HYDROLOGICALLY INFORMED DEVELOPMENT:

A LANDSCAPE ANALYSIS OF THE IMPACTS OF RURAL RESIDENTIAL DEVELOPMENT ON DRINKING WATER QUALITY IN THE LOWER MCKENZIE WATERSHED, OREGON

by

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A THESIS

Presented to the Department of Planning,
Public Policy and Management
and the Graduate School of the University of Oregon
in partial fulfillment of the requirements
for the degree of
Master of Community and Regional Planning

June 2011

THESIS APPROVAL PAGE

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Title: Hydrologically Informed Development: A Landscape Analysis of the Impacts of Rural Residential Development on Drinking Water Quality in the Lower McKenzie Watershed, Oregon

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Exurban growth is prevalent in watersheds nationwide and of special concern in areas important for their undeveloped qualities. The McKenzie River, Oregon, is a natural amenity of great public, aesthetic and recreational value and provides drinking water for much of the southern Willamette Valley. These qualities also make the basin an attractive place to live, and their preservation is often in conflict with the rights and gains of private landowners. However, current containment strategies of development can be arbitrary from a hydrological perspective, especially when adapted from urban contexts. This study introduces a spatially-explicit and physically-based approach for identifying hydrologically sensitive lands in periurban watersheds and then applies that model as a framework for assessing current risk to municipal drinking water sources from exurban residential development.

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ACKNOWLEDGEMENTS

I thank each of my advisors for their assistance in the preparation of this manuscript. In particular, I thank Robert Parker for his oversight and as a personal example of holistic thinking. I thank David Hulse for the clarity and structure of his ideas and for his passion and dedication to understanding complex landscape problems. I thank Scott Bridgham for his exceptional eye for detail that he applies to all his work. Finally, I thank Marc Schlossberg as a personal and professional mentor: May he encourage and inspire many students to come. Special thanks are due to the Environmental Studies program, which provided financial support for a great deal of my educational tenure at the University of Oregon.

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CHAPTER I

INTRODUCTION

Air, soil and water constitute the environmental continuum, and are vital components for sustaining life on earth. These components are interactive and interactions amongst them are complex. … Change imposed on one component in the environmental continuum has effects that propagate to other components, and some of these effects are unknown and cannot be quantified. This interactive nature requires that the environment is managed and protect as a cohesive whole (or as a system)

– V.P. Sign. Watershed Modeling

Water degradation emerges as a problem at the scale of the landscape, even as its causes are ultimately tied to actions that are site-specific. In this way, nonpoint source water pollution is linked to larger socioeconomic forces, including changes in development pattern, population increase, and land use conversion. Perhaps no socioeconomic force is more pervasive than the growth of the modern city, which requires ecosystem support areas many times larger than the urban land base. What is more, cities invariably exert development pressures onto surrounding lands, and sometimes into those areas most important for their undeveloped qualities. Few ecosystem services are more important, nor more sensitive to landscape change, than drinking water.

Like many rural counties nationwide, Oregon rural development code seeks to minimize development in rural lands. These regulations focus primarily on preserving large, intact lots of farm and forestland. The use of zoning and subdivision controls to protect water quality is less well developed, and often relies on land use controls that are potentially arbitrary from a hydrologic perspective. New development can create risks to water quality many times what it otherwise would solely as consequence of poor siting. For this reason, location-specific restrictions may offer more promise to protecting water quality than do widespread growth moratoria. Yet zoning, lot size and form, topography, climate, and soils vary significantly across a watershed. The challenge therefore becomes how to systematically define areas inappropriate for development in a systematic and hydrologically informed manner.

"The task of controlling land use to protect water quality and watershed health necessitates finding the right matches between various ecological and hydrological scales and functions, on the one hand, and various land use planning and regulatory scales and functions, on the other" (Arnold, Clean Water-Land Use 2006). Since land use controls focus on design standards at the site level, cumulative impacts of rural residential development on water quality at the scale of the watershed may remain unaddressed. Clustered development, permeable soils, and proximity to streams and rivers can all compromise the capacity of onsite septic systems to treat domestic waste. Given the nested and spatially-distributed nature of processes involved in pollution transport and delivery, a more robust strategy to contain current (and future) water quality risks may require considering current residential development against underlying hydrogeomorphic dynamics—dynamics that only emerge at the landscape scale.

With population increasing at a rate above the national average, Oregon continues to balance growth against intensifying and expanding urban land-uses. The potential for exurban growth is particularly strong in watersheds adjacent to major urban centers, and of special concern in those watersheds that provide municipal drinking water. Since urban growth is closely linked to public infrastructure, Oregon law prohibits the construction of community waste treatment systems in drinking water source watersheds (OAR 340-041 2010). Primary treatment of residential waste therefore becomes dependent on onsite septic systems. As a result, rural development code implicitly serves both as a means to limit contamination risks to drinking water *and* as a means to constrain rural growth.

The McKenzie Basin is one of three such watersheds in the state of Oregon, and local water samples suggest that bacteria and nutrient concentrations are higher below clustered residential housing (EWEB 2009). This study looks at residential development patterns in the Camp Creek subwatershed, located within the McKenzie Watershed, and the nearest subwatershed to the Eugene Water and Electric Board (EWEB) water intake at Hayden Bridge. Camp Creek provides an important case study of land use and water quality due to its proximity to the water intake for the City of Eugene, its proximity to the eastern edge of the Springfield Urban Growth Boundary (UGB), and the mixture of residential and agricultural land uses it contains.

CHAPTER II

LITERATURE REVIEW

Water is the one substance from which the earth can conceal nothing; it sucks out its innermost secrets and brings them to our very lips.

– Jean Giraudoux (1882-1944), The Madwomen of Chaillot, 1946

A. Impacts of Rural Residential Development

The influence of the metropolis has always extended well beyond the urban boundary. Productive ecosystems located outside the city provide the food, water, and other renewable resources that are consumed within. Modern cities can claim ecosystem support areas 500 to 1,000 times the size of the city itself (Boland and Hanhammer 1999), creating a vast dichotomy between the physical space occupied and the geographic region across which associated demographic, political, and economic processes take place (Brenner 2000). New means of defining and understanding city boundaries may be required.

The relationship between urban areas and surrounding lands, however, is not unidirectional, and urban land uses associated with the urban core increasingly infiltrate into surrounding rural areas. This expanding ring of development is discontinuous in shape, mixed in content, and multi-functional in purpose (Bourne 2001, Iaquinta and Drescher 2000). It is the periurban interface.

Rural residential development, in particular, emerges in the urban periphery well before, or completely apart from, other urban land uses. While rural growth is typically motivated on the one hand by lower housing costs and the presence of landscape amenities, its growth potential is often constrained by the distance to services and urban employment centers. Presumably for this reason rural counties are most likely to grow when they are adjacent to metropolitan areas, which link housing consumers who possess higher than average income with access to both urban and natural amenities (Johnson, Nucci and Long 2005).

With awareness of the importance of the surrounding support areas growing, local governments increasingly address residential growth in periurban zones as it relates to maintain these ecosystem services. Loss of agricultural lands, wildland fire danger, and habitat fragmentation are commonly named within the 'smart growth' literature. However, conversion of land can also have strong ecological impacts on aquatic resources, causing increased flows and concentrations of nutrients, heavy metals, sediment, and bacteria (Paul and Meyer 2008).

What is more, the lots within the watershed that are most attractive to homeowners are often those locations most important to the continuing functions of the water system—riverfront lands, wetlands, aquifer recharge areas, and hillside and mountain slopes (Arnold, Wet Growth 2005). Development in hydrologically sensitive areas further reshapes and redirects water bodies, which can further degrade their natural health and functioning. Left unchecked, the cumulative impact of rural development and associated runoff from lawns, roads and septic systems can become principle drivers of the biological, chemical, and physical composition of water.

B. Bacteria Contamination in Rural Water Bodies

Bacteria are one of the most common pollutants threatening the health of US rivers and streams (USEPA 2002) and are strongly associated with urban development (Hascic and Wu 2006), including residential land use (Schoonover and Lockaby 2006). The relationship between bacteria loads and residential development in rural watersheds is less clear, yet examples of surface and ground water contamination resulting from failing or poorly placed septic systems are common. Septic systems require frequent maintenance and often fail, especially with age. This possibility poses greater risks to environmental and public health as septic systems are increasingly located on marginal lands.

Site-level waste treatment relies on relatively simple technology, which is then adapted to specific site conditions. The design and engineering of septic systems is supported by a large body of academic and professional knowledge, and a complete discussion is beyond the scope of this thesis. Nonetheless, a brief overview follows. All septic system share three elements—the septic tank, the distribution box, and the adsorption drainfield—each of which can be modified to ensure that waste effluent is sufficiently treated. The tank primarily serves to separate solids, liquids and insoluble components of waste. The effluent (i.e. the liquid portion of the waste)—which composes the largest volume—flows from the septic tank into the drainfield, where it is absorbed or metabolized by naturally occurring microbial populations.

The efficiency of onsite waste treatment is highly dependent on the hydrologic conditions of the drainfield (Dawes and Goonetilleke 2003). As a result, the siting and design of septic systems is highly regulated, and requires landowners to obtain permits for the installation, modification, or repair of any system (OAR 340-041 2010). Soils which are extremely porous have limited absorptive capacity and conduct water rapidly, result in highly mobile effluent. By contrast, poorly-drained soils frequently become saturated, which interferes with the aerobic breakdown of waste and can further become hydrologically connected with receiving waters.

By addressing regulation with *site-level* design standards, local governments implicitly treat onsite waste treatment as a *closed-system*. Yet given the heterogeneity of the physical conditions and land ownership within a given watershed, many septic systems *do not* function in a self contained manner, especially during wet times of the year (Chin, et al. 2009). In these instances, waste effluent leaves the drainfield, travels offsite, and possibly becomes connected to receiving waters. Yet rarely will a local government deny development rights outright, even if the property is highly constrained by onsite conditions (Community Planning Workshop 2009).

Part of the difficulty of effective regulation arises from the differences in geographic scale between the land use policies that regulate *site-level* development, on the one hand, and the larger *watershed-scale* physical and social processes that lead to water quality degradation, on the other hand. However, land use controls that focus exclusively on design standards at the site level may fail to address cumulative impacts occurring at the watershed-scale. What is more, the literature in this area is lacking, since researchers have either concentrated on hillslope-scale processes, focused on agricultural practices (and above-ground transport pathways), or used statistical analysis that provide few insights into the spatial distribution of transport and delivery processes.

C. Bacteria Delivery Pathways from Nonpoint Sources

Transport and delivery processes are important to understanding water quality impacts in watersheds with diverse hydrogeology and irregular rural development (Novotny and Olem 1994). Delivery of bacteria to receiving waters can occur via both above ground and below ground pathways (Jamieson, et al. 2002). In both cases, adsorption is presumed to limit the movement of microorganisms (ibid). Adsorption refers to the capacity of soil to physically or chemically bond to various nearby substances, including negatively charged bacteria and viruses. This capacity affects all soil processes and is closely related to soil productivity, weathering and leaching.

The presence of bacteria, it follows, is often positively correlated with water turbidity (Mallin, et al. 2001), presumably when above-ground transport pathways predominate. Most bacteria transport models, particularly those focused on agricultural or pasturelands, are based on sediment erosion (Fraser, Barten and Pinney 1998, Leon, et al. 2002). However, Jamieson et al. (2002) noted how several studies have questioned the validity of linking microbial transport directly to erosion.

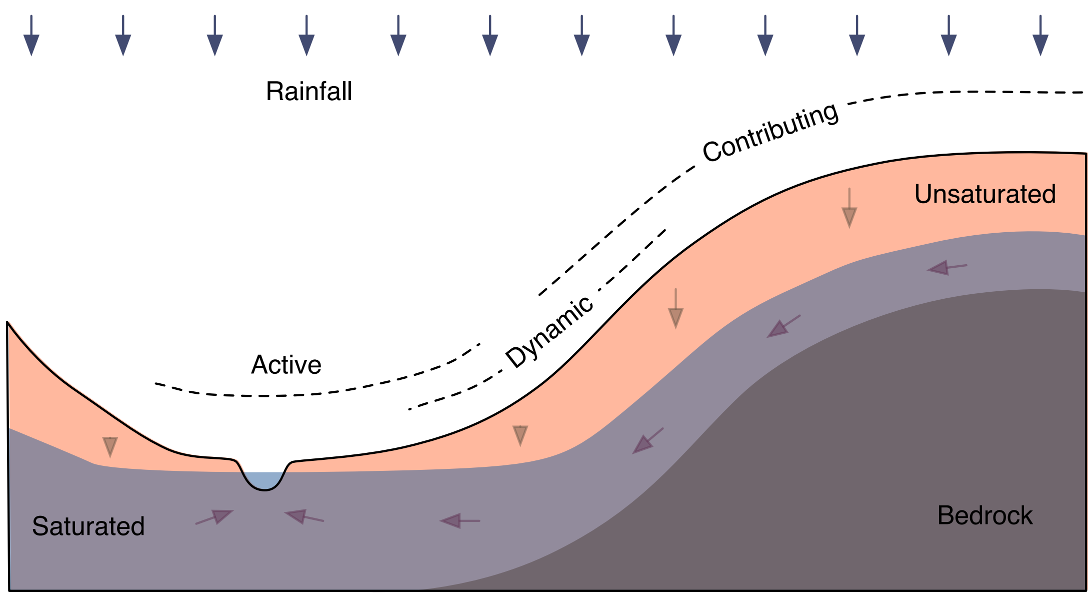
A growing number of studies indicate that bacteria loading may occur via subsurface transport pathways (Jamieson, et al. 2002), especially when considering the impacts of rural septic systems (which are located subsurface). While it does appear true that physical straining and adsorption can effectively block the movement of bacteria located within a true soil matrix—resulting in huge die-offs (Gerba and Bitton 1984)—field experiments indicate that few soils indeed operate as a perfect matrix (Thomas and Phillips 1979, Zehe and Fluhler 2001). When soils become saturated, macropores and cracks (Benham, et al. 2006) allow bacteria to move quite far (greater than 60 m) along preferential pathways (Chin, et al. 2009), and may represent the dominant means of transport and delivery of fecal bacteria in many watersheds.

Still, the physical processes involved in pollution transport and delivery are poorly understood and lack regulatory guidelines for framing analysis (Jamieson, et al. 2002, Lane, et al. 2006, Arnold, Clean Water-Land Use 2006). Yet planning for sustainable development requires addressing water quality *proactively*, often before consensus is achieved; a contradiction frequently termed as a ‘wicked problem’ (Mcharg 1969, Rittel and Webber 1973). In light of these uncertainties, Walters (2000) argues that the most robust framework for assessing and mitigating water quality risks, may be to frame analysis on underlying runoff processes, which can be broadly generalized across a watershed.

D. Hydrologically Sensitive Areas

Runoff is produced by a combination of infiltration-excess and saturation-excess overland flows (Figure 1), and is highly dependent on climatic, geomorphic, and land use factors (Chorley 1978). Processes that contribute to runoff include the movement of water both above and below ground, which is continuously exchanged between surface and subsurface zones.

Figure 1. Overview of hillslope hydrologic processes. While various hillslope hydrological processes can contribute to basin runoff, most runoff is formed as saturation excess and variable source area flow in the Pacific Northwest.



In watersheds with impervious soils, such as in arid regions or urban drainages, precipitation is likely to exceed infiltration capacity (Kirby 1985). Runoff forms as thin sheets of water that runs downslope until it infiltrates more pervious soil or eventually reaches a stream or river. Water quality problems have been historically defined under the paradigm of soil conservation and sediment transport (T. Walter 2000), and overland surface flow has frequently dominated discussions of watershed management, especially in the land use planning literature (Leopold 1968).

In regions where humid, well-vegetated, and topographically steep areas combine with shallow soils with high infiltration capacity, runoff tends to originate within areas of the landscape that are prone to saturation (Dunne 1978). As rainfall falls and infiltrates the soil surface, groundwater tables rise and begin to flow laterally, accumulating at locations where hydrologic conductivity suddenly drops. Thus, some areas saturate more frequently, particularly during wet times of the year, and become primary sources for basin runoff through a combination of water returning to the ground surface (and subsequently forced overland), and additional precipitation that falls and is unable to infiltrate the soil. These saturation excess processes define much of the runoff dynamics in the Pacific Northwest.

Saturation events are more likely to occur in some areas of the watershed than others, particularly in places with shallow restricting layers, where the downhill slope decreases (e.g. the toe-slope of a hill), or where multiple hillslopes converge (e.g. in gullies or valleys) (Beven and Kirkby, 1979). All three incidences occur where the interflow capacity is reduced by a local decrease in hydraulic transmissivity. During periods of extended rainfall, increased interflow causes the aerial extent of saturation around these areas to expand. Since saturation strongly depends on previous soil moisture conditions, these saturation areas can fluctuate widely in extent, even during a single storm.

Soils which are prone to saturation strongly influence the timing and location of runoff generated during wet times of the year. As such, Walter (2000) described these areas as *hydrologically sensitive*. Hydrologically sensitive areas (HSAs) become ‘active’ as rain falls on their surface, fails to infiltrate because of soil saturation, and flows downslope until reaching a nearby water body.

Pollution that originates in saturation-prone soils is likely to become hydrologically connected to surface water. Not only have bacteria been shown to travel quite far along preferential pathways in saturated soils (Benham, et al. 2006), but drainfields also become much less effective at treating waste due to the anaerobic conditions that interfere with the breakdown of waste. What is more, saturated soils cannot accept additional wastewater leaving the septic tank. For this reason, the identification of such soils (i.e. Critical Source Areas) is particularly important in understanding and managing present pollution loading, since monitoring and remediation efforts can be concentrated to key areas in the watershed (Walter, et al. 2000). It further follows that it is equally important to prevent future CSAs from forming. Doing so requires summing hydrologic processes over large and geographically diverse areas in a *spatially-explicit* manner.

E. Pollution Transport in Watershed Modeling

Addressing the complexities inherent to nonpoint source pollution (as described earlier) requires aligning policies across a of range scales. Rural governments often lack a means for integrating site-level land use decisions with watershed-scale hydrology and pollution transport processes. Spatially-explicit modeling approaches can account for the distributed generation, transport, and attenuation of pollution loads (Jamieson, et al. 2002). As such, catchment areas become natural units for understanding, analyzing and managing water quality issues (Singh and Frevert 2006, Dougherty, et al. 2003).

Models capture the emergent nature of watersheds by simulating hydrologic processes according to a number of simplifying assumptions (Singh and Frevert 2006). These assumptions create the conceptual boundaries for describing how complex systems are presumed to act.

The first examples of watershed modeling used empirical approaches to describe relationships between explanatory variables that are spatially-aggregated (e.g. the physical environment, development, or land use, etc.) and response variables (e.g. water flows or water quality indicators). Results from empirical modeling have provided the foundation for much of the hydrological sciences, and have been successfully applied to soil conservation and the identification of pollution sources. Yet statistical models often prove limited when exploring spatial relationships in policy analysis since they allow only limited inferences to be made into underlying hydrologic processes (Atasoy, Palmquist and Phaneuf 2006). Further, statistical models are not spatially-explicit, thereby making it difficult to identify sensitive areas and pollution transport and delivery pathways.

Physical-based watershed models, by contrast, provide insights into the transport and delivery of pollution. For instance, Chin et al. (2009) applied two common watershed hydrology models (HSPF and SWAT) to the same set of hydrology and water quality data, and found that while both models were equally capable of predicting discharge, SWAT was significantly better at modeling bacteria counts. Since each model generated discharge from different respective contributions of runoff, interflow and groundwater, pollution delivery in the study could be assumed to take place via the pathways that SWAT had predicted.

Physical models explain variables via governing equations related to the conservation of mass, momentum or energy, and are typically run across a geographic space, over a period of time, and sometimes both. The degree of detail with which space and time are handled varies by model. For instance, a simple physical model may average spatial values across an entire watershed, while a more complex distributed model may require dividing a single watershed into thousands of smaller parts (represented as pixels or hydrologically similar units).

Not surprisingly, the complexity of physical models is often prohibitive, both in terms demand for data and processing power. Even with the increasing availability of both, many researchers argue that as the number of parameters increase – particularly those that cannot be directly measured or those that co-vary with other modeling parameters – the results of the model become increasingly difficult to substantiate

(Merritt, Letcher and Jakeman 2003). Merrit et al. (ibid.) therefore recommends that in most instances a simple watershed model of 2 to 3 parameters suffices, and that while a more complex model may provide a better fit of observed data, it may not predict future behavior to any higher degree of certainty.

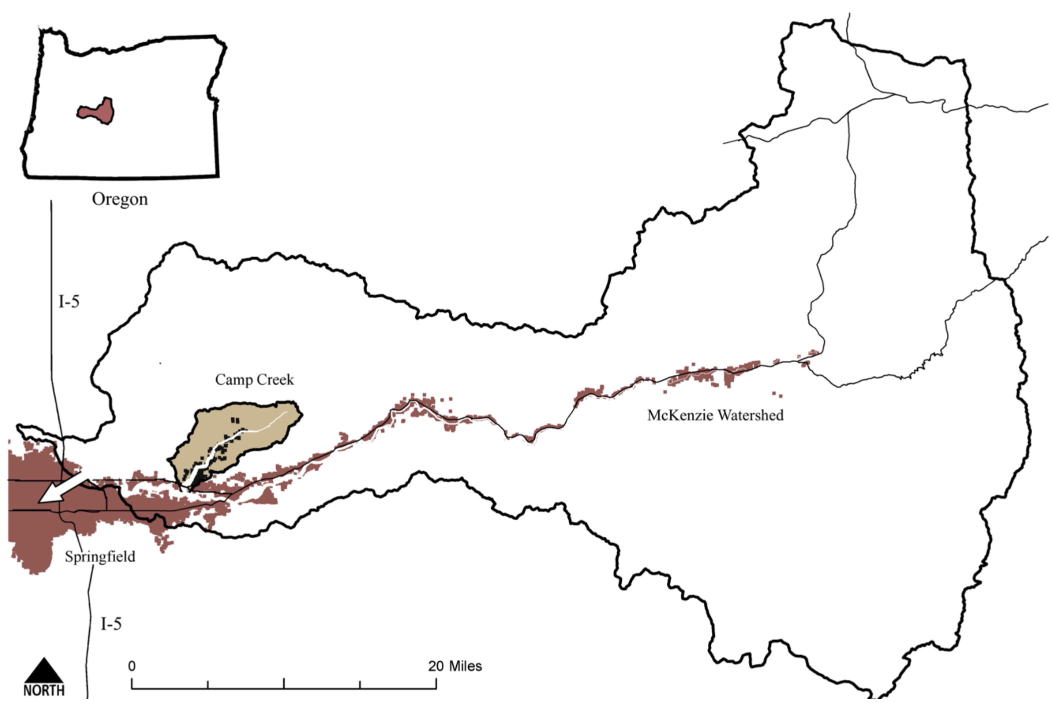
CHAPTER III

METHODS

A. Regional Context

The McKenzie River is a natural amenity of great recreational and aesthetic value within the state of Oregon. Yet these same values also make the watershed of the McKenzie River an attractive place to live. Currently, more than 6000 primary dwellings are located within the McKenzie Watershed, concentrated along the flat valley bottoms and historic transportation corridors (Figure 2). Property value is highly associated with proximity to the McKenzie River: 75% of the most expensive quarter of residential lots are within 300m of the river.

Figure 2. Map of the McKenzie Watershed. The McKenzie Watershed provides drinking water for 200,000 people while also accommodating more than 6,000 households (shown in red). Camp Creek, a sixth order subwatershed located in the lower part of the watershed, contains a share of this development (shown in black). The sub-watershed is located 2 miles northwest of the Springfield UGB and 5 miles upstream from the municipal water intake (white arrow).



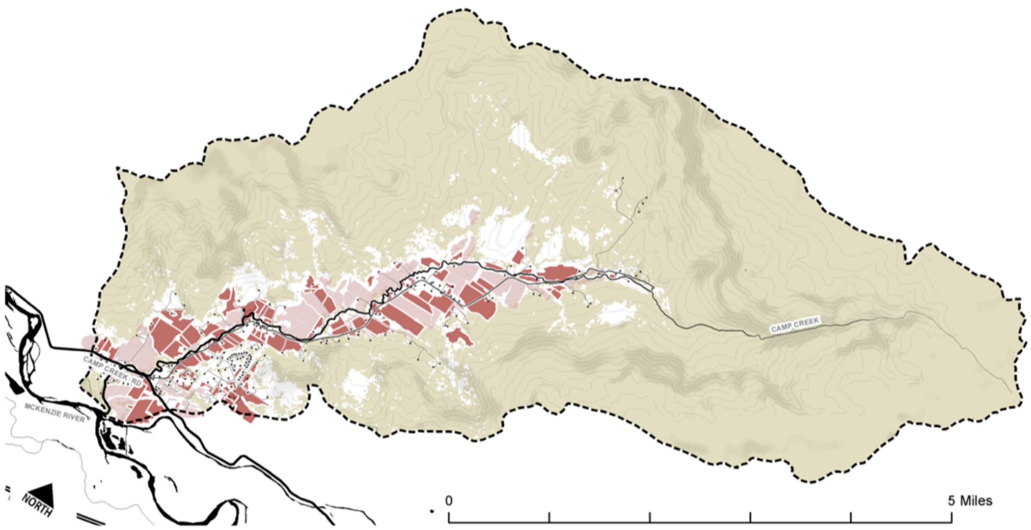
The capacity for additional residences in the watershed remains a key regional planning question. With comprehensive zoning codes well established within Lane County, development in the basin is presumably constrained to a limited set of development possibilities, including additional lots divisions (where minimum acreage requirements allow) and limited new constructed on agricultural and on impacted forests lands (LCOG 2009). Other research, however, indicates that development may also occur outside of prescribed paths (Community Planning Workshop 2009). For instance, roughly one third of land use decisions within the last 30 years were made under the administrative discretion of the planning director (ibid).

B. Study Area

Camp Creek is one of 38 sixth-order subwatersheds within the McKenzie Watershed and located just 2 miles away from the edge of the Springfield UGB (Figure 3). Land use and land cover is representative of the landscape features found throughout the lower McKenzie Watershed. Low-lying foothills gradually flatten into the valley bottom, which were historically shaped by meandering channels and frequent floods. The basin rises from 45m at its lowest to 260m along its westerly divide. Annual precipitation (152 cm/yr) is higher than the adjacent Willamette Valley (100 cm/yr) but lower than the wettest areas of the high Cascades (230 cm/yr).

Development of Camp Creek is representative of the lower reaches of the McKenzie Basin: Covering just 2% of the total McKenzie drainage, Camp Creek accounts for 4% of all residences (n=216) in the basin and 14% of all agricultural lands. Dwellings are concentrated in the lower half of the watershed alongside agriculture and pasturelands. Despite its proximity to Springfield, housing sales data from the lower McKenzie Watershed shows that land value is closely associated with pricing fluctuations found in the housing market higher in the watershed (see Figure 4).

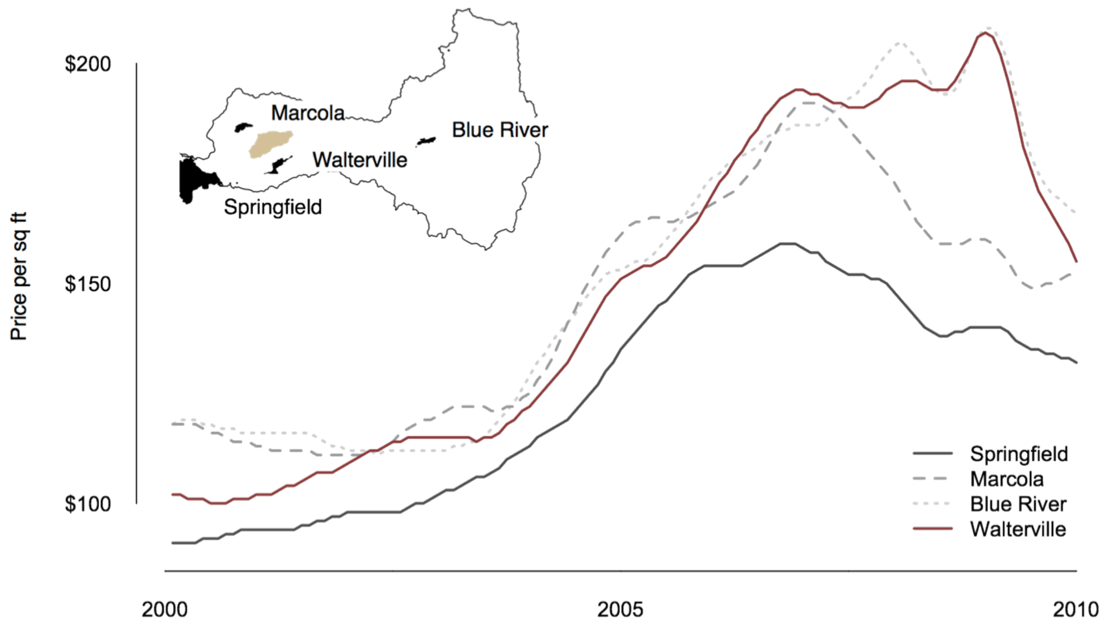
Figure 3. Principal land use and land cover in Camp Creek Basin. Camp Creek contains a mixture of land uses, including residential, agricultural and forestry lands. Houses are shown as black squares; agricultural fields are shown along valley bottom in red (pasturelands are shown in darker tone than hay fields); forested areas on surrounding hillslopes are shown in light brown.



Considering the proximity of Camp Creek to nearby urban employment centers (15-25 minutes), the disproportionate share of Measure 37 claims[1], and the increased demand shown in housing sales, Camp Creek may be susceptible to future exurban growth. Combined with its proximity to the municipal water intake, Camp Creek subwatershed can provide unique insights on the relationship between competing urban demands on surrounding rural land.

[1] Measure 37 allowed landowners who had bought land before the passage of statewide zoning laws the opportunity to demonstrate their original intent to develop their property. If approved, the landowner could either request payment for loss of vested property rights, or bypass local zoning and use the land as originally intended. Nearly all claims lie in the most populated region of the state, the Willamette Valley, and the density of those claims is highly associated with their proximity to existing Urban Growth Boundaries.

Figure 4. Home sales within the McKenzie Watershed between 2000 and 2010. Homes in the lower McKenzie Watershed (including Camp Creek) are worth significantly more than houses located within the Springfield Urban Growth Boundary—an indicator of latent rural demand.



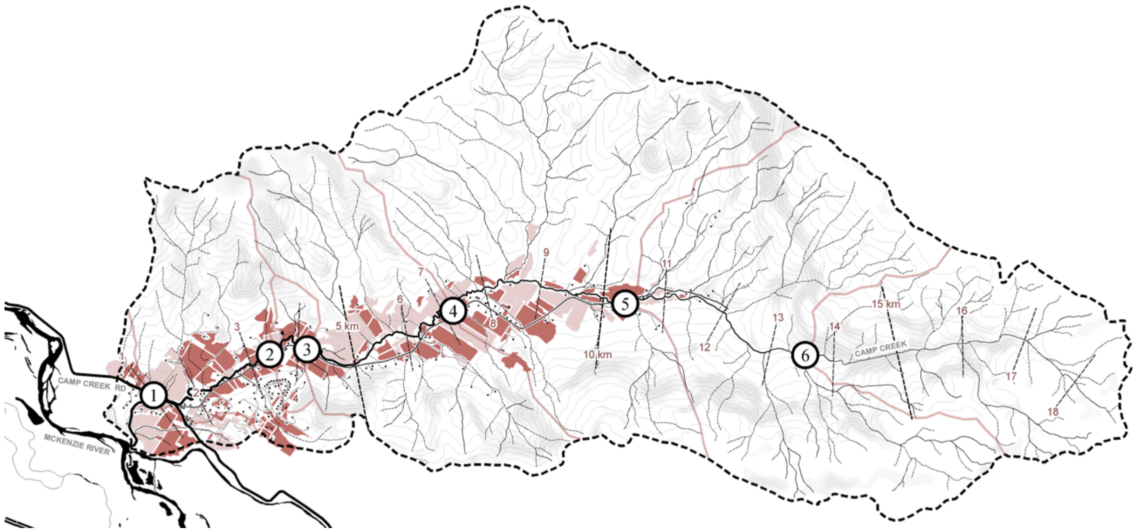
Data Source: http://www.zillow.com 2010

C. Empirical Analysis Methodology

Ten monitoring stations have been established in the Camp Creek watershed previous to this study (Figure 5). Six of the monitoring stations are located along the main channel of Camp Creek, and the remaining four along upper tributaries. Monitoring stations along the main stem of Camp Creek were placed with reference to different land uses found within the basin: the lower two stations contain a greater relative proportion of rural residences; the middle two stations are located in areas of agricultural land use; the highest stations are more heavily forested.

Various partners have aided in the collection of water samples over the last 10 years but the data had yet to be comprehensively analyzed. The majority of data exists between June 2007 to February 2010. During this time, teams of students from the local Thurston High School traveled to Camp Creek once each month to collect water samples

Figure 5. Water-quality monitoring stations located on the main stem of Camp Creek. Water quality stations are represented as large white circles. From left to right, they are: (1) E310, (2) E313, (3) E314, (4) E312, (5) E315 and (6) E317.



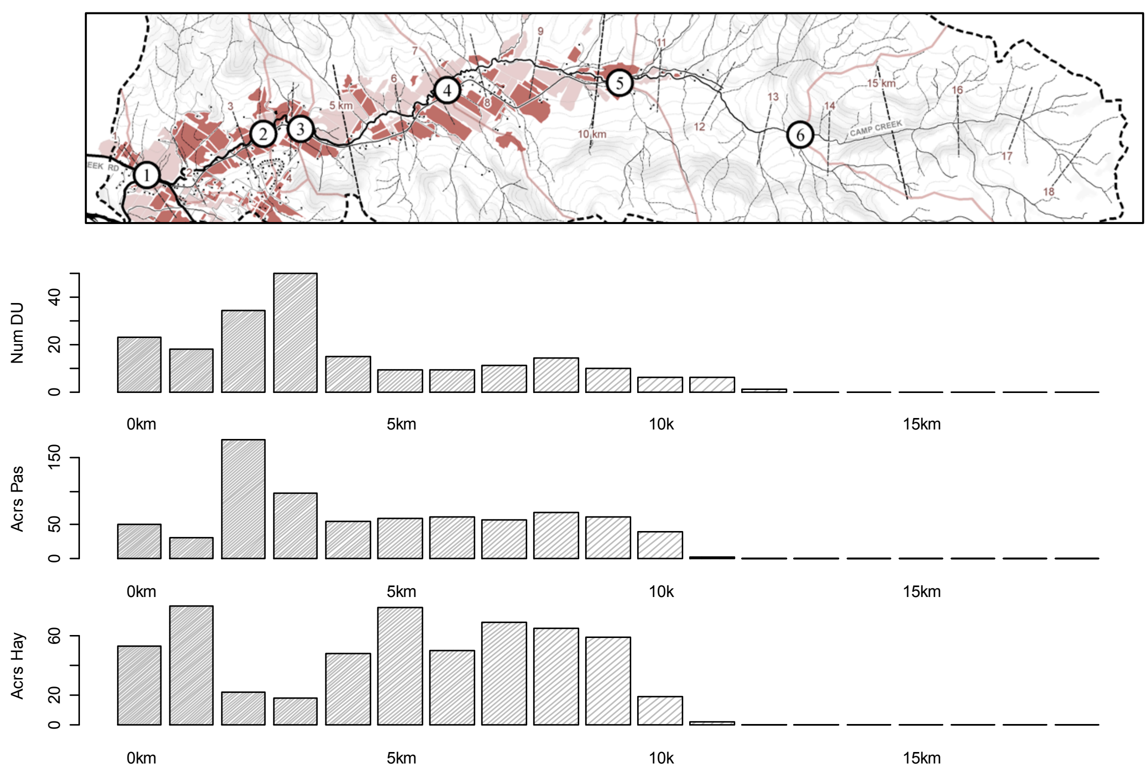
from each of the monitoring stations. The students then tested water samples for the following water quality indicators, including concentration of total coliform, concentration of *E. coli*, conductivity, turbidity, pH, nitrates, and phosphorous. All together, the water quality data is composed of approximately 300 water samples. It is worth noting here that the distribution of many water quality parameters is highly skewed (e.g. bacteria, nutrients). Such data was log transformed since most statistical tests assume normally distribution.

The first phase of this study sought to identify spatial trends between water quality indicators and the upstream distance from which the water samples were taken. The significance of this relationship was assessed using linear regression modeling. Additional tests to describe changes in water quality throughout the year for those water quality indicators found to have statistical significance, .

Since elevated water quality parameters can occur naturally, it was also important to ensure that higher readings aren't just the result of a greater drainage area. When cumulative development increases with drainage area, as in Camp Creek, it can be difficult to separate the influence of development on water quality from background sources. To address the point, a subsample of the sampling areas was created composed of only those drainages without upstream development (either residential or agricultural). Background sources can be ruled out as a dominant pollution source if water quality trends that are found in downstream developed gauges do not hold true for upper undeveloped tributaries.

Assuming anthropogenic pollution loads, the second phase of the empirical analysis sought to establish proportional pollution loads between agricultural and residential sources (Figure 6). Since main stem samples taken above E315 have little housing development or pasture upstream, E315 and E317 are assumed it to act as a baseline. Gauges E310 and E313 (which have a higher relative proportion of rural residential development) should show elevated readings relative to central stations E314

Figure 6. Distribution of development in Camp Creek by development features count per each kilometer reach. The left side of the chart represents reaches closer to the catchment outlet. Numbered circles indicate water quality monitoring stations along Camp Creek. Bold red lines indicate the upstream divide for each station.



and E312 (higher relative proportion of agricultural lands) if pollution loading occurs solely as the result of residential sources. GIS was used to identify the amount of primary dwelling, pastureland and hayfields that lie above each monitoring station. I then ran a number of regression analyses, and compared competing models for added explanatory power. Significance was assessed by examining variances within residual error of each iteration of the multiple regression model (i.e. residential vs. agricultural).

D. Distributed Hydrologic Modeling Methodology

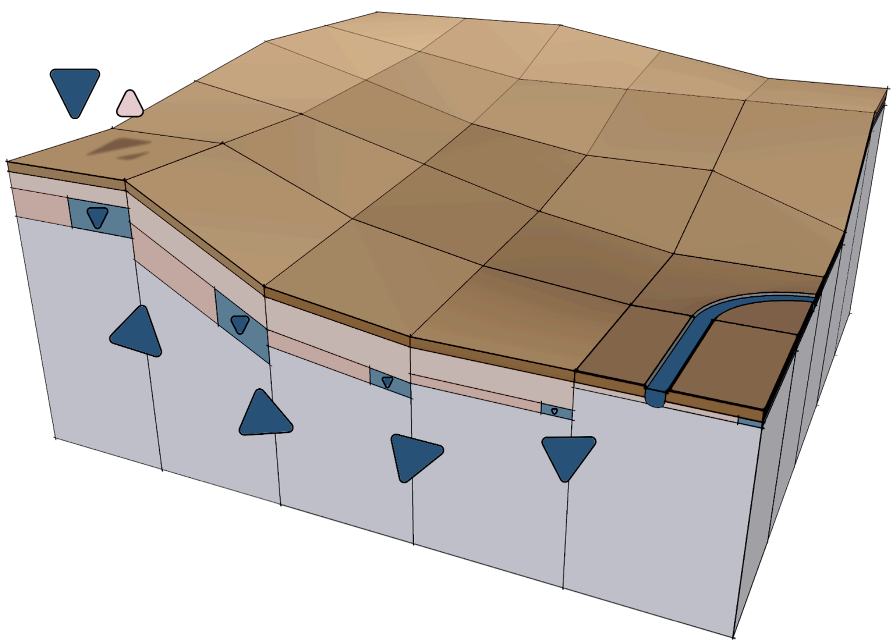
> *Remember that all models are wrong; the practical question is how wrong they need to not be useful.*
>
> *– George E. P., Empirical Model-Building, p. 74*

TOPMODEL (Beven and Kirkby, 1979) is a parsimonious and spatially-explicit hydrologic model that is based on the assumption that a watershed-wide water table intersects the landscape to produce runoff generating areas. TOPMODEL has been shown to successfully identify saturated areas (Holko and Lepisto 1997). TOPMODEL is raster-based, which requires dividing the study watershed into a grid, where each pixel represents a column of soil capable of holding a discrete quantity of water (Figure 7). By linking together neighboring soil columns, TOPMODEL simulates the subsurface flow of water through the landscape. Flow rates are lower when the slope of soil column is relatively flat, which will cause the water table to rise over time. By contrast, the water table will lower in areas with steeper slopes since more water escapes laterally into downslope cells. Via these processes, water tends to accumulate in downslope cells, particularly those at the concave portion of the hillslope or where multiple hillslopes converge.

The dynamics described above are defined by two simplifying assumptions on ground water flow: (1) that the underground hydrologic gradient follows surface topography, and; (2) that hydrologic conductivity fluctuates with the depth of the local water table. Each is discussed in detail below.

1. Like all physical processes, water moves along energy gradients from high to low. Within mountainous watersheds, topography is the predominant energy gradient that controls the movement of both surface and subsurface flows. While hydrologic processes are invariably complex—complicated by macropores, soil heterogeneity, and local pockets of saturation—TOPMODEL assumes low rate is directly proportional to the slope gradient; steeper slopes discharge a greater quantity of water than shallower slopes.
2. TOPMODEL assumes that lateral outflow decreases exponentially as the water table recedes. Stated another way, the topographic impacts on water movement are greater the closer the water is to the ground surface. By combining an estimated recharge rate (i.e. rainfall), respective contributing area, local slope, and local soil properties, both ... t any point in the

ure 7. Organization of water stores used in TOPMODEL. As rain falls, water ins (arrows) through vertical zones before flowing downslope. Downslope cells d to be more saturated (darker) than hillslope cells (lighter). Saturated cells ve basin discharge during wet times of the year.



TOPMODEL is easily translated to other programming/scripting languages. In this study, TOPMODEL was written with the Python scripting language (http://www.python.org/). Python is an open-source programming language popular because of its use of high-level scripting, its efficient handling of data arrays, and its easy integration with GIS software. Since a number of modifications were made to the TOPMODEL framework (which are discussed below), the Python version used in this study is referenced as TOPyMODEL.

i. Model Framework

TOPMODEL requires dividing the landscape into a number of discrete reservoirs (i.e. raster cells). Each pixel (i.e. vertical reservoir) within the watershed is further divided into three vertical zones: (1) the intercept zone; (2) the infiltration zone; and (3) the saturation zone. Described in the simplest terms, water enters the watershed in form of precipitation, sequentially passes through each vertical zone, and flow laterally once it has reached the water table, eventually leaving the watershed as stream flow (Figure 8). Each phase of this process is discussed in detail below.

a. Intercept Zone – Infiltration and Evaporation Functions

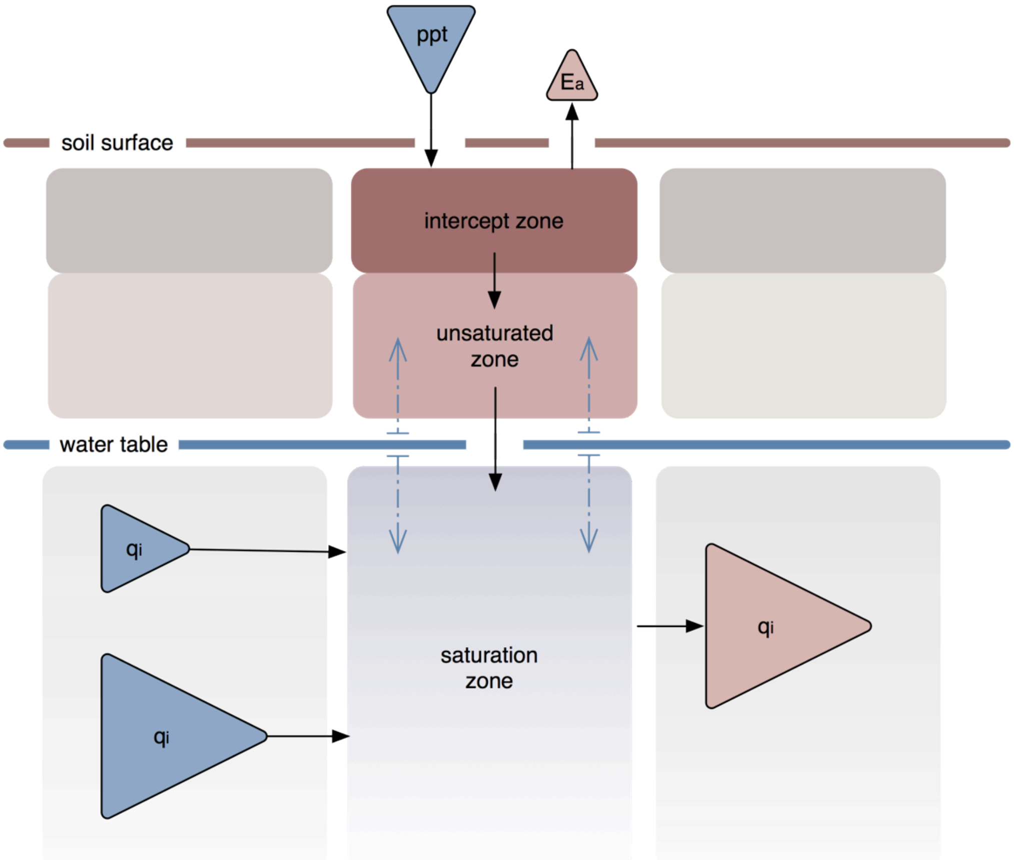
Precipitation first enters the upper intercept zone – a simplified representation of the vegetation and above soil features that trap water at the start of a rainstorm. It is from this zone that evapotranspiration losses occur. Evapotranspiration losses (E_a) are proportional to the amount of water in the intercept zone (S_i) compared to its maximum potential capacity (S_{imax}). If the intercept zone is fully saturated, water is lost at the maximum potential rate (E_p).

Equation 1. Evapotranspiration loss (E_a)

$$E_a = E_p \cdot \left(1 - \frac{S_i}{S_{imax}}\right)$$

Evapotranspiration potential also varies by season. Summer losses exceed winter losses by an order of magnitude because of difference in solar radiation. This variation is

ure 8. Water exchange between vertical and horizontal stores in TOPMODEL. ter flows laterally with downslope gradient (see Figure 10) after it enters the uration zone (water table). The rate of movement is based on moisture nditions, depth to the water table, and the slope of the cell.



especially true in higher latitudes where the day length and directness of radiation are more variable. The TOPMODEL framework was modified to adjust for seasonal changes in evapotranspiration losses. The potential evaporation loss for each time step was calculated using a modified version of the Hamon PET formula (Hamon 1963). Values were lowest in the winter (10 mm/day) and highest during the summer (40 mm/day).

b. Unsaturated Zone – Moisture Content and Vertical Drainage Functions

Water then drains into the unsaturated zone once falling precipitation exceeds the capacity of the infiltration zone. The unsaturated zone functions by adding a time delay to the basin runoff response. Water leaves the unsaturated zone by draining into the underlying water table (Equation 2). Vertical drainage (q_v) is highest when the unsaturated zone contains more moisture (s_{uz}) and when the water table (s_{wt}) is close to the ground surface. Moisture within the unsaturated zone drains only after the field capacity (fc) has been reached

Equation 2. Moisture content of unsaturated zone (mc)

$$\mathrm{mc} = \left(\left(1 - \frac{s_{uz}}{s_{wt}}\right) - fc\right) \cdot \ s_{wt}$$

When both the moisture and depth functions are combined, vertical drainage (q_v) is expressed as Equation 3. If the saturated zone rises to depth 0, the soil has become completely saturated and additional rainfall produces surface runoff (saturation excess).

Equation 3. Vertical drainage (q_v)

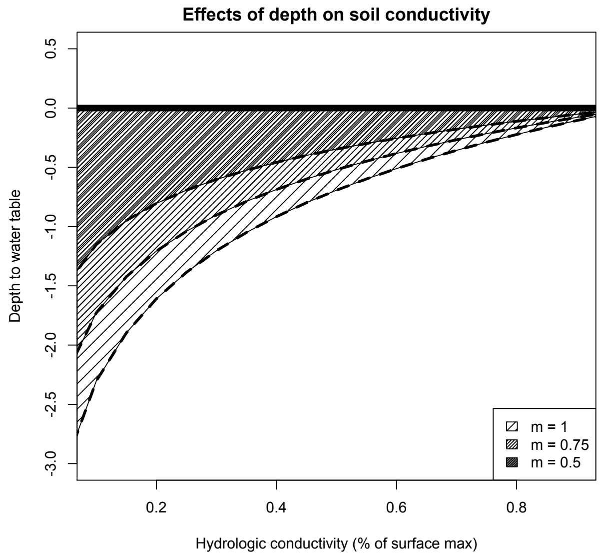
$$q_v = \ \mathrm{mc} \cdot e^{-\frac{s_{wt}}{m_v}}$$

c. Saturation Zone – Saturation Deficit and Lateral Flow Functions

The rate of lateral subsurface flow (q_i) is dependent on both local slope and the depth of the saturated zone. Darcy's law predicts that flow is a function of hydraulic gradient (i.e. difference in pressure head). TOPMODEL assumes that topography is the sole determinant of the hydraulic gradient: the steeper the gradient, the greater the outflow. In reality, pressure heads can be affected by many other variables.

Like vertical drainage, subsurface flow also changes as an inverse exponential function of the depth to the water table (s_{wt}). The exact shape of relationship between water table depth and outflow is defined by the decay-depth coefficient (m). A larger coefficient increases the effective depth at which water moves readily downslope, while a smaller value allows only water in the upper reaches of the soil profile to flow (Figure 9).

ure 9. Hydrologic connectivity and depth to the water table in TOPMODEL. drologic conductivity (K_{sat}) decreases as the water table (s_{wt}) drops. The depth ay coefficient (m) defines the strength of this relationship.



Discharge from any one cell, then is equal to the depth-adjusted rate ($e^{-\frac{S_{wt}}{m}}$) at which completely saturated soil conducts water at surface level (K_{sat}), slope ($tan(\beta)$) and the cross sectional width (c) across which the flow is passing (30m in this study).

Equation 4. Lateral subsurface flow (q_i)

$$q_i = K_{sat} \cdot c \cdot e^{-\frac{s}{m}} \cdot tan(\beta)$$

In the original TOPMODEL framework, the entire system is held in a steady state (inflow = outflow). As the time step is shortened equilibrium becomes increasingly unlikely due to the slow rate of conductivity seen in most soils. As a result, the TOPMODEL framework was made to be more fully dynamic by eliminating the assumption of static state. During each time step, each raster is ‘visited’ recursively. Once the ‘moving analysis window’ has climbed to the highest points of the catchment (i.e. those which have no upslope neighbors), calculations of water movements are handled in the same sequence as the water flows. Thus, at each time step, the analysis window of the model will have traveled from the outlet to the farthest reaches of the watershed, and then sequentially calculated fluxes through infiltration, drainage and lateral subsurface flow. Expressed mathematically, the local storage deficit (s_{wt}) at each time step is equal to the difference between the local outflow (i.e. all incoming flows from neighboring cells) and any vertical discharge from the overlying infiltration zone.

Equation 5. Change in saturation deficit (s_{wt}) with each time step.

$$s_{wt} = s_{(x-1)} + q_{out} - q_v - \sum_{n=1}^{8} q_{in}$$

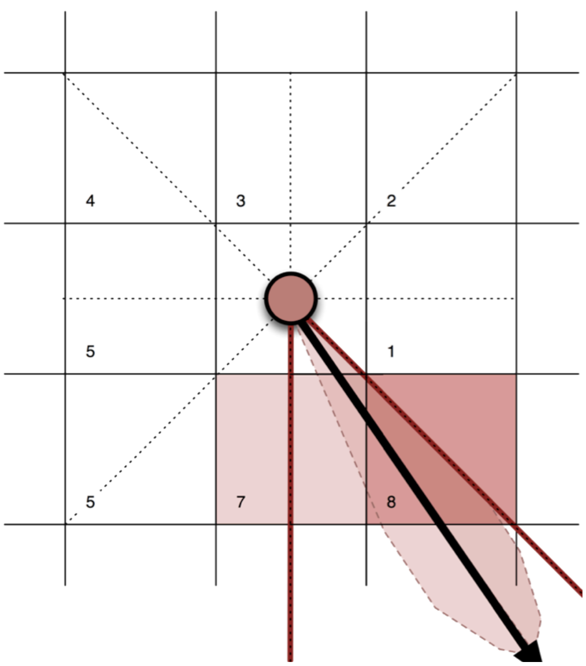
The summation function ($\sum_{n=1}^{8} q_{in}$) of Equation 5 represents the total volume of water that flows into a cell from upslope neighbors. The direction of outflow is expressed as a vector between 0 and 2π and is based on the steepest slope angle between all

ighboring cells.

function of the

eighboring cell

ure 10. Subsurface flow direction in TOPMODEL. Below, water flows southeast h the steepest slope and enters neighboring cells 7 and 8. The proportion of low received is calculated from the difference between the flow vector (black) and nearest two reference angles (red).



with the smallest difference between the direction of flow (i.e. the black line) and the nearest reference angle (i.e. the red lines).

ii. Model Setup

In comparison to other hydrologic models, particularly those that are spatially-explicit and temporally dynamic, TOPMODEL is parsimonious with regards to setup data, and moreover, contains relatively few calibration parameters. The following section describes the derivation of input data and the parameters used in model processes.

a. Input Data

Digital Elevation Model (DEM): As noted above, TOPMODEL assumes that topography plays a dominant feature in determining the hydrologic character of the basin. The hydrologic gradient is calculated as the tangent of slope, and slope is easily calculated based on elevation values. USGS DEM (1/3rd degree second) was reprojected to UTM 10N and resampled to 30m raster cells. All subsequent raster images were aligned to these reprojected elevation raster maps (Figure 11).

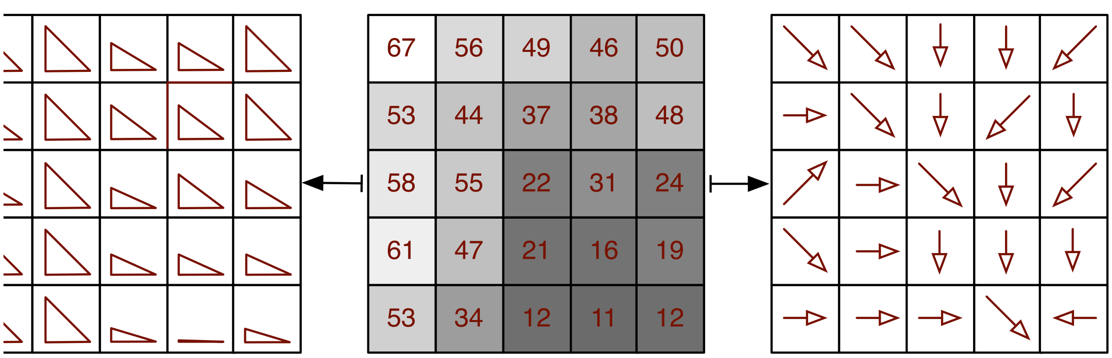
Slope and Directional Flow Maps: Slope values and directional flow values for each pixel were calculated using a hydrologic extension for ArcGIS (Tarboton 2010). Slope values were calculated as a ratio of rise over run. Flow directions were calculated from steepest subsequent angle. The TOPyMODEL then uses the directional flow values to direct subsurface flow into neighboring cells.

Hydrologic Conductivity Map: Soil conductivity influences subsurface flow dynamics by specifying the velocity at which water will move through saturated soil. Data were adapted from NRCS soil maps for Lane County. NRCS hydrologic conductivity was then resampled to a 30m resolution. Gaussian blur was applied to the image using neighborhood statistics to average cell values over a 250m radius. This process created soil maps that maintained the overall distribution patterns but eliminated the unnaturally sharp breaks created by the NRCS data.

Precipitation Data: Precipitation data are the principle input for any hydrologic model. All precipitation in Camp Creek was assumed to fall as rain. Since no precipitation data was available within the Camp Creek basin, nearby stations served as proxy data. The nearest climate station to Camp Creek with long-term daily precipitation data is located north of Eugene (Eugene Mahlon Sweet Field). A scaling-factor was calculated using the average annual rainfall data from the PRISM climate model between 1979 and 2010 (http://www.prism.oregonstate.edu/). Since rainfall for the Camp Creek basin averaged 158 cm and the Eugene Mahlon weather station at 114 cm, precipitation data was transformed linearly using a scaling coefficient of 1.39.

Septic Systems: Septic systems represent the origin of bacteria populations within the model. No comprehensive set of GIS data exists on septic system locations within the McKenzie watershed. A study of septic systems by the local water provider provides some guidance (EWEB 2009). Septic tanks are located a mean distance of 30m from primary residences. All properties surveyed had at least one drainfield, while some properties had two or three. For the sake of this study, I assumed that septic systems are ... unit for each tax

ure 11. Derivation of TOPyMODEL base maps. USGS Digital Elevation Models EMs) depict the elevation of each pixel as a single value (middle). Slope (left) and v direction (right) are calculated by comparing the height value of each cell to ghbors.



b. Modeling Parameters

Intercept zone depth (s_i): The amount of precipitation absorbed before draining into the groundwater system. This capacity also represents the water in the system upon which evapotranspiration losses are assumed to work. Increasing s_i impacts sporadic rain events to a greater degree than continuous precipitation. This is most evident during fall and spring rain events, during which previous moisture is less likely to have fallen and 'pre-loaded' the soil. Forest cover is assumed to double this capacity.

Field capacity (fc): Water drains vertically into the water table as a function of the moisture content of the soil. Capillary forces can cause a great deal of moisture to be retained in the soil, depending on the soil structure. The percentage of moisture at which vertical discharge ceases to occur is called the field capacity.

Depth decay coefficient – drainage (m_v): The water table depth also affects vertical drainage. Less water reaches a lower water table. The decay is assumed to occur as an inverse exponential function. Small changes in the drainage decay coefficient had a large effect on the overall discharge throughout the season.

Depth decay coefficient – lateral flow (m): The flow decay coefficient works similarly to the coefficient above, but relates to the horizontal subsurface flow of water in the saturated zone. The lateral decay coefficient had a more nuanced effect on overall discharge than the drainage coefficient m_v, affecting the discharge of some single-storm events more than others. In addition, smaller values tended to cause a slightly less steep slope on the descending part of the hydrograph after storm responses.

Maximum bacteria mixing depth (b_d): Studies have shown a sharp decline in bacteria populations beyond a certain soil depth. This value is used to describe the range at which bacteria can survive, and therefore, considered potentially mobile. The b_d coefficient sets the depth above which bacteria is assumed to be active and evenly distributed.

Daily bacteria survival rate (b_{srv}): Bacteria survival is affected by a litany of interrelated factors. Most studies indicate half-live of fecal coliforms in soil varies between 8 and 12 days, although values from each study vary widely. The survival rate was applied as a function of time since introduction (rather than distance traveled). Setting the survival rate to 0.95 causes 5% of all bacteria populations are killed each day system wide. Using this value, 50% of bacteria will have died 14 days after introduction into the system.

iii. Model Initialization

Distributed hydrologic models require calibrating many parameters at many locations, but with only limited observations. Hydrologic observations are usually limited to the discharge data taken from stream gauges at the basin outlet and precipitation data taken from rain gauges scattered over the nearby region. Calibration is performed by inserting a set of input data (i.e. precipitation) into the model and then adjusting modeling parameters to achieve the best fit possible to a set of reference data (i.e. discharge data). The reference data is typically split into two parts: the first half of the data is used to calibrate the model while the second half is reserved to validate those results. Through calibration, modelers rely on 'proxy' parameters that cannot be measured directly, but can otherwise be estimated. Calibration requires adjusting the parameter values until the model output matches observed reference data. Hydrologists accept the model as valid if the model produces results that are sufficiently similar to the second set of reference data (i.e. the set not used during calibration).

While it may seem possible that a superior physical model might bypass calibration by relying on well-established physical relationships and accurately measured field data, a number of factors make this scenario unlikely. First, this model would require presupposing that all parameters are absolutely known and can be directly measured. Second, the operation of this model would be cost-prohibitive, if not impossible, due to the scale and resolution of data required. Complex models, in turn, can contain so many degrees of freedom that it becomes difficult establish the actual validity of modeling results (as discussed in the Chapter II).

a. Discharge Data

Discharge data is important for calibrating and validating the functionality of the model, yet in many watersheds worldwide, no long-term discharge data is available. Such is the case with Camp Creek. The following section describes the methods used to construct a proxy discharge dataset.

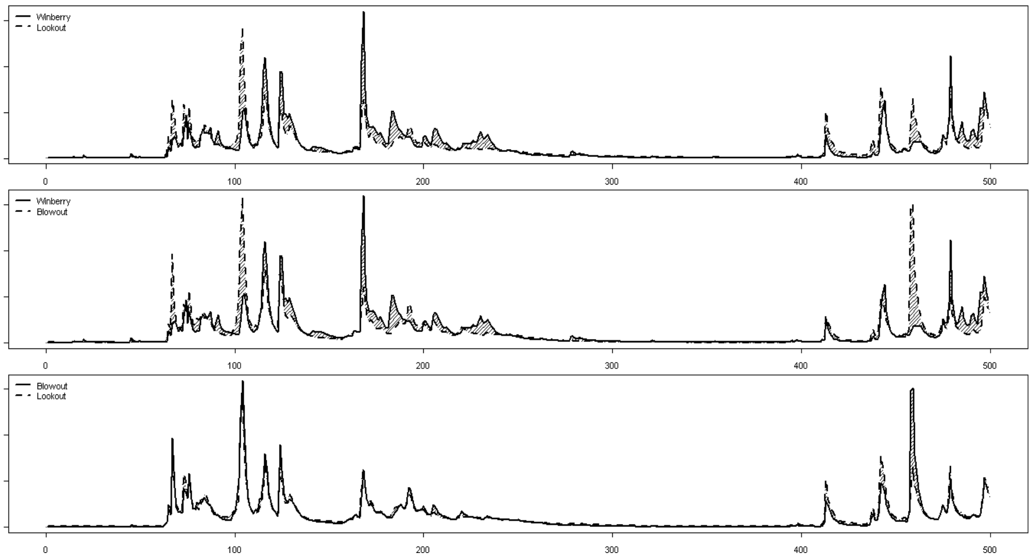
Assuming the central principle of geography – that two nearby phenomenon will tend to be more similar than two similar phenomenon more distant – discharge can be approximated using nearby gauged watersheds (Figure 12). A simple regression model was developed from a small sample of similarly sized watersheds. The sample was created from USGS data using the following criteria: located in central western Oregon, east of the coast and west of the Cascade divide; and with a drainage area between 2,600 and 13,000 hectares. Average daily discharge was modeled against elevation, drainage area, and precipitation. The regression model revealed that the majority of change in discharge could be explained using two variables: drainage area and precipitation ($R^2 = 0.8$, $p < 0.01$). Elevation had only marginal significance ($p > 0.05$).

From the twenty subwatersheds surveyed, Elk Creek was chosen as a proxy data set for Camp Creek due to its proximity, size (7,000 ha vs. 9,000 ha), precipitation (160 cm/yr vs. 142 cm/yr) and elevation (80 m vs. 83 m). Discharge data was scaled using a coefficient of 0.74, as calculated using the previous regression model. Using scaled-data from Elk Creek as a template, discharge in Camp Creek was predicted to peak at 750 cfs and 300 cfs during storm events, fall to between 45 cfs and 23 cfs during intermittent winter dry periods, and then return to between 5 cfs and 1 cfs during the summer dry season.

b. Model Validation

The *Nash–Sutcliffe model efficiency* (NSE) coefficient was used to assess the validity of TOPyMODEL. The NSE measure is functionally equivalent to a regression model coefficient of determination (R^2). The literature contains examples of using NSE to calibrate and validate results for simulations of both discharge and water quality

ure 12. Comparison of basin discharge between three nearby sub-watersheds. terogeneous rainfall patterns, rather than differences in topography and soil nposition, are most responsible for discrepancies between the relative flows of two sub-watersheds at a given point in time.

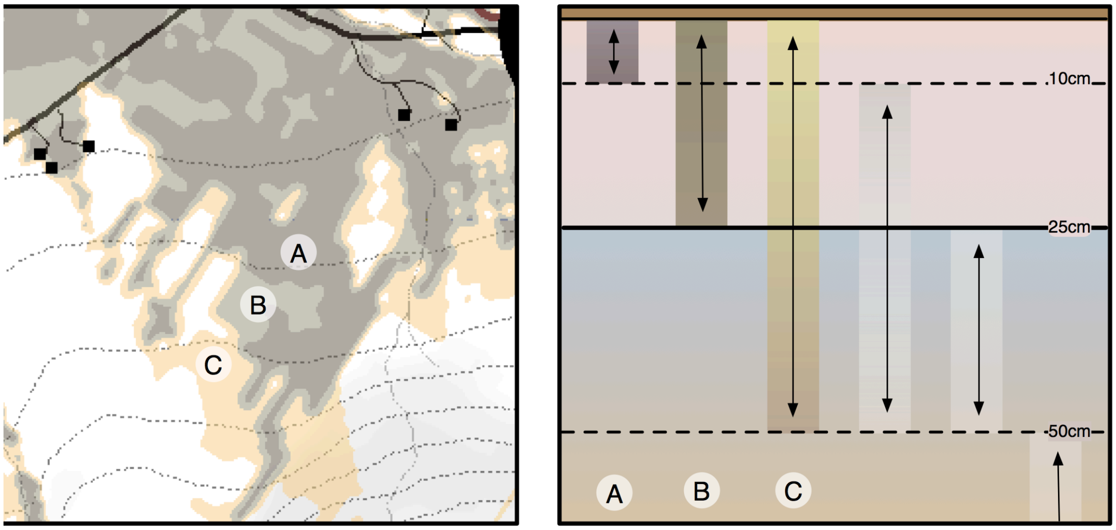


constituents (Chin, et al. 2009). The NSE coefficient can range from $-\infty$ to 1. An efficiency of 1 (NSE = 1) occurs when modeled discharge perfectly matches observed data. An efficiency of 0 (NSE = 0) indicates that the model predictions are as accurate as the mean of the observed data. An efficiency less than zero (NSE < 0) occurs when the residual variance (described by the nominator in the expression above), is larger than the data variance (described by the denominator).

iv. Risk Assessment Framework

Once the modeling output had been verified, hydrologically sensitive soils were identified by mapping saturation dynamics within Camp Creek. Saturation is expressed in terms of saturation deficit, or the depth of precipitation required to completely saturate the soil column. Soils that maintain a lower saturation deficit throughout the year are more likely to become saturated than soils that maintain higher values. Since fluctuations of saturation deficits occur within a definite range, soils can be grouped according to similar hydrologic dynamics (Figure 13).

gure 13. Classification schema used to define hydrologically sensitive areas SAs). Each cell in the watershed was classified according to the rise and fall of the ter table throughout the simulation period, as reported in terms of the amount of ecipitation required before saturation occurs. Highly sensitive classes contained uration deficits that fluctuated between: (A) 0-10 cm, (B) 0-25 cm, (C) 0-50 cm.



Those areas of the watershed actively contributing to pollution loading are those (a) where effluent is present and mobile and (b) where the surrounding area is likely to become hydrologically connected. Mobility was quantified based on the distance effluent traveled before bacteria were eliminated. As such, bacteria loads generated from a home where effluent is extremely mobile creates a higher risk to surface water quality[2]. The attenuation of bacteria was based on a simple first-order decay function set to a 5% mortality rate ($b_{srv} = 0.95$).

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[2] The rate of effluent discharge was set to an arbitrary value. This approach allows the relative contributions of each home septic system to be assessed relatively to one another, holding all other factors equal. This approach does not serve to determine actual loads (as would be required in a TMDL-style approach), nor could it be used to predict build-out capacity based on a maximum concentration limit.

The capacity of soils to ‘treat’ effluent is dependent on the effluent remaining within the soil matrix. If the ground flow containing effluent reaches saturated soils, the dwelling where the effluent originated was flagged as hydrologically connected and assigned a risk index. The index value for a given dwelling was calculated by multiplying the raster representing the concentration of bacteria found in effluent pathways against the HSA raster for subwatershed. All values on the resulting raster were then summed and assigned to the risk index. The exact degree that a HSA class affected the total sum was based on a simple assumption that each class was twice as likely to result in return flow as the next lowest HSA class. It is important to note that the risk index does not mean that loading actually occurs, but rather that contamination is much more likely to occur if a septic system begins to fail, or during very wet times of the year.

ɔle 1. Water quality trends in Camp Creek. Water quality indicators were gged for future analysis if longitudinal trends were statistically significant. cteria concentrations showed the strongest fit of all indicators, and were thus ected as the principal water quality indicator used to track the impacts of idential dwellings on water quality in this study.

CHAPTER IV

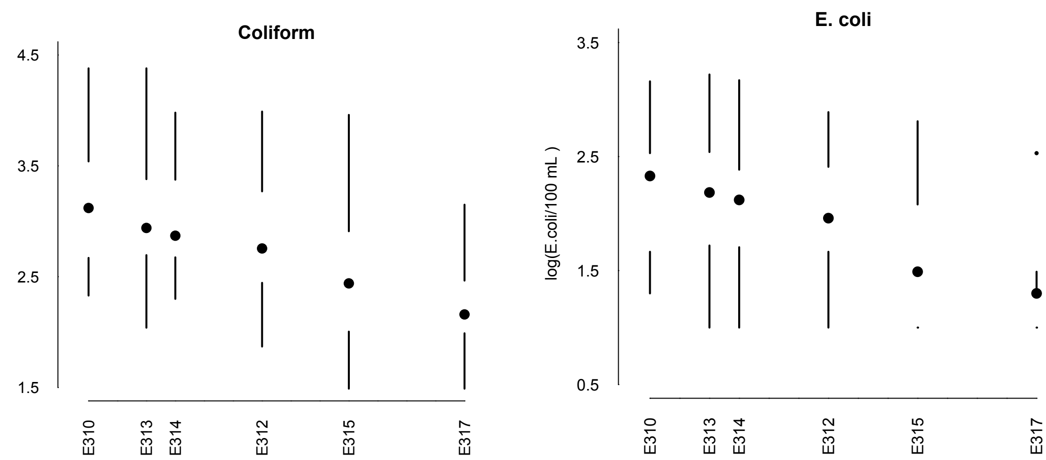
FINDINGS

.RIABLE	DESCRIPTION	P		R^2	+/-
:al coliforms‡	Number of colonies per 100mL	< 0.001	***	0.27	+
:oli‡	Number of colonies per 100mL	< 0.001	***	0.43	+
nductivity	Electric conductivity	< 0.001	***	0.12	+
:bidity	Cloudiness of water	< 0.001	***	0.08	+
$)_2$ saturation	Percent of potential capacity	< 0.01	**	0.02	-
	Acidity of water	> 0.05		n/a	
rates	Water-soluble NO_3	> 0.05		n/a	
ɔsphorous	Soluble & particulate states	> 0.05		n/a	

‡ Log transformed

Most watersheds are more developed at their lower elevations, closer to the basin outlet. Thus, water quality data that shows significant and highly correlated longitudinal trends can indicate water degradation that results from upstream development. Table 1 reveals that concentration of total coliform and *E. coli* were strongly correlated with upstream drainage area ($p < 0.001$, $R^2 = 0.27 / 0.43$ respectively), but contained much less noise than other physical measures associated with drainage, including conductivity, turbidity and oxygen saturation ($R^2 = 0.27 - 0.43$ vs. $R^2 = 0.12 - 0.02$ respectively).

ure 14. Longitudinal trends of bacteria concentration along Camp Creek. liform and *E. coli* measures are inversely related to the distance of monitoring tion from the basin outlet (see Figure 5). Concentrations are log transformed.

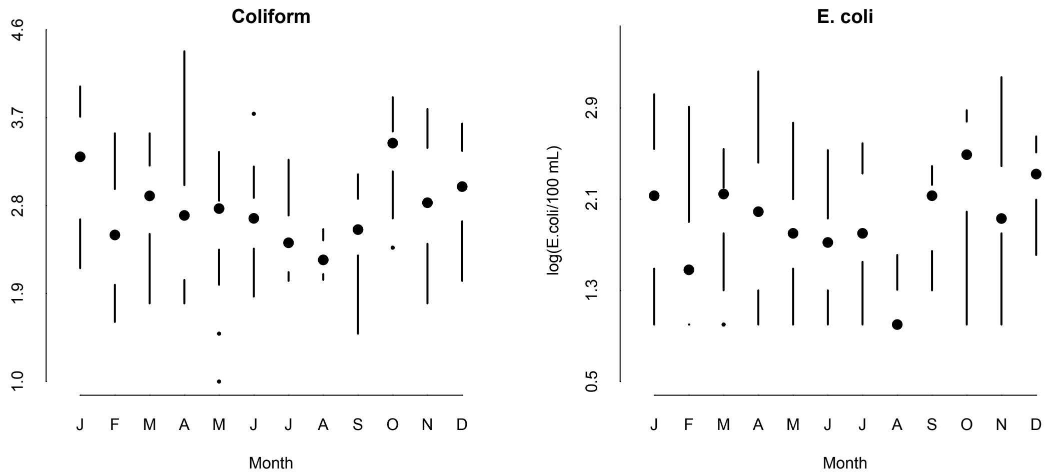


ed no significant

tal coliform and *E.*
4 shows how
n *E. coli* and total
quality data taken
t correlated in the
ssociation (p =
ne larger basin,
and moreover, *E. coli* concentrations completely unrelated to upper tributary drainage area (p = 0.65); in strong contrast to the high significance observed in the lower drainage (p < 0.01). These results strongly indicate that bacteria concentrations, especially those of *E. coli*, are linked to agricultural land uses and / or residential development found in the lower two thirds of the watershed.

Total coliform and *E. coli* fluctuate with the time of the year. Figure 15 reveals that: (1) highest bacteria loads are found in October; (2) bacteria decline through the winter and springs months (December through April); (3) counts begin to rise at a gradual

**ure 15. Annual fluctuations of bacteria concentration in Camp Creek. Mean
icentration of coliform and *E. coli* peak in October and are lowest in April.
nter months show greater variation between maximum (top of line) and
nimum readings (bottom of line). Concentrations are log transformed.**



rate during the summer. Concentrations of bacteria in Camp Creek were significantly higher during the hottest four months of the summer (July through October) than the rest of the year (Total Coliform: $p < 0.01$; E. coli: $p = 0.03$), suggesting that summer water temperatures may cause regrowth in bacteria populations (SCCWRP 2008). Bacteria concentrations were more varied during the wetter and cooler seasons of the year ($p < 0.01$), suggesting rapid but temporal changes immediately following storm events (ibid).

Bacteria concentrations therefore appear to be associated with development, but the data are inconclusive as to its source. Each development indicator (e.g. number of upstream residential dwellings vs. acreage of upstream agricultural lands) added only limited predictive power to the regression model ($R^2 = 0.33$ from $R^2 = 0.27$, $p < 0.01$) once seasonal fluctuations in bacteria levels were accounted for. However, no benefit was observed from combining multiple development variables ($p > 0.1$). Thus, empirical data *does not* indicate that residential development contributes to pollution loads to a greater degree than other anthropogenic sources of bacteria, namely hay fields and pasturelands.

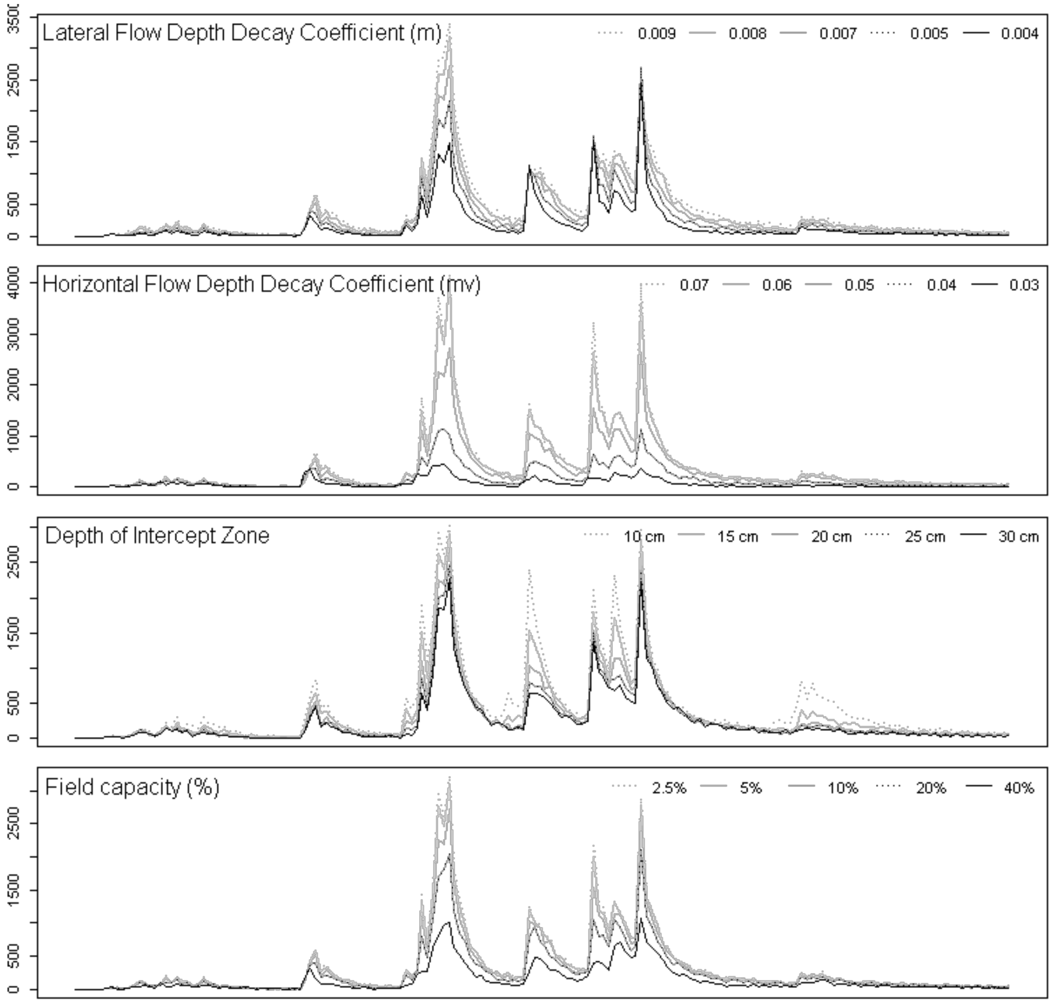
B. Calibration and Validation of TOPyMODEL

Using the concepts of subsurface flow and hydrologic conductivity simulated by TOPyMODEL, saturation levels can be calculated for any point in the landscape. Saturation levels are reported as saturation deficit, or the amount of water necessary to fully saturate the soil in that cell. Due to their location in the watershed, some areas require very little moisture in order to become completely saturated. As a result, these soils contribute more to overall discharge – they are more hydrologically active. A hydrograph describes the summation of these processes. By calibrating a model so that simulated output closely matches observed discharge, those hydrological processes can be spatially mapped upstream of the gauging station. The following section describes observations from the calibration of TOPyMODEL to historic discharge data.

Calibrating the hydrologic model requires adjusting unknown parameters until simulated discharge closely matches a reference set of discharge data. Calibration was achieved by adjusting the four parameters (Figure 16). The calibrated values used in the final simulation are indicated in each row of the hydrograph. Sensitivity analysis provide additional insight into how variables work in response to each other, and, which parameters exert greatest control over simulated output.

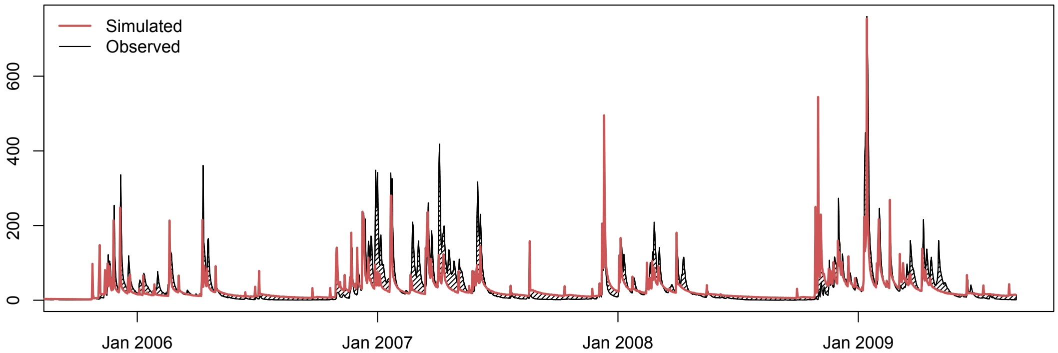
Whereas the depth decay coefficient (m_v) exerts strong control over simulated discharge throughout the season, variations in the third and fourth parameters highlighted above are more specific to certain times of the year. For instance, the effective intercept capacity (S_i) impacted discharge most pronouncedly after periods of little rain. Likewise, little change was observed during those times of the year when precipitation had already saturated the intercept zones. Adjusting field capacity (fc) had a somewhat different result. Higher field capacities greatly reduced discharge during winter storm events but had little impact on the baseflow simulated during drier parts of the year.

gure 16. Discharge sensitivity to changes in modeling parameters. Sensitivity of scharge is shown in relation to four model parameters: depth decay coefficient lateral flow (m); depth decay coefficient for vertical drainage (m_v); storage pacity of the intercept zone (s_i), and; field capacity (fc).



I validated the calibrated TOPyMODEL against observed flow data from between September 1st, 2005 and August 31st, 2009 (Figure 17). The Nash-Efficiency Coefficient was used to assess the predictive power of the hydrology model. Simulated results

ure 17. Hydrograph of TOPyMODEL validation run. The TOPyMODEL was idated against a 3-year set of data from September 1st, 2005 to August 31st, 2009. e NSE coefficient is 0.55 and was accepted as a satisfactory fit for observed data.



.55)[3]. The NSE
e data than would
ıer, the NSE value
Γ and HSPF

C. Hydrologically Sensitive Areas of Camp Creek

i. Incidence of Hydrologically Sensitive Areas

In many watersheds of the Pacific Northwest, most runoff originates from a relatively small proportion of soils in the basin that are saturation-prone. As such, soils can be most broadly classified within one of two groups: soils that may become saturated,

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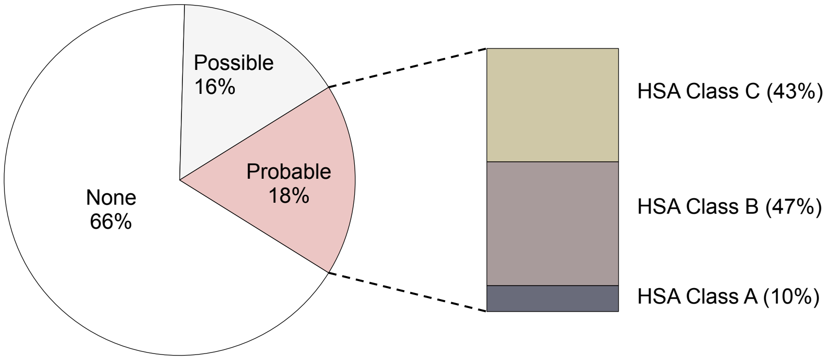
[3] Note that the NSE test is more sensitive to differences in values during extreme flow periods of the year. Some studies log transformed discharge flows to accommodate for this effect. Since it is often during peak discharges that water quality becomes most impacted, I chose to preserve these peaks and leave data untransformed.

ure 18. Composition of hydrologically sensitive soil (i.e. HSA) classes within mp Creek (refer to Figure 13).



Creek (~66%)
r, these soils
lows. Unsaturated
cit of these soils is
of 17 cm.

that are potentially active (i.e. HSAs). These areas generally conform to floodplains and stream corridors, but notable exceptions exist. Saturation-deficits within these zones demonstrate a slightly left-skewed distribution that range between 0 cm and 40 cm, with a mean deficit of 20 cm and a standard deviation of 9 cm.

Not all saturation-prone soils are equally likely to produce runoff. Soils that maintain a lower saturation deficits throughout the year are more likely to generate runoff. Further, when saturation deficits at a given point vary less, the soil takes less time 'wet-up' after long dry spells. As wet periods increase in magnitude or duration, a greater proportion of HSA soils contribute in basin discharge. During the simulation period used in this study, slightly less than half (45% of all HSAs or 15% of basin area) of all HSAs became completely saturated (i.e. 'HSA-probable soils'). On the other hand, 'HSA-possible soils' (55%) never saturated, but maintained saturation deficits small enough that complete saturation *could* occur during extreme wet periods. Again, HSA-possible soils are classified based on variation in saturation values.

One third (33%) of *HSA-probable* soils (15% of all HSAs or 5% of basin area) exhibited large swings in saturation deficit values throughout the year, functioning as *variable source areas* (VSAs). For the majority of the year, VSA soils maintain large saturation deficits. Yet after a large storm, these soils cause a rapid increase (50%) in the areal extent of saturated soils, sharply increasing overall basin discharge. The majority of these transitional cells are located in the mid to upper half of the watershed and found above the valley floor (see below).

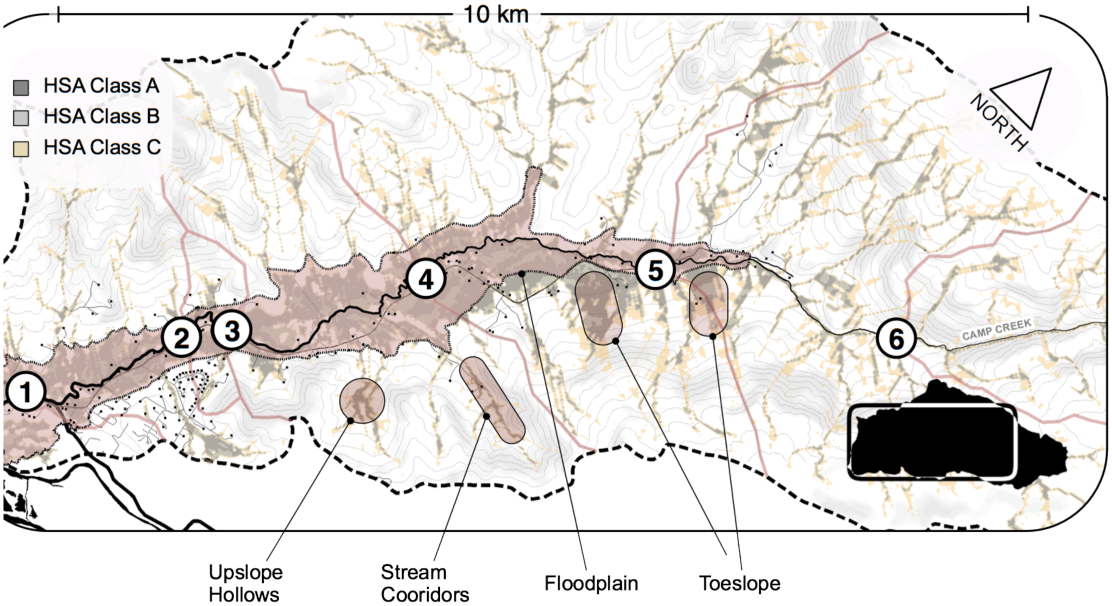
ii. Spatial Distribution of Hydrologically Sensitive Soils

Since previous studies have demonstrated that runoff is associated with riparian areas at the hillslope level (Dosskey, et al. 2002), HSA soils should coincide with the general form of riparian areas of a watershed – a broad trunk that covers the valley floor, tapering as it rises, and subsequently branching into multiple hillslope corridors. Indeed, TOPyMODEL results generally conform to this prediction and show that HSA soils can be geographically classified according to four zones: the floodplain, mid-valley toeslopes, riparian corridors, and upslope hollows (Figure 19).

One fifth (21%) of HSA soils lie within the 100 year floodplain. More than four fifths of the areal extent of the floodplain is hydrologically sensitive (83%). Saturation conditions tend to fall drastically in soils immediately above the floodplain, making for a sharp transition between HSA floodplain soils and surrounding hillslopes (see discussion on *Mid-valley toeslopes* below). While the floodplain is largely contiguous with saturation-prone soils, a number of isolated patches remain unsaturated throughout the year. These 'dry islands' are formed from natural levies, and because of the way that TOPMODEL models subsurface flow, cause subsurface-water to flow around these areas. Builders of both roads and homes have historically taken advantage of these relative high points.

Two fifths (41%) of HSA soils in Camp Creek occur in narrow corridors 30 m to 90 m in width, and which climb from the valley bottom into surrounding hillsides. While

gure 19. Hydrologically sensitive areas in lower Camp Creek. Darker-shaded soils come saturated before lighter-shaded soils.



nearly all streams recorded in GIS by the county overlie these saturated corridors, results showed an additional 66% saturated corridors than would be predicted by streams alone. Since saturation conditions of saturation corridors that carry perennial streams differ little from those that carry intermittent streams, results indicate that the existing stream networks may be inadequate for managing water quality risks.

Approximately 160 hectares of HSA soils located mid-valley (7% of all HSAs) is worth noting, since this swath rises into the surrounding foothills yet is not confined to narrow hillside corridors. These ‘saturated toeslopes’ appear to be caused by a gradual decrease in hydrologic conductivity, which differs markedly from the sharp transition between floodplain and hillslope soils (noted above).

Much of the remaining 31% of HSA soils in Camp Creek occurs upslope of the saturation corridors in laterally concave hollows. Since saturated hollows form upslope of saturation corridors, these broad zones are geographically separated from both the floodplain and toeslope saturation areas. For this same reason, saturated hollows also occur away from development.

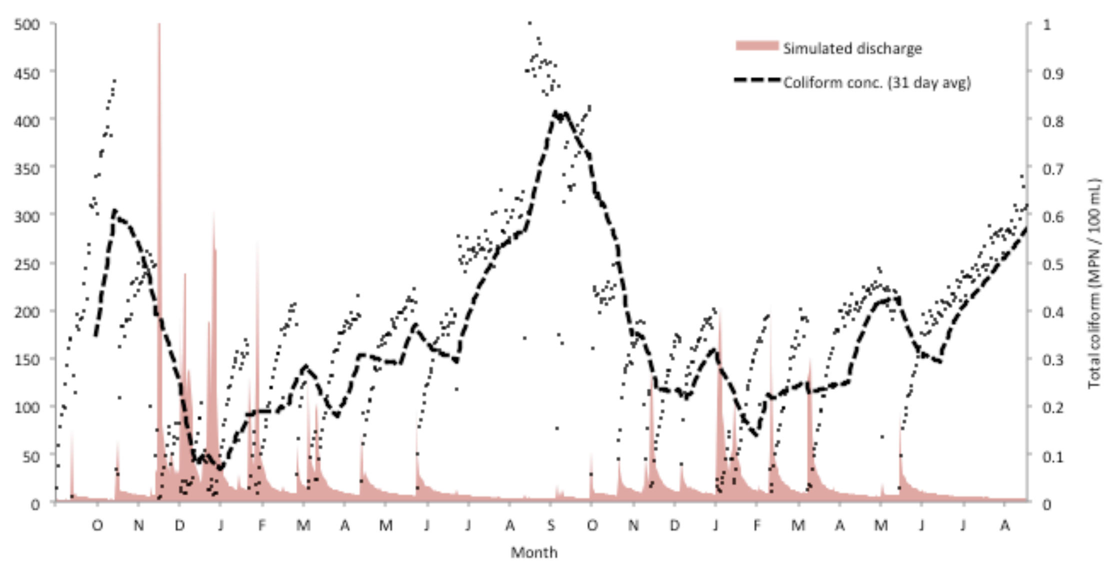
D. Bacteria Transport and Loading in Camp Creek

i. Simulated Bacteria Loads in Camp Creek

Bacteria levels can be described according to absolute load (i.e. total number of bacteria) or by concentration (i.e. number of bacteria per 100mL flow). Simulated bacteria load were highest during times when flow is high (i.e. during the wet season and during storm events), and are lowest during the end of summer (Figure 20). Results further show that bacteria loads respond more quickly to rainfall than respective changes in discharge, that loads increase quickly reach a maximum threshold, and that variation in bacteria loads is much less than the respective variation in discharge.

Unlike absolute bacteria load, simulated bacteria concentration are highest when flows are lowest. Concentrations drop precipitously during storm events due to dilution, ... ir peak level ... : two orders of

ure 20. Simulated annual fluctuations of bacteria concentration (black dots) in mp Creek, shown between September 1st, 2006 and September 1st, 2008. Dashed : shows the 31 day running average of bacteria concentration against simulated charge (red). Highest relative concentrations occur in October and September. mpare to temporal patterns in observed bacteria concentration (Figure 15).



magnitude, which agrees with water samples taken from Camp Creek (see Figure 15) and variations reported in other watersheds (e.g. SCCWRP 2008).

ii. High-Risk Dwellings in Camp Creek

The distance that bacteria travels from its point of origin is directly related to underlying subsurface conditions. Effluent that is generated in hydrologically sensitive areas is more likely to enter cells that are also prone to saturation, is more likely to resurface, and is therefore able to reach water bodies that quickly carry the bacteria out of the basin.

TOPyMODEL shows that more than one third of primary dwellings in Camp Creek (n=79) create a high risk of surface water contamination because of their location (Figure 21). Ten percent of these dwellings (n=23) presented only marginal risk; effluent produced at these sites traveled offsite, but was eliminated within 10 m of the drainfield. Slightly more than half of these dwellings (n=45) are located in HSA-probable soils, where complete saturation occurred at least once during the simulation period. Effluent from these highest-risk dwellings traveled between 20 m and 60 m before being eliminated during the simulation run[4].

Dwellings built in saturation-prone soils are more common in upper parts of the watershed than lower. Over 50% of primary dwellings are located in HSA soils above stations E314 and E312. What is more, approximately 25% of these dwellings are located in VSA-class HSA soils – areas where the propensity of the soil to become saturated may be missed during onsite evaluations due to large annual fluctuations in the local water table throughout the year. In general, newer dwellings are less likely to have been built in HSAs than older dwellings. For instance, between 1930 and 1960, more than half of

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[4] As a general rule for these pathways, bacteria loads decreased by an order of magnitude for every 10m traveled.

ure 21. Map of dwellings that create greatest water quality risks. Water quality ks result from a minority of dwellings. Some dwellings create greater tamination risks due to the propensity of surrounding soils to become saturated. ellings that create contamination risk are marked as dark circles. Larger circles oict greater risk.

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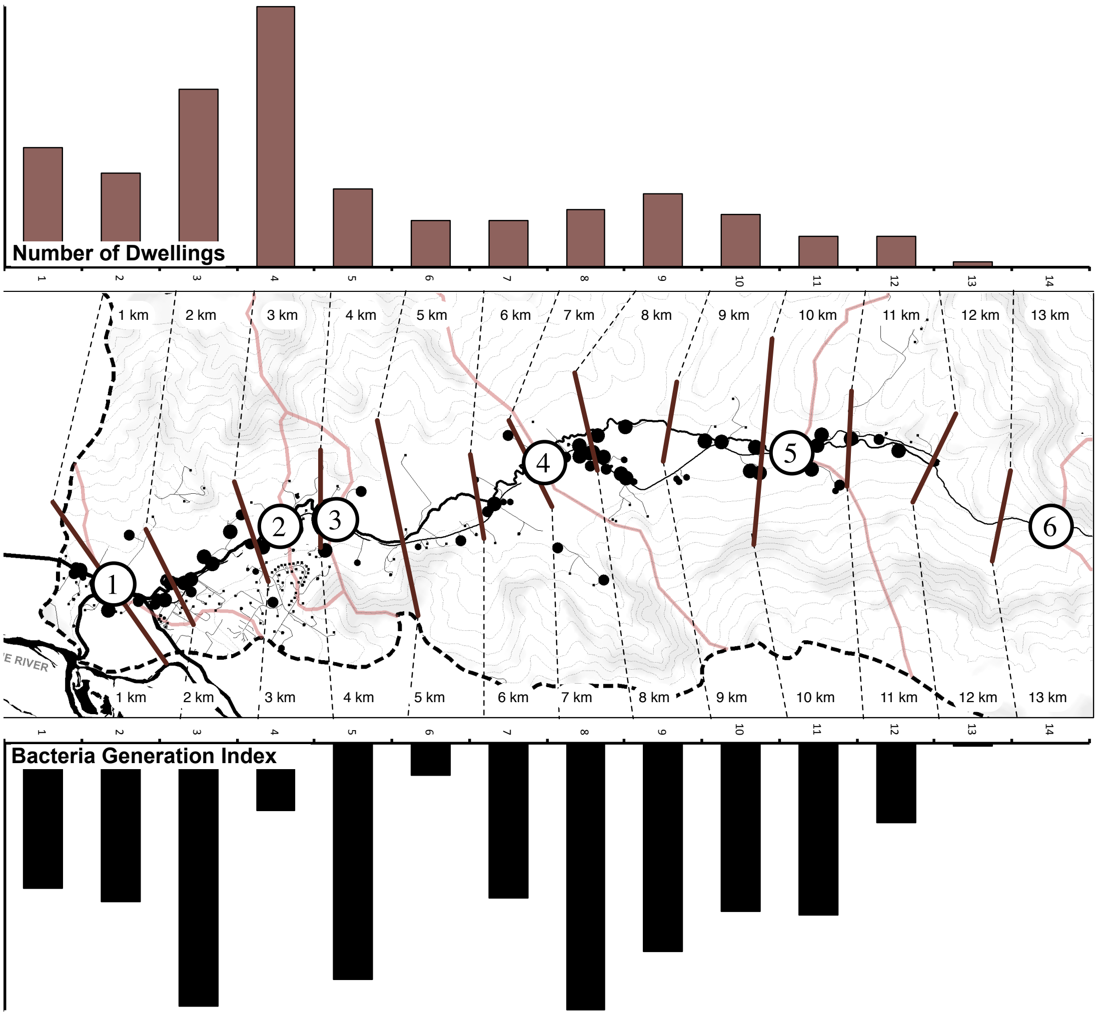
ited regions.



CHAPTER V
DISCUSSION

A. Planning and Private Property Rights

All land use planning evolves from nuisance law—the general prohibition that one property owner interfere with the rights enjoyed by their neighbor, whether by the creation of smells, sounds, or pollution. Some nuisances may become so widespread that they impact the general health, safety and welfare of a community. Government may 'police' such public nuisances accordingly. Actions that impact drinking water clearly qualify as a public nuisance.

The unreasonableness of a nuisance is both normative and contextual. Lacking a formal definition, courts have considered the nature of the offending act, including how long, how bad, and the impacts of the nuisance (D. Mandelker 2005). Resolving nuisances in a case by case manner is resource intensive and comprehensive land use plans provide a system for proactively avoiding nuisance claims by systematically defining community values. Most local governments in the US now implement some form of comprehensive land use planning.

Oregonians affirm that natural beauty, recreational opportunities, and environmental quality are important ingredients to their quality of life (Davis and Hibbits, Inc. 1996). Such elements are strongly aligned with Oregon's rivers and lakes (ibid). Polls further suggest that of all the qualities associated with Oregon's waterways, clean rivers and clean drinking water are those values enjoyed most (Davis, Hibbits and McCaig, Inc. 2002). At the same time, housing consumers—the same members of the public who are worried about the costs of sprawl—demand and monetarily value homes that are located in those areas that impact water-related values most (Ziegler 2003). These contradictions of public values are consequently translated into local comprehensive land use plans.

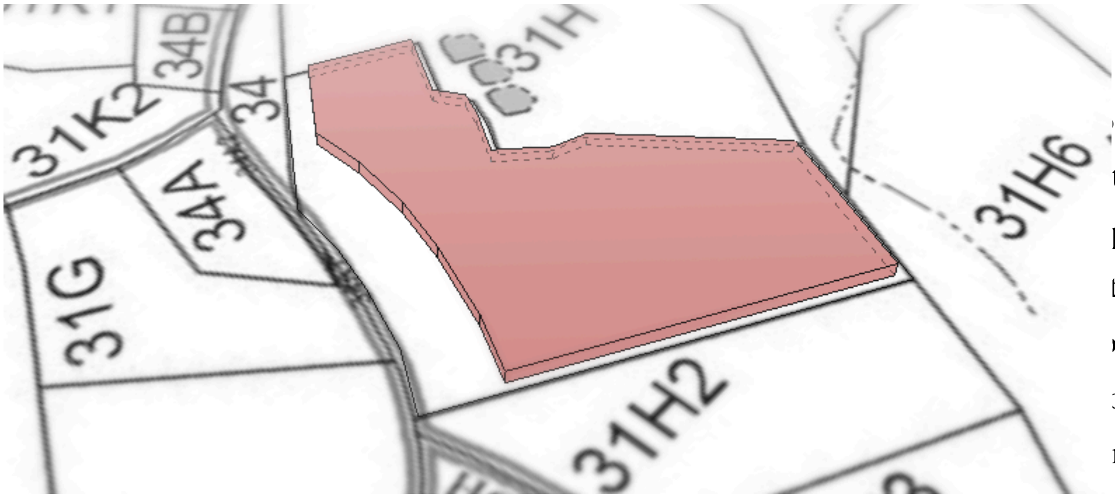
B. Connecting Scale and Function Between Clean Water and Land Use

Such contradictions also result from the difficulties of matching the various scales of hydrologic systems with the various scales of land use planning and regulation (Arnold, Clean Water-Land Use 2006). While land use planning may gain its legal-basis from its capacity to minimize hazard to public welfare, planning systems often become politically feasible only when they preserve and enhance existing land values. Limitations of private land rights are felt acutely by landowners in the present, while costs suffered by the public appear abstract and distant. As such, local governments arguably "have incentives to promote development, especially when the impacts are downstream, or at least outside, the political jurisdiction of the local unit of government" (D. Mandelker 1989).

Many public nuisances emerge only at the landscape scale and often span more than a single jurisdiction. Local governments are often hamstrung limiting nonpoint source pollutants, which place private land rights against the complex task of linking specific activities with specific impacts that are both spatially and temporally disconnected (Novotny and Olem 1994). As a result, landscape-level goals meant to protect water quality are often left to state or federal mandates, resulting in inventory systems that are narrow in focus. As a result, federal and state governments are increasingly relying on regional performance standards (i.e. Total Maximum Daily Limits) to manage water quality and nonpoint sources of pollution. Between 1998 and 2008, the number of TMDLs accepted by the EPA grew by 50% each year (USEPA 2010). These standards, in turn, will obligate local governments to address land use with a broader lens.

The emerging nexus of land use planning (where decisions occur at the site-scale) and water quality management (where decisions occur at the landscape scale) seeks to maximize land value *without* compromising water resources (Arnold, Wet Growth 2005). The land use planning techniques traditionally applied by local governments are inherently limited in protecting water quality in the absence of more a integrated, system-based approach.

,ure 22. Building envelope defined by setbacks on a typical rural lot. As can be n from the example below, minimum setbacks in rural areas often do not address ilding siting relative to the intermittent streams or wetlands found off property.



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Regional planning goals (e.g. wildland fire safety, urban containment, protecting natural resource lands, and drinking water) frequently require local governments to maintain rural development at low densities. However, local governments often focus solely on maintaining minimum lot sizes on purposed land divisions – a policy approach that is often seen as least restrictive and most politically feasible. In the absence of additional controls, development may still concentrate in areas of the lot that create higher risks to water quality, as buildable envelopes remain quite large (Figure 22). As this study has shown, the poor siting of development can exacerbate water quality impacts by many orders of magnitude.

The development envelope remains the most direct means for controlling the siting of primary and ancillary structures (PCGDER 2000). Development envelopes in rural areas are typically defined through two approaches: natural resource overlays and setbacks. Yet rural development envelopes may be often ineffective at protecting water quality as commonly applied. First, while natural resource inventories can protect sensitive lands from certain land uses, their design is often externally motivated and limited to a very specific focus. As a result, many environmental overlays fail to capture the dynamics of natural systems. For instance, floodplain restrictions pertain only to preventing structures from impeding waters during a 100-year flood event. Few floodplain ordinances will regulate subsurface structures (e.g. septic tanks), even though flood plains are hydrologically active. Second, setback rules in rural areas often originate, not from environmental concerns, but from the desire to preserve the rural character of the landscape.

Today, it is also common for local governments to use setbacks to establish uniform width buffers around environmentally sensitive areas, such as wetlands, streams and steep slopes. While buffers do afford protection to sensitive areas, uniform-width setbacks are devices that have evolved from a framework of urban land use planning. Extrapolating their use to the protection of natural resources may result in rules that may be arbitrary from a scientific standpoint, water quality protection measures that have little physical basis, some areas along the stream being under protected and other areas overprotected (Dosskey, et al. 2002).

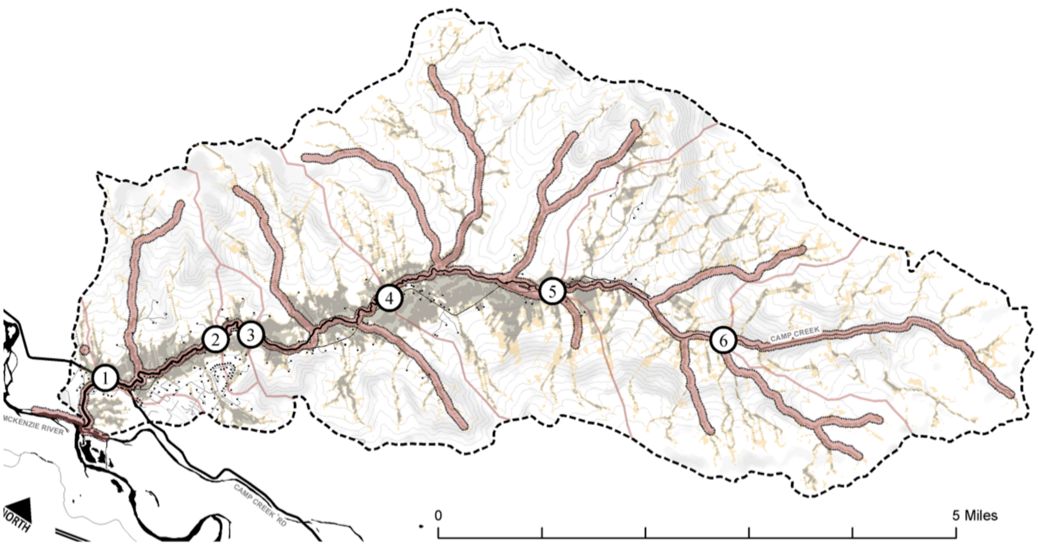
D. The Potential for a Hydrogeomorphic Planning Approach

To be more effective as a tool for protecting water resources, it is important that building envelopes be sculpted based on an integrated watershed model. This argument has also been addressed within research of variable-width buffering, which varies buffering distances on a set of related environmental variables (e.g. slope, soil, etc). Their use has been discussed broadly in context of habitat conservation (Berry, et al. 2003) and management of agricultural lands (Polyakov, Fares and Ryder 2005), but is equally appropriate to managing rural residential development. Integrated approaches for protecting natural resources are not only more physically accurate, but are better at identifying an optimal balance between development and land conservation.

This study has shown how a relatively simple distributed watershed hydrology model can be used to quickly and accurately delineate those areas inappropriate for certain types of development. By defining hydrologic risks in regard to hydrologically sensitive areas, this study also suggests how management of water quality risks can be focused on areas where targeted land use coincides with locations were discharge is generated. Not surprisingly, this study shows that the majority of water quality risk within a basin comes from a minority of residential dwellings. In the Camp Creek basin, for instance, approximately one third of dwellings constitute the entirety of risk to water quality from residential sources. Further, results from this study suggest offsite saturation conditions are often just as important to understanding pollution transport as conditions on the property itself. Effluent commonly carried bacteria 20-60 m from their point of origin; in extreme instances, pollution pathways extended up to 200 m.

Hydrologically sensitive areas in Camp Creek are found predominately within floodplains and river corridors, a relationship that has been well documented in the scientific literature. These areas should be protected from future development. Yet maps of hydrologically sensitive areas produced in this study show that current inventories of stream networks are not comprehensive predictors of sensitive lands; approximately 40% of saturated corridors outside of the Camp Creek floodplain were not officially protected in a recently purposed drinking water overlay (Figure 23). TOPyMODEL also predicted the existence of extensive areas of saturated lands that exist outside of both floodplains and riparian corridors. Further, approximately 50% of saturated soils are highly variable in nature, making these areas more difficult to identify during the permitting process.

ure 23. Purposed 2010 Lane County Drinking Water Overlay. Maps shows how st saturation-prone areas are left unprotected.



<u>E. Conclusions</u>

Hydrologic modeling allows for numerous variables related to the siting of new development and water quality to be simultaneously considered over an extended geographic area. This study successfully related changes in basin discharge with fluctuations in bacteria concentrations based on the presence and location of septic systems, thereby indicating the physical processes involved along specific pollution transport pathways. This study has further suggested how complex loading processes can be explained according to a relatively simple set of rules, and how those rules allow for a more integrated means of assessing water quality risks and regulating future development.

The fact that TOPyMODEL successfully approximates bacteria fluctuations supports cursory finding from water samples: waste from residential dwellings play and important role in bacteria loading, but that not all dwellings create equal risk to water quality. Moreover, since modeling results indicate a strong delineation of hydrologically sensitive soils (actual, probable and potential), this study suggests a spatially-specific means for regulating areas inappropriate for development.

Still, these predictions also imply potential and significant impacts on both existing property owners and the distribution of future population. It is important that the modeling results described in this study be further substantiated. What is more, this study did not account for all sources of bacteria. It does not rule out the contribution, nor the importance of understanding, bacteria coming from agriculture and pasturelands. However, such modeling approaches could easily be applied to pasturelands, although may require considering different transport pathways, particularly those above ground.

Since the TOPyMODEL process allows delivery pathways to be identified recursively, it also provides a means for identifying sections of the main stem of Camp Creek where concentrations of bacteria should increase markedly. Study results could be further validated by taking water quality samples above and below these reaches of Camp Creek where TOPyMODEL indicates concentrated pollution loading is occurring. If samples showed a significant increase in load, especially as compared to the basin-wide loading rate described in this study, results would further indicate that transport and hydrological processes suggested by the model are correct.

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