.0 | -100.0 |
| Alfalfa | 353.8 | 590.2 | 66.8 | 3640.9 | 928.8 | 0.0 | -100.0 |
| Pasture | 1217.9 | 0.0 | -100.0 | 1689.4 | 38.7 | 0.0 | -100.0 |
| Pond/Lake | 33.5 | 31.7 | -5.4 | 38.8 | 15.8 | 41.6 | 24.1 |
| Fencerow | 123.4 | 0.0 | -100.0 | 366.4 | 196.8 | 0.0 | -100.0 |
| Riparian Area | 51.6 | 95.1 | 84.4 | 384.4 | 644.8 | 1113.8 | 2057.6 |
| Intercropping | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 5432.1 | 100.0 |
| Biodiversity G | 0.0 | 0.0 | 0.0 | 80.0 | 100.0 | 117.2 | 100.0 |
| Organic Crops | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 28.0 | 100.0 |
| Setaside | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 233.4 | 100.0 |
| Wetland | 0.0 | 0 | 0.0 | 0.0 | 0.0 | 0.0 | 100.0 |
| Other | 1051 | 538.6 | -48.8 | 830.3 | -21.0 | 792.5 | -24.5 |

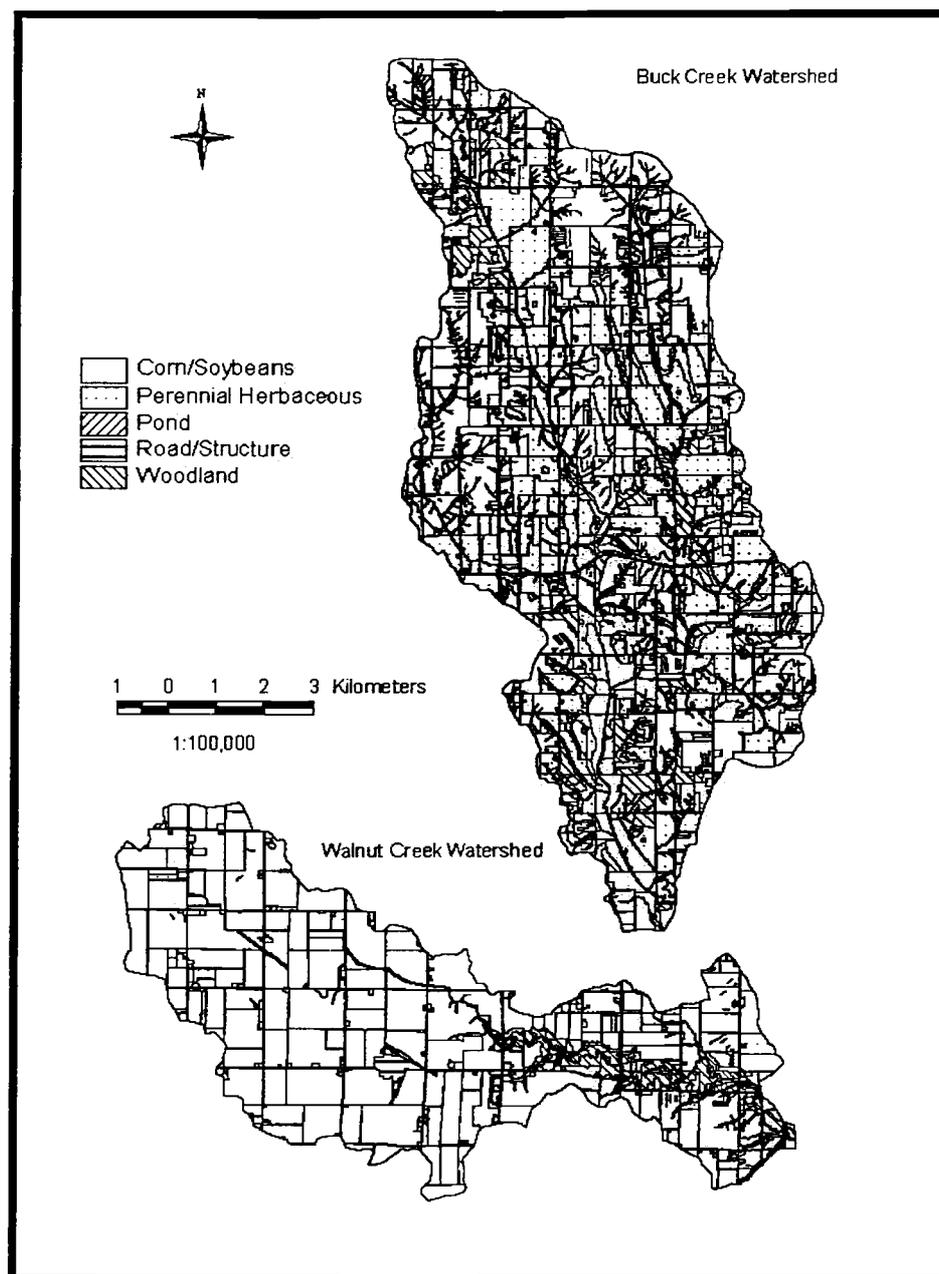


Figure 2.2. Current (1994) land use in the Buck and Walnut Creek watersheds.

2.4 Methods

2.4.1 Water Quality Data

Walnut Creek is one of the study sites in the United States Department of Agriculture (USDA) Management Systems Evaluation Area (MSEA) project. As part of MSEA, five sites within the watershed have been monitored for water quality monthly since 1990. Additionally, a network of stream gauges and tipping buckets have been operating in the basin since 1991. Data have been collected on stream nitrate concentrations since 1990 [Hatfield et al., 1999]. Nitrate concentrations during the study period often exceeded the 10 mg/l federal limitation, and watershed losses ranged from 4 to 66 kg/ha. These losses represented 4 to 115% of the nitrogen fertilizer applied each year. [Jaynes et al., 1999]. These data were used in this study assist in the calibration of the SWAT model in Walnut Creek watershed. However, because the MSEA project focused on nitrogen and pesticide dynamics in agricultural systems, there are few data available on sediment in Walnut Creek. Two samples were collected in 1999 to assist in the determination of sediment loading in Walnut Creek.

Data collection in the Buck Creek watershed began in March of 1998 and continued through June of 1999. A single surface-water monitoring site was selected near the base of the watershed. Samples were collected from late winter to early spring in 1998 and 1999, and an effort was made to collect multiple

samples during periods of high discharge. Grab samples were collected at an approximate monthly interval, while an ISCO™ 6700 sampler was used to collect close interval samples representing periods of rapidly fluctuating discharge. The sampler was tripped manually and collected at four-hour increments. Samples from six precipitation and snowmelt events were collected during the study period.

All samples were transported to either the Iowa State Health Lab or the Central Analytic Laboratory at Oregon State University in iced coolers. At a minimum, each sample was analyzed for total suspended solids (TSS), pH, specific conductance, nitrate+nitrite nitrogen as N, and total phosphate as P. A recording pressure transducer (Solinst 3400) was installed at the site in July of 1998, and a rating curve developed to generate hourly discharge data. Additionally, rainfall data were collected with a standard tipping bucket gage and an Onset data logger. The daily Buck Creek hydrograph was extended to a synthetic eight-year dataset, using a transform function based on a linear regression with data collected by USGS near Hartwick, IA in Walnut Creek (USGS Station Number 05452200) [Schoup, 1999].

Nitrate-nitrogen and total suspended solids data are summarized in Table 2.2. For the 116 samples, nitrate concentrations in Buck Creek never exceeded the 10 mg/l federal maximum even in samples collected during the spring and summer of 1998 and 1999, when nitrate concentrations tend to be greatest. In Walnut Creek, nitrate is a more significant issue, with concentrations exceeding

10 mg/l about 30% of the time between 1992 and 1995. The number increases to 45% exceedence when looking only at spring and early summer months (Table 2.2).

In studies of Midwestern water quality, TSS is often not considered, a fact commonly attributed to the lack of salmonid bearing streams [Waters 1995]. But links between agricultural activities and TSS are well established [Menzel et al., 1984 ; Freemark and Smith, 1995], as are detrimental effects on warm water fishes [Mathews, 1984; Lyons and Courtney, 1990]. In addition, increases in stream sediments represent increases in soil loss. 116 samples were collected for TSS analysis from Buck Creek. These data are summarized in Table 2.2. Collection was targeted towards storm events, and these values therefore represent likely maximums. The median value was approximately 1.0 g/l, with a maximum of 26.0 g/l. These values are highly elevated, suggesting that erosion is a much more significant problem in the Buck Creek watershed than surface water nitrate runoff. The evaluation is limited because of the relatively small sample size. This suggests the need for additional watershed-scale studies, similar in scope to the MSEA work in Walnut Creek, that include the evaluation of erosion and TSS.

The relationship between nitrate and TSS during events appears consistent with other studies. Nitrate concentrations decrease during high flows due to dilution and TSS concentrations increase due to the increased erosive potential of the higher discharges (Figure 2.3).

Table 2.2. TSS and NO³-N summary for Buck Creek and Walnut Creek outlet stations. Walnut Creek data provided by J. Hatfield. A. All MSEA data collected between 1992 and 1995. B. MSEA data taken during March, April, May, June, July, between 1992 and 1995.

| | <i>Mean</i> | <i>Std.</i> | <i>Median</i> | <i>Maximum</i> | <i>n</i> | <i>% > 10</i> <i>mg/l</i> |
|---------------------------------|-------------|-------------|---------------|----------------|----------|---------------------------------|
| Buck Creek | | | | | | |
| NO ₃ -N (mg/l) | 6.02 | 1.85 | 6.45 | 9.6 | 116 | 0 |
| TSS (mg/l) | 2208 | 3939 | 1012 | 27200 | 116 | Na |
| Walnut Creek | | | | | | |
| A. NO ₃ -N (mg/l) | 8.0 | 3.6 | 8.3 | 20.9 | 966 | 33 |
| B. NO ₃ -N (mg/l) | 9.0 | 3.4 | 9.3 | 20.9 | 628 | 45 |

2.4.2 The SWAT Model

The SWAT (Soil and Water Assessment Tool) model is a continuous, spatially explicit simulation model. The model is designed to quantify the effects of land use and management change on water quality in agricultural basins [Arnold et al., 1995].

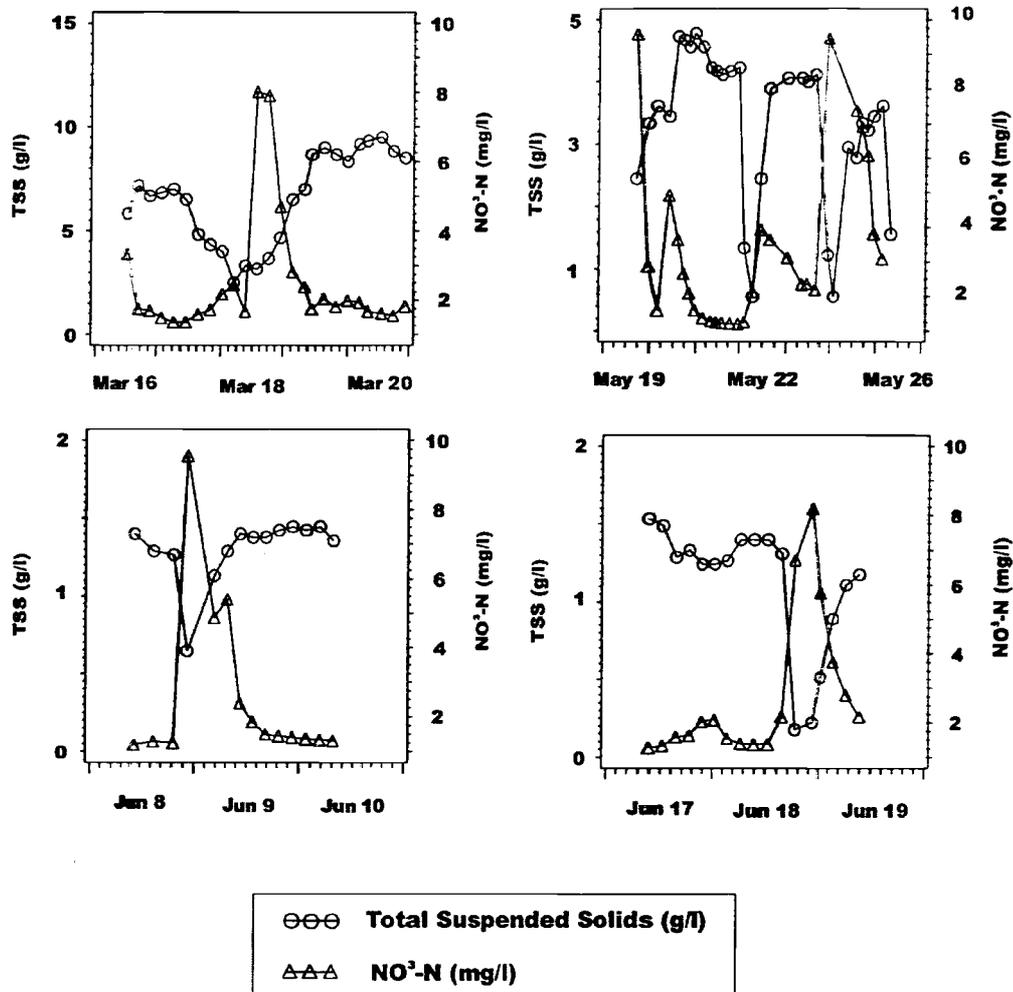


Figure 2.3. Total Suspended Sediment and Nitrate-Nitrogen for four storm events in the Buck Creek watershed.

It was developed by the USDA Agricultural Research Service (USDA- ARS) and is based on the ROTO, CREAMS, and SWRRB models. SWAT represents an improvement upon its predecessors through incorporation of increased spatial detail and routing of water and sediment [Binger, 1996]. A version of the model incorporating a Geographic Information System (GIS) was utilized. The GIS

interface simplifies the process of watershed discretization and parameter assignment through the use of spatially-explicit data sets representing elevation surface, soils, land use, and management [Di Luizio et al., 2000]. SWAT simulates hydrology using a mass balance with terms representing surface runoff, percolation, lateral surface flow, and evapotranspiration. Surface runoff is estimated with the SCS formula [Di Luizio et al., 2000]. The percolation component is treated using a storage model in combination with a crack flow model to simulate rapid macropore flow through desiccated soils [Arnold et al., 1995]. Lateral surface flow is generated using a kinematic storage model, and evapotranspiration is modeled after methods described by Hargreaves and Samani, [1985]. The model uses a modified version of the Universal Soil Loss Equation (MUSLE), developed by Williams and Berndt, [1977] to estimate sediment yield. The nitrogen subroutine includes estimates of surface water nitrate loss, nitrate leaching, denitrification, mineralization, immobilization, organic nitrogen movement with soils and crop uptake.

Recent developments in computer hardware, software, and remote sensing techniques have greatly accelerated the use of data-rich distributed models, including SWAT. The SWAT model provides users with the choice of a detailed grid based approach, including cell to cell routing, or a less detailed hydrologic response unit (HRU) configuration. Manguerra and Engel, [1998] concluded that for most cases, the simpler HRU configuration provided essentially the same results as a more complicated grid based scheme. Given this, and the difficulty in

obtaining grid-cell specific data related to current and future management practices, we employed the HRU scheme.

2.4.3 Use of Future Scenarios

Evaluations of the local effects of specific restoration and management changes on surface water quality are common in the literature [Daniels and Gilliam, 1996]. Many of these studies include plot or field-scale experiments employing detailed sampling schemes, intensive chemical analyses, and field-scale manipulation of land cover and management practices. The current study is not designed to simulate effects of field-scale change. Rather, the intent is to quantify the cumulative effects of changes in watershed land use and management practices, using a modeling framework. The basin wide changes are expressed as spatially-explicit GIS datasets that embody the agricultural and conservation practices of the alternative future scenarios developed by Nassauer et al., [2001].

Alternative future scenarios have been used for years in Europe to assist in land-use planning and the evaluation of the potential environmental consequences of different choices for landowners and policymakers [Schoonenboom, 1995; Harms et al., 1996]. The use of future scenarios, whose land use and land cover can be represented in GIS databases, coupled with GIS-based modeling approaches, allows the exploration of the outcomes of different management alternatives on real landscapes, using data on soils, topography, and biota of those

landscapes. Alternative future scenarios coupled with GIS-based evaluative approaches can be powerful tools for guiding policy and land-use decision making [Ahern, 1999; Santelmann et al., 2001]. In the U.S., only a handful of such studies have been done [Steinitz et al., 1994; Hulse et al., 2000] and this project is the first to focus on specific changes in agricultural practices, and to use a spatially distributed model to evaluate water quality responses.

2.4.4 Future Scenarios for Iowa watersheds

Scenario 1 represents a projection of current trends in Midwestern agriculture to the year 2025 applied to the study watersheds. Under this scenario, management decisions are based largely on maximizing production of agricultural commodities, although in cases where no decrease in production is projected, management practices that enhance water quality are employed. The most significant of these management practices with projected water quality benefits employed in Scenario 1 are precision agriculture and no-till cultivation. The increased area in production envisioned in Scenario 1 results in an increase in the mass of fertilizer applied to each basin.

Scenario 2 includes a variety of management techniques designed specifically to address water quality concerns. The most noteworthy of these design elements was the implementation of riparian buffers to 30 m on both sides of perennial streams and 15 m on both sides of ephemeral streams. Other

management strategies include no-till cultivation, strip cropping where appropriate, and alfalfa production on all fields adjacent to streams.

Development of Scenario 3 proceeded under the assumption of broad-based public support for restoration of native biodiversity and improvement in water quality, with an emphasis on maximizing terrestrial biodiversity. Monocultures of corn and soybean rotations are significantly reduced under this scenario, replaced over large areas by strip intercropping, including strip intercropping that incorporates a strip of native perennials in fields of corn and soybeans. The widths of riparian buffers in Scenario 3 are, in all instances, doubled compared to Scenario 2. Additionally, production of organic crops is increased and large areas are set-aside as high quality habitat reserves [Nassauer et al., submitted].

Coiner et al., [2001] provide an economic analysis of the current and three future scenarios for the Walnut Creek watershed. Among other results, they indicate that in Walnut Creek, Scenario 3, with a 58% decrease in the area under production of corn and soybeans, results in approximately the same economic return as current practices. The suggestion is that carefully chosen management decisions can both significantly reduce NPS pollution and maintain the economic viability of the Corn Belt region.

The land cover changes among the scenarios are summarized in Tables 2.1a and 2.1b for Walnut Creek and Buck Creek, respectively. Assumptions

regarding the agricultural practices (amount and timing of fertilizer, tillage etc.) used in each scenario are listed in Appendix 2.1.

Historic land cover was generated using a detailed (1:12000) regional soils database. Classifications are based on characteristics of soils formed under prairie and forest vegetation. Areas defined as prairie were sub-divided into upland prairie, ephemeral wetland, seasonal wetland, semi-permanent wetland, permanent wetland, and pond according to the relationships between soil types and wetland types described in Galatowitsch and van der Valk, [1994] [Rustigian, 1999].

2.4.5 Model Calibration

SWAT is often described as a physically based model designed for use in ungauged basins [Arnold et al., 1995; Rosenthal et al., 1995; Binger, 1996]. While the claim has merit, calibration was considered an important step in this study. The studies cited above concentrated on the hydrology components of SWAT, whereas our project required estimation of sediment and nutrient export as well as hydrologic change. The nutrient components of SWAT are highly parameterized and there is no reason to assume that default values correctly account for watershed-specific nutrient dynamics. Although a complete long-term dataset including discharge, TSS and NO₃-N was unavailable, we attempted to make reasonable comparisons to all available data.

Figure 4 represents calibration results for Walnut and Buck Creek. The stream discharge results match measured values reasonably well. The coefficient of determination (r^2) based on the monthly discharge data for the Buck Creek simulation is 0.64. For the Walnut Creek simulations the r^2 for the discharge results is 0.67. The Buck Creek sediment plot was developed based on an average TSS stream concentration of 150 mg/l to represent the entire seven year time period. This average was developed from low flow samples collected in 1997 and 1998. Though a more complete TSS dataset might allow for significant improvement in these results, we present the figure as qualitative evidence that SWAT can effectively reproduce instream sediment concentrations for watersheds in Iowa. Very little TSS data was available in the Walnut Creek watershed and we used the Buck Creek calibrations in the Walnut Creek simulations.

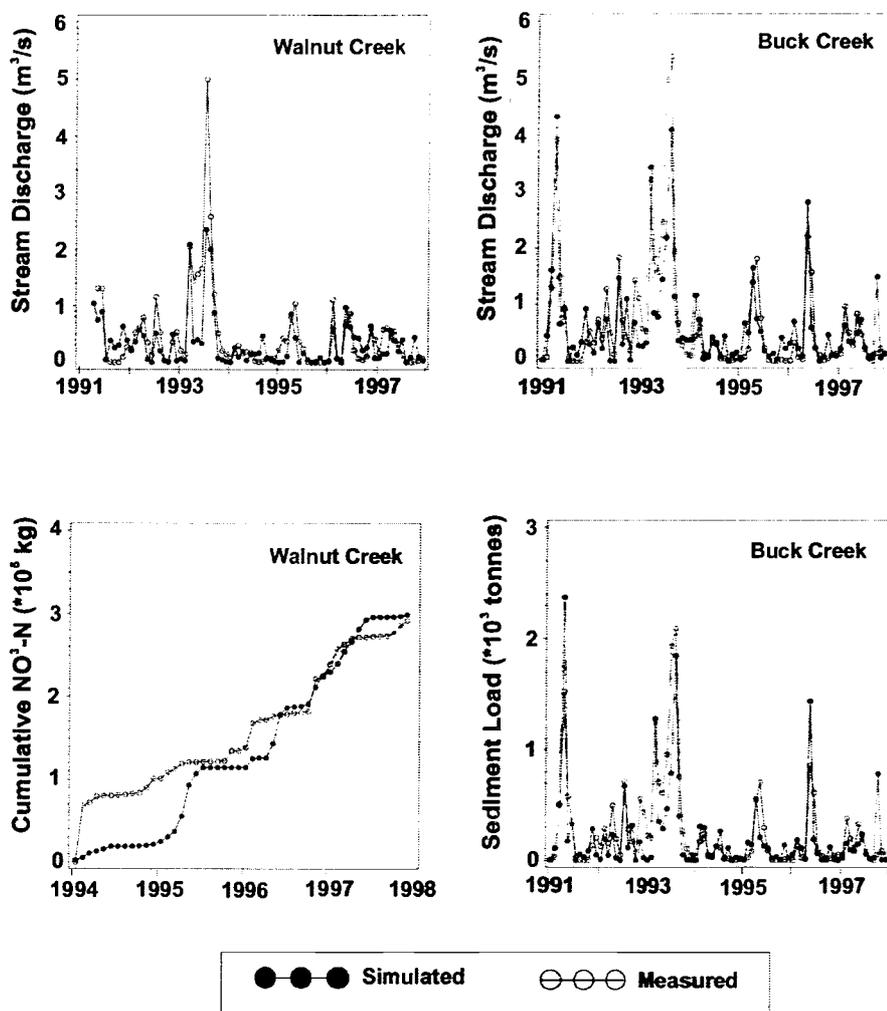


Figure 2.4. Calibration results for Walnut and Buck Creek.

The cumulative distribution of $\text{NO}_3\text{-N}$ for the period when nitrate measurements were available in Walnut Creek is also included in Figure 3. The model accounts for most of the nitrate in Walnut Creek for the period, though

discrepancies in the timing and magnitude are apparent. One of the difficulties encountered in these relatively large watersheds was a lack of field-specific data on the magnitude and timing of fertilizer applications. Model simulations represent best estimates of current nutrient management practice, on average, for the region [Hatfield et al., 1999] but no attempt was made to verify nutrient application regimes on a field-scale basis. Improved estimates of nitrate inputs would likely reduce model error, but given the complexity of nitrogen dynamics and the successful long-term mass balance of NO₃-N demonstrated by Figure 4, we elected to accept this calibration of the SWAT model as representative of general watershed practices.

2.5 Results and Discussion

2.5.1 Simulation Results

The scenarios evaluated in this project were designed to improve water quality, albeit with different emphases on human priorities and watershed practices. The model was used to quantify the improvement that might be expected with the implementation of each scenario. Each scenario for both Walnut and Buck Creeks was simulated on a daily basis for an eight-year period from 1992 through 1998. Results from these long-term simulations are, throughout the text, presented as percentage change of median yearly loading, for

the simulated six-year period. The median values were used in an effort to provide an indication of how water quality responded to changes in land use and not to extreme events.

2.5.2 Evaluation of Future Scenarios

In all cases, the scenarios were forecast to decrease upland erosion and TSS concentrations in Walnut and Buck Creeks relative to current conditions (see Figures 5 and 6). The decreases forecast for Scenarios 2 and 3 were substantially lower than those for Scenario 1.

The major difference between the present landscape and Scenario 1 is the basin-wide implementation of no-till cultivation. There is an approximately 15% decrease in the median TSS loading in each of the two streams, while the area in corn and soybean production actually increases from 80% of the total watershed area to 86% of the watershed area in Walnut Creek and from 43% to 62% in the Buck Creek watershed. These results suggest that moderate reductions in soil loss could be achieved in this setting though the widespread implementation of no-till farming.

The simulated decreases in erosion from Scenarios 2 and 3 were significantly greater than from Scenario 1, ranging from 35% to 60% reductions in the median sediment yield. We attribute this improvement to a more complete set of management practices combined with decreased production of corn and

soybeans. Surprisingly, TSS loadings under Scenario 2 are forecasted to exhibit the greatest decline from current values, whereas an examination of the scenarios might suggest Scenario 3 would show the most significant decreases. (Scenario 3 buffers are twice the width, strip intercropping Often replaces corn and soybean rotation, and larger areas are set-aside as habitat reserves).

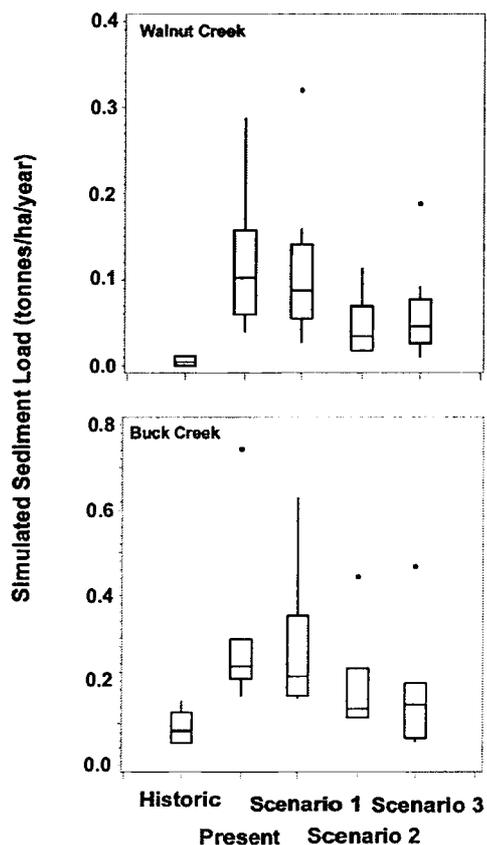


Figure 2.5. Boxplot representing sediment loading in Walnut and Buck Creek watersheds.

This result suggests that increasing the width of riparian buffers from 30 to 60 meters along perennial streams is unlikely to provide any significant decrease

of in-stream sediment concentrations, beyond what is expected under the narrower buffer strips. Set-asides and alternative crops are not simulated to decrease erosion more than maintenance of perennial cover on erodible land, either in the form of alfalfa/hayfields or as carefully managed rotational pasture.

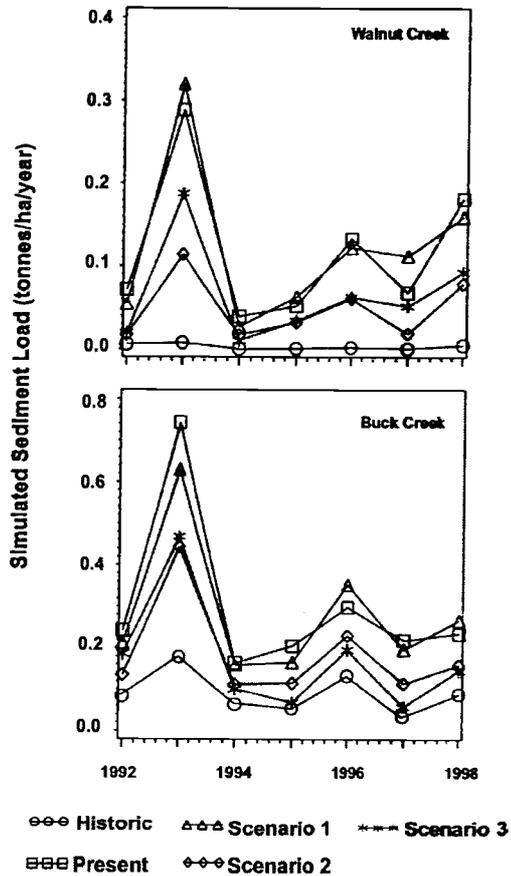


Figure 2.6. Time series plots of yearly sediment loading in Walnut and Buck Creek watersheds for the study period. The time series help explain the variation noted in figure 6, particularly the high values which occurred in 1993, a year of widespread flooding throughout the Midwest.

The pattern of erosion reduction is similar between Buck Creek and Walnut Creek, but the magnitude of the decrease is simulated to be larger in Walnut Creek. Buck Creek is a well-developed stream system with much greater relief than Walnut Creek. Overland flow moves over steeper terrain in Buck Creek, which increases its potential to erode. Additionally, upland flow paths tend to be shorter in Buck Creek due to the dendritic nature of the channel. These shorter flow-paths result in higher TSS values, as sediment is less likely to be re-deposited before reaching the stream.

The pattern of results for the nitrate simulations is somewhat different than that for the TSS simulations (Figures 2.7 and 2.8). One of the most notable differences occurs in Scenario 1, where nitrate concentrations are forecast to increase in both Walnut and Buck Creeks. In each watershed, the area in monoculture production of corn and soybeans increases but the management assumption is that current application rates of nitrate do not change. Fertilizer applications are expected to be targeted to locations where they will produce greatest yield increases, but average amounts per hectare of cropland are expected to remain about the same. This greater mass of applied nitrogen results in increased nitrate runoff for Scenario 1.

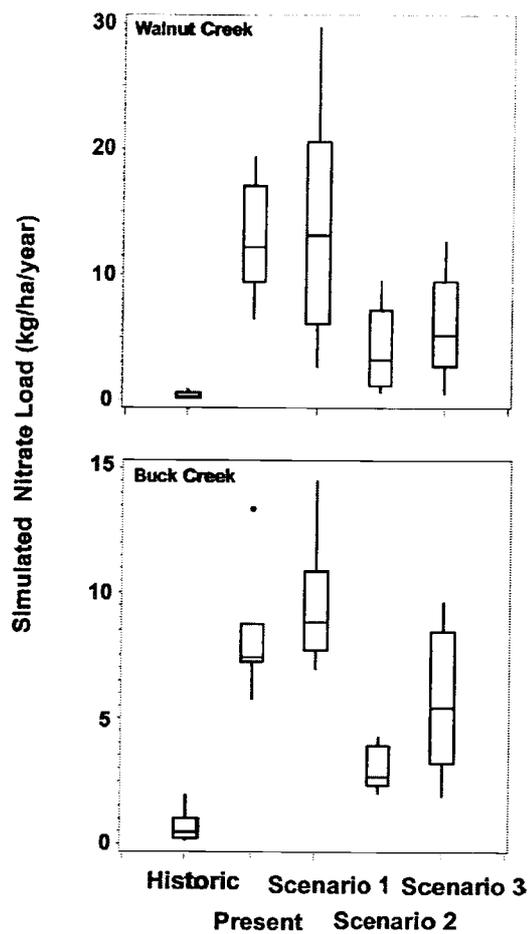


Figure 2.7. Boxplots representing nitrate loading in Walnut and Buck Creek watersheds.

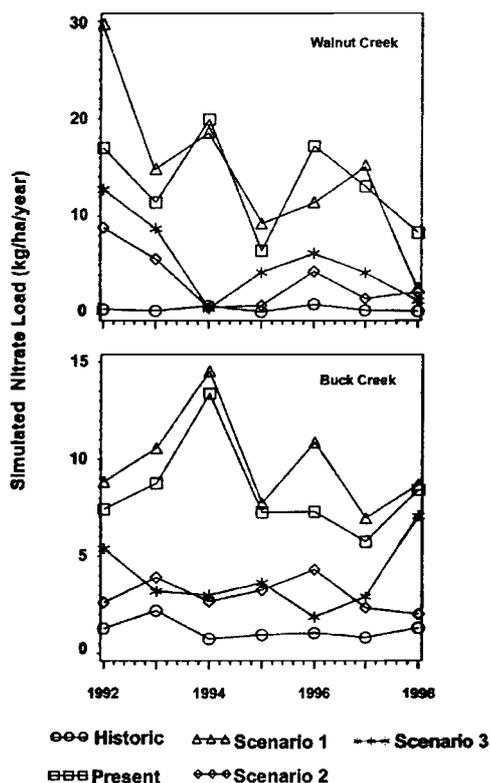


Figure 2.8. Time series plots of yearly nitrate loading in Walnut and Buck Creek watersheds for the study period. The time series help explain the variation noted in figure 2.7

The pattern of nitrate loading in Scenarios 2 and 3 mimics that of TSS, with significant decreases forecast in both cases, ranging from 57% to 70% reduction in the median nitrate load. In Scenario 2, much of the watershed area is still assumed to produce corn and soybeans (Table 2.1a) but the area is not in continuous corn and soybean rotation. Rather, the assumption is for a corn – soybean – alfalfa – alfalfa rotation. The two years of alfalfa are unfertilized and we assumed that application of nitrate to corn could be reduced by a modest 10%

(to 120 kg/ha/year of rotation as anhydrous NH_3). Additionally, a significant area of both watersheds (400 ha in Walnut and 3640 ha in Buck Creek) is converted to alfalfa production under the assumptions of Scenario 2. Pasture land also increases substantially in Scenario 2. Over the simulation period, significantly less commercial nitrate is applied to these systems and as a result, significantly less nitrate is simulated being exported from the watershed in the stream system.

Nitrate results from Scenario 3 suggest significant decreases from the present though, as with TSS, the largest improvements to water quality occur under Scenario 2. The area in corn and soybean production in both watersheds is greatly reduced in Scenario 3, replaced primarily with riparian areas and strip intercropping. In Scenario 3, the area in row crop production (as intercropping) is greater than in Scenario 2 and nitrate runoff is consistently higher in Scenario 3 as a result.

In Buck Creek, the design rules result in an extreme decrease in corn and soybean production as monocultures (it drops from 3823 ha in the present to 88 ha under the assumptions of Scenario 3). This decrease is due, in large part, to conversions from corn and soybean monocultures to strip intercropping and to the implementation of a 60m wide riparian buffer along perennial streams; this removes a major portion of Buck Creek watershed from conventional row crop production. The dendritic nature of the stream system in Buck Creek gives this 60 meter buffer an area of 1114 ha, or approximately 13% of the basin area. In Walnut Creek, again in part due to a less complicated stream network, design

rules result in a riparian buffer system that is approximately 263 ha or approximately 5% of the watershed area. Additionally, in Walnut Creek the area in corn and soybean production (including strip intercropping) in Scenario 3 is higher than in Buck Creek. Walnut Creek has 3656 ha or 70% of its area in enterprises that produce corn and soybeans whereas 5520 ha (62%) of Buck Creek is under corn and soybean production (intercropped) in Scenario 3. As a result of the physiographic differences between the watersheds, significantly less nitrate is applied to Buck Creek. Even given its larger area – it is 59% larger than Walnut Creek – model results and calibration data indicate loadings of nitrate are generally smaller than for Walnut Creek.

2.5.3 Evaluation and Comparison of Water Quality in Historic and Current Landscape

We include this analysis of historical water quality to provide a context for other simulation results and current water quality in the watersheds. All model results, and especially those attempting to model historical (or future) landcover have some degree of uncertainty – both in terms of accuracy of historical land use and model treatment of it. Despite this, we feel the analysis is useful in that it provides a benchmark to compare to the magnitude of change simulated for the future scenarios.

Simulated water quality under pre-development conditions appears significantly different than under current conditions. In Walnut Creek, the simulations using pre-development land cover yielded values representing reductions of 90% for TSS and 96% for nitrate. In Buck Creek, modeled reductions were 96% for TSS and 87% for nitrate. These very large simulated differences appear to result from three related factors. The first is our reconstruction of historical land-cover. Walnut Creek, with its relatively flat slopes, is modeled as a watershed with 2404 ha of wetlands. This represents 46% of the total watershed area and greatly changes the hydrology of the watershed. Estimated historic wetland area in Buck Creek is considerably less than in Walnut Creek, due for the most part to the greater topographic relief as compared to Walnut Creek. As a result, estimates of historic nitrate and sediment loading in Buck Creek are higher than similar estimates for Walnut Creek. The second factor involved in the significantly improved water quality under historic land cover is that nutrients were not added as chemical fertilizers to the systems, and so the potential for nutrient runoff was considerably less. The last factor responsible for the result is that in pristine prairie ecosystems, bare land surfaces are never developed through tillage. The relatively dense perennial cover results in reduced erosion potential and TSS concentrations. Historically, these watersheds stored more water in the upland wetlands, significantly reducing peak flows and development of stream channels. These reductions acted to reduce the potential for erosion and sediment delivery to streams, as well as bank erosion.

2.6 Summary and Conclusions

The results of this study have implications for improving water quality in rural areas. They lend support to an emerging view that to restore water quality and achieve substantial reductions (ca. 30-75%) in nutrient export in areas such as the Corn Belt will require major reconsideration of approaches. More specifically, this study indicates that achievement of major benefits in water quality will require major alterations of activities that take place in the watershed, in particular, modifications of the agricultural practices. In the current study, scenarios that included widespread implementation of agricultural enterprises in which nitrogen applications were substantially reduced (i.e., decreased by 10-33%) along with implementation of other BMPs, resulted in reductions in nitrogen export of >50%.

Scenario 1, in which BMPs including 3-6 m wide riparian buffers, filter strips, conservation tillage and precision agriculture were employed, but without substantial reductions in the total amount of nitrogen applied, showed increased nitrate export. These results indicate that continued use of present levels of nitrogen to fertilize crops will continue to result in nitrogen export to surface and ground waters greatly in excess of historical values.

Results indicate that conversion to no-till cultivation and residue management across large areas is, not surprisingly, an effective measure to reduce sediment concentrations. Only modest reductions in TSS occurred with exclusive

no-till cultivation, as envisioned in Scenario 1. Significant reductions of up to 65% occurred when other changes (including decreased row crop production and wider riparian areas) were also incorporated.

The results presented here are for two watersheds in Iowa, representative of the Des Moines Lobe and the Southern Iowa Drift Plain physiographic provinces. We encourage the further testing of these ideas, both with modeling and with watershed-level experiments [Likens et al., 197], on a wider selection of watersheds in other physiographic and agricultural settings. Such research is needed to understand the extent, costs, and benefits of various practices aimed at improving water quality in agricultural regions and downstream.

2.7 Acknowledgements

We acknowledge support from the U.S. EPA/NSF Partnership for Environmental Research STAR grants program, grant number R825335-01. Special thanks to Michael Shoup and Dr. Jerald Schnoor at the University of Iowa for assistance in the Buck Creek sampling program. We also thank Dr. Gerald Hatfield of the USDA-ARS Tilth Laboratory for providing us data collected in Walnut Creek, Story County, Iowa. Conversations with Dr. John Bolte regarding the use of simulation models significantly improved this manuscript.

2.8 Appendix - Key Land-use and Management Assumptions

Current Scenarios

Corn/Soybeans

Two year rotation of corn and beans on each field.

Single N application to corn in October. 183 kg/ha injected as anhydrous NH₃.

Single elemental P application to both corn and soybeans. 56 kg/ha to corn and 40 kg/ha to soybeans.

Chisel plow, with a mixing ratio of 0.25 – approx. 75% residue left on field. Corn plowed under in the fall, soybean crop is not.

Pasture

Generic grazing application on all pastures. Grazing occurs during two 28 day cycles (beginning 6/1 and 8/1). Pasture is killed in fall and replanted in early April.

Alfalfa

Alfalfa is managed on a generic 5 year multiple cut rotation. Planted early April of year 1, harvested twice in the first year. Harvested four times in each of the 2nd and 3rd years. Harvested twice in the 4th, crop is killed, but is harvested and killed again in the 5th year.

No fertilization.

Riparian Areas

SWAT provides a method to incorporate the quality of riparian areas. Two coefficients provide a quality index related to cover density and to the ability of the streambed to resist erosion. The current riparian area in both watersheds is generally thin and disconnected. The cover density index was set to a value representing average cover density in an effort to reflect riparian density.

Scenario 1

Corn/Soybeans

Same Rotation as current. Single N application to corn. 143 kg/ha anhydrous NH₃.

Single P application to both corn and soybeans. Same amounts as in current.

No Till conservation practices, with a mix ratio of 0.05 (leaving 95% residue)

Riparian Areas

Design rules indicate that the riparian buffers are well established and maintained throughout all perennial and ephemeral streams. In an effort to incorporate these buffers the cover coefficient was set to 1.0 (on a scale from 0.0 to 1.0, with 1.0 representing a well-vegetated riparian area)

Scenario 2

Corn/Soybeans

Corn crop is on a four year rotation of corn/soybeans/alfalfa/alfalfa. There is no application of N. The 3 year rotations of N fixing crops provide available N for

corn. All tillage practices are assumed to be conservation no till with a mix ration of 0.05.

Riparian Areas

As with Scenario 1, we assumed riparian areas were in very good condition and set the value of the cover coefficient to 1.0.

Scenario 3

Corn/Soybeans

The corn crop in Scenario 3 is modeled as a 2 year rotation with soybeans, using the same timing and amounts of fertilization as in Scenario 1.

Organic Crops

Modeled as a corn/soybean rotation, with no addition of N or P

Strip Intercropping

SWAT does not have the ability to simulate the growth of two different crops in a single field at the same time. In an effort to simulate plausible effects of strip intercropping, the practice was modeled as a corn soybean oat rotation. Corn was fertilized at 25 kg/ha of anhydrous NH₄ and P was applied at 56 kg/ha.

Riparian Areas

The coefficient was set to reflect well-maintained and complete riparian buffer systems.

3.0 Tree Data Structures with Hydrologic Process Models

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For Submission to
Computers and Geosciences

3.1 Abstract

A tree analogy of vector networks representing streams was developed and applied, with a focus on the modeling stream physical processes. Analyses indicate that the tree provides a useful basis for developing solutions to the partial differential equations representing stream dynamics. To illustrate this functionality, an example developed around St. Venant's equations for one-dimensional fluid flow was developed. Additional examples demonstrate the utility of this data structure in querying and summarizing spatially distributed data. These examples outline tree traversal methods which summarize upstream lengths and calculate Strahler stream orders. The algorithms are presented as object-oriented C++ code, and include interface code with map import/export and display capability, vector to tree conversion routines, and the hydrologic routing model. These concepts represent a step towards further integration of hydrologic process models, spatially distributed data, and Geographic Information Systems (GIS).

3.2 Introduction

Detailed digital representations of stream networks are increasingly used in the study of hydrological processes. One of the reasons behind this is the proliferation of readily available digital elevation data and methods to convert those data to

stream networks [Costa-Cabral and Burges, 1994; Peckham, 1998; Renssen and Knoop, 2000]. These detailed representations are clearly useful for mapping purposes and for the development of a variety of stream metrics (e.g. slope, aspect). In addition, it would appear that direct application of these datasets within numerical models would also be useful. The methods presented here facilitate the use of vector networks in numerical models through the automatic conversion of detailed vector networks to a generalized tree data structure. In cases where the networks originate from DEM data, corollary information such as slope or aspect can be calculated and stored within the data structure.

The goal of this study was to develop methods for directly using non-topological vector networks in models of hydrologic process. We begin with a description of a tree data structure well-suited for the representation of stream networks. We then introduce a solution to a common streamflow routing equation utilizing both derived data from the DEM and the tree structure. The paper concludes with an example designed to reflect the utility of the approach. This set of algorithms adds to a growing list of applications designed to fully integrate geographic information systems, the data developed with them, and models of environmental transport.

3.3 The Tree Data Structure

The tree is a well-established data structure used to store hierarchical information [Shaffer 2001]. Trees are recursive structures consisting of a root node with zero or more subtrees (children) connected to that node. Each subtree is similarly defined. Each node is connected by a link establishing the topology of the tree structure, and contains a pointer to each child subtree originating from it. Nodes with greater than zero subtrees are referred to as branches; nodes with zero subtrees are referred to as leaves.

Trees generally fall into one of two categories. The first is the set of trees where each node has the same number of direct children. In the case of a binary tree, each node possesses references to two children, known as the left and right children. These two pointers are valid for all branches but are both empty in the case of leaves. Figure 3.1 is a depiction of a binary tree and a set of corresponding data.

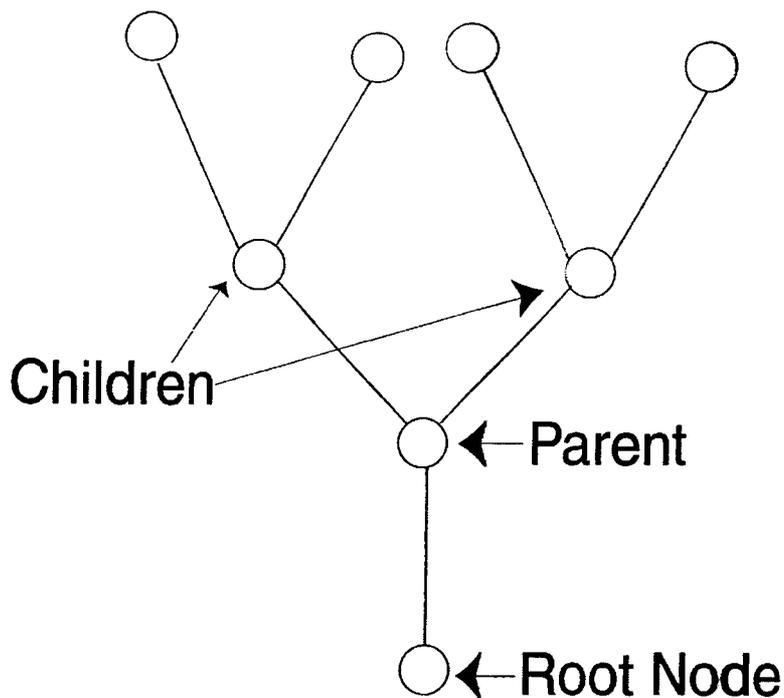


Figure 3.1. An example of a binary tree.

The special case of the binary tree is straightforward, and appropriate for many sets of hierarchical data. But in many instances, the nature of the data does not adhere to the binary assumptions – some branches may not have exactly two children. Tertiary trees consist of branches with three children; quad trees are those with four children. More generally, this set of trees is commonly referred to as k -ary trees, where k represents the number of children allocated per node. K -ary trees represent the simplest tree data structure because a collection of elements, all with a single unique definition, sufficiently characterizes the entire tree.

The second, and more complicated, case is often referred to as a general tree. A general tree is defined as a tree with a variable number of children per node (Figure 3.2)

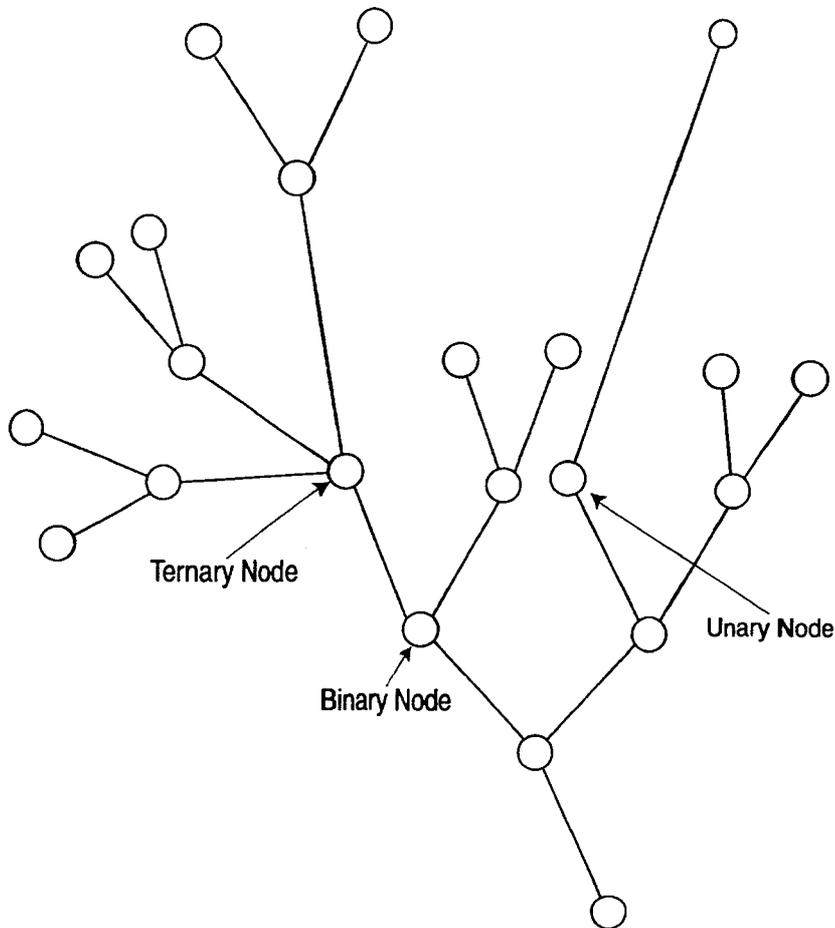


Figure 3.2. An example of a general tree.

Because each element potentially maintains information relating to a different number of children, implementation of the general case is more difficult. One solution is to simply include references to a maximum number of children, using a k -ary tree. Implementation of this case is straightforward, but would

result in a potentially large number of unnecessary pointers. These null pointers significantly slow the process of querying the tree and this is therefore a non-optimal solution. The parent pointer concept is another solution for the general tree. In this approach, each node stores data identifying itself and its downstream node – and no information about its children. The parent-pointer tree is a compact data structure – each link needs only store information about itself and its parent, and these values are null only for the single case of the root node. The drawback to the parent-pointer tree is that recursive search towards children is not possible – the required information is not maintained.

Despite differences between different tree implementations, trees are an appropriate structure for the storage and maintenance of hierarchical data. The major reason for the suitability is that the data structure mirrors the hierarchy of the data it represents. Building known relationships between data points in the data structure greatly facilitates the processes of query and insertion.

As an example of the structure's utility, consider querying the tree representing stream reaches for the total length of the stream network, given that each node stores information about the length of its corresponding length. Total length can be computed by traversing the entire tree, accumulating lengths across the traversal process. A recursive traversal through the tree structure begins at the root of the tree, and can proceed in either a depth-first or breadth-first fashion. Depth-first searches traverse the tree along its left-most branches until a leaf is found; it then “backs up” to the nearest parent node not fully traversed. This

search process is repeated until the entire tree has been traversed. Breadth-first search proceeds by exhaustively examining a nearest node, then recursively examining child nodes, until the entire tree has been examined. We can apply either approach to efficiently compute total stream length in a network. For example, a depth-first search would start at the root of the tree. The search recursively moves from the current node to its left child, until a leaf of the tree is found. The cumulative length is computed based on the length so far plus the length of the current node. The algorithm then backs up to the nearest uncomputed node and repeats, until all nodes have been computed. At this point, the cumulative length will contain the total reach length of the network.

3.4 Trees and Stream Networks

The generation of trees to represent non-topological vector GIS formats imposes a set of constraints that must be incorporated into the development of the data structure. The most important consideration is that stream networks are best represented as general trees. Of course, most elements in a stream network will tend to be binary. But, in cases where stream networks are generated from DEM data, tertiary nodes are common and up to 8-ary nodes are possible, and would occur, by definition, in the case of a sink. Because the number of children may vary depending upon the topology of the surface, most stream networks must be treated as general trees.

Non-topological vector networks are commonly defined through relatively simple collections of vertices, where the vertices are collected into links representing stream segments. Topologic information is inherent in this geographic data, but is not stored directly. Each stream segment knows which vertices represent it, but not where it exists in the network. The development of a tree from this dataset requires an explicit statement of the inherent topology. An example of non-topological vector network definitions is ESRI's ArcView™ shapefiles. These are a common format for the storage of vector networks, and are used as the basic input vector datasets in this study. It should be noted that the following methods could also be directly applied to vector networks where topology is defined.

Here, we present a tree representation for creating and storing topological and attribute data for these non-topological vector networks using an efficient two-way binary tree structure. The method explicitly accounts for the non-binary cases of parents with more than two children. The use of a binary tree provides for a simple, compact storage mechanism and rapid traversal capabilities. We extend the binary concept to effectively address networks containing n-order relationships, while maintaining the efficiency of binary trees using the concept of "phantom nodes" described below. The development of a simple binary tree representation, with special treatment of the unusual, yet very important, non-

binary cases common in DEM-derived stream networks, is useful for a broad range of hydrological applications.

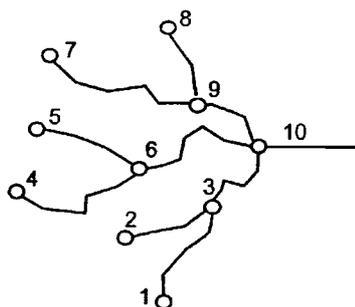
As noted above, stream networks derived from DEMs may contain nodes with more than two intersecting stream reaches. To handle the case of a node with more than two children, we introduce the concept of a phantom node. The phantom node is an introduced node with no physical representation. It allows the traversal algorithm presented below to represent stream networks, which are general trees, as simple binary trees. This removes the necessity to provide storage for more than two children, and its implementation is simpler than other general tree solutions. If the tree-building algorithm encounters a parent with more than two children, it adds a phantom-node to the tree. This additional node acts as the second child of the current node and the parent of the would-be third child. Figure 3.3 depicts the phantom node concept. The phantom node is in effect a zero length reach. Figure 3.4 represents the complete set of steps necessary to develop binary tree topology from a set of vertices.

The conversion of sets of vertices in non-topological vector networks to trees requires a decision regarding the number of vertices in the vector network represented by each node in the tree. If we define a reach as the length between any two junctions in the vector network, a reasonable choice is to use the vertices representing each reach. Under this definition, each node in a binary tree represents a reach in the vector network. A drawback is that the lengths of reaches vary depending on the stream network. For numerical purposes it may be

useful to maintain smaller length reaches, over which finite difference solutions can be solved. To overcome this potential drawback, each node maintains an array representing smaller length sub nodes of the reach. The size of the sub node array is calculated before each model run and equals a user-specified Δx divided by the length of the reach. In cases where the Δx is longer than the reach length, the sub node array contains a single element, with a length equal to the stream length.

Step 1 - The General Tree Problem

Assume the goal is to convert the network below to a binary tree. Each node maintains pointers to two upstream nodes and one downstream node. But the binary assumption fails for node number 10.

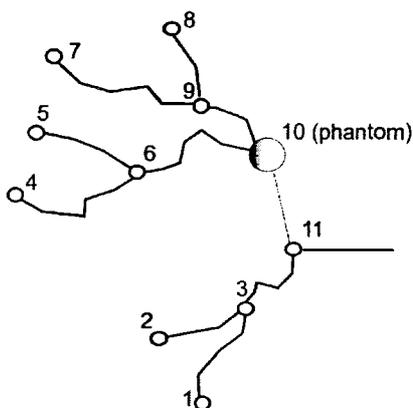


| NodeID | LeftNode | RightNode | DownstreamNode |
|--------|----------|-----------|----------------|
| 1 | NULL | NULL | 3 |
| 2 | NULL | NULL | 3 |
| 3 | 1 | 2 | 10 |
| 4 | NULL | NULL | 6 |
| 5 | NULL | NULL | 6 |
| 6 | 4 | 5 | 10 |
| 7 | NULL | NULL | 9 |
| 8 | NULL | NULL | 9 |
| 9 | 7 | 8 | 10 |
| 10 | 3 | 6 | |

What happens to node 9 which is also upstream of node 10?

Step 2.

Add the phantom node and reconnect pointers to maintain the binary tree assumption. Note how the general tree depicted above effectively becomes a binary tree.



| NodeID | LeftNode | RightNode | DownstreamNode |
|-------------|----------|-----------|----------------|
| 1 | NULL | NULL | 3 |
| 2 | NULL | NULL | 3 |
| 3 | 1 | 2 | 10 |
| 4 | NULL | NULL | 6 |
| 5 | NULL | NULL | 6 |
| 6 | 4 | 5 | 10 |
| 7 | NULL | NULL | 9 |
| 8 | NULL | NULL | 9 |
| 9 | 7 | 8 | 10 |
| 10 | 6 | 9 | 11 |
| 10(phantom) | 3 | 10 | |

Figure 3.3. The phantom node solution to the general tree problem.

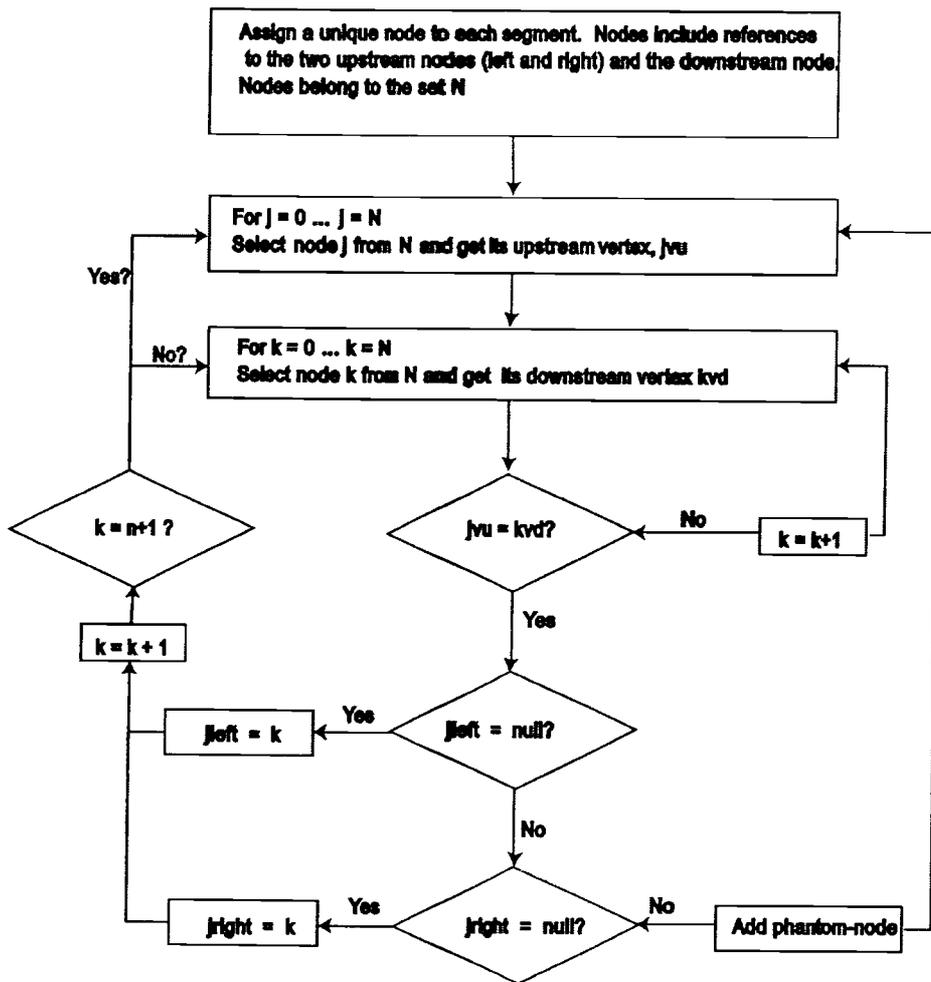


Figure 3.4. A flowchart representing the steps in the development of a phantom node binary tree from non-topological stream network models

3.5 Streamflow Routing

St. Venant developed the equations used to represent one-dimensional unsteady open channel flow in 1871. [Chow et al., 1988] The equations are essentially restatements of the conservation of mass (continuity) and of momentum, commonly reported as follows:

Continuity

$$\frac{\partial Q}{\partial x} + \frac{\partial Q}{\partial A} = q \quad (3.1)$$

Conservation of Momentum

$$\frac{1}{A} \frac{\partial Q}{\partial t} + \frac{1}{A} \frac{\partial}{\partial x} \left(\frac{Q^2}{A} \right) + g \frac{\partial y}{\partial x} - g(S_o - S_f) = 0 \quad (3.2)$$

Local
Convective
Pressure
Gravity
Friction
Acceleration
Acceleration
Force
Force
Force

where Q is discharge (m^3/s), x is distance (m), A is channel cross sectional area, (m^2), q is the discharge per length of channel (m^2/s), g is the acceleration due to gravity, S_o is the slope of the water surface, and S_f is the slope of the channel bed. These two equations fully describe discharge as a function of both time and space. The solution to these equations requires the estimation of a variety of hydraulic properties and is beyond the scope of this analysis. Simplifications are common though, using a variety of assumptions about the movement of water and the importance of different terms. The kinematic wave is one such solution [Wooding, 1965; Ponce, 1978]. If the kinematic wave dominates the hydrograph,

the acceleration and pressure force terms can effectively be ignored [Ponce, 1977]. Removing them from the momentum equation simplifies Equation 3.2 to the assumption that gravity forces equal friction forces. In other words, at any point in time and space the discharge does not change; the flow is uniform. Because the flow is uniform we can replace the momentum equation with a standard uniform resistance equation such as Manning's or the Chezy equation [Bedient and W. 1992]. A commonly cited version of the kinematic flow equation [eg. Martin and McCutcheon, 1999], using Manning's equation, is:

$$\frac{\partial Q}{\partial x} + \alpha\beta Q^{*\beta-1} \left(\frac{\partial Q}{\partial t} \right) = q \quad (3.3)$$

where Q is the discharge (m^3/s), x is the horizontal distance (m), α and β are parameter combinations derived from Manning's equation, t is time, and q is the unit discharge (m^2/s).

3.5.1 Solution Procedure

An implicit finite difference solution is as follows [Chow et al., 1988]:

$$\frac{\partial Q}{\partial t} = \frac{Q_{time} - Q_{time-1}}{\Delta t} \quad (3.4)$$

$$Q^* = \frac{Q_{time-1} + Q^{in}}{2} \quad (3.5)$$

$$\frac{\partial Q}{\partial x} = \frac{Q_{time} - Q_{time-1}^{in}}{\Delta x} \quad (3.6)$$

Solving for the unknown value Q_{time} and setting

$$z = \alpha\beta Q^{\beta-1} \quad (3.7)$$

$$Q_{time} = \frac{q_{time} + \frac{Q^{in}}{\Delta x} + z \frac{Q_{time-1}}{\Delta t}}{\left[\frac{1}{\Delta x} + \frac{z}{\Delta t} \right]} \quad (3.8)$$

This equation represents the unknown value of Q as a function of known values related to upstream discharges (labeled with a superscript 'in'), previous time discharge (label with a subscript 'time-1'), and normal flow parameters, which make up the z term. The binary tree representation provides the topology necessary to solve the equation over the entire network through a traversal algorithm similar to that presented in section 3.2.

3.6 Application and Results

Code representing the algorithms presented above was developed using C++. The software uses a simple interface that provides a shapefile reader and map display capabilities. A simple application was developed to characterize the kinematic wave routing model describe above over a tree representation of a realistic stream network. The stream network represents Bear Creek, an approximately 100 square kilometer watershed in the Long Tom basin of the Willamette Valley, Oregon and originated from a mosaic of two USGS DEM datasets (Figure 3.4). Arc/Info™ software was used to develop the flow direction

and accumulation grids, and gridded streams were then generated using a 10 ha threshold area. This grid was converted to Environmental Systems Research Institute (ESRI™) shapefiles, which corresponds to stream network portrayed in figure 3.5. All subsequent processing occurred within the software framework described here. The steps taken during processing are outlined in Figure 3.6.

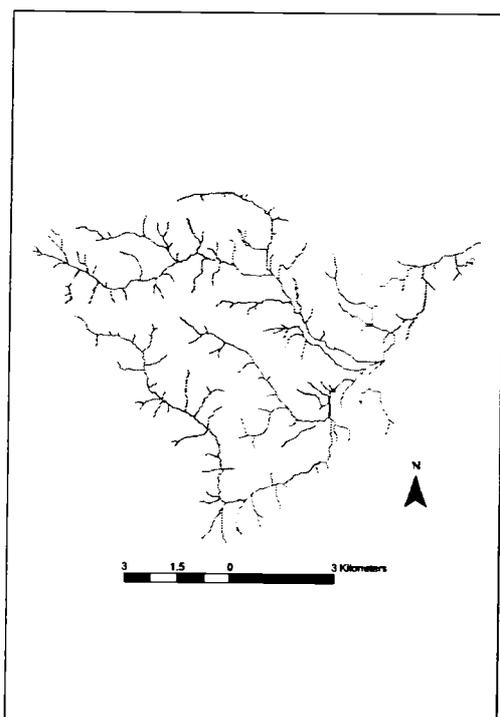


Figure 3.5. The Bear Creek stream network. The network is made up of 351 individual reaches. A tree representation of this dataset is used as the basis for a solution to the kinematic wave equation.

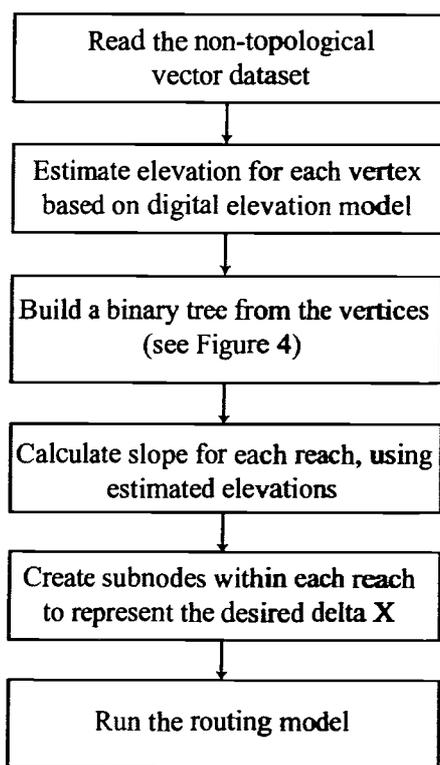


Figure 3.6. A flow chart outlining the steps involved in processing ArcView™ shapefiles.

Evaluations of the network model and the use of a tree as a finite difference solution matrix were performed using a simple synthetic lateral inflow hydrograph. These synthetic data were routed through the more realistic hydrologic network. The use of a synthetic dataset facilitates visualization of the model's capacity to accumulate and translate a potential flood wave over detailed networks, without the complications more realistic inputs might introduce. In addition to representation of time series outputs at select locations in the basin, we

also present a set of statistics which are calculated using the binary tree and represent useful spatial quantities describing all of the reaches in the basin.

For this simulation, each reach received the time series lateral inflow shown in Figure 3.7. This simple inflow is designed to simulate a spatially uniform input rate of water, without using the more detailed upslope model outlined in Chapter 4 of this document. In this way we are better able to confine the analysis to the network routing model.

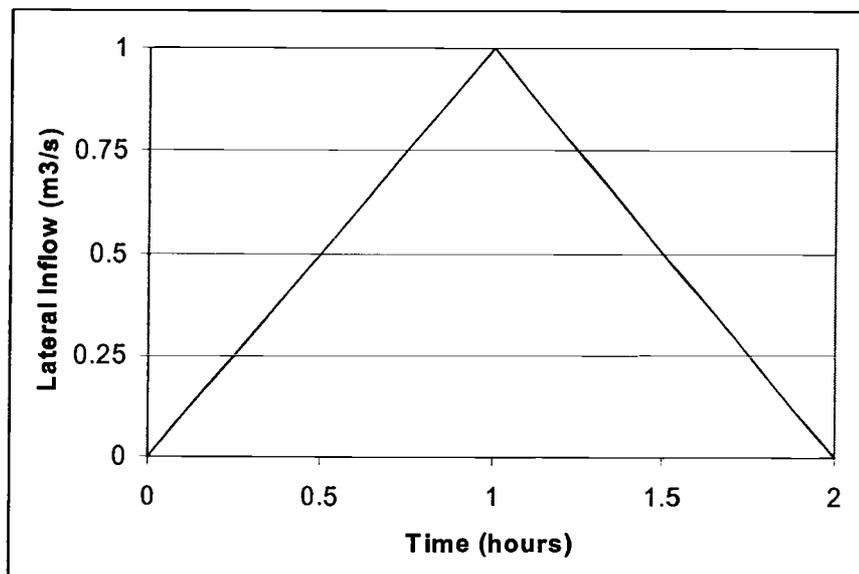


Figure 3.7. The triangular inflow hydrograph imposed upon each reach in the Bear Creek network.

The model was run with a maximum Δx of 500 meters and a Δt of 0.001 days. In the frequent cases where the reach length was less than the maximum Δx , the reach length was used as the space step. Results indicate that the tree

structure, in combination with a finite difference approximation of the kinematic wave, reasonably simulates the dynamics of simple flood wave (figure 3.8). In figure 3.8, a representative reach was selected for each stream order. The accumulated area above each of the reaches is reflected in the increase in discharge magnitude with order. The figure also reflects the channel residence time, with peak flows occurring later in time in the higher order reaches. The series representing first order streams (and to some degree second) are difficult to interpret in Figure 3.8 because of the large peak flow differences between the different stream orders.

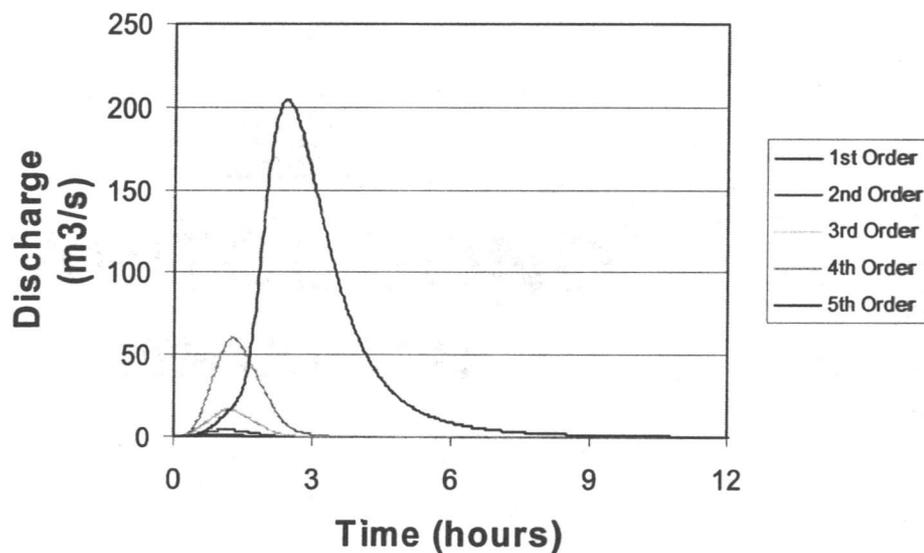


Figure 3.8. Stream discharge from a variety of reaches corresponding to the inflow hydrograph in figure 3.7. Each series represents an individual stream order.

The important result is that the first order output is essentially equivalent to the first order input, which is more clearly outlined in Figure 3.7. The tree datasets also lend themselves to calculations related to spatially distributed physical characteristics of watersheds. These include values such as upstream channel lengths and drainage areas. The imposition of a tree structure allows calculation of these values for every point within a basin. To demonstrate this functionality, distributed model results corresponding to figure 3.8, but representing every reach in the Bear Creek network were calculated and summarized for each stream order (Table 3.1). Referring to table 3.1, note that for first order streams the mean peak discharge is approximately $1.0 \text{ m}^3/\text{s}$, corresponding to the maximum input rate of water (figure 3.7).

Table 3.1. Statistical descriptions of model results for all stream reaches in the Bear Creek watershed.

| | Area (ha) | Total Upstream Area (ha) | Time to Peak Discharge (hours) | Lateral Inflow (m3) | Stream Inflow (m3) | Stream Outflow (m3) |
|------------------|--------------|-----------------------------------|---|---------------------------|--------------------------|---------------------------|
| Stream Order = 1 | | | | | | |
| N | 177 | 177 | 177 | 177 | 177 | 177 |
| Mean | 23.6 | 23 | 1.03 | 3888 | 0 | 3882 |
| Std Dev | 13.5 | 13 | 0.04 | 0 | 0 | 5.6 |
| Min | 10.2 | 10 | 1.01 | 3888 | 0 | 3855 |
| Max | 93.1 | 93 | 1.44 | 3888 | 0 | 3910 |
| Stream Order = 2 | | | | | | |
| N | 75 | 75 | 75 | 75 | 75 | 75 |
| Mean | 19.9 | 102 | 1.07 | 3888 | 14865 | 18841 |
| Std Dev | 20.5 | 61 | 0.06 | 0 | 10332 | 10400 |
| Min | 0.0 | 30 | 1.0 | 3888 | 7751 | 11642 |
| Max | 85.7 | 293 | 1.39 | 3888 | 55032 | 58955 |
| Stream Order = 3 | | | | | | |
| N | 67 | 67 | 67 | 67 | 67 | 67 |
| Mean | 17.6 | 807 | 1.27 | 3888 | 163703 | 167983 |
| Std Dev | 16.9 | 552 | 0.18 | 0 | 115448 | 115585 |
| Min | 0.0 | 192 | 1.05 | 3888 | 23376 | 27555 |
| Max | 84.5 | 2114 | 1.78 | 3888 | 426059 | 430391 |
| Stream Order = 4 | | | | | | |
| N | 20 | 20 | 20 | 20 | 20 | 20 |
| Mean | 17.6 | 2257 | 1.49 | 3888 | 443652 | 448991 |
| Std Dev | 16.0 | 814 | 0.23 | 0 | 140353 | 141202 |
| Min | 0.7 | 1100 | 1.2 | 3888 | 240069 | 244061 |
| Max | 55.6 | 3473 | 1.94 | 3888 | 662501 | 667300 |
| Stream Order = 5 | | | | | | |
| N | 12 | 12 | 12 | 12 | 12 | 12 |
| Mean | 18.7 | 6859 | 2.11 | 3888 | 1342099 | 1349606 |
| Std Dev | 16.9 | 411 | 0.22 | 0 | 88636 | 89055 |
| Min | 0.2 | 6148 | 1.8 | 3888 | 1200600 | 1204600 |
| Max | 47.6 | 7428 | 2.42 | 3888 | 1460890 | 1468800 |

The first order reaches receive no upstream input and outflows from them are essentially equivalent to lateral inflows. As the stream order increases, upstream inputs become increasingly dominant, while lateral inflows remain constant.

3.7 Conclusions

Processes designed to generate networks are well established, but simple and effective methods to directly utilize the derived streams in distributed models of hydrology are not generally available. The tree definition and routing procedure outlined here provide an opportunity to improve hydrologic models through the direct use of detailed representations of stream networks.

The paper demonstrates that through the development of binary trees, with the phantom node concept, we have automated the process of generating a topology necessary to solve finite difference equations across complex stream networks. The structure has the added benefit that stream order and upstream contributing areas of each reach can be quickly calculated. In a general sense, the simulations indicate that the processes outlined to generate binary trees and recursively traverse them perform well across even large stream networks. In addition, the binary tree data structure employing phantom nodes automates the process of calculating spatially distributed metrics describing watershed. These calculations have significant utility in studies focused on hydrologic modeling, but also may be of use in a more standard GIS context, where the calculation of data such as upstream area is not always standard.

Here, we presented a solution for the kinematic wave; implementation of other solutions would proceed in a similar fashion. Possibilities of other applicable models include storage routing techniques, link-node unit hydrograph,

as well as somewhat more complex diffusion analogies for fluid flow. In addition, the procedure may have utility in modeling the movement of other hydrologically interesting parameters including sediment, nutrients and stream temperatures. Further extensions of the tree data structure are presented in Chapter 4 and applied again in Chapter 5. These extensions include the addition of an upslope model that uses the tree to spatially distribute model calculations.

**4.0 The WET Hydro Model: Development, Application and Analysis of
the Effects of Scale and Multiple Output Criteria on Model
Parameterization**

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For Submission to Water Resources Research

4.1 Abstract

Research on modeling of hydrologic processes continues to evolve and has potential to benefit significantly from improvements in Geographic Information Systems (GIS) and from experimental studies focused on the development of datasets that can be used to constrain or quantify model uncertainty. In order to realize this potential, a focus on development of methods to incorporate these datasets into hydrologic models is needed. We present a newly developed model of hydrologic process and apply it in three different regions. Application occurs over a variety of temporal and spatial scales, and dominant hydrologic processes vary considerably across the basins. This rigorous test of the model functioning suggests that overall model performance is similar to other hydrologic models, with model efficiencies of over 0.9 for some basins and events. We also demonstrate that the uncertainty of these results, from a parameter and structural standpoint, is highly significant. The effect of numerical error on simulation results is shown and numerical methods designed to limit dispersion are presented. Additionally, simple particle tracking methods are implemented and used to develop model based hydrograph separations. We show how these extensions allow for the rejection of unacceptable model structures that would otherwise appear acceptable from standard time series analysis of output hydrographs. The effect of model scale is demonstrated and a set of simple model derived statistics is shown to provide further insights into the distributed processing of state variables.

4.2 Introduction

The translation of rainfall into stream runoff involves the interaction of a wide variety of potential flowpaths. These flowpaths can be broadly categorized into Hortonian overland flow, saturated overland flow, macropore flow, subsurface matrix flow, and groundwater movement [Dunne and Leopold, 1978].

The contribution of each of these pathways to a hydrograph varies with the physical characteristics of the watershed, and it changes under different rainfall intensities and soil water conditions. Understanding of these processes has progressed significantly in recent years due to a variety of hillslope and headwater catchment experiments focused on distributed point scale measurements and tracer studies [Sklash et al., 1986; McDonnell et al., 1990; Bonell, 1993; Anderson et al., 1997; Montgomery et al., 1997]. The body of work seeking to experimentally quantify flowpath dynamics in larger watershed-scale basins (1 to 1000 km²) is considerably smaller. This is due at least in part to the recognition that the utility of point scale measurements decreases as basin size increases [Klemes, 1986; Beven, 1989; Grayson et al., 1992]. The cost of such projects is also a concern. Those experimental studies that do focus on watershed-scale flow path dynamics tend to concentrate on the information content of watershed-scale watershed tracers [Kirchner et al., 2000; Kirchner et al., 2001; Uhlenbrook et al., 2001]. The tracer methodologies outlined in these studies have potential to

significantly increase understanding of watershed-scale flowpath dynamics, but the work is still in its initial stages.

Modeling studies are commonly undertaken as either alternatives to, or extensions of measurement-based examinations of watershed flowpath dynamics. Input requirements, spatial discretization, and simulated processes define a continuum over which watershed models fall [Woods and Sivapalan, 1999]. Physics-based and fully distributed watershed models are well represented in the literature. One of the reasons for their popularity appears to be the conceptual clarity of taking measurable, well-founded point-scale concepts, applying them at many points, and making statements about watershed-scale hydrologic behavior. Two general groups of physically-based models have evolved. The first group is includes the fully-distributed process-based models, and is represented by models such as the System Hydrologique European (SHE) [Abbott et al., 1986]. These models have been applied to hydrologic problems and have also been extended to evaluate water quality [La March and Lettenmaier, 2001]. A second group freely combine empirical models, like the SCS equation, with physics-based models (eg. Penman Monteith equations for evapotranspiration (ET)), and a variety of routing mechanisms. Examples include SWAT [Arnold et al., 1995], AGNPS [Binger and Theurer, 2001], and ANNAGNPS [Yuan et al., 2001]. These models extend the empirical equations in time and space to produce continuous simulations of a wide variety of water quality and quantity metrics. The distributed empirical models have generally been developed specifically to evaluate landuse change at

watershed-scales, especially in agricultural areas. A variety of studies have utilized these models to produce useful statements regarding landuse change and the effects on a variety of hydrologic metrics [Eheart and Tornil, 1999].

Despite widespread acceptance of these models, an alternative view has emerged with the understanding that complications arising from non-linear dynamics and uncertainty significantly reduce the utility of the fully distributed physically/empirically based models [Beven, 1989]. In response to this concern, a variety of models have appeared which focus on conceptual modeling of hydrology [Beven and Kirkby, 1979; Bergstrom, 1995]. These models distinguish themselves from the former by avoiding point-scale, first-principle descriptions of water movement. Rather, they focus on understanding important flow paths, and rely on calibrated rate equations to represent model fluxes. They have shown significant potential to incorporate more directly the information gained through experimental field studies [Seibert and McDonnell, 2002]. This is due in part to the fact that the models can maintain explicitly water volume and tracer mass across time and space. This information corresponds to the data derived from watershed tracer studies, and this link between field data and model state provides at least the opportunity to compare internal model state with measured quantities representing real world dynamics.

Development of *fully-distributed* conceptual models is a relatively new idea that has arisen from limitations identified in their aspatial or lumped counterparts [Vertessy and Elsenbeer, 1999]. The most commonly cited

limitations center around the steady-state assumptions necessary to produce a simple quasi-spatial model (TOPMODEL is the most commonly referenced example) [Wigmosta and Lettenmaier, 1999; Seibert et al., 2003]. As a result of this steady-state assumption, hillslopes wet and dry at a constant rate that parallels stream discharge. While this may not diminish the capacity to simulate the hydrograph, extensions focused on watershed biogeochemistry, and the landuse changes that play a role, may be significantly affected.

Conceptual models that explicitly include *distributed* volumes and important flow pathways have significant potential to improve the internal hydrologic dynamics of such simulations. At the same time, distributing the models increases the number of parameters and in turn, reduces the ability to produce estimates of model uncertainty. Despite this drawback, the ability to incorporate spatially-distributed input data, whose accuracy and precision is continually improving, and produce spatially-distributed outputs is strongly sought, especially, as noted above, to underpin watershed-scale simulations of watershed chemistry and sediment movement. For this reason, application of these models in future studies involving landuse change seems likely. A variety of studies have evaluated different variants of the conceptual box model at the hillslope scale, but considerably less work has provided detailed application of conceptual box models at the watershed-scale.

The focus of this paper is the description of a fully (and flexibly) distributed conceptual model representing a variety of important flow pathways

commonly identified in watershed-scale basins. We include results of a global mass balance to demonstrate the effectiveness of vector based input routines and model functioning and introduce important components of the model. In order to address the significant challenge to fully characterize a watershed-scale model, we present a range of simulation results. Applications focus on four separate and quite different sites over a wide variety of hydrologic conditions. The frequency and duration and input and output measurement varies, along with the size of the basins and the dominant hydrologic regime. Our goal is not so much to introduce these basins and the data to describe them, as it is to fully characterize the strengths and weaknesses of the model.

Several model runs are designed to characterize the effect of model scale on output. In this case we refer to scale not as the size of the basins being simulated, but the size and number of upland units used to represent watersheds. We have developed simulation code that can quickly update unit sizes between model runs under the realization that scale may be a parameter as much as, for instance, saturated hydraulic conductivity. Here we provide some initial guidelines on how to approach the decision.

In addition to comparing measured versus modeled hydrograph outputs, we also identify a variety of model outputs that can be used to more fully characterize the quality of the simulation results. These results include the percent of area and time over which overland flow occurs and estimates of new water contribution to single event hydrographs. New water contribution is

calculated using simulated tracer and two-component hydrograph separation using average storm concentrations. We show how these results can be used as additional criteria to identify 'non-behavioral' simulations and further reduce the number of acceptable parameter sets, or reject model structures.

4.3 The WET Hydro Model

4.3.1 Upland Soil Water Model

The upland model employs a conservation of mass equation:

$$\frac{dV}{dt} = P + SS_{in} + SOF_{in} - ET - G - SS_{out} - SOF_{out} \quad (4.1)$$

where V is the volume of water in each element (cubic meters), t is current time (days), P is the precipitation rate (m/d), SS_{in} is the rate of subsurface inflow (m/d), SOF_{in} is the input rate of saturation overland flow (m/d), ET is the evapotranspiration rate (m/d), G is the loss to groundwater (m/d), SS_{out} is the rate of subsurface outflow (m/d) and SOF_{out} is the output rate of overland flow (m/d).

Assuming the density of water is 1.0, this is a statement of the conservation of mass and the solution to the equation represents an estimate of the volume of water for each unique element through time. Each upslope element is treated as fully mixed, with two conceptual buckets representing the saturated and unsaturated zones. The differential equation is solved using a variable time step Runge-Kutta-Felberg solution procedure outlined in Press et al., [1992].

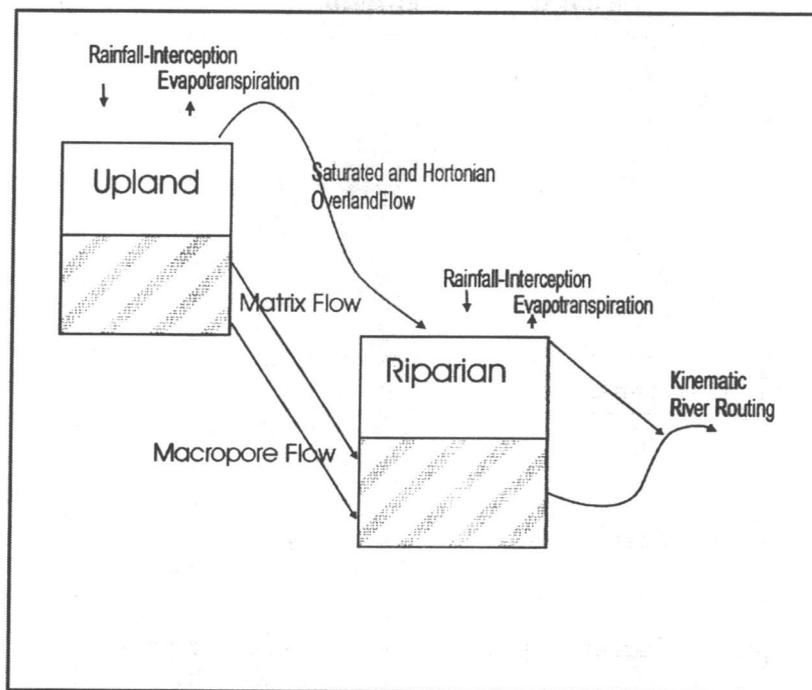


Figure 4.1. A conceptual diagram representing the processes simulated by WET_Hydro.

(Figure 4.1) The buckets are defined by their vertical lengths. An increase in water volume results in an increase in the depth of the saturated zone, and a corresponding decrease in the depth of the unsaturated zones. The relationship between the two zones is defined as:

$$z_t = z_u + z_s \quad (4.2)$$

where z_t is the depth of the unit, z_u is the depth of the unsaturated zone and z_s is the depth of the saturated zone. All infiltrating water is assumed to enter and exit the element from the saturated zone. The top of the z_u store represents the water table, and the saturation deficit of each unit can be calculated as:

$$S_d = (z_t * \phi) - (z_t * \theta) \quad (4.3)$$

where ϕ is the porosity of the soil and θ is the soil water content, which is calculated from the current water volume as:

$$\theta = V / A * z_t \quad (4.4)$$

where V is the water volume, A is the area of the unit and z_t is the depth of the soil.

4.3.2 Precipitation Input

The large spatial variability in precipitation data has been established by a number of researchers [Koren et al., 1999]. It is reasonable to assume that in the large, topographically diverse watersheds simulated as part of this study, a small number of point precipitation measurements are likely not representative of spatially-distributed rainfall volumes. To provide more accurate estimates of the spatial variability in rainfall, we utilize, where feasible, output from The Polynomial Regression on Independent Slopes Model (PRISM) model. PRISM was developed to provide spatially-distributed estimates of climatic variables over a variety of time scales [Daly et al., 1997]. WET_Hydro uses the spatially-explicit, long-term, 30-year average climate data to develop a rainfall modification factor that varies in space.

$$\sum_{i=0}^{i < num.UpslopeUnits} Precip_i = Precip_i * \frac{LongTerm Precip_i}{LongTerm Precip_{measurementSite}} \quad (4.5)$$

More measures of meteorological data in both space and time could significantly reduce a large source of uncertainty in the model. The collection of additional data, and/or the development of better models of climatic variability, was beyond the scope of this study. However, the system is designed to readily use any additional information that can be supplied. This is an area where further testing is warranted.

4.3.3 Matrix Flow

Lateral flux of soil water is assumed to be a function of the land surface slope, approximating the hydraulic gradient and the effective conductivity. Conductivity is assumed to decline exponentially with depth, and the rate is expressed as follows:

$$SS = K_{eff} * Slope * e^{-\frac{s_d}{m}} \quad (4.6)$$

where SS is the rate of subsurface downslope movement (m/d), K_{EFF} is the effective hydraulic conductivity (m/d), slope is the ground surface slope as a ratio, s_d is the saturation deficit (m), and m is a parameter that expresses the change in conductivity with depth. Note that when the soil is fully saturated, the saturation deficit is zero and the soil drains at the saturated hydraulic conductivity. The drainage rate declines exponentially with a decrease in the saturation. The shape

of the exponential function defining this decrease is modulated by the parameter m .

4.3.5 Overland Flow

Infiltration is assumed to occur when the soil is not saturated. But during periods of high rainfall, or where the lateral movement of soil water is low (low conductivity soils, and or shallow sloping lands), the saturation deficit is often zero and infiltration cannot occur. In these instances, excess precipitation is ponded and subsequently delivered directly to the adjacent downslope unit. This delivered water infiltrates if the downslope unit is unsaturated, and ponds if the downslope unit is fully saturated. In the case of an upland unit that drains into a reach, the ponded volume is routed directly into the channel. Currently, rainfall excess overland flow is not simulated.

4.3.6 Evapotranspiration

Estimates of evapotranspiration were available for Maimai, but model estimates were necessary in the other three catchments. In cases where estimates were necessary, we use the Hargreaves equation [Hargreaves and Samani, 1985]. The model assumes that air temperature is generally correlated with radiation, which is the most significant driver of evapotranspiration. The equation is as follows:

$$E_{rc} = 0.0023 * S_0 \delta_r (T + 17.8) \quad (4.7)$$

where E_{rc} is the reference crop evapotranspiration (mm/d), S_0 is the water equivalent of radiation (mm/d) for the location, T is temperature (C), and δ^t is the difference between mean monthly maximum and mean monthly minimum temperatures. Actual ET is estimated using a soil moisture extraction function.

The function is defined by [Shuttleworth, 1993] as follows:

$$f(\theta) = \left(\frac{\theta - \theta_w}{\theta_f - \theta_w} \right) * \frac{\theta_d}{\theta_f} \quad (4.8)$$

where θ is the current soil water content, θ_w is the wilting point of the soil, θ_f is the field capacity of the soil and θ_d is used to define the function outlined in Figure 3.2 The value of potential evapotranspiration is then modified by the following equation to estimate actual evapotranspiration.

$$E = E_{rc} * f(\theta) \quad (4.9)$$

where E is the evaporation rate (m/d), E_{rc} is the reference crop evapotranspiration rate (m/d) and $f(\theta)$ is the soil moisture extraction function defined in Figure 3.2.

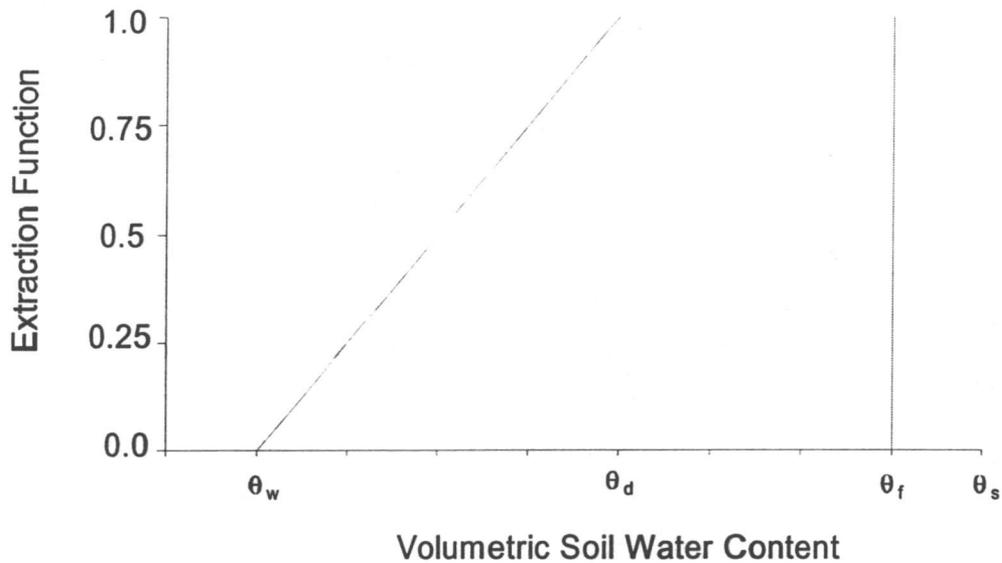


Figure 4.2. The soil moisture extraction function defining the evapotranspiration rate as a function of volumetric soil water content. The evapotranspiration rate is equal to the potential evapotranspiration rate until water content reaches θ_d , at which point the rate decreases linearly.

4.3.7 Instream Routing

The instream routing components of the model are presented in chapter 3, hence, here only an overview is provided. The downslope movement of in-channel water is assumed to proceed kinematically in one dimension. This assumption allows for significant simplification of the St. Venant equations, which describe one-dimensional fluid flow. From the simplified equations we develop an algebraic equation for discharge and solve the equation using known

boundary conditions and traversing the binary tree from the leaves towards the root.

4.4 Study Regions

Five separate watersheds representing three hydrologic regimes are presented in this paper. Sites were chosen in order to provide a rigorous test for a model designed to function at the watershed-scale, in areas where topography plays an important role in hydrologic functioning. We chose to focus on a variety of different locations in an effort to provide as broad a set of testing scenarios as might be expected to encounter. Invariably this broad focus reduces the ability of the study to provide detailed understanding of the hydrologic response of any one of the basins. But we feel this is consistent with the goal of presenting a conceptual model that provides an opportunity to understand how watershed-scale basins may respond to distributed restoration.

4.4.1 Wiley and Schaefer Creek Watershed, Oregon

Wiley and Schaefer Creeks are characteristic of the Eastern Willamette Valley. This area is dominated by a temperate marine climate with warm dry summers and colder wet winters, with approximately 80 percent of precipitation falling, on average, between October and May. Tertiary basalt and andesite in the

Cascades, transitioning to thick quaternary deposits in lower elevation areas, characterize basin geology. Soils are dominantly fine-textured silt and clay loams. The basins contain two type III ecoregions [Omernik, 1991], the Cascade and Willamette Valley.

Wiley Creek is a 600 sq km basin that is part of the South Santiam River watershed, a 3400 km² basin draining the western slope of the Cascade Mountains in Oregon. Landuse in the Wiley Creek basin is approximately 75% forested, 20% agricultural, and the remaining 5% in low-density urban development. Upper catchments tend towards steeply sloping forested topography, with headwaters in the Cascade Mountains. As is typical of basins in the Willamette Valley, lower portions of the catchment tend towards a more rolling topography, and the majority of the agricultural landuse are found within them. Long term discharge measurements have been collected in the basin the United States Geological Survey (USGS) since 1947 (USGS Station Number 141887000).

Schaefer Creek is a small (304 ha) tributary of Crabtree Creek, also in the South Santiam River watershed. Simulations for Schaefer Creek are included because of its combination of relatively small size and the availability of a long term daily stream discharge record (USGS Station Number 14188610).

4.4.2 The Maimai Watershed – Northern New Zealand.

The Maimai M8 watershed is located in Northern New Zealand and has been the site on ongoing hydrologic research since the 1970s. The area receives approximately 2600 mm of rainfall each year and runoff is closely coupled to rainfall, with total runoff values of approximately 1950 mm. A firmly compacted conglomerate sits below relatively complex soils classified as Blackball Hill soils, with mean depths of approximately 0.6 m. The forested watershed is steeply sloped, with a mean value of 34 degrees.

Simulations at Maimai provide a useful test for models of hydrologic process because of the variety of measured datasets and published descriptions of the watershed. These complete datasets provide an opportunity to evaluate model performance with respect to discharge with significant detail, and at the same time evaluate model performance using other datasets related to hydrologic process. Additionally, the site conforms closely to the one of the key assumptions defining hydrologic models that are sensitive to topographic gradients. That assumption is that topography is the dominant control on down slope flow. This is likely the case at Maimai because of the high rainfall rate, steeply sloping topography and relatively shallow soils with a well-defined impermeable horizon. Ongoing research in at Maimai has been recently reviewed by [McGlynn et al., 2002].

4.4.3 The San Jose Watershed – Northern Chile

The San Jose basin is located in the coastal mountain range of the 8th Region of Chile. It encompasses approximately 750 hectares of hilly terrain that has been intensively cultivated. Wheat is the primary crop, although pine and eucalyptus tree plantations are increasingly common. The climate in the area is Mediterranean with negligible snowfall. Streams are generally dry in the summer, while in the winter they respond in a flashy manner to large rainfall events, returning within hours to baseflow levels. The San Jose basin was instrumented in May of 2001 with stream gages at three locations, where discharge is being recorded every five minutes, and three tipping-bucket rain gages of 0.2 mm resolution. Soils in the region are characterized by very low permeabilities (1 – 1000 cm/day) and percolation to deeper aquifers is limited by shallow (2 -12 meters) granitic bedrock.

4.5 Results and Discussion

The overall goal of this research is the development and application of a flexibly distributed hydrologic model. Here we include both an analysis of model code and of more standard hydrologic model results.

4.5.1 Simulation Code

Wet_Hydro is an entirely new hydrologic modeling system, and while the concepts behind it are by design, quite simple, the code to implement these ideas is somewhat complex. Because the model is under development, we have included this section, which is designed to provide an analysis of the simulation environment. Programming oversights, faulty logic, and incorrect processing occur, and can be difficult to identify. The removal of both compile and runtime errors is the first step towards the development of error free code, but running without failure is not necessarily a good indication of the successful operation of model code because faulty, incomplete, or incorrectly coded logic can render a simulation ineffective.

A more useful indication, at least in the case of a model dealing with conservative quantities, is the explicit maintenance of a global mass balance. A correct mass balance is evidence of a variety of successes. In terms of WET_Hydro, three of these successes seem to stand out. (1) The model implements conversion routines to derive arrays representing detailed vector maps (refer to Appendix A.1 for details on this component of the study). The mass balance indicates that this process does not result in 'holes' in the watershed. (2) Units within the variety of algorithms are consistent and correct throughout time and space. (3) The tolerance threshold for error in the solution procedure does not result in significant numerical dispersion. Establishing these three elements is a

step towards code validation, although it is important to note that a mass balance does not provide any evidence that the model successfully describes the hydrologic functioning of the watershed. Its use here is to verify only that the code is consistent, and mass is conserved for any combination of model parameters and algorithms.

The mass balance, in the case of WET_Hydro is written as follows:

$$\sum_{i=0}^{i<upslopeCount} \left(\sum_{t=0}^{t<stopTime} (P_{t,i} - GW_{t,i} - ET_{t,i}) - D_t \right) + \left(\sum_{t=0,i=0}^{i<upslopeCount} SW_i - \sum_{t=n,i=0}^{i<upslopeCount} SW_i \right) + \left(\sum_{t=0,j=0}^{j<reachCount} RW_j - \sum_{t=n,j=0}^{j<reachCount} RW_j \right) + \left(\sum_{t=0,i=0}^{i<upslopeCount} PV_i - \sum_{t=n,i=0}^{i<upslopeCount} PV_i \right) = 0 \quad (4.10)$$

where t refers to time, i to upland unit, P is precipitation volume (m^3), GW is groundwater loss volume (m^3), ET is the evapotranspiration volume (m^3), PV is the ponded volume (m^3), D is the volume of water leaving the system as discharge (m^3), SW is the volume of water in the soil (m^3) and RW is the volume of water in the channel (m^3).

In figure 4.3, six separate simulations are presented, each with a progressively lower tolerance for error. During these simulations, rainfall and initial soil water content were the only model inputs. Stream discharge and final soil water content were the only model outputs. The statistic labeled “Volume Unaccounted” is a measure of the amount of input water that could not be explained by the summation of discharge and final soil water content. The two

series do not line up exactly because of storage within the landscape. The statistics do reflect this storage.

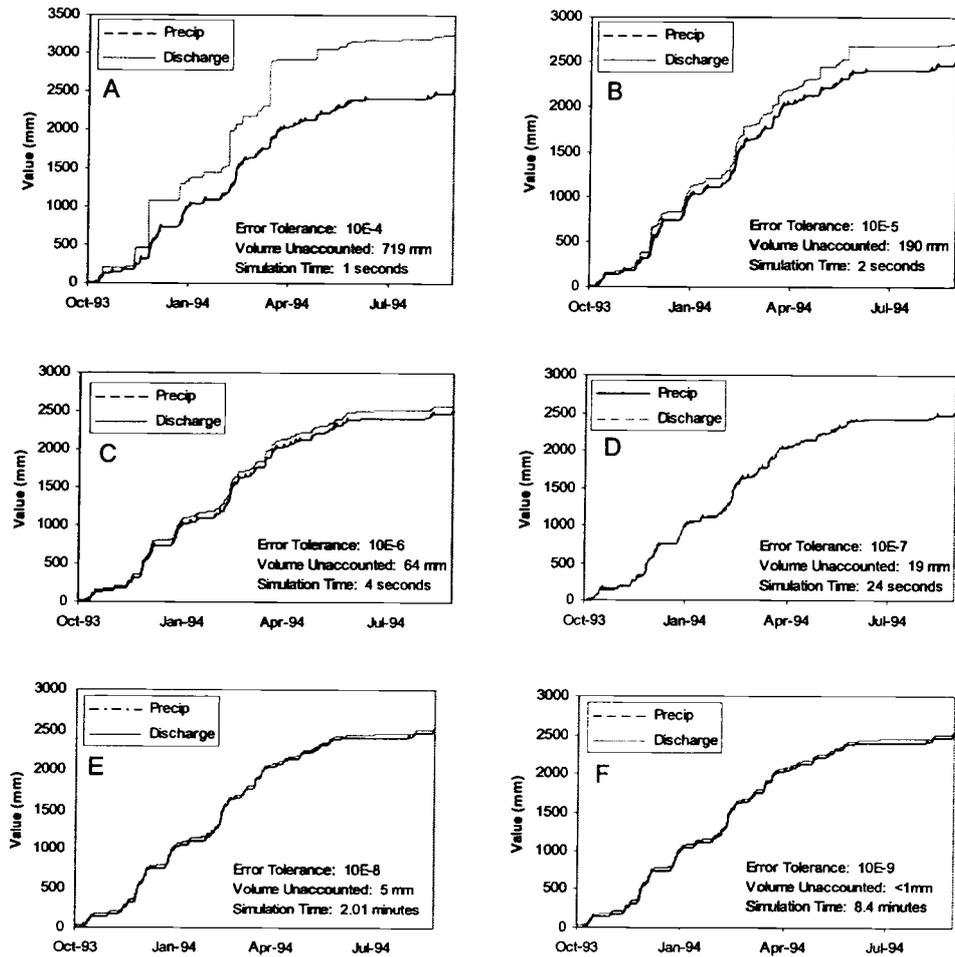


Figure 4.3. Results from six separate models runs at Schaefer Creek representing different numerical error tolerance

These data indicate that given large error tolerance (figures 4.3A-4.3C), the model functions poorly (but quickly – simulation times are between 1 and 4 seconds). These data suggest that an error tolerance of $\sim 10^{-6}$ (shown in figure 4.3D) is reasonable for these simulations. At this level, numerical error is reasonably small, and any further reductions (as shown in figure 4.2E and 4.2F) require considerably more simulation time. The mass balance provides a useful method of developing reasonable assumptions regarding the appropriate level of error tolerance during simulations. Because the error approaches zero as the error tolerance decreases, these results also clearly establish that numerical error, and not code errors, are responsible for the performance.

4.5.2 Model Calibration

Wet_Hydro employs upon a variety of parameters. In its simplest conception, and assuming that spatially-distributed parameters are single valued, the model has six parameters. (Table 4.1) This relatively small number of parameters is designed to provide the potential to develop some understanding of the uncertainty resulting from them. In an effort to understand how different combinations of those parameters affect model results we developed a Uniform Random Sampling (URS) approach. We use this approach to characterize parameter space and identify, where feasible, appropriate parameter sets. Clearly a wide variety of other more advanced approaches are available. We chose to

present these basic URS simulations for three reasons. (1) The development of algorithms is not dependent upon parameter array derivatives, (2) many realizations can be processed in a reasonably short amount of time, and (3) while the process is rather unintelligent, results provide a clear picture of the variety of results, both good and bad, that the code is capable of developing.

Table 4.1. WET_Hydro parameters and ranges.

| Param | Definition | Units | Range |
|------------------|--------------------------------|-------|---------------|
| m | Scaling Parameter | M | 0.1-20 |
| Init Sat | Initial Saturation | - | 0.7-1.0* |
| K _{Eff} | Rate of movement at saturation | M/d | 0.001-5 |
| n | Manning's n | - | 0.01-0.15 |
| phi | Soil storage capacity | - | 0.25-0.35 |
| kDepth | Loss rate to groundwater | M/d | 0.00001-0.001 |

One of the first decisions when measured and modeled data are compared is the choice of goodness-of-fit measures. We present the results using four different likelihood functions. Each of these measures focuses on a somewhat different aspect of the fit between modeled and measured data. We present multiple criteria in an effort to provide a more complete characterization of model functionality. The test statistics are the root mean square error (RMSE):

$$RMSE = \sqrt{\frac{1}{n} \sum_{t=0}^{t=n} (d_t - o_t(\theta))^2} \quad (4.11)$$

the coefficient of efficiency (Nash-Sutcliffe efficiency criterion)

$$R_{eff} = 1 - \frac{\sum_{t=0}^{t=n} (d_t - o_t(\theta))^2}{\sum_{t=0}^{t=n} (d_t - \bar{d})^2} \quad (4.12)$$

Willmott's Index of Agreement:

$$D = 1 - \frac{\sum_{t=0}^{t=n} (d_t - o_t(\theta))^2}{\sum_{t=0}^{t=n} (|o_t(\theta) - \bar{d}| + |d_t - \bar{d}|)^2} \quad (4.13)$$

and the coefficient of determination (R^2) test statistic:

$$R^2 = \left(\frac{\sum_{t=0}^{t=n} (d_t - \bar{d})(o_t(\theta) - \bar{o}_t(\theta))}{\left[\sum_{t=0}^{t=n} (d_t - \bar{d}) \right]^{0.5} \left[\sum_{t=0}^{t=n} (o_t(\theta) - \bar{o}_t(\theta)) \right]^{0.5}} \right)^2 \quad (4.14)$$

where n is the number of observations, t is time, d is the measured value of discharge, $o_t(\theta)$ is the modeled value of discharge, and $\bar{o}_t(\theta)$ is the mean value of the modeled data, given the parameter array θ , and \bar{d} is the average measured runoff over the observed time period.

The index of agreement [Willmott, 1981] and coefficient of determination range from 0.0 to 1.0, with higher values indicating stronger agreement between the modeled and measured data (closer correspondence). We present the coefficient of determination because it is generally familiar, but offer the index of agreement as a more useful alternative. The utility of the R^2 is limited because it

measures only colinearity. High values indicate similar patterns through time, but do not necessarily indicate similar magnitudes. The index of agreement maintains the familiar 0 to 1 range, but has been shown to better reflect actual differences.

Units of the root mean square are equivalent to the data and so reporting the statistic along with time series data can be quite useful. And the last statistic, the coefficient of efficiency [Nash and Sutcliffe, 1970], is most commonly cited and used in studies of hydrologic response [Loague and Freeze, 1985; Legates and McCabe Jr., 1999; Leavesley et al., 2002]. Values of the statistic range from 1.0 to minus infinity. A value of 1.0 indicates exact agreement and 0.0 indicates that the observed mean is as good a predictor as the model. Because it is normalized by the measurements, it can be used to quickly compare results from different time series in different basins. The statistic relies on least squares techniques, and so weights peak flow values more heavily than low flows. In this study we consider a number of different basins, and for this reason we use the coefficient of efficiency more frequently than other statistics. All scatter plots are provided with the Nash-Sutcliffe (R_{eff}) statistic as the Y-axis. We report the other measures only for simulations using a single parameter set, where time series simulation results are also included.

Here we use Monte Carlo simulation at a relatively simple level. Other researchers have begun to focus more intently on how results such as these can be extended to provide more formal analyses of models and model acceptability [Duan et al., 1992; Freer et al., 1996; Gupta et al., 1998; Boyle et al., 2000;

Madsen, 2000; Bates and Campbell, 2001; Aronica et al., 2002]. Included in these more formal analyses are the Generalized Linear Uncertainty Estimation (GLUE) [Romanowicz et al., 1994] procedure which formalizes the development of statistical definitions of confidence from the Monte Carlo sample space and the MOCOM-UA [Yapo et al., 1998] procedures also extend which these kinds of Monte Carlo analyses and use other likelihood functions (some of which are defined in Section 4.2.4.1) in formal multi-criteria analysis of model functionality. Recent work by [Seibert and McDonnell, 2002] has introduced methods which, beginning with Monte Carlo simulation, allow for the incorporation of 'soft' data into model identification and selection. In sections 4.4 –4.5 we introduce alternative criteria used with Monte Carlo simulation to further constrain the set of acceptable models.

The calibration procedures used in the study were standardized as follows:

- Select a small number of events from each of the four basins and for each of those eight events perform a detailed Monte Carlo analysis.
- Use the results of the Monte Carlo simulations to provide an idea of overall parameter identifiability and also to determine maximum Nash-Sutcliffe statistics. A Nash-Sutcliffe efficiency of 0.0 is used as the lower cutoff in each of the Monte Carlo scatter plots (figures 4.2 to 4.8).

- Use the parameter sets responsible for higher error statistics in model verification procedures over seasonal time frames.

Table 4.2. A listing of all calibration events.

| Catchment | ID | Begin Date | End Date | Input |
|-----------|----|------------|----------|--------|
| Wiley | a | 02/15/94 | 03/15/94 | Daily |
| Schaefer | c | 02/15/94 | 03/15/94 | Daily |
| Maimai | e | 09/02/87 | 09/05/87 | 20 min |
| Maimai | f | 10/23/87 | 10/26/87 | 20 min |
| San Jose | g | 5/26/01 | 5/28/01 | 15 min |
| San Jose | h | 5/28/01 | 5/30/01 | 15 min |

The Maimai watershed is steep and underlain by shallow and essentially impermeable bedrock. These characteristics tend to favor strong topographic controls on watershed hydrology. Models, like TOPMODEL and WET_Hydro, which rely on topographic gradient as a key driver of down slope flow, would be expected to function effectively under these conditions. In addition, the area is wet, highly responsive to rainfall, and the key hydrologic inputs and outputs are well documented. We focus on the period from September to December of 1987, using 20 minute rainfall, evapotranspiration, and discharge measurements.

The most notable aspect of the Monte Carlo simulations at Maimai is the parameter space similarity between events. M and initial saturation are relatively well constrained and indicate regions over which measured and modeled hydrographs are more closely linked. The sampling provides very limited utility in selecting values for k_{eff} , ϕ , k_{Depth} , or n , indicating rather than maximum

model efficiencies are, under these conditions, sensitive only to initial conditions and m . It is likely that the inclusion of other criteria, either different periods of time during the hydrograph or entirely different data, might be used to further constrain these parameters. Also it is worthy to note that maximum values of efficiency are relatively high and certainly within the realm of other 'acceptable' simulations.

The hydrology in the San Jose watershed differs considerably from that at Maimai and in the Willamette Valley. Here the low conductivities and short intense rain events produce a flashy hydrologic system dominated by Hortonian overland flow. The model does not currently simulate directly this infiltration excess overland flow, but we selected this watershed to more fully examine the saturated overland flow components of the model. Two considerably different events were selected for Monte Carlo simulation. The first was the initial runoff event for the winter 2001 wet season, the duration of which was approximately a day and a half. The second occurred approximately 12 hours after and lasted for approximately the same time.

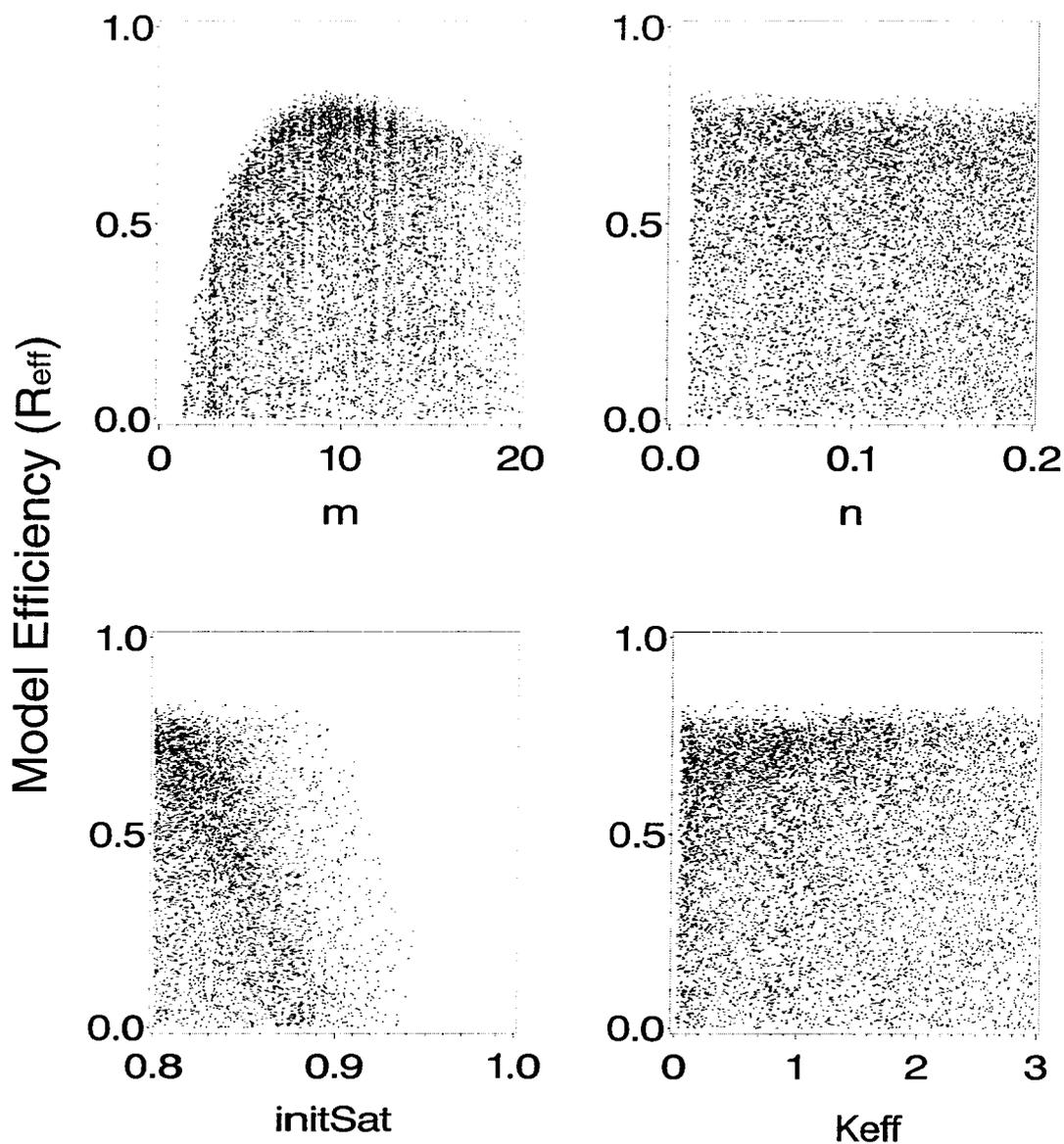


Figure 4.4. Scatter plot of model efficiency versus parameter value for four model parameters in the Maimai watershed. The event over which these efficiencies were developed began on September 2, 1987 and ended on September 5, 1987.

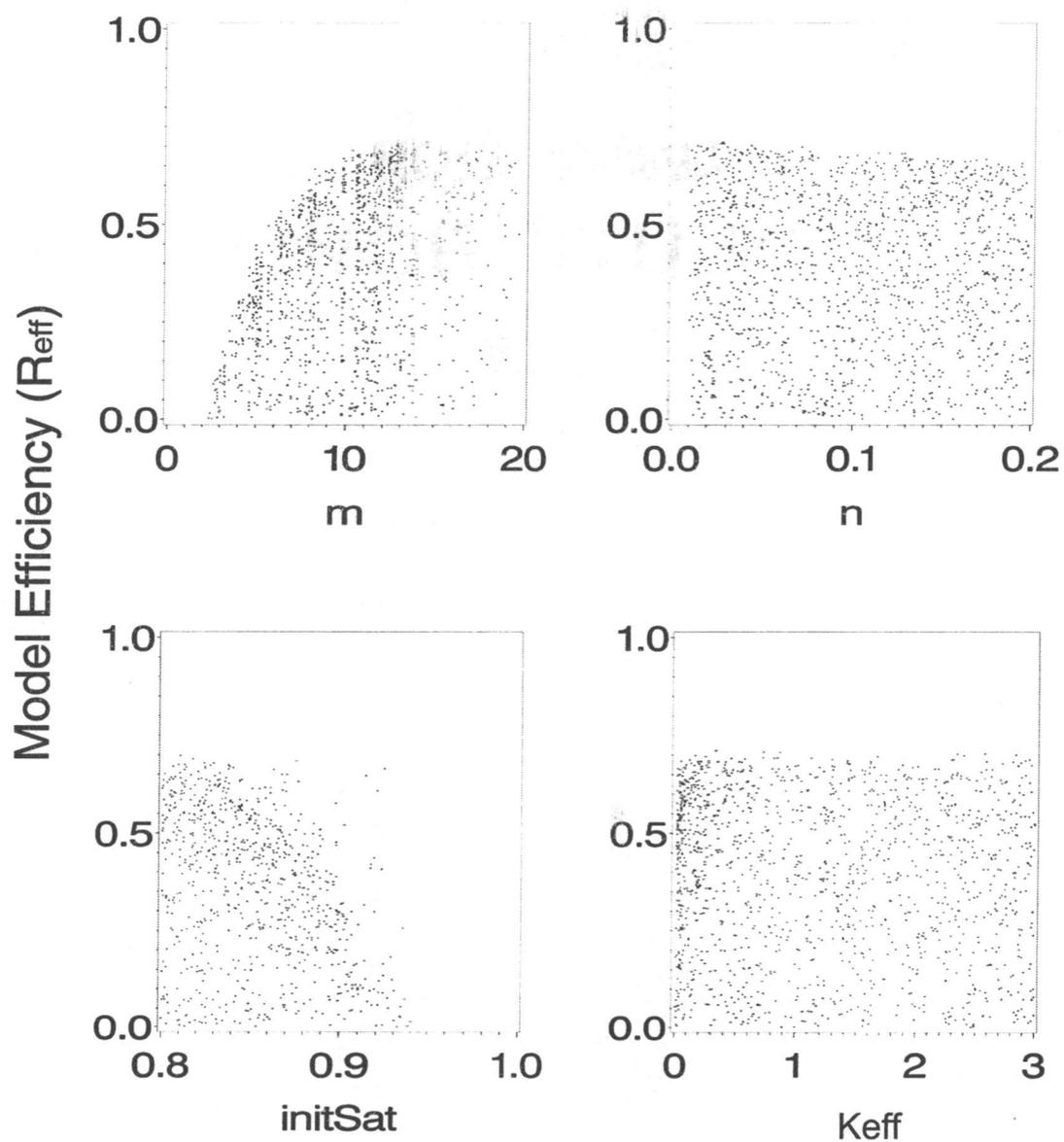


Figure 4.5. Scatter plot of model efficiency versus parameter value for four model parameters in the Maimai watershed. The event over which these efficiencies were developed began on October 23, 1987 and ended on October 26, 1987.

Three nested stream flow data series were available and we calculated a weighted model efficiency which includes each of these data points. Maximum model efficiencies for the first event are the lowest of all of the four regions (~ 0.50), indicating a relatively poor performance at the San Jose for the event. In addition, the lower density of points when compared with the Maimai suggest that the model is more capable of the production of non-behavioral results within feasible parameter space at San Jose than at the other three basins. While efficiencies do increase over the second event (up to ~ 0.70), the values are lower than maxima found at other sites. This result suggests that model structure (and not simply parameters) is less adequate for simulations at San Jose than for other areas. We anticipate that the inclusion of algorithms corresponding to dominant flow pathways (Hortonian overland flow, routing of overland flow, etc) has potential to increase maximum efficiencies. We pursue these results further in section 4.4.4, using additional criteria to more completely character model efficiency in the San Jose basin.

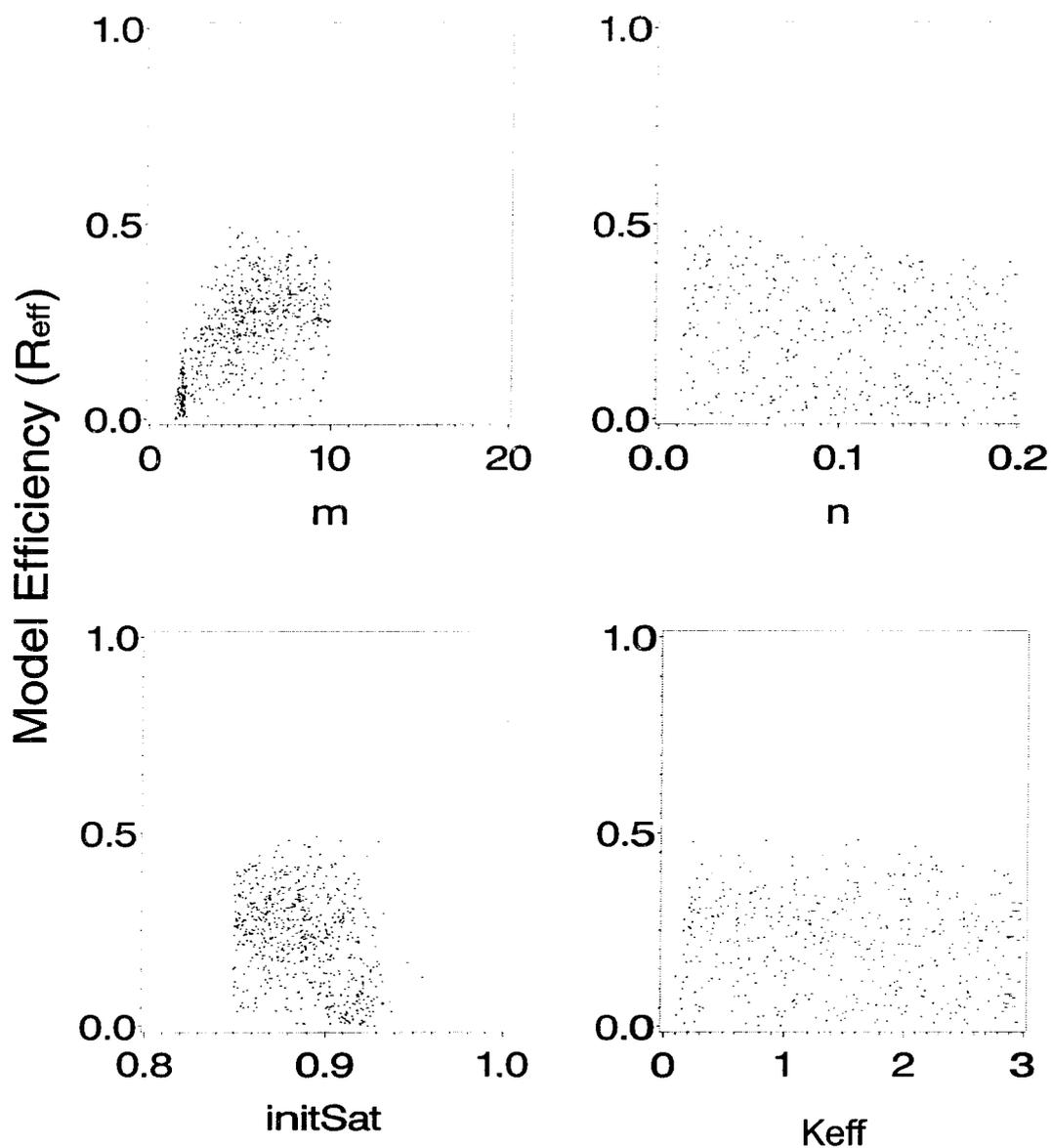


Figure 4.6. Scatter plot of model efficiency versus parameter value for four model parameters in the San Jose watershed. The event over which these efficiencies were developed began on May 25, 2001 and ended on May 28, 2001.

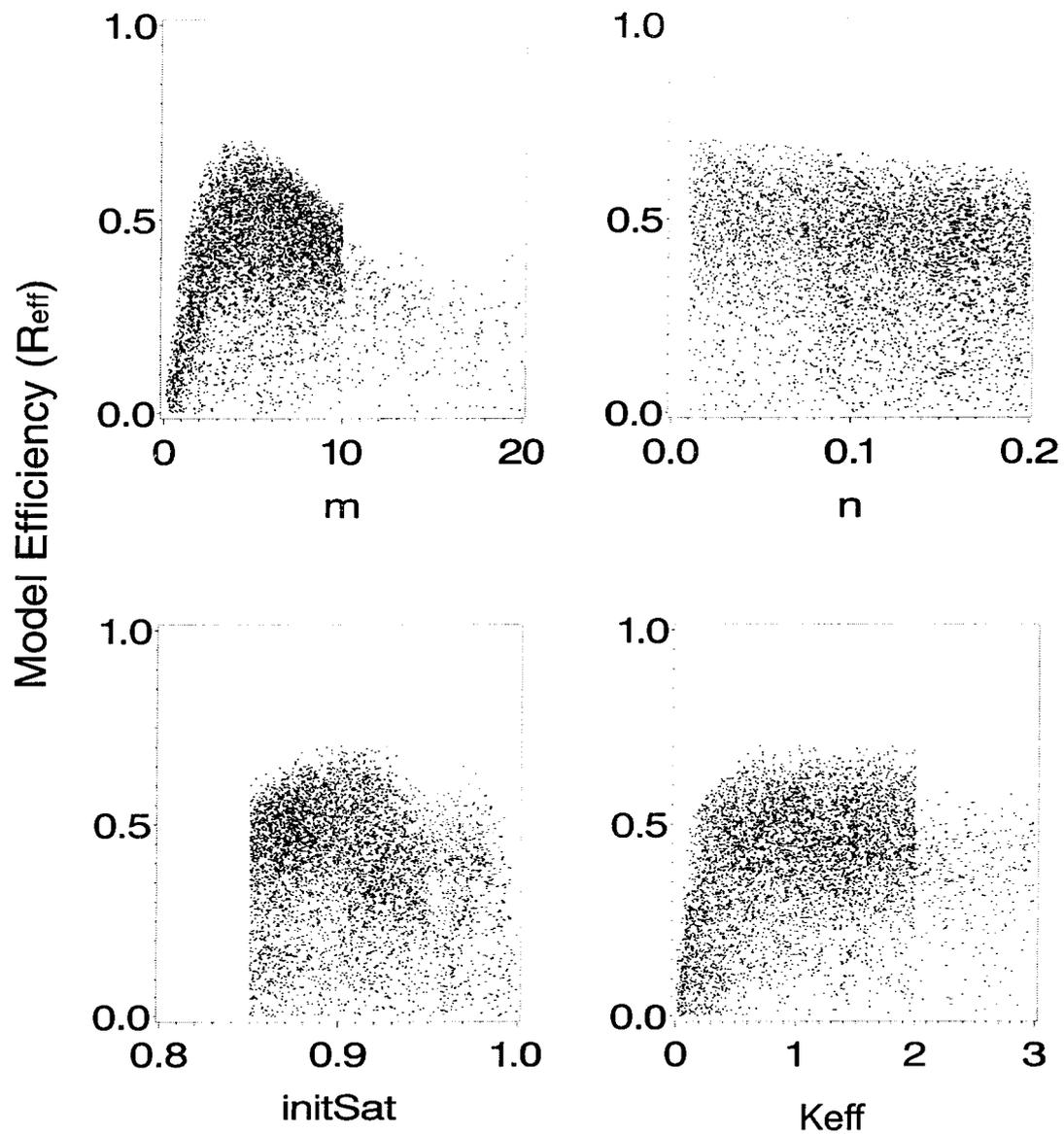


Figure 4.7. Scatter plot of model efficiency versus parameter value for four model parameters in the San Jose watershed. The event over which these efficiencies were developed began on May 28, 2001 and ended on May 30, 2001.

For each of the two watersheds in the Willamette Valley, a single Monte Carlo sampling was developed. For the time period from February 15, 1993 to March 15, 25000 realizations were generated. The model was run at sub daily time steps, but efficiency was calculated for average daily values, corresponding to the measured values.

Results from Schaefer Creek indicate relatively high efficiencies of over 0.80, but that those high values can be found across a wide range of feasible parameter space. As with most of the scatter plots, these results are indicative of the lack of parameter identifiability that has been noted by various authors evaluating conceptual models. This result strongly suggests that the use of single output calibrations for distributed models results in highly uncertain parameter choices.

Similar results are found at Wiley Creek, most notably the decrease in efficiency between for initial saturation values of between 0.90 and 0.97. The parameter m , which defines the decrease in conductivity with depth, displays an identifiable maximum at about 12.0 and we used this value in subsequent verification runs.

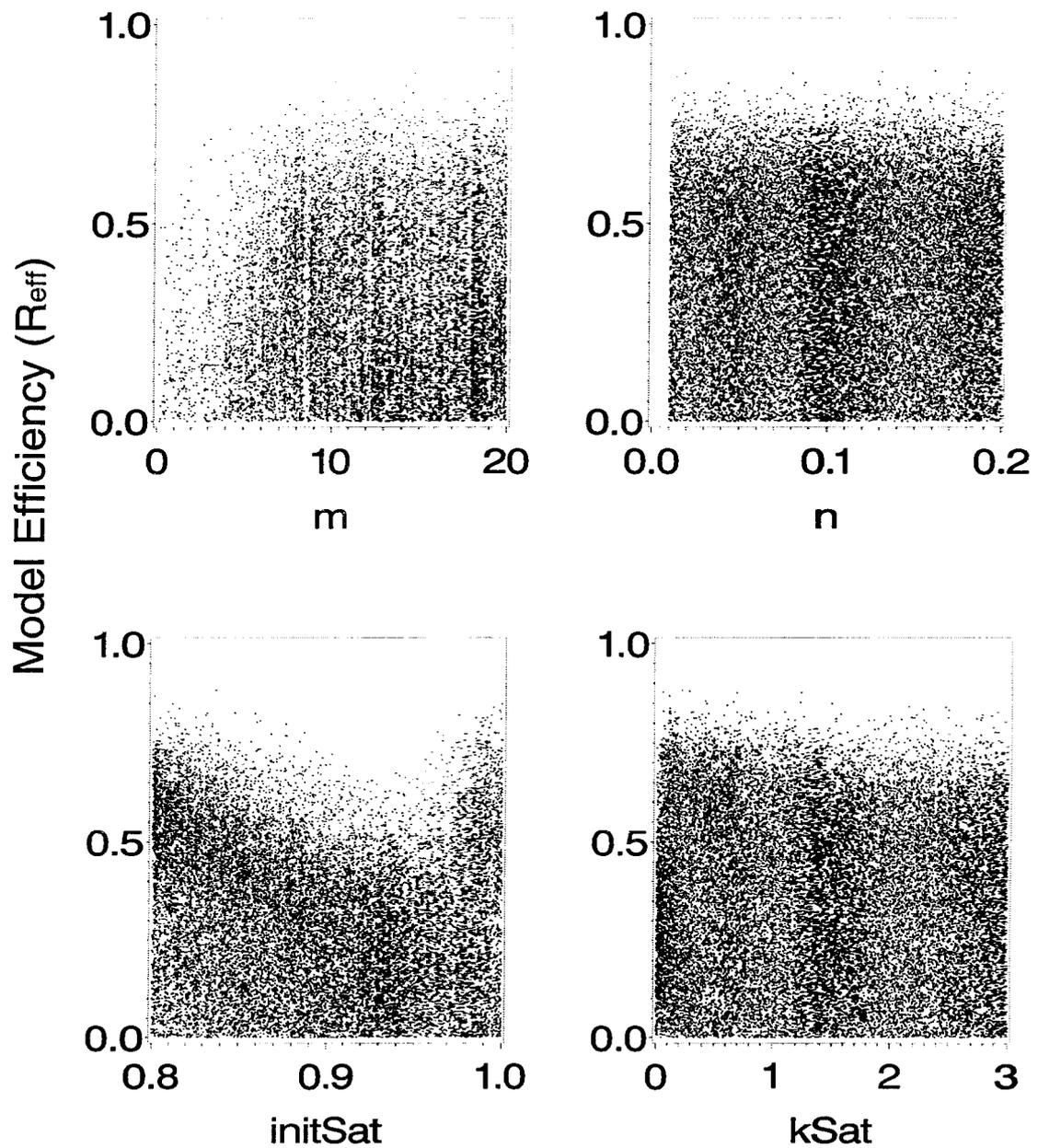


Figure 4.8. Scatter plot of model efficiency versus parameter value for four model parameters in the Schaefer watershed. The event over which these efficiencies were developed began on February 15, 1993 and ended on March 15, 1993.

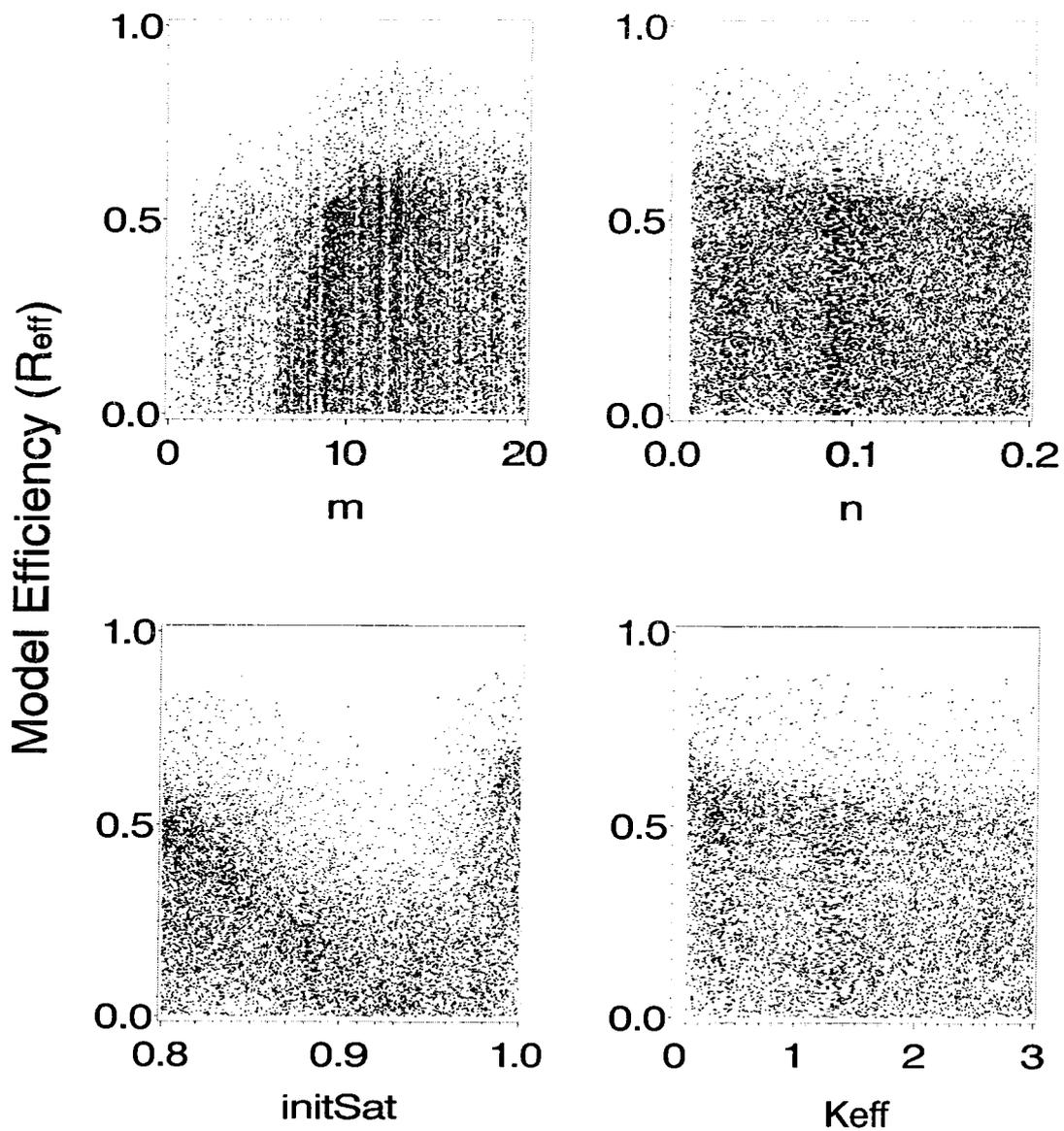


Figure 4.9. Scatter plot of model efficiency versus parameter value for four model parameters in the Wiley watershed. The event over which these efficiencies were developed began on February 15, 1993 and ended on March 15, 1993.

4.5.3 Model Verification

The process of verification involves the application of the model using the parameter set representing the highest efficiency from the Monte Carlo procedures. The events chosen for verification are outlined in table 4.3. For the short term verification events, adjustments for initial soil saturation were allowed because the value clearly changes through time. In all cases, we calculated efficiencies at the same frequency as in the calibration periods (either hours or days). Clearly, the selection of a single parameter set does tend to exclude much of the information derived by the Monte Carlo sampling, most notably the fact that many different parameter sets provide essentially equivalent model efficiencies [Beven, 2001]. But our goal is to indicate how a successful parameter set derived over a relatively short event performs over other short term events and also seasons. That other parameter sets also produce similar results is important to understand, but should not preclude further model analyses. Figures 4.10 - 4.17 provide time series results of a number of modeled outputs. These include rainfall inputs, measured and modeled discharge, volumetric soil water content, the model mass balance and the model time step. The water content represents, in most cases, a single spatial unit and provides a qualitative understanding of upland model output. It corresponds to equation 4.4.

Table 4.3. An overview of the verification events

| Catchment | Ident | Begin Date | End Date | Input |
|-----------|-------|------------|----------|--------|
| Wiley | 1 | 10/1/93 | 10/1/94 | Daily |
| | 2 | 10/1/95 | 10/1/96 | Daily |
| Schaefer | 3 | 10/1/93 | 10/1/94 | Daily |
| | 4 | 10/1/95 | 10/1/96 | Daily |
| Maimai | 5 | 10/28/87 | 12/1/87 | 20 min |
| | 6 | 09/02/97 | 12/31/97 | 20 min |
| San Jose | 7 | 7/7/01 | 7/9/01 | 15 min |
| | 8 | 5/26/01 | 9/01/01 | 15 min |

Table 4.4. Results from each of the verification events outlined in table 4.3. The four likelihood statistics correspond to those outlined in Equations 4.11 through 4.14.

| Basin | Parameters | | | | | | likelihood | | | | |
|----------|------------|------|------|------|-----|--------|------------------|----------------|------|-------|--|
| | IS | m | n | kEff | phi | kdepth | R _{eff} | R ² | D | RMSE | |
| Wiley | 0.77 | 12.6 | 0.10 | 1.7 | 0.3 | 0.0001 | 0.45 | 0.71 | 0.85 | 3.926 | |
| Wiley | 0.77 | 12.6 | 0.10 | 1.7 | 0.3 | 0.0001 | 0.61 | 0.64 | 0.89 | 9.331 | |
| Schaefer | 0.77 | 19.5 | 0.13 | 0.28 | 0.3 | 0.0001 | 0.15 | 0.56 | 0.85 | 0.248 | |
| Schaefer | 0.77 | 19.5 | 0.13 | 0.28 | 0.3 | 0.0001 | 0.35 | 0.55 | 0.83 | 0.463 | |
| Maimai | 0.97 | 9.8 | 0.01 | 1.1 | 0.3 | 0.0001 | 0.89 | 0.91 | 0.97 | 0.002 | |
| Maimai | 0.97 | 9.8 | 0.01 | 1.1 | 0.3 | 0.0001 | 0.89 | 0.96 | 0.98 | 0.002 | |
| San Jose | 0.95 | 3.9 | 0.02 | 0.6 | 0.3 | 0.0001 | 0.14 | 0.41 | 0.76 | 0.198 | |
| San Jose | 0.92 | 3.9 | 0.02 | 0.6 | 0.3 | 0.0001 | 0.57 | 0.58 | 0.86 | 0.521 | |

Highest model efficiencies ($R_{\text{eff}} = 0.89$) occur at Maimai, where there are few of the complications that occur in other watersheds (such as relatively poor inputs, very large and unmeasured heterogeneities in soils, meteorology, and anthropogenic influences, as well as overland flow, and the need for streamflow

routing). Despite these reasonably clear explanations for the higher likelihood values, we feel that demonstration of high model efficiencies at this site is consistent with the assumption that the model development, and the strategy for representing temporal and spatial aspects do not result in significant errors. This leads us to infer that the lower model efficiencies seen in other regions are likely due to inadequate input and oversimplification of the dominant hydrologic processes, both of which might be improved upon with further research.

A fully-distributed model of the Maimai catchment was implemented and the two data series included in the soil water content figure (figure 4.10c) represent a near stream model element and an upland model element, as noted in figure 4.10c. Soil water content tends to be higher in riparian areas than in upland areas, a result that could not be achieved with a semi-distributed solution. Also note that soil water in the two regions tends to increase and decrease in parallel. Seibert et al., [2002] showed that upland areas tend to show more of a disconnection to stream flow than near stream areas. Figure 4.10 appears to suggest that the model does not capture these dynamics, and may point to a limitation in the model structure. At the same time, the result is in part due to the lumped parameterization pursued throughout this paper. In cases where more information is utilized to describe the spatially-distributed nature of parameters (most notably those defining transmissivity dependence on water content) the apparent steady-state drainage rates may disappear.

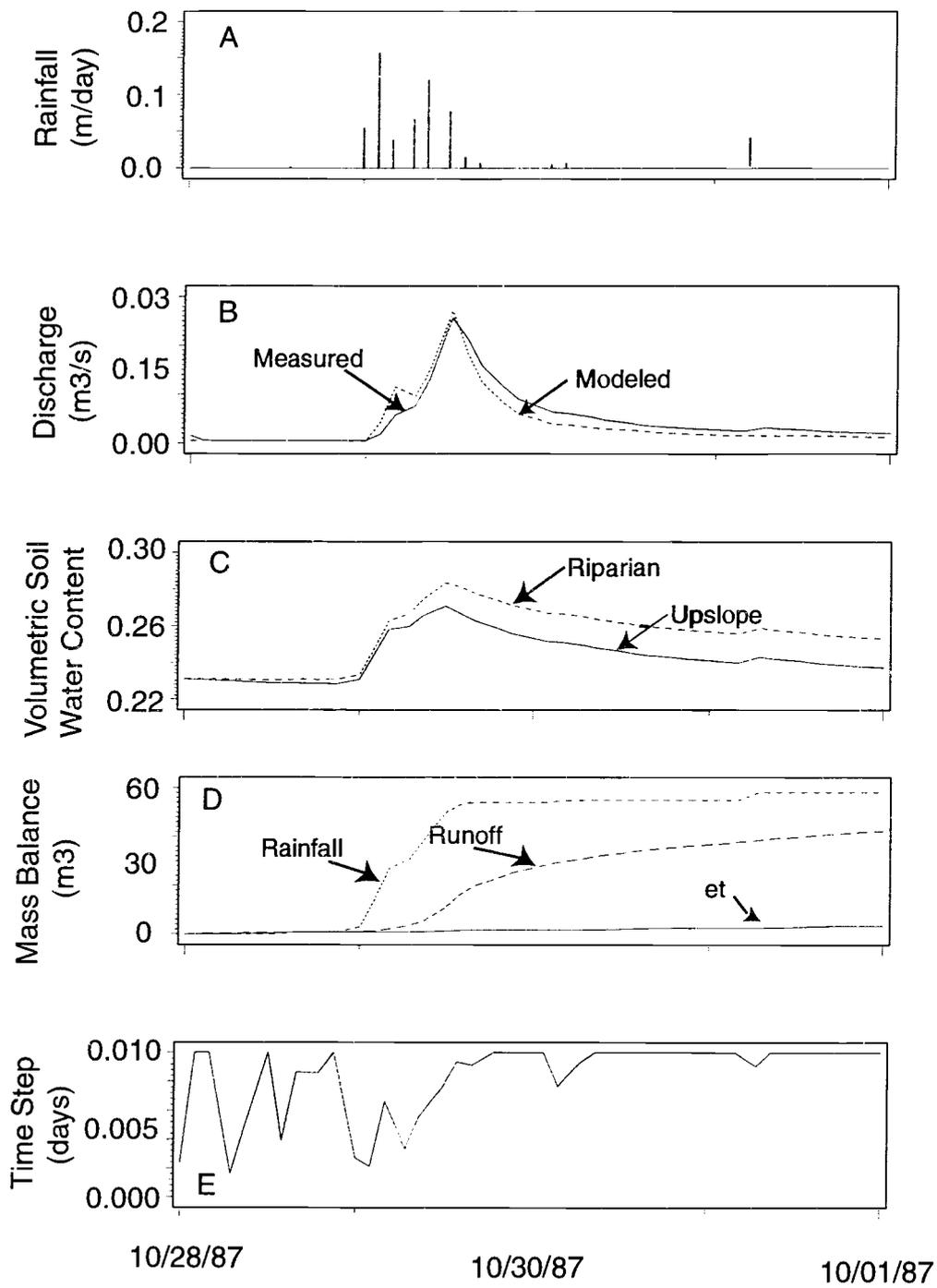


Figure 4.10. Verification results from Maimai from October 28 through November 1, 1987. All results are simulated except for the series labeled 'Measured' in part B.

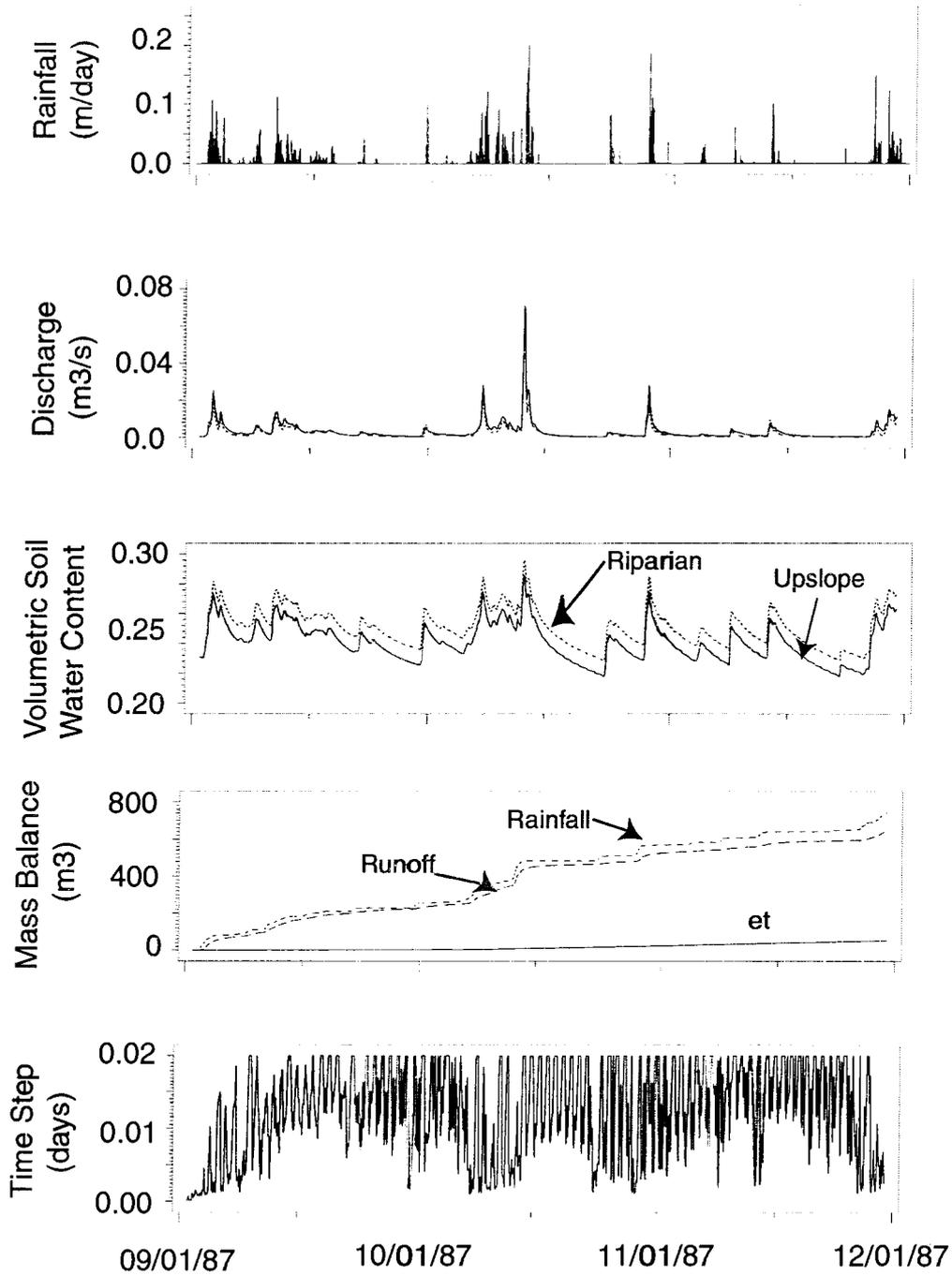


Figure 4.11. Verification results from Maimai from September 1 through December 1, 1987. All results are simulated except for the series labeled 'Measured' in part B. The maximum time step was larger for this simulation than presented in 4.10 to decrease model simulation time over the longer 3 month time period. Note that the time period over which figure 4.10 was developed is included as a small portion of this figure.

Verification results from the San Jose Basin indicate a decrease in model performance as compared with Maimai. The short event modeled in figure 4.12 appears to demonstrate the model focus on soil water movement, as opposed to overland flow. The modeled discharge increased, along with measurements, commencing approximately 4 hours after the onset of rainfall, however the modeled rates of discharge change much more gradually than the measured values. The model does not correctly simulate the flashiness of the measurements, and the low value of R_{eff} (0.14) reflects this. Results for the longer term simulation improve, and this appears to result in part from the longer discharge 'tails' associated with larger storms, which correspond more closely to the exponential decreases in conductivity assumed by the model. (figure 4.13)

The wet season measured hydrograph appears to indicate a complex relationship between watershed storage, matrix discharge and overland flow. Large rain events occurring after long dry periods do not translate into stream discharge and appear to be completely stored, allowing for significant runoff from subsequent events. In most cases, the model produces lower peak discharges, most likely due to a relatively low model influence of overland flow. With different parameter choices, the influence of overland flow can be more accurately captured, but Monte Carlo simulations indicate that the chosen parameters produced 'best' results for two short storms in May and June of 2001.

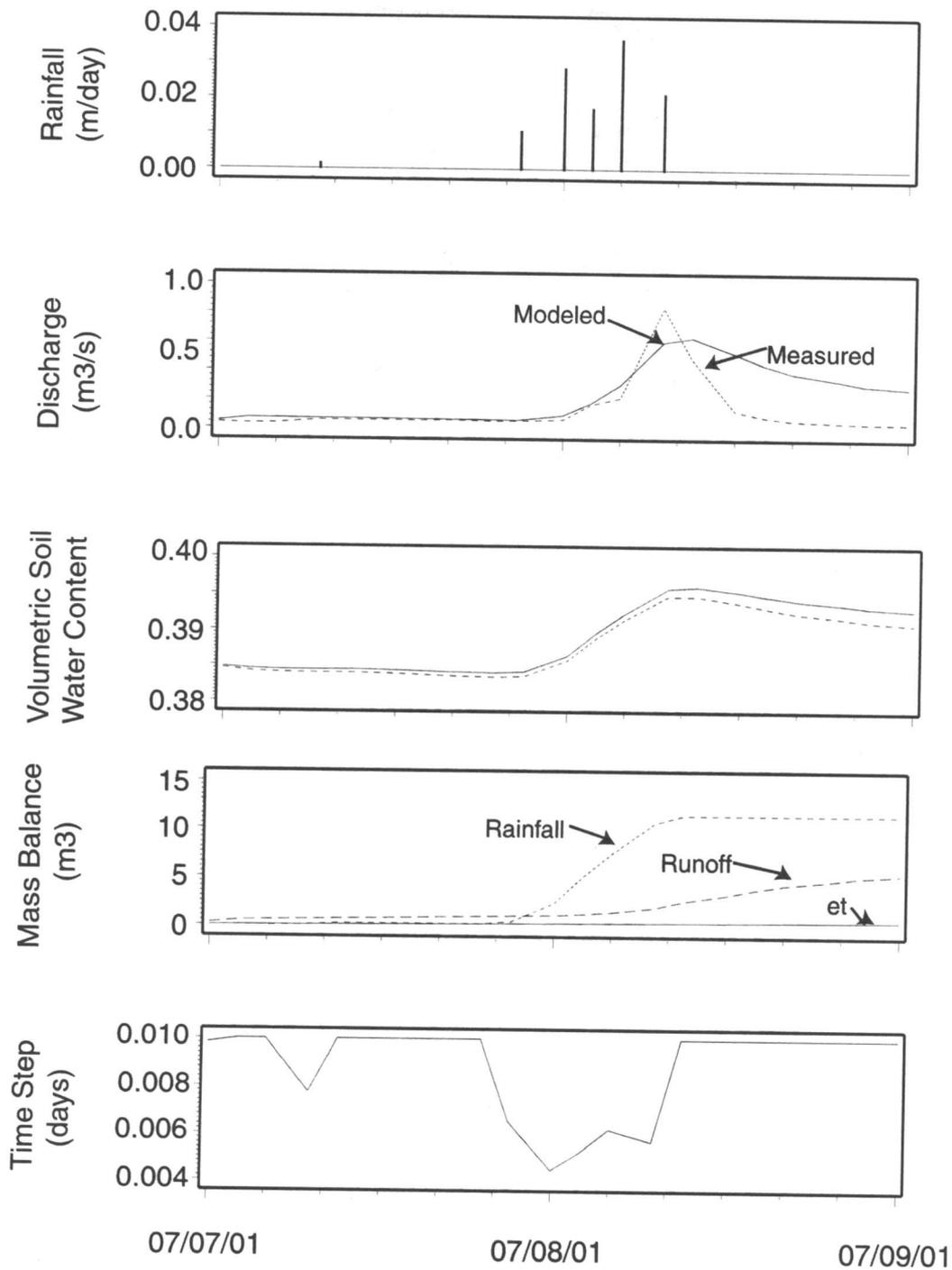


Figure 4.12. Verification results from San Jose from July 7 through July 9, 2001. All results are simulated except for the series labeled 'Measured' in part B.

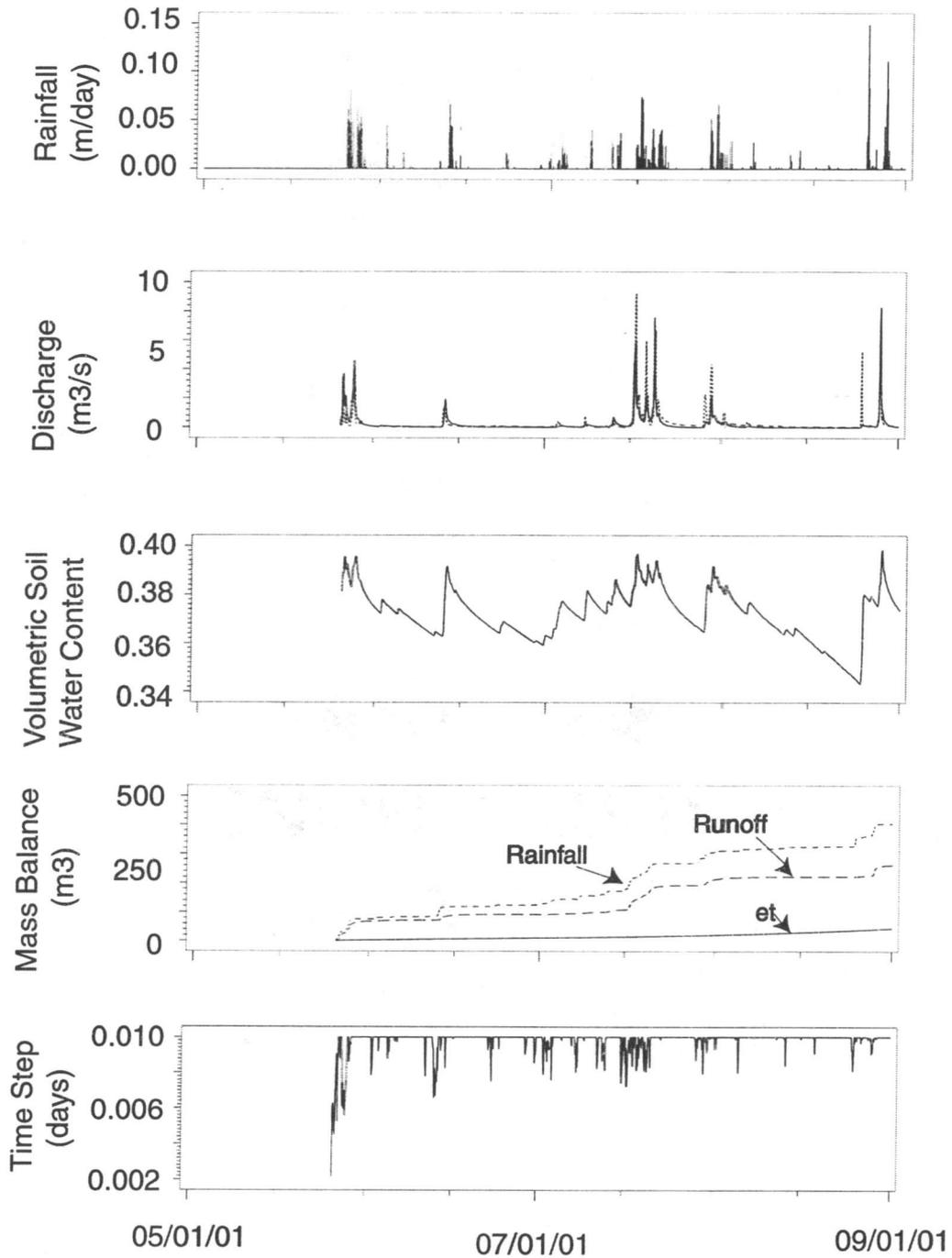


Figure 4.13. Verification results from San Jose from May 22, 2001 through September 1, 2001. All results are simulated except for the series labeled 'Measured' in part B. The small event outlined in Figure 4.12 is a small portion of the simulation outlined in this figure.

In section 4.4.4 we explore in more detail the relationship between efficiency, parameterization and model functioning, but we finish this section by addressing the degree to which the model has been 'verified' for the San Jose. The basin is larger than Maimai, and clearly does not correspond nearly as well to the assumptions about hydrology implemented in WET_Hydro, so lower efficiencies were to some degree expected. Efficiency over the first event suggests the model is only a marginally better predictor of the hydrograph than the observed mean, and this value ($R_{\text{eff}} = 0.14$) essentially constitutes a rejection of the model for this event. But, over the longer event, efficiency increases significantly, although the relatively low value (0.56) still suggests the need for further evaluation of model structures, with the inclusion of overland flow routing and Hortonian overland flow generation a seemingly good starting point.

Two model runs, corresponding to water years 1994 and 1996, demonstrate model application to the basins in the Willamette Valley (figures 4.14 through 4.17). Year long events were chosen to correspond with the average daily values used as model input. Focusing on discharge, where measured and modeled values exist, along with error statistics (Table 4.4), notice a relatively low efficiency (0.15 to 0.61) in both basins, but that values of R^2 are above 0.55, suggesting the patterns in discharge are generally correct, but that magnitudes differ.

Results from Schaefer Creek tend to suggest poorer model performance than in Wiley Creek. In most cases it appears that R_{eff} is strongly affected by

periods of time where the model estimates storm discharge that the measurements simply do not indicate. Note that in each of the four hydrographs (figures 4.14 through 4.17), measurements are generally under predicted. The model does not currently implement a snowmelt/accumulation algorithm, and these differences are most likely due to snowfall accumulation. In addition, the Schaefer basin is 10.2 kilometers from, and 0.9 kilometers above the nearest meteorological station. (The station is located within the Wiley Creek Basin). To more successfully apply the model to the Schaefer Creek basin, more precise inputs and a snowmelt algorithm appear necessary. Simulation at Schaefer for WY1996 are better than for 1993, quite possibly due to the higher winter temperatures and corresponding decrease in snowfall accumulations. The average winter temperatures measured at Foster Reservoir were 48.24 degrees C in WY 1994 and 51.79 degrees C in WY 1996

Model efficiencies are higher in Wiley Creek, a likely reason being the much larger basin area, and greater elevation range. Approximately forty eight percent of Wiley lies below 600 meters elevation (approximately 2000 ft), while 100% of Schaefer lies above 880 meters. Because of the larger percentage of lowland, snowfree areas, snowfall accumulations should have a lesser effect on Wiley Creek discharges. In addition, our meteorological inputs were collected within the Wiley Creek basin, a fact that should also produce higher model efficiencies. As with Schaefer Creek, model efficiencies for the 1996 period at Wiley are higher than in 1994, potentially corresponding again to warmer average temperatures.

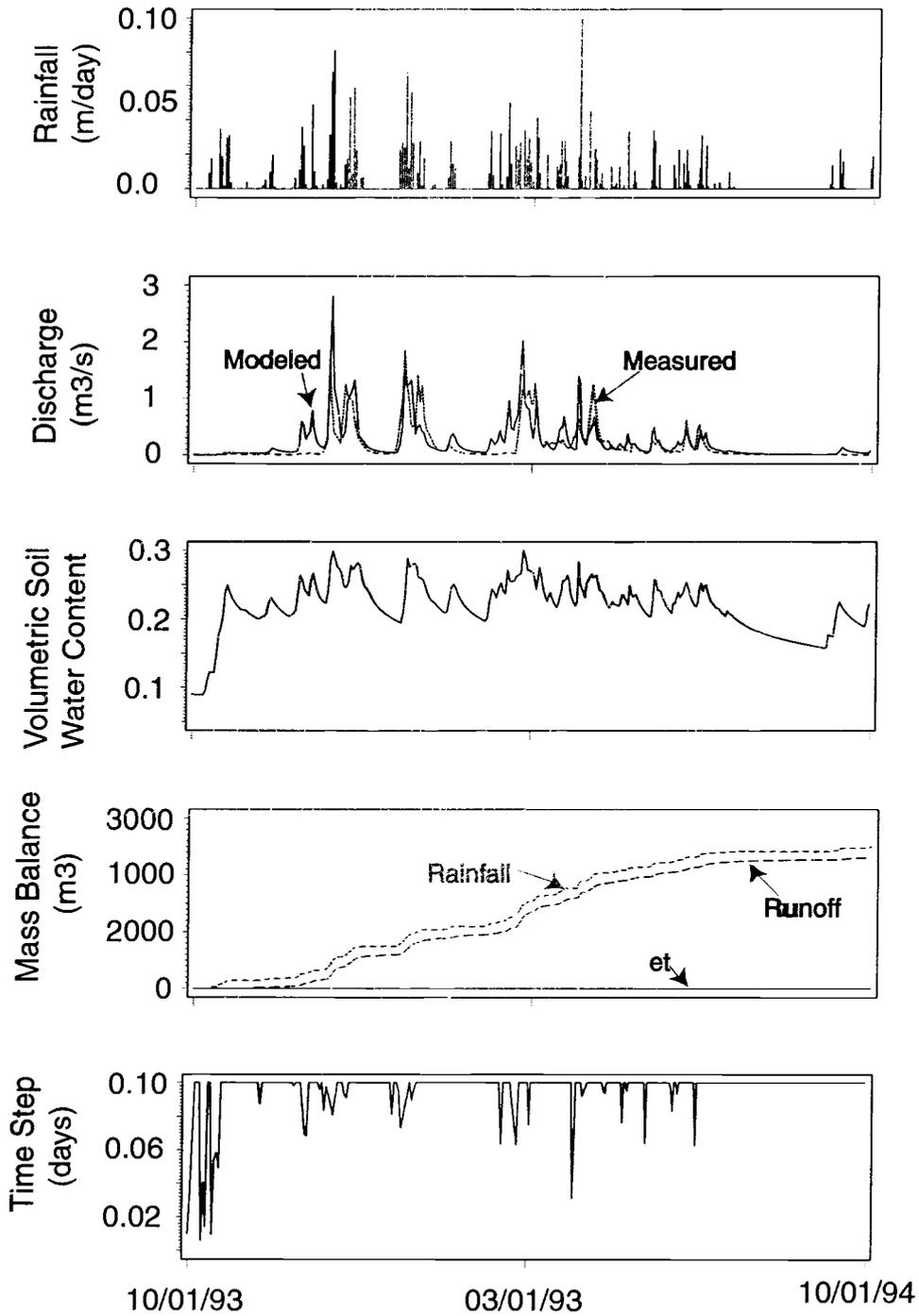


Figure 4.14. Verification results from Schaefer Creek for the 1993 water year. All results are simulated except for the series labeled 'Measured' in part B.

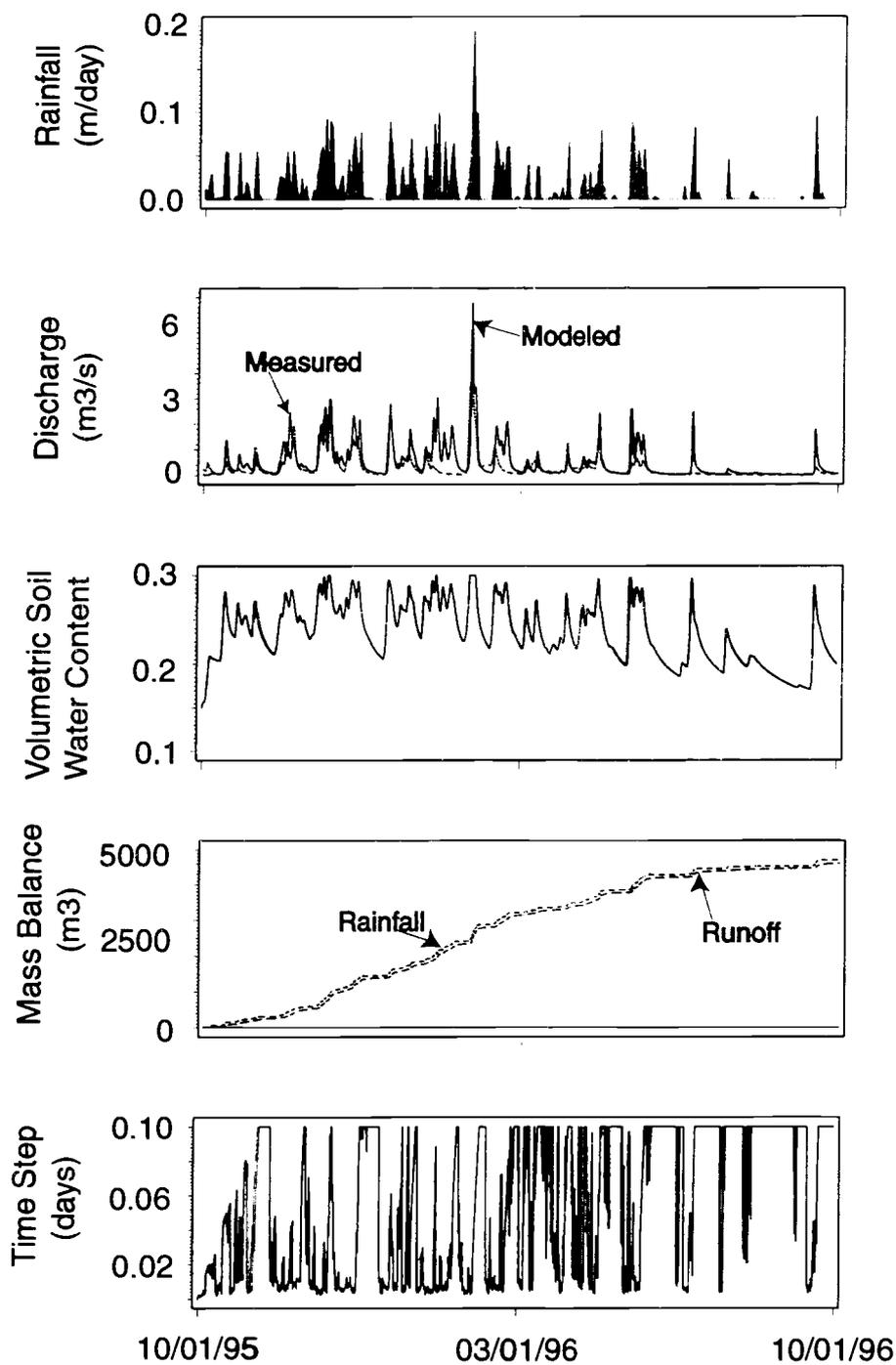


Figure 4.15. Verification results from Schaefer Creek for the 1993 water year. All results are simulated except for the series labeled 'Measured' in part B.

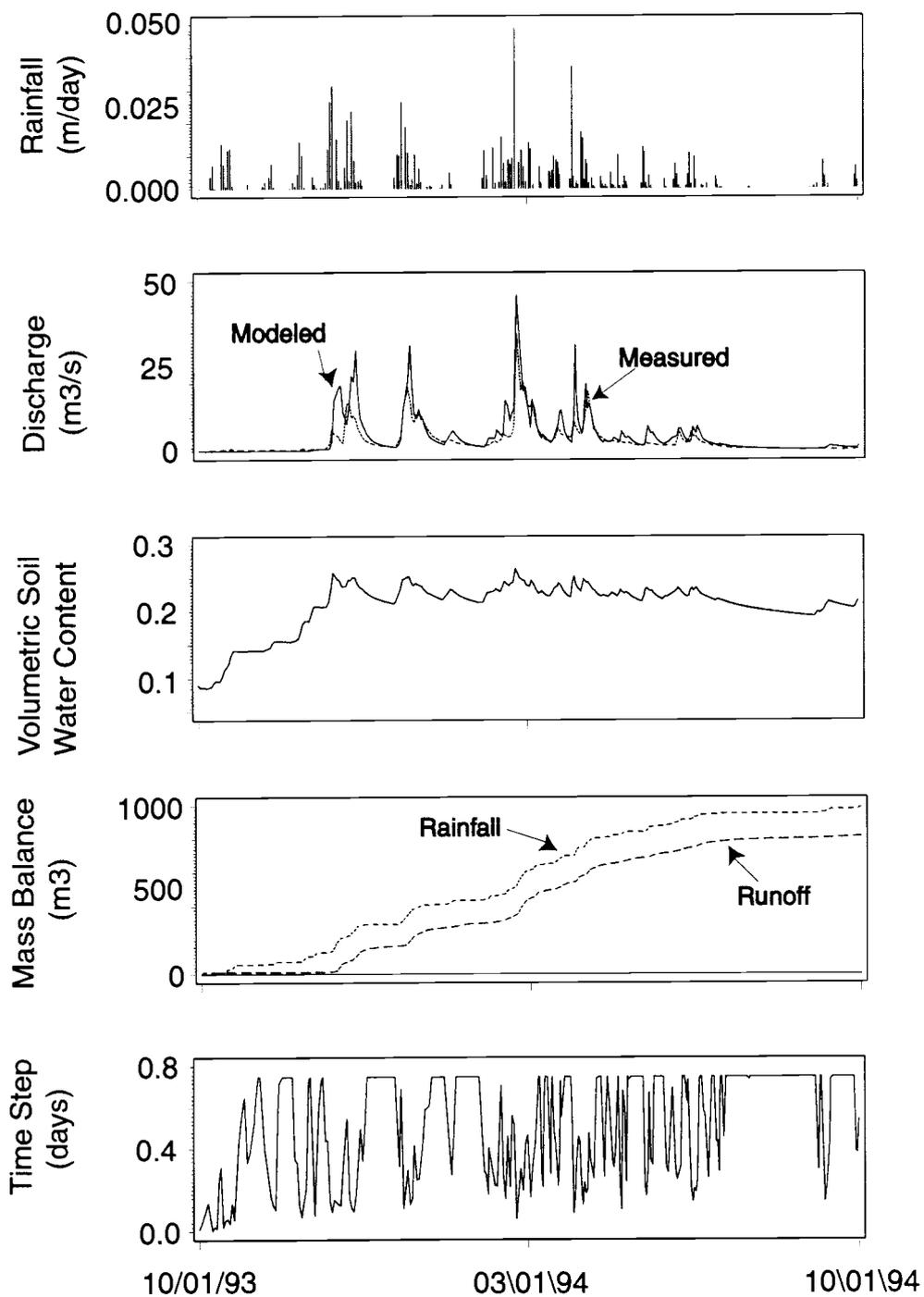


Figure 4.16. Verification results from Wiley Creek for water year 1994. All results are simulated except for the series labeled 'Measured' in part B.

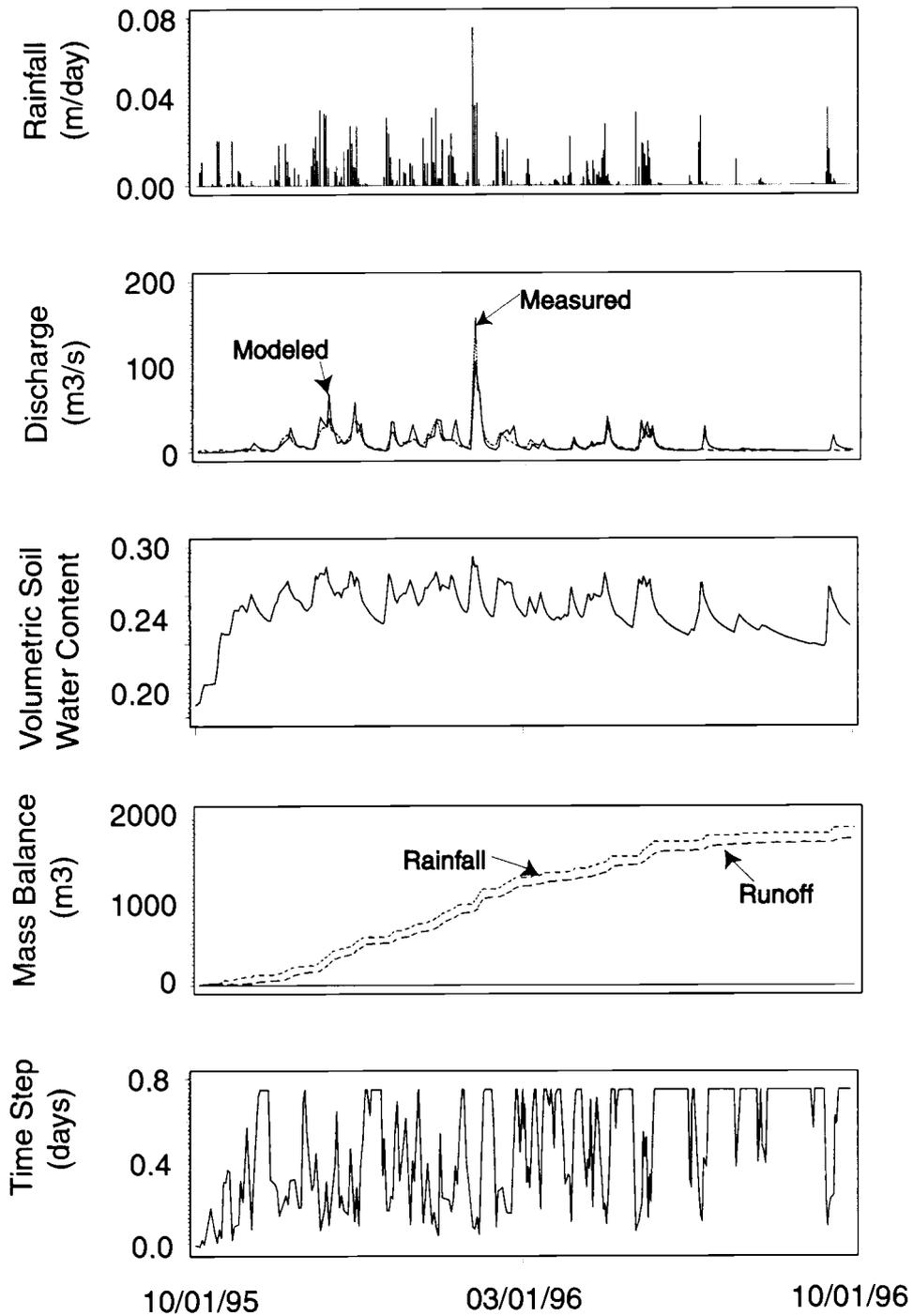


Figure 4.17. Verification results from Wiley from for water year 1996. All results are simulated except for the series labeled 'Measured' in part B.

4.5.4 The value of additional criteria

The need for multi-criteria calibrations of hydrologic models is well established in the hydrologic literature [Yapo et al., 1998; Freer et al., 2002] and a number of methods exist to formally incorporate multiple criteria into calibration procedures [Martin and McCutcheon, 1998; Boyle et al., 2000; Freer et al., 2002; Misirli et al., 2002]. These methods generally focus on the degree to which additional calibration data can reduce parameter uncertainty. A wide variety of criteria have been utilized, including multiple components of discharge data [Boyle et al., 2000; Turcotte et al., 2002], various objective functions [Gan and Biftu, 1996], remotely sensed saturated area estimates [Franks et al., 1998], observed saturated estimates [Beven and Kirkby, 1979; Ambroise et al., 1996], water table depth [Lamb et al., 1998; Blazkova et al., 2002], snow covered areas [Leavesley et al., 2002], and isotopic signals [Hooper et al., 1988]. Uhlenbrook et al., [2002] used an extensive analysis of tracer concentration data to develop flowpath distribution underlying the Tracer Aided Catchment model (TAC), producing, at least conceptually, a priori reductions in output uncertainty. The first study to formally incorporate the results of tracer analysis (through hydrograph separations and estimates of percent new water) into an a posteriori parameter and uncertainty estimation procedure appears in Seibert and McDonnell, [2002]. They focused attention on a small catchment (the same Maimai watershed outlined in other areas of this document), and were able to

demonstrate how the incorporation of these additional criteria produced significant reductions in output uncertainty.

Here, in a manner similar to Seibert and McDonnell, [2002], we utilize hydrograph separation and a model of tracer movement to provide concentrations and produce simulated values of percent new water. This work differs from the former through application at larger scales and in an area with considerably less data for model analysis. We show how this additional state variable can be used to more fully characterize model functioning and the set of acceptable parameter sets. Though we do not formalize the incorporation of this new water estimate into parameter uncertainty reductions, we do demonstrate how the statistic may be used to reject models that appear to provide otherwise behavioral responses.

Estimation of new water discharge through hydrograph separation is an increasingly common undertaking [Sklash et al., 1986; McDonnell, 1990; Montgomery et al., 1997; Uhlenbrook et al., 2001] though it generally relies upon field measures of conservative tracer concentrations. Two component hydrograph separations (where the components are pre-event (old) water and event (new) water) are based on the following basic mass balance:

$$\frac{Q_{old}}{Q_{total}} = \frac{C_{total} - C_{new}}{C_{old} - C_{new}} \quad (4.15)$$

where Q refers to discharge and C the concentration of tracer in each component. To implement this mass balance within the model we assume initial concentrations in the system (C_{old}) and a distinct concentration in rainfall (C_{new}).

This difference corresponds to the different isotopic ratios or concentrations relied on during measurement based tracer hydrograph separations. Mass balances of each of these components, along with the tracer in simulated runoff are maintained in a fashion similar to that described for water in section 4.4.1. Upon completion of an event simulation, these integrated masses are converted back to the concentrations (Equation 4.15) and the hydrograph is separated into old and new water contributions.

The focus of this section is on the San Jose watershed where the flashy hydrology and dominance of overland flow can be assumed to produce relatively high new water contributions to storm hydrographs. While there is no chemical data to corroborate this assumption, the correspondence with experimental observation justifies it. In section 4.4 we demonstrated the model's capability to simulate hydrographs in the San Jose. Here we further evaluate those simulations by assessing the internal dynamics of the model through hydrograph separation. Figure 4.18 displays the same data as Figure 4.8, but in addition includes an estimate of the new water discharge over the event.

This estimate of percent new water is classified into those simulations producing less than 50 percent new water and those producing greater than 50 percent new water. Given the predominance of Hortonian overland flow in the basin, it is reasonable to assume large new water contributions, in all likelihood greater than 50 percent. This figure suggests very strongly that the model performs considerably worse, across parameter space, than can be documented

using discharge based error calculations alone. The maximum R_{eff} reported from discharge based error was 0.57, but to achieve a new water content of at least 50 percent, the value decreases to at most 0.45. In the event that our interest includes the internal spatial dynamics of the San Jose basin, this highly instructive information provides additional evidence that the model structure is inadequate for simulations in this region. This result suggests the need to incorporate more explicitly the unique hydrologic pathways operating in the San Jose. These very likely include infiltration excess overland flow and more sophisticated routing of overland flow.

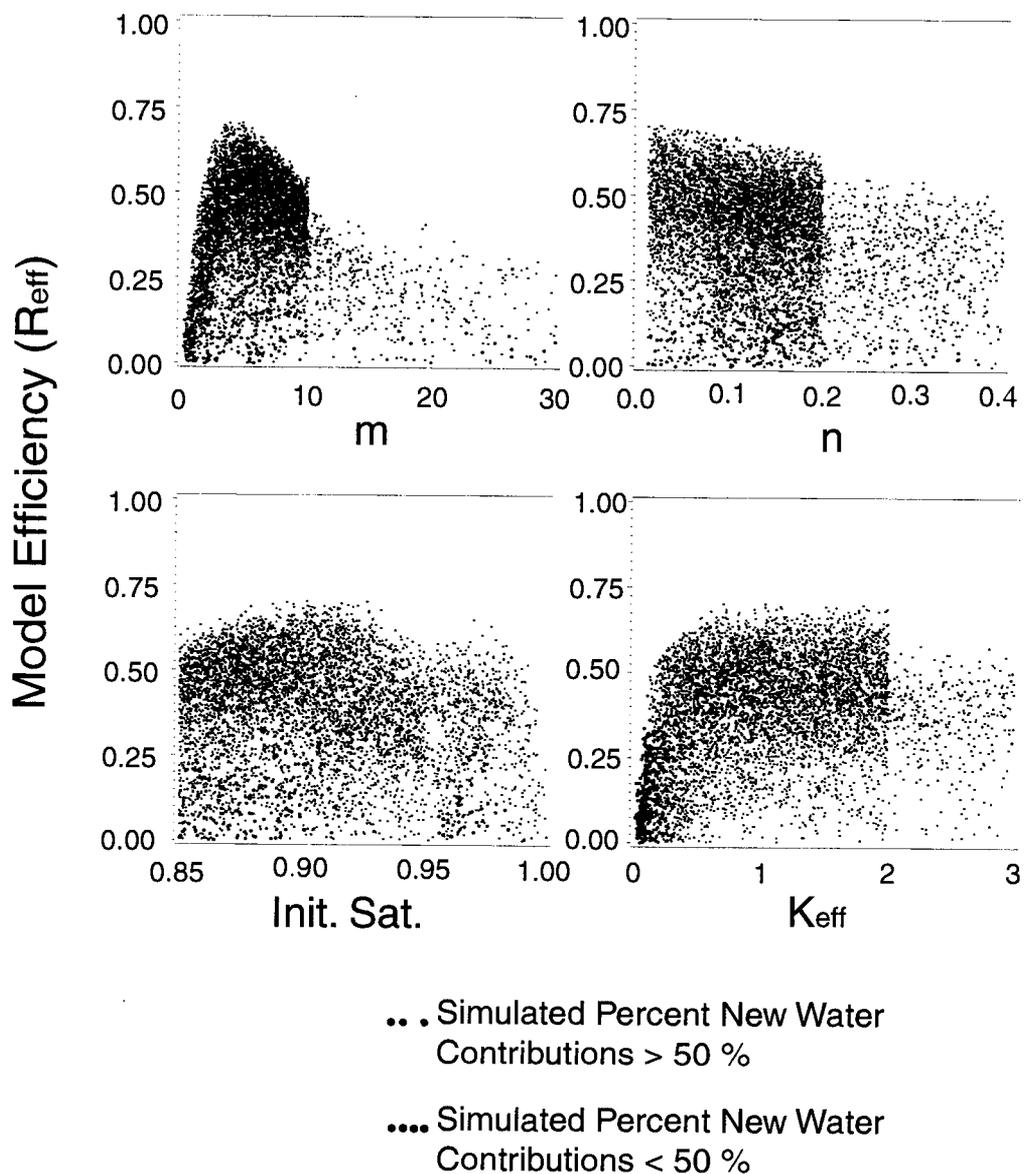


Figure 4.18. A scatter plot with model efficiency for each parameter, color coded as percent new water contributions to overall streamflow for the May 28 to May 30 2001 event. Maximum efficiency decreases significantly to approximately 0.45 in the presence of this additional model derived information.

4.5.6 Scale effects on parameter space

The scale of discretization chosen during the development of hydrologic model input has potential to effect model results [Bloschl and Sivapalan, 1995; van Loon and Keesman, 2000]. Models that include topographic controls on the downslope movement of water appear quite scale dependent. One of the reasons appears to be the increased precision in land surface slope estimates as the scale over which they are calculated decreases. Much of the work that attempts to quantify the issue of scale in hydrologic models has focused on grid size effects on the extraction of information from Digital Elevation Models (DEMs) [Freeman, 1991]. Specific studies have evaluated grid size effects on the generation of the topographic index and how that affected TOPMODEL simulations [Wolock and Price, 1994; Zhang and Montgomery, 1994]. The significant scale dependence of a fully distributed grid based model of hydrology has also been noted [Kuo et al., 2001]. This work also demonstrated that this significance was due the more the precise estimates of slope derived from finer scale grids.

These studies appear to have two significant similarities (1) a focus on grid cell size as the definition of scale and (2) for those studies including hydrologic models, the identification of scale effects under a constant set of model parameters. We add to the discussion of scale effects on hydrologic models through the use of a flexible vector definition of scale, and through analysis of how parameters might effectively hide the effect of scale.

Given that most distributed models are grid based, it is not surprising to find grid size as the most common definition of model scale. But that definition is somewhat restrictive and is unacceptable to the evaluation of scale in a vector based model. Here we define scale more loosely as relatively large to relatively small, and interpretation of those quantities relies on maps and the map scale bar (figure 4.19). Given this definition, model discretization can reflect more directly the landscape slope, with more model units in steeper areas and fewer units on shallower slopes.

Throughout this paper we have evaluated model functioning through fairly complete characterizations of parameter space. We pursue this path because, as argued by [Freer et al., 1996], the parameter arrays are as important to model functioning as the model algorithms themselves. We continue with this approach as we evaluate scale. Under most modeling methodologies, after the modeler assumes a scale (generally through the choice of a grid size), the effects of that assumption on model results are often not interpreted. One of the reasons for this may be that discharge-based model parameterization can effectively make up for any deficiencies introduced by scale.

We argue that scale is essentially an additional model variable, and that it can be treated in a similar manner to parameters. WET_Hydro flexibly defines scale over which calculations occur, and code-based scale changes between model runs are easy to implement. We use these features to evaluate two questions related to scale. Can standard single-criterion parameter estimation effectively

hide the effect of model scale on model output? If so, can other model outputs find it?

We focus now on the Maimai watershed because of its relatively high efficiency, detailed inputs, and relatively fast simulation times. To answer the question we include two additional Monte Carlo simulations. The first simulation represents a medium size model scale and the second represents a small model scale (Figure 4.19).

Topographic representations improve as we move to the smallest scale, and intuition might suggest that model efficiency should improve along with it. If efficiency improves, then we conclude that the given set of parameters and model structure cannot hide the effects of scale. If efficiency does not improve, such a conclusion cannot be made. The distinction is important because if we can effectively remove the scale effect through parameter choices, then there is significant potential for an efficient model with incorrect internal dynamics. Figures 4.20 and 4.21 represent model efficiency versus key parameters for each of the two additional simulations. These figures correspond to the watershed configurations outlined in Figure 4.19b and 4.19c, respectively. Figure 4.4 corresponds the watershed configuration outlined in 4.19a. For each of the three configurations, maximum efficiencies are essentially equivalent, but occur at different places in parameter space. This appears to indicate that while scale does change model results for any given parameter vector, single criterion hydrograph calibrations will simply adjust the vector and result in an essentially equivalent

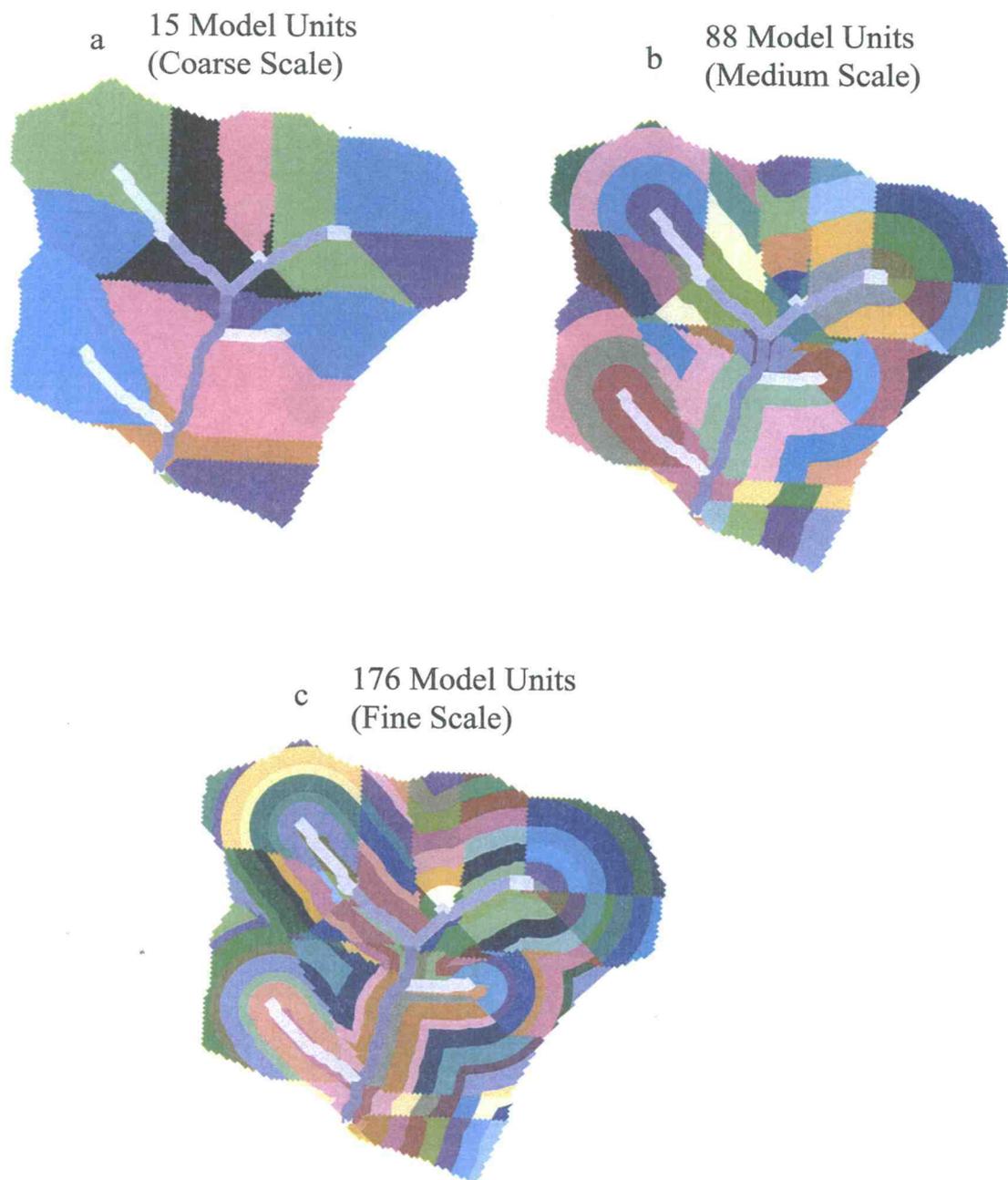


Figure 4.19. A set of simple maps depicting the three different model unit configurations for the Maimai catchment. The colors are used to delineate individual model units. Map a represents a coarse model scale, map b a medium model scale, and map c a fine model scale.

efficiency. This suggests the need to evaluate models in a multi-criteria sense, where the additional criteria are developed from datasets other than the hydrograph.

Here we approach these additional data from a simulation standpoint, and suggest model derived output that appear to show a large degree of scale dependence. The output we focus on is the frequency of saturation over an event of the three model configurations. We employ the same Monte Carlo type simulations, but report the frequency of saturation for each realization, in addition to the other values reported elsewhere. Figure 4.22 shows the frequency of saturation across all realizations occurring for each model scale. Figure 4.22a establishes that no parameter combinations produced saturated area in the watershed at any point during the simulation for the simplest watershed configuration. Alternatively, Figure 4.22c indicates the opposite – that for the most detailed watershed configuration, every parameter combination resulted in a model that that produced saturation. To some extent, of course, this result is intuitive, and is one of the reasons a modeler might choose a distributed simulation over lumped equivalent. But the quantification of this feature provides information that may be used to make decisions beyond the choice between lumped versus distributed and into the realm of model unit scale. These kinds of model applications have potential to guide the development of watershed datasets that might be both simple to measure and prove directly useful in the modeling process.

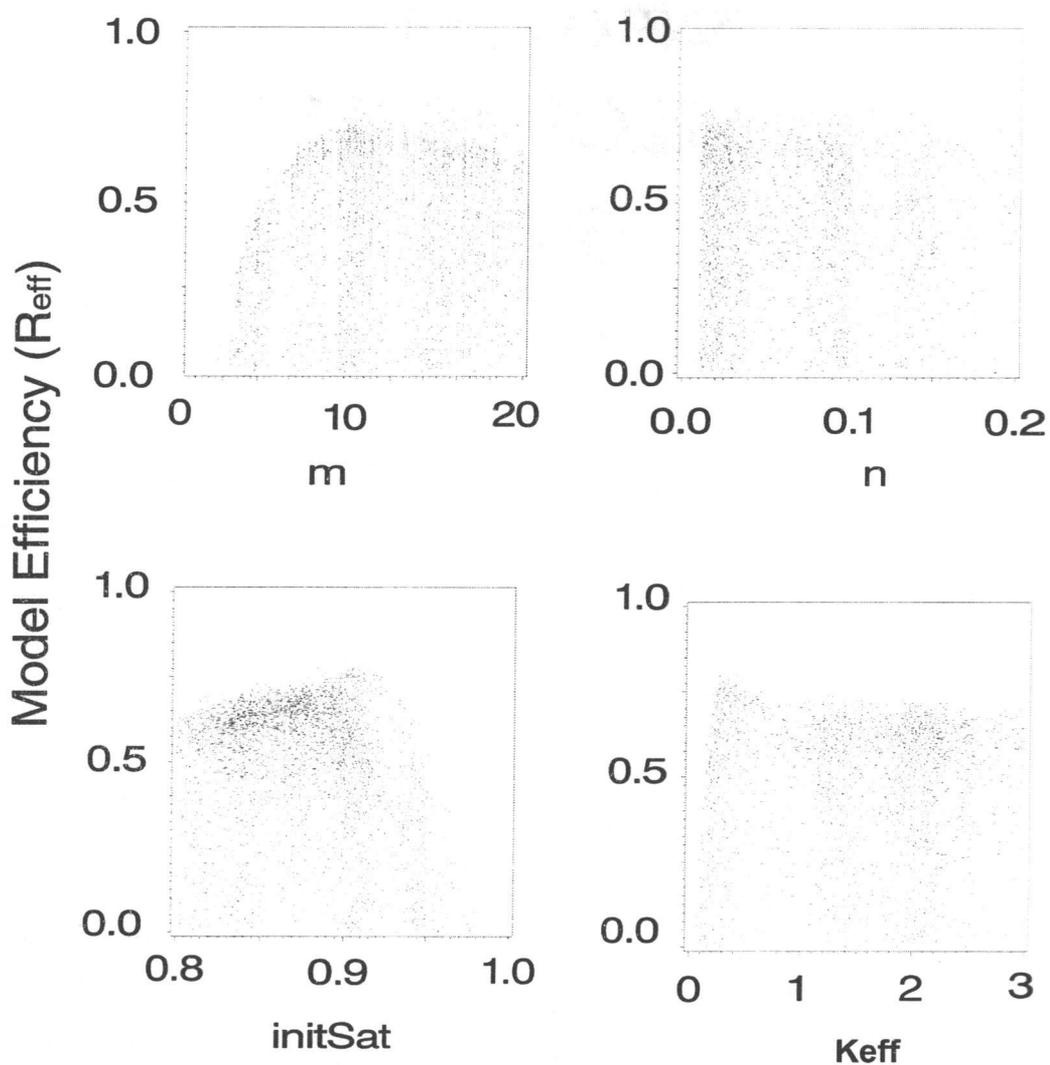


Figure 4.20. Scatter plot of model efficiency versus parameter value for four model parameters in the Maimai watershed, using the medium model scale that corresponds to figure 4.19b. The event over which these efficiencies were developed began on September 2, 1987 and ended on September 5, 1987. The maximum value of efficiency is approximately 0.77.

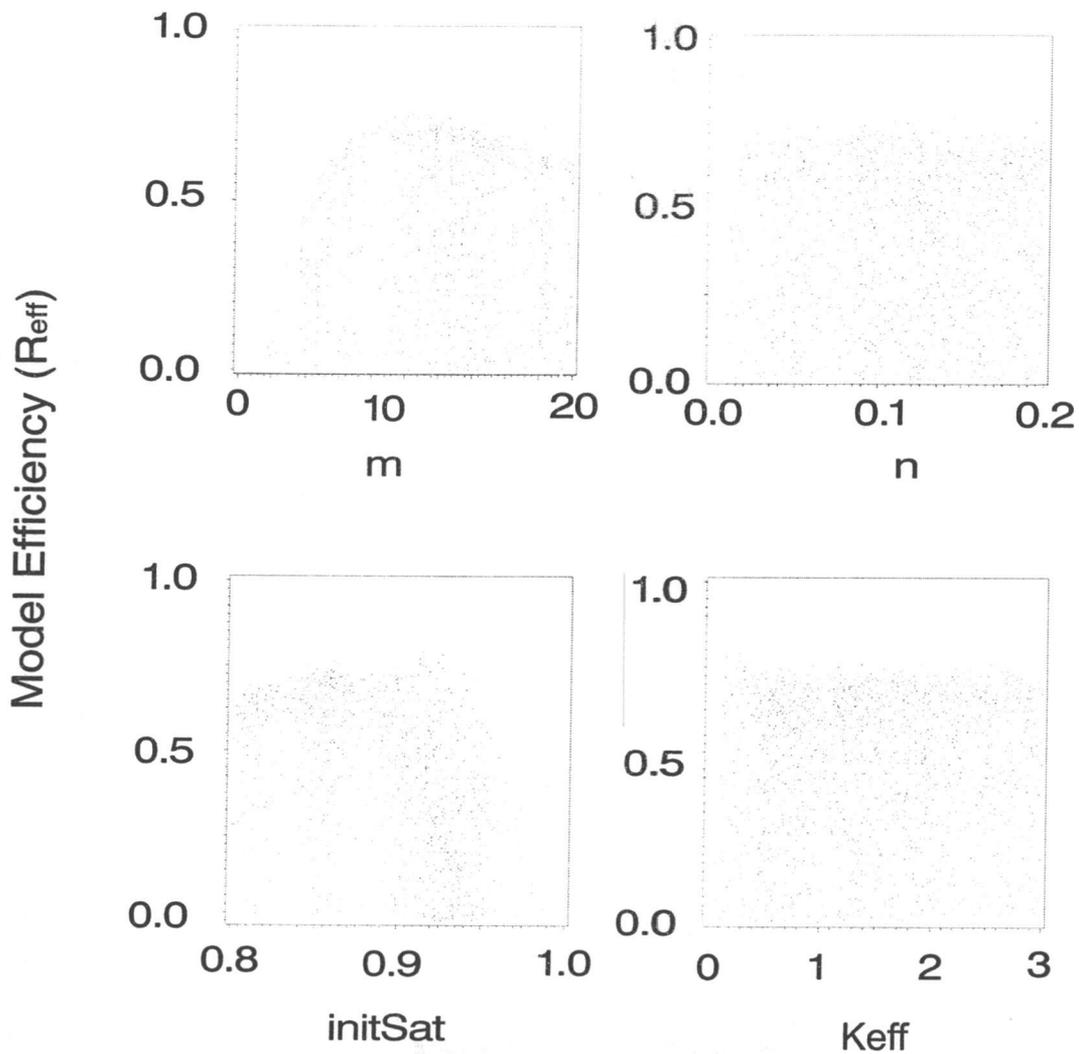


Figure 4.21. Scatter plot of model efficiency versus parameter value for four model parameters in the Maimai watershed, using the fine model scale that corresponds to figure 4.19c. The event over which these efficiencies were developed began on September 2, 1987 and ended on September 5, 1987. The maximum value of efficiency is approximately 0.77.

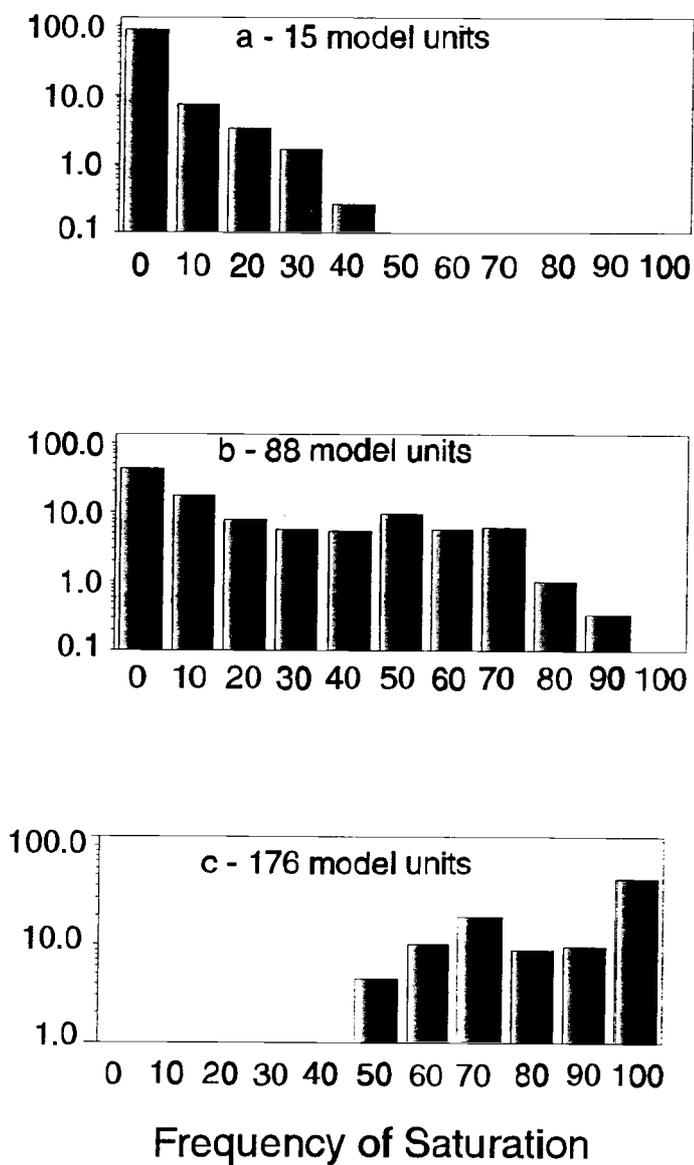


Figure 4.22. The frequency of saturation under different model resolutions and parameter sets at Maimai.

4.6 Summary and Conclusions

This paper presents the development and application of a flexibly distributed hydrologic model. We demonstrate model application at a variety of sites across different hydrologic regimes and scales. The model was able to provide reasonable replication of outflow characteristics. The ability to simulate measured hydrographs is a prerequisite for any hydrologic model, and so this result is certainly useful. But the conversion of rainfall to runoff is handled quite nicely by a wide variety of other models and so this result, while expected and necessary, does not represent the more interesting aspects of the study.

Variable time step Runge-Kutta solutions have been applied in a wide variety of simulation contexts for years, but appear to have been avoided in the development of most hydrologic models. This is somewhat surprising given the rapidly changing and essentially discontinuous nature of rainfall input, which has significant potential to benefit from the maintenance of numerical error. The inclusion of a variable time step, and its implicit control on numerical dispersion, significantly improve the efficiency of the simulation code, and we recommend further use and testing of the concepts.

The use of Monte Carlo simulation also is not, in and of itself, a new contribution. However, our emphasis on Monte Carlo characterization of a variety of basins provides further evidence for the wide variety of potentially acceptable results, and the significant parameter uncertainty given discharge based model

efficiencies. We encourage the use of Monte Carlo simulations because of the uncertainty these results tend to highlight. Extending these analyses in order to quantify parameter uncertainty is an area that has received increased attention in recent years [Romanowicz et al., 1994; Yapo et al., 1998] and we see this positive trend continuing.

We have also documented a set of model derived criteria that have potential to improve dialog between field and modeling components of hydrologic research. The evaluation of internal model dynamics through the implementation of a virtual tracer model allowed us to reject model structures that appeared from the evaluation of output hydrographs reasonably acceptable. The hydrograph was also shown to be independent of scale, and therefore of limited use in making determinations regarding the most appropriate model configurations in. We proposed a simple model result and established that its value was scale dependent and therefore has potential to assist in model configuration. There are no doubt many other model derived output variables that can be used to better direct modeling exercises. We feel that the development of these additional criteria, and the formalization of methods to utilize them will remain an area of worthwhile research.

**5.0 Watershed Planning and the Use of Distributed Modeling Concepts –
Assessing the Cumulative Hydrologic Effects of Potential Landuse
Change**

Kellie B. Vache

John P. Bolte

Mary V. Santelmann

5.1 Abstract

Evaluation of the effects of landuse change on the hydrology and water quality of watersheds is an important component of watershed management. In this paper we present a new modeling approach designed to facilitate the evaluation of the effects of proposed landuse changes. The focus is on water quantity and erosional changes. This model of change is built upon the hydrologic model described in Chapter 3, and focuses on simple, and spatially distributed, volume or mass based accounting procedures. This process results in a model with significantly fewer parameters when compared with other standard models relating landuse change and water quality. A version of the universal soil loss equation is used to provide estimates of sediment movement.

The model focuses on watershed scale basins (1 to 1000 km²) and the cumulative effects of multiple restoration sites. Sensitivity to a set of restoration options, including riparian buffers, wetlands, stream discharge augmentation, as well as management alternatives designed to reduce erosion, is an integral component of the model. We combine the dynamic, physically-based models with an existing decision-support system (DSS), and simulate the effects of a wide variety of DSS derived watershed management plans. This combination provides direct evaluation of DSS proposed landuse plans. We show how these evaluations provide additional understanding of the watershed plans through qualitative assessment of their effects on water quantity and in-stream concentrations of total suspended solids. The code is written using an object-

oriented programming language and a complete user interface is included and provides run time access to model parameters and algorithms.

5.2 Introduction

The concept of multi-use watershed management originated with the understanding that material resources including timber, water and agricultural commodities are only a portion of the valued uses of watersheds [Brooks et al., 1997]. Biodiversity, recreation, and ecosystem function are additional benefits that interact with water quality and are derived from watersheds. Under the multi-use concept, watershed management becomes a process of balance, weighing the perceived costs and benefits of individual restoration actions to make decisions. Various authors have demonstrated the utility of distributed, scenario-based future changes studies [Hulse and Gregory, 2001; Steinitz and McDowell, 2001] designed to inform planning decisions. A more direct method of facilitating the complex decision making process involves the use of expert systems and decision support systems. [Janssen, 1992; Engel et al., 1993; Zhu et al., 1998; Reynolds et al., 1999; Koutsoyiannis et al., 2002; Lamy et al., 2002] These data driven software tools process spatially-distributed data and, given constraints, rules, and objectives, provide managers with some informed idea of the kinds of activities and operations that are most suitable. The result is a more dynamic

conceptualization of the future than is developed through static scenario development.

Expert systems tend to maintain a local focus because they rely heavily upon site level characteristics in support of rules. Interactions that may occur between restored sites can be difficult to integrate into a rule-based framework. But these interactions – essentially, the larger context into which a decision is developed - may be as important to the success of that decision as the local conditions. More complete decision support systems act as extensions to the expert systems, including additional components beyond rule based decision making [Bolte et al., 2000]. These additional components may include environmental simulation models which are often designed, through endpoint measures, to specifically quantify the cumulative effects of the state of the upland areas.

Considering the issues of hydrology and water quality, this potential exists because models simulate water as it transitions from the atmosphere, to the soil, into channels, and then out of the watershed, at which point outputs reflect the watershed state. In addition, the evaluation of landuse change effects on hydrologic process through modeling is a well established concept [James, 1965; Bicknell et al., 1997; Wooldridge et al., 2001; Miller et al., 2002]. Direct combination of rule-based decision-making and hydrologic simulation models presents one potential method of incorporating spatial interactions into the decision-making process.

In this paper we present a spatially-distributed model of erosion designed to work directly within an established rule-based decision support framework, known as RESTORE [Lamy et al., 2002]. The model provides an assessment of the cumulative effects of watershed restoration plans designed by the decision support system. The hydrologic model described in chapter 4 and the network model from Chapter 3 provides the basis for this set of models designed to evaluate landuse change effects. The state variables and processes of interest have been carefully defined, and represent only a small subset of potential quantities of interest. This study focuses on erosion and peak flow changes and the analysis of other water quality concerns, including pesticides and nutrients, are not included in the paper. We are continuing with research necessary to add some of these additional components, but their inclusion is beyond the scope of the current study.

The paper begins with a description of the RESTORE Decision Support System and its use of simulation to further extend the utility of the rule base. Included in this section is a description of the restoration options that the model is designed to effectively simulate. It then proceeds to define the model and concludes with an example application at a sub-watershed of the Willamette River in Northwest Oregon.

5.3 Decision Support

The decision support system described by Lamy et al., [2002] is designed to provide watershed councils and planners the capability to develop and evaluate watershed scale restoration plans. The system relies on user-provided restoration goals and objectives, along with a complete set of rules relating spatial data and the capability of various restoration options to improve watershed conditions. The rules indicate the degree to which any option might satisfy given objectives. The current version of the DSS includes five objectives: water quality, water quantity, habitat improvement, economic benefit and social conditions. These objectives are further classified into 28 sub-objectives. As an example, a user might focus decisions based on reduction of stream temperature, which is a sub-objective of the water quality objective. RESTORE processes the rules to rank the effectiveness of individual restoration alternatives for each place in the environment, given prescribed objectives. The result of this process is a spatially-explicit dataset representing the rule-generated watershed scale restoration plan. The plan includes both landuse and landcover alternatives, and the system is designed as an interactive tool where the effects of different planning goals can be quickly translated into additional potential landscapes. In this regard the tool can be used to compare and contrast the effects of combinations of different watershed goals and objectives on optimum restoration strategies. A challenge in making use of RESTORE is the difficulty in qualitatively expressing the

differences between the varieties of different landscapes that might be developed. Simulation models which are able to quantify the effect of each landscape on some defined state have potential to address this limitation.

5.4 Model Description

A simulation model (WET_Hydro) acts as the basis for this research. The hydrologic components of the model are outlined in chapters 3 and 4 of this thesis. To evaluate the effects of different landscapes on water quality, a variety of additional models have been constructed. These models include a simple erosion simulation, along with a variety of models designed to incorporate a heuristic understanding of water quality restoration into the simulation context.

The need to establish uncertainty measures along with model derived estimation of ecological endpoints has been well documented [Wu and Marceau, 2000; Santelmann et al., 2001; Brugnach et al., 2003]. The models proposed in this study are in many cases overly simplified (see equations 5.10-5.15 for example). This simplification, though, is by design and reflects the fact that the consequences of small scale restoration are highly uncertain.

Consider riparian buffers and their effects on sediment export. The fact that they trap sediment is well documented, (see the review by [Wegner, 1999]) but the amount of that reduction varies widely based on site level characteristics. Here we simply estimate the value and (because the model is run on a distributed

landscape, where there may be hundreds of separate restoration sites) the effect of this estimate provides an indication of the integrated effects of basin wide restoration. Results do *not* indicate how effectively individual projects achieve their goals, but that is not the question we seek to answer. Instead the focus is on cumulative effects of basin wide restoration given *assumptions* about how effectively individual projects operate. This approach lends itself to estimates of the uncertainty in the endpoint as a function of uncertainty in buffer potential. A complete Monte Carlo simulation of various model parameters was beyond the scope of this project but we do provide a basic example of how multiple model runs across a sample of parameter space can be used to provide an indication of the endpoint uncertainty.

5.4.1 Erosion Model

The model of erosion is based on the Modified Universal Soil Loss Equation (MUSLE) [Williams and Berndt, 1977]. The original Universal Soil Loss Equation is given by [Wischmeier and Smith, 1965]:

$$A = R * K * LS * C * P \quad (5.1)$$

where A is the soil loss rate in mass/area time, R is the rainfall erosivity factor, K is the soil erodibility factor, LS is the length-slope, C is the crop management factor and P is the erosion control factor.

The length slope factor represents that idea that steeper slopes have increased erosional potential. The longer these slopes are, the greater that potential. Moore and Burch, [1986] developed an equation relating the length slope factor to area and slope as:

$$LS = \left(\frac{A}{22130} \right)^{0.4} \left(\frac{\sin(S_0)}{0.0835} \right) \quad (5.2)$$

where A is area (m²) and S₀ is the slope of the landscape unit. The k factor can be found in the variety of digital soil databases produced by the National Resources Conservation Service (NRCS), including the SSURGO data which we include as input to the hydrologic model. The crop management factor reflects vegetative cover, plant litter, soil surface and management. The erosional control factor represents explicitly those landuse practices that reduce slopes. These include terracing and strip cropping and have not been published for non-agricultural areas [Brooks et al., 1997]. In most applications [Jackson et al., 1986 ; Tetra Tech 1995], these factors are treated as calibration parameters.

Our interest in the USLE is to evaluate, in very simple terms, erosion and the cumulative effects of a small set of restoration options, applied in different configurations over watershed scale basins. For this reason significant resources were not allocated toward calibrating these values. The focus is instead on comparisons between current and restored landscapes. Table 5.1 provides a listing of the C factor used in the modeling.

Table 5.1. USLE coefficients implemented in the study.

| LULC A | Landuse | C | P |
|--------|--------------------|-------|---|
| 1 | Rural_Residential | 0.400 | 1 |
| 2 | Urban | 0.050 | 1 |
| 3 | Agriculture | 0.01 | 1 |
| 4 | Forest | 0.007 | 1 |
| 5 | Wetlands | 0.001 | 1 |
| 6 | Natural Vegetation | 0.001 | 1 |
| 7 | Water | 0.001 | 1 |
| 8 | Roads | 0.100 | 1 |

The MUSLE assumes that storm discharge can provide a time variant estimate of the erosivity of a rainfall event. The equation replaces the rainfall erosivity factor with the combination of accumulated discharge and peak flows ($Q \cdot Q_p$) over an event time scale of some arbitrary number of days. This factor is commonly referred to as the Runoff Factor [Dunne and Leopold 1978]. We further modified the conceptualization of erosivity to develop an estimate of erosion that corresponded to our treatment of time, space and discharge. We use the estimate of discharge rate and volume at each point in time and space as estimates of Q and Q_p . In the original MUSLE conception these values represented stream discharge and we have simply assumed that peak discharges and volumetric rates from the upslope units on which erosion occurs are reasonable substitutes.

5.4.2 Sediment Routing

The erosion model consists of two state variables, which are similar to those in the hydrologic model. The first state is **upland** soils and the second state is instream sediments. The rate of change of each of these states is the variable of interest.

The rate of erosion from upland areas is calculated from the MUSLE, and these rates are integrated to calculate the mass of soil in each upland unit. The differential equation, which utilizes the rates calculated by the USLE, is written as follows:

$$\frac{dM}{dt} = \text{input} - \text{output} \quad (5.3)$$

or

$$\begin{aligned} \frac{dM}{dt} = & (Q * Q_p * K * LS * C * P * A)_{above} \\ & - (Q * Q_p * K * LS * C * P * A)_{this} \end{aligned} \quad (5.4)$$

The solution to this differential equation is calculated at each time through the use of the Runge-Kutta-Felberg solution procedure, and gives an indication of the quantity of soil within each upslope. It is worth noting that this model is not intended as a landscape evolution procedure, although it does bear some resemblance to one. We included the fully integrated mass balance to provide a

degree of continuity throughout the calculations of state. Despite this, our focus is on only the rates of downslope sediment movement.

Two distinct instream sediment routing procedures have been developed. The first of these procedures explicitly routes suspended (and assumed dissolved) sediment in the channel, from sub-reach to sub-reach. The second procedure involves a simple summation.

The explicit routing procedure is analogous to the instream water routing procedure outlined in Chapter 3, and providing a model estimate of state for each sub-reach in the network. Whereas the stream routing procedure includes momentum and mass conservation, here we employ only a statement of mass conservation. The conservation of mass for a conservative substance can be stated as [Schnoor, 1997]

$$\frac{\partial C}{\partial t} = -u_i \frac{\partial C}{\partial x_i} + \frac{\partial}{\partial x_i} E_i \frac{\partial C}{\partial x_i} - R \quad (5.5)$$

where the first term on the right side represents the rate of change of mass due to advection, the second the rate of change of mass due to diffusion, and the last term refers to rates of degradation or reaction. C is concentration (kg/m^3), t is time, u_i is average velocity in the i th direction, m/s , x_i is the distance in the i th direction and R is the reaction transformation rate ($\text{kg}/\text{m}^3\text{s}$). This equation, commonly referred to as the advection-dispersion equation, can be significantly simplified through discretization into a series of completely mixed compartments. With this assumption, the partial differential equation can be recast as a simpler

ordinary equation, with exchange flows between boxes providing the spatial distribution. The equation is as follows [Schnoor, 1997]

$$V_j \frac{dC_j}{dt} = \sum_{k=1}^n Q_{j,k} C_k + \sum_{k=1}^n Q'_{j,k} C_k - \sum_{k=1}^n Q_{k,j} C_j - \sum_{k=1}^n Q'_{k,j} C_j - k C_j V_j \quad (5.6)$$

where V_j is the volume of the j box, C_j is the concentration in the j box, t is time, n is the number of compartments adjacent to j , $Q_{j,k}$ is the inflow from box k to box j , C_k is the concentration in box k , $Q'_{j,k}$ is the dispersive flow from k to j , $Q_{k,j}$ is the outflow from j to k , $Q'_{k,j}$ is the dispersive flow from j to k , k is the first order rate constant for the transformation and $Q'_{j,k}$ and $Q'_{k,j}$ are symmetric matrices with zero diagonal. In this equation, the first two terms on the right side are the mass and dispersive inflows, respectively. The third and fourth terms are the mass and dispersive outflow, respectively.

If we assume that advection is the dominant transport process and that the suspended sediment is both conservative and non-reactive, a further simplification gives:

$$V_j \frac{dC_j}{dt} = \sum_{k=1}^n Q_{j,k} C_k - \sum_{k=1}^n Q_{k,j} C_j \quad (5.7)$$

The Runge-Kutta solution procedure, along with the spatially distributed 'boxes' defined by the hydrologic model, provides a simple explicit solution to this equation. Grouping unknown values on the left, and rewriting the equation to include lateral sediment inflow and sediment inflow from the upstream box, the equation becomes:

$$\frac{dM_j}{dt} = Q_{in,upslope} \frac{M_{in,upslope}}{V_{in,upslope}} + Q_{in,upstream} \frac{M_{in,upslope}}{V_{in,upslope}} - Q_{out} \frac{M_{out}}{V_{out}} \quad (5.8)$$

This equation is solved subsequent to the flow routing equation, with all values on the right hand side known.

The routing procedure given above provides an estimate of total suspended solids through the length of the network system. But in many instances, there may be an interest in simulating only upland sediment movement. This choice could be made for at least two reasons. First, the time of travel in the stream system is, for smaller basins, much smaller than travel times on upslopes. For watershed scale basins (< 1000 km²), it can often be reasonably assumed instantaneous [Beven and Kirkby, 1979]. If the goal of a simulation is to look at relatively long times, in watershed scale basins, then stream routing of suspended sediment might be overlooked. A second reason may be the need to minimize the real time over which a simulation occurs. The explicit routing procedure outlined above is stable, but only at very small time steps. Including the routing greatly increases simulation time. The alternative to routing the sediment is a simple summation. In this case, the 'routing' equation becomes:

$$\frac{dM}{dt} = \sum_{i=0}^n M \quad (5.9)$$

where n = number of upslope units, M is the mass of sediment entering each reach from the upslope structure. Implementation of this equation provides an estimate of the rate at which sediment leaves a basin over a given time period, and ignores the in-channel time of travel.

5.4.3 Model Sensitivity to Landuse Changes

The model is sensitive to landuse changes characterized directly by parameters in the USLE. As an example, if under current conditions a landscape unit is in agriculture and no erosion control is practiced, the parameter c might be assigned a value of 0.1. But if under future scenarios significant erosion control practices are implemented, the value may decrease to 0.01. The value of sediment movement, derived from the USLE for that individual landscape unit, would decrease by a factor of 10, given the new scenario. But the model is insensitive to landuse changes which are not characterized by parameters. Sensitivity to the changes has been built into the model through the inclusion of alternative terms that modify sediment and water movement.

5.4.4 Wetland Model

Wetland areas can provide important hydrologic and water quality services to watersheds [Hammer' 1989; Coleman et al., 2001]. The capacity of wetland to trap sediment and reduce export is well documented [Hemond and Benoit, 1988; Johnston 1991; Hupp et al., 1993; Gilliam 1994]. They are important components of the restoration scenarios defined by the RESTORE DSS and the hydrologic model is designed to explicitly simulate the effects of these areas on hydrology and water quality. Various researchers have proposed wetland specific hydrologic models [Sun et al., 1998; McKillop et al., 1999]. In all cases, the research we identified which focused on simulation of the hydrologic response of wetlands concluded with data intensive and highly parameterized models. We propose an alternative conceptualization that adheres to the concept of simple statements regarding the local effects of spatially-distributed restoration and how they might be upscaled through model application, to provide basin-wide insights.

In cases where an upland model unit includes a polygon with alternative landuse class "wetlands", a select group of parameters and procedures related to the hydrologic model are manipulated as follows:

$$K_{\text{altern}} = k_1 * K_{\text{eff}} \quad (5.10)$$

$$S_{\text{altern}} = k_2 * S_0 \quad (5.11)$$

where K_{eff} is the effective basin conductivity, k_1 is a user-defined coefficient, S_0 is the slope of the polygon and k_2 is a user-defined coefficient. In addition, water in

wetland units that is unable to infiltrate is not routed directly to the nearest stream channel. Rather, the water is left standing on the unit, and infiltrates as the area dries out. These simple and consistent changes reflect how wetlands, under ideal conditions, tend to function. Clearly in some instances more information might be available in order to parameterize the effect of wetlands, and additional understanding has some potential to improve results. But we justify the simplifications implemented here on the familiar grounds that our goal is to estimate the cumulative effects of spatially distributed wetlands in potential watershed scale landscapes – not the functioning of any single wetland unit.

5.4.5 Flow Augmentation Model

The idea of increasing late summer discharges in streams draining agricultural lands is an increasingly cited restoration alternative. It is generally accepted that uses of groundwater and stream flow for domestic, but more importantly agricultural irrigation, tend to decrease late summer discharge in stream channels. These decreases reduce the potential for fish passage and increase the potential for elevated stream temperature [ODEQ, 2001]. Because these anthropogenic influences on discharge are most significant during periods when discharge would naturally be at a minimum, a useful measure of evaluation is simply the minimum discharge.

In recognition of restoration potential of summer discharge increases, the rule base included in RESTORE includes an option referred to as 'Increase Late Summer Flows'. Incorporation of this qualitative, yet spatially distributed alternative requires a variety of assumptions. The proposed model corresponding to late summer flow increases is similar to that outlined for wetlands, and can be described as follows:

$$Q_{\text{altern}} = k_3 + Q \quad (5.12)$$

where k_3 is, in this case, a rate of discharge increase (m^3/s). The user specifies the value of k_3 and the time period over which that water is added to the network and the DSS specifies the location where the input occurs. It is assumed that the volume of flow augmentation is derived from deep groundwater sources, and therefore does not affect the overall model mass balance. The rate is added to those streams identified by the DSS as receiving late summer flow improvement, and all other routing algorithms proceed without change.

5.4.6 Stream Buffer Model

The effects of stream buffers on in-stream water quantity and quality are characterized by a set of complex interactions. These interactions involve upland landuse, land surface slopes and soils types, the physical features defining the buffer, as well as local hydrologic conditions [Phillips, 1989; Muscutt et al., 1993]. Buffers tend to decrease the concentrations of in-sediment through

increased roughness, entrapment, and bank protection [Peterjohn, 1984]. This is used to implement a simple local model of the effect of buffers on erosion:

$$A_{altern} = A * k_4 \quad (5.13)$$

Where A is the erosion rate (mass/area time) from the model unit and k_4 is a user defined reduction coefficient representing the capability of the buffer to trap sediment and reduce bank erosion. The use of a coefficient allows for the exploration of the cumulative affects of watershed restoration, given assumptions about the quality of individual sites. In addition to the coefficient k_4 , the fraction of the total stream bank receiving riparian buffer is also necessary for operation. This value is used in the calculation of the length of each reach that is subject to restoration:

$$L_a = \frac{A_a}{(L_t * W)} \quad (5.13a)$$

where L_a is percent of the total reach length that is buffered by RESTORE, A_a is the area of buffer in each RESTORE polygon, and is given by RESTORE, L_t is the total length of each reach, and W is a user defined parameter indicated the width of the assumed buffer. This equation is necessary because the hydrologic model does not use directly every polygon making up the DSS derived scenarios. In all other cases, simple areal averaging is used to convert from polygons (the structural unit on which DSS decision are made) to model structures. This additional step is necessary here because the DSS, in this case, makes assumptions about the area of prescribed buffers and that area is not the same as

the total area of each polygon. Additional information regarding the different datasets used by each of these components is provided in Appendix A.1.

Determination of the value of parameter k_4 is difficult, with a wide range of results reported in the literature [Peterjohn, 1984; Dillaha et al., 1988; Dillaha et al., 1989; Magette et al., 1989]. To incorporate the uncertainty that characterizes this value, we allow it to range, and base it on best guesses. Clearly these assumptions regarding the site scale effectiveness are not necessarily 'correct', but again we argue that they are appropriate for watershed-scale analyses of possible future scenarios.

5.4.7 Tile Drainage Model

Tile drains are commonly used in agricultural lands, and the watersheds in the Willamette Valley are no exception. Tile drains effectively act to increase the hydraulic conductivity of watersheds, in much the same way as other preferential flow pathways. A variety of studies have evaluated the effects of tile drains on hydrology and solute transport. [Richard and Steenhuis, 1988; Booltink, 1995; Shalit and Steenhuis, 1996; Singh et al., 1996]. From a modeling context, a variety of studies have been developed, and generally focus on detailed field-scale modeling of well-defined tile drain systems [Khan and Rushton, 1996; Munster et al., 1996; Singh et al., 1996]. The increasing uncertainty of model predictions at regional scales has been established [Mohanty et al., 1998]. Additionally, they

note that site specific findings have some potential to be included in regional scale models, but focus their research on a 24 ha farm, and do not make statements at larger watershed scales.

Here we propose a model of tile drainage applicable at the watershed scale over which this study focuses. The model is an extension of recently proposed ideas from the hillslope hydrology literature, where conceptual box models are being used to more explicitly simulate a widely variable flowpath distribution. [Seibert and McDonnell, 2002; Uhlenbrook and Leibundgut, 2002]. The model is based on a heuristic understanding of how tile drains act in the environment, and specific information on location, description and functioning of local tile drain systems is not incorporated. In lieu of this information, we make the following assumptions.

- Agricultural lands are tiled drained and these tile drains systems effectively reduce the local soil water content.
- Tile drain systems 'turn on' only under relatively wet conditions

To incorporate these assumptions, an additional tile drain flow path is included for Upslope units that include agricultural landuse. The path way is defined by the following equation:

$$T = k_5 * (\theta - \theta_t) \quad \text{if } \theta > \theta_t \quad (5.14)$$

$$T = 0 \quad \text{if } \theta < \theta_t \quad (5.15)$$

where T is the tile drainage rate (m/d), k_5 is tile drainage rate coefficient (m/d), and θ_t is the unitless wetness threshold over which the tile drains activate.

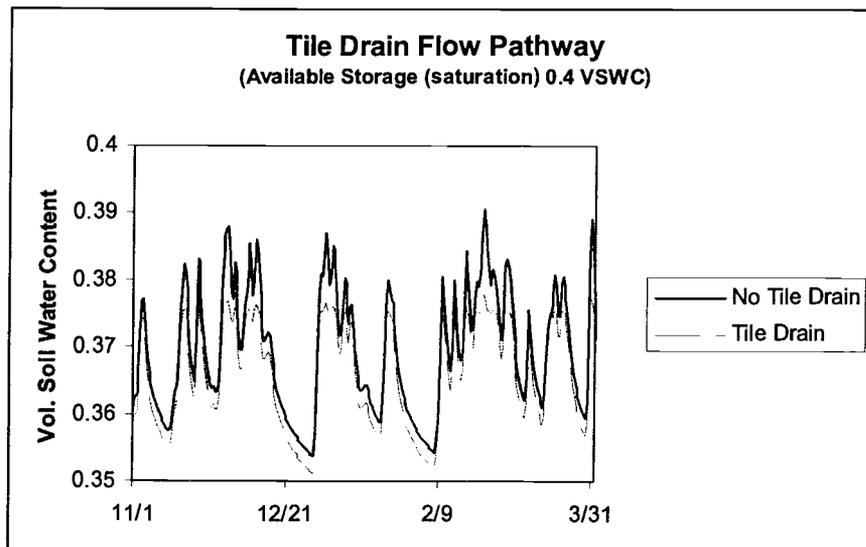


Figure 5.1. Results derived from the tile drain model.

Figure 5.1 provides an indication of the effect of these equations on soil water content. Without implementation of equations 5.14 and 5.15, the two series would be equivalent. It is shown that the model effectively reduces soil water content during relatively wet periods.

5.5 Site Description

Bear Creek is a 74.2 km² subbasin of the Long Tom, and Willamette Rivers, near Eugene, Oregon. Elevations in the basin range from 92 to 525 m, with a predominance of agriculture in the lower elevation regions. The higher elevation regions are mostly managed timberlands, and include a well-established road network designed in support of timber operations. The area has a maritime climate, typical of the Willamette Valley, with cool wet winters and relatively dry warm summers. Snowfall occurs infrequently in the lowlands, and with a maximum elevation of 525 m, snowpack does not persist between storms. Water quality is a concern in the region, with 91 km of the Long Tom River listed under the 1998 Oregon Department of Environmental Quality (ODEQ) 303(d) list of impaired Oregon waterways.

5.6 Results and Discussion

Applications are focused on demonstration of the capacity of the models to simulate the cumulative effects of restoration activity on in-stream water quality. Two sets of DSS derived scenarios were developed. Set 1 focuses solely on the implementation of riparian buffers. Different configurations of buffers are developed and the effects of these configurations on sediment yields are

compared. The second set relies directly on the DSS for inputs. It is designed to evaluate the effectiveness of the rule-based DSS at developing scenarios that accomplish the goals of the user. Table 5.2 lists the values of the major coefficients introduced in this chapter. In addition, note that the sediment routing model outlined by equation 5.9 was utilized in all simulations.

5.6.1 Buffer Analysis

The analysis of buffer scenarios provides an indication of how the system employs simple models across large areas to provide estimates of the cumulative effects of restoration. Here we focus only on buffers in an effort provide an indication of how a single restoration strategy modifies model results - as more restoration strategies are included in the simulations, the effect of individual strategies becomes more difficult to interpret. Four separate coverages representing different buffer configurations were generated. The rules used to generate these landscapes are outlined in table 5.3. As an example, the Upland Buffer scenario was developed by implementing stream buffers on all first order reaches for all landuse categories. Figures 5.2 and 5.3 present a set of maps depicting the results of the design rules outlined in table 5.3.

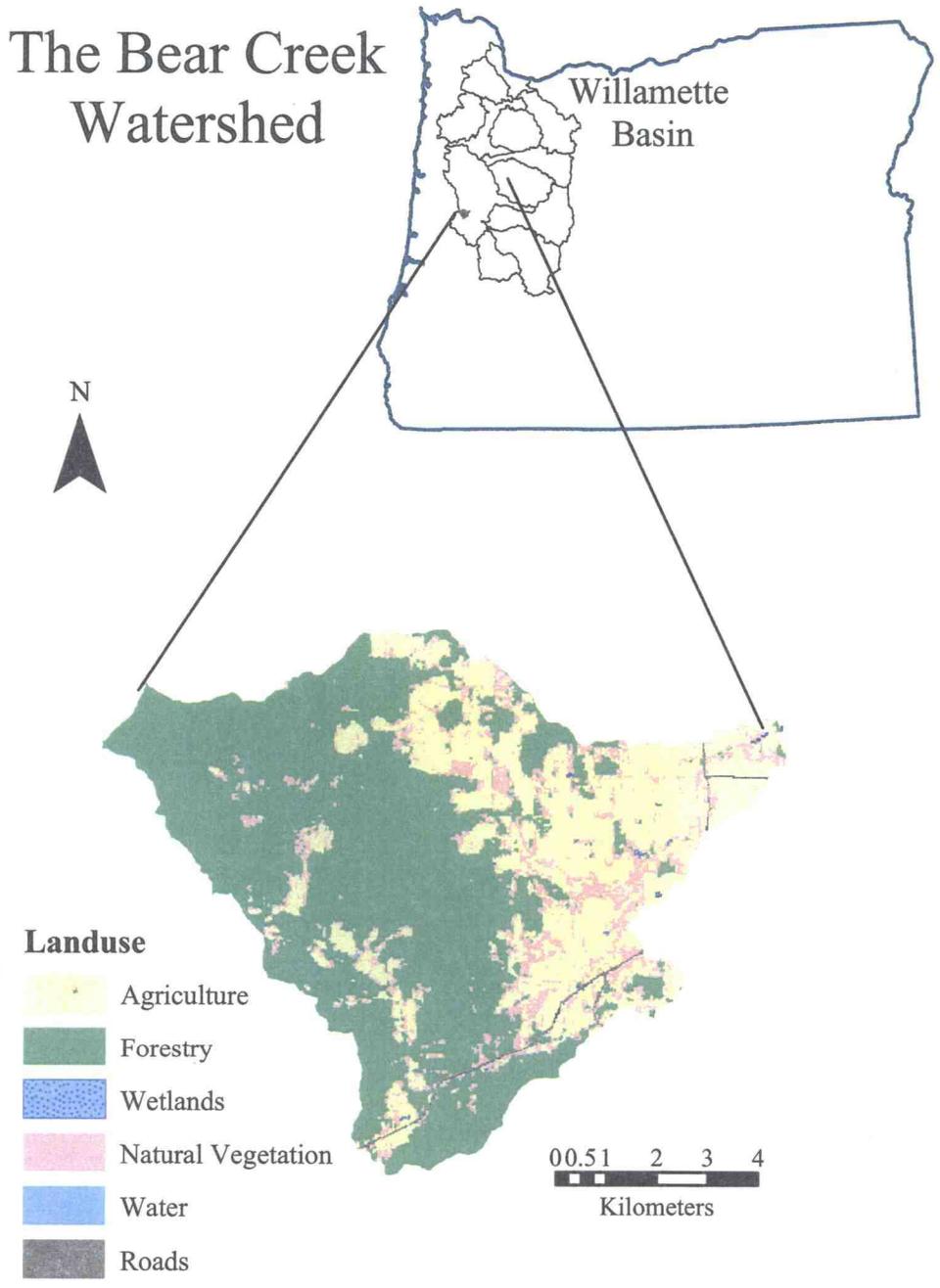


Figure 5.2. Location of the Bear Creek watershed. The map is classified on a coarse description of landuse.

Table 5.2. Landuse change model coefficients. The values of k_4 were allowed to range from 0.5 to 0.35 in some analyses as noted in section 4.4.1.

| <i>Landuse Parameter</i> | <i>Associated Model</i> | <i>Equation number</i> | <i>Value</i> |
|--------------------------|-------------------------|------------------------|--------------|
| k_1 | Wetland | 5.10 | 0.8 |
| k_2 | Wetland | 5.11 | 0.8 |
| k_3 | Flow Augmentation | 5.12 | 0.00005 |
| k_4 | Buffer | 5.13 | 0.20 |
| k_5 | Tile Drain | 5.14 | 2 |
| θ_t | Tile Drain | 5.14 | 30 |

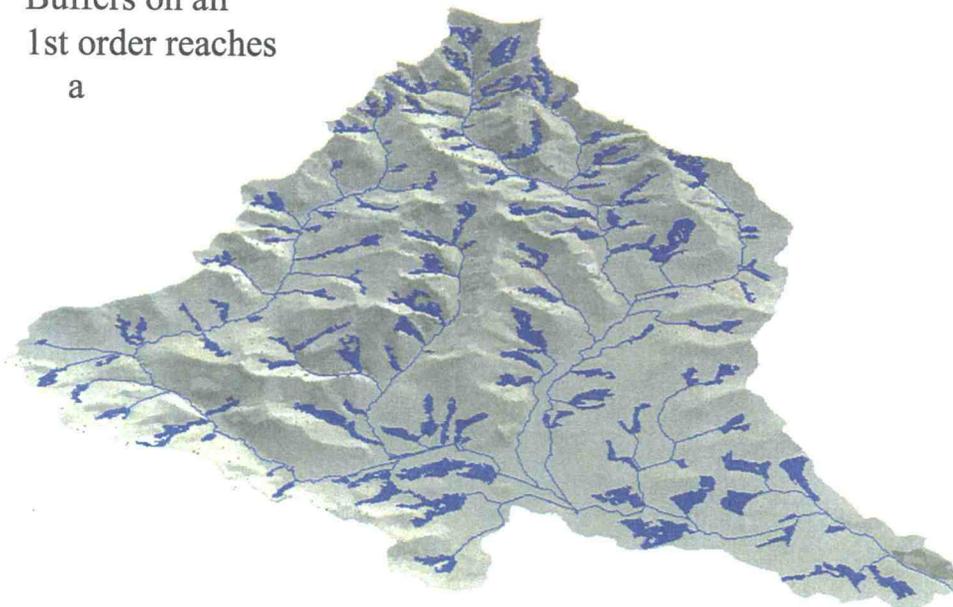
Table 5.3. Design rules for buffer scenarios.

| <i>Scenario</i> | <i>Stream Order</i> | <i>Landuse</i> |
|---------------------|---------------------|----------------|
| Upland Buffer | 1 | ALL |
| Complete Buffer | ALL | ALL |
| Lowland Buffer | ≥ 2 | ALL |
| Agricultural Buffer | ALL | Agriculture |

The watershed was modeled for January of 1994. Percent change comparisons were made using model results of a simulation with no restoration alternatives as the baseline. Data to derive the statistics represented by the box and whisker plots in Figure 5.4 were developed through multiple simulation runs with different values of k_4 in equation 5.13. These figures provide a basic idea of how uncertainty in buffer potential manifests itself in our chosen output. Values of k_4 were allowed to range from 5 to 35 percent and provide an indication of how

uncertainty in the capacity of a buffer to reduce sediment loading manifests itself in the results. Not surprisingly, the complete buffer coverage produces the greatest sediment output reduction. Buffers along all agricultural stream reaches (figure 5.4, bottom) provide relatively large sediment reductions and require the least amount of buffer conversion.

Buffers on all
1st order reaches
a



Buffers on all
2nd order reaches
b

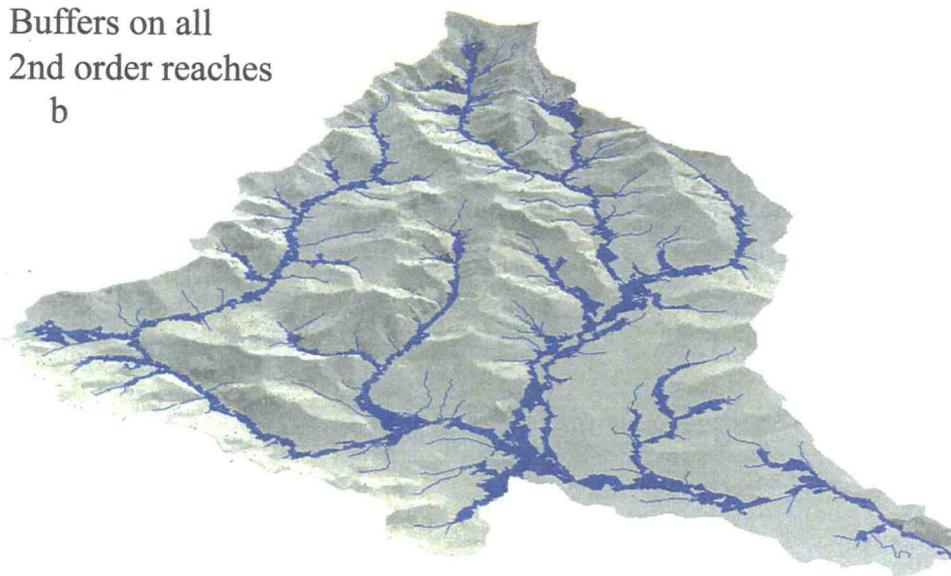
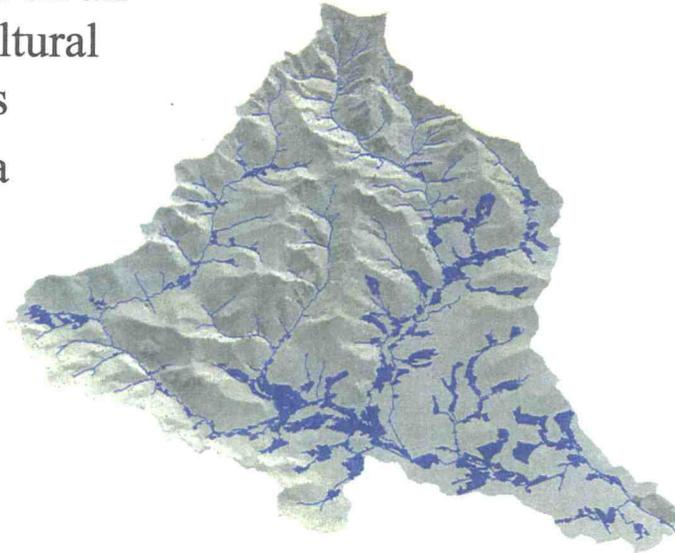


Figure 5.3. a) Results from buffers on all 1st order reaches. b) results from buffers on all 2nd order reaches. The blue color represents the polygons treated by the model as containing near stream buffers. The vertical exaggeration is 1.5 times.

Buffers on all
Agricultural
reaches

a



Buffers on all
reaches

b

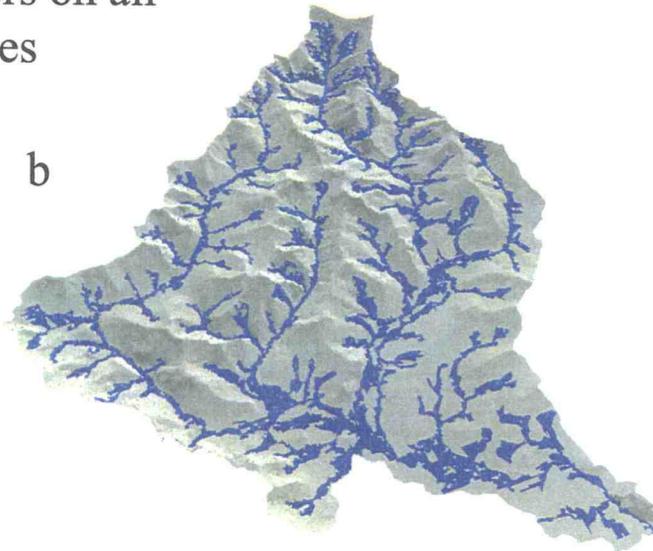


Figure 5.4. a) Results from buffers on all reaches draining agricultural lands. b) results from buffers on all 2nd order and higher reaches. The blue color represents the polygons treated by the model as containing near stream buffers. The vertical exaggeration is 1.5 times.

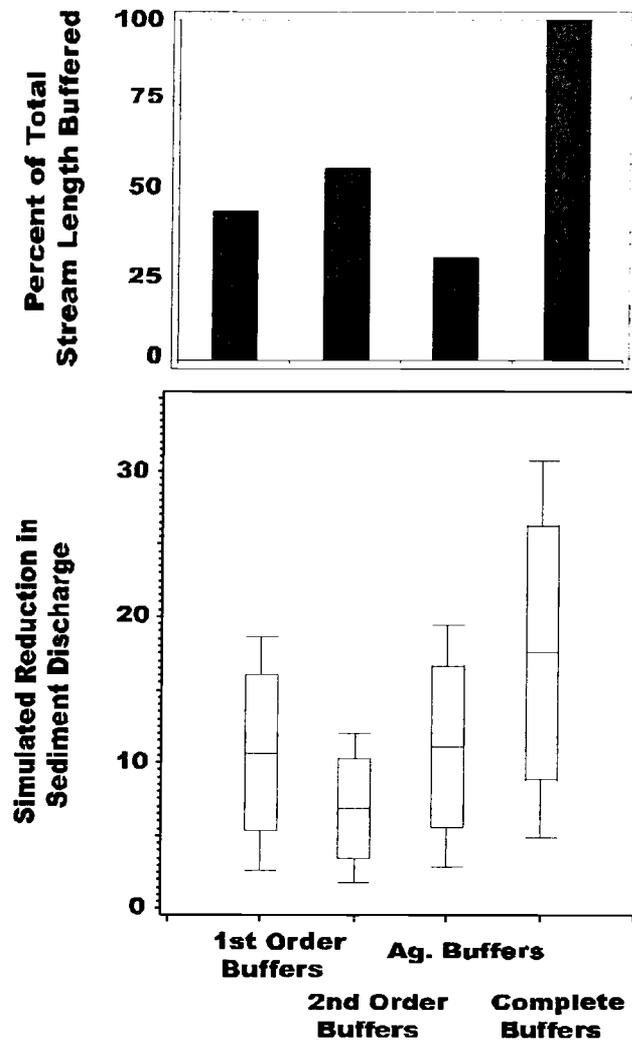


Figure 5.5. Results from each of the four buffer scenarios using a non-buffered landscape as the baseline from which reductions are calculated.

This is due to the greater erosion potential (as characterized by the USLE) of agricultural lands. The results also indicate that restoration in headwater basins, characterized here by the 1st order landscape, has potential for greater reductions in sediment export than restoration in higher order reaches. In addition, the total

length of 1st order streams is smaller than the total length of 2nd order streams (figure 5.4, top box). In other words, greater reduction potential exists for a buffer configuration which, in this case, requires considerably less area. The explanation appears to be that 1st order basins contribute a disproportionate amount of stream flow to the overall hydrograph. The spatially explicit volume based simulation provides the means to demonstrate this, and results such as these indicate the utility of the approach.

5.6.2 DSS derived scenarios

RESTORE provides users the opportunity to focus restoration simulations based on user-defined goals and the objectives of restoration. We developed a series of scenarios utilizing the goal driven multi-attribute decision maker implemented in the DSS, and then evaluated these scenarios using the simulation model. This section demonstrates the types of results available through this combination of simulation and decision support. Figures 5.5 and 5.6 represent the DSS generated restoration plans, given different watershed objectives. These maps are quite useful in conveying the kinds of restoration suited to the stated objectives and the spatial dependence of those restoration options. For instance, note that given a water quality objective, RESTORE suggests wetland restoration is useful across a wide area, but that this area tends to be located in lower elevation regions of the basin. These maps clearly contain a significant amount of information, but it

remains a challenge to interpret on a quantitative level how they each might affect water quality, and how they might compare with one another. Simulation models provide one means of developing an improved understanding of these landscapes. Here we focus on endpoint measures of sediment export and late summer discharge, but note that the simulation framework provides values of these state variables at each point in time and in space.

Objective:
General Restoration

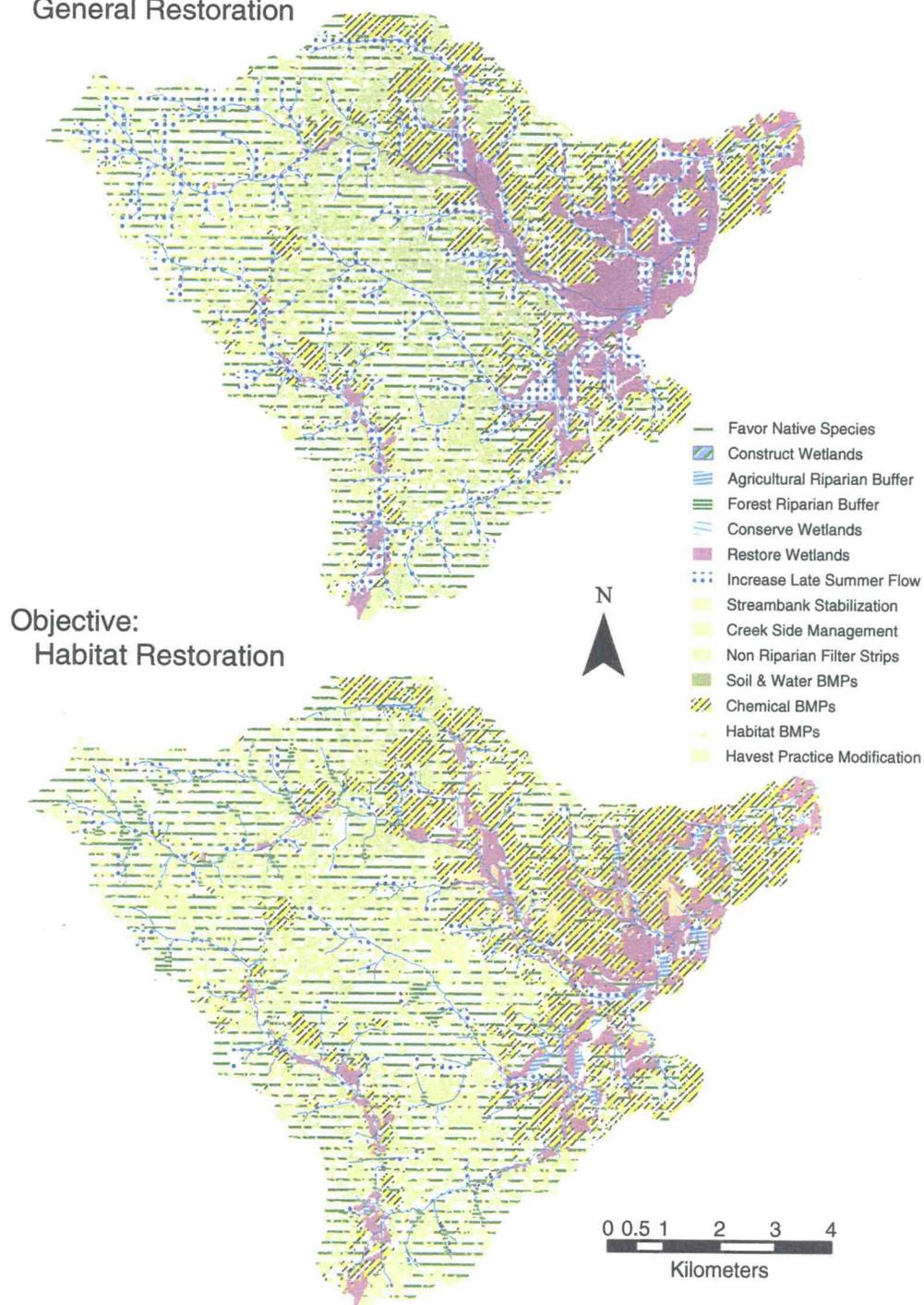
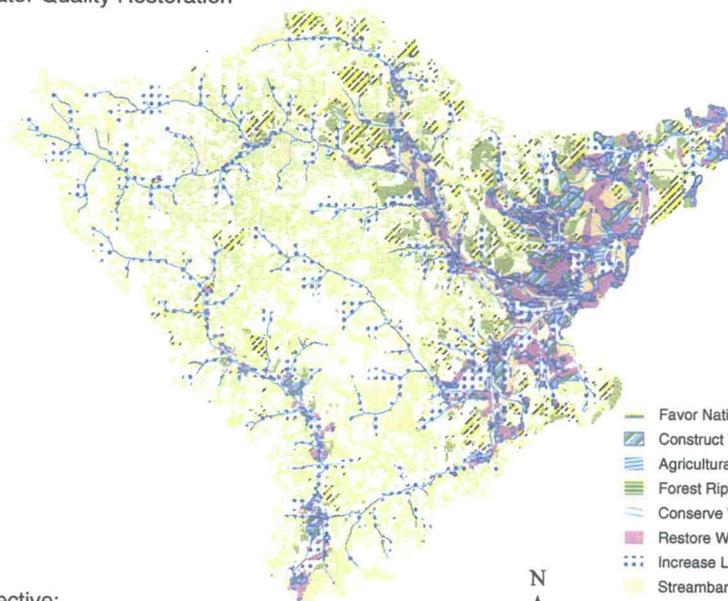
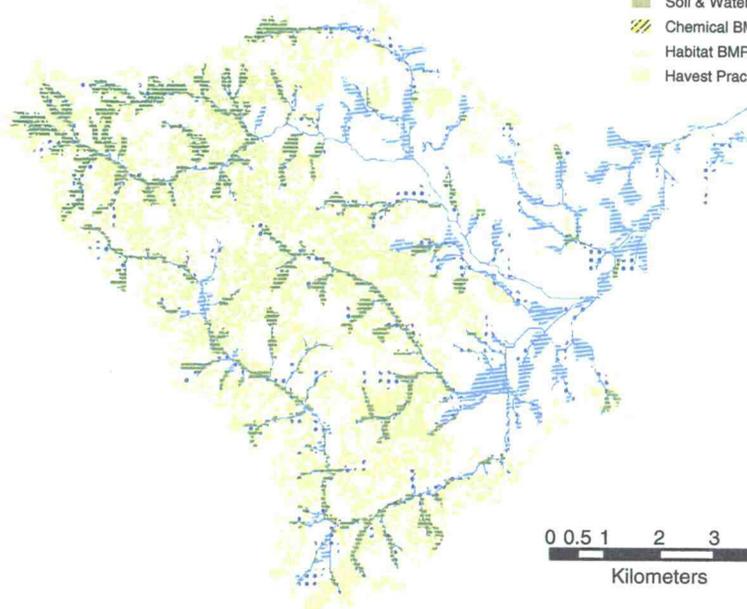


Figure 5.6. A Map depicting the general and habitat focus restoration plans derived by RESTORE. Part a is the general plan and part b is the habitat plan.

Objective:
Water Quality Restoration



Objective:
Temperature Restoration



- Favor Native Species
- ▨ Construct Wetlands
- ▨ Agricultural Riparian Buffer
- ▨ Forest Riparian Buffer
- ▨ Conserve Wetlands
- ▨ Restore Wetlands
- ⋯ Increase Late Summer Flow
- ▨ Streambank Stabilization
- ▨ Creek Side Management
- ▨ Non Riparian Filter Strips
- ▨ Soil & Water BMPs
- ▨ Chemical BMPs
- ▨ Habitat BMPs
- ▨ Harvest Practice Modification



0 0.5 1 2 3 4
Kilometers

Figure 5.7. A map depicting the water quality and temperature focused restoration plans derived by RESTORE. Part a is the water quality plan and part b is the temperature plan

Two simulation periods were chosen for examination, each designed to be somewhat typical of Oregon weather. The first is representative of winter conditions, when discharge, and consequently sediment export, tend to be elevated. We developed these simulations using January 1994 as employed in section 4.4.1. The second set of simulations utilized data from May 1, 1994 to October 1, 1994 and is representative of the dry, low flow conditions often associated with water quality limitation. Model results indicate that the four scenarios have potential to reduce sediment export by anywhere from approximately 8 to 35 %, depending upon the scenario and the period of time over which simulations occur (figure 5.10). This suggests that scenarios developed under different objectives result in significantly different outputs.

Comparisons among the scenarios suggest that the rules defining the habitat objective produce the largest decrease in export. We attribute this result to the large areal extent of wetland restoration implemented under the habitat objective (see figure 5.5, bottom). The temperature objective produces the smallest reductions - a smaller amount of the landscape is 'restored' under the objective.

Sediment export is most directly a water quality concern, so it seems reasonable to assume that a water quality objective would result in the largest reductions. However, results indicate that the objective produced a landscape that appears, in comparison with others, modest in its sediment reduction capacity. This apparent discrepancy suggests very strongly one of the reasons for not only

including simulation models in DSSs, but using those models to iteratively inform the landscape generation components. The endpoint information available from simulation is of a different variety than that available to a rule based DSS as general input (broadly, this includes spatially distributed data used to describe the physical landscape and the rules that are based on that data). Both pieces of information are valuable individually, but feedback from simulations might be used to further refine rule sets, and ultimately produce derived landscapes that satisfy objectives at both the site and watershed scale. These sediment export results indicate this potential quite clearly.

Clearly, the model used to simulate the effect of individual buffers (equation 5.13) is a gross simplification of the processes occurring within the buffered area. Given this fact, it is worth reiterating here the utility of this model. The magnitude of the model results is by and large controlled by two important factors. The first is the reduction potential of individual restoration sites. Here we are suggesting that the uncertainty surrounding sites precludes the use of a more detailed simulation of site level change. In lieu of that we suggest the use of simple heuristic models. But, this means the magnitude of the reduction is largely controlled by our assumptions, and so the focus of interpretation of these results should not be that magnitude of reduction. Rather, interpretations focus on the second controlling factor – the number, location, and type of restoration activities implemented across watershed-scale basins. In this case, we are interested in the differences between scenarios. The value of this modeling exercise then, is to

provide planners and stakeholders with methods to evaluate different watershed scale restoration plans for basins, providing one piece of important decision-making material.

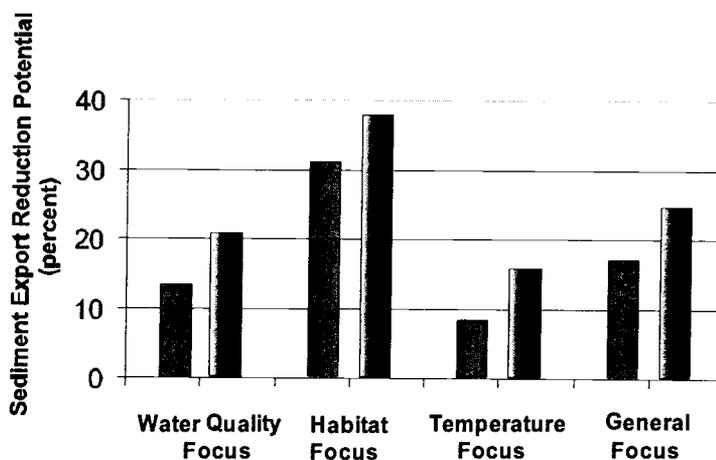


Figure 5.8. A histogram representing change in sediment discharge simulated as a result of each of the four DSS derived scenarios, for each objective. The solid gray bars represent summer time reduction potential, the mixed bars represent winter reductions.

The results of the summer time simulations (Figure 5.9) indicate that each of the scenarios did effectively increase discharge, although in this case, distinguishing among the scenarios is difficult. The reason for the similarity appears to be that the total number of reaches receiving late summer flow augmentation, in each of the scenarios, is very similar (Table 5.4). For each of these reaches, discharge is increased by a constant rate, during the period of concern. A next step might include the inclusion of additional water uses

(including treatment of irrigation and domestic and municipal water use), and also the evaluation of a wider variety of scenarios with a larger range in the number of streams receiving late summer flow increases.

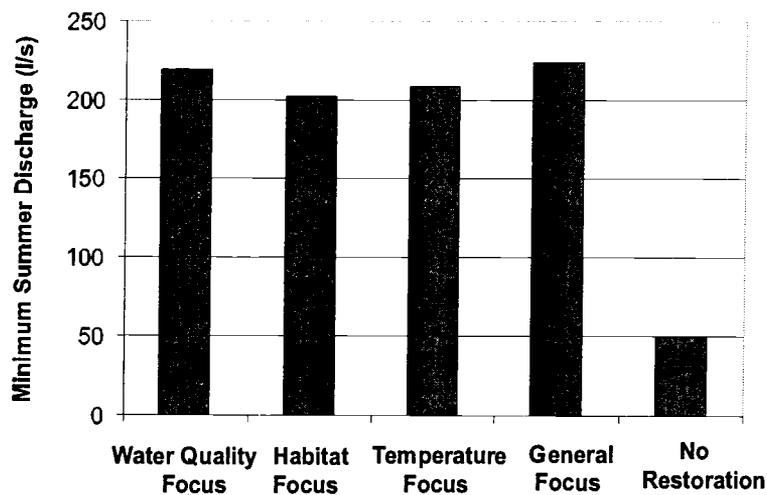


Figure 5.9. The minimum summer time discharge from Bear Creek simulated for each of the four RESTORE derived scenarios and for the current landscape, indicated as 'No Restoration'.

Table 5.4. Number of reaches receiving late summer flow increases for each of the four RESTORE derived scenarios

| Scenario | Number of reaches |
|---------------|-------------------|
| Water Quality | 25 |
| Temperature | 24 |
| Habitat | 23 |
| General Focus | 26 |

The more direct utilization of this information has potential to improve simulations of late summer discharge, and show more meaningful differences between different landscape designs. This information might allow for better approximations of appropriate values of k_3 , the model parameter indicating the numeric affect of the 'improve late summer discharge' alternative. Here we simply increased discharge by 0.5 liters/second at each location, and these results suggest that may be too simple a model.

5.7 Conclusions

In this study we provide a set of tools designed to explore, through application, the utility of linkages between decision support, distributed landuse scenarios, and simulation modeling. We demonstrate that model analysis of DSS derived scenarios can provide useful summaries of the effects of the proposed changes. In addition we have found that the ability to dynamically develop new scenarios using the expert guidance facilitated by RESTORE is a significant improvement over more standard procedures where a single set of static scenarios are evaluated.

An area of further research that will build upon this work is in landscape optimization, where the DSS and simulation model interact more directly using the summarizations provided by the model to direct the analysis implemented by the DSS. The result of this interaction would be a landscape scenario that was optimized with respect to the model outputs and the DSS derived objectives and goals. These types of procedures also have the potential to more fully characterize the range of results that might be expected from the DSS, and how these in turn affect simulation results. This process has potential to more fully characterize the uncertainty associated with restoration, and it is precisely this information that may be of the most use to decision makers.

A more thorough characterization of uncertainty is also warranted. Here we suggest a framework that relies on single parameter assumptions of site scale restoration effectiveness. We have shown that, along with multiple model runs, this has potential to provide some understanding of endpoint uncertainty. But including additional parameters and random sampling *within* model runs would be necessary to make full use of the ideas.

Additionally, the set of state variables and landuse practices defined in this paper is relatively small. Stream temperature, pesticides, nutrients as well as other variables, are of interest to watershed users and planners, and the analysis of these components is a potentially useful addition. Furthermore, other land covers and practices have the potential to influence any of the state variables mentioned

in the paper, sediment and water quantity included. We chose a subset to provide data necessary to develop and analyze the model, but encourage development of model sensitivity to other factors.

Along these lines, the work stands to benefit from inclusion in a component based modeling framework. Component based simulation provides an interface to the model structure, and has potential to greatly facilitate the process of model development. This includes simplification of process necessary to add different state variables and rate equations, more sophisticated interface generated model summaries, and potentially, an optimization framework as outlined above.

6.0 Conclusions

This thesis is focused on the development and implementation of tools designed to improve characterizations of watershed scale landscapes and the potential for restoration. Chapter 2 centers around the application of a standard simulation model to a set of thoughtful, yet static, landscape scenarios. Subsequent chapters were developed in part based on the experience gained from the research described in Chapter 2. The focus of these following chapters ranges considerably, but a common theme is the development of new techniques designed to improve both the technical and scientific basis for landuse planning related to water concerns. In part, this is a novel venture, because much of the recent work focused on advancing hydrologic modeling has not been incorporated into tools that are of use to the planning community.

The initial portions of the work (Chapter 2) focus on the implementation of a standard, watershed-scale model simulating landscape change measured as changes in sediment and nitrogen export. Data describing two watersheds, located in central Iowa and considered representative of the Corn Belt Region, are used as input for the modeling work. The chapter demonstrates the utility of future scenarios and the information that can be developed from alternative landscapes through simulation modeling. The most compelling finding from the work is that very significant changes, beyond standard BMP applications, will be necessary to induce relatively modest (10 – 50 percent) reductions in high non-

point source pollution export that characterizes much of the Midwest. Beyond this though, the study represents one of the first attempts to coordinate a set of highly planned watershed-scale restoration scenarios and simulation. This procedure represents one approach to integrate more fully the realms of policy and of science, but this avenue of research remains in its infancy.

The process of informing the planning process through scenario development is still quite rough. While part of this can be explained by the wide variation in planning goals and objectives, the continuing need to establish a consistent framework, specifying for instance available options and the methods to place them and evaluate the resulting landscapes, also contributes. To some degree, the lack of appropriate tools designed to facilitate both the development of potential landscapes and the simulation of them contributes to the difficulty of establishing a consistent method. In addition, a lack of quantitative understanding of uncertainty in the modeling process also plays a role. The bulk of this thesis is designed to address these issues, and while we do not 'solve' them, the work does suggest a plausible path with which to better inform the planning process.

Chapters 3 – 5 were developed in an effort to provide alternatives to the current state of the art (SWAT) scenario based planning studies. The following areas of improvement were identified as the development of Chapter 2 progressed. (1) The process of working with SWAT was challenging and suggested the need for technical improvements to models designed for use by planners. (2) SWAT is clearly overparameterized, meaning there is not enough information in measured

datasets to identify the parameter values that make up a large portion of the model. Solutions to watershed scale problems with a focus on parameter minimization would appear to be of some utility because as the dimensions of parameter space increase, understanding, even conceptually, the uncertainty becomes increasingly difficult. (3) Generation of the landscape scenarios was a time-consuming process that resulted in three static maps of the potential future landscape. Linkage with a more sophisticated DSS, where scenario generation might be responsive to both watershed goals and potentially, model results, was identified as a potentially useful improvement.

The third chapter lays the technical groundwork for the model. The binary tree is identified as a concise data structure with the capability to maintain the topology, or spatial relationships of data, necessary to store information in way that is useful to distributed models. A detailed description of a unique version of the binary tree that can represent all standard network datasets was provided. Included in this description is a concept of phantom nodes which allows for a binary definition of general trees. Synthetic data are processed through a realistic stream network using the tree and a solution to the kinematic wave equation. This example application demonstrates the utility of the structure and the routines to utilize it.

Chapter 4 introduces the model WET_Hydro, designed to incorporate many recently established ideas, centered on uncertainty, regarding hydrologic modeling. Chapter 4 was intended to place the work within these recent ideas and

to establish its functionality as a model of hydrologic processes. The paper is composed of three major sections, the first two entailing model description and application. The third focuses on the development of model-derived criteria used to provide increased understanding of model operation. Model applications at a wide variety of sites and model scales were established. This range of areas provide a better test than application at a single location because, in this case, the goal is a general model that might be applied with a minimum of data in locations with disparate hydrology. Not all together surprising, the model performance was very high at Maimai, where rainfall and runoff are very much related and input data are well constrained. But model performance, as quantified through discharge, was quite reasonable for all locations, even given the wide variation in the temporal scale of inputs, the spatial scale of model operation, and known differences in the dominant hydrologic processes among sites. This suggests that there is significant utility in distributed, conceptual, and yet physically based models.

The chapter continues by introducing the utility of hydrograph separations, which provide a degree of understanding regarding the internal processing of the model. These analyses focused on the San Jose basin, where the new water contribution to stream flow is considered to be quite high. The inclusion of percent new water in the same Monte Carlo simulations used to explore parameter space and identify highest efficiencies was shown to significantly reduce the acceptable parameter space. Of course, the efficiency also was reduced through

these analyses, but this highlights one of the key concerns related to hydrologic models, namely that the model structure and parameter vector that makes up the highest efficiency using discharge data may be quite inefficient in regards to the internal processing of water. If simulated discharge is the only concern, then this is not a problem. But in that case, much simpler lumped models may produce equally efficient solutions and may do so more quickly and with fewer input requirements.

The last portion of Chapter 4 begins by establishing, through simulations at Maimai, the effect of model scale (or resolution) on efficiency and parameter space. At this location the results of the model (defined as the static structure and parameters) are shown to be scale dependant, but discharge-based efficiency is not. This adds further evidence suggesting that alternative evaluation criteria for models should be established. In this case, the need arises because discharge data provide no guidance for the development of model resolution. Certainly, most decisions regarding model resolution have not been made using discharge data, and in fact are most often established on a purely operational basis – often using the scale of the input data or in the case of a semi distributed simulation, a configuration of subwatersheds. But, as the effects of scale are better understood, methods which provide some thoughtful suggestion of appropriate choices will be useful. Here we suggest that measures other than discharge, along with Monte Carlo simulations across model scales, may have some utility in that regard. The measured data we suggest include the qualitative and informal ‘sticky boot test’,

where a hydrologist may walk portions of a watershed to develop a feel for the degree of saturation across different areas. The model value with which this corresponds might be a simple summary, over each simulation, of the percent of time or space that was saturated during the model run. This modeled output is shown to be scale dependent, and as such could be used to help guide the model resolution which best captures watershed dynamics.

A variety of questions remain to be answered in regards to the hydrologic components of this work. Applications to areas represented by the San Jose basin have significant potential for improvement with the inclusion of an infiltration excess overland flow mechanism and routing of the overland flow component. Along these same lines, a snow routine has the potential to improve the quality of winter season simulations – this deficiency is apparent in the results from Schaefer and, to a somewhat lesser degree, Wiley Creek. From a verification standpoint, it would be useful to apply the model to a relatively large basin with distributed parameters and evaluations of performance at multiple points, including time series data representing upland soil water content. We did have the opportunity in the San Jose to calibrate to three nested discharge sites, but used a single site for verification and did not include upland data. This idea is very much related the assertion that multiple criteria, including hard data, like discharge and piezometer traces, as well as more qualitative information collected through field experiences, should be made more standard components of hydrologic simulations. We have suggested some model-derived quantities and

corresponding field measures as additional criteria, but the development of methods to formalize their use is only just now beginning.

In Chapter 5 the research returns to the theme of landuse change and water quality. A modified version of the USLE is implemented, using the distributed estimates of water movement to represent the erosivity factor. This basic model is used to produce estimates of sediment movement which are moved through the system and translated into values representing watershed sediment export. Simple models relating site level landuse and land practice characteristics to site level erosion reductions were then developed and implemented. The watershed model then is used to establish, for a variety of different landscape scenarios, the cumulative restoration potential of the distributed site level models. Given this approach, no information can be developed from the model regarding the site level effectiveness of restoration. But given the known wide range in restoration potential of sites, and the project's focus on watershed scales, we are comfortable leaving those analyses to field-based studies. The approach does provide an opportunity to evaluate, given a range of plausible restoration potential, the overall effects representative of watershed scale restoration. It also facilitates the process of developing an idea of how uncertainty in the potential of sites to improve water quality manifests itself in the endpoint measures.

The model is coordinated with the RESTORE DSS to evaluate a series of rule-based landscape plans. A variety of insights are derived from these analyses, not the least of which is the documentation of the fact that the rules making up

many of the objectives have unintended consequences. The clearest example is that landscape scenarios derived with a habitat improvement objective are simulated to have the highest erosion reduction potential of any of the other objectives, including water quality. The major benefit of this coordination then, is the idea that the model can guide the site level focused rule engine. Considerable work remains in the development of more dynamic interactions between models and the rule engine. It would seem that the most useful target for this work (we are not there now) might be in landscape optimization through automatic landscape generation and evaluation with direct feedback between the two.. In this case, the model (or models) would provide the kinds of analyses we have developed in this paper, along with uncertainty, back to the landscape generator, which would then use that information to reevaluate the interaction between the rules and landscape data. This process would produce a new landscape more likely to satisfy objectives at the watershed scale, then the evaluations would again occur and the process would begin again. The results presented here are relatively static, but clearly suggest the feasibility of these ideas.

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Appendix A.1. Development of a vector based distributed hydrologic model

Wet_Hydro is a hydrologic modeling system that flexibly incorporates data and process descriptions of hydrologic fluxes. Data sources include both spatially-distributed, lumped and point data. Estimates of vertical water fluxes are used to maintain a distributed soil water balance and heuristic watershed scale algorithms define horizontal soil water fluxes. Simulations involving large areas, where stream residence times become significant can include a kinematic river routing algorithm. The model was developed using C++ and includes an interface designed to assist with input data development and interpretation and presentation of model results. See figure A1.

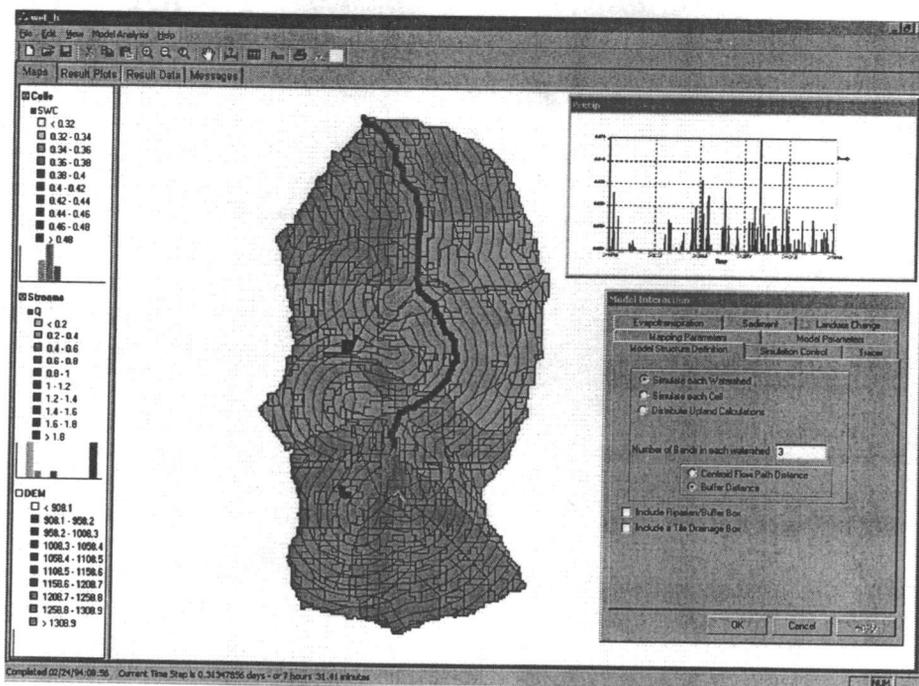


Figure A.1. The WET_Hydro model interface.

The model relies upon its distributed nature to simulate varying flowpath distances and travel times. GIS datasets are used to define a digital landscape of spatially distributed homogeneous land areas and these data fall into two categories. Here we describe briefly the data structures and concepts necessary to support the vector based datasets describing the environment. The first category consists of pre-processed information provided to the model prior to a simulation. This includes spatial data representing assumed known properties of the landscape such as soil properties and land surface slopes. The model derives the second category, using the data provided by the first. It is worth noting that under WET_Hydro, the landscape is redundantly represented in three separate, broadly defined, datasets. The redundant representation of space is necessary for three major reasons. First, proprietary GIS systems, and their data formats, are necessary because of the complex, well-tested methods they have implemented. Consider the process of overlaying two spatially explicit vector datasets to produce a single dataset which incorporates all of the information. The process is quite complicated and proprietary GISs exist that accomplish the task. Implementation of this type of functionality withing WET_Hydro was unnecessary. Despite their clear utility, proprietary GIS systems do not perform all the tasks necessary for the model, *and* they require users purchase separate and potentially expensive software. For these reasons it was important that we separated the model from standard GIS systems. Because of this separation, the

model requires its own definition of spatial data. And a last representation of spatial data is necessary because visualization of spatial data and efficient numerical simulation are two very different goals for computer software. To accomplish both, separate data representations are necessary.

A.1.1 Preprocessed Data Sets

Elevation, soils, landuse, streams, and watersheds comprise the input data that must be preprocessed. In addition, the model can make use of spatially distributed estimates of rainfall and temperature. Wet_Hydro reads, as initial input, two data formats. The first is a spatial data format developed by Environmental Systems Research Institute (ESRI) known as a shapefile. The second format is common ASCII gridded data. Shapefiles represent a non-topological, openly published representation of spatial data. Development of input data for WET_Hydro can occur in any Geographic Information System that can produce shapefiles, although to date all preprocessing has occurred under the ESRI ArcGIS 8.0 software. WET_Hydro requires two separate input shapefiles. The first is polygon file that represents the land area of interest. The second is a line file representing the stream system. The required ASCII grid data represents a Digital Elevation Model (DEM). The model can also use ASCII grids representing flow directions and, but these grids are not necessary for operation.

The model defines a stream segment as the length of stream between any two stream junctions, and requires explicit definition of the watershed draining into each stream, as a polygon coverage that becomes part of the composite upland shapefile. Each upland polygon (or watershed) must be tied to its corresponding reach through the tables of data that are part of the shapefile designation. Additional upland data including landuse, soils, meteorological data, and distance to the stream can also be specified, and included through additional overlay processes, in the shapefile representing upland areas. This shapefile is made up of a series of homogenous polygons that represent all of the spatial information available to the model. It is important to note that no attempt, beyond the development of watersheds, is made to define these units based on some plausible understanding of their hydrologic relationships specifically. Table A.1.1 lists the data sets used in this study to produce the two necessary shapefiles. Table A.1.2 is a definition of the required fields in the upland and reach coverages. A separate database incorporating some basic meteorological data is also necessary. See table A.1.3.

Table A.1.1 A listing of the spatially explicit datasets used by WET_Hydro.

| Data Set Name | Source | Format | Required? |
|----------------|---------------------------|------------|-----------|
| DEM | USGS | ASCII grid | Y |
| Flow Direction | Derived | ASCII grid | N |
| Stream Grid | Derived | ASCII grid | N |
| Stream Vector | Derived | Shapefile | Y |
| Landuse | ERC | Shapefile | Y |
| Soils | SUURGO | Shapefile | Y |
| Watersheds | Derived | Shapefile | Y |
| Precipitation | Oregon Climate Service | Shapefile | N |
| Buffers | Derived | Shapefile | N |

Table A.1.2. Input data fields required by the WET_Hydro model.

| Name | Type | Data Set | Preprocess? | Description |
|----------|-------|----------|-------------|---|
| Area | Float | Upland | Y | The area of the polygon in square meters |
| Slope | Float | Both | N | Maximum feature slope, as a ratio. Calculated through development of 3 dimensional shapefiles based on DEM. |
| Depmax | Float | Upland | Y | The maximum depth of soil in the polygon |
| Hydro_id | Int | Both | Y | A id unique to each reach and the polygon within the reach's watershed. |
| Abbrev | char | Upland | Y | Abbreviation for the soil type |
| Lulc_a | Int | Upland | Y | Landuse code |
| Centroid | Float | Upland | N | Space to store the calculated distance from the polygon centroid to the nearest point on the polygon's reach |
| Buffdist | Int | Upland | Y | The buffered distance from the polygon to the reach |
| Side | Int | Upland | N | Space to store the side of the stream on which a polygon exists. 0 = first order, 1 = left side, looking downstream, 2 = right side, looking downstream |
| Ltprec | Float | Upland | Y | The 30 year long term yearly precipitation for the polygon |
| Length | Float | Reach | Y | Length of the reach in meters |
| Order | int | Reach | N | Space to store the calculated stream order |

Table A.1.3. Fields required in the climate database. The file is read as a comma delimited text file. The model can only simulate time which is between the start and end time in this file. No specific time step is required for these data.

| Field | Required? | Description |
|--------|-----------|--|
| Time | Y | The datetime of the data. A floating-point value, measuring days from midnight, 30 December 1899. Corresponds to the Microsoft datetime definition |
| Precip | Y | Precipitation rate over the time interval. Must reported as m/d |
| Tmax | N | Maximum Daily temperature (C) |
| Tmin | N | Minimum Daily temperature (C) |
| Evap | N | Evaporation Rate over the interval. Must be reported as m/d |

The soils database provides information on the spatial location of different soil types. WET_Hydro makes use of these data in a variety of means. One of the most important uses of the soils data are relationships between soil textural class and hydrologically important soil properties. The model includes a database used to relate soil textural class, from the SSURGO datasets, to these parameters. The version of this table is listed in table A.1.4. In the event that better data becomes available, this table can be edited.

Table A.1.4. A listing of the hydrologic properties inferred from the distributed soils datasets.

| Soil Texture | Abbreviation | Phi | Ksat (m/d) | WP | FC |
|--------------------|--------------|-------|---------------|------|------|
| sand | S | 0.395 | 0.0176 | 0.07 | 0.11 |
| loamy sand | LS | 0.41 | 0.0156 | 0.09 | 0.19 |
| sandy loam | SL | 0.435 | 0.00347 | 0.09 | 0.19 |
| silt loam | SIL | 0.485 | 0.00072 | 0.15 | 0.29 |
| loam | L | 0.451 | 0.000695 | 0.1 | 0.25 |
| sandy clay loam | SCL | 0.42 | 0.00063 | 0.17 | 0.32 |
| silty clay loam | SICL | 0.477 | 0.00017 | 0.17 | 0.32 |
| clay loam | CL | 0.476 | 0.000245 | 0.17 | 0.32 |
| sandy clay | SC | 0.426 | 0.000217 | 0.17 | 0.32 |
| silty clay | SIC | 0.492 | 0.000103 | 0.17 | 0.32 |
| clay | C | 0.482 | 0.000128 | 0.22 | 0.33 |
| Weathered | WB | 0.482 | 0.000128 | 0.17 | 0.32 |
| Unknown | UNKNOWN | 0.482 | 0.000128 | 0.17 | 0.32 |
| unweathered | UWB | 0.495 | 0.0176 | 0 | 0 |
| water | WATER | 0.495 | 0.0176 | 0 | 0 |

A.1.2 Model Derived Data Sets

The derived datasets, which make up the second broad category of model input data, consist of two separate C++ classes. The first class bears some resemblance to a shapefile and is known as a MapLayer. The second class, known as a ReachTree, represents the same data as both the MapLayer and the shapefile, but does so in an entirely different manner.

A.1.2.1 The MapLayer Class

The MapLayer class is an integral component of the model. A MapLayer is closely aligned with a shapefile, but is unique. The most important difference is that a MapLayer is a well-defined C++ class that is made up not only of vertices and a database, but also a host of methods designed to evaluate the spatial data. A shapefile is a data format made up of a collection of vertices and the data describing them. The MapLayer class, along with a variety of other classes defined by the Biosystems Analysis Group (BAG) at Oregon State University, is essentially a stand alone, open source Geographic Information System that acts a bridge between proprietary GISs, like ARC/Info, and the data structures which comprise the foundation of the WET simulation package.

A.1.2.2 The ReachTree Class

The ReachTree class is derived entirely from the MapLayer class and is necessary from a model standpoint. This is because both the shapefile and the MapLayer are designed primarily as visualization and data maintenance tools, not as numerical simulation units. The shapefile, for instance, is made up of up to five separate files, two of which are absolutely necessary. The first is a binary file containing vertex information for every polygon used to describe space. The second is a database file, with a separate record corresponding to each shape and defining the properties of that area. The vertices making up a polygon 'know' about the other vertices in the same polygon, but not about other polygons – shapefiles are non-topological datasets, where relationships among polygons are undefined. This format is useful from a display perspective, but from a distributed modeling standpoint, the relationships between adjacent units must be defined. The ReachTree class, which is unique to the Watershed Evaluation Toolkit, is the definition of topology for each simulation. The ReachTree is designed for numerical efficiency - at the expense of visualization, and is essentially made up of a series of large, well organized floating point arrays. The class contains input information related to both streams and upland zones. This information is a combination of physical characteristics and topology.

A.1.2.3 Reach Definition

Reaches in the ReachTree class are defined as branches of a binary tree, and are documented in chapter 3. It is worth noting that for every reach defined in the shapefile (as well as the MapLayer), an equivalent reach exists in the ReachTree. The major difference is that each reach in the ReachTree explicitly maintains information about its location within the network, but does not maintain a detailed spatial description of the vertices that define it spatially. In addition, the ReachTree class is designed to store data of a generic nature. This is because a variety of other environmental models make use of the ReachTree class. In the case of WET_Hydro, this generic information consists of a structure containing data related to the reach that is unique to a hydrologic model. This structure is referred to as a REACH_INFO_HYDRO. In the case of other models that make use of the ReachTree class, the generic pointer represents other, model specific, data.

A.1.2.4 Upland Definitions

Each REACH_INFO_HYDRO maintains a pointer to an array of structures referred to as UPLAND_INFOS. The array represents a set of conceptually realistic areas that are linked to one another and the channel, through rules of adjacency and elevation. Each reach maintains a pointer to an *array* of

UPLAND_INFOs because the model supports three separate, and progressively more complex levels of upland discretization. This flexible architecture is designed to facilitate analysis of the effect of model resolution on results, as described in Chapter 4.

As part of the ReachTree definition, each branch in the tree corresponds, by definition, to exactly one reach in the GIS data structures (Maplayers and shapefiles). But this one-to-one correspondence is relaxed for the Upland definition. This relaxation occurs because while the stream network is explicitly developed to define hydrologic relationships, the upland polygons are not. It is during the development of the upland portions of the ReachTree (and not during the development of the GIS coverages) that hydrologic relationships between upland units are specified. Because polygons and their associated data must be placed together into hydrologically related groups, which in turn relate to a specific stream segment, there is not a one-to-one correspondence between polygons and reaches in the Upland definition.

There are three steps in the development of the ReachTree upland definition. The first step is the determination of the appropriate number of upland units. The second step is the assignment of polygons (data from the MapLayer) to each of the upland units. The last step is the calculation of representative parameter values for each upland unit. These representative values are based on the polygons that the upland unit represents.

In the simplest topological description, each watershed (or stream segment) is represented by a single upland unit that represents the average properties of all polygons in the watershed. A somewhat more complicated definition divides the watershed into a set of adjacent zones. These zones can be based either on user input or a model calculation related to the distance of each polygon to the stream. Because the distance is simply a magnitude, there is no simple method to distinguish polygons on one side of the stream from polygons on the other. For this reason, each sub-watershed is configured as a symmetric system of bands.

The third, and most complicated method is developed through the calculation of the side of the stream on which each cell exists. While the distance from each cell to a stream defines the length of a vector, this additional information provides the magnitude of that vector. This allows us to differentiate one side of a stream from the other. WET_Hydro assumes that first order streams have a single "side", and that this 'side of stream' calculation is unnecessary in that instance. The justification for this assumption is that the majority of land area in a first order watershed is essentially unchannelled. There is therefore no reasonable basis to define the side of the stream on which a point in such an area exists. In the case of second and larger order streams, the side on which a point falls is more easily defined. WET_Hydro includes code that follows a grid-based flowpath and uses the angle with which cell discharge intersects stream discharge to distinguish one side of the stream from another.

In those cases where upland data is not a simple average of all the polygons in each watershed, the calculation of the upland structures is dependent upon some measure of the distance from each cell to the stream. Here we define that distance. In the current version of WET_Hydro, code has been implemented to make use of two such measures. In the first case, these measures must be defined in the input shapefiles, and in most cases represents a buffered distance. Under this scheme, each cell maintains information in the cell database related to the cell's distance to the stream. In the alternative case, where these data were not preprocessed, the model has capability to calculate alternative estimates of the distance from each cell to the stream segment into which it drains. To make these calculations, the centroid of each cell is determined, and then the nearest straight line distance from the centroid to the nearest point on the stream is calculated.

The use of this distance information is always in concert with the subwatershed identification that is a part of the input data set definitions. The user specifies the number of bands to be modeled in each subwatershed. This value is dependent upon the number of polygons and the size of the basins under consideration, but might acceptably range from one to twenty. The model uses this combination of distances from the polygon coverage and band widths from the model structures to assign the appropriate polygons to each model unit. Once the polygons have been mapped into the model units, representative parameter values are estimated by area weighted averaging of the values associated with each polygon.

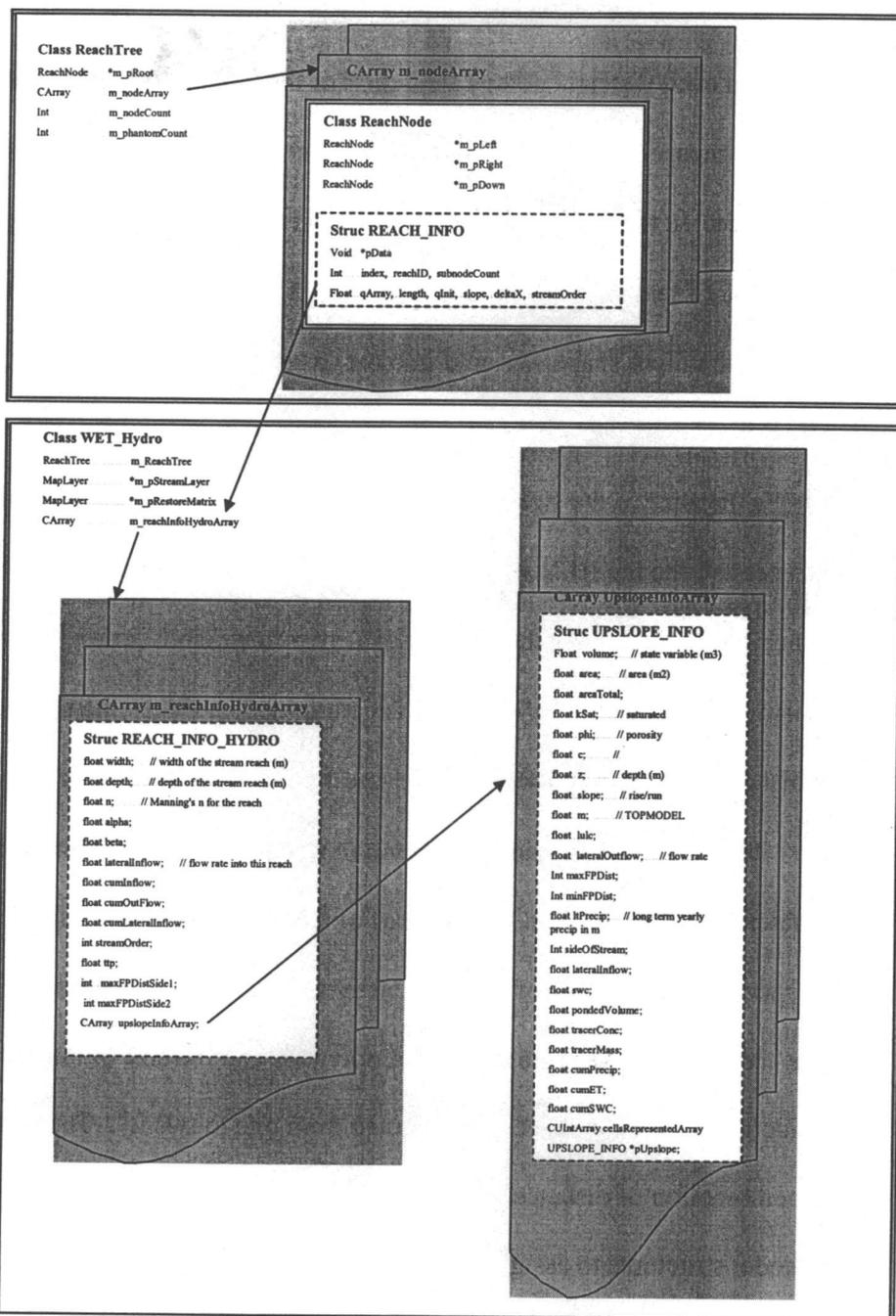


Figure A.2. A chart representing the REACHTREE data structure designed to store topological definitions and spatially distributed data.