AN ABSTRACT OF THE DISSERTATION OF

Cara J. Poor for the degree of Doctor of Philosophy in Civil Engineering presented on May 31, 2006.
Title: The Effects of Land Use on Stream Nitrate Concentrations: From the Catchment Scale to the Regional Scale

Signature redacted for privacy.

Abstract approved: ______________________________

Peter O. Nelson

This work furthers the understanding of processes occurring in catchments that affect stream nitrate concentrations using two different approaches: a temporally intensive case study of three headwater catchments with varying land use (through storm event monitoring) and a spatially intensive study on the regional scale (through statistical modeling) of 1st- 4th order catchments. At the catchment scale, stream nitrate concentrations during three storm events were monitored in three catchments with different land uses (forested, agricultural, residential) to determine how land use affects nitrate "patterns" during storm events. Overall, results of storm event nitrate concentrations suggest that varying nitrate inputs have a large affect on nitrate dynamics. While within-storm nitrate concentration response patterns in the residential catchment were the same as the patterns in the reference forested catchment (a "concentration" pattern throughout the year), a "dilution" pattern was observed in the fall and winter and a "concentration" pattern was observed in the spring in the agricultural catchment. At the regional scale, a statistical model was developed using land use and either topographic index (TI) or hydrologic landscape regions (HLRs) to predict stream nitrate concentrations during lowflow. Including TI and HLRs (in the form of primary hydrologic flowpaths) significantly improved chloride predictions,

but did not improve nitrate predictions. Results of the linear regressions imply that the hydrologic setting of the catchments are adequately represented (from chloride, which is tightly linked to hydrology), and nitrate is more strongly affected by processes such as denitrification and plant uptake during lowflow. Agricultural effects were seen both on the smaller catchment scale and the regional scale. Different patterns were observed in the agricultural catchment during storm events, and chloride was elevated in the Willamette Valley where agricultural activity is concentrated. The temporal pattern of nitrate during storm events was found to be largely controlled by the spatial organization of land cover, whereas the spatial pattern of land cover did not control stream nitrate concentrations sufficiently to improve predictions of nitrate during lowflow. Future work should determine whether or not the spatial pattern of land cover, TI, and HLRs improves nitrate predictions during storm events.

The Effects of Land Use on Stream Nitrate Concentrations: From the Catchment Scale to the Regional Scale

by
Cara J. Poor

A DISSERTATION

submitted to

Oregon State University

in partial fulfillment of
the requirements for the
degree of

Doctor of Philosophy

Presented May 31, 2006
Commencement June 2007

Doctor of Philosophy dissertation of Cara J. Poor presented on May 31, 2006.

APPROVED:

Signature redacted for privacy.

Major Professor, representing Civil Engineering

Signature redacted for privacy.

Chair of the Department of Civil, Construction, and Environmental Engineering

Signature redacted for privacy.

Dean of the Graduate School

I understand that my dissertation will become part of the permanent collection of Oregon State University libraries. My signature below authorizes the release of my thesis to any reader upon request.

Signature redacted for privacy.

Cara J. Poor, author

ACKNOWLEDGEMENTS

Support for this research was provided by the United State Geological Survey, the Center for Water and Environmental Sustainability, and a cooperative grant from the U.S. Environmental Protection Agency (USEPA).

I particularly wish to thank Jeff McDonnell, who acted as research advisor and has provided a lot of guidance and encouragement to complete this research. Pete Nelson, as my major advisor, was very helpful during my time in the laboratory and during this whole process. I would also like to thank John Bolte for finding funding for the last two years of this work, and for all of the modeling help he provided. Wayne Huber, John Selker, and Richard Cuenca deserve thanks for the knowledge they have imparted through classes and conversations, which has significantly improved my understanding of hydrology and science and thus contributed to this work.

I would also like to thank the many student colleagues that have supported me throughout this experience. Students in Jeff McDonnell's Hydrolab, especially April James, contributed significantly through conversations of hydrogeochemical processes. Kellie Vache vastly improved my knowledge of GIS and ArcHydro, without which I would not have been able to complete several analyses. Finally, I would like to thank my family for their tremendous support. In addition for encouraging me and always supporting me with everything I do, my father provided me with valuable practical experience and financial support by letting me work for his firm. Thanks Dad, without you I would not be who I am. Not only has my husband, Aaron, been a great emotional support, he helped me with many field installations and

flow gauging. He has been by my side during the tough times, and always been readily available to help me blow off steam with a hike or bike ride. And Ella has been the joy of my existence the last 12 months, and been good enough to let me finish this degree.

CONTRIBUTION OF AUTHORS

Jeff McDonnell was instrumental in the development of Chapter 2. Dr. McDonnell also provided extensive editing and suggestions for the content in Chapters 3 and 4. John Bolte participated in the initial development of many components making up the models outlined in Chapters 3 and 4.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

TABLE OF CONTENTS (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF TABLES

LIST OF APPENDIX TABLES

The Effects of Land Use on Stream Nitrate Concentrations: From the Catchment Scale to the Regional Scale

Chapter 1

Introduction

Background

Nitrogen inputs have increased sharply in the last 50 years, doubling in the United States from 1961 to 1997 (Pimental, 1993; Howarth et. al., 2002). This has caused great concern for the health of stream ecosystems. In general, approximately one-third of nitrogen inputs to catchments are exported, with the majority exported to surface waters (Howarth et. al., 2002). This increase in export to surface waters has caused a significant increase in ecosystem degradation. Algal blooms have resulted from upstream agricultural fertilizer runoff, which in turn cause hypoxia and "dead" zones for fish (National Science and Technology Council, 2000; Rabalais et. al., 2002; National Research Council, 2000). Episodic acidification of streams has also resulted from increased nitrate levels (Wigington et. al., 1996a; Wigington et. al., 1996b; Wellington and Driscoll, 2004). Degradation of stream and surrounding ecosystems has created the need to study how land use is affecting the natural system and what we can do to repair or prevent some of the damage.

Not surprisingly, land use has been found to have a large effect on the amount of nitrogen exported to the stream (Salvia-Castellvi et. al., 2005; Schilling, 2002; Jordan et. al., 1997; Owens et. al., 1991; Howarth et. al., 2002; Jordan and Weller, 1996; Johnson et. al., 1997; Herlihy et. al., 1998; Wernick et. al., 1998; Arheimer and Liden, 2000; Jones et. al., 2001; Wayland et. al., 2003; Donner et. al., 2004; Woli et. al., 2004; Buck et. al., 2004; Lattin et. al., 2004; Little et. al, 2003). The majority of the work on land use effects has focused on baseflow or a small number of sampling events correlating land use and nitrate (Johnson et. al., 1997; Herlihy et. al., 1998; Wernick et. al., 1998; Arheimer and Liden, 2000; Jones et. al., 2001; Wayland et. al., 2003; Donner et. al., 2004; Woli et. al., 2004; Buck et. al., 2004; Lattin et, al., 2004; Little et. al., 2003; Schilling, 2002). While it is clear that land use affects the magnitude of nitrate and other nutrients exported from catchments, it is not clear how it affects nutrient dynamics throughout the catchment.

Recent studies in watersheds with a single land use (forested or agricultural) reveal distinct nitrate patterns during storm events. These studies either show a "concentration" pattern, where nitrate concentrations increase with increasing flow rates and essentially mimic the storm hydrograph, or a "dilution" pattern, where nitrate concentrations decrease with increasing flow rates as a mirror image of the hydrograph (Salvia-Castellvi et. al., 2005; Bolstad and Swank, 1997; Webb and Walling, 1985; Petry et. al., 2002; Vanni et. al., 2001; Inamdar et. al., 2004; McHale et. al., 2002; Burns et. al., 1998). During storm events, nitrate may be quickly mobilized to the stream and "flushed" from the catchment (Creed et. al., 1996; Creed and Band 1998; McHale et. al., 2002). The magnitude of nitrate concentrations undoubtedly vary throughout the year due to the "wetting-up" and "drying-down" of the catchment, but how do these storm patterns change with season? While the strong links between hydrology and nitrate are well established, most studies to date have been conducted predominantly in undisturbed environments. Watersheds with multiple land uses need to be studied to determine whether or not these same patterns (and processes) occur under different conditions and scales.

In addition to gaining knowledge of catchment processes controlling stream nitrate concentrations, this knowledge needs to be incorporated into models to improve nitrate predictions. Recent discussions in the global hydrologic community have called for an improvement to predictions in ungauged basins (Clarke 2005; Littlewood et. al., 2003; Sivapalan, 2003; Sivapalan et. al., 2003). Many basins throughout the world are ungauged or inadequately gauged, creating a need for extrapolation of knowledge from gauged basins to ungauged basins for watershed management decisions. Due to the heterogeneity of climate and landscape and our current lack of understanding of basin responses, extrapolating calibrated models from a gauged to ungauged basin has proven to be woefully unsuccessful (Sivapalan, 2003). Adequate water quantity and quality predictions are needed to make informed, sustainable management decisions to prevent further ecosystem degradation and promote human life and health (Sivapalan et. al., 2003).

Improving predictions may be accomplished using the knowledge we currently have about the processes controlling nitrate concentrations. Several studies have examined the processes involved in nitrate transport, transformations, and storage (Hjerdt et. al., 2004; Petry et. al., 2002; Creed and Band; 1998; Jordan et. al., 1997; McHale et. al., 2002; Hornberger et. al., 1994). One somewhat common finding from this work is that hot spots (patches of the catchment with relatively high reaction rates, often enhanced at the terrestrial-aquatic interfaces) exert a profound control on streamwater nitrate dynamics (McClain et. al, 2003). The interface between oxic and anoxic zones (i.e., the interface between upland and riparian zones), is typically a hot spot for denitrification (Dahm et. al., 1998; McClain et. al., 2003; Peterjohn and Correll, 1984; Lowrance et. al., 1984). Topography and topographic position are simple measures that may aid the identification of hotspots in catchments. The well known topographic index (TI) of Beven and Kirkby (1979) has been found in studies conducted in headwater forested catchments to show a positive correlation with nitrate concentrations (Creed and Band, 1998; Welsch et. al., 2001). In addition, land use near the stream has been found to be a better predictor of water quality than land use over the entire watershed (Peterjohn and Correll, 1984; Lowrance et. al., 1984). Adding topographic and distance weighting effects to empirical catchment analyses is one way to begin to add a dimension of process representation and hotspot (patches of the catchment with relatively high denitrification rates) influence on streamwater nitrate predictions.

Another way to improve predictions is using a classification scheme. As a result of many hydrologic studies, characterization of a catchment's hydrologic setting is possible. This in turn may improve predictions of nitrate, which has been found to be strongly linked to hydrology (Creed et. al., 1996; Creed and Band, 1998; McHale et. al., 2002). Classification of catchments or regions based on various sets of criteria has been discussed for some time (Chapman, 1987; Winter, 2001; Omernik and

Griffith, 1991; Preston, 2000; Baker et. al., 2001). The hydrologic landscape regions (HLRs) classification scheme developed by Wolock et. al. (2004) is the first objective classification for the entire United States that can be tested with independent data. This classification scheme needs to be further tested with a range of water quality parameters in different regions to determine its usefulness for improving predictions.

Objectives

The general goal of this study was to determine how land use affects stream nitrate concentrations. This work furthers the understanding of nitrate dynamics in catchments with varying land use, and incorporates process-based knowledge into statistical models to improve predictions of nitrate concentrations. Two different approaches to understanding the processes occurring in the catchments were used: a temporally intensive case study of three catchments with varying land use (through storm event monitoring) and a spatially intensive study on the regional scale (through statistical modeling). Both approaches provided insights into processes. The first portion of this work identified patterns of nitrate concentrations during storm events and how they vary seasonally, and the second portion was focused on improving predictions of nitrate concentrations based on land use and different hydrologic indices (TI and the hydrologic landscape region classification scheme). If the hydrologic index or classification does improve predictions, then certain processes may be accounted for. Both portions of this work further the goal of understanding processes controlling nitrate export from a catchment, and how land use may be altering these processes. This work also furthers the global hydrologic community's goal of improving predictions in ungauged basins.

The specific objectives of this dissertation are to:

1. Determine how land use affects export of nitrate and nitrate "patterns" during storm events on the catchment scale, and whether or not behaviors are different from the "flushing" seen in pristine catchments.
2. Develop a statistical water quality model using land use to determine how land use affects stream nitrate concentrations on the regional scale.
3. Include process-based knowledge of the controls hydrology exerts over nitrate with topographic index (TI) and a classification scheme (HLRs).

The dissertation is organized into five chapters, with chapters 2 through 4 describing each portion of the research. Chapter 1 is a general introduction. Chapter 2 describes the catchment scale portion of the research, which explores how human activity alters the export of nitrate, how the input of nitrate changes throughout the year, affecting storm response (i.e., depletion of soil water nitrate, addition of fertilizer, etc.), and how the changing contribution of source waters throughout the year affect streamflow concentrations. Chapters 3 and 4 describe the development of the statistical water quality model using topographic index (TI) and the hydrologic landscape region classification scheme. Chapter 3 explores whether identifying areas with a high denitrification potential using TI improves the prediction of stream nitrate concentrations in Western Oregon and Northern California, and whether models using in-stream and out-of-stream inverse distance (1/d) and inverse-distance squared ($1/d^2$)

measures improve nitrate predictions. Chapter 4 explores the correlation of nitrate with the HLR classification scheme of Wolock et. al. (2004) in Western Oregon and Northern California, and whether the primary hydrologic flowpaths identified from HLRs help predict nitrate, which is controlled by transformation processes as well as the hydrology of the catchment. Chapter 5 is a general conclusion, with a synthesis of the results from the catchment scale and regional scale approaches.

References

Arheimer, B. and Liden, R., 2000. Nitrogen and phosphorous concentrations from agricultural catchments – influence of spatial and temporal variables. Journal of Hydrology 227, 140-159.

Baker, M.E., Wiley, M.J., and Seelbach, P.W., 2001. GIS-based hydrologic modeling of riparian areas: implications for stream water quality. Journal of the American Water Resources Association 37, 1615-1627.

Beven, K.J. and Kirkby, 1979. A physically based, variable contributing area model of basin hydrology. Hydrological Sciences 24, 43-69.

Bolstad, P.V., and Swank, W.T., 1997. Cumulative impacts of landuse on water quality in a Southern Appalachian watershed. Journal of the American Water Resources Association 33, 519-533.

Buck, O., Niyogi, D.K., and Townsend, C.R., 2004. Scale-dependence of land use effects on water quality of streams in agricultural catchments. Environmental Pollution 130, 287-299.

Burns, D.A., Murdoch, P.S., Lawrence, G.B., and Michel, R.L., 1998. Effect of groundwater springs on NO_3^- concentrations during summer in Catskill Mountain streams. Water Resources Research 34, 1987-1996.

Chapman, T., 1987. Classification of regions *in* Comparative Hydrology: An ecological approach to land and water resources, M. Falkenmark and T. Chapman eds. Unesco, Stockholm, pp. 50-74.

Clarke, R.T., 2005. The PUB decade: how should it evolve? Hydrological Processes 19, 2865-2869.

Creed, I.F. and Band, L.E., 1998. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research 11, 3105-3120.

Creed, I.F., Band, L.E., Foster, N.W., Morrison, I.K., Nicolson, J.A., Semkin, R.S., and Jeffries, D.S., 1996. Regulation of nitrate-N release from temperate forests: A test of the N flushing hypothesis. Water Resources Research 32, 3337-3354.

Dahm, C.N., Grimm, N.B., Marmonier, P., Valett, H.M., Vervier, P., 1998. Nutrient dynamics at the interface between surface waters and groundwaters. Freshwater Biology 40, 427-451.

Donner, S.D., Kucharik, C.J., and Foley, J.A., 2004. Impact of changing land use practices on nitrate export by the Mississippi River. Global Biogeochemical Cycles 18, GB1028, 1-21.

Herlihy, A.T., Stoddard, J.L., and Johnson, C.B., 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic Region, U.S. Water, Air, and Soil Pollution 105, 377-386.

Hjerdt, K.N., McDonnell, J.J., Seibert, J., and Rodhe, A., 2004. A new topographic index to quantify downslope controls on local drainage. Water Resources Research 40, W05602, 1-6.

Hornberger, G.M., Bencala, K.E., and McKnight, D.E., 1994. Hydrological controls on dissolved organic carbon during snowmelt in the Snake River near Montezuma, Colorado. Biogeochemistry 25, 147-165.

Howarth, R.W., Boyer, E.W., Pabich, W.J., and Galloway, J.N., 2002. Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31, 88-96.

Inamdar, S.P., Christopher, S.F., and Mitchell, M.J., 2004. Export mechanisms for dissolved organic carbon and nitrate during summer storm events in a glaciated forested catchment in New York, USA. Hydrological Processes 18, 2651-2661.

Johnson, L.B., Richards, C., Host, G.E., and Arthur, J.W., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37, 193-208.

Jones, K.B., Neale, A.C., Nash, M.S., Van Remortel, R.D., Wickham, J.D., Riitters, K.H., and O'Neill, R.V., 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16, 301-312.

Jordan, T.E., and Weller, D.E., 1996. Human contributions to terrestrial nitrogen flux. Bioscience 46, 655-663.

Jordan, T.E., Correll, D.L., Weller, D.E., 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33 2579-2590.

Lattin, P.D., Wigington, P.J., Moser, T.J., Peniston, B.E., Lindeman, D.R., and Oetter, D.R., 2004. Influence of remote sensing imagery source on quantification of riparian land cover/land use. Journal of the American Water Resources Association 40, 215-227.

Little, J.L., Saffran, K.A., and Fent, L., 2003. Land use and water quality relationships in the lower Little Bow River watershed, Alberta, Canada. Water Quality Research Journal of Canada 38, 563-584.

Littlewood, I.G., Croke, B.F., Jakeman, A.J., and Sivapalan, M., 2003. The role of 'top-down' regressioning for Prediction in Ungauged Basins (PUB). Hydrological Processes 17, 1673-1679.

Lowrance, R., Todd, R., Fail J. Jr., Hendrickson, O. Jr., Leonard, R., 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34, 374-377.

McClain, M.E., Boyer, M.E., Dent, C.L., Gergel, S.E., Grimm, N.B., Groffman, P.M., Hart, S.C., Harvey, J.W., Johnston, C.A., Mayorga, E., McDowell, W.H., and Pinay, G., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6, 301-312.

McHale, M.R., McDonnell, J.J., Mitchell, M.J., and Cirmo, C.P., 2002. A field-based study of soil water and groundwater nitrate release in an Andirondack forested watershed. Water Resources Research 38, 1031, 1-17.

National Research Council, 2000. Clean Coastal Waters: Understanding and reducing the effects of nutrient pollution. National Academy Press, Washington D.C..

National Science and Technology Council, 2000. Integrated assessment of hypoxia in the northern Gulf of Mexico. Committee on Environment and Natural Resources, Washington D.C.

Peterjohn, W.T. and Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65, 1466-1475.

Preston, S.D., 2000. Statistical identification of hydrochemical response units for hydrologic monitoring and modeling in Maryland. USGS Water-Resources Invesitigations Report 00-4232.

Omernik, J.M., and Griffith, G.E., 1991. Ecological regions versus hydrologic units: frameworks for managing water quality. Journal of Soil and Water Conservation 46, 334-340.

Owens, L.B., Edwards, W.M., and Keuren, R.W., 1991. Baseflow and stormflow transport of nutrients from mixed agricultural watersheds. Journal of Environmental Quality 20, 407-414.

Petry, J., Soulsby, C., Malcolm, I.A., and Youngson, A.F., 2002. Hydrological controls on nutrient concentrations and fluxes in agricultural catchments. The Science of the Total Environment 294, 95-110.

Pimentel, D. (ed.), 1993. World soil erosion and conservation. Cambridge University Press, Cambridge, Massachusetts.

Rabalais, N.N., Turner, R.E., and Scavia, D., 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. Bioscience 52, 129-141.

Salvia-Castellvi, M., Iffly, J.F., Borght, P.V., and Hoffman, L., 2005. Dissolved and particulate nutrient export from rural catchments: a case study from Luxembourg. Science of the Total Environment 344, 51-65.

Schilling, K.E., 2002. Chemical transport from paired agricultural and restored prairie watersheds. Journal of Environmental Quality 31, 1184-1193.

Sivapalan, M., 2003. Prediction in ungauged basins: a grand challenge for theoretical hydrology. Hydrological Processes 17, 3163-3170.

Sivapalan, M., Takeuchi, K., Franks, S.W., Gupta, V.K., Karambiri, H., Lakshmi, V., Liang, X., McDonnell, J.J., Mendiondo, E.M., O'Connell, P.E., Oki, T., Pomeroy, J.W., Schertzer, D., Uhlenbrook, S., and Zehe, E., 2003. IAHS Decade on Predictions in Ungauged Basins (PUB), 2003-2012: Shaping an exciting future for the hydrological sciences. Hydrological Sciences Journal 48, 857-880.

Vanni, M.J., Renwick, W.H., Headworth, J.L., Auch, J.D., and Schaus, M.H., 2001. Dissolved and particulate nutrient flux from three adjacent agricultural watersheds: a five-year study. Biogeochemistry 54, 85-114.

Wayland, K.G., Long, D.T., Hyndman, D.W., Pijanowski, B.C., Woodhams, S.M., and Haack, S.K., 2003. Identifying relationships between baseflow geochemistry and land use with synoptic sampling and r-mode factor analysis. Journal of Environmental Quality 32, 180-190.

Webb, B.W., and Walling, D.E., 1985. Nitrate behaviour in streamflow from a grassland catchment in Devon, U.K.. Water Resources 19, 1005-1016.

Wellington, B.I. and Driscoll, C.T., 2004. The episodic acidification of a stream with elevated concentrations of dissolved organic carbon. Hydrological Processes 18, 2663-2680.

Welsch, D.L., Kroll, C.N., McDonnell, J.J., and Burns, D.A., 2001. Topographic control on the chemistry of subsurface stormflow. Hydrological Processes 15, 1925-1938.

Wernick, B.G., Cook, K.E., and Schreier, H., 1998. Land use and streamwater nitrate dynamics in an urban-rural fringe watershed. Journal of the American Water Resources Association 34, 639-650.

Wigington, P.J., Baker, J.P., DeWalle, D.R., Kretser, W.A., Murdoch, P.S., Simonin, H.A., Van Sickle, J., McDowell, M.K., Peck, D.V., Barchet, W.R., 1996a. Episodic acidification of small streams in the Northeastern United States: episodic response project. Ecological Applications 6, 374-388.

Wigington, P.J., DeWalle, D.R., Murdoch, P.S., Kretser, W.A., Simonin, H.A., Van Sickle, J., and Baker, J.P., 1996b. Episodic acidification of small streams in the Northeastern United States: ionic controls of episodes. Ecological Applications 6, 389-407.

Winter, T.S., 2001. The concept of hydrologic landscapes. Journal of the American Water Resources Association 37, 335-349.

Woli, K.P., Nagumo, T., Kuramochi, K., and Hatano, R., 2004. Evaluating river water quality through land use analysis and N budget approaches in livestock farming areas. Science of the Total Environment 329, 61-74.

Wolock, D.M., Winter, T.C., and McMahon, G., 2004. Delineation and evaluation of hydrologic-landscape regions in the United States using geographic information system tools and multivariate statistical analyses. Environmental Management 34, Suppl. 1, S71-S88.

Chapter 2

The Effects of Land Use on Stream Nitrate Dynamics

Cara J. Poor

Jeffrey J. McDonnell

Submitted to the Journal of Hydrology

in review

Abstract

The effects of land use and land use change on stream nitrate are poorly understood. While case studies have been presented, most process work has been done in areas with one land use (minimally disturbed or agricultural) and areas with substantial atmospheric deposition. In this paper we present results from 3 neighboring headwater catchments in western Oregon with similar (low) atmospheric deposition, size, and geology but with different, spatially consistent land use expressions: forest, agriculture and residential. The climate in western Oregon has a distinct pattern of a three month rainless period in the summer, a wetting up with many storms in the fall and winter, and a decrease of storms in the spring. We investigate how human activity alters the export of nitrate, whether the input of nitrate changes throughout the year, affecting storm response (i.e., depletion of soil water nitrate, addition of fertilizer, etc.), and how the changing contribution of source waters throughout the year affects streamflow concentrations. Our results showed marked differences in export rates between the three catchments. The forested catchment showed minimal export for three monitored storms (fall, winter, spring) through the seasonal wetting up of the catchments, and the residential catchment showed high export for all three storms. While the agricultural catchment displayed elevated export in the fall (similar to the residential catchment), exports decreased progressively throughout the rainy period (following late summer manure and green bean application). Overall, our results of storm event nitrate concentrations suggest that varying nitrate inputs have a large effect on nitrate dynamics. While within-storm nitrate concentration response

patterns in the residential catchment were the same as the patterns in the reference forested catchment (a "concentration" pattern throughout the year), a "dilution" pattern was observed in the fall and winter and a "concentration" pattern was observed in the spring in the agricultural catchment.

Introduction

Increases in nitrogen inputs in the last 50 years have caused great concern for the health of stream ecosystems (Pimental, 1993; Howarth et. al., 2002). Nitrogen inputs from human activity have doubled in the United States from 1961 to 1997 (Howarth et. al., 2002). In general, approximately one-third of nitrogen inputs to catchments are exported, with the majority exported to surface waters (Howarth et. al., 2002). This increase in export to surface waters has been shown to cause algal blooms, which in turn cause hypoxia and "dead" zones for fish (National Science and Technology Council, 2000; Rabalais et. al., 2002; National Research Council, 2000). Episodic acidification of streams has also resulted from increased nitrate levels (Wigington et. al., 1996a; Wigington et. al., 1996b; Wellington and Driscoll, 2004).

Not surprisingly, land use has been found to have a large effect on the amount of nitrogen exported to the stream (Salvia-Castellvi et. al., 2005; Schilling, 2002; Jordan et. al., 1997; Owens et. al., 1991; Howarth et. al., 2002; Jordan and Weller, 1996; Johnson et. al., 1997; Herlihy et. al., 1998; Wernick et. al., 1998; Arheimer and Liden, 2000; Jones et. al., 2001; Wayland et. al., 2003; Donner et. al., 2004; Woli et. al., 2004; Buck et. al., 2004; Lattin et. al., 2004; Little et. al, 2003). Since a significant portion of nitrogen export from catchments is due to non-point source fertilizer runoff, the proportion of agricultural land in a catchment is often correlated to stream nitrate export (Howarth et. al., 2002). Nitrogen export is generally greater in rivers draining more densely populated catchments (Jordan and Weller, 1996). This may be due to sewage inputs or deposition and subsequent runoff of NO_x emissions. The majority of the work on land use effects has focused on baseflow or a small number of sampling events correlating land use and nitrate (Johnson et. al., 1997; Herlihy et. al., 1998; Wernick et. al., 1998; Arheimer and Liden, 2000; Jones et. al., 2001; Wayland et. al., 2003; Donner et. al., 2004; Woli et. al., 2004; Buck et. al., 2004; Lattin et, al., 2004; Little et. al., 2003; Schilling, 2002). While it is clear that land use affects the magnitude of nitrate and other nutrients exported from catchments, it is not clear how it affects nutrient dynamics or the nutrient concentration pattern during storm events.

A few studies have been conducted in catchments with mixed land use during storm events; however, much of the work has been concerned with monthly exports, and little is shown of nitrate concentrations varying with discharge dynamics (Jordan et. al., 1997; Owens et. al., 1991; Bolstad and Swank, 1997; Salvia-Castellvi et. al., 2005). Results are shown as a baseflow index or monthly averages (Jordan et. al., 1997; Owens et. al., 1991; Salvia-Castellvi et. al., 2005). Alternatively, one event or the "typical" response for a catchment is shown (Salvia-Castellvi et. al., 2005; Bolstad and Swank, 1997). These studies, in addition to studies conducted in forested or agricultural catchments, either show a "concentration" pattern, where nitrate concentrations increase with increasing flow rates and essentially mimic the storm hydrograph, or a "dilution" pattern, where nitrate concentrations decrease with increasing flow rates as a mirror image of the hydrograph (Salvia-Castellvi et. al., 2005; Bolstad and Swank, 1997; Webb and Walling, 1985; Petry et. al., 2002; Vanni et. al., 2001; Inamdar et. al., 2004; McHale et. al., 2002; Burns et. al., 1998). During storm events, nitrate may be quickly mobilized to the stream (Creed et. al., 1996; Creed and Band 1998; McHale et. al., 2002). The magnitude of nitrate concentrations undoubtedly vary throughout the year due to the "wetting-up" and "drying-down" of the catchment, but how do these storm patterns change with season? While the strong links between hydrology and nitrate are well established, most studies to date have been conducted predominantly in either minimally disturbed environments or agricultural areas.

We argue that further investigation of the seasonality of nitrate dynamics during storm events should occur in catchments with varying land uses. In order to understand the behavior of solutes during storm events, studies need to be conducted in areas with major disturbances (Burns, 2005). Here, we present a study that examines the seasonality of nitrate dynamics in three catchments with similar physical characteristics (area, geographic proximity, geology, soils, topography, elevation) but different land uses. Storm events were monitored in this Mediterranean climate from the end of a 3-month rainless period through a clear progression of wet-up and potential flushing events. We explore how human activity alters the export of nitrate, whether or not the input of nitrate changes throughout the year, affecting storm response (i.e., depletion of soil water nitrate, addition of fertilizer, etc.), and how the changing contribution of source waters throughout the year affect streamflow concentrations.

Site Description

The three study catchments are each on the order of 50 ha and are sub-basins of the 33 km^2 Oak Creek Watershed, located near Corvallis, Oregon, U.S.A (Figure 2.1). This area is located in the Pacific Northwest of the United States in a region virtually devoid of atmospheric nitrogen deposition (annual rate of approximately 1.52 kg/ha/yr, http://nadp.sws.uiuc.edu/nadpdata/annualReq.asp?site=OR97). The climate in the Pacific Northwest is relatively mild and often described as a mediterranean climate, with dry summers and wet winters. Average temperature in the Oak Creek Watershed is 11.5 °C, and mean annual precipitation is approximately 111 cm/year (Oregon Climate Service, www.ocs.oregonstate.edu). The majority of the precipitation falls during the rainy season (November through June). Minimal snowfall occurs in the catchment, with snowmelt occurring 1-2 days after the event. The Oak Creek Watershed has clear and well-defined land uses expressed within its subcatchments. The upper portion of the watershed is a minimally disturbed, second growth Douglas Fir forest. The mid-portion of the watershed is primarily agricultural (sheep and cattle grazing, growth of clover, wheat, and fescue) with small inholdings of residential areas. Land use in the lower portion of the watershed consists of urban residential and the Oregon State University campus. Each study catchment has a clean expression of land use (forested, agricultural, residential) and shares approximately the same headwater divide.

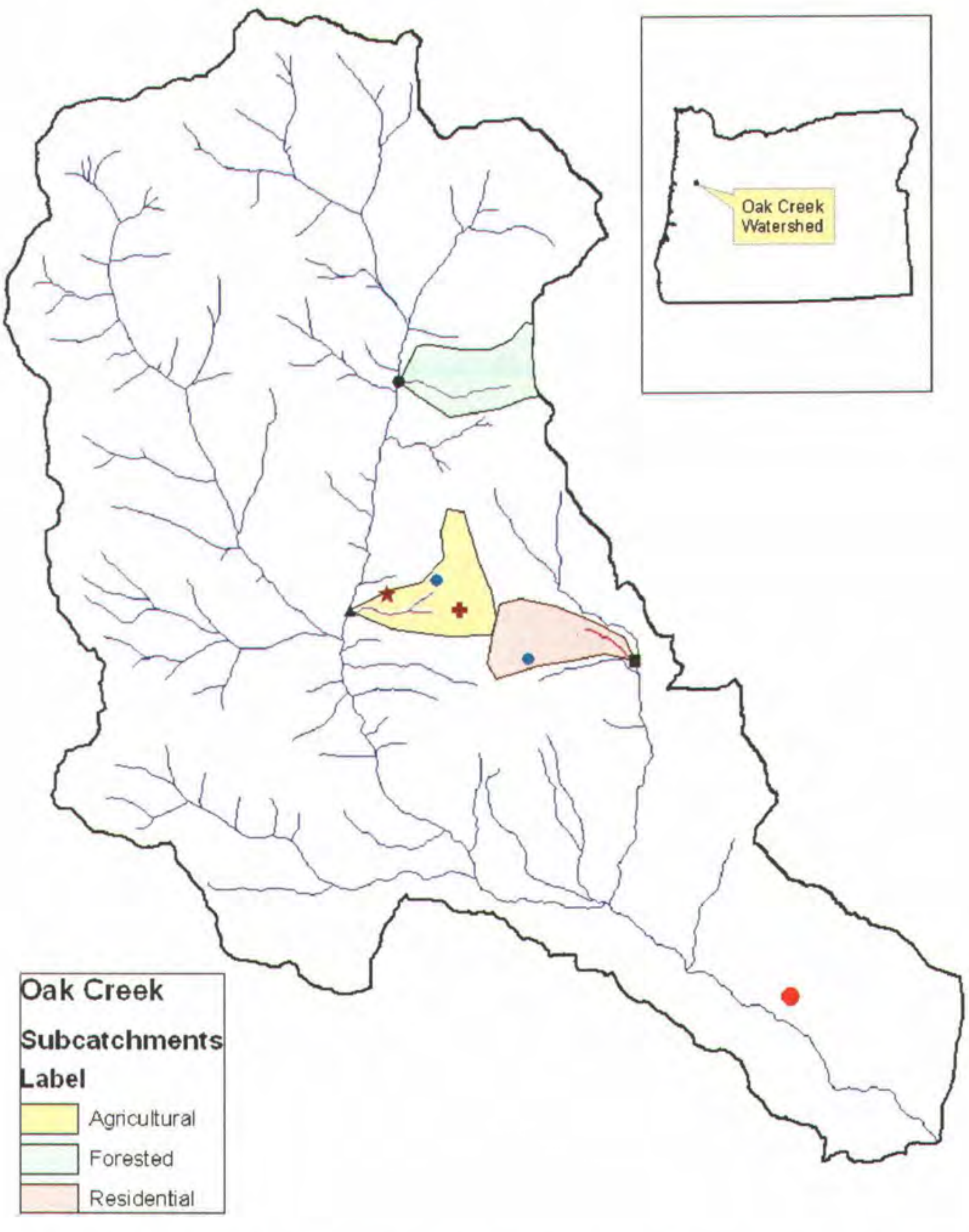

Figure 2.1. Oak Creek Watershed and study catchments. Rain gage location = ●, well locations in agricultural and residential catchments = ●, groundwater seep = ★, soil pipe = ✚, sampling point in the forested catchment = ●, sampling point in the agricultural catchment = ▲, and sampling point in the residential catchment = ■.

Forested Catchment. The 49.5-ha forested catchment is minimally disturbed (Figure 2.1) and is drained by a first-order stream. Land use is entirely forested with approximately 1750 m of abandoned gravel roads. Elevations range from 152 to 450 m. Additional physical features of the forested catchment are listed in Table 2.1. Vegetation consists mainly of Douglas Fir, alder, ash, sword ferns, blackberry, and various weed species. The soil in the catchment is classified as the Dixonville-Philomath association, which is moderately deep (approximately 1 m of weathered basalt bedrock), well-drained silty clay loams and shallow, well-drained silty clays (Soil Conservation Service, 1975). The ~30 cm-thick surface layer consists of silty clay loam and silty clay, and the ~60 cm-thick subsurface layer consists of silty clay and clay. Underlying geology is mafic volcanic.

		Site	
	Forested[1]	Agricultural[1]	Residential[2]
Watershed Area (ha)	49.5	52.2	42.9
% Tree Cover	98.1	52.8	83.1
Mean Slope (%)	22.7	12.4	15.1
Mean TI	6.42	6.84	6.56
Elevation Difference (ft)	298	158	84
Drainage Density (km/km^2)	1.95	1.78	1.20
Road Density (km/km^2)	3.55	1.20	5.64

[1]all roads are unpaved

[2]all roads are paved

Table 2.1. Physical features of study catchments.

Agricultural Catchment. The 52.2-ha agricultural catchment is located within the Wilson Sheep Farm, where 325 sheep are rotated through the catchment and neighboring 100 ha of pasture land. The sheep are confined in a building for several weeks when the ewes are lambing, and graze in the catchment the rest of the year, rotating weekly to bi-weekly amongst the fields. The manure generated during the lambing period is kept under roof throughout the winter and applied to the fields in the summer when conditions are dry. Green bean waste is also applied to fields in the summer. The sampling site for the agricultural catchment is shown in Figure 2.1. This catchment is drained by a second-order ephemeral tributary to Oak Creek that flows through grass fields. Land use is entirely agricultural with approximately 625 m of gravel road leading to the main sheep barn and one outbuilding. The catchment varies in elevation from 116 to 274 m. Additional physical features of the agricultural catchment are listed in Table 2.1. Approximately 62 kg N/ha of manure and green beans are spread in the summer onto fields. Grazing animals input approximately 0.25 kg N/ha/day as manure to the catchment throughout the year (except in February and March during the lambing period), based on data supplied to us by the Oregon State University sheep farm manager (Tom Nichols, personal communication, 2004) and published numbers for average manure production per sheep and quantity of nitrogen per kg of manure (American Society of Agricultural Engineers, 2003). This large input of nitrogen will likely affect streamflow quality during the grazing period and when stored manure is applied. Unlike the perennial flow in the forested catchment, stream flows in the agricultural catchment are continuous during the rainy season but discontinuous in the summer months. The main vegetation consists of blackberry adjacent to the stream and grass fields interspersed with oak and ash throughout the catchment. Soil type is classified as the Waldo-Bashaw association, which include poorly drained silty clay loams and clays (Soil Conservation Service, 1975). Approximate depth to bedrock is 2 m, and the underlying geology is mafic volcanic.

Residential Catchment. The 42.9-ha residential catchment shares a portion of the agricultural catchment's drainage divide, and is heavily wooded. Land use is entirely residential, including a park with woodlands and marshes in the lower portion of the catchment. Some sections of the catchment are hardened (i.e., runoff or rainfall cannot infiltrate into the soil in these areas) by paved streets, concrete lining of the stream, and storm drains that empty directly into the stream channel. Impervious areas cover approximately 15% of the catchment. Housing density (2.7 houses/ha) is relatively low compared to most residential neighborhoods, with houses on 0.1-ha lots and approximately 3,950 m of sanitary sewer lines in the upper portion of the catchment. Although much of the catchment is wooded and some natural features have remained, land use in the catchment is significantly different from the forested catchment due to the paved streets, concrete lining of the stream, storm drains, and houses. Elevation varies from 116 to 200 m. Additional physical features of the residential catchment are listed in Table 2.1. The main vegetation consists of Douglas Fir, alder, ash, sword ferns, and blackberry mixed with lawns and ornamental shrubs. In the lower portion of the catchment, the stream flows through a park, which contains a marshy area, baseball fields, and lawns. Soil type is classified as the Waldo-Bashaw association, which are poorly drained silty clay loams and clays (Soil Conservation Service, 1975). Approximate depth to bedrock is 2 m, and the underlying geology is mafic volcanic. The only known anthropogenic input of nitrogen is sporadic fertilization of lawns and bushes by homeowners in the upper portion of the catchment.

Methods

Stream chemistry was sampled at the outlets of the three catchments during one of the first fall storms (12/9/2003), following a 3-month summer drought. We refer to this as the "wetting up" period (i.e., the beginning of the 2003-2004 water year). A winter storm on 2/23/2004 was sampled at each catchment outlet when water tables at each of the sites were close to the surface. A spring storm on 4/13/2004, when each catchment was beginning to dry out, was also sampled. Three ISCO Model 1672 autosamplers were used at sampling locations for hourly sampling on the rising limb of the hydrograph and a bi-hourly sampling on the falling limb. Biweekly grab samples were taken at each site during the 2003-2004 field season (and when the agricultural stream was flowing--November 2003 through June 2004).

Biweekly soil water samples were also taken at each site during the 2003-2004 field season from porous-cup tension lysimeters. Lysimeters for each site were located <2 m laterally and <10 m longitudinally from the stream sampling point, and were approximately 53, 76, and 48 cm deep in the forested, agricultural, and residential catchments, respectively. Groundwater samples were taken on 2/19/2004 from an existing shallow well ~24 m deep in the agricultural catchment (Tom Nichols, personal communication, 2004), and on 7/20/2003 and 7/23/2004 from a deeper residential well in the residential catchment (see Figure 2.1 for locations). The exact depth of the residential well is unknown; however, several other wells in the area are ~60 m deep. Additional samples were taken from a groundwater seep and soil pipe in the agricultural catchment on 2/19/2004.

Flow was gauged at stream sections with good natural flow control. We used TRUTRACK Inc. capacitance rods to measure stage height at 10-minute intervals throughout the year. We used the salt-dilution technique of Gordon et. al. (1992) to establish rating curves for each gauging position. From these relationships, flow rates were determined for the sampling period. There is some uncertainty in flow rates due to uncertainties in the rating curves from the natural flow control sections and due also to the estimates of watershed area based on a 30-m DEM. Uncertainty was quantified from the difference in flow measurement data and the approximated rating curve, and was $\pm$ 14.87%, $\pm$ 6.87%, and $\pm$ 4.48% for the forested, agricultural, and residential hydrographs, respectively. Uncertainties are shown as a percentage due to the nature of stream flow gauging; fewer measurements were made at the higher flows and therefore there is more uncertainty. This uncertainty is shown in the hydrographs as a band of upper and lower flow rates. Precipitation data was obtained from a Met One model 385 tipping bucket rain gauge located within the Oak Creek Watershed at the point shown in Figure 2.1. We computed seven-day and 30-day API using the method of Mosley (1979). Biweekly precipitation chemistry data was obtained from the NADP Hyslop Farm site (Latitude 44.6347, Longitude -123.19), which is approximately 10 km northeast of the Oak Creek Watershed (http://nadp.sws.uiuc.edu/nadpdata/annualReq.asp?site=OR97).

Samples were preserved and collected according to Standard Methods for the Examination of Water and Wastewater (Clesceri et. al., 1998). Sample bottles were rinsed with hydrochloric acid and deionized water before use. When the autosamplers were used, the bottles filled automatically through a rinsed tube. Samples were taken from the middle of the channel at mid-depth in fast-moving water to ensure adequate mixing. When grab samples were taken, the bottle was rinsed three times with stream water, then filled beneath the water surface. Conductivity, pH, and temperature were measured during sampling. All collected samples were analyzed for nitrate, DOC, and major anions and cations. Samples were filtered with Whatman 0.7 μm glass fiber syringe filters within 24 hours of collection. Dissolved organic carbon was measured using a Dohrmann DC-190 Total Organic Carbon Analyzer. Nitrate, sulfate, chloride, and fluoride were measured using a Dionex Model DX 500 Ion Chromatograph. A Varian Liberty 150 ICP Atomic Emission Spectrophotometer was used to determine potassium, calcium, sodium, magnesium, and silica concentrations. All samples were measured in duplicate to determine the reliability of methods, and uncertainty was quantified from the standard deviation. Due to the accuracy of the instruments, uncertainty was very small and thus not decipherable in the resulting plots.

Results

Hydrologic Response to Storm Events. Characteristics of storm 1 (12/9/2003 to 12/12/2003), storm 2 (2/23/2004 to 2/25/2004), and storm 3 (4/13/2004 to 4/16/2004) are shown in Table 2.2. Rainfall duration and total rainfall ranged from

15-24 hours and 17-26 mm, respectively. Ranges for the 7-day and 30-day API were 0-18 and 0.5-31 mm, respectively. The 7-day API for storm 3 was zero, indicating that no precipitation occurred 7 days before the rain event. Storm 1 had the lowest intensity (22.9 mm in 24 hours) and highest API (7-day API of 18.3), whereas storm 3 had the highest intensity (25.5 mm in 17 hours) and lowest API (7-day API of 0.0). Although the API is higher during storm 1 (fall) than storm 2 (winter), the total precipitation is higher in the winter (Table 2.3). The wetter conditions in the winter are also reflected in the higher baseflows. The discrepancy between API and seasonal precipitation/baseflow is due to the fall wet-up; after the 3-month dry period fall storms wet-up the catchment. A significant response to these storms is delayed until the winter, when additional rainfall creates saturated conditions.

Storm	Total Rainfall (mm)	Rain Duration (hr)	7-day API (mm)	30-day API (mm)
1	22.9	24	18.3	30.6
2	16.7	15	3.7	13.6
3	25.5	17	0.0	0.6

Table 2.2. Characteristics of storm 1 (12/9/2003 to 12/12/2003), 2 (2/23/2004 to 2/25/2004) and 3 (4/13/2004 to 4/16/2004).

		Season		
		Fall	Winter	Spring
Total Precipitation (mm)		415	521	153
	Forested	0.019	0.061	0.039
Average Baseflow (mm/hr)	Agricultural	0.002	0.064	0.018
	Residential	0.075	0.130	0.086

Table 2.3. Seasonal precipitation and baseflow for the three catchments.

Catchment response to each storm was variable (Table 2.4). Runoff ratios increased with increasing development, with the highest ratios in the residential catchment (0.23-0.33) and the lowest ratios in the forested catchment (0.05-0.10). One exception was the runoff ratio in the agricultural catchment during storm 1 (0.09), which was about the same as the runoff ratio in the forested catchment (0.10). Runoff ratio in the agricultural catchment ranged from 0.09-0.17. Runoff ratios were still highest in the residential catchment during storm 1 (0.33). Baseflow was highest in the residential catchment during all storms (0.071-0.141 mm/hr), and increased through the rainy period as the catchment became more hydrologically connected. Baseflow in the forested catchment stayed relatively constant, ranging from 0.040 to 0.067 mm/hr. With the exception of storm 2, the agricultural catchment had the lowest baseflow (0.007-0.084 mm/hr). Baseflow increased to 0.084 mm/hr in the winter then decreased, which reflects the ephemeral nature of the stream. Peak discharge also generally increased with increasing development, except for storm 2, where peak flows in the agricultural catchment (0.50 mm/hr) were higher than peak flows in the residential catchment (0.42 mm/hr). Peak discharge ranged from 0.08 to 0.17 mm/hr in the forested catchment, 0.21 to 0.50 mm/hr in the agricultural catchment, and 0.26-0.42 mm/hr in the residential catchment.

		Site		
		Forested	Agricultural	Residential
Nitrate Export Rate (kg/ha/storm)	Storm 1	0.012	0.121	0.131
	Storm 2	0.005	0.040	0.108
	Storm 3	0.010	0.021	0.131
Runoff Ratio	Storm 1	0.105	0.094	0.326
	Storm 2	0.092	0.168	0.319
	Storm 3	0.056	0.145	0.229
Peak Discharge (mm/hr)	Storm 1	0.173	0.208	0.256
	Storm 2	0.142	0.497	0.422
	Storm 3	0.077	0.216	0.289
Baseflow (mm/hr)	Storm 1	0.062	0.007	0.071
	Storm 2	0.067	0.084	0.110
	Storm 3	0.040	0.017	0.141
Hydrograph Response Time (hrs)	Storm 1	3.7	1.0	1.3
	Storm 2	7.2	0.0	2.3
	Storm 3	2.7	2.0	0.0
Time to Peak (hrs)	Storm 1	15.0	11.3	19.3
	Storm 2	7.0	13.7	15.3
	Storm 3	25.0	22.3	35.2

*Spring storm produced two peaks. Time to peak and peak flow is shown for the second peak in the hydrograph.

Table 2.4. Catchment response to storms.

Time to peak (defined as the time from the start of the rising limb of the hydrograph to the peak) ranged from 7-25 hours in the forested catchment, 11-22 hours in the agricultural catchment, and 15-35 hours in the residential catchment. Time to peak was longer for the forested catchment than the agricultural catchment during storms 1 and 3, but increased with increasing development for storm 2. Time to peak was the longest in the residential catchment during all storms. Hydrograph response time was the longest for the forested catchment during all storms (2.7-7.2 hours). Hydrograph response time (defined as the time from the onset of rainfall to the start of the rising limb of the hydrograph) varied from 0.0-2.0 and 0.0-2.3 hours in the agricultural and residential catchments, respectively. Although it appears that time to peak increased with increasing development during storm 2, the hydrograph from the agricultural catchment peaks first when taking hydrograph response time into account (Figures 2.2-2.4). During all storms, the agricultural hydrograph peaks first, followed by the forested hydrograph, then the residential hydrograph. Only the agricultural hydrograph is shown for clarity, with arrows showing the peaks of the forested and residential hydrograph.

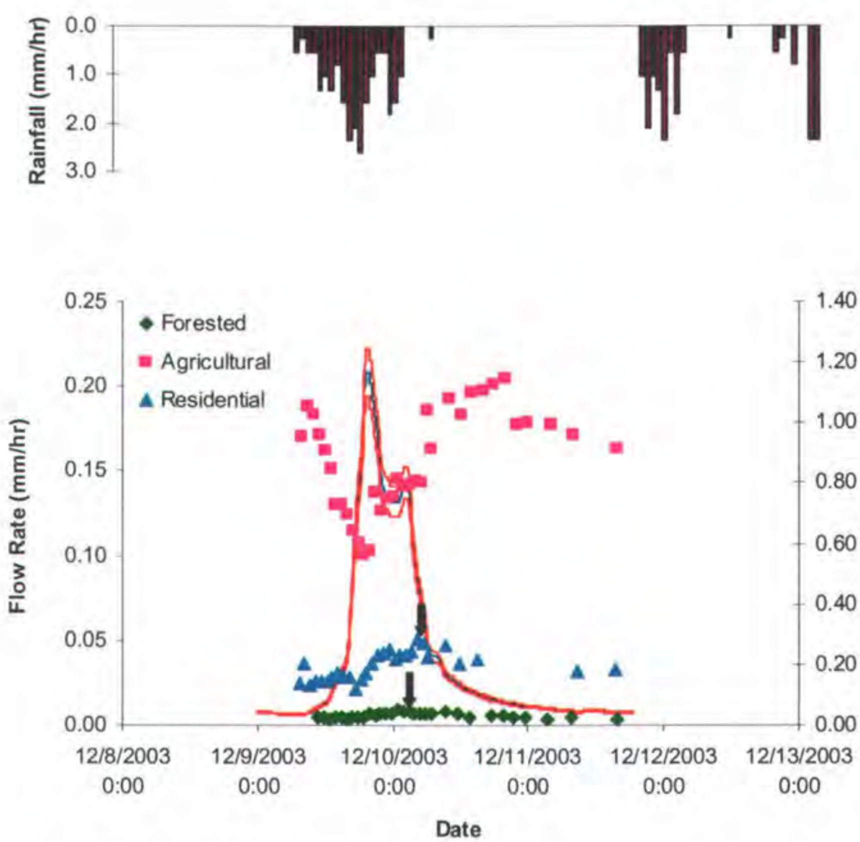

Figure 2.2. Nitrate response to storm 1 in the three study catchments. Only the agricultural hydrograph is shown for clarity. Arrows indicate the hydrograph peak for the forested and residential catchments.

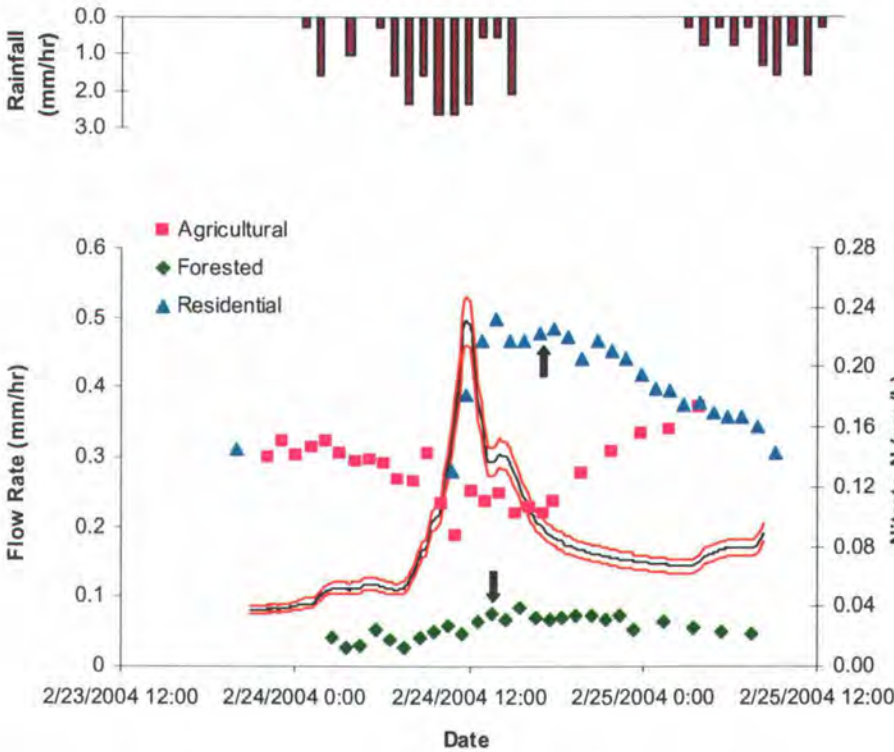

Figure 2.3. Nitrate response to storm 2 in the three study catchments. Only the agricultural hydrograph is shown for clarity. Arrows indicate the hydrograph peak for the forested and residential catchments.

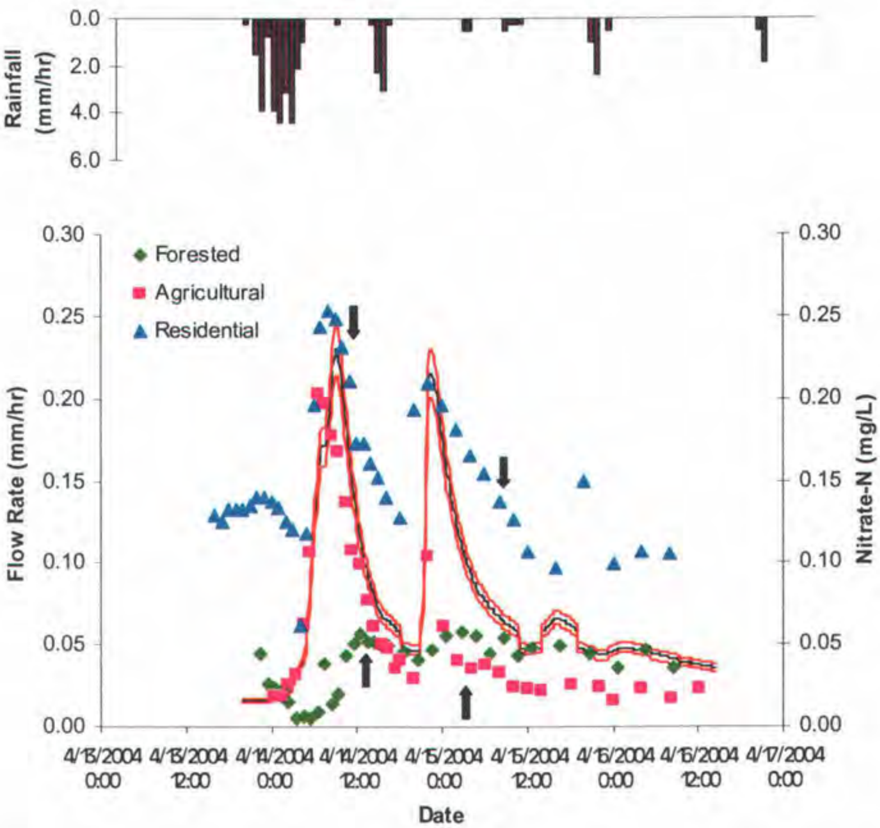

Figure 2.4. Nitrate response to storm 3 in the three study catchments. Only the agricultural hydrograph is shown for clarity. Arrows indicate the hydrograph peak for the forested and residential catchments.

Nitrate Response to Storm Events. Export rates of nitrate in all catchments were highest during storm 1. Rates in the forested, agricultural, and residential catchments were 0.012, 0.121, and 0.131 kg/ha/storm, respectively. Figure 2.5 shows export rates for each catchment during the three storms, with uncertainty bars that have been carried through from the hydrograph uncertainty. Uncertainty due to analytic methods was also included, but the quantified uncertainty is so small it is insignificant (on the order of 1E-7 kg/ha). The highest nitrate concentrations in the agricultural catchment were in the fall, due to the summer buildup of nitrogen (62 kg

N/ha applied). Biweekly samples, storm event samples, and export rates revealed a progressive decrease of nitrate concentrations throughout the year. Export rates in the agricultural catchment were 0.121, 0.040, and 0.021 kg/ha/storm for storms 1, 2, and 3, respectively. Nitrate export rates in the forested and residential catchments were relatively constant. The highest export rates occurred in the residential catchment during all three events (0.108-0.131 kg/ha/storm), and the lowest export rates occurred in the forested catchment (0.005-0.012 kg/ha/storm). The high export rates in the residential catchment are likely due to the high baseflow observed in the residential catchment throughout the year and not high nitrate concentrations, which is evident from the baseflow (Tables 2.3 and 2.4) and concentration plots (Figures 2.2 through 2.4).

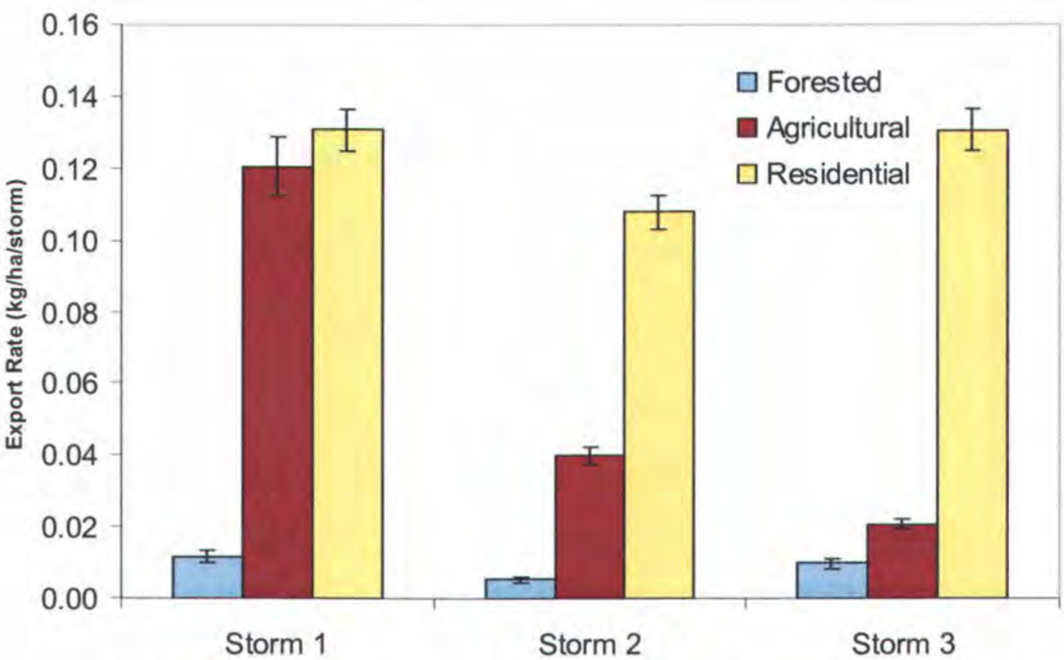

Figure 2.5. Nitrate export rates in the three study catchments during storms 1, 2, and 3.

The nitrate response to the storm events in each catchment are shown in Figures 2.2 through 2.4. In the forested and residential catchments, nitrate increased with increasing flow rates during storms 1, 2, and 3. A "concentration" pattern was observed during all storm events. Concentrations ranged from 0.005 to 0.06 mg/L as N and 0.06 to 0.29 mg/L as N in the forested and residential catchments, respectively. In the agricultural catchment, nitrate concentrations decreased with increasing flow rates during storms 1 and 2, and increased with increasing flow rates during storm 3. A "dilution" pattern was observed during storms 1 and 2, and a "concentration" pattern was observed during storm 3. Nitrate concentrations progressively decreased through the rainy period, from 0.6 to 1.1 mg/L as N in the fall, 0.09 to 0.17 mg/L as N in the winter, to 0.02 to 0.20 mg/L as N in the spring. Nitrate concentrations were lowest in the forested catchment during all storms. During storm 1 nitrate concentrations were highest in the agricultural catchment. Baseflow concentrations were about the same in the agricultural and residential catchments prior to storm 2 (~0.15 mg/L as N). Peak nitrate concentrations are therefore higher in the residential catchment during storm 2, since baseflow concentrations are about the same and nitrate concentrations exhibit a "concentration" pattern in the residential catchment and a "dilution" pattern in the agricultural catchment. Nitrate concentrations were highest in the residential catchment during storm 3. Baseflow nitrate concentrations in the agricultural catchment were much lower than the residential catchment during storm 3 (~0.017 mg/L as N in the agricultural catchment and ~0.14 mg/L as N in the residential catchment), although the peak nitrate concentration in the agricultural catchment is on the order of the peak nitrate concentration in the residential catchment (0.20 and 0.25 mg/L as N in the agricultural and residential catchments, respectively).

Discussion

Land use change effect on stream nitrate is poorly understood despite the increasing concerns for stream ecosystem health (Howarth et. al., 2002). The majority of the work on land use effects has focused on baseflow or a small number of sampling events correlating land use and nitrate (e.g. Schilling, 2002). While it is clear that land use affects the magnitude of nitrate and other nutrients exported from catchments, it is not clear how it affects nutrient dynamics or the nutrient concentration pattern during storm events. The few studies that have been conducted in catchments with mixed land use during storm events have reported mainly monthly exports, with little analysis of nitrate concentrations under varying discharge dynamics. Those studies that have analyzed concentration-discharge responses and coupled hydrobiogeochemical processes have been focused on exclusively forested or agricultural catchments. Our work in this paper presents results from 3 neighboring headwater catchments in western Oregon with similar (low) atmospheric deposition, size and geology but with different, consistent land use expressions: forest, agriculture and residential. This follows work that we have presented in other parts of the USA (see Burns et al., 2005) where land use change effects on hydrological and biogeochemical processes have been quantified.

Seasonal Trends. Nitrate concentrations and export rates are always low in the forested catchment. This is likely due to the lack of anthropogenic inputs in this region. Inputs of nitrate in the forested catchment include atmospheric deposition and nitrogen fixation/microbial processing in the soil. Both of these inputs are relatively low locally (Sylvia et. al., 1998), resulting in low stream concentrations. In addition, baseflow in the forested catchment is lower than baseflow in the agricultural catchment during storm 2 and lower than baseflow in the residential catchment during all storms. Peak flow in the forested catchment is lower than peak flows in the agricultural and residential catchments during all storms as well. The lower flow rates and nitrate concentrations produce low export rates.

In the agricultural catchment, nitrate concentrations and export rates are high in fall, medium in the winter, and low in the spring. This is likely due to the activities occurring in the agricultural catchment that are absent in the other two catchments. During the summer months, when the streambed is dry, approximately 62 kg N/ha are applied to fields within the catchment (Tom Nichols, personal communication, 2004). We would expect that this nitrogen is incorporated into the soil before the catchment "wets up." An estimated 56 kg N/ha of this applied source is taken up by grass growth, transformed to gas via volatilization and dentrification, and binds to soil particles as organic nitrogen based on averages for the area (Moore and Gamroth, 1993). Excess nitrogen at the end of the growing season is estimated to be approximately 6 kg/ha. Peak flows also occur much more quickly in the agricultural catchment, and are higher during storms 1 and 3. A quicker time to peak may be due to less throughfall occurring in the catchment; only 52.8% of the catchment is covered with trees compared to 83.1% and 98.1% in the residential and forested catchments, respectively. Less interception will occur in the agricultural catchment, causing a more rapid input of rainfall to the catchment (other things being equal). The higher peaks in the agricultural catchment in the fall and spring (storms 1 and 3) may be due to smaller subsurface storage zones in this catchment as evidenced through lower baseflow generally, and flow disappearance in the summer months. We estimated the agricultural catchment storage using the recession curve analysis of Vitvar et. al. (2002). Hydrographs of the three catchments were plotted in log space, and the recession limbs of four storm events were used to determine the recession coefficient. Recession coefficients were 0.119, 0.141, and 0.087 d^{-1} for the forested, agricultural, and residential catchments, respectively. We then used the recession coefficient, baseflow, and peak flow to derive mean hydraulic conductivity, storage coefficient, and subsequently estimate storage volume. Vitvar et. al. (2002) showed that this method provides similar results to the more rigorous convolution integral approach relating rainfall $\delta^{18}O$ to streamflow $\delta^{18}O$. Calculations using this approach suggest that storage volume in the agricultural catchment is approximately 64% and 88% less than that of the forested and residential catchments, respectively. These faster, higher peak flows in the agricultural catchment likely deliver nitrate to the stream more quickly, eventually depleting the applied fertilizer source.

Nitrate export rates are consistently high due to high flow rates in the residential catchment. Baseflow is consistently higher than in the other two catchments, although it is lower before storm 1 than before storms 2 and 3. Increasing baseflow throughout the year can be attributed to the "wetting up" of the catchment; source waters are replenished as the rainy period progresses. Baseflow concentrations prior to each storm stay relatively constant (0.15, 0.15, and 0.13 mg/L as N prior to storms 1, 2, and 3, respectively), and peak concentrations are very similar (0.27, 0.23, and 0.25 mg/L as N for storms 1, 2, and 3, respectively) which controls the relatively constant export rates throughout the year. The main source of nitrogen in the residential catchment is fertilizer application to lawns and ornamental shrubs. Although it appears that a large amount of nitrate is applied and exported to the stream, the high export in the fall is largely due to relatively high flow rates and not high nitrate concentrations. One surprising result of this study is the time to peak of the three catchments; the time to peak was the longest in the residential catchment for all storms. We would expect the time to peak to be shorter in the residential catchment compared to the forested catchment because of the impervious area, storm drains, lined portions of the stream, and the lower proportion of forest cover. We suspect that the marshy area, which is approximately 20 m upstream of the outlet of the catchment and downstream of the housing development, delays peak flows. The stream channel becomes split into many channels in the marshy area, and many pools are formed. This increase in complexity likely delays the time to peak observed at the outlet of the catchment.

Export rates increased with increasing development, which is in general agreement with other land use studies (Salvia-Castellvi et. al., 2005; Schilling, 2002;

Jordan et. al., 1997; Owens et. al., 1991). Maximum nitrate concentrations were found in the watershed with the highest proportion of agriculture in a study conducted in Luxembourg with watersheds having various proportions of land use (Salvia-Castellvi et. al., 2005). In a paired agricultural and restored prairie watershed study, more nitrate and chloride was exported in the agricultural than the restored watershed (Schilling, 2002). Nitrate concentrations increased as the proportion of cropland increased in a study of 27 watersheds with varying proportions of cropland and forested land uses (Jordan et. al., 1997). In a study of four mixed agricultural watersheds, nitrate export generally increased with increasing watershed development (Owens et. al., 1991). Typical seasonal trends in nitrate were found to correspond with streamflow; high export rates occurred during wet periods and periods of high flow and low export rates occurred during dry periods (Salvia-Castellvi et. al., 2005; Schilling, 2002; Owens et. al., 1991). However, none of these studies show the marked difference in export rates between the agricultural catchment, which progressively decreases throughout the rainy period, and the relatively constant export rates of the residential and forested catchments. The decrease in nitrate export in the agricultural catchment further indicates that varying nitrate inputs have a large affect on nitrate dynamics.

Sources of Streamflow. To determine whether differences in nitrate dynamics were due to differences in hydrology and flowpaths, sources of streamflow need to be identified. Mixing diagrams were used to determine sources of streamflow in the forested, agricultural, and residential catchments (Figures 6 through 8). Biweekly streamflow, storms 1, 2, and 3 streamflow, soil water, groundwater, and rainfall sulfate and chloride concentrations are presented. Sulfate and chloride are considered quasi-conservative chemicals within these catchments, especially during the short duration of the storm events. Although sulfate can undergo transformations in the soil and chloride has been found to sorb to some types of soil, some useful trends are still revealed with these plots. Other studies have also used sulfate in EMMA analyses (Hooper et. al., 1990; Christophersen and Hooper, 1992). The groundwater shown in Figures 6 and 8 refers to the samples taken in the residential catchment. In Figure 7, shallow groundwater refers to the sample taken in the agricultural catchment and deep groundwater refers to the samples taken in the residential catchment. Since the well in the residential catchment is relatively deep and the other catchments are in close proximity, we assume that this sample is representative of the groundwater for the entire area.

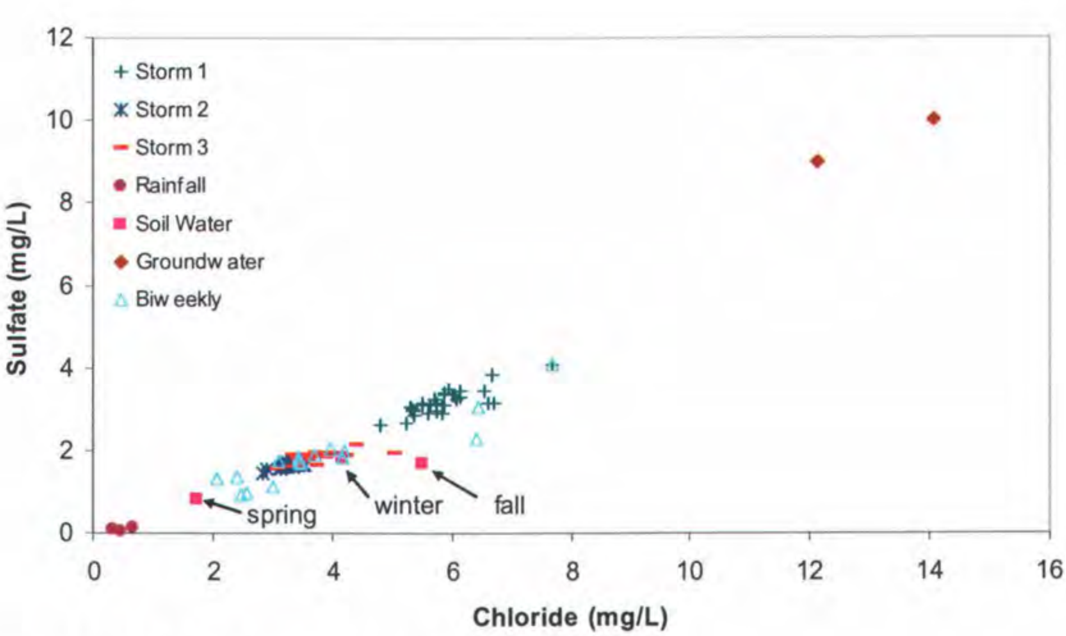

Figure 2.6. Mixing diagram for forested catchment.

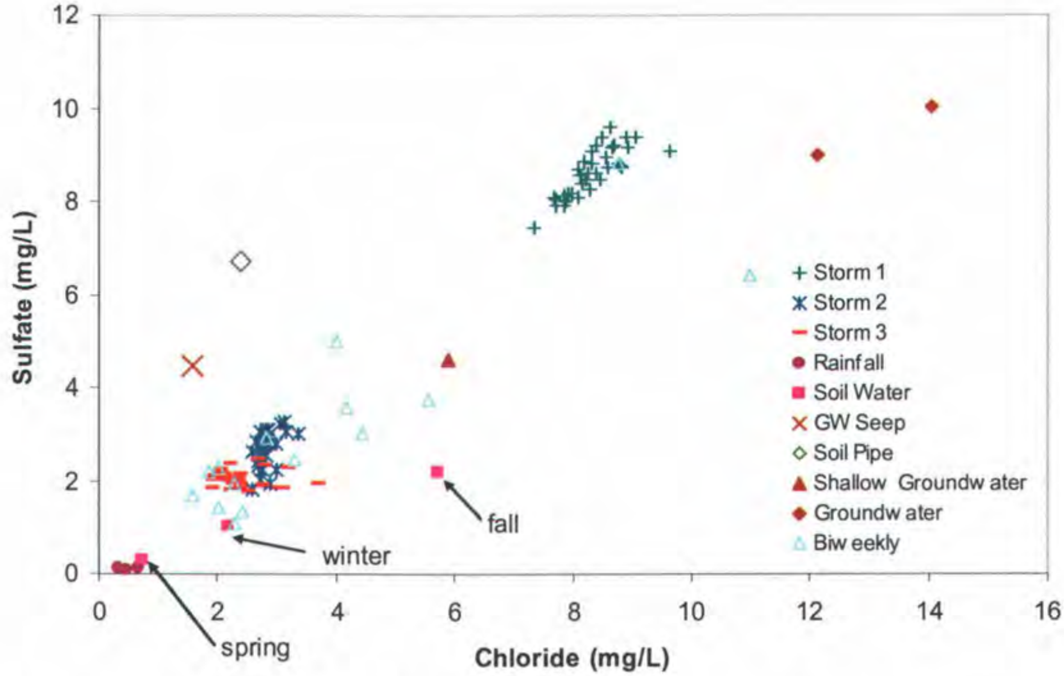

Figure 2.7. Mixing diagram for agricultural catchment.

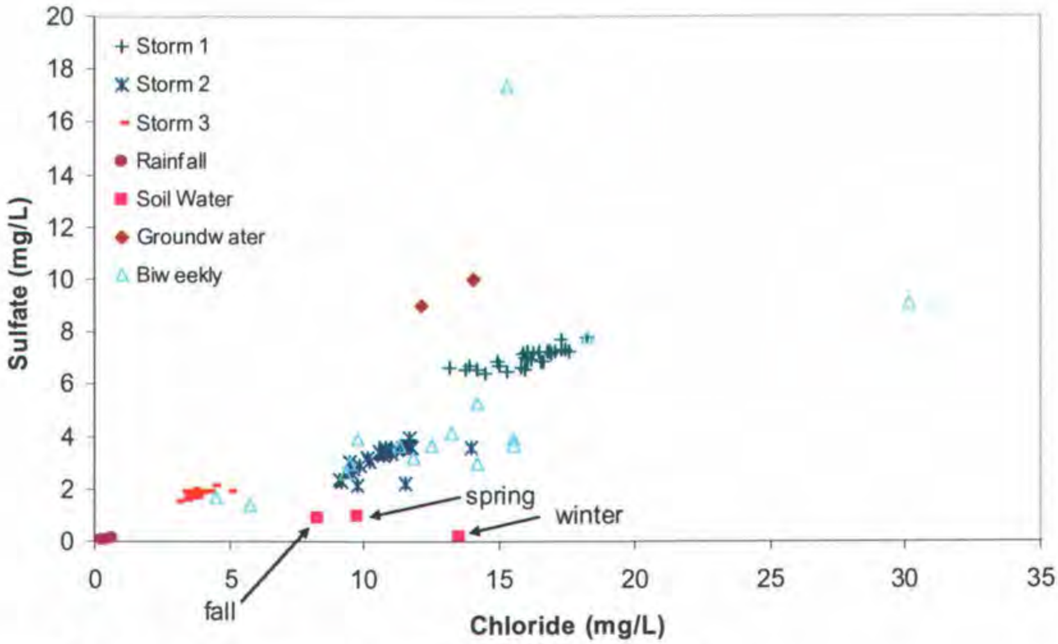

Figure 2.8. Mixing diagram for residential catchment.

Although streamflow is not entirely bound by end members, some inferences can still be made from the mixing diagrams. In the forested catchment, the mixing diagram is strongly linear, suggesting two sources of streamflow: groundwater and rainfall (Figure 2.6). Soil water is essentially the same as stream water. The source of streamflow migrates towards rainfall from fall to spring, suggesting that rainfall becomes a more dominant source as the rainy period progresses. During the dry periods, groundwater is the main source of streamflow. Rainfall becomes a larger source as the catchment wets up, which is evident from the progression of storm and biweekly samples towards rainfall concentrations. Soil water also shows this shift from groundwater to rainfall throughout the year. This is likely due to dilution from rainfall, and shows that soil water cannot be considered a constant source.

In the residential catchment, storms 2, 3, and winter and spring biweekly streamflow are bound by groundwater, soil water, and rainfall (Figure 2.8). The linear progression of storm and biweekly streamflow (similar to the progression in the forested catchment) suggests that streamflow is shifting from groundwater-dominated in the fall to rainfall-dominated in the spring. Soil water also shifts toward rainfall from winter to spring. The fall soil water is closer to the rainfall source, but this may be an anomaly; the general trend throughout the year for soil water is towards decreasing chloride concentrations (data not shown). Storm 1 and fall biweekly streamflow exhibit higher chloride concentrations than soil and groundwater, thus placing them outside of the bounds formed by the end members. Although further sampling is needed to validate end members, we surmise that there is another source of groundwater (perhaps shallower/deeper?) in the catchment with higher chloride and sulfate concentrations.

The mixing diagram for the agricultural catchment indicates that additional sources contribute to streamflow besides groundwater, rainfall, and soil water (Figure 2.7). The soil pipe and groundwater seep samples (see Figure 2.1 for locations) have a different chemical makeup than the other sources, and contribute to streamflow as well. Although it is not clear whether one or both are sources for streamflow, it is evident that the streamflow from the agricultural catchment has a different source apportionment. Storm 1 and the fall biweekly samples are dominated by groundwater and the soil pipe/groundwater seep, which is significantly different from the rainfall dominance during storms 2, 3, and the remaining biweekly samples. The difference between nitrate concentrations in fall streamflow and winter and spring streamflow may be due to a shift in sources from groundwater to rainfall dominated (similar to the behavior in the forested and residential catchments), or it may be due to a depletion of nitrogen in the soil. It is likely that both mechanisms are occurring. Although the seasonal progression of streamflow and soil water is similar to the progression in the forested and residential catchments, the difference in nitrate concentrations in storm 1 and fall biweekly samples and storms 2, 3, and the remaining biweekly samples is much larger in the agricultural catchment than the other two catchments.

Hydrology vs. Land use. It is evident from the analysis above that in addition to differences in land use, there are differences in hydrology between the three catchments. Study catchments are three previously ungauged catchments that have common headwaters, similar geology and soil type, atmospheric deposition, and are in close proximity. However, these catchments have slight differences that are reflected in the hydrology; the agricultural stream is ephemeral, the residential catchment has higher baseflows and a slower time to peak, and the forested catchment has steeper slopes. The distribution of topographic index (TI) also shows the slight difference in the three catchments (Figure 2.9). Although distributions are similar, the forested catchment has more areas with low TI (which indicates steep slopes and a small upslope contributing area), and the agricultural and residential catchments have more areas with high TI (areas with flatter slopes and a large upslope contributing area). The residential catchment has more areas with low TI than the agricultural catchment, which likely reflects the steep upper portion of the residential catchment. The skewness of the TI distribution was also smaller for the forested catchment (1.11) compared to the agricultural and residential catchments (1.22 and 1.39 for the agricultural and residential catchments, respectively).

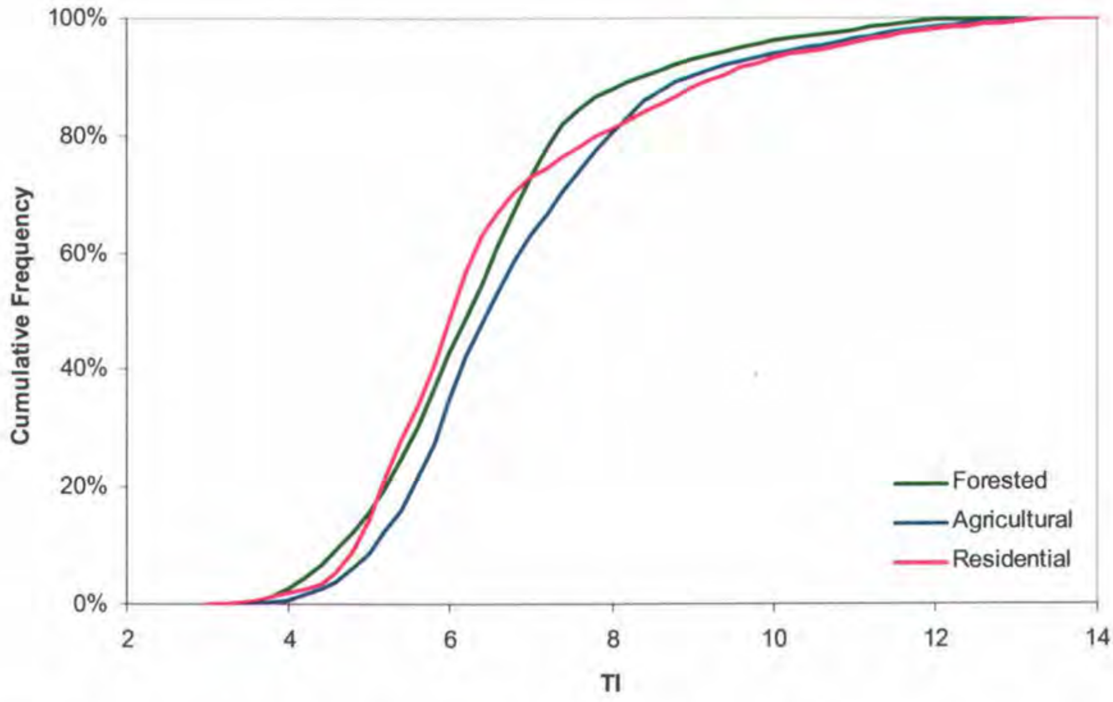

Figure 2.9. Cumulative frequency distribution of TI for the three study catchments.

We believe that these differences, in addition to the other sources of streamflow and lower storage volume in the agricultural catchment, contribute somewhat to the differences in nitrate dynamics. Different sources will contribute different amounts of nitrate to the stream, and flowrates will be different (i.e., groundwater flowrates are expected to be much smaller than overland flowrates). However, land use is also affecting nitrate dynamics. In the agricultural catchment, nitrogen builds up in the soil from the significant summer application of manure and green bean waste. This buildup is depleted from the soil by the end of the year. In contrast, a significant buildup of nitrogen in the soil does not occur in the forested and residential catchments. Separating the effects of hydrology and land use is difficult, since hydrology and land use are linked (in terms of impervious areas, storm drains, compaction of the soil, and/or removal of riparian vegetation). The shift in stream water sources are similar in the three catchments, shifting from groundwater dominated in the fall to rainfall dominated in the spring, indicating that the hydrology is somewhat similar. We argue that the difference between nitrate patterns in the three catchments during the three storm events is attributed more to land use than background hydrological differences.

Flushing of Nitrate. In the forested and residential catchments, a "concentration" response occurred during storms 1, 2, and 3, which is indicative of accumulated soil nitrate being flushed to the stream. This "concentration" response is similar to the results of other studies conducted in undisturbed catchments (Inamdar et. al., 2004; McHale et. al., 2002; Burns et. al., 1998). The mechanism behind this response has been described as the flushing hypothesis, where the N-enriched upper soil layer is flushed after a period of low demand, often during spring snowmelt and fall storms, when the water table rises to previously unsaturated portions of the N-enriched soil layer (Creed et. al., 1996; Creed and Band, 1998). Flushing would thus be expected during the first few fall storms in our catchments.

No seasonal variation in nitrate behavior during storm events was observed. The mixing diagram for the forested catchment shows streamflow shifting from groundwater-dominated to rainfall-dominated, and soil water is not a significant source (Figure 2.6). Flow rates in the Pacific Northwestern United States are characterized by a long and pronounced rain-free low flow period followed by a defined wetting-up period in the fall and then constantly fluctuating flow from multiple rain events (with little seasonal increase following wet-up). These hydro-period differences result in mechanisms for labile nutrient mobilization that are different to those described by Creed et. al. (1996) for the Northeastern areas of the United States. Their flushing hypothesis is predicated upon microbial activity and slow plant uptake over a quasi-dormant period (winter, sub-snow) which allows nitrate buildup. Nitrate is then flushed out as the catchment wets up during spring melt and storm events. Other draining mechanisms described by Burns et. al. (1998) and McHale et. al. (2002) for the Northeastern United States also do not appear to occur in our catchment.

A marked difference in nitrate patterns occurred in the agricultural catchment, with a seasonal shift from a "dilution" storm pattern in the fall and winter to a "concentration" pattern in the spring. Webb and Walling (1985) observed the same seasonal response of a "dilution" response during the winter period and a "concentration" response in the spring and summer months in a grassland (agricultural) catchment. They found that seasonal change in nitrate behavior during storm events could be explained by the influence of soil throughflow. Soil moisture increased and saturated areas expanded in their catchment, which resulted in higher contributions of storm water to the stream, diluting nitrate concentrations. As their variable source saturated areas decreased and the catchment drained in the spring and summer, rainfall was more likely to enter the soil profile and displace stored water with higher nitrate concentrations that then became the more dominant source of streamflow. These processes likely operate in our agricultural study catchment, since saturated areas, groundwater seeps, and soil pipes were observed throughout the catchment during the winter months. However, at our agricultural site nitrate concentrations also decrease throughout the year, indicating that soil nitrate pools become depleted. Although the shift from a "dilution" pattern to a "concentration" pattern indicates more water is coming into contact with soil nitrate pools before reaching the stream, these same pools are also exhausted as the rainy season progresses.

Implications for Watershed Development. Human activity has altered the movement of nitrate in the agricultural catchment to a larger degree than the residential catchment. Nitrate concentrations are higher in the agricultural catchment in the fall, which is a clear result of manure and green bean application and quick delivery to the stream. Nitrate concentrations in the agricultural catchment are high until the source is depleted, which is in contrast to the consistent nitrate concentrations in the residential catchment. Results from the residential catchment are not as easy to deconstruct; we were unable to locate one of the sources of streamflow during this study. This source may be anthropogenic or may be natural. Export rates were also highest compared to the other two catchments, but this is largely due to high flow rates and not high nitrate concentrations. Nonetheless, compared to our forested reference site, the nitrate dynamics or patterns are not significantly altered in the residential catchment. The residential catchment, with its low density housing and well-wooded yards (and no septic inputs), may be a model for minimal impact on nutrient dynamics within a watershed. Tree cover has been retained (83.1% compared to 52.8% in the agricultural catchment), and the lower portion of the catchment is used as a park. Portions of the park are well developed, with baseball fields and lawns, but other portions are relatively natural, with marshy areas and significant riparian woods acting as control measures. It is these control measures that prevent nitrate patterns from being affected by the development; residential areas with a higher housing density but similar spatial layout may have higher nitrate export rates but the patterns during storm events would likely stay the same. The marshy areas and riparian woods delay runoff response and may retain some of the exported nitrogen from the upper portion of the catchment. We could imagine a scenario, however, where there is so much alteration of the catchment from development that these control measures could be "swamped" out and no longer useful. More work would be needed to determine the ideal housing density and spatial layout of control measures.

Conclusions

Most process work to date that deals with the effect of land use and land use change on stream nitrate has been done in areas with one land use (minimally disturbed or agricultural) and areas with substantial atmospheric deposition. Our analysis of 3 neighboring headwater catchments in western Oregon with similar (low) atmospheric deposition, size, and geology but with different, consistent land use expressions revealed the following:

1. Human activity altered the patterns of stream nitrate concentrations during storm events in the agricultural catchment to a larger extent compared to the residential catchment. Nitrate response patterns in the residential catchment were the same as the patterns in the reference forested catchment (a "concentration" pattern throughout the year), whereas a "dilution" pattern was observed in the fall and winter and a "concentration" pattern was observed in the spring in the agricultural catchment.
2. Manure and green bean application in the agricultural catchment significantly increased nitrate concentrations and exports in the fall, which decreased throughout the year as the source became depleted. This is in contrast to the relatively constant export rates in the forested and residential catchments, which likely had a more constant source of nitrate (i.e., no large source inputs).
3. Streamflow in the forested, agricultural, and residential catchments moved from groundwater-dominated to rainfall-dominated as the rainy period progressed. Additional streamflow sources were identified in the agricultural catchment, which may include a groundwater seep and soil pipe.

Acknowledgements

We thank the following colleagues for field support: Derek Godwin, John Pennington, Jeff Mutti, Jesse Melick, and Michael Dewey. We thank Arne Skaugset for flow data, the Environmental Protection Agency for precipitation and residential groundwater data, Pete Nelson for help with laboratory techniques, and Kellie Vache for useful conversations. Many thanks to Doug Burns for reviewing an early version of this manuscript and providing valuable editorial advice. This research was funded by the United States Geological Survey and the Center for Water and Environmental Sustainability.

References

American Society of Agricultural Engineers, 2003. Manure Production and Characteristics, ASAE D384.1 FEB03, ASAE Standards. American Society of Agricultural Engineers.

Arheimer, B. and Liden, R., 2000. Nitrogen and phosphorous concentrations from agricultural catchments – influence of spatial and temporal variables. Journal of Hydrology 227, 140-159.

Bolstad, P.V., and Swank, W.T., 1997. Cumulative impacts of landuse on water quality in a Southern Appalachian watershed. Journal of the American Water Resources Association 33, 519-533.

Buck, O., Niyogi, D.K., and Townsend, C.R., 2004. Scale-dependence of land use effects on water quality of streams in agricultural catchments. Environmental Pollution 130, 287-299.

Burns, D., 2005. What do hydrologists mean when they use the term flushing? Hydrological Processes 19, 1325-1327.

Burns, D.A., Murdoch, P.S., Lawrence, G.B., and Michel, R.L., 1998. Effect of groundwater springs on NO_3^- concentrations during summer in Catskill Mountain streams. Water Resources Research 34, 1987-1996.

Burns, D., Vitvar, T., McDonnell, J., Hassett, J., Duncan, J., Kendall, C., 2005. Effects of suburban development on runoff generation in the Croton River basin, New York, USA. Journal of Hydrology 311, 266-281.

Christophersen, N., and Hooper, R.P., 1992. Multivariate analysis of stream water chemical data: the use of principal components analysis for the end-member mixing problem. Water Resources Research 28, 99-107.

Clesceri, L.S., Greenberg, A.E., and Eaton, A.D. (eds.), 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition. American Public Health Association, American Water Works Association, Water Environment Federation, Washington, D.C..

Creed, I.F. and Band, L.E., 1998. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research 11, 3105-3120.

Creed, I.F., Band, L.E., Foster, N.W., Morrison, I.K., Nicolson, J.A., Semkin, R.S., and Jeffries, D.S., 1996. Regulation of nitrate-N release from temperate forests: A test of the N flushing hypothesis. Water Resources Research 32, 3337-3354.

Donner, S.D., Kucharik, C.J., and Foley, J.A., 2004. Impact of changing land use practices on nitrate export by the Mississippi River. Global Biogeochemical Cycles 18, GB1028, 1-21.

Gordon, N.D., McMahon, T.A., Finlayson, B.L., 1992. Stream Hydrology: an introduction for ecologists. Wiley, New York, pp. 163-167.

Herlihy, A.T., Stoddard, J.L., and Johnson, C.B., 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic Region, U.S. Water, Air, and Soil Pollution 105, 377-386.

Hooper, R.P., Christophersen, N., and Peters, N., 1990. Modelling streamwater chemistry as a mixture of soilwater end-members – an application to the Panola Mountain catchment, Georgia, U.S.A. Journal of Hydrology 116, 321-343.

Howarth, R.W., Boyer, E.W., Pabich, W.J., and Galloway, J.N., 2002. Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31, 88-96.

Inamdar, S.P., Christopher, S.F., and Mitchell, M.J., 2004. Export mechanisms for dissolved organic carbon and nitrate during summer storm events in a glaciated forested catchment in New York, USA. Hydrological Processes 18, 2651-2661.

Johnson, L.B., Richards, C., Host, G.E., and Arthur, J.W., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37, 193-208.

Jones, K.B., Neale, A.C., Nash, M.S., Van Remortel, R.D., Wickham, J.D., Riitters, K.H., and O'Neill, R.V., 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16, 301-312.

Jordan, T.E., and Weller, D.E., 1996. Human contributions to terrestrial nitrogen flux. Bioscience 46, 655-663.

Jordan, T.E., Correll, D.L., Weller, D.E., 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33, 2579-2590.

Lattin, P.D., Wigington, P.J., Moser, T.J., Peniston, B.E., Lindeman, D.R., and Oetter, D.R., 2004. Influence of remote sensing imagery source on quantification of riparian land cover/land use. Journal of the American Water Resources Association 40, 215-227.

Little, J.L., Saffran, K.A., and Fent, L., 2003. Land use and water quality relationships in the lower Little Bow River watershed, Alberta, Canada. Water Quality Research Journal of Canada 38, 563-584.

McHale, M.R., McDonnell, J.J., Mitchell, M.J., and Cirmo, C.P., 2002. A field-based study of soil water and groundwater nitrate release in an Andirondack forested watershed. Water Resources Research 38, 1031, 1-17.

Moore, J.A., and Gamroth, M.J., 1993. Calculating the fertilizer value of manure from livestock operations. OSU Extension Service, Corvallis, Oregon.

Mosley, M.P., 1979. Streamflow generation in a forested watershed, New Zealand. Water Resources Research 15, 795-806.

National Research Council, 2000. Clean Coastal Waters: Understanding and reducing the effects of nutrient pollution. National Academy Press, Washington D.C..

National Science and Technology Council, 2000. Integrated assessment of hypoxia in the northern Gulf of Mexico. Committee on Environment and Natural Resources, Washington D.C.

Owens, L.B., Edwards, W.M., and Keuren, R.W., 1991. Baseflow and stormflow transport of nutrients from mixed agricultural watersheds. Journal of Environmental Quality 20, 407-414.

Petry, J., Soulsby, C., Malcolm, I.A., and Youngson, A.F., 2002. Hydrological controls on nutrient concentrations and fluxes in agricultural catchments. The Science of the Total Environment 294, 95-110.

Pimentel, D. (ed.), 1993. World soil erosion and conservation. Cambridge University Press, Cambridge, Massachusetts.

Rabalais, N.N., Turner, R.E., and Scavia, D., 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. Bioscience 52, 129-141.

Salvia-Castellvi, M., Iffly, J.F., Borght, P.V., and Hoffman, L., 2005. Dissolved and particulate nutrient export from rural catchments: a case study from Luxembourg. Science of the Total Environment 344, 51-65.

Schilling, K.E., 2002. Chemical transport from paired agricultural and restored prairie watersheds. Journal of Environmental Quality 31, 1184-1193.

Soil Conservation Service, 1975. Soil Survey of Benton County Area, Oregon. USDA.

Sylvia, D.M., Fuhrmann, J.J., Hartel, P.G., and Zuberer, D.A., 1998. Principles and Applications of Soil Microbiology. Prentice Hall.

Vanni, M.J., Renwick, W.H., Headworth, J.L., Auch, J.D., and Schaus, M.H., 2001. Dissolved and particulate nutrient flux from three adjacent agricultural watersheds: a five-year study. Biogeochemistry 54, 85-114.

Vitvar, T., Burns, D.A., Lawrence, G.B., McDonnell, J.J., and Wolock, D.M., 2002. Estimation of baseflow residence times in watersheds from the runoff hydrograph recession: method and application in the Neversink watershed, Catskill Mountains, New York. Hydrological Processes 16, 1871-1877.

Wayland, K.G., Long, D.T., Hyndman, D.W., Pijanowski, B.C., Woodhams, S.M., and Haack, S.K., 2003. Identifying relationships between baseflow geochemistry and land use with synoptic sampling and r-mode factor analysis. Journal of Environmental Quality 32, 180-190.

Webb, B.W., and Walling, D.E., 1985. Nitrate behaviour in streamflow from a grassland catchment in Devon, U.K.. Water Resources 19, 1005-1016.

Wellington, B.I. and Driscoll, C.T., 2004. The episodic acidification of a stream with elevated concentrations of dissolved organic carbon. Hydrological Processes 18, 2663-2680.

Wernick, B.G., Cook, K.E., and Schreier, H., 1998. Land use and streamwater nitrate dynamics in an urban-rural fringe watershed. Journal of the American Water Resources Association 34, 639-650.

Wigington, P.J., Baker, J.P., DeWalle, D.R., Kretser, W.A., Murdoch, P.S., Simonin, H.A., Van Sickle, J., McDowell, M.K., Peck, D.V., Barchet, W.R., 1996a. Episodic acidification of small streams in the Northeastern United States: episodic response project. Ecological Applications 6, 374-388.

Wigington, P.J., DeWalle, D.R., Murdoch, P.S., Kretser, W.A., Simonin, H.A., Van Sickle, J., and Baker, J.P., 1996b. Episodic acidification of small streams in the Northeastern United States: ionic controls of episodes. Ecological Applications 6, 389-407.

Woli, K.P., Nagumo, T., Kuramochi, K., and Hatano, R., 2004. Evaluating river water quality through land use analysis and N budget approaches in livestock farming areas. Science of the Total Environment 329, 61-74.

Chapter 3

Exploring the effects of land use, terrain, and spatial organization on lowflow nitrate concentration: Analysis of 135 catchments in Oregon and California

Cara J. Poor

John Bolte

Jeffrey J. McDonnell

For Submission to Environmental Science and Technology

Abstract

Predictions of streamwater nitrate in ungauged basins are still largely unsuccessful due to our lack of understanding of the processes controlling nitrate within the catchment, and our inability to mathematically represent these processes. In addition, statistical models using land use to predict stream nitrate concentrations are somewhat de-coupled from the body of literature that examines processes of catchment-scale biogeochemical cycling. This process knowledge needs to be incorporated into models in a simple mathematical way to improve predictions of nitrate. This paper presents our development of a statistical model with inverse-distance weighting of land use and an added index (TI) to predict hot spots for nitrate. We did not see an improvement of nitrate predictions using TI and inverse-distance weighting; the best models for both the Western Oregon and Northern California datasets were the %Landuse-Elevation models. We did, however, see an improvement of chloride predictions using TI and inverse-distance weighting; adding TI significantly improved model predictions and the best models for both Western Oregon and Northern California were the In*Out^{-1} Landuse-Elevation models. One interesting result of this study is the significance of elevation for predicting both nitrate and chloride. Elevation was originally used as a proxy for TI, and it is shown in the model results that it consistently predicts nitrate better than TI.

Introduction

The effect of catchment hydrological processes, catchment-scale biogeochemical processes, and land use on streamwater nitrate concentrations is poorly understood. While many investigators have examined coupled hydrobiogeochemical processes in upland forested catchments where atmospheric deposition is the only anthropogenic input (Mitchell et. al., 2003), it is still exceedingly difficult to make predictions of streamwater nitrate in ungauged basins. For catchments with mixed landuse, including agricultural development, urban, and suburban development, landuse has been found to exert a dominant control on streamwater nitrate concentrations (Schilling, 2002; Jordan et. al., 1997; Owens et. al., 1991; Howarth et. al, 2002). Many spatial statistical models have been proposed that regress proportions of different landuses in a catchment against stream nitrate concentration (Arheimer and Liden, 2000; Wickham et. al. 2002; Jones et. al., 2001). While correlation between land use and stream nitrate is well documented in a variety of climate and geographical settings, model regression coefficients (e.g., R^2) reported in such studies are rarely in excess of 0.5 (Norton and Fisher, 2000; Herlihy et. al., 1998; Johnson et. al., 1997). Catchment models using land use plus additional measures such as nitrogen inputs and annual flow rate have shown slightly more predictive power than models using land use alone (Hunsacker and Levine, 1995; Norton and Fisher, 2000; Arheimer and Liden, 2000; Jones et. al., 2001).

While empirical models are useful tools in many instances to predict concentrations in streams, they are somewhat de-coupled from the body of literature that examines processes of catchment-scale biogeochemical cycling. Several studies have examined the processes involved in nitrate transport, transformations, and storage (Hjerdt et. al., 2004; Petry et. al., 2002; Creed and Band; 1998; Jordan et. al., 1997; McHale et. al., 2002; Hornberger et. al., 1994). One somewhat common finding from this work is that hot spots (patches of the catchment with relatively high reaction rates, often enhanced at the terrestrial-aquatic interfaces) exert a profound control on streamwater nitrate dynamics (McClain et. al, 2003). The interface between oxic and anoxic zones (i.e., the interface between upland and riparian zones), is typically a hot spot for denitrification (Dahm et. al., 1998; McClain et. al., 2003; Peterjohn and Correll, 1984; Lowrance et. al., 1984). Microbial studies have found that hot spots for denitrification occur at sites where groundwater flowpaths transport nitrate to supplies of available organic carbon within the riaparian zone (Hill et. al., 2000; Sebilo et. al., 2003), and at sites with flooded or moist soils (Christensen et. al., 1990). Pinay et. al. (1989) and many others have shown that elevational differences can significantly affect soil saturation levels, which in turn affects the denitrification potential. Within wet zones of the catchment (e.g. riparian areas), the seasonality of soil saturation levels (perennially saturated, seasonally inundated, and dry or rarely inundated) has also been shown to affect denitrification potential and thus patterns of nutrient export (Baker et. al., 2001).

Topography and topographic position are simple measures that may aid the identification of hotspots in catchments. The well known topographic index (TI) of Beven and Kirkby (1979) (where $TI = \ln\left(\frac{a}{\tan\beta}\right)$, a is the upslope contributing area per unit contour length, and β is the local slope angle) has been found in studies conducted in headwater forested catchments to show a positive correlation with nitrate concentrations (Creed and Band, 1998; Welsch et. al., 2001). Creed and Band (1998) used TOPMODEL, the hydrologic model developed by Beven and Kirkby (1979) that incorporates TI to predict soil saturation, to predict the seasonal fluctuation in stream nitrate concentrations. Welsch et. al. (2001) found a positive correlation between soil water nitrate concentrations and TI. In general, low values of TI indicate steep areas with a low contributing area (likely the hillslope on the edge of a catchment) and a high TI value indicates flat areas with a high contributing area (likely the areas near the stream). For larger watersheds with mixed landuse, clearly landuse will mask the subtleties of topographic position seen in pristine forest catchments controlling streamwater nitrate concentrations. Nevertheless, topography and topographic position may still play a role. For instance, land use near the stream has been found to be a better predictor of water quality than land use over the entire watershed (Peterjohn and Correll, 1984; Lowrance et. al., 1984). Recent studies have used distance weights to account for the effects of spatial arrangement of land cover (King et. al., 2005; Hunsacker and Levine, 1995; Soranno et. al., 1996; Canham et. al., 2004; Kehmeier, 2000; Smith et. al., 1997). The majority of these studies use a mass-balance approach, with flowpath distances and an export coefficient for each type of land use (Canham et. al., 2004; Soranno et. al., 1996; Hunsacker and Levine, 1995).

Adding topographic and distance weighting effects to empirical catchment analyses is one way to begin to add a dimension of process representation and hotspot (patches of the catchment with relatively high denitrification rates) influence on streamwater nitrate predictions. To date, published export coefficients from studies that have begun to combine landuse and distance weighting have relied on values measured either in the lab or obtained from separate, published studies. This is similar to the decay coefficient that is sometimes added to account for transformation reactions, which is usually measured either in the lab or borrowed from the literature (Smith et. al., 1997). Obtaining a coherent dataset to test hypotheses of landuse, terrain attributes, and spatial organization has been difficult due to the large number of samples required to produce reliable results. Few studies use more than 50 or so catchments in their analysis, which calls into question the models' validity and parsimony when using several predictor variables. In this paper, we take advantage of an extensive 135 catchment database of low flow water quality sampling (modeling 76 catchments from one region and 59 catchments from another region separately) from the U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP), Regional Environmental Monitoring and Assessment Program (REMAP; Stoddard et. al., 2005), an EPA agricultural-riparian study database (Moser et. al., 1997), and a pre-pilot EMAP study (Herlihy et. al., 1997; Peck et. al., 2005a; Peck et. al., 2005b). This is the first use of the extensive EMAP and REMAP dataset for stream water quality research. A subset of this dataset has been used for biological research (Van Sickle et. al., 2004; Kehmeier, 2000) and water quality research (Pan et. al., 2004); however, no water quality research has been conducted on the entire dataset to date. We use these data to explore the use of TI, landuse, and distance weighting to predict stream nitrate concentrations. Low flow samples may help identify anthropogenic sources of stream nitrate concentrations more accurately than samples taken during wet conditions, which is based on the premise of previous studies showing land use significantly altering stream nitrate concentrations during baseflow conditions (i.e., Heisig, 2000). A strong correlation between unsewered housing density and stream nitrate concentrations during baseflow conditions has also been observed (Heisig, 2000). Although samples were collected during the dry summer months, TI may still be effective in predicting nitrate concentrations. Areas identified by TI as "hot spots" for nitrate will still be a source/sink, especially if build-up has occurred.

Using this large dataset, we go beyond what has been done to date in the hydrological literature with distance weights and borrow approaches used in biological studies using more sophisticated inverse-distance weighting that might aid in bringing more process representation into the regression model approach. We use an inverse-distance (1/d) and inverse-distance squared ($1/d^2$) method with in-stream and out-of-stream flowpath distances of land use to estimate streamwater nitrate concentration, following fish biomass study approaches developed by Kehmeier (2000). Out-of-stream distances are defined as the distance (following the flow paths in the catchment) from a location in the catchment to a point in the stream, and in-stream distances are defined as the distance from that point in the stream to the outlet of the catchment (following the stream network). We hypothesize that using in-stream and out-of-stream inverse-distance or inverse-distance squared flowpath distances of land use may improve nitrate predictions by accounting for near-stream processes (i.e., uptake by riparian vegetation, denitrification, etc.).

While the land use coverage, topography, and the new inverse distance weighting will help predict stream nitrate concentrations, we first need to separate the effects of the physical flowpath of water (and the nitrate in the water) from anthropogenic effects and biogeochemical processes. Chloride is often used as a conservative tracer in catchment studies and for comparison with reactive nutrients such as nitrate (Kirchner et. al., 2001; Neal and Rosier, 1990; Nyberg et. al., 1999). Under typical catchment conditions, chloride is nonreactive (Neal and Rosier, 1990), although chloride retention in the soil layer has been noted (Mulder et. al., 1990; Cook et. al., 1994; Nyburg et. al., 1999). Chloride varies seasonally and annually according to rainfall inputs and contribution from groundwater (Kirchner et. al., 2001; Neal and Rosier, 1990), which makes it useful as a hydrologic tracer (i.e., tracer of the water itself). Chloride has been used to calculate residence times of groundwater (Kirchner et. al., 2001; Nyberg et. al., 1999). With the use of topographic index (TI), which indicates saturated areas (Beven and Kirkby, 1979), chloride may be useful in separating the effects of hydrology from anthropogenic inputs for nitrate.

We developed a statistical model to predict stream nitrate concentrations using inverse-distance weighting of land use and an added index (TI) to identify and account for hot spots for nitrate. Inverse-distance (1/d) and inverse-distance squared ($1/d^2$) models, with in-stream and out-of-stream flowpath distances of land use and TI, were developed and evaluated against more traditional models using areal-weighted percentages of land use. Models substituting site elevation and slope for TI were also developed to determine whether TI significantly improves model results. Chloride was modeled using the same techniques for comparison. To evaluate the effects of region on model performance and coefficient selection, watershed land-cover and nitrate data from two different areas, Northern California and Western Oregon, were used to calibrate and validate the models. We explore the following questions:

1. Will identifying areas with a high denitrification potential using TI improve the prediction of stream nitrate concentrations?
2. Will models using in-stream and out-of-stream inverse distance (1/d) and inverse-distance squared ($1/d^2$) measures improve nitrate predictions?
3. Will models using TI improve nitrate predictions compared to models using a simple measure such as elevation or slope?
4. Due to the conservative nature of chloride, will chloride models help separate the effects of hydrology from anthropogenic inputs for nitrate?
5. Will topographic index be more influential predicting stream nitrate concentrations in the mostly forested catchments in Northern California, compared to the mixed land use of Western Oregon?

Methods

Study Areas. Catchments in Northern California and the Willamette River Basin in Western Oregon were used for model development. Both Northern

California and Western Oregon have a mediterranean climate, with wet winters and dry summers. Average long-term precipitation in Northern California and Western Oregon is 1528 mm and 1653 mm, respectively. Figures 3.1 and 3.2 show the location of sampling sites at the outlet of study catchments in Northern California and Western Oregon. In general, Northern California study catchments have a lower road density, higher elevation, and are mostly forested (Table 3.1). The dominant geology in Norhtern California is eugosynclinal, with granite, ultramafic, and andesite also present in significant amounts. Soil types include various types of clay, weathered alluvium, and volcanic soils. Western Oregon study catchments have a higher proportion of agricultural and urban land uses. Average long-term precipitation is slightly higher in Western Oregon, and the extremes (minimum and maximum long-term precipitation) are higher as well. The dominant geology in Western Oregon is calcerous-alakaline volcanics, with mafic volcanics, lake sediment, sandstone, and shale and mudstone also present in significant amounts. Soil types include various types of clay, weathered alluvium, and volcanic soils.

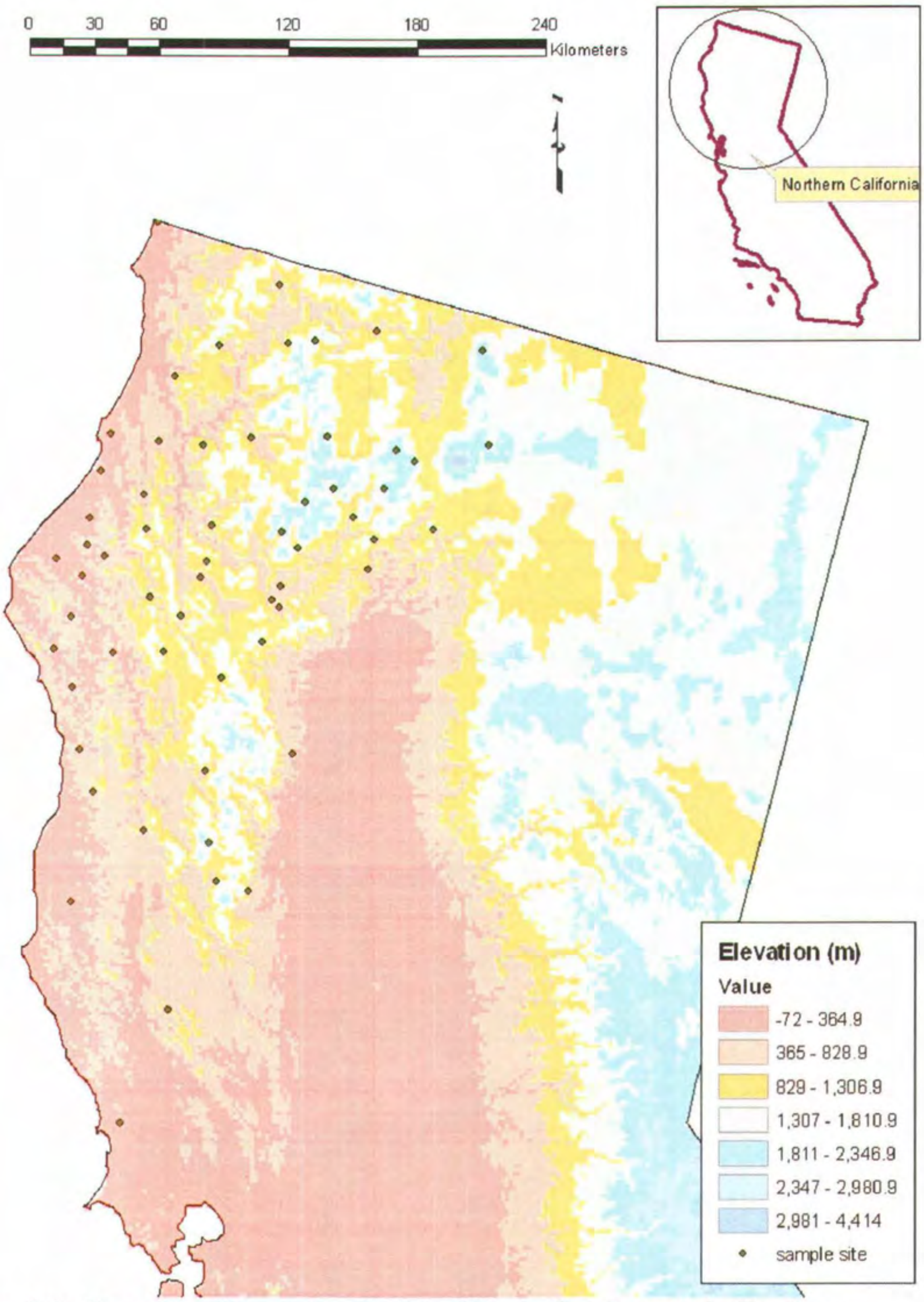

Figure 3.1. Sampling Sites at the Outlet of Study Catchments in Northern California.

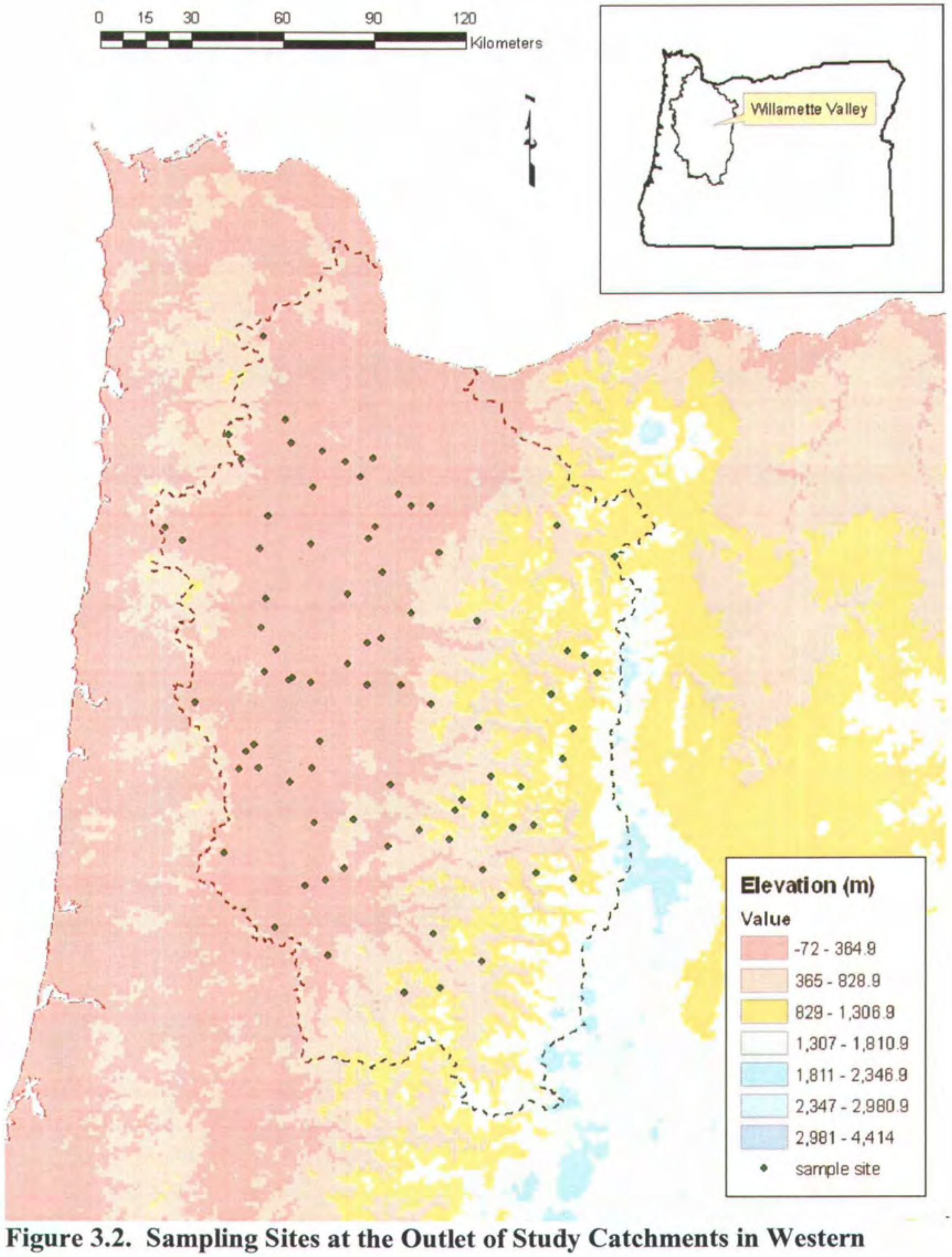

Figure 3.2. Sampling Sites at the Outlet of Study Catchments in Western Oregon.

	Statistic		
	Mean	Minimum	Maximum
	Western Oregon		
Watershed Area (ha)	3848	59	45867
Site Elevation (m)	339	24	1213
Mean Elevation (m)	544	49	1872
Slope at Site (%)	3.63	0.00	15.00
Mean Slope (%)	20.24	0.30	55.05
Road Density (m/m^2)	0.00264	0.00020	0.01100
Average Precipitation (mm)	1653	1029	3319
%Forest	73.32	0.70	100.00
%Agriculture	21.17	0.00	96.90
%Urban	5.52	0.00	82.81
	Northern California		
Watershed Area (ha)	5330	112	62299
Site Elevation (m)	709	14	1761
Mean Elevation (m)	1089	101	2173
Mean Slope (%)	19.41	7.70	27.21
Road Density (m/m^2)	0.00111	0.00000	0.00308
Average Precipitation (mm)	1528	691	2912
%Forest	99.79	96.01	100.00
%Agriculture	0.14	0.00	3.54
%Urban	0.08	0.00	2.13

Table 3.1. General Catchment Characteristics for Western Oregon and Northern California.

The Northern California dataset (71 catchments) was obtained from the Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP) and Regional Environmental Monitoring and Assessment Program (REMAP). Sites were selected using an unequal-probability, spatially-balanced survey design, which provides a measurement of a subset of all waters (Stoddard et. al., 2005). This measurement is intended to estimate the condition of streams and rivers in the western United States. The REMAP sampling area is nested within the EMAP sampling area, and provides a more intensive sampling in a smaller area. Sites for REMAP were also selected using unequal probability. All samples were collected during the summer low-flow period (May through September) in 2000, 2001, and 2002. The one-time sampling event involved taking one 4-L and 2 syringe (60 ml) samples in the middle of the stream at the sample point. Samples were analyzed for pH, major anions and cations, total nitrogen, and total phosphorous. Laboratory duplicates, a split of the 4-L sample, and analytical duplicates, putting the sample in different sample tubes on the instrument, were conducted for quality assurance.

The Western Oregon dataset (all samples located within the boundaries of the Willamette River Basin) includes 97 catchments with chemistry data and land use coverage, and is comprised of data from EMAP, REMAP, an EPA agricultural-riparian study (Moser et. al., 1997), and a pre-pilot EMAP study (Herlihy et. al., 1997; Peck et. al., 2005a; Peck et. al., 2005b). Site selection for the agricultural-riparian study was designed to determine the effect of riparian areas on the ecological condition of small, perennial streams in agricultural landscapes of Western Oregon, and was conducted as part of EPA's Pacific Northwest Program (Baker et. al., 1995). The study watersheds were restricted to watersheds within the prairie terraces or foothill subecoregions (Pater et. al., 1998), streams without major impoundments, watersheds with agriculture as the primary land cover and limited urban development, and riparian corridors with a wide range of woody vegetation cover. The pre-pilot EMAP study was a survey in the mid-Willamette River Basin designed to develop quantifiable and reproducible indicators (i.e., fish, macroinvertabrates, and physical habitat) of ecological condition. The first year focused on development of indicators in 16 randomly selected stream sites and two hand-picked sites. The following year, the same sites were sampled (except for one site where access was denied), an additional 13 sites were randomly selected, and five stream sites were chosen to represent the extremes in disturbance. A subset of the sites sampled during the first two years was sampled during the last few years of the study. Since samples were taken during multiple years for the pre-pilot EMAP study, the most recent samples were used in this study (which coincides with the time of EMAP sample collection). All sampling and laboratory analysis was conducted in the same manner as the EMAP and REMAP sampling.

Sampling locations from the agricultural-riparian study are distributed based on study objectives, and sampling locations from the pre-pilot study are distributed partially based on study objectives and partially randomly selected, which is in contrast to the completely randomly selected sites from EMAP and REMAP. Samples for the agricultural-riparian study and EMAP were collected in 1997. Pre-pilot samples were collected 1993-1997, and REMAP samples were collected in 1994 and 1995. All samples were collected during the summer low-flow period (June through September).

Land use/land cover data (30-m resolution) and digital elevation models (DEMs) were also provided by the EPA (Oetter et. al., 2000; Hulse et. al., 2002; Vogelmann et. al., 2001; Wickham et. al., 2004). In Western Oregon, satellite images were taken in 1990 for the land use/land cover data, with a classification accuracy varying from 50-100% (Hulse et. al., 2002). This accuracy is based on a comparison of the satellite images and aerial photos; some land cover classifications are easier to detect from satellite images (e.g., snow, water) than others (e.g., sugar beet crop, grass). Land cover is divided into 65 classifications, including different types of agricultural crops, forested/natural vegetation, and urban/human development. In Northern California, satellite images were taken in 1992 for the land use/land cover data, with an overall classification accuracy of 70% (Wickham et. al., 2004). Accuracy is based on the agreement between the reference land cover label and a mode class present in a 3x3 block centered on the sample point. See Wickham et. al. (2004) for details. Land cover is divided into 21 classifications, including different types of agricultural crops, forested/natural vegetation, and urban/human development. Both coverages are raster images or grids, which divide catchments into cells (one cell is 30-m x 30-m) then assign a value of land use/land cover or elevation to each cell. Several sites were removed from the dataset due to errors in the DEM in low gradient regions that caused inaccurate delineations of the watershed boundary and stream network. Some catchments also had missing water quality data. This resulted in 76 sites for the Western Oregon dataset (39 in the Willamette Valley and 37 in the upland Cascade and Coast Ranges) and 59 sites for the Northern California dataset.

Model Description. All land use characterizations were grouped into three land use categories: forested, agricultural, and urban. Natural vegetation, riparian vegetation, and forest were placed in the forest category. The agricultural category includes orchards, row crops, and any other type of farming activity. Roads, housing developments, and urban areas are included in the urban category. Classifications were grouped into these categories to decrease the number of variables in the resulting model. Topographic index was calculated for each cell (as defined by the DEM) in a watershed.

Using the DEM files for each catchment, a flow direction grid was created that describes the direction of flow (north, south, east, west, northeast, northwest, southeast, or southwest) for each cell in the catchment. The steepest gradient between a cell and the eight neighboring cells determined the direction of flow. The flow direction grid was then used to create a stream network with a minimum upstream drainage area of 500 cells (according to the DEM coverage). To calculate area, inverse-distance (1/d), and inverse-distance squared ($1/d^2$), an algorithm developed by Kehmeier (2000) was used. The algorithm calculated in-stream and out-of-stream total inverse-distance (1/d) and inverse-distance squared ($1/d^2$) for each cell in the catchment, then summed up the distance values (in-stream, out-of-stream, and total (in-stream + out-of-stream) 1/d and $1/d^2$) of each category of land use and TI. Out-of-stream distances were calculated according to flowpaths from the flow direction grid, and are defined as the flowpath distance from a location (cell) in the catchment to the point of entry to the stream. In-stream distances were calculated according to stream networks, and are defined as the distance in the stream from the point of entry (identified from the out-of-stream flowpath) to the outlet of the catchment. To minimize the effects of watershed size, land use variables were normalized by the total sum of all cells in the watershed (i.e., forested area was divided by total area, forested (in-stream distance)$^{-1}$ was divided by total (in-stream distance)$^{-1}$, etc.), and a weighted average was calculated for TI (i.e.,

$\sum_{i=1}^{n}$ [TI_i*(out-of-stream distance)$_i^{-2}$]/$\sum_{i=1}^{n}$ [(out-of-stream distance)$_i^{-2}$]). As a result, the three land use measures summed to one and the resulting solution was not unique. One of the land use measures had to be removed, and we chose the urban land use since it comprises such a small percentage of land use in the Western Oregon and Northern California catchments (5.52% and 0.08% for Western Oregon and Northern California, respectively).

Two general model forms were created to describe the various models. The first general model form is:

$$\log(NO_3^-) = a_0 + a_1(F_{eff}) + a_2(A_{eff}) + b_1(TI_{eff}) \qquad (1)$$

where a_0 is the intercept, a_1 and a_2 are land use coefficients, and b_1 is the TI coefficient. F_{eff}, A_{eff}, and TI_{eff} are the effects of forested land use, agricultural land use, and TI, respectively, which can be %Landuse, total inverse-distance ($Total^{-1}$), or total inverse-distance squared ($Total^{-2}$). Total inverse-distance and inverse-distance squared were calculated by summing in-stream and out-of-stream inverse-distance and inverse-distance squared, respectively. For the in-stream and out-of-stream models, the general model form is:

$$\log(NO_3^-) = a_0 + a_1(F_{eff1}) + a_2(F_{eff2}) + a_3(A_{eff1}) + a_4(A_{eff2}) + b_1(TI_{eff1}) + b_2(TI_{eff2}) \qquad (2)$$

where the subscript eff1 and eff2 denote the in-stream and out-of-stream effects, respectively. Variables with the eff1 subscript can be in-stream inverse-distance (1/d) or in-stream inverse-distance squared (1/d^2), and variables with the eff2 subscript can be out-of-stream inverse-distance (1/d) or out-of-stream inverse-distance squared

($1/d^2$). Models were created incorporating the in-stream and out-of-stream inverse-distance (In+Out^{-1}) and the in-stream and out-of-stream inverse-distance squared (In+Out^{-2}). To determine the relationship between in-stream and out-of-stream effects, a multiplicative model was also tested:

$$\log(NO_3^-) = a_0 + a_1(F_{eff1})*(F_{eff2}) + a_3(A_{eff1})*(A_{eff2}) + b_1(TI_{eff1})*(TI_{eff2}) \qquad (3)$$

The multiplicative models were depicted as In*Out^{-1} for inverse-distance and In*Out^{-2} for inverse-distance squared. In all models, elevation, slope, and watershed area were separately substituted for TI variables to determine whether TI significantly improves model results. The same model forms were used for the chloride models.

Since nitrate data in Western Oregon and Northern California are highly skewed toward zero (and not normally distributed), nitrate data were $\log_{10}$-transformed. Chloride data were also $\log_{10}$-transformed.

Model Determination and Calibration. Using the model forms described above, SAS version 9.1 (SAS Institute, 2003) was used to perform linear regressions. Variable correlations were performed to determine which independent variables have the potential to predict nitrate and chloride concentrations (Tables 3.2 and 3.3). Correlations between nitrate/chloride and all available data (percentage land use, slope, elevation, watershed area, and watershed road density) were initially calculated for both the Western Oregon and Northern California datasets. Additional correlations between nitrate/chloride and mean annual flow rate and the total length of all upstream streams were conducted for the Western Oregon dataset, and between nitrate/chloride and population density for the Northern California dataset. Mean annual flow rate and the total length of all upstream streams were not available for the Northern California dataset, and population density was not available for the Western Oregon dataset. Correlations between nitrate/chloride and explanatory variables greater than 0.2 were considered large enough to include the variables in the models; correlations less than 0.2 will likely be too weak to have significance. For the Western Oregon dataset, the largest correlations to nitrate were with slope, elevation, natural/forested, agriculture, and developed land use variables (Table 3.2). Slope, elevation, agriculture, natural/forested, and developed land use had the largest correlations to chloride. This indicates that TI (which is derived from slope) and land use variables might successfully predict nitrate and chloride. The largest correlations to nitrate were with wetlands, developed land use, and elevation for the Northern California dataset (Table 3.3). Chloride had the highest correlations with elevation, population density, agriculture, wetlands, and developed land use. Although correlations were not strong, land use may still be a useful predictor of nitrate and chloride. Both site and mean elevation were highly correlated to nitrate and chloride in each dataset. Models were created with site and mean elevation to determine which variable predicts nitrate and chloride better.

Variable	$\log(NO_3^-)$	$\log(Cl^-)$	%Ag	%For	%Nat Veg	%ADA	%Ag + ADA	%Ag + LDD	Rd Den	Site Slope	Site Elev	Mean Slope	Mean Elev	Wshd Area	Strm Len	Mean Q
$\log(NO_3^-)$	1.000	0.473	0.475	-0.398	-0.523	0.287	0.531	0.516	0.024	-0.394	-0.574	-0.467	-0.618	-0.029	-0.008	-0.100
$\log(Cl^-)$	0.473	1.000	0.630	-0.529	-0.636	0.263	0.652	0.661	0.008	-0.632	-0.754	-0.655	-0.793	0.027	0.037	-0.098
%Ag	0.475	0.630	1.000	-0.822	-0.842	0.111	0.898	0.977	-0.183	-0.546	-0.590	-0.725	-0.657	0.007	0.034	-0.116
%For	-0.398	-0.529	-0.822	1.000	0.906	-0.461	-0.902	-0.880	-0.241	0.486	0.504	0.659	0.583	0.077	0.057	0.195
%Nat Veg	-0.523	-0.636	-0.842	0.906	1.000	-0.589	-0.976	-0.925	-0.207	0.566	0.567	0.715	0.625	0.054	0.048	0.165
%ADA	0.287	0.263	0.111	-0.461	-0.589	1.000	0.537	0.315	0.726	-0.269	-0.259	-0.368	-0.316	-0.090	-0.114	-0.120
%Ag + ADA	0.531	0.652	0.898	-0.902	-0.976	0.537	1.000	0.968	0.166	-0.585	-0.619	-0.778	-0.697	-0.034	-0.022	-0.151
%Ag + LDD	0.516	0.661	0.977	-0.880	-0.925	0.315	0.968	1.000	-0.023	-0.579	-0.621	-0.767	-0.694	-0.013	0.008	-0.136
Rd Den	0.024	0.008	-0.183	-0.241	-0.207	0.726	0.166	-0.023	1.000	-0.040	-0.076	-0.132	-0.136	-0.100	-0.121	-0.101
Site Slope	-0.394	-0.632	-0.546	0.486	0.566	-0.269	-0.585	-0.579	-0.040	1.000	0.744	0.546	0.711	-0.421	-0.433	-0.269
Site Elev	-0.574	-0.754	-0.590	0.504	0.567	-0.259	-0.619	-0.621	-0.076	0.744	1.000	0.630	0.952	-0.329	-0.352	-0.113
Mean Slope	-0.467	-0.655	-0.725	0.659	0.715	-0.368	-0.778	-0.767	-0.132	0.546	0.630	1.000	0.740	0.035	0.017	0.170
Mean Elev	-0.618	-0.793	-0.657	0.583	0.625	-0.316	-0.697	-0.694	-0.136	0.711	0.952	0.740	1.000	-0.094	-0.114	0.043
Wshd Area	-0.029	0.027	0.007	0.077	0.054	-0.090	-0.034	-0.013	-0.100	-0.421	-0.329	0.035	-0.094	1.000	0.981	0.926
Strm Len	-0.008	0.037	0.034	0.057	0.048	-0.114	-0.022	0.008	-0.121	-0.433	-0.352	0.017	-0.114	0.981	1.000	0.925
Mean Q	-0.100	-0.098	-0.116	0.195	0.165	-0.120	-0.151	-0.136	-0.101	-0.269	-0.113	0.170	0.043	0.926	0.925	1.000

Table 3.2. Correlation Matrix for Western Oregon. Ag = Agriculture, For = Forest, Nat Veg = Natural Vegetation, ADA = All Developed Areas, LDD = Low-Density Development, Rd Den = Road Density (m roads/m2 watershed area), Elev = Elevation (m), Pop Den = Population Density (# people/km2), Whsd Area = Watershed Area (ha), Strm Len = Length (m) of all stream in watershed upstream of site, Mean Q = estimated mean annual discharge (cfs). Shaded areas indicate correlation coefficients between variables and nitrate/chloride that are >0.2.

Variable	$\log(NO_3^-)$	$\log(Cl^-)$	%Ag	%Rng	%For	%Barren	%Wetlands	%Urban	%NRU	%Ag + ADA	Rd Den	Site Elev	Mean Slope	Mean Elev	Pop Den	Wshd Area
$\log(NO_3^-)$	1.000	0.298	0.190	-0.042	0.011	0.079	-0.291	0.141	0.020	0.222	0.005	-0.329	0.118	-0.302	0.180	-0.162
$\log(Cl^-)$	0.298	1.000	0.258	0.024	0.017	-0.164	-0.299	0.102	0.036	0.263	-0.058	-0.626	-0.014	-0.547	0.217	0.165
%Ag	0.190	0.258	1.000	-0.005	-0.018	-0.093	-0.055	0.122	0.642	0.903	-0.144	-0.299	0.259	-0.344	0.620	-0.043
%Rng	-0.042	0.024	-0.005	1.000	-0.962	0.119	-0.197	0.002	-0.066	-0.004	-0.131	0.027	-0.029	0.138	0.021	0.134
%For	0.011	0.017	-0.018	-0.962	1.000	-0.378	0.139	-0.031	0.024	-0.029	0.215	-0.099	0.029	-0.218	-0.051	-0.129
%Barren	0.079	-0.164	-0.093	0.119	-0.378	1.000	0.084	0.009	0.076	-0.075	-0.317	0.282	0.068	0.347	-0.034	0.018
%Wetlands	-0.291	-0.299	-0.055	-0.197	0.139	0.084	1.000	-0.048	-0.008	-0.067	-0.135	0.421	-0.202	0.341	-0.066	-0.093
%Urban	0.141	0.102	0.122	0.002	-0.031	0.009	-0.048	1.000	0.642	0.537	0.099	-0.099	-0.163	-0.058	0.605	0.122
%NRU	0.020	0.036	0.642	-0.066	0.024	0.076	-0.008	0.642	1.000	0.333	0.164	-0.034	-0.174	0.031	0.294	0.234
%Ag + ADA	0.222	0.263	0.903	-0.004	-0.029	-0.075	-0.067	0.537	0.333	1.000	-0.079	-0.297	-0.297	-0.317	0.789	0.016
Rd Den	0.005	-0.058	-0.144	-0.131	0.215	-0.317	-0.135	0.099	0.164	-0.079	1.000	-0.042	-0.154	-0.118	-0.058	-0.023
Site Elev	-0.329	-0.626	-0.299	0.027	-0.099	0.282	0.421	-0.099	-0.034	-0.297	-0.042	1.000	-0.151	0.910	-0.259	-0.137
Mean Slope	0.118	-0.014	0.259	-0.029	0.029	0.068	-0.202	-0.163	-0.174	-0.297	-0.154	-0.151	1.000	0.035	-0.354	-0.137
Mean Elev	-0.302	-0.547	-0.344	0.138	-0.218	0.347	0.341	-0.058	0.031	-0.317	-0.118	0.910	0.035	1.000	-0.309	0.092
Pop Den	0.180	0.217	0.620	0.021	-0.051	-0.034	-0.066	0.605	0.294	0.789	-0.058	-0.259	-0.354	-0.309	1.000	-0.009
Wshd Area	-0.162	0.165	-0.043	0.134	-0.129	0.018	-0.093	0.122	0.234	0.016	-0.023	-0.137	-0.137	0.092	-0.009	1.000

Table 3.3. Correlation Matrix for Northern California. Ag = Agriculture, Rng = Rangeland, For = Forest, Nat Veg = Natural Vegetation, NRU = Non-residential Urban, ADA = All Developed Areas, Rd Den = Road Density (m roads/m2 watershed area), Elev = Elevation (m), Pop Den = Population Density (# people/km2), Whsd Area = Watershed Area (ha). Shaded areas indicate correlation coefficients between variables and nitrate/chloride that are >0.2.

Separate models for the Northern California and Western Oregon datasets were created to compare model coefficients and model performance. The significance of the addition of TI to the models was evaluated using the partial-F test. Models from each region were compared using Akaike's Information Criterion (AIC). Akaike's Information Criterion is a measure of how well a model fits the data, with an added penalty for the complexity of the model, that is, the number of estimated model coefficients (Burnham and Anderson, 1998). Thus, AIC helps identify small (parsimonious) models that fit the data better than other models. The model that fits the data best is indicated by the smallest value of AIC. Differences in AIC (ΔAIC), which is the AIC of the model subtracted from the minimum AIC among all candidate models, were used to compare models. Models with a $\Delta AIC < 2$ are considered good and models with a $\Delta AIC < 7$ are considered okay compared to the best model (Burnham and Anderson, 1998).

Due to the relatively small datasets, multiple cross validation was conducted. Cross validation, specifically multiple cross validation, involves using a subset of the whole dataset to determine the stability of model performance (Mosteller and Tukey, 1977). Five percent of the dataset was randomly removed 10 times, and changes in the regression were noted. This process is more appropriate for small datasets and models with a large number of variables than setting aside a portion of the data for validation, since the model will have more predictive power if a larger dataset is used for calibration.

Results

Chloride Modeling. Tables 3.4 and 3.5 show chloride modeling results for the Western Oregon and Northern California datasets. Both the agricultural and forested variables help model predictions, but because of the high collinearity between the agricultural and forested variables (on average, $r = -0.900$ for area, 1/d, and $1/d^2$ variables) it is difficult to interpret the resulting coefficients. Thus, models using only the forested variables are shown and discussed. Using land use proportions, variability in chloride concentrations were explained better for Western Oregon compared to the Northern California dataset (R^2 of 0.41 and 0.07 for Western Oregon and Northern California, respectively). The partial-F test ($\alpha = 0.05$) showed that adding TI significantly improved results in both regions (R^2 of 0.45 and 0.14 for Western Oregon and Northern California, respectively). Site elevation also significantly improved ($\alpha = 0.05$) prediction of chloride (R^2 of 0.61 and 0.40 for Western Oregon and Northern California, respectively). The inverse-distance (1/d) and inverse-distance squared ($1/d^2$) calculations slightly improved model results; on average the ΔAIC values were lower compared to the proportional models. Although improvements were not large, the multiplicative models (In*Out^{-1} and In*Out^{-2}) predicted chloride better than the additive models (In+Out^{-1} and In+Out^{-2}). According to the ΔAIC values, the best model was the In*Out^{-1} Landuse-Elevation model for both Western Oregon and Northern California. The full equation is:

$$\log(NO_3^-) = 2.56 - 0.573(IS\ F_{1/d}) * (OS\ F_{1/d}) - 0.0115(site\ elevation) \qquad (4)$$

For Western Oregon, the R^2 value increased from 0.61 to 0.64 and ΔAIC decreased from 4.6 to 0.0 when comparing the %Landuse-Elevation and In*Out^{-1} Landuse-Elevation models. Improvements for the Northern California dataset were not as large; the R^2 stayed at 0.40 and ΔAIC decreased slightly from 0.2 to 0.0 when comparing the %Landuse-Elevation and In*Out^{-1} Landuse-Elevation models, and the R^2 value increased slightly from 0.14 to 0.15 and ΔAIC decreased from 21.4 to 20.2 when comparing the %Landuse-TI and In*Out^{-1} Landuse-TI models.

Model	Terms (coeff., SE)				R^2	ΔAIC
%Landuse					0.41	34.5
	Forested					
	(-1.20, 0.168)					
%Landuse-TI					0.45	30.9
	Forested,	TI				
	(-1.83, 0.314)	(-0.174, 0.0739)				
%Landuse-Elevation					0.61	4.6
	Forested,	Site Elevation				
	(-0.518, 0.176)	(-0.00108, 0.000170)				
Total^{-1} Landuse-TI					0.51	22.1
	Forested,	TI				
	(-1.87, 0.292)	(-0.177, 0.0723)				
Total^{-1} Landuse-Elevation					0.63	1.7
	Forested,	Site Elevation				
	(-0.607, 0.177)	(-0.000990, 0.000180)				
In+Out^{-1} Landuse-TI					0.54	21.7
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(-1.92, 0.818)	(0.0768, 0.812)	(-0.0141, 0.131)	(-0.186, 0.139)		
In+Out^{-1} Landuse-Elevation					0.63	3.1
	IS Forested,	OS Forested,	Site Elevation			
	(-0.382, 0.566)	(-0.241, 0.576)	(-0.000981, 0.000185)			
In+Out^{-2} Landuse-TI					0.53	24.1
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(-0.440, 0.345)	(-1.31, 0.368)	(0.0477, 0.0455)	(-0.191, 0.0732)		
In+Out^{-2} Landuse-Elevation					0.64	1.2
	IS Forested,	OS Forested,	Site Elevation			
	(-0.0181, 0.249)	(-0.661, 0.281)	(-0.000994, 0.000176)			
In*Out^{-1} Landuse-TI					0.58	11.3
	IS*OS Forested,	IS*OS TI				
	(-1.64, 0.202)	(-0.0115, 0.00354)				
In*Out^{-1} Landuse-Elevation					0.64	0.0
	IS*OS Forested,	Site Elevation				
	(-0.573, 0.154)	(-0.000910, 0.000186)				
In*Out^{-2} Landuse-TI					0.54	17.2
	IS*OS Forested,	IS*OS TI				
	(-1.40, 0.185)	(-0.00692, 0.00309)				
In*Out^{-2} Landuse-Elevation					0.64	0.4
	IS*OS Forested,	Site Elevation				
	(-0.555, 0.152)	(-0.000919, 0.000186)				

Table 3.4. Chloride Modeling Results for Western Oregon using Land Use, TI, and Site Elevation.

Model	Terms (coeff., SE)				R^2	ΔAIC
%Landuse					0.07	23.9
	Forested					
	(-25.3, 12.2)					
%Landuse-TI					0.14	21.4
	Forested,	TI				
	(-29.0, 12.0)	(-0.36, 0.17)				
%Landuse-Elevation					0.40	0.2
	Forested,	Site Elevation				
	(-8.29, 10.3)	(-0.000861, 0.000156)				
Total^{-1} Landuse-TI					0.13	21.9
	Forested,	TI				
	(-6.19, 2.65)	(-0.300, 0.145)				
Total^{-1} Landuse-Elevation					0.40	0.2
	Forested,	Site Elevation				
	(-1.80, 2.27)	(-0.000864, 0.000155)				
Total^{-2} Landuse-Elevation					0.39	0.4
	Forested,	Site Elevation				
	(-0.380, 0.625)	(-0.000874, 0.000154)				
In+Out^{-1} Landuse-TI					0.18	22.6
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(-1.86, 3.65)	(-8.40, 16.2)	(0.136, 0.258)	(-0.398, 0.201)		
In+Out^{-1} Landuse-Elevation					0.40	2.0
	IS Forested,	OS Forested,	Site Elevation			
	(-1.70, 3.08)	(1.52, 13.5)	(-0.000862, 0.000156)			
In+Out^{-2} Landuse-TI					0.18	22.1
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(-0.405, 1.99)	(-10.6, 16.3)	(-0.0214, 0.0790)	(-0.241, 0.0946)		
In+Out^{-2} Landuse-Elevation					0.40	2.3
	IS Forested,	OS Forested,	Site Elevation			
	(-0.371, 1.69)	(-1.09, 13.9)	(-0.000869, 0.000156)			
In*Out^{-1} Landuse-TI					0.15	20.2
	IS*OS Forested,	IS*OS TI				
	(-3.67, 1.55)	(-0.0250, 0.0105)				
In*Out^{-1} Landuse-Elevation					0.40	0.0
	IS*OS Forested,	Site Elevation				
	(-1.21, 1.35)	(-0.000860, 0.000155)				
In*Out^{-2} Landuse-TI					0.14	21.1
	IS*OS Forested,	IS*OS TI				
	(-1.66, 0.784)	(-0.0161, 0.00681)				
In*Out^{-2} Landuse-Elevation					0.40	0.3
	IS*OS Forested,	Site Elevation				
	(-0.481, 0.680)	(-0.000869, 0.000155)				

Table 3.5. Chloride Modeling Results for Northern California using Land Use, TI, and Site Elevation.

With the exception of the In+Out^{-1} Landuse-TI model, an increase in forested land use, site elevation, and TI decreased chloride concentrations in Western Oregon. Site elevation was consistently a better predictor of chloride than TI (i.e., %Landuse-TI had an R^2 of 0.45 and ΔAIC of 30.9 compared to %Landuse-Elevation which had an R^2 of 0.61 and ΔAIC of 4.6). For Northern California, the ΔAIC values were much higher for the land use and TI models (20.2-22.6) compared to the land use and elevation models (0.0-2.3). Similar to the Western Oregon models, an increase in site elevation, forested land use, and TI decreased chloride concentrations. Exceptions include the positive relationships between chloride and in-stream inverse-distance TI in the In+Out^{-1} Landuse-TI model and chloride and out-of-stream inverse-distance forested land use in the In+Out^{-1} Landuse-Elevation model.

From these results, it appears that TI and site elevation are strong variables for predicting chloride in Western Oregon. Site elevation was also a strong predictor of chloride for the Northern California dataset. As site elevation increases, chloride concentrations gradually decrease (Figure 3.3). Site elevation and mean elevation were found to produce very similar modeling results, because they are so closely related (r = 0.952 and r = 0.910 for the Western Oregon and Northern California datasets, respectively). In this study, site elevation produced better model results, but either site elevation or mean elevation could be used in model simulations. Slope also significantly improved Western Oregon chloride models ($\alpha = 0.05$) but not as much as elevation (R^2 of 0.51 compared to 0.61 for the %Landuse-Slope and %Landuse-Elevation models, respectively). Since the correlation between chloride and slope is relatively weak for the Northern California dataset (r = -0.014), models using slope were not created. Only the results from models using site elevation are shown and discussed.

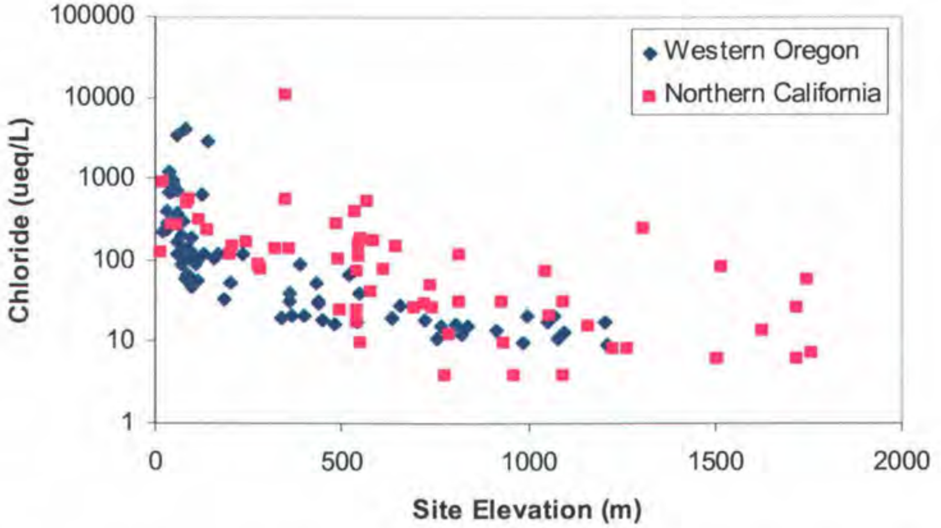

Figure 3.3. Variation in Chloride Concentrations with Site Elevation for Western Oregon and Northern California.

Nitrate Modeling. Tables 3.6 and 3.7 show nitrate modeling results for the Northern California and Western Oregon datasets. Using land use proportions, nitrate concentrations were predicted better for the Western Oregon dataset compared to the Northern California dataset (R^2 = 0.28 and 0.05 for Western Oregon and Northern California, respectively). Adding TI did not significantly (α = 0.05) improve nitrate predictions for Western Oregon. For the Northern California dataset, the TI term did not significantly improve (α = 0.05) the %Landuse model, but did significantly improve the inverse-distance and inverse-distance squared models. Accounting for inverse-distance or inverse-distance squared did not improve overall model results for the Western Oregon dataset; the best model was the %Landuse-Elevation model according to the ΔAIC. A slight improvement was observed when just looking at forested land use and TI as independent variables though; nitrate predictions improved from $R^2 = 0.29$ and ΔAIC = 10.6 using area proportions to $R^2 = 0.33$ and ΔAIC = 10.2 using in-stream and out-of-stream inverse-distance squared ($1/d^2$). Except for the In*Out^{-1} Landuse-Elevation model, the additive models predicted nitrate for Western Oregon better than the multiplicative models. In contrast, the multiplicative models predicted nitrate for Northern California better than the additive models.

Model	Terms (coeff., SE)				R^2	ΔAIC
%Landuse					0.28	9.0
	Forested					
	(-2.44, 0.449)					
%Landuse-TI					0.29	10.6
	Forested,	TI				
	(-2879, 0.873)	(-0.122, 0.205)				
%Landuse-Elevation					0.38	0.0
	Forested,	Site Elevation				
	(-1.33, 0.535)	(-0.00179, 0.000530)				
$Total^{-1}$ Landuse-TI					0.28	11.8
	Forested,	TI				
	(-2.80, 0.865)	(-0.134, 0.215)				
$Total^{-1}$ Landuse-Elevation					0.37	1.8
	Forested,	Site Elevation				
	(-1.14, 0.552)	(-0.00184, 0.000560)				
$In+Out^{-1}$ Landuse-TI					0.33	10.2
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(1.51, 2.40)	(-4.56, 2.39)	(0.710, 0.386)	(-0.963, 0.409)		
$In+Out^{-1}$ Landuse-Elevation					0.38	2.3
	IS Forested,	OS Forested,	Site Elevation			
	(1.33, 1.78)	(-2.56, 1.82)	(-0.00200, 0.000574)			
$In+Out^{-2}$ Landuse-TI					0.31	12.8
	IS Forested,	OS Forested,	IS TI,	OS TI		
	(-0.0385, 1.02)	(-3.50, 1.09)	(0.0980, 0.134)	(-0.483, 0.216)		
$In+Out^{-2}$ Landuse-Elevation					0.38	2.2
	IS Forested,	OS Forested,	Site Elevation			
	(0.960, 0.795)	(-2.03, 0.899)	(-0.00210, 0.000554)			
$In*Out^{-1}$ Landuse-TI					0.29	10.1
	IS*OS Forested,	IS*OS TI				
	(-2.47, 0.639)	(-0.00997, 0.0112)				
$In*Out^{-1}$ Landuse-Elevation					0.36	2.0
	IS*OS Forested,	Site Elevation				
	(-0.987, 0.487)	(-0.00178, 0.000589)				
$In*Out^{-2}$ Landuse-TI					0.28	11.8
	IS*OS Forested,	IS*OS TI				
	(-2.62, 0.566)	(-0.0176, 0.00947)				
$In*Out^{-2}$ Landuse-Elevation					0.35	4.1
	IS*OS Forested,	Site Elevation				
	(-0.677, 0.485)	(-0.00203, 0.000596)				

Table 3.6. Nitrate Modeling Results for Western Oregon using Land Use, TI, and Site Elevation.

Model	Terms (coeff., SE)				R^2	ΔAIC
%Landuse					0.05	2.9
	Forested (-43.4, 25.2)					
%Landuse-TI					0.09	2.6
	Forested, (-49.0, 25.2)	TI (-0.547, 0.364)				
%Landuse-Elevation					0.12	0.0
	Forested, (-26.7, 25.5)	Site Elevation (-0.000844, 0.000384)				
In+Out^{-1} Landuse-TI					0.18	0.4
	IS Forested, (1.14, 7.46)	OS Forested, (-47.7, 33.1)	IS TI, (-1.41, 0.528)	OS TI (0.577, 0.411)		
In+Out^{-1} Landuse-Elevation					0.12	2.1
	IS Forested, (1.51, 7.62)	OS Forested, (-22.5, 33.4)	Site Elevation (-0.000859, 0.000386)			
In+Out^{-2} Landuse-TI					0.16	1.9
	IS Forested, (0.233, 4.14))	OS Forested, (-29.6, 33.9)	IS TI, (-0.413, 0.164)	OS TI (0.0152, 0.197)		
In+Out^{-2} Landuse-Elevation					0.12	2.1
	IS Forested, (1.24, 4.16)	OS Forested, (-22.9, 34.3)	Site Elevation (-0.000869, 0.000385)			
In*Out^{-1} Landuse-Elevation					0.12	0.4
	IS*OS Forested (-2.74, 3.35)	Site Elevation (-0.000876, 0.000383)				
In*Out^{-2} Landuse-TI					0.12	0.3
	IS*OS Forested (-2.64, 1.62)	IS*OS TI (-0.0334, 0.0141)				
In*Out^{-2} Landuse-Elevation					0.12	0.5
	IS*OS Forested (-1.27, 1.68)	Site Elevation (-0.000886, 0.000383)				

Table 3.7. Nitrate Modeling Results for Northern California using Land Use, TI, and Site Elevation.

Site elevation was consistently a better predictor of nitrate than TI (i.e., for the Western Oregon dataset, %Landuse-TI had an R^2 of 0.29 and ΔAIC of 10.6 compared to %Landuse-Elevation which had an R^2 of 0.38 and ΔAIC of 0.0). Site elevation significantly (α = 0.05) improved nitrate models for both the Western Oregon and Northern California datasets. Slope did not significantly improve nitrate models (α = 0.05) for the Western Oregon dataset. Slope was not used in the Northern California models due to the weak correlation between nitrate and slope (r = 0.118). Slope models are not shown since elevation was a better predictor and thus better for comparison with TI.

Regardless of any statistical significance, models produced using the Northern California dataset are basically useless because of their low R^2 values. The best model was the %Landuse-Elevation model according to the ΔAIC value (R^2 value of 0.12), although the other models using TI and elevation had ΔAIC values very close to zero and R^2 values are about the same or slightly better. The R^2 values of models produced using the Western Oregon dataset were higher, with the best model having an R^2 value of 0.38 (%Landuse-Elevation model).

In all of the models for both regions, an increase in forested land use, TI, and site elevation decreased nitrate concentrations, and an increase in in-stream inverse-distance and inverse-distance squared forested land use increased nitrate concentrations. Exceptions include the positive relationship between nitrate and in-stream inverse-distance and inverse-distance squared TI for Western Oregon, and between nitrate out-of-stream inverse-distance and inverse-distance squared TI for Northern California.

Validation. Model validation was conducted for all of the nitrate models developed for Western Oregon and Northern California using multiple cross validation. Models were considered validated if the signs of the coefficients stayed the same and coefficients did not vary by more than 100%. With the exception of the forested (in-stream distance)$^{-2}$ coefficient in the In+Out^{-2} Landuse-TI model, the signs of all of the coefficients (positive or negative) stayed the same for the Western

Oregon models. The forested (in-stream distance)$^{-2}$ coefficient, which had a high standard error, was positive when using all of the data and shifted to negative when using a subset of the data. The coefficient also varied 294% during validation. The TI (out-of-stream distance)$^{-2}$ coefficient varied by 157% in the same model, indicating that this model is not stable. Coefficients in all of the other models varied by less than 74%. The signs of all of the coefficients stayed the same for the Northern California models, except for the forested (in-stream distance)$^{-2}$ and the TI (out-of-stream distance)$^{-2}$ coefficients in the In+Out^{-2} Landuse-TI model, and the forested (in-stream distance)$^{-1}$ coefficient in the In+Out^{-1} Landuse-TI model. The forested (in-stream distance)$^{-2}$ and TI (out-of-stream distance)$^{-2}$ coefficients in the In+Out^{-2} Landuse-TI model varied by 979% and 1444%, respectively, and the forested (in-stream distance)$^{-1}$ coefficient in the In+Out^{-1} varied by 734%. The forested (in-stream distance)$^{-1}$*(out-of-stream distance)$^{-1}$ coefficient varied by 687% in the In*Out^{-1} Landuse-Elevation model. Coefficients in all other models varied by less than 68%.

We saw similar validation results for the Western Oregon and Northern California chloride models. The signs of all of the coefficients stayed the same for the Western Oregon models, except for the forested (out-of-stream distance)$^{-1}$ and TI (in-stream distance)$^{-1}$ coefficients in the In+Out^{-1} Landuse-TI model, the forested (out-of-stream distance)$^{-1}$ coefficient in the In+Out^{-1} Landuse-Elevation model, and the forested (in-stream distance)$^{-2}$ coefficient in the In+Out^{-2} Landuse-Elevation model. In these models, the coefficients also varied by more than 100%. Coefficients in all other models varied by less than 50%. The signs of all of the coefficients stayed the same for the Northern California models, except for the forested (in-stream distance)$^{-2}$ and TI (in-stream distance)$^{-2}$ coefficients in the In+Out^{-2} model, and the forested (out-of-stream distance)$^{-1}$ coefficient in the In+Out^{-1} model. Coefficients also varied by more than 100% in these models. In addition, the forested (total distance)$^{-2}$ coefficient in the Total^{-2} Landuse-Elevation model varied by 195%, the TI (in-stream distance)$^{-1}$ coefficient in the In+Out^{-1} model varied by 104%, and the forested (in-stream distance)$^{-2}$ coefficient in the In+Out^{-2} model varied by 489%. The coefficients in all other models varied by less than 80%.

Discussion

This paper presents our development of a statistical model with inverse-distance weighting of land use and an added index (TI) to predict hot spots for nitrate. We did not see an improvement of nitrate predictions using TI and inverse-distance weighting; the best models for both the Western Oregon and Northern California datasets were the %Landuse-Elevation models. We did, however, see an improvement of chloride predictions using TI and inverse-distance weighting; adding TI significantly improved model predictions and the best models for both Western Oregon and Northern California were the In*Out^{-1} Landuse-Elevation models. One interesting result of this study is the significance of elevation for predicting both nitrate and chloride. Elevation was originally used as a proxy for TI, and it is shown in the model results that it consistently predicts nitrate better than TI.

Chloride as tracer of water. As a conservative tracer, chloride is largely affected by the water cycle, which includes rainout/orographic effects, saturation conditions, and varying contributions of streamwater from different sources such as groundwater, soil water, and rainfall (flowpaths). Rainout has been described as the decrease in chloride deposition with distance inland observed in most coastal areas (Van Leeuwen et. al., 1996; Li, 1992; Gustafsson and Larsson, 2000; Hingston and Gailitis, 1976; Carratala et. al., 1998). An increase in chloride deposition has also been observed with altitude (Gustafsson and Larsson, 2000; Fowler et. al., 1988). The general topography of Western Oregon includes the Willamette Valley, which is bordered by a mountain range on the west side (often called the coast range) and the Cascade Range on the east side. Distance from sampling sites to the coast range from 23 to 180 km. Table 3.8 shows rainwater chloride deposition from three sites in Western Oregon, which demonstrates the general decrease of chloride deposition with distance inland and the increase with altitude (data from http://nadp.sws.uiuc.edu/nadpdata). A different trend is apparent in stream chloride concentrations with distance inland (Figure 3.4). For the Western Oregon dataset, chloride and distance from the coast were found to be highly correlated ($r = -0.714$). A general decrease in chloride with distance from the coast occurs, but there is a peak along the valley floor where most of the agricultural activity is concentrated. This is similar to findings from other studies, where elevated chloride concentrations in groundwater and stream water in agricultural areas were observed (Pionke and Urban, 1985; Smart et. al., 1998). No clear trend for the Northern California dataset was observed.

Site	Elevation (m)	Distance from Coast (km)	Average Annual Concentration (mg/L)	Average Annual Wet Deposition (kg/ha)
OR02	104	42.8	1.42	22.64
OR97	69	77.6	0.69	5.97
OR10	436	170	0.33	6.60

Table 3.8. Chloride Rainfall Concentration and Deposition Data from NADP sites within Western Oregon.

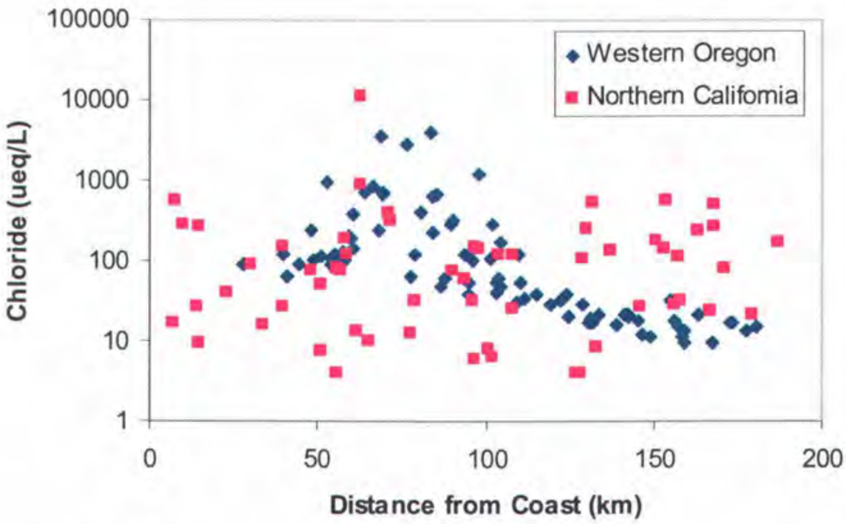

Figure 3.4. Variation in Chloride Concentrations with Distance from the Coast.

To further determine how rainout and orographic mechanisms affect stream chloride concentrations, precipitation and elevation were plotted against distance from the coast (Figures 3.5 and 3.6, respectively). For the Western Oregon dataset, elevation increases over the coast range, decreases over the valley floor, and increases over the Cascade Range. Elevation was found to be highly correlated to both chloride and distance from the coast ($r = -0.754$ and $r = 0.801$, respectively). Chloride was modeled using distance from the coast, and model results were about the same as models using elevation. This same pattern was observed for precipitation; precipitation amounts decreased with distance inland, with a minimum at the valley floor, then increased as it reached the Cascade Range (where elevation increases). The similarities between elevation and depositional patterns is likely a regional phenomenon due to the unique topography of Western Oregon. The difference between stream chloride concentrations and chloride deposition/precipitation trends may be due to the agricultural land use in Western Oregon.

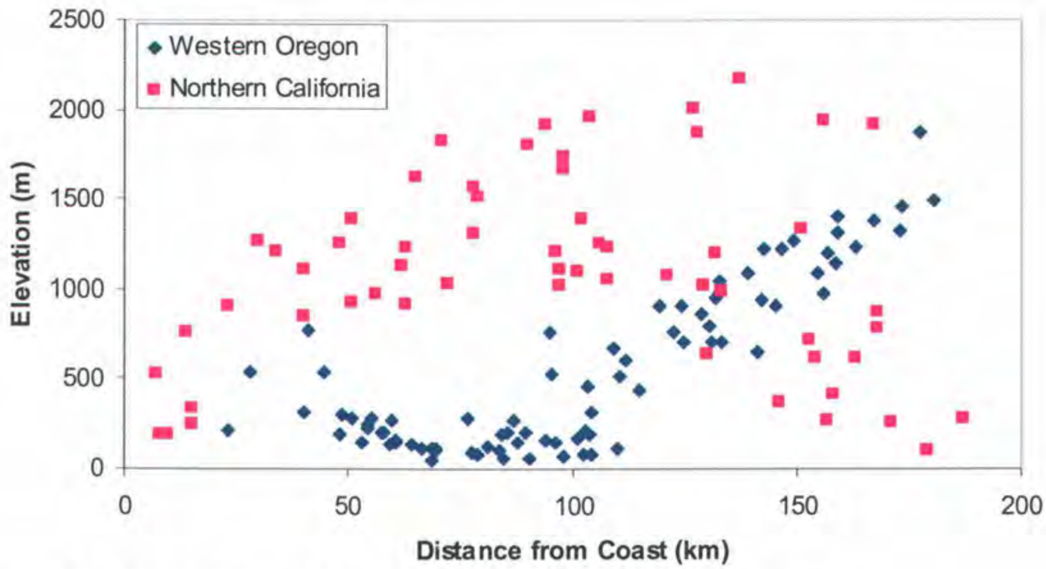

Figure 3.5. Variation in Elevation with Distance from the Coast.

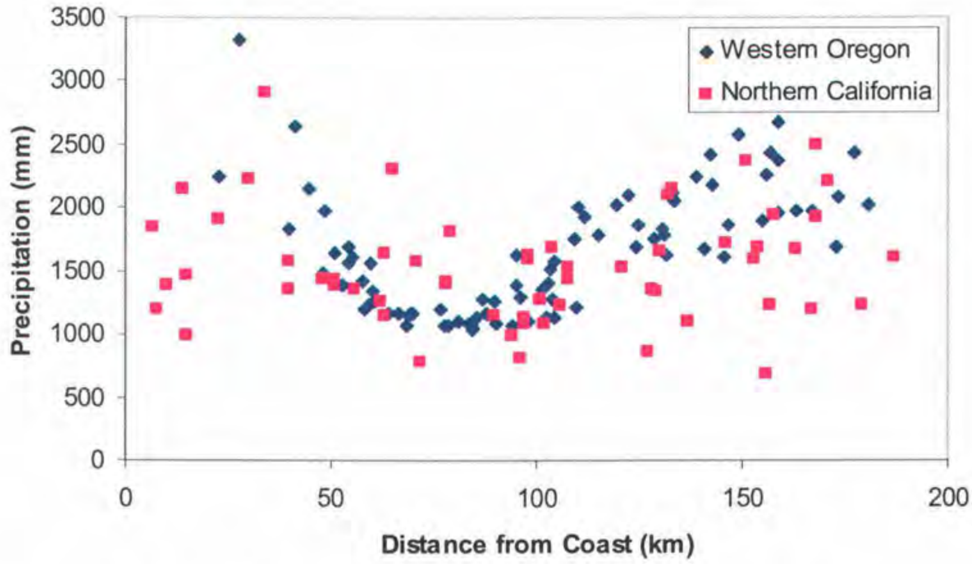

Figure 3.6. Variation in Precipitation with Distance from the Coast.

This impact of agricultural land use on chloride concentrations may render chloride an inadequate tracer of water. Land use has been found to affect chloride concentrations in other studies (Herlihy et. al., 1998; Smart et. al., 1998). Chloride concentrations in groundwater beneath cropland have been found to be 5-7 times higher than in forested areas (Pionke and Urban, 1985), and watersheds dominated by agriculture have produced elevated concentrations of chloride (Smart et. al., 1998). Nonetheless, the significance of TI and inverse-distance weighting in the chloride models indicates that some of the catchment processes may be represented (water saturation or hotspots in terms of TI and flowpaths in terms of inverse-distance).

Landuse. Relationships found in this study are similar to those found in other studies using percent area of land use, with a negative correlation between forested area and nitrate (Norton and Fisher, 2000; King et. al., 2005; Arheimer and Liden, 2000; Herlihy et. al., 1998). Reasons for the relatively poor nitrate model performance in this study could be the low flow sampling conditions or the lack of variability in land use. Mixed results have been shown on the relative importance of wet vs. dry sampling conditions. In contrast to Heisig (2000) who saw a significant relationship between nitrate concentrations and land use during low flow conditions, Johnson et. al. (1997) saw no relationship between nitrate and land use during dry conditions but were able to produce a significant nitrate model for wet weather data using %agriculture. It appears that watersheds and rivers are still hydrologically connected during dry conditions in some areas and not in others. Unless sampling is conducted during wet conditions in Western Oregon and Northern California, it is unknown whether or not a more significant model can be produced if wet season data is used.

Poor model results could also be due to the lack of variability in land use. Studies have found that models created by watersheds dominated by forest land cover do not predict nitrate concentrations as well as watersheds in lowland areas where agriculture is more common and land cover is more diverse (Herlihy et. al., 1998). This is similar to the model results from the Western Oregon and Northern California regions; the dataset from Northern California, which is dominated by forest land cover, did not predict nitrate concentrations as well as the dataset from Western Oregon. To determine whether or not we could improve model results for Western Oregon, the dataset was split into "valley" and "upland" datasets. The resulting models had a poorer performance than the models using the entire dataset (best R^2 of 0.18). Poor model performance when modeling valley and upland sites separately may be due to the narrow range of site elevations (21-170 m and 109-1213 m for the valley and upland sites, respectively). Since site elevation explains the most variability in nitrate concentrations, a better model using site elevation was produced when using the entire dataset.

Distance Relationship. Accounting for distance (using in-stream and out-of-stream inverse-distance or inverse-distance squared) in the model did not significantly improve nitrate predictions. The %Landuse-Elevation model was the best model for both Western Oregon and Northern California according the ΔAIC value. The reason for the similarity between models using %landuse and inverse distance weighting is evident when %landuse is plotted against 1/d and $1/d^2$ (Figure 3.7). Normalized values of %forested area and 1/d are very similar. The fitted trendline shows an approximate 1:1 relationship with %forested area and 1/d for Western Oregon, and values are closely related in both region (R^2 of 0.85 and 0.96 for Northern California and Western Oregon, respectively). Values for %forested area and $1/d^2$ are not as similar, but are close with an R^2 value of 0.76 for Northern California and 0.57 for Western Oregon. Although inverse-distance weighting did improve chloride models, the improvement is relatively small (R^2 of 0.61 to 0.64 and ΔAIC of 4.6 to 0.0 for the %Landuse-Elevation and $In*Out^{-1}$ Landuse-Elevation models, respectively) and comparable to the small difference between %landuse and 1/d. Since chloride is strongly affected by flowpaths, the slight difference between %landuse and 1/d is enough to make a difference in the models. In contrast, nitrate is controlled by processes other than the hydrology of the catchment (i.e., denitrification) which likely overwhelms any improvement using inverse-distance weighting.

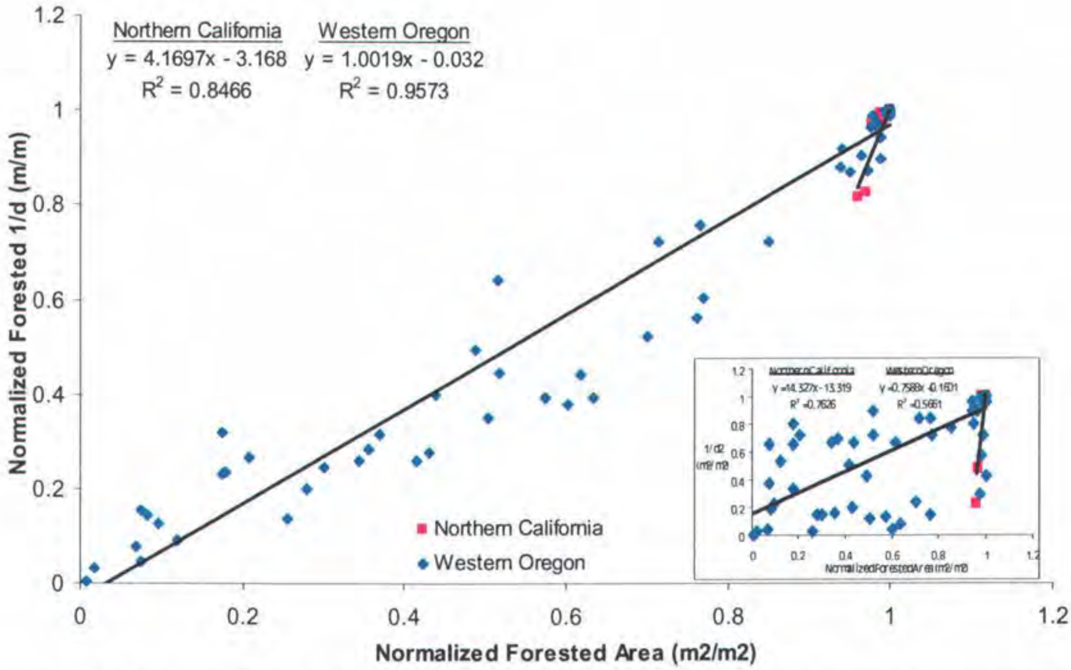

Figure 3.7. Comparison of Forested Area and Distance for Western Oregon and Northern California.

The similarity between %landuse and inverse distance weighting may be due to the catchments used; modeling literature shows mixed results when accounting for distance. Some studies have shown an improvement in predictions when accounting for spatial patterns (Soranno et. al., 1996; Kehmeier, 2000), whereas other studies have shown that spatial pattern did not greatly influence model results (Hunsacker and Levine, 1995; King et. al., 2005; Canham et. al., 2004; Norton and Fisher, 2000). Biota, TN, TP, and/or DOC were modeled in these studies. One can argue that these models are not comparable to a nitrate model since different processes affect biota, TN, TP, DOC, and nitrate. Biota is indirectly affected by land cover and catchment processes, and TN, TP, and DOC are more likely to be sorbed to the soil or incorporated into organic matter than nitrate. For nitrate, inverse distance weighting improved predictions in small watersheds but not medium or large watersheds in one study (King et. al., 2005). In our study, the Western Oregon watersheds were grouped into small (<500 ha), medium (500-3000 ha), and large (>3000 ha) watersheds, but no difference was found in predictions for the various size classes. It appears that the effect of distance weighting depends heavily on the catchment setting (i.e., a significant difference between %landuse and 1/d or $1/d^2$).

Topographic Index. Lack of significance of TI in the model may have been due to the period that focused on dry sampling conditions (September). Using just the Western Oregon September samples, the %Landuse-TI nitrate model had a much lower R^2 (0.14 for September samples compared to 0.29 using all of the data). Summer low flows (especially in September after a 3-month rainless period) are likely from a deep groundwater source that is largely unaffected by land use or topography. Studies have shown that during summer low flows in Western Oregon, the dominant source of streamwater is from groundwater sources (see results from Chapter 2; Hinkle et. al., 2001; Wondzell and Swanson, 1999). Streamwater during summer low flows in Northern California is also from groundwater (Rademacher et. al, 2005; Ahearn et. al., 2004). During wet conditions, storm runoff through the soil and on the surface will also contribute to streamwater. Interflow through the soil zone will likely be more affected by surface nitrate inputs (i.e., fertilizer application), whereas groundwater will be more affected by subsurface inputs (i.e., leaky sewers, septic tanks, etc.).

In addition, temporal changes in discharge have been found to be a major factor affecting the rate and extent of biogeochemical reactions such as denitrification (Dahm et. al., 1998). Although denitrification and other transformation processes can occur in groundwater aquifers, results suggest that nitrate in the groundwater undergoes relatively little transformation (in hot spots) as it travels to the stream. Topographic index is evidently not adequately predicting the hot spot at the interface between groundwater and riparian water or stream water. Hot spots may only occur when water is flowing through previously unsaturated areas and saturated areas are formed within the catchment. During the dry summer months when sampling was conducted, shallow flowpaths disappear. We hypothesize that TI may have better predictive power with winter or storm samples when shallow flowpaths form, saturated areas are formed, and topography plays a stronger role in water movement. More work is needed to determine whether or not TI can improve nitrate predictions during wet conditions.

In our study, elevation, slope, and watershed area were substituted for TI variables in all models to determine whether TI is a strong explanatory variable. These simple measures are possible proxies for TI. Elevation, which significantly improved nitrate and chloride models in both the Western Oregon and Northern California regions, predicted nitrate and chloride concentrations as well as or better than TI (R^2 of 0.38 compared to 0.29 for nitrate). Slope significantly improved chloride models, but not nitrate models for Western Oregon. This shows that for these datasets, TI is not a strong explanatory variable.

Regional Differences. Nitrate model performance using the Western Oregon dataset was better than nitrate model performance using the Northern California dataset. This may be due to the differences in land cover of the two regions; Northern California is mostly forested, with its watersheds having an average of 99.79% forest cover. Northern California also has a lower road density and higher elevation, which is typically found in areas with a higher proportion of forest land cover. In contrast, watersheds in Western Oregon have an average of 73.32% forest, 21.17% agriculture, and 5.52% urban land cover. Herlihy et. al. (1998) found that areas with predominantly forested land cover had weaker relationships than areas with more agriculture and/or urban land cover. The lack of a strong gradient in Northern California likely created poorer models than those created for Western Oregon.

Prediction of chloride for the Western Oregon dataset was much more successful than predictions for the Northern California dataset as well (best R^2 of 0.64 compared to 0.40, respectively). Different elevation trends occur for the two regions with distance inland (Figure 3.5); in Western Oregon the same pattern occurs for elevation and precipitation, whereas elevation increases and precipitation decreases with distance inland in Northern California (Figures 3.5 and 3.6). Elevation is likely a good representation of rainout/orographic effects in Western Oregon due to the similarity in precipitation and elevation variations.

Although model performance was significantly different for the two regions, relationships between variables were similar. A negative correlation between the forested variable and nitrate/chloride existed for both Western Oregon and Northern

California models. Forested areas are likely acting as a nitrate sink (as indicated by the negative correlation between forested land use and nitrate). In addition, the site elevation coefficient for the two regions was similar (-0.0008 to -0.0009 and -0.002 in the nitrate models and -0.0009 and -0.0009 to -0.001 in the chloride models for Northern California and Western Oregon, respectively). The negative correlation between forest land cover and nitrate/chloride and the relationship between site elevation and nitrate/chloride evidently do not vary between regions.

Nitrate vs. Chloride. The ability to predict chloride concentrations better than nitrate concentrations using land use and site elevation or TI is likely due to rainout/orographic effects and the conservative nature of chloride. The pattern of chloride deposition can be identified in stream chloride concentrations, which is likely the reason for successful predictions using site elevation (distance to the coast and site elevation were highly correlated). Although agriculture in the Willamette Valley affects the chloride pattern, the pattern of decreasing chloride concentrations with increasing distance to the coast is still discernable. Nitrate and distance to the coast were also found to be correlated, although the correlation is very similar to that between nitrate and chloride ($r = -0.474$ and $r = 0.473$, respectively). These correlations indicate that stream nitrate concentrations may be controlled somewhat by chloride deposition.

Improved modeling results for chloride compared to nitrate may be due to the conservative nature of chloride. Chloride, which is largely affected by the hydrology of the catchment, is predicted better with TI than nitrate (R^2 of 0.29 and 0.45 for the

%Landuse-TI nitrate and chloride models, respectively). The chloride models reflect the rainout/orographic effects and the hydrologic setting (saturated areas in the case of TI and flowpaths in the case of inverse-distance weighting) within the catchment that likely affect nitrate concentrations as well. Transformation processes, such as denitrification, plant uptake, nitrogen fixation, and nitrification, also affect nitrate concentrations (Sylvia et. al., 1998). Chloride does not undergo these processes, and is generally controlled by atmospheric deposition, anthropogenic inputs (i.e., fertilizer or irrigation in agricultural areas, sewage input in urban areas), and catchment hydrology. The decrease in ability to predict nitrate concentrations compared to chloride is likely due to these transformation processes affecting nitrate concentrations more than land use and catchment hydrology. An in-stream decay coefficient has been used in other models to account for transformation processes (e.g., Smith et. al., 1997), but we feel that adding a coefficient that needs to be measured in the field or borrowed from the literature would take away from the simplicity of the model.

Conclusions

Our statistical model development incorporates process-based knowledge (using TI) into empirical models, creating a link between statistical modeling and process literature. Model results revealed the following:

1. The identification of hot spots with TI did not significantly improve nitrate predictions. This may be due to the source of streamwater; summer low flows are likely from a deep groundwater source that is largely unaffected by activity

occurring on the surface. Although denitrification and other transformation processes can occur in groundwater aquifers, results suggest that nitrate in the groundwater undergoes relatively little transformation (in hot spots) as it travels to the stream.

2. In-stream, out-of-stream, and total inverse-distance and inverse-distance squared calculations provided no improvement in nitrate predictions. This is likely due to the similarity between %landuse and 1/d and $1/d^2$ landuse. Different catchment settings, where there is a significant difference between %landuse and inverse-distance weighted land use, may improve model predictions.
3. Site elevation was the most significant predictor of nitrate and chloride concentrations. In Western Oregon, chloride was successfully predicted using land use and site elevation. Site elevation is likely representing regional rainout/orographic effects due to the similarities between precipitation and elevation variations, which is more strongly linked to chloride than nitrate. Correlations between nitrate/chloride and nitrate/distance to the coast indicate that stream nitrate concentrations may be controlled somewhat by chloride deposition. More work needs to be done to determine the mechanism involved.
4. Stream chloride concentrations were elevated in agricultural areas in Western Oregon, which differed from the observed rainout/orographic pattern of atmospheric chloride deposition. This may render chloride an inadequate

tracer of water. Nonetheless, the use of TI and inverse-distance weighting significantly improved chloride models, indicating that the hydrologic setting of the catchment (identification of saturated areas in the case of TI and flowpaths in the case of inverse-distance weighting) is represented. Nitrate models were not significantly improved using TI and inverse-distance weighting, which indicates that transformation processes may be a more significant control of nitrate.

5. The dominance of one explanatory variable (forested land use in this case) can significantly affect model performance. The models created from the Northern California dataset, which is dominated by forest land cover (average 99.79%), did not predict nitrate as well as the models created from the Western Oregon dataset.

Future work is needed to determine whether or not distance calculations (specifically in-stream, out-of-stream, and total inverse-distance and inverse-distance squared) and TI can improve the prediction of stream nitrate concentrations. The type of catchment setting that provides a significant difference between %landuse and 1/d or $1/d^2$ land use needs to be identified to further investigate the utility of inverse-distance weighting. This study has shown that nitrate traveling in the deeper groundwater (the source of streamwater during lowflow) is not significantly affected by the identification of hot spots using TI. Hot spots at the interface between the groundwater and riparian water near the stream evidently do not significantly transform groundwater nitrate before it reaches the stream, or groundwater may bypass the riparian area altogether (so there is no interface for a hot spot). The saturated areas (and hot spots) that are formed from shallow flowpaths during storm events may be better identified by TI than the hot spots at the interface between groundwater and riparian water. Alternatively, TI may not be adequately identifying these hot spots. Topographic index does aid in the prediction of chloride, which indicates that TI is identifying saturation conditions within the catchment. More work, possibly with a different index and/or using TI to predict nitrate using storm samples, is needed.

Acknowledgements

We thank Dan Heggem, Jim Wigington, and John Van Sickle of the Environmental Protection Agency for chemistry and GIS data, and Alan Herlihy, Jim Wigington, and Pete Nelson for useful discussions. Many thanks to John Van Sickle for reviewing an early version of this manuscript and providing valuable editorial advice. This research was funded by the Environmental Protection Agency cooperative program.

References

Ahearn, D.S., Sheibley, R.W., Dahlgren, R.A., and Keller, K.E., 2004. Temporal dynamics of stream water chemistry in the last free-flowing river draining the western Sierra Nevada, California. Journal of Hydrology 295, 47-63.

Arheimer, B. and Liden, R., 2000. Nitrogen and phosphorous concentrations from agricultural catchments – influence of spatial and temporal variables. Journal of Hydrology 227, 140-159.

Baker, J.P., Landers, D.H., Lee, H., Ringold, P.L., Sumner, R.R., Wigington, P.J., Bennett, R.S., Preston, E.M., Frick, W.E., Sigleo, A.C., Specht, D.T., and Young, D.R., 1995. Ecosystem management research in the Pacific Northwest: five-year research strategy. EPA/600/R-95/069, U.S. EPA, Corvallis, Oregon, 251 pp.

Baker, M.E., Wiley, M.J., and Seelbach, P.W., 2001. GIS-based hydrologic modeling of riparian areas: implications for stream water quality. Journal of the American Water Resources Association 37, 1615-1627.

Beven, K.J. and Kirkby, 1979. A physically based, variable contributing area model of basin hydrology. Hydrological Sciences 24, 43-69.

Burnham, K.P., and Anderson, D.R., 1998. Model selection and inference, a practical information-theoretic approach. Springer-Verlag New York Inc.

Canham, C.D., Pace, M.L., Papaik, M.J., Primack, A.G.B., Roy, K.M., Maranger, R.J., Curran, R.P., and Spada, D.M., 2004. A spatially explicit watershed-scale analysis of dissolved organic carbon in Adirondack Lakes. Ecological Applications 13, 839-854.

Carratala, A., Gomez, A., and Bellot, J., 1998. Mapping rain composition in the East of Spain by applying kriging. Water, Air, and Soil Pollution 104, 9-27.

Christensen, S., Simkins, S., and Teidje, J.M., 1990. Spatial variation of denitrification: dependency of activity centers on the soil environment. Soil Science Society of American Journal 54, 1608-1613.

Cook, P.G., Jolly, I.D., Leaney, F.W., Walker, G.R., Allan, G.L., Fifield, L.K., and Allison, G.B., 1994. Unsaturated zone tritium and chlorine 36 profiles from southern Australia: their use as tracers of soil water movement. Water Resources Research 30, 1709-1719.

Creed, I.F. and Band, L.E., 1998. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research 11, 3105-3120.

Dahm, C.N., Grimm, N.B., Marmonier, P., Valett, H.M., Vervier, P., 1998. Nutrient dynamics at the interface between surface waters and groundwaters. Freshwater Biology 40, 427-451.

Fowler, D., Cape, J.N., Leith, I.D., Choularton, T.W., Gay, M.J., and Jones, A., 1988. The influence of altitude on rainfall composition at Great Dun Fell. Atmospheric Environment 22, 1355-1362.

Gustafsson, M.E.R. and Larsson, E.H., 2000. Spatial and temporal patterns of chloride deposition in Southern Sweden. Water, Air, and Soil Pollution 124, 345-369.

Heisig, P.M., 2000. Effects of residential and agricultural land uses on the chemical quality of baseflow of small streams in the Croton Watershed, Southeastern New York. U.S. Geological Survey Water-Resources Investigations Report 99-4173, 16p.

Herlihy, A., Kaufmann, P., Reynolds, L., Li, J., and Robison, G., 1997. Developing indicators of ecological condition in the Willamette Basin: and overview of the Oregon prepilot study for EPA's EMAP program, *in* River Quality, Dynamics and Restoration. A Laenen and D.A. Dunnette (eds.), Lewis Publishers, CRC Press, Boca Raton, pp. 275-282.

Herlihy, A.T., Stoddard, J.L., and Johnson, C.B., 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic Region, U.S. Water, Air, and Soil Pollution 105, 377-386, 1998.

Hill, A.R., Devito, K.J., Campagnolo, S., and Sanmugades, K., 2000. Subsurface denitrification in a forest riparian zone: interactions between hydrology and supplies of nitrate and organic carbon. Biogeochemistry 51, 193-223.

Hingston, F.J. and Gailitis, V., 1976. The geographic variation of salt precipitated over Western Australia. Australian Journal of Soil Research 14, 319-335.

Hinkle, S.R., Duff, J.H., Triska, F.J., Laenen, A., Gates, E.B., Bencala, K.E., Wentz, D.A., Silva, S.R., 2001. Linking hyporheic flow and nitrogen cycling near the Willamette River – a large river in Oregon, U.S.A. Journal of Hydrology 244, 157-180.

Hjerdt, K.N., McDonnell, J.J., Seibert, J., and Rodhe, A., 2004. A new topographic index to quantify downslope controls on local drainage. Water Resources Research 40, W05602, 1-6.

Hornberger, G.M., Bencala, K.E., and McKnight, D.E., 1994. Hydrological controls on dissolved organic carbon during snowmelt in the Snake River near Montezuma, Colorado. Biogeochemistry 25, 147-165.

Howarth, R.W., Boyer, E.W., Pabich, W.J., and Galloway, J.N., 2002. Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31, 88-96.

Hulse, D.W., Gregory, S.V., and Baker, J.P., 2002. Willamette River basin planning atlas: trajectories of environmental and ecological change. Oregon State University Press, Corvallis, Oregon, USA.

Hunsaker, C.T., and Levine, D.A., 1995. Hierarchical approaches to the study of water quality in rivers. Bioscience 45, 193-203.

Johnson, L.B., Richards, C., Host, G.E., and Arthur, J.W., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37, 193-208.

Jones, K.B., Neale, A.C., Nash, M.S., Van Remortel, R.D., Wickham, J.D., Riitters, K.H., and O'Neill, R.V., 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16, 301-312.

Jordan, T.E., Correll, D.L., Weller, D.E., 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33, 2579-2590.

Kehmeier, J.W., 2000. A spatially explicit method for determining the effects of watershed scale land use on stream conditions. Master's Thesis, Oregon State University.

King, R.S., Baker, M.E., Whigham, D.F., Weller, D.E., Jordan, T.E., Kazyak, P.F., and Hurd, M.K., 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15, 137-153.

Kirchner, J.W., Feng, X., and Neal, C., 2001. Catchment-scale advection and dispersion as a mechanism for fractal scaling in stream tracer concentrations. Journal of Hydrology 254, 82-101.

Li, Y., 1992. Seasalt and pollution inputs over the continental United States. Water, Air, and Soil Pollution 64, 561-573.

Lowrance, R., Todd, R., Fail J. Jr., Hendrickson, O. Jr., Leonard, R., 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34, 374-377.

McClain, M.E., Boyer, M.E., Dent, C.L., Gergel, S.E., Grimm, N.B., Groffman, P.M., Hart, S.C., Harvey, J.W., Johnston, C.A., Mayorga, E., McDowell, W.H., and Pinay, G., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6, 301-312.

McHale, M.R., McDonnell, J.J., Mitchell, M.J., and Cirmo, C.P., 2002. A field-based study of soil water and groundwater nitrate release in an Andirondack forested watershed. Water Resources Research 38, No. 4, 1-17.

Mitchell, M.J., Driscoll, C.T., Inamdar, S., McGee, G.G., Mbila, M.O., and Raynal, D.J., 2003. Nitrogen biogeochemistry in the Adirondack Mountains of New York: hardwood ecosystems and associated surface waters. Environmental Pollution 123, 355-364.

Moser, T.J., Wigington, P.J., Schuft, M.J., Kaufmann, P.R., Herlihy, A.T., Van Sickle, J., and McAllister, L.S., 1997. The effect of riparian areas on the ecological condition of small, perennial stream in agricultural landscapes of Western Oregon. Research Plan, Environmental Protection Agency, EPA/600/R-97/074.

Mosteller, F. and Tukey, J.W., 1997. Data Analysis and Regression, A Second Course in Statistics. Addison-Wesley, Reading, Massachusetts.

Mulder, J., Christophersen, N., Hauhs, M., Vogt, R.D., Andersen, S., and Andersen, D.O., 1990. Water flow paths and hydrochemical controls in the Birkenes catchment as inferred from rainstorm high in seasalts. Water Resources Research 26, 611-622.

Neal, C. and Rosier, P.T.W., 1990. Chemical studies of chloride and stable oxygen isotopes in two conifer afforested and moorland sites in the British Uplands. Journal of Hydrololgy 115, 269-283.

Norton, M.M. and Fisher, T.R., 2000. The effects of forest on stream water quality in two coastal plain watersheds of the Chesapeake Bay. Ecological Engineering 14, 337-362.

Nyberg, L., Rodhe, A., and Bishop, K., 1999. Water transit times and flow paths from two line injections of ^{3}H and ^{36}Cl in a microcatchment at Gardsjon, Sweden. Hydrological Processes 13, 1557-1575.

Oetter, D. R., W. B. Cohen, M. Berterretche, T. K. Maiersperger, and R. E. Kennedy, 2000. Land cover mapping in an agricultural setting using multiseasonal Thematic Mapper data. Remote Sensing of Environment 76, 139–155.

Owens, L.B., Edwards, W.M., and Keuren, R.W., 1991. Baseflow and stormflow transport of nutrients from mixed agricultural watersheds. Journal of Environmental Quality 20, 407-414.

Pan, Y, Herlihy, A., Kaufman, P., Wigington, J., Van Sickle, J., and Moser, T., 2004. Linkages among land-use, water quality, physical habitat conditions and lotic diatom assemblages: a multi-scale assessment. Hydrolobiologica 515, 59-73.

Pater, D.E., Bryce, S.A., Thorson, T.D., Kagen, J., Chappell, C., Omernik, J.M., Azevedo, S.H., and Woods, A.J., 1998. Ecoregions of Western Washington and Oregon (two-sided, color poster with map). United States Geological Survey, Reston, Virginia.

Peck, D.V., Averill, D.K., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, M., McCormick, F.H., Peterson, S.A., Cappaert, M.R., Magee, T., and Monaco, A., 2005a. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for non-wadeable rivers and streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Washington, D.C.

Peck, D.V., Herlihy, A.T., Hill, B.H., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, J.M., McCormick, S.A., Peterson, S.A., Ringold, P.L., Magee, T., and Cappaert, M.R., 2005b. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for wadeable streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Peterjohn, W.T. and Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65, 1466-1475.

Petry, J., Soulsby, C., Malcolm, I.A., and Youngson, A.F., 2002. Hydrological controls on nutrient concentrations and fluxes in agricultural catchments. The Science of the Total Environment 294, 95-110.

Pinay, G., Decamps, H., Arles, C., and Lacassin-Seres, M., 1989. Topographic influence on carbon and nitrogen dynamics in riverine woods. Archive fur Hydrobiologie 114, 401-414.

Pionke, H.B. and Urban, J.B., 1985. Effect of agricultural land use on ground-water quality in a small Pennsylvania watershed. Ground Water, 23, 80-98.

Rademacher, L.K., Clark, J.F., Clow, D.W., and Hudson, G.B., 2005. Old groundwater influence on stream hydrochemistry and catchment response times in a small Sierra Nevada catchment: Sagehen Creek, California. Water Resources Research 41, WO2004, 1-10.

SAS Institute, Inc., 2003. Getting Started with SAS. Cary, North Carolina, U.S.A.

Schilling, K.E., 2002. Chemical transport from paired agricultural and restored prairie watersheds. Journal of Environmental Quality 31, 1184-1193.

Sebilo, M., Billen, G., Grably, M., and Mariott, A., 2003. Isotopic composition of nitrate-nitrogen as a marker of riparian and benthic denitrification at the scale of the whole Seine River system. Biogeochemistry 63, 35-51.

Smart, R.P., Soulsby, C., Neal, C., Wade, A., Cresser, M.S., Billett, M.F., Langan, S.J., Edwards, A.C., Jarvie, H.P., and Owen, R., 1998. Factors regulating the spatial and temporal distribution of solute concentrations in a major river system in NE Scotland. The Science of the Total Environment 221, 93-110.

Smith, R.A., Schwarz, G.A., and Alexander, R.B., 1997. Regional interpretations of water-quality monitoring data. Water Resources Research 33, 2781-2798.

Soranno, P.A., Hubler, S.L., Carpenter, S.R., and Lathrop, R.C., 1996. Phosphorous loads to surface waters: a simple model to account for spatial pattern of land use. Ecological Applications 6, 865-878.

Stoddard, J.L, Peck, D.V., Olsen, A.R., Larson, D.P., Van Sickle, J., Hawkins, C.P., Hughes, R.M., Whittier, T.R., Lomnicky, G., Herlihy, A.T., Kaufman, P.R., Peterson, S.A., Ringold, P.L., Paulsen, S.G., and Blair, R., 2005. Environmental Monitoring and Assessment Program (EMAP): Western Streams and Rivers Statistical Summary. EPA 620/R-05/XXX, U.S. Environmental Protection Agency, Washington, D.C.

Sylvia, D.M., Fuhrmann, J.J., Hartel, P.G., and Zuberer, D.A., 1998. Principles and Applications of Soil Microbiology. Prentice Hall.

Van Leeuwen, E.P., Draaijers, G.P.J., and Erisman, J.W., 1996. Mapping wet deposition of acidifying components and base cations over Europe using measurements. Atmospheric Environment 20, 2495-2511.

Van Sickle, J., Baker, J., Herlihy A., Bayley, P., Gergory, S., Haggerty, P., Ashkenas, L., and Li, Judith, 2004. Projecting the biological condition of streams under alternative scenarios of human land use. Ecological Applications 14, 368-380.

Vogelmann, J.E., Howard, S.M., Yang, L., Larson, C.R., Wylie, B.K., and Van Driel, N., 2001. Completion of the 1990s National Land Cover Data Set for the Conterminous United States from Landsat Thematic Mapper Data and Ancillary Data Sources. Photogrammetric Engineering and Remote Sensing 67, 650-662.

Welsch, D.L., Kroll, C.N., McDonnell, J.J., and Burns, D.A., 2001. Topographic control on the chemistry of subsurface stormflow. Hydrological Processes 15, 1925-1938.

Wickham, J.D., O'Neill, R.V., Riitters, K.H., Smith, E.R., Wade, T.G., and Jones, K.B., 2002. Geographic targeting of increases in nutrient export due to future urbanization. Ecological Applications 12, 93-106.

Wickham, J.D., Stehman, S.V., Smith, J.H., and Yang, L., 2004. Thematic accuracy of MRLC-NLCD land cover for the western United States. Remote Sensing of Environment 91, 452-468.

Wondzell, S.M., and Swanson, F.J., 1999. Floods, channel change, and the hyporheic zone. Water Resources Research 35, 555-567.

Chapter 4

Evaluation of the new Hydrologic Landscape Regions in Oregon and California using Lowflow Stream Chloride and Nitrate Concentrations

Cara J. Poor

Jeffrey J. McDonnell

John Bolte

For Submission to Environmental Management

Abstract

One of the main science themes of the IAHS Decade on Prediction in Ungauged Basins is watershed classification. The first major watershed classification scheme for the entire United States was published recently in this journal by Wolock et. al. (2004). This paper represents the first formal test of the Wolock et. al. (2004) hydrologic landscape regions. We bring to bear 124 catchments from two regions with low atmospheric deposition; Western Oregon and Northern California. We test the new hydrologic landscape regions with linear regression models using the primary hydrologic flowpaths that are most likely to dominate catchment hydrology in each region. Including hydrologic landscape regions (in the form of primary hydrologic flowpaths) significantly improved chloride predictions, but did not improve nitrate predictions for both the Western Oregon and Northern California datasets. Results of the linear regressions imply that the hydrologic setting of the catchments are adequately represented (from chloride, which is tightly linked to hydrology), and nitrate is more strongly affected by transformation processes such as denitrification. Hydrologic landscape regions and land use predicted chloride relatively well for the Western Oregon dataset, whereas poor chloride and nitrate prediction occurred using the Northern California dataset. This is likely due to the dominance of one or two explanatory variables (forested land use and the shallow groundwater flowpath). The catchments in Northern California are dominated by forest land cover (average 99.79%) and all catchments except one have the shallow groundwater flowpath. This study shows that hydrologic landscape regions are a useful tool for predicting quasi-

conservative anion concentrations in regions with significant variation in primary hydrologic flowpaths.

Introduction

The IAHS International Decade on Predictions in Ungauged Basins (PUB) has become a major focus of the global hydrologic community (Clarke 2005; Sivapalan et. al., 2003). Many basins throughout the world are ungauged or inadequately gauged, creating a need for extrapolation of knowledge from gauged basins to ungauged basins for watershed management decisions. Due to the heterogeneity of climate and landscape and our current lack of understanding of basin responses, extrapolating calibrated models from a gauged to ungauged basin has proven to be woefully unsuccessful (Sivapalan, 2003). Adequate water quantity and quality predictions are needed to make informed, sustainable management decisions to prevent further ecosystem degradation and promote human life and health (Sivapalan et. al., 2003). A call for improvements to predictions in ungauged basins has recently been made in the hydrologic literature (Littlewood et. al., 2003; Sivapalan, 2003; Sivapalan et. al., 2003). One proposed method for improvements has been termed the "hybrid top-down" modeling. This method establishes relationships that control the behavior of environmental systems such as streamwater quality responses to changes in landscape, in addition to including simple conceptual representations of processes (Littlewood et. al., 2003). These hybrid top-down models can be more useful than purely statistical models (e.g., linear regression models); mathematical representations of processes will aid in understanding the response of the basin while eliminating the need for parameterization that is required in process models. Processes that affect input and output variables are not represented in purely statistical models (sometimes called "black box" models), and while this type of model may be useful for some environmental management purposes, it cannot contribute to the scientific understanding of within-system processes.

It is well established that hydrologic response within a catchment varies depending on the region and corresponding climatic inputs. For instance, a catchment in an arid, low latitude region will respond differently to a storm event than a catchment in a humid, temperate region. To determine the appropriate modeling approach for predictions in ungauged basins, recent calls have been made for watershed classification based on the hydrologic response (McDonnell and Woods, 2004). The scientific community has not developed an organized way of acknowledging the wide variance of hydrological processes from one region to the next (McDonnell and Woods, 2004). Classification would group together catchments that have similar hydrologic settings, providing an initial screening of catchments that may be successfully modeled when grouped together. It is the first step to resolving the challenge of predicting stream water quantity and quality in ungauged basins.

Classification of catchments or regions based on various sets of criteria has been discussed for some time (Chapman, 1987; Winter, 2001; Omernik and Griffith, 1991; Preston, 2000; Baker et. al., 2001). These catchments/regions have similar ecologic, water quality, or hydrologic characteristics, such as runoff ratios or %surface runoff. Chapman (1987) presents a catchment classification scheme based on climate (average precipitation/evaporation) and type of catchment response which resulted in catchment descriptions of humid vs. dry, temperate vs. snow and ice vs. warm, and areas with a catchment response to precipitation inputs vs. flatland. Preston (2000) defined a hydrochemical response unit for Maryland, which identifies areas with similar land use, soil type, slope, and geology at the catchment scale. The hydrologic setting of catchments has been classified using hydrologic units, which identify areas with similar soil, geology, slope, precipitation, and potential evapotranspiration (Wolock et. al., 2004; Winter, 2001). While these classification schemes have been proposed, none have been independently tested. Results have shown relationships between classification and the water quality parameters that helped develop them, but separate datasets have not been used to test classifications (i.e., Preston, 2000). The hydrologic landscape regions (HLRs) classification scheme developed by Wolock et. al. (2004) is the first objective classification for the entire United States that can be tested with independent data.

Areas with similar hydrologic settings will likely have a similar response to chemical inputs, especially chloride and nitrate, which have been strongly linked in past studies to catchment hydrology (Creed et. al., 1996; Creed and Band, 1998; McHale et. al., 2002). We choose chloride and nitrate out of all possible water quality parameters due to their unique properties; chloride is relatively conservative and largely affected by the hydrology of a catchment and nitrate is a very reactive nutrient that is affected by both catchment hydrology and transformation processes (i.e., denitrification, nitrification, plant uptake, etc.). While land use has been shown to affect chloride concentrations (Herlihy et. al., 1998; Smart et. al., 1998), chloride is often used in pristine catchments as a conservative tracer of water (Kirchner et. al., 2001; Neal and Rosier, 1990; Nyberg et. al., 1999). Comparison between the highly reactive nitrate and the quasi-conservative chloride may provide useful insights. Elevated nitrate concentrations in streams due to agricultural activity has also caused much concern and created the need to study the processes that affect nitrate transport to streams (Pimental, 1993; Howarth et. al., 2002). If the HLR classification system can successfully predict hydrologic conditions within a catchment, the ability to predict nitrate and chloride concentrations and other water quality parameters at the outlet of a catchment may be vastly improved.

Hydrologic landscape regions were developed to account for the fundamental hydrologic processes that will likely affect water quality in small watersheds (approximately 200 km^2), and were based on the hydrologic landscapes concept of Winter (2001). The fundamental hydrologic processes occurring in a catchment - surface runoff, which is controlled by slope and permeability; groundwater flow, which is controlled by geologic characteristics; and atmospheric-water exchange, which is controlled by climate - were accounted for in the hydrologic landscapes concept (Winter, 2001). To determine HLRs for the United States, Wolock et. al. (2004) delineated a set of small watersheds (on the order of 200 km^2) from digital elevation models. Land surface form, geologic texture, and climate characteristics were then quantified. Land surface form was described by relief (maximum elevation minus minimum elevation in the watershed), total percentage flatland located in the upland areas of the watershed, and percentage of flatland in the lowland areas of the watershed (Wolock et. al., 2004). Geologic texture was described by soil and bedrock permeability. Soil permeability was estimated using the percentage of sand in the soil, and bedrock permeability was quantified by assigning permeability classes to the general lithologic groups. Climate characteristics were described by mean annual precipitation minus potential evapotranspiration (PET). Land surface form, geologic texture, and climate characteristics were used to assign the watersheds to similar groups that define the HLRs using principal components analysis (PCA). The PCA resulted in five principal components that explained at least 10% of the total variance in the data, which were retained for further analysis (Wolock et. al., 2004). Twenty HLRs were defined based on similar characteristics. A cluster analysis of the scores of the five principal components was then used to assign each watershed to one of 20 HLR groups. The resulting HLRs are identified by numbers and vary from 1-20, with areas of similar hydrologic settings located in different geographic locations. Wolock et. al. (2004) hypothesized that these HLRs had four combinations of primary hydrologic flowpaths: 1) shallow groundwater and deep groundwater (SGW-DGW), 2) overland flow and deep groundwater (OF-DGW), 3) shallow groundwater (SGW), and 4) overland flow (OF). These primary hydrologic flowpaths can be expected to affect water quality; regions with overland flow may transport nutrients/pollutants from the land surface directly to streams, regions with shallow groundwater flow may "flush" nutrients/pollutants from the soil layer to the stream, or regions with deep groundwater flow may transport nutrients/pollutants from the land surface to groundwater.

Hydrologic landscape regions appear to be a potentially successful tool for predicting water quantity and quality, however this classification scheme needs to be further tested with a range of water quality parameters in different regions. Testing of these classifications is necessary to determine their validity and usefulness in the quest to improve the understanding of hydrologic processes. Wolock et. al. (2004) determined how well HLRs explained the variance in fish species richness and nitrogen transport efficiency (estimated percentage of total nitrogen inputs to the basin that is exported from the basin in the stream) using analysis of variance (ANOVA) R^2 values, but did not attempt any linear regressions using HLRs to predict water quality. Hydrologic landscape regions need to be used in linear regressions or simple statistical models to test their validity for predicting water quality parameters. Linear regressions using a multiple of variables that affect water quality parameters are the most practical way to predict water quality. Determining the relationship between two variables provides insights, but the complex nature of water quality requires the use of linear regressions with multiple parameters. In addition, water quality parameters that are frequently measured and used for management decisions, such as chloride and nitrate, need to be tested in the HLR classification framework. The next step towards improving our ability to predict water quantity and quality in ungauged basins is to use the classification developed by Wolock et. al. (2004) in a linear regression using measured water quality data. We hypothesize that defining areas with similar hydrologic settings using HLRs will improve prediction of stream nitrate and chloride concentrations. Areas with similar hydrologic settings may have similar responses to a storm event (e.g., streams in one hydrologic setting may be flashy due to intense rainfall and small storage in groundwater aquifers, whereas streams in another hydrologic setting may have a slow response to a rainfall event due to large groundwater stores and high rates of evaporation). During lowflow, a catchment whose primary hydrologic flowpath is shallow groundwater may be impacted more by the transformation processes occurring in the shallow subsurface (denitrification, nitrification, plant uptake, etc. for nitrate) compared to a catchment whose primary flowpath is deeper groundwater, which bypasses biologically active areas. These different hydrologic settings will in turn affect the mobilization of nitrate or chloride to the stream.

This study brings together 124 catchments in Northern California and Western Oregon, which represent regions with three of the four primary hydrologic flowpaths predicted by Wolock et. al. (2004). Lowflow water quality sampling, in addition to land cover and attribute data, was used to test the HLR groupings. Signals from primary flowpaths (shallow groundwater, deep groundwater, etc.) may be clearer during lowflow when the catchment is not completely saturated. A clear signal may also come from the different land uses within the catchment during lowflow. Linear regression models using HLRs and inverse-distance weighting of land use to predict nitrate and chloride were developed. To determine how region affects regression performance and regression coefficients, watershed land cover, nitrate, and chloride data from two different regions, Northern California and the Willamette Valley in Western Oregon, were modeled. We explore the following questions:

1. Will the HLR classification scheme of Wolock et. al. (2004) be correlated to nitrate and chloride concentrations in Western Oregon and Northern California?
2. Will the primary hydrologic flowpaths identified from HLRs help predict a quasi-conservative anion (chloride), which is largely controlled by the hydrology of a catchment?
3. Will the primary hydrologic flowpaths identified from the HLRs help predict a very reactive nutrient (nitrate), which is controlled by transformation processes as well as the hydrology of the catchment?
4. How does region affect regression coefficients and predictive ability? Are primary hydrologic flowpaths more significant in a region dominated by one land use (Northern California catchments are 99.79% forested on average) compared to a region with more variable land use (Western Oregon catchments are 73.32% forested on average)?

Methods

Study Areas. The study catchments are located in Northern California and Western Oregon, and are described in detail in Chapter 3. Both regions have low atmospheric deposition of nitrate (annual rate of approximately 1.57 kg N/ha/yr and 1.52 kg N/ha/yr for Northern California and Western Oregon, respectively, http://nadp.sws.uiuc.edu/nadpdata). The 59 Northern California study catchments range in area from 112 – 62,299 ha and are largely forested (99.79% forested on average). The 76 Western Oregon study catchments range in area from 59 – 45,867 ha and include a mix of agricultural land use (21.17%) and urban (5.53%) land use in addition to predominant forested land use (73.30%). The Northern California dataset was obtained from the Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP) and Regional Environmental Monitoring and Assessment Program (REMAP; Stoddard et. al., 2005). The Western Oregon dataset is comprised of data from EMAP, REMAP, an EPA agricultural-riparian study (Moser et. al., 1997), and a pre-pilot EMAP study (Herlihy et. al., 1997; Peck et. al., 2005a; Peck et. al, 2005b). Due to some catchments having more than one HLR grouping within the catchment boundary, only 53 catchments from the Northern California dataset and 71 catchments from the Western Oregon dataset were ultimately used in linear regression models. Sampling locations of modeled catchments for Western Oregon and Northern California are shown in Figure 4.1.

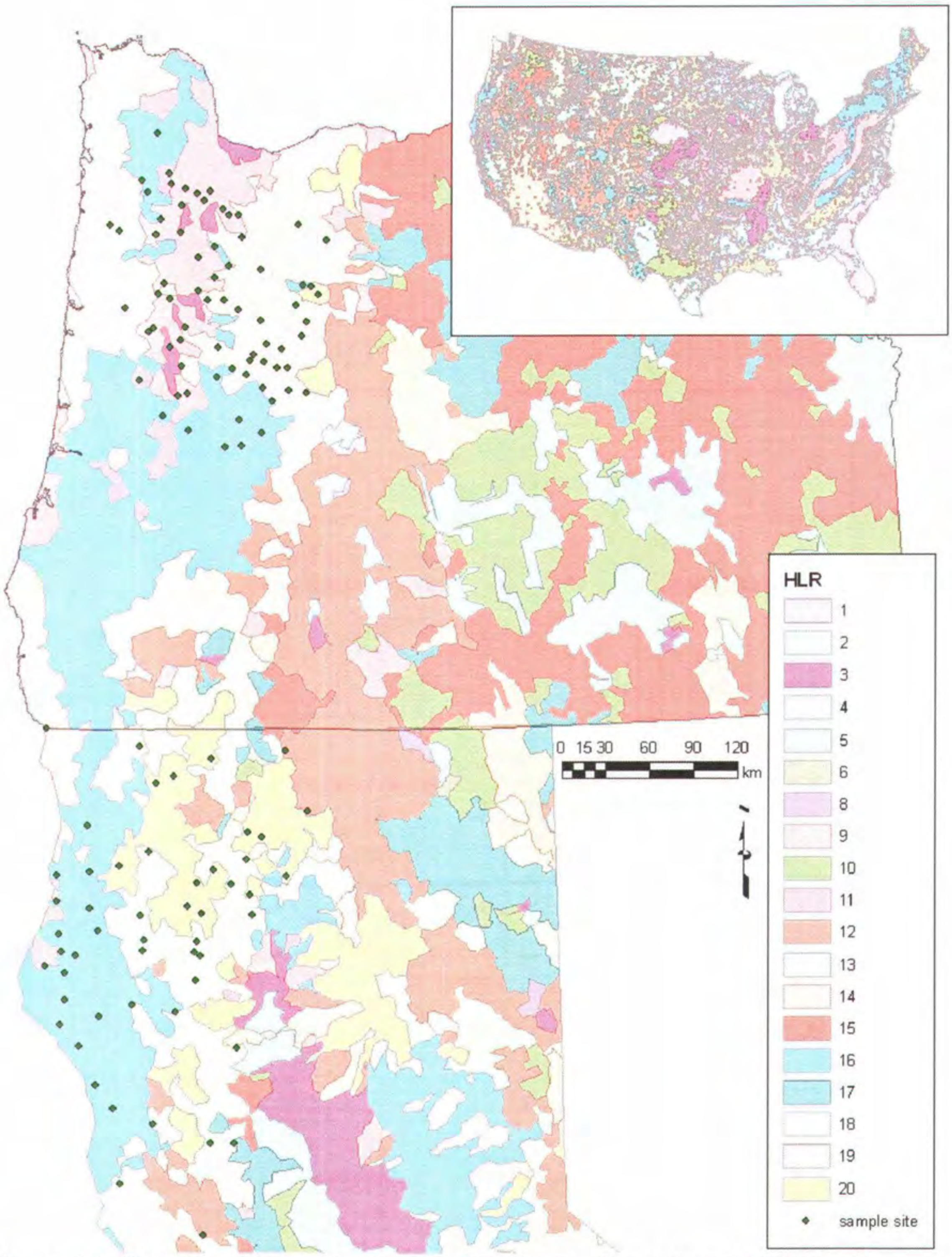

Figure 4.1. Sampling Sites at the Outlet of Study Catchments and Hydrologic Landscape Regions in Western Oregon and Northern California. Inset: Hydrologic Landscape Regions in the United States.

Regression Description. Equations and distance calculations are described in detail in Chapter 3. Briefly, land use was grouped into three categories: forested, agricultural, and urban. Area, inverse-distance (1/d), and inverse-distance squared ($1/d^2$) were then calculated for each cell in the catchment, and summed for each category of land use. To minimize the effects of watershed area on results, each land use measure (area, 1/d, and $1/d^2$) was normalized (i.e., total forested area/total area). As a result, the three land use measures summed to one and the resulting solution was not unique. To resolve the non-unique solution, we removed one of the land use measures. We chose the urban land use since it comprises such a small percentage of land use in the Western Oregon and Northern California catchments (5.52% and 0.08% for Western Oregon and Northern California, respectively).

Maps of HLRs were provided by the United States Geological Survey (USGS), and are grouped after Wolock et. al. (2004). To include HLR, the four combinations of primary hydrologic flowpaths were added to the linear regression equations. The resulting variables were qualitative; catchments were assigned a value of one (the catchment has these hydrologic flowpaths) or zero (the catchment does not have these hydrologic flowpaths).

Hydrologic landscape regions for Western Oregon and Northern California are shown in Figure 4.1. Not all of the HLRs are represented, since Western Oregon and Northern California do not have the complete range of geologic, soil, and precipitation types found in the United States. In Western Oregon, only groups 3, 9, 11, 12, 16, 19, and 20 are represented (see definitions below). All of these groups are present in the modeled catchments except for group 12. These groups reflect characteristics of subhumid plains with overland flow and deep groundwater (group 3), humid plateaus with overland flow and deep groundwater (group 9), humid plateaus with overland flow (group 11), semiarid plateaus with shallow groundwater (group 12), humid mountains with shallow groundwater (groups 16 and 20), and very humid mountains with shallow groundwater (group 19). All of these groups have a positive precipitation minus potential evapotranspiration (PET) value, except for group 12 (Wolock et. al, 2004). Group 12 is on the southern tip of the Willamette Valley in Western Oregon, which is typically drier than the rest of the valley and is not present in any of the modeled catchments. Since only three of the four primary hydrologic flowpath combinations are present in Western Oregon; overland flow and deep groundwater (OF-DGW), shallow groundwater (SGW), and overland flow (OF), , the fourth (shallow groundwater and deep groundwater) was removed from the linear regressions. In addition, the third primary hydrologic flowpath (OF) was removed from the linear regressions to minimize the effects of collinearity. Because a catchment will have one of these variables when it doesn't have the other two, OF-DGW, SGW, and OF are all somewhat collinear. The removal of OF resulted in only OF-DGW and SGW represented in the equations.

All of the HLRs are present in Northern California except 1 (subhumid plains with shallow and deep groundwater), 2 (humid plains with shallow and deep groundwater) and 7 (humid plains with shallow groundwater). Groups 9, 12, 16, 18, 19, and 20 are present in the modeled catchments. These groups reflect characteristics of humid plateaus with overland flow and deep groundwater (group 9), semiarid plateaus with shallow groundwater (group 12), humid mountains with shallow groundwater (group 16), semiarid mountains with shallow groundwater (group 18), very humid mountains with shallow groundwater (group 19), and humid mountains with shallow groundwater (group 20). All of these groups have a positive precipitation minus PET value, except for groups 12 and 18 (Wolock et. al., 2004). Both of these groups represent semiarid areas. Only two of the four primary hydrologic flowpath combinations are present in Northern California; SGW and OFW-DGW. One was removed to eliminate the effects of collinearity. Thus, for the Northern California regressions, only one flowpath was represented (SGW).

The resulting linear regressions include land use (forested and agricultural) and the OF-DGW and SGW flowpaths. The general regression form is:

$$\log(NO_3^-) = a_0 + a_1(F_{eff1}) + a_2(F_{eff2}) + a_3(A_{eff1}) + a_4(A_{eff2}) + b_1(OF - DGW) + b_2(SGW) \tag{1}$$

where NO_3^- represents stream nitrate concentration and the subscripts eff1 and eff2 denote the in-stream and out-of-stream effects, respectively. Variables with the eff1 subscript can be in-stream inverse-distance (1/d) or in-stream inverse-distance squared ($1/d^2$), and variables with the eff2 subscript can be out-of-stream inverse-distance (1/d) or out-of-stream inverse-distance squared ($1/d^2$). Regressions including in-stream and out-of-stream inverse-distance were depicted as $In+Out^{-1}$ and regressions including in-stream and out-of-stream inverse-distance squared were depicted as $In+Out^{-2}$. Only eff1 was used for the area, total inverse-distance (in-stream 1/d + out-of-stream 1/d), and total inverse-distance squared (in-stream $1/d^2$ + out-of-stream $1/d^2$) regressions. Area regressions were depicted as %Landuse or %Landuse-HLR, total inverse-distance regressions were depicted as Total^{-1}, and total inverse-distance squared regressions were depicted as Total^{-2}. To determine the relationship between in-stream and out-of-stream effects, a multiplicative regression was also tested:

$$\log(NO_3^-) = a_0 + a_1(F_{eff1})*(F_{eff2}) + a_3(A_{eff1})*(A_{eff2}) + b_1(OF - DGW) + b_2(SGW) \quad (2)$$

Since incorporation of in-stream and out-of-stream effects is relatively new, the exact interaction and relationship between these effects and nitrate is still unknown. Here we test an additive and multiplicative interaction. The multiplicative regressions were depicted as In*Out^{-1} for inverse-distance and In*Out^{-2} for the inverse-distance squared. The same form was used for the chloride regressions (please see Chapter 3). Although the same equations and distance calculations were used as in Chapter 3, the focus of this paper is on testing the HLR classification scheme, whereas the paper consisting of Chapter 3 tests the use of inverse-distance weighting and topographic index (an index used to predict the saturated areas within a catchment). Both papers are attempting to improve empirical nitrate and chloride predictions, but taking distinctly different approaches.

SAS version 9.1 (SAS Institute, 2003) was used to perform linear regressions. Separate regressions for the Northern California and Western Oregon datasets were created to compare coefficients and regression performance. To determine whether or not the HLR classification significantly improved regressions, a partial F-test was conducted. All regressions were compared using Akaike's Information Criterion

(AIC). The difference between the regression with the lowest AIC and the AIC of the regression of concern (ΔAIC) determined which model was the most accurate, based on model performance with an added penalty for using a large number of variables. Due to the relatively small datasets, multiple cross validation was conducted.

Results

Regressions. Tables 4.1 and 4.2 show nitrate and chloride regression results, respectively, using land use and HLR for Western Oregon. The partial F-test (α = 0.05) indicated that including HLR did not significantly improve nitrate regressions (R^2 of 0.26 and 0.27 for the %Landuse and %Landuse-HLR regressions, respectively). The ΔAIC revealed that the inverse-distance regressions also did not improve prediction of nitrate; the best regression was the %Landuse regression. For chloride, regressions using HLR had a higher R^2 value than the regression using just %land use (R^2 of 0.42 and 0.52 for the %Landuse and %Landuse-HLR regressions, respectively), and the partial F-test (α = 0.05) indicated that the regressions were significantly improved with HLR. Combining HLR with land use improved regression results compared to using just HLR or land use (R^2 of 0.42, 0.49, and 0.52 for the %Landuse, HLR, and %Landuse-HLR regressions, respectively). Using inverse-distance weighting for land use slightly improved results (R^2 of 0.52, 0.58, and 0.58 for the %Landuse-HLR, In+Out^{-1} Landuse-HLR, and In*Out^{-2} Landuse-HLR regressions, respectively). The ΔAIC value indicated that the best regression was the In*Out^{-2} Landuse-HLR regression. For the Northern California datasets, poor regressions resulted for both nitrate and chloride (best R^2 of 0.07 and 0.08 for nitrate and chloride, respectively). The addition of HLR did not significantly improve regressions (α = 0.05).

Model	Terms (coeff., SE)				R^2	ΔAIC
HLR					0.15	12.4
	OF-DGW,	SGW				
	(0.659,1.11)	(-0.662,1.09)				
%Landuse					0.26	0.0
	Forested					
	(-2.51, 0.503)					
%Landuse-HLR					0.27	3.7
	Forested,	OF-DGW,	SGW			
	(-2.65, 0.806)	(-0.552, 1.10)	(-0.427, 1.02)			
$Total^{-1}$ Landuse-HLR					0.25	5.3
	Forested,	OF-DGW,	SGW			
	(-2.35, 0.779)	(-0.460, 1.11)	(-0.461, 1.03)			
$Total^{-2}$ Landuse-HLR					0.17	12.3
	Forested,	OF-DGW,	SGW			
	(-0.937, 0.659)	(0.253, 1.14)	(-0.675, 1.08)			
$In+Out^{-1}$ Landuse-HLR					0.25	7.1
	IS Forested,	OS Forested,	OF-DGW,	SGW		
	(-0.410, 2.15)	(-2.08, 2.35)	(-0.452, 1.12)	(-0.435, 1.04)		
$In+Out^{-2}$ Landuse-HLR					0.24	8.3
	IS Forested,	OS Forested,	OF-DGW,	SGW		
	(0.534, 0.949)	(-2.80, 1.25)	(-0.182, 1.12)	(-0.373, 0.949)		
$In*Out^{-1}$ Landuse-HLR					0.26	4.1
	IS*OS Forested,	OF-DGW,	SGW			
	(-2.08, 0.644)	(-0.377, 1.09)	(-0.282, 1.02)			
$In*Out^{-2}$ Landuse-HLR					0.22	8.3
	IS*OS Forested,	OF-DGW,	SGW			
	(-1.46, 0.599)	(-0.0196, 1.11)	(-0.404, 1.05)			

Table 4.1. Nitrate Regression Results using Land Use and HLRs for the Western Oregon Dataset.

Model	Terms (coeff., SE)				R^2	ΔAIC
HLR					0.49	12.4
	OF-DGW, (0.339, 0.339)	SGW (-0.599, 0.333)				
%Landuse					0.42	18.6
	Forested (-1.25, 0.176)					
%Landuse-HLR					0.52	9.4
	Forested, (-0.567, 0.256)	OF-DGW, (0.0799, 0.350)	SGW (-0.550, 0.324)			
Total^{-1} Landuse-HLR					0.56	4.3
	Forested, (-0.757, 0.237)	OF-DGW, (-0.022, 0.338)	SGW (-0.536, 0.313)			
Total^{-2} Landuse-HLR					0.53	8.3
	Forested, (-0.482, 0.196)	OF-DGW, (0.130, 0.338)	SGW (-0.607, 0.321)			
In+Out^{-1} Landuse-HLR					0.58	2.1
	IS Forested, (-1.72, 0.634)	OS Forested, (1.10, 0.694)	OF-DGW, (-0.0390, 0.330)	SGW (-0.555, 0.306)		
In+Out^{-2} Landuse-HLR					0.57	4.8
	IS Forested, (-0.482, 0.283)	OS Forested, (-0.253, 0.372)	OF-DGW, (-0.000531, 0.333)	SGW (-0.545, 0.312)		
In*Out^{-1} Landuse-HLR					0.58	0.7
	IS*OS Forested, (-0.726, 0.192)	OF-DGW, (-0.0225, 0.324)	SGW (-0.468, 0.306)			
In*Out^{-2} Landuse-HLR					0.58	0.0
	IS*OS Forested, (-0.671, 0.173)	OF-DGW, (0.026, 0.319)	SGW (-0.483, 0.304)			

Table 4.2. Chloride Regression Results using Land Use and HLRs for the Western Oregon Dataset.

For Western Oregon, chloride regression results were significantly better than nitrate regression results, with the chloride In*Out^{-2} Landuse-HLR regression having the best R^2 value of 0.58 (Figure 4.2). The nitrate %Landuse-HLR regression produced the best R^2 value of 0.27, although the ΔAIC value indicated that the %Landuse regression was the best predictor of nitrate (R^2 of 0.26). The standard error for OF-DGW and SGW in the nitrate regressions was also very high (on the order of 1.0). In all of the nitrate and chloride regressions, the sign of the SGW variable was negative. The sign of the OF-DGW variable varied from positive to negative, indicating that the OF-DGW variable may not be a stable predictor of nitrate and chloride concentrations.

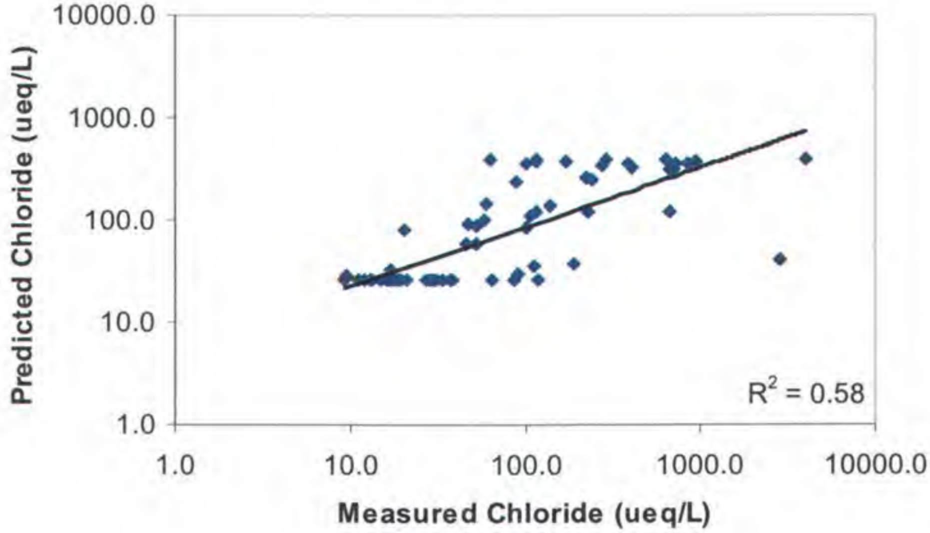

Figure 4.2. Measured vs. Predicted Stream Chloride Concentrations for Western Oregon (In*Out^{-2} Landuse-HLR Regression).

Validation. Multiple cross-validation was used to validate the regressions. See Chapter 3 for a detailed discussion of validation methods. Models were considered validated if the signs of the coefficients stayed the same and coefficients did not vary by more than 100%. For Western Oregon, the signs of all coefficients in all nitrate regression models stayed the same except for the forested (in-stream distance)$^{-1}$ coefficient in the In+Out^{-1} Landuse-HLR regression, and the OF-DGW and SGW coefficients in the In+Out^{-2} Landuse-HLR regression. The coefficients of the forested variables in the nitrate regressions varied less than 31%. The exceptions were the forested in-stream 1/d and 1/d^2 variables, which varied by 169% and 96%, and the forested out-of-stream 1/d and 1/d^2 forested variables, which varied by 61% and 37%

in the In+Out^{-1} Landuse-HLR and In+Out^{-2} Landuse-HLR regressions, respectively. High variability occurred with the SGW and OF-DGW coefficients in almost all regressions, ranging from 29-254% for SGW and 59-198% for OF-DGW. Regressions with SGW and OF-DGW coefficients varying by more than 100% included the Total^{-1} Landuse-HLR, Total^{-2} Landuse-HLR, In+Out^{-1} Landuse-HLR, In+Out^{-2} Landuse-HLR, In*Out^{-1} Landuse-HLR, and In*Out^{-2} Landuse-HLR regressions.

The variables in the chloride regressions varied to a somewhat lesser degree. The coefficient of the forested variable in all regressions varied less than 33%, and the SGW coefficient varied by less than 12%. The OF-DGW coefficient varied considerably, ranging from 13-437%. In the In+Out^{-2} Landuse-HLR regression, OF-DGW varied by 14,000% due to the small coefficient (-0.00053). The overall difference of the coefficient was 0.11. Regressions with the OF-DGW coefficient varying by more than 100% included the %Landuse-HLR, Total^{-1} Landuse-HLR, In+Out^{-1} Landuse-HLR, In+Out^{-2} Landuse-HLR, In*Out^{-1} Landuse-HLR, and In*Out^{-2} Landuse-HLR regressions. The signs of all coefficients in all chloride models stayed the same except for OF-DGW in the Total^{-1} Landuse-HLR, In+Out^{-1} Landuse-HLR, In+Out^{-2} Landuse-HLR, and In*Out^{-1} regressions.

Discussion

Recent calls for improvements to predictions in ungauged basins have been made in the hydrologic literature (Littlewood et. al., 2003; Sivapalan, 2003; Sivapalan et. al., 2003). Watershed classification is the first step to resolving the challenge of predicting stream water quantity and quality; classification would group together catchments that have similar hydrologic settings, providing an initial screening of catchments that may be successfully modeled when grouped together. Hydrologic landscape regions are an objective way to begin classifying catchments based on the attributes that produce characteristic hydrologic responses. This classification may help predict hydrologic settings, which in turn can aid in the prediction of water quality. In this study, HLRs did not significantly improve nitrate predictions, but the classification did improve chloride predictions. Land use was a better predictor of nitrate than HLRs, which shows that for the Western Oregon dataset human modifications to the catchment affect nitrate more than the underlying hydrology. Chloride may be affected to a lesser degree by human modifications and more by the catchment's hydrologic flowpaths. Human modifications in a catchment can include surface disturbance/fertilizer application associated with agricultural activity, hardening of the catchment/leaky septic and sewer systems associated with housing developments, and hydrologic modifications to the streams including lining of the stream and surface impoundments. The effect of surface impoundment was not explored in this study, but we would expect impoundments to affect nitrate more than chloride. Residence times behind the impoundment would be longer, creating more opportunity for denitrification and other transformation processes to occur that would affect nitrate concentrations and not chloride. Although more work needs to be done to understand the reasons for the differences in prediction, these regressions are the first test of Wolock et. al.'s (2004) hydrologic landscape regions.

HLR Parameters. Hydrologic landscape regions were moderately correlated with chloride and nitrate from the Western Oregon dataset ($r = -0.699$, $p<0.0001$ and $r = -0.378$, $p = 0.012$ for chloride and nitrate, respectively), but not with chloride and nitrate from the Northern California dataset. The primary hydrologic flowpaths associated with HLRs significantly improved chloride predictions in Western Oregon, but did not improve nitrate predictions ($\alpha = 0.05$ significance level). The difference between chloride and nitrate regressions for Western Oregon may be due to the way HLRs are grouped; HLR groupings are based on land surface form, geologic texture, and climate in terms of %flatland, bedrock permeability class, %sand, and precipitation minus PET. In some cases, slope was used instead of %flatland. To better understand the functional relationships between HLRs and the parameters used to group them, we looked at correlations between nitrate/chloride and the HLR parameters. We did not have access to the same datasets used to determine HLR parameters, but we looked at parameters we thought were similar. Nitrate and chloride were found to be correlated to slope, precipitation, soil, and geology for the Western Oregon dataset (Table 4.3), indicating that HLRs should adequately predict nitrate and chloride concentrations. Nitrate regressions did not perform as well as chloride regressions (best R^2 of 0.27 and 0.58 for nitrate and chloride, respectively), which is due to the poorer correlations between nitrate and HLR parameters. Correlations with slope, precipitation, soil, and geology were consistently lower for nitrate compared to chloride, which coincides with the ability of HLR to predict chloride and not nitrate.

Variable	Pearson's Coefficient
Nitrate	
Mean Precipitation	-0.460 **
Mean Slope	-0.467 **
%Type D Hydrosoil	0.367 *
Calc-alkaline Volcanics	-0.446 **
Lake Sediments	0.377 *
Glacial Drift	-0.372 *
Chloride	
Mean Precipitation	-0.683 **
Mean Slope	-0.655 **
%Type D Hydrosoil	0.580 **
Calc-alkaline Volcanics	-0.655 **
Lake Sediments	0.484 **

Table 4.3. Correlations of HLR Grouping Parameters with Stream Nitrate and Chloride Concentrations for the Western Oregon Dataset (* = $p<0.001$, ** = $p<0.0001$).

Land surface form determines the gravity flow of water within a catchment. Correlations with chloride and slope were higher than correlations with nitrate and slope ($r = -0.655$, $p<0.0001$ and $r = -0.467$, $p<0.0001$ for chloride and nitrate, respectively). The higher correlation between chloride and slope indicates that chloride is more tightly coupled to the flowpaths (water transport) of the catchment, and nitrate is possibly affected by additional processes.

Chloride had a higher correlation with precipitation than nitrate with precipitation ($r = -0.683$, $p<0.0001$ and $r = -0.460$, $p<0.0001$ for chloride and nitrate, respectively). This may be due to the orographic/rainout effects that control chloride deposition to a greater extent than nitrate. Figure 4.3 shows the variation in precipitation with distance from the coast. For Western Oregon, precipitation is high close to the coast, decreases in the Willamette Valley, and increases in the Cascade

Range (which borders the valley on the east side). The same pattern is observed for chloride deposition (see Chapter 3). Rainout effects have been observed in other coastal areas (Van Leeuwen et. al., 1996; Li, 1992; Gustafsson and Larsson, 2000; Hingston and Gailitis, 1976; Carratala et. al., 1998), and orographic effects, with an increase in chloride deposition with altitude, have also been observed (Gustafsson and Larsson, 2000; Fowler et. al., 1988). The correlation between chloride and nitrate is very similar to the correlation between precipitation and nitrate ($r = -0.473$, $p<0.0001$ and $r = -0.460$, $p<0.0001$ for chloride and precipitation, respectively), which indicates that nitrate concentrations are controlled somewhat by chloride deposition. The lower correlation between nitrate and precipitation (and the weaker relationship between rainout/orographic effects and stream nitrate concentrations) may partially account for the poor nitrate predictions. However, it is unclear from this study what mechanism is controlling these correlations.

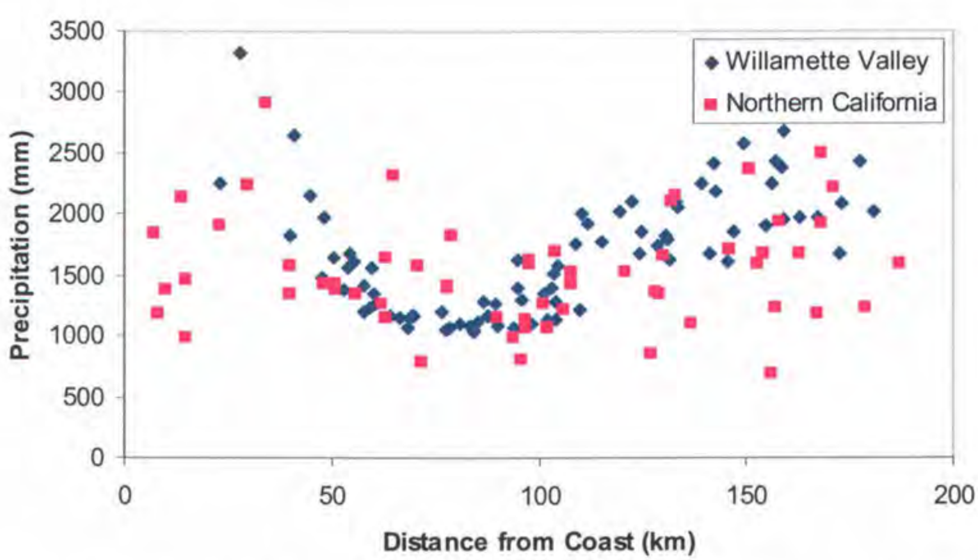

Figure 4.3. Variation in Precipitation with Distance from the Coast (from Chapter 3).

Geologic texture was determined by %sand (soil type) and bedrock permeability. Nitrate and chloride were correlated to soils, especially the Type D hydrologic soil classification ($r = 0.367$, $p = 0.0007$ and $r = 0.580$, $p<0.0001$ for nitrate and chloride, respectively). Hydrologic soil types classify soils based on the relative runoff potential of soils in catchments (USDA SCS, 1967). Four types, A, B, C, and D, are defined based on soil properties that influence runoff. Type A has high infiltration rates, type B moderate infiltration rates, type C slow infiltration rates, and type D very slow infiltration rates. Type D consists mainly of clay soils with a high swelling potential, soils with a high permanent water table, soils with claypan or clay layer at or near the surface, and shallow soils over nearly impervious materials. Again, chloride has a higher correlation with soils than nitrate. Soil type predicts the ease with which water is transported through soil, whether it is %sand or %Type D hydrosoil. Nitrate, which is often removed from the soil via plant uptake and microbial denitrification, is not transported with water as efficiently as chloride. Geologic type, which is similar to the bedrock permeability class in Wolock et. al. (2004), is also correlated with nitrate and chloride. Correlations between chloride and geology were higher than correlations between nitrate and geology (i.e., $r = -0.446$, $p<0.0001$ and $r = -0.655$, $p<0.0001$ between calcerous-alkaline volcanics and nitrate and chloride, respectively). Since these parameters predict hydrologic settings, a better correlation with chloride, which as a conservative tracer is more tightly linked to hydrology, is expected.

HLRs vs. Land use. Land use was a better predictor of nitrate than HLR. Regression results indicate that nitrate in the Western Oregon catchments were more affected by surface activities (i.e., agricultural, urban activity) than hydrologic flowpaths in a region with low atmospheric inputs (annual deposition rate of approximately 1.52 kg N/ha/yr for Western Oregon, http://nadp.sws.uiuc.edu/nadpdata). In contrast, chloride predictions in Western Oregon were significantly improved with the addition of HLRs, and the regression using HLRs alone explained more variability than the regression using land use alone (R^2 of 0.49 and 0.42 for the HLR and %Landuse regressions, respectively). Using in-stream and out-of-stream inverse-distance squared land use and HLRs explained the most variability of chloride. Primary hydrologic flowpaths in addition to land use appear to control chloride concentrations. Although the poor nitrate predictions makes it difficult to make definite statements on the mechanisms controlling stream nitrate concentrations, the nitrate response may be "swamped" by land use and obscuring the effects of hydrology on nitrate.

Both HLR variables used in the Western Oregon regressions did not have the same significance, however. The OF-DGW variable had a high p-value in all chloride/nitrate regressions (on the order of 0.9), and OF-DGW was not significantly different from zero when taking into account the standard error. In addition, the sign of OF-DGW was not consistent; in some regressions there is a positive relationship between chloride/nitrate and OF-DGW and in other regressions the relationship is negative. The validation process also showed a high variability in the OF-DGW coefficient. It may be best to remove the OF-DGW variable from the regressions. When the regressions are run without OF-DGW, the R^2 stays the same (0.52 for the chloride %Landuse-HLR regression). Reasons for the low significance of OF-DGW may include the lack of variability in the three primary hydrologic flowpaths; only two catchments are in the third category (OF). This leads us to believe that the regressions may be more stable if the OF-DGW variable is removed.

Regional Differences. Nitrate and chloride predictions using land use and HLRs were very poor for the Northern California dataset. In addition, there were no correlations between HLR, nitrate, and chloride. In contrast, adding HLR to regressions significantly improved chloride predictions for the Western Oregon dataset. Better performance for all regressions in general occurs with the Western Oregon dataset, which may be due to the higher variation in land use and HLR. Catchments in Northern California are mostly forested (average 99.97% forested) and the variation in slope is lower in Northern California than in Western Oregon (mean slope of 7.70-27.21% vs. 0.30-55.05% for Northern California and Western Oregon, respectively). In addition, 52 of the 53 catchments in Northern California had the shallow groundwater primary hydrologic flowpath, whereas three of the four primary hydrologic flowpaths were represented in the Western Oregon catchments. This lack of variability impedes any efforts to predict nitrate and chloride concentrations for the Northern California catchments. The variation in primary hydrologic flowpaths, upland/valley, and land use of the catchments used in the Western Oregon dataset likely improved modeling results.

Predictions in Ungauged Basins. The use of a classification scheme such as HLR is the first step towards predicting water quantity and quality in ungauged basins. This study furthers the initiative to improve predictions in ungauged basins, with an independent test of the HLR classification scheme developed by Wolock et. al. (2004). We choose chloride and nitrate out of many possible water quality parameters due to their unique properties; chloride is relatively conservative and is largely affected by the hydrology of a catchment and nitrate is a very reactive nutrient that is affected by both catchment hydrology and transformation processes (i.e., denitrification, nitrification, plant uptake, etc.). Nitrate and chloride are also commonly measured in most water quality studies, thus an abundance of data is available for testing. Prediction of nitrate, which is affected by hydrologic flowpaths and transformation reactions, was not improved using HLRs. Prediction of chloride, which is largely affected by hydrologic flowpaths, was significantly improved using HLRs. The difference in regression predictions may be due to the lack of variability in nitrate concentrations compared to chloride concentrations (many sites had stream nitrate concentrations at or below detection limits) in addition to transformation reactions affecting nitrate. It appears from both the Western Oregon nitrate regressions and Northern California regressions that a significant variation in all variables is needed to produce successful results. Although poor predictions are rarely reported in the literature, it is still important to show the negative result. Nonetheless, the chloride regressions show the potential for using HLRs to predict water quality in ungauged basins. Future classifications and water quality models will need to take these mixed results into account. Some water quality parameters may be successfully predicted using HLR classifications whereas other may not. Classification of hydrologic settings has the potential to be a very powerful tool, and more studies attempting to predict water quality and quantity should test the utility of HLRs. Although this hybrid modeling effort is still in the exploratory stages, we have shown that a simple linear regression can be conducted to estimate the catchment response for chloride using spatial GIS data (digital elevation models, land cover, HLR classification).

Conclusions

Our hybrid statistical model development incorporates process-based knowledge (hydrologic landscape regions) into empirical regressions, creating a link between statistical modeling and process literature. Comparison of chloride, a conservative tracer, and nitrate provided useful insights. Hydrologic landscape regions were moderately correlated with nitrate and chloride for the Western Oregon dataset, but not with nitrate and chloride for the Northern California dataset. Regression results revealed that HLRs via primary hydrologic flowpaths significantly improved chloride predictions, but did not improve nitrate predictions. Improved predictions of chloride compared to nitrate are likely due to the conservative nature of chloride, which creates a tighter link between chloride and the hydrology of the catchment (with the caveat that poor nitrate regression performance may also be due to low variability of nitrate concentrations). Poor predictions occurred using the Northern California dataset; whereas HLR and land use predicted chloride relatively well for the Western Oregon dataset. This may be due to the lack of variation in primary hydrologic flowpaths and land use in the Northern California dataset; nitrate and chloride variation in Northern California could not be explained by these variables.

Future work is needed to determine whether or not HLR can improve the prediction of stream nitrate and chloride concentrations. Hydrologic landscape regions identify areas of similar hydrologic settings, where similar water quality characteristics may occur. Hydrologic landscape regions may be more suitable for predicting water quality parameters when the catchment is saturated or "wetting up" during storm events. Sampling during wet conditions will show whether or not HLR can improve predictions under different conditions. Regression results were relatively successful in one region (Western Oregon), whereas regression results in the other region (Northern California) were very poor. Testing of the hydrologic landscape regions in other areas of the United States may provide some insights into its suitability for predicting water quality in ungauged basins.

Acknowledgements

We thank Dan Heggem, Jim Wigington, and John Van Sickle of the Environmental Protection Agency for chemistry and GIS data, Dave Wolock of the United States Geological Survey for hydrologic landscape region maps, and Alan Herlihy, Jim Wigington, and Pete Nelson for useful discussions. Many thanks to John Van Sickle for providing valuable statistical advice. This research was funded by the Environmental Protection Agency cooperative program.

References

Baker, M.E., Wiley, M.J., and Seelbach, P.W., 2001. GIS-based hydrologic regressioning of riparian areas: implications for stream water quality. Journal of the American Water Resources Association 37, 1615-1627.

Carratala, A., Gomez, A., and Bellot, J., 1998. Mapping rain composition in the East of Spain by applying kriging. Water, Air, and Soil Pollution 104, 9-27.

Chapman, T., 1987. Classification of regions *in* Comparative Hydrology: An ecological approach to land and water resources. M. Falkenmark and T. Chapman eds., Unesco, Stockholm, pp. 50-74.

Clarke, R.T., 2005. The PUB decade: how should it evolve? Hydrological Processes 19, 2865-2869.

Creed, I.F. and Band, L.E., 1998. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research 11, 3105-3120.

Creed, I.F., Band, L.E., Foster, N.W., Morrison, I.K., Nicolson, J.A., Semkin, R.S., and Jeffries, D.S., 1996. Regulation of nitrate-N release from temperate forests: A test of the N flushing hypothesis. Water Resources Research 32, 3337-3354.

Fowler, D., Cape, J.N., Leith, I.D., Choularton, T.W., Gay, M.J., and Jones, A., 1988. The influence of altitude on rainfall composition at Great Dun Fell. Atmospheric Environment 22, 1355-1362.

Gustafsson, M.E.R. and Larsson, E.H., 2000. Spatial and temporal patterns of chloride deposition in Southern Sweden. Water, Air, and Soil Pollution 124, 345-369.

Herlihy, A., Kaufmann, P., Reynolds, L., Li, J., and Robison, G., 1997. Developing indicators of ecological condition in the Willamette Basin: an overview of the Oregon prepilot study for EPA's EMAP program *in* River Quality, Dynamics and Restoration. A Laenen and D.A. Dunnette (eds.), Lewis Publishers, CRC Press, Boca Raton, pp. 275-282.

Herlihy, A.T., Stoddard, J.L., and Johnson, C.B., 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic Region, U.S. Water, Air, and Soil Pollution 105, 377-386, 1998.

Hingston, F.J. and Gailitis, V., 1976. The geographic variation of salt precipitated over Western Australia. Australian Journal of Soil Research 14, 319-335.

Howarth, R.W., Boyer, E.W., Pabich, W.J., and Galloway, J.N., 2002. Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31, 88-96.

Kirchner, J.W., Feng, X., and Neal, C., 2001. Catchment-scale advection and dispersion as a mechanism for fractal scaling in stream tracer concentrations. Journal of Hydrology 254, 82-101.

Li, Y., 1992. Seasalt and pollution inputs over the continental United States. Water, Air, and Soil Pollution 64, 561-573.

Littlewood, I.G., Croke, B.F., Jakeman, A.J., and Sivapalan, M., 2003. The role of 'top-down' regressioning for Prediction in Ungauged Basins (PUB). Hydrological Processes 17, 1673-1679.

McDonnell, J.J., and Woods, R., 2004. On the need for catchment classification. Journal of Hydrology 299, 2-3.

McHale, M.R., McDonnell, J.J., Mitchell, M.J., and Cirmo, C.P., 2002. A field-based study of soil water and groundwater nitrate release in an Andirondack forested watershed. Water Resources Research 38, No. 4, 1-17.

Moser, T.J., Wigington, P.J., Schuft, M.J., Kaufmann, P.R., Herlihy, A.T., Van Sickle, J., and McAllister, L.S., 1997. The effect of riparian areas on the ecological condition of small, perennial stream in agricultural landscapes of the Willamette Valley. Research Plan, Environmental Protection Agency, EPA/600/R-97/074.

Neal, C. and Rosier, P.T.W., 1990. Chemical studies of chloride and stable oxygen isotopes in two conifer afforested and moorland sites in the British Uplands. Journal of Hydrololgy 115, 269-283.

Nyberg, L., Rodhe, A., and Bishop, K., 1999. Water transit times and flow paths from two line injections of ^{3}H and ^{36}Cl in a microcatchment at Gardsjon, Sweden. Hydrological Processes 13, 1557-1575.

Omernik, J.M., and Griffith, G.E., 1991. Ecological regions versus hydrologic units: frameworks for managing water quality. Journal of Soil and Water Conservation 46, 334-340.

Pimentel, D. (ed.), 1993. World soil erosion and conservation. Cambridge University Press.

Peck, D.V., Averill, D.K., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, M., McCormick, F.H., Peterson, S.A., Cappaert, M.R., Magee, T., and Monaco, A., 2005a. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for non-wadeable rivers and streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Washington, D.C.

Peck, D.V., Herlihy, A.T., Hill, B.H., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, J.M., McCormick, S.A., Peterson, S.A., Ringold, P.L., Magee, T., and Cappaert, M.R., 2005b. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for wadeable streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Preston, S.D., 2000. Statistical identification of hydrochemical response units for hydrologic monitoring and regressioning in Maryland. USGS Water-Resources Invesitigations Report 00-4232.

SAS Institute, Inc., 2003. Getting Started with SAS. Cary, North Carolina, U.S.A.

Sivapalan, M., 2003. Prediction in ungauged basins: a grand challenge for theoretical hydrology. Hydrological Processes 17, 3163-3170.

Sivapalan, M., Takeuchi, K., Franks, S.W., Gupta, V.K., Karambiri, H., Lakshmi, V., Liang, X., McDonnell, J.J., Mendiondo, E.M., O'Connell, P.E., Oki, T., Pomeroy, J.W., Schertzer, D., Uhlenbrook, S., and Zehe, E., 2003. IAHS Decade on Predictions in Ungauged Basins (PUB), 2003-2012: Shaping an exciting future for the hydrological sciences. Hydrological Sciences Journal 48, 857-880.

Smart, R.P., Soulsby, C., Neal, C., Wade, A., Cresser, M.S., Billett, M.F., Langan, S.J., Edwards, A.C., Jarvie, H.P., and Owen, R., 1998. Factors regulating the spatial and temporal distribution of solute concentrations in a major river system in NE Scotland. The Science of the Total Environment 221, 93-110.

Stoddard, J.L, Peck, D.V., Olsen, A.R., Larson, D.P., Van Sickle, J., Hawkins, C.P., Hughes, R.M., Whittier, T.R., Lomnicky, G., Herlihy, A.T., Kaufman, P.R., Peterson, S.A., Ringold, P.L., Paulsen, S.G., and Blair, R., 2005. Environmental Monitoring and Assessment Program (EMAP): Western Streams and Rivers Statistical Summary. EPA 620/R-05/XXX, U.S. Environmental Protection Agency, Washington, D.C.

United States Department of Agriculture Soil Conservation Service, 1967. Hydrology *in* USDA National Engineering Handbook. Berkeley, California.

Van Leeuwen, E.P., Draaijers, G.P.J., and Erisman, J.W., 1996. Mapping wet deposition of acidifying components and base cations over Europe using measurements. Atmospheric Environment 20, 2495-2511.

Winter, T.S., 2001. The concept of hydrologic landscapes. Journal of the American Water Resources Association 37, 335-349.

Wolock, D.M., Winter, T.C., and McMahon, G., 2004. Delineation and evaluation of hydrologic landscape regions in the United States using geographic information system tools and multivariate statistical analyses. Environmental Management 34, Suppl. 1, S71-S88.

Chapter 5

Summary and Conclusions

This dissertation focused on how land use affects nitrate concentrations on both the catchment scale and regional scale. A temporally-intensive study was first conducted on the catchment scale to determine how land use affects processes controlling nitrate dynamics. The majority of studies to date on the processes controlling nitrate dynamics from catchments have been conducted in mostly pristine catchments, creating a need to study different types of land use. Chapter 2 describes the analysis of 3 neighboring headwater catchments in western Oregon with similar (low) atmospheric deposition, size, and geology but with different, consistent land use expressions. This analysis revealed that human activity altered the patterns of stream nitrate concentrations during storm events in the agricultural catchment to a larger extent compared to the residential catchment. Nitrate response patterns in the residential catchment were the same as the patterns in the reference forested catchment (a "concentration" pattern throughout the year), whereas a "dilution" pattern was observed in the fall and winter and a "concentration" pattern was observed in the spring in the agricultural catchment. Manure and green bean application in the agricultural catchment significantly increased nitrate concentrations and exports in the fall, which decreased throughout the year as the source became depleted. This is in contrast to the relatively constant export rates in the forested and residential catchments, which likely had a more constant source of nitrate (i.e., no large source inputs). Streamflow in the forested, agricultural, and residential catchments moved from groundwater-dominated to rainfall-dominated as the rainy period progressed. Additional streamflow sources were identified in the agricultural catchment, which may include a groundwater seep and soil pipe. Although the exports in the residential catchment were high, the patterns of nitrate concentrations in the residential catchment were minimally impacted compared to the agricultural and forested catchments.

Chapter 3 describes an alternative approach to understanding processes controlling nitrate through statistical modeling. Improved prediction of nitrate concentrations using mathematical representations of processes occurring within the catchment would indicate that these processes are accurately represented. The statistical model incorporates process-based knowledge (TI) into linear regressions, creating a link between statistical modeling and process literature. The identification of hot spots with TI did not significantly improve nitrate predictions. This may be due to the source of streamwater; summer low flows are likely from a deep groundwater source that is largely unaffected by activity occurring on the surface. Although denitrification and other transformation processes can occur in groundwater aquifers, results suggest that nitrate in the groundwater undergoes relatively little transformation (in hot spots) as it travels to the stream. In-stream, out-of-stream, and total inverse-distance and inverse-distance squared calculations provided no improvement in nitrate predictions. This is likely due to the similarity between %landuse and $1/d$ and $1/d^2$ landuse. Different catchment settings, where there is a significant difference between %landuse and inverse-distance weighted land use, may improve model predictions. Models also revealed that the dominance of one explanatory variable (forested land use in this case) can significantly affect model performance. The models created from the Northern California dataset, which is dominated by forest land cover (average 99.79%), did not predict nitrate as well as the models created from the Western Oregon dataset.

One surprising finding in Chapter 3 was that site elevation was the most significant predictor of nitrate and chloride concentrations. In Western Oregon, chloride was successfully predicted using land use and site elevation. Site elevation is likely representing regional rainout/orographic effects due to the similarities between precipitation and elevation variations, which is more strongly linked to chloride than nitrate. Correlations between nitrate/chloride and nitrate/distance to the coast indicate that stream nitrate concentrations may be controlled somewhat by chloride deposition. More work needs to be done to determine the mechanism involved. Stream chloride concentrations were elevated in agricultural areas in Western Oregon, which differed from the observed rainout/orographic pattern of atmospheric chloride deposition. This may render chloride an inadequate tracer of water. Nonetheless, the use of TI and inverse-distance weighting significantly improved chloride models, indicating that the hydrologic setting of the catchment (identification of saturated areas in the case of TI and flowpaths in the case of inverse-distance weighting) is represented. Nitrate models were not significantly improved using TI and inverse-distance weighting, which indicates that transformation processes may be a more significant control of nitrate.

Future work is needed to determine whether or not distance calculations (specifically in-stream, out-of-stream, and total inverse-distance and inverse-distance squared) and TI can improve the prediction of stream nitrate concentrations. The type of catchment setting that provides a significant difference between %landuse and 1/d or $1/d^2$ land use needs to be identified to further investigate the utility of inverse-distance weighting. This study has shown that nitrate traveling in the deeper groundwater (the source of streamwater during lowflow) is not significantly affected by the identification of hot spots using TI. Hot spots at the interface between the groundwater and riparian water near the stream evidently do not significantly transform groundwater nitrate before it reaches the stream, or groundwater may bypass the riparian area altogether (so there is no interface for a hot spot). The saturated areas (and hot spots) that are formed from shallow flowpaths during storm events may be better identified by TI than the hot spots at the interface between groundwater and riparian water. Alternatively, TI may not be adequately identifying these hot spots. Topographic index does aid in the prediction of chloride, which indicates that TI is identifying saturation conditions within the catchment. More work, possibly with a different index and/or using TI to predict nitrate using storm samples, is needed.

Chapter 4 incorporates the hydrologic landscape region classification scheme into empirical regressions, which is another way to create a link between statistical modeling and process literature. Hydrologic landscape regions were correlated with nitrate and chloride for the Western Oregon dataset, but not with nitrate and chloride for the Northern California dataset. Regression results revealed that HLRs via primary hydrologic flowpaths significantly improved chloride predictions, but did not improve nitrate predictions. Improved predictions of chloride compared to nitrate are likely due to the conservative nature of chloride, which creates a tighter link between chloride and the hydrology of the catchment (with the caveat that poor nitrate regression performance may also be due to low variability of nitrate concentrations). Poor predictions occurred using the Northern California dataset, whereas HLR and land use predicted chloride relatively well for the Western Oregon dataset. This may be due to the lack of variation in primary hydrologic flowpaths and land use in the Northern California dataset; nitrate and chloride variation in Northern California cannot be explained by these variables.

Future work is needed to determine whether or not HLR can improve the prediction of stream nitrate and chloride concentrations. Hydrologic landscape regions identify areas of similar hydrologic settings, where similar water quality characteristics may occur. Hydrologic landscape regions may be more suitable for predicting water quality parameters when the catchment is saturated or "wetting up" during storm events. Sampling during wet conditions will show whether or not HLR can improve predictions under different conditions. Regression results were relatively successful in one region (Western Oregon), whereas regression results in the other region (Northern California) were very poor. Testing of the hydrologic landscape regions in other areas of the United States may provide some insights into its suitability for predicting water quality in ungauged basins.

Agricultural effects were seen both on the smaller catchment scale (Oak Creek) and the regional scale (Western Oregon). Nitrate dynamics during storm events in the agricultural catchment were significantly different than the dynamics in the forested and residential catchments. Significant summer nitrate inputs and the lack of significant riparian vegetation likely caused the changing patterns of nitrate concentrations (a dilution pattern during the fall and winter storms and a concentration pattern in the spring). On the regional scale, chloride was elevated in the valley where agricultural activity is concentrated.

The temporal pattern of nitrate during storm events was found to be largely controlled by the spatial organization of land cover. Nitrate dynamics or patterns during storm events in the residential catchment, with its marshy area in the lower portion of the catchment and significant riparian vegetation, were not significantly impacted. Land use impacts on nitrate dynamics may not be mitigated as well if the marshy area were farther upstream or vegetation near the stream was located elsewhere within the catchment. In contrast, nitrate dynamics in the agricultural catchment, with its limited riparian vegetation and small water storage, was significantly impacted. On the regional scale and during lowflow, the spatial pattern of land cover did not control stream nitrate concentrations sufficiently to improve predictions of nitrate. This may indicate that spatial pattern of land cover may be significant during storm events when areas within the catchment become saturated and hydrologic connectivity occurs, but not during lowflow when deeper groundwater (which is not affected by surface activities) is the source of streamwater.

Bibliography

Ahearn, D.S., Sheibley, R.W., Dahlgren, R.A., and Keller, K.E., 2004. Temporal dynamics of stream water chemistry in the last free-flowing river draining the western Sierra Nevada, California. Journal of Hydrology 295, 47-63.

American Society of Agricultural Engineers, 2003. Manure Production and Characteristics, ASAE D384.1 FEB03, ASAE Standards. American Society of Agricultural Engineers.

Arheimer, B. and Liden, R., 2000. Nitrogen and phosphorous concentrations from agricultural catchments – influence of spatial and temporal variables. Journal of Hydrology 227, 140-159.

Baker, J.P., Landers, D.H., Lee, H., Ringold, P.L., Sumner, R.R., Wigington, P.J., Bennett, R.S., Preston, E.M., Frick, W.E., Sigleo, A.C., Specht, D.T., and Young, D.R., 1995. Ecosystem management research in the Pacific Northwest: five-year research strategy. EPA/600/R-95/069, U.S. EPA, Corvallis, Oregon, 251 pp.

Baker, M.E., Wiley, M.J., and Seelbach, P.W., 2001. GIS-based hydrologic modeling of riparian areas: implications for stream water quality. Journal of the American Water Resources Association 37, 1615-1627.

Beven, K.J. and Kirkby, 1979. A physically based, variable contributing area model of basin hydrology. Hydrological Sciences 24, 43-69.

Bolstad, P.V., and Swank, W.T., 1997. Cumulative impacts of landuse on water quality in a Southern Appalachian watershed. Journal of the American Water Resources Association 33, 519-533.

Buck, O., Niyogi, D.K., and Townsend, C.R., 2004. Scale-dependence of land use effects on water quality of streams in agricultural catchments. Environmental Pollution 130, 287-299.

Burnham, K.P., and Anderson, D.R., 1998. Model selection and inference, a practical information-theoretic approach. Springer-Verlag New York Inc..

Burns, D., 2005. What do hydrologists mean when they use the term flushing? Hydrological Processes 19, 1325-1327.

Burns, D.A., Murdoch, P.S., Lawrence, G.B., and Michel, R.L., 1998. Effect of groundwater springs on NO_3^- concentrations during summer in Catskill Mountain streams. Water Resources Research 34, 1987-1996.

Burns, D., Vitvar, T., McDonnell, J., Hassett, J., Duncan, J., Kendall, C., 2005. Effects of suburban development on runoff generation in the Croton River basin, New York, USA. Journal of Hydrology 311, 266-281.

Canham, C.D., Pace, M.L., Papaik, M.J., Primack, A.G.B., Roy, K.M., Maranger, R.J., Curran, R.P., and Spada, D.M., 2004. A spatially explicit watershed-scale analysis of dissolved organic carbon in Adirondack Lakes. Ecological Applications 13, 839-854.

Carratala, A., Gomez, A., and Bellot, J., 1998. Mapping rain composition in the East of Spain by applying kriging. Water, Air, and Soil Pollution 104, 9-27.

Chapman, T., 1987. Classification of regions *in* Comparative Hydrology: An ecological approach to land and water resources, M. Falkenmark and T. Chapman eds. Unesco, Stockholm, pp. 50-74.

Christensen, S., Simkins, S., and Teidje, J.M., 1990. Spatial variation of denitrification: dependency of activity centers on the soil environment. Soil Science Society of American Journal 54, 1608-1613.

Christophersen, N., and Hooper, R.P., 1992. Multivariate analysis of stream water chemical data: the use of principal components analysis for the end-member mixing problem. Water Resources Research 28, 99-107.

Clarke, R.T., 2005. The PUB decade: how should it evolve? Hydrological Processes 19, 2865-2869.

Clesceri, L.S., Greenberg, A.E., and Eaton, A.D. (eds.), 1998. Standard Methods for the Examination of Water and Wastewater, 20th Edition. American Public Health Association, American Water Works Association, Water Environment Federation, Washington, D.C..

Cook, P.G., Jolly, I.D., Leaney, F.W., Walker, G.R., Allan, G.L., Fifield, L.K., and Allison, G.B., 1994. Unsaturated zone tritium and chlorine 36 profiles from southern Australia: their use as tracers of soil water movement. Water Resources Research 30, 1709-1719.

Creed, I.F. and Band, L.E., 1998. Export of nitrogen from catchments within a temperate forest: Evidence for a unifying mechanism regulated by variable source area dynamics. Water Resources Research 11, 3105-3120.

Creed, I.F., Band, L.E., Foster, N.W., Morrison, I.K., Nicolson, J.A., Semkin, R.S., and Jeffries, D.S., 1996. Regulation of nitrate-N release from temperate forests: A test of the N flushing hypothesis. Water Resources Research 32, 3337-3354.

Dahm, C.N., Grimm, N.B., Marmonier, P., Valett, H.M., Vervier, P., 1998. Nutrient dynamics at the interface between surface waters and groundwaters. Freshwater Biology 40, 427-451.

Donner, S.D., Kucharik, C.J., and Foley, J.A., 2004. Impact of changing land use practices on nitrate export by the Mississippi River. Global Biogeochemical Cycles 18, GB1028, 1-21.

Fowler, D., Cape, J.N., Leith, I.D., Choularton, T.W., Gay, M.J., and Jones, A., 1988. The influence of altitude on rainfall composition at Great Dun Fell. Atmospheric Environment 22, 1355-1362.

Gordon, N.D., McMahon, T.A., Finlayson, B.L., 1992. Stream Hydrology: an introduction for ecologists. Wiley, New York, pp. 163-167.

Gustafsson, M.E.R. and Larsson, E.H., 2000. Spatial and temporal patterns of chloride deposition in Southern Sweden. Water, Air, and Soil Pollution 124, 345-369.

Heisig, P.M., 2000. Effects of residential and agricultural land uses on the chemical quality of baseflow of small streams in the Croton Watershed, Southeastern New York, U.S. Geological Survey Water-Resources Investigations Report 99-4173, 16p.

Herlihy, A., Kaufmann, P., Reynolds, L., Li, J., and Robison, G., 1997. Developing indicators of ecological condition in the Willamette Basin: an overview of the Oregon prepilot study for EPA's EMAP program, *in* River Quality, Dynamics and Restoration, A Laenen and D.A. Dunnette (eds.). Lewis Publishers, CRC Press, Boca Raton, pp. 275-282.

Herlihy, A.T., Stoddard, J.L., and Johnson, C.B., 1998. The relationship between stream chemistry and watershed land cover data in the Mid-Atlantic Region, U.S. Water, Air, and Soil Pollution 105, 377-386.

Hill, A.R., Devito, K.J., Campagnolo, S., and Sanmugades, K., 2000. Subsurface denitrification in a forest riparian zone: interactions between hydrology and supplies of nitrate and organic carbon. Biogeochemistry 51, 193-223.

Hingston, F.J. and Gailitis, V., 1976. The geographic variation of salt precipitated over Western Australia. Australian Journal of Soil Research 14, 319-335.

Hinkle, S.R., Duff, J.H., Triska, F.J., Laenen, A., Gates, E.B., Bencala, K.E., Wentz, D.A., Silva, S.R., 2001. Linking hyporheic flow and nitrogen cycling near the Willamette River – a large river in Oregon, U.S.A. Journal of Hydrology 244, 157-180.

Hjerdt, K.N., McDonnell, J.J., Seibert, J., and Rodhe, A., 2004. A new topographic index to quantify downslope controls on local drainage. Water Resources Research 40, W05602, 1-6.

Hooper, R.P., Christophersen, N., and Peters, N., 1990. Modelling streamwater chemistry as a mixture of soilwater end-members – an application to the Panola Mountain catchment, Georgia, U.S.A. Journal of Hydrology 116, 321-343.

Hornberger, G.M., Bencala, K.E., and McKnight, D.E., 1994. Hydrological controls on dissolved organic carbon during snowmelt in the Snake River near Montezuma, Colorado. Biogeochemistry 25, 147-165.

Howarth, R.W., Boyer, E.W., Pabich, W.J., and Galloway, J.N., 2002. Nitrogen use in the United States from 1961-2000 and potential future trends. Ambio 31, 88-96.

Hulse, D.W., Gregory, S.V., and Baker, J.P., 2002. Willamette River basin planning atlas: trajectories of environmental and ecological change. Oregon State University Press, Corvallis, Oregon, USA.

Hunsaker, C.T., and Levine, D.A., 1995. Hierarchical approaches to the study of water quality in rivers. Bioscience 45, 193-203.

Inamdar, S.P., Christopher, S.F., and Mitchell, M.J., 2004. Export mechanisms for dissolved organic carbon and nitrate during summer storm events in a glaciated forested catchment in New York, USA. Hydrological Processes 18, 2651-2661.

Johnson, L.B., Richards, C., Host, G.E., and Arthur, J.W., 1997. Landscape influences on water chemistry in Midwestern stream ecosystems. Freshwater Biology 37, 193-208.

Jones, K.B., Neale, A.C., Nash, M.S., Van Remortel, R.D., Wickham, J.D., Riitters, K.H., and O'Neill, R.V., 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: a multiple watershed study from the United States Mid-Atlantic Region. Landscape Ecology 16, 301-312.

Jordan, T.E., and Weller, D.E., 1996. Human contributions to terrestrial nitrogen flux. Bioscience 46, 655-663.

Jordan, T.E., Correll, D.L., Weller, D.E., 1997. Relating nutrient discharges from watersheds to land use and streamflow variability. Water Resources Research 33, 2579-2590.

Kehmeier, J.W., 2000. A spatially explicit method for determining the effects of watershed scale land use on stream conditions. Master's Thesis, Oregon State University.

King, R.S., Baker, M.E., Whigham, D.F., Weller, D.E., Jordan, T.E., Kazyak, P.F., and Hurd, M.K., 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15, 137-153.

Kirchner, J.W., Feng, X., and Neal, C., 2001. Catchment-scale advection and dispersion as a mechanism for fractal scaling in stream tracer concentrations. Journal of Hydrology 254, 82-101.

Lattin, P.D., Wigington, P.J., Moser, T.J., Peniston, B.E., Lindeman, D.R., and Oetter, D.R., 2004. Influence of remote sensing imagery source on quantification of riparian land cover/land use. Journal of the American Water Resources Association 40, 215-227.

Li, Y., 1992. Seasalt and pollution inputs over the continental United States. Water, Air, and Soil Pollution 64, 561-573.

Little, J.L., Saffran, K.A., and Fent, L., 2003. Land use and water quality relationships in the lower Little Bow River watershed, Alberta, Canada. Water Quality Research Journal of Canada 38, 563-584.

Littlewood, I.G., Croke, B.F., Jakeman, A.J., and Sivapalan, M., 2003. The role of 'top-down' regressioning for Prediction in Ungauged Basins (PUB). Hydrological Processes 17, 1673-1679.

Lowrance, R., Todd, R., Fail J. Jr., Hendrickson, O. Jr., Leonard, R., 1984. Riparian forests as nutrient filters in agricultural watersheds. Bioscience 34, 374-377.

McClain, M.E., Boyer, M.E., Dent, C.L., Gergel, S.E., Grimm, N.B., Groffman, P.M., Hart, S.C., Harvey, J.W., Johnston, C.A., Mayorga, E., McDowell, W.H., and Pinay, G., 2003. Biogeochemical hot spots and hot moments at the interface of terrestrial and aquatic ecosystems. Ecosystems 6, 301-312.

McDonnell, J.J., and Woods, R., 2004. On the need for catchment classification. Journal of Hydrology 299, 2-3.

McHale, M.R., McDonnell, J.J., Mitchell, M.J., and Cirmo, C.P., 2002. A field-based study of soil water and groundwater nitrate release in an Andirondack forested watershed. Water Resources Research 38, 1031, 1-17.

Mitchell, M.J., Driscoll, C.T., Inamdar, S., McGee, G.G., Mbila, M.O., and Raynal, D.J., 2003. Nitrogen biogeochemistry in the Adirondack Mountains of New York: hardwood ecosystems and associated surface waters. Environmental Pollution 123, 355-364.

Moore, J.A., and Gamroth, M.J., 1993. Calculating the fertilizer value of manure from livestock operations. OSU Extension Service, Corvallis, Oregon.

Moser, T.J., Wigington, P.J., Schuft, M.J., Kaufmann, P.R., Herlihy, A.T., Van Sickle, J., and McAllister, L.S., 1997. The effect of riparian areas on the ecological condition of small, perennial stream in agricultural landscapes of Western Oregon, Research Plan. Environmental Protection Agency, EPA/600/R-97/074.

Mosley, M.P., 1979. Streamflow generation in a forested watershed, New Zealand. Water Resources Research 15, 795-806.

Mosteller, F. and Tukey, J.W., 1997. Data analysis and regression, a second course in statistics. Addison-Wesley, Reading, Massachusetts.

Mulder, J., Christophersen, N., Hauhs, M., Vogt, R.D., Andersen, S., and Andersen, D.O., 1990. Water flow paths and hydrochemical controls in the Birkenes catchment as inferred from rainstorm high in seasalts. Water Resources Research 26, 611-622.

National Research Council, 2000. Clean Coastal Waters: Understanding and reducing the effects of nutrient pollution. National Academy Press, Washington D.C..

National Science and Technology Council, 2000. Integrated assessment of hypoxia in the northern Gulf of Mexico. Committee on Environment and Natural Resources, Washington D.C.

Neal, C. and Rosier, P.T.W., 1990. Chemical studies of chloride and stable oxygen isotopes in two conifer afforested and moorland sites in the British Uplands. Journal of Hydrology 115, 269-283.

Norton, M.M. and Fisher, T.R., 2000. The effects of forest on stream water quality in two coastal plain watersheds of the Chesapeake Bay. Ecological Engineering 14, 337-362.

Nyberg, L., Rodhe, A., and Bishop, K., 1999. Water transit times and flow paths from two line injections of ^{3}H and ^{36}Cl in a microcatchment at Gardsjon, Sweden. Hydrological Processes 13, 1557-1575.

Oetter, D. R., W. B. Cohen, M. Berterretche, T. K. Maiersperger, and R. E. Kennedy. 2000. Land cover mapping in an agricultural setting using multiseasonal Thematic Mapper data. Remote Sensing of Environment 76, 139–155.

Omernik, J.M., and Griffith, G.E., 1991. Ecological regions versus hydrologic units: frameworks for managing water quality. Journal of Soil and Water Conservation 46, 334-340.

Owens, L.B., Edwards, W.M., and Keuren, R.W., 1991. Baseflow and stormflow transport of nutrients from mixed agricultural watersheds. Journal of Environmental Quality 20, 407-414.

Pan, Y, Herlihy, A., Kaufman, P., Wigington, J., Van Sickle, J., and Moser, T., 2004. Linkages among land-use, water quality, physical habitat conditions and lotic diatom assemblages: a multi-scale assessment. Hydrolobiologica 515, 59-73.

Pater, D.E., Bryce, S.A., Thorson, T.D., Kagen, J., Chappell, C., Omernik, J.M., Azevedo, S.H., and Woods, A.J., 1998. Ecoregions of Western Washington and Oregon (two-sided, color poster with map). United States Geological Survey, Reston, Virginia.

Peck, D.V., Averill, D.K., Herlihy, A.T., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, M., McCormick, F.H., Peterson, S.A., Cappaert, M.R., Magee, T., and Monaco, A., 2005a. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for non-wadeable rivers and streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Washington, D.C.

Peck, D.V., Herlihy, A.T., Hill, B.H., Hughes, R.M., Kaufmann, P.R., Klemm, D.J., Lazorchack, J.M., McCormick, S.A., Peterson, S.A., Ringold, P.L., Magee, T., and Cappaert, M.R., 2005b. Environmental monitoring and assessment program – surface waters western pilot study: field operations manual for wadeable streams. EPA 600/R-05/xxx, U.S. Environmental Protection Agency, Office of Research and Development, Washington, D.C.

Peterjohn, W.T. and Correll, D.L., 1984. Nutrient dynamics in an agricultural watershed: observations on the role of a riparian forest. Ecology 65, 1466-1475.

Petry, J., Soulsby, C., Malcolm, I.A., and Youngson, A.F., 2002. Hydrological controls on nutrient concentrations and fluxes in agricultural catchments. The Science of the Total Environment 294, 95-110.

Pimentel, D. (ed.), 1993. World soil erosion and conservation. Cambridge University Press, Cambridge, Massachusetts.

Pinay, G., Decamps, H., Arles, C., and Lacassin-Seres, M., 1989. Topographic influence on carbon and nitrogen dynamics in riverine woods. Archive fur Hydrobiologie 114, 401-414.

Pionke, H.B. and Urban, J.B., 1985. Effect of agricultural land use on ground-water quality in a small Pennsylvania watershed. Ground Water 23, 80-98.

Preston, S.D., 2000. Statistical identification of hydrochemical response units for hydrologic monitoring and modeling in Maryland. USGS Water-Resources Invesitigations Report 00-4232.

Rabalais, N.N., Turner, R.E., and Scavia, D., 2002. Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. Bioscience 52, 129-141.

Rademacher, L.K., Clark, J.F., Clow, D.W., and Hudson, G.B., 2005. Old groundwater influence on stream hydrochemistry and catchment response times in a small Sierra Nevada catchment: Sagehen Creek, California. Water Resources Research 41, WO2004, 1-10.

Salvia-Castellvi, M., Iffly, J.F., Borght, P.V., and Hoffman, L., 2005. Dissolved and particulate nutrient export from rural catchments: a case study from Luxembourg. Science of the Total Environment 344, 51-65.

SAS Institute, Inc., 2003. Getting Started with SAS. Cary, North Carolina, U.S.A.

Schilling, K.E., 2002. Chemical transport from paired agricultural and restored prairie watersheds. Journal of Environmental Quality 31, 1184-1193.

Sebilo, M., Billen, G., Grably, M., and Mariott, A., 2003. Isotopic composition of nitrate-nitrogen as a marker of riparian and benthic denitrification at the scale of the whole Seine River system. Biogeochemistry 63, 35-51.

Sivapalan, M., 2003. Prediction in ungauged basins: a grand challenge for theoretical hydrology. Hydrological Processes 17, 3163-3170.

Sivapalan, M., Takeuchi, K., Franks, S.W., Gupta, V.K., Karambiri, H., Lakshmi, V., Liang, X., McDonnell, J.J., Mendiondo, E.M., O'Connell, P.E., Oki, T., Pomeroy, J.W., Schertzer, D., Uhlenbrook, S., and Zehe, E., 2003. IAHS Decade on Predictions in Ungauged Basins (PUB), 2003-2012: Shaping an exciting future for the hydrological sciences. Hydrological Sciences Journal 48, 857-880.

Smart, R.P., Soulsby, C., Neal, C., Wade, A., Cresser, M.S., Billett, M.F., Langan, S.J., Edwards, A.C., Jarvie, H.P., and Owen, R., 1998. Factors regulating the spatial and temporal distribution of solute concentrations in a major river system in NE Scotland. The Science of the Total Environment 221, 93-110.

Smith, R.A., Schwarz, G.A., and Alexander, R.B., 1997. Regional interpretations of water-quality monitoring data. Water Resources Research 33, 2781-2798.

Soil Conservation Service, 1975. Soil Survey of Benton County Area, Oregon. USDA.

Soranno, P.A., Hubler, S.L., Carpenter, S.R., and Lathrop, R.C., 1996. Phosphorous loads to surface waters: a simple model to account for spatial pattern of land use. Ecological Applications 6, 865-878.

Stoddard, J.L, Peck, D.V., Olsen, A.R., Larson, D.P., Van Sickle, J., Hawkins, C.P., Hughes, R.M., Whittier, T.R., Lomnicky, G., Herlihy, A.T., Kaufman, P.R., Peterson, S.A., Ringold, P.L., Paulsen, S.G., and Blair, R., 2005. Environmental Monitoring and Assessment Program (EMAP): Western Streams and Rivers Statistical Summary. EPA 620/R-05/XXX, U.S. Environmental Protection Agency, Washington, D.C.

Sylvia, D.M., Fuhrmann, J.J., Hartel, P.G., and Zuberer, D.A., 1998. Principles and Applications of Soil Microbiology. Prentice Hall.

United States Department of Agriculture Soil Conservation Service, 1967. Hydrology *in* USDA National Engineering Handbook. Berkeley, California.

Van Leeuwen, E.P., Draaijers, G.P.J., and Erisman, J.W., 1996. Mapping wet deposition of acidifying components and base cations over Europe using measurements. Atmospheric Environment 20, 2495-2511.

Vanni, M.J., Renwick, W.H., Headworth, J.L., Auch, J.D., and Schaus, M.H., 2001. Dissolved and particulate nutrient flux from three adjacent agricultural watersheds: a five-year study. Biogeochemistry 54, 85-114.

Van Sickle, J., Baker, J., Herlihy A., Bayley, P., Gergory, S., Haggerty, P., Ashkenas, L., and Li, Judith, 2004. Projecting the biological condition of streams under alternative scenarios of human land use. Ecological Applications 14, 368-380.

Vitvar, T., Burns, D.A., Lawrence, G.B., McDonnell, J.J., and Wolock, D.M., 2002. Estimation of baseflow residence times in watersheds from the runoff hydrograph recession: method and application in the Neversink watershed, Catskill Mountains, New York. Hydrological Processes 16, 1871-1877.

Vogelmann, J.E., Howard, S.M., Yang, L., Larson, C.R., Wylie, B.K., and Van Driel, N., 2001. Completion of the 1990s National Land Cover Data Set for the Conterminous United States from Landsat Thematic Mapper Data and Ancillary Data Sources. Photogrammetric Engineering and Remote Sensing 67, 650-662.

Wayland, K.G., Long, D.T., Hyndman, D.W., Pijanowski, B.C., Woodhams, S.M., and Haack, S.K., 2003. Identifying relationships between baseflow geochemistry and land use with synoptic sampling and r-mode factor analysis. Journal of Environmental Quality 32, 180-190.

Webb, B.W., and Walling, D.E., 1985. Nitrate behaviour in streamflow from a grassland catchment in Devon, U.K.. Water Resources 19, 1005-1016.

Wellington, B.I. and Driscoll, C.T., 2004. The episodic acidification of a stream with elevated concentrations of dissolved organic carbon. Hydrological Processes 18, 2663-2680.

Welsch, D.L., Kroll, C.N., McDonnell, J.J., and Burns, D.A., 2001. Topographic control on the chemistry of subsurface stormflow. Hydrological Processes 15, 1925-1938.

Wernick, B.G., Cook, K.E., and Schreier, H., 1998. Land use and streamwater nitrate dynamics in an urban-rural fringe watershed. Journal of the American Water Resources Association 34, 639-650.

Wickham, J.D., O'Neill, R.V., Riitters, K.H., Smith, E.R., Wade, T.G., and Jones, K.B., 2002. Geographic targeting of increases in nutrient export due to future urbanization. Ecological Applications 12, 93-106.

Wickham JD, Stehman SV, Smith JH, and Yang L. 2004. Thematic accuracy of MRLC-NLCD land cover for the western United States. Remote Sensing of Environment 91, 452-468.

Wigington, P.J., Baker, J.P., DeWalle, D.R., Kretser, W.A., Murdoch, P.S., Simonin, H.A., Van Sickle, J., McDowell, M.K., Peck, D.V., Barchet, W.R., 1996a. Episodic acidification of small streams in the Northeastern United States: episodic response project. Ecological Applications 6, 374-388.

Wigington, P.J., DeWalle, D.R., Murdoch, P.S., Kretser, W.A., Simonin, H.A., Van Sickle, J., and Baker, J.P., 1996b. Episodic acidification of small streams in the Northeastern United States: ionic controls of episodes. Ecological Applications 6, 389-407.

Winter, T.S., 2001. The concept of hydrologic landscapes. Journal of the American Water Resources Association 37, 335-349.

Woli, K.P., Nagumo, T., Kuramochi, K., and Hatano, R., 2004. Evaluating river water quality through land use analysis and N budget approaches in livestock farming areas. Science of the Total Environment 329, 61-74.

Wolock, D.M., Winter, T.C., and McMahon, G., 2004. Delineation and evaluation of hydrologic-landscape regions in the United States using geographic information system tools and multivariate statistical analyses. Environmental Management 34, Suppl. 1, S71-S88.

Wondzell, S.M., and Swanson, F.J., 1999. Floods, channel change, and the hyporheic zone. Water Resources Research 35, 555-567.

Appendix

Appendix A

Datasets used for Linear Regressions

STRM_ID	Nitrate (ueq/L)	logNitrate	Chloride	log Cl	Valley Site	HLR	OF-DGW	SGW	XSLOPE	XELEV	F LU instream SUM(1/d)
AR06	1.285	0.108903128	4038.563	3.606226862	1	9	1	0	0.0888889	275	0.573999
AR08	5.354	0.728678367	289.986	2.462377031	1	9	1	0	0.113	260	4.232568
AR13	164.561	2.216326918	717.222	2.855653603	1	9	1	0	0.161	200	3.764964
AR14	3.141	0.497067936	3488.659	3.542658521	1	mixed			0.29	200	2.918564
AR16	0.785	-0.105130343	384.335	2.584709936	1	9	1	0	0.19	200	3.469332
AR22	108.16	2.034066679	101.767	2.007606972	1	9	1	0	0.33	265	1.116508
AR23	36.982	1.567990394	53.761	1.730467338	1	mixed			1.06	380	2.388007
AR27	27.772	1.443607157	46.681	1.669140151	1	16	0	1	1.46	320	5.650085
AR28	2.07	0.315970345	952.855	2.979026817	1	9	1	0	0.1	170	6.158718
AR31	1.499	0.175801633	662.503	2.821187849	1	9	1	0	0.27	140	3.390751
AR32	6.425	0.807873132	324.566	2.511303023	1	mixed			0.12	150	2.152301
AR34	99.308	1.996984236	242.007	2.383827928	1	9	1	0	0.477	100	1.335015
AR37	428.786	2.632240597	1208.119	3.082109714	1	mixed			0.123	130	1.665014
AR38	195.26	2.290613285	222.179	2.346703008	1	9	1	0	0.812	80	2.410079
AR40	548.869	2.739468703	275.065	2.439435333	1	9	1	0	0.29	100	3.271997
AR46	1.856	0.268577972	855.403	2.932170769	1	9	1	0	0.454	165	2.575553
AR47	7.568	0.878981123	705.178	2.848298755	1	9	1	0	0.263	150	5.481608
AR49	7.71	0.887054378	673.39	2.828266663	1	9	1	0	0.99	120	8.433628
OR0035	7.853	0.895035597	115.645	2.06312686	1	11	0	0	1.1111111	560	17.175789
OR0055	4.284	0.631849462	87.439	1.941705182	0	16	0	1	9.6666667	1270	3.741226
OR0075	19.99	1.300812794			0	19	0	1	1	430	5.379436
OR0095	0.714	-0.146301788	138.209	2.140536325	1	11	0	0	1.4148148	280	5.925029
OR7905	10.709	1.029748919	2820.6	3.450341502	1	16	0	1	1.0555556	470	7.628138
OR8235	4.998	0.698796252	90.259	1.955490518	0	19	0	1	1.125	360	12.840698
ORC01	0	-3	17.798	1.250371202	0	19	0	1	1.83425	1470	14.672888
ORC02	0.7	-0.15490196	21	1.322219295	0	19	0	1	3.85	1315	11.378997
ORC03	0	-3	15	1.176091259	0	19	0	1	7.912	2750	17.746061
ORC04	1.6	0.204119983	31	1.491361694	0	19	0	1	3.4	1190	4.607985
ORC05	2.1	0.322219295	19	1.278753601	0	19	0	1	1.57	1110	14.265885
ORC06	8.567	0.932828767	15.006	1.176264942	0	19	0	1	9.75	2520	13.11901
ORC07	5.7	0.755874856	17	1.230448921	0	19	0	1	10.15	1780	8.315907
ORC08	0.8	-0.096910013	21	1.322219295	0	19	0	1	6.3	3510	6.732928
ORC09	0	-3	16.134	1.207742053	0	19	0	1	13.3	2640	13.099044
ORC10	17.134	1.233858763	13.172	1.119651722	0	19	0	1	9.8	3600	10.98594
ORC28	0	-3	21	1.322219295	0	16	0	1	8.2415556	3280	3.461816
ORC32	0	-3	17	1.230448921	0	19	0	1	3.6111111	3460	10.724606
ORC36	0	-3	20.788	1.317812708	0	19	0	1	6.945	1200	3.173661
ORC40	0	-3	19.462	1.289187468	0	19	0	1	8.2	2080	3.165075
ORC43	0	-3	9.646	0.984347258	0	19	0	1	1.575	3230	25.072987
ORC46A	0	-3	27.557	1.440231936	0	19	0	1	11.8	2160	10.847807
ORST97-013	25.987	1.414756146	408.028	2.610689967	1	9	1	0	0.55	105	3.88513
ORST97-023	4.069	0.60948769	111.075	2.045616322	0	16	0	1	0.96	540	13.139116

Table A1. Western Oregon Dataset used for Linear Regressions.

STRM_ID	Nitrate (ueq/L)	logNitrate	Chloride	log Cl	Valley Site	HLR	OF-DGW	SGW	XSLOPE	XELEV	F LU instream SUM(1/d)
ORST97-032	0	-3	17.149	1.2342388	0	19	0	1	7.789	3960	5.852964
ORST97-035	2.213	0.344981414	30.97	1.490941205	0	19	0	1			8.086283
ORST97-304	1.285	0.108903128	37.232	1.570916366	0	16	0	1			38.185123
ORST97-305	0	-3	16.388	1.214525955	0	16	0	1	9.2	1580	7.39307
ORST97-307	0.214	-0.669586227	17.939	1.25379823	0	16	0	1	3.6	2370	5.55119
ORST97-309	0.785	-0.105130343	12.157	1.084826417	0	16	0	1	15	2700	6.858321
ORST97-310	1.428	0.154728207	28.432	1.45380741	0	16	0	1	4.79	1440	7.246092
ORST97-422	6.64	0.822168079	64.874	1.812070676	0	19	0	1	10.8	1710	4.567722
ORST97-424	116.299	2.06557598	169.292	2.228636436	1	9	1	0	0.163	195	0.598754
ORST97-425	52.474	1.719944171	60.615	1.782580109	1	9	1	0	3.4925	310	2.915865
ORST97-428	12.422	1.094191525	230.471	2.362616286	1	16	0	1	0.55	115	6.847821
ORST97-431	2.07	0.315970345	101.062	2.004587889	0	mixed					12.247591
ORST97-432	3.498	0.543819805	118.014	2.071933531	0	19	0	1	1.0375	770	5.621232
ORST97-434	2.213	0.344981414	19.462	1.289187468	0	19	0	1			30.05537
ORST97-435	17.206	1.235679919	52.153	1.717279295	0	9	1	0	13.7	1420	2.5351
ORST97-436	27.772	1.443607157	38.304	1.583244129	0	19	0	1	0.1	1190	6.403546
ORST97-440	0.643	-0.191789027	11.113	1.045831314	0	19	0	1	4.67	2480	9.622324
ORST97-441	2.07	0.315970345	9.28	0.967547976	0	20	0	1	14.03	3980	9.068167
ORST97-443	0.286	-0.543633967	13.341	1.125188384	0	20	0	1	4.385	3000	3.893172
ORST97-444	0.286	-0.543633967	20.252	1.306467919	0	19	0	1			19.75477
ORST97-445	1.856	0.268577972	11.141	1.046924174	0	19	0	1	13.56	3540	6.487093
ORST97-448	9.567	0.980775774	101.288	2.005557996	1	9	1	0	3.025	360	3.790991
ORST97-451	3.284	0.516403148	33.396	1.523694452	0	19	0	1	3.23	620	10.744524
ORST97-460	0.643	-0.191789027	38.699	1.587699743	0	mixed					34.264236
ORST97-461	5.997	0.777934049	38.586	1.58642976	0	mixed			8.9	1800	13.453437
ORST97-462	2.356	0.372175286	30.124	1.478912639	0	16	0	1	13.333333	1440	4.929804
ORST97-464	5.783	0.762153192	53.112	1.725192656	0	16	0	1	2.55	670	5.421547
ORV01	2.3	0.361727836	64	1.806179974	1	3	1	0	0.15	280	0.042842
ORV02	19.9	1.298853076	59	1.770852012	1	19	0	1	0	270	5.616872
ORV03	5.4	0.73239376	124	2.093421685	1	mixed			0.28	252	10.761968
ORV04	25.2	1.401400541	47	1.672097858	1	19	0	1	0.265	332	3.600328
ORV08	6	0.77815125	98	1.991226076	1	mixed			0.42499	385	8.677629
ORV20	496.181	2.69564013	105.547	2.023445894	1	19	0	1	0.1	523	0.800123
ORV21	6	0.77815125	199	2.298853076	1	mixed			1.0375	235	2.105021
ORV22	67.4	1.828659897	116	2.064457989	1	9	1	0	0.2	336	0.21247
ORV25	12.6	1.100370545	637	2.804139432	1	9	1	0	0	410	1.060132
ORV27	2.6	0.414973348	188	2.274157849	1	19	0	1	0.87	328	9.229014
ORV28	0	-3	89.159	1.950165189	1	9	1	0	0.285	235	6.135762
ORV30	33.769	1.5285182	116.434	2.066079818	1	9	1	0	0.55	440	0.347098
ORV31	70.965	1.851044207	118.775	2.074725039	1	mixed			0.85	205	0.253319

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	F LU outstream SUM(1/d)	F LU total SUM(1/d)	F LU instream SUM(1/d2)	F LU outstream SUM(1/d2)	F LU total SUM(1/d2)	F LU area (m2)	F LU length(m)
AR06	12.649866	0.532004	0.004603	0.137215	0.004565	3744000	41065772
AR08	162.974625	3.924257	0.000603	1.841623	0.000408	41009400	568892096
AR13	68.432213	2.207906	0.094351	0.795816	0.005914	16514100	203815616
AR14	43.023308	2.409876	0.023078	0.535747	0.007884	10354500	77515176
AR16	87.386726	3.080567	0.004512	1.021161	0.002074	22464000	231423968
AR22	41.940098	0.943273	0.004454	0.622335	0.001471	6802200	91336264
AR23	72.321785	1.74304	0.0153	1.183597	0.00153	10197900	105118360
AR27	178.445145	4.441555	0.048265	2.738385	0.004583	27466200	359837248
AR28	227.987488	5.28739	0.027195	3.035704	0.001552	43669800	551354688
AR31	58.806702	2.824321	0.018029	0.777344	0.007729	12785400	100621192
AR32	16.562279	0.81828	0.084545	0.341411	0.006822	1818000	13554528
AR34	27.628624	1.166615	0.007862	0.549319	0.00667	3376800	23840264
AR37	23.157583	1.151083	0.018416	0.446254	0.002857	3275100	24569708
AR38	34.757893	1.549033	0.032224	0.68021	0.003498	3701700	27032282
AR40	77.97937	2.567806	0.037874	1.483876	0.008318	9117000	83858648
AR46	40.26453	2.103724	0.01148	0.540288	0.001801	7468200	40167052
AR47	90.256287	4.020838	0.077259	1.127483	0.009865	19161000	159161552
AR49	90.688255	5.137809	0.123592	1.308652	0.010393	13550400	65230836
OR003S	300.267761	12.72224	0.13657	3.817722	0.012224	58828500	462262976
OR005S	4.60626	1.733455	0.03953	0.051898	0.009219	1042200	1080310
OR007S	29.238756	3.680859	0.042505	0.418846	0.010836	4637700	9955565
OR009S	11.941216	3.02677	0.040667	0.132363	0.009894	2628900	3533347
OR790S	102.748169	6.143599	0.010941	1.165564	0.008119	22706100	152572944
OR823S	289.102783	9.976834	0.054495	3.741296	0.011996	58342500	657799872
ORC01	284.653992	10.876713	0.046308	3.494323	0.011066	62069400	656574528
ORC02	66.32061	6.778218	0.140539	0.889259	0.012858	12219300	38125052
ORC03	337.013275	9.744173	0.428826	4.36266	0.012172	70917408	1058648832
ORC04	79.102898	3.782357	0.036913	1.137856	0.009031	13774500	88705840
ORC05	165.188889	9.081038	0.137178	2.141106	0.012047	31494600	197549248
ORC06	24.392706	4.365714	0.181204	0.266212	0.011357	5652900	13234300
ORC07	14.10805	3.503721	0.096014	0.17649	0.012181	2990700	4581078
ORC08	24.727875	3.284199	0.090261	0.336084	0.011808	4518900	14802860
ORC09	14.711508	3.402439	0.536725	0.155405	0.010434	3819600	7723121
ORC10	22.335823	4.748909	0.185875	0.249866	0.012114	5048100	8952511
ORC28	13.814625	2.145322	0.050477	0.178679	0.008906	2623500	5939833
ORC32	252.066025	9.039868	0.049986	3.097872	0.003955	55164600	582413056
ORC36	8.264432	1.870582	0.036376	0.103516	0.008969	1836000	3595522
ORC40	12.726727	2.159528	0.024815	0.166072	0.008235	2717100	6073909
ORC43	105.583923	9.157369	0.469795	1.171939	0.014113	27162900	162197312
ORC46A	17.808308	3.90539	0.307528	0.198324	0.01172	4581900	9555145
ORST97-013	44.063255	2.721929	0.071988	0.734795	0.009896	6991200	32283644
ORST97-023	110.68399	7.845942	0.137716	1.439008	0.007382	21189600	97194016

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	F LU outstream SUM(1/d)	F LU total SUM(1/d)	F LU instream SUM(1/d2)	F LU outstream SUM(1/d2)	F LU total SUM(1/d2)	F LU area (m2)	F LU length(m)
ORST97-032	12.15355	3.079814	0.05768	0.117463	0.009385	3407400	6237743
ORST97-035	27.010042	3.262292	0.255554	0.313116	0.010818	5676300	24673540
ORST97-304	603.693481	21.571354	0.83005	7.820423	0.016191	120008352	1151900928
ORST97-305	28.760269	3.886356	0.082246	0.357593	0.011945	5596200	15583322
ORST97-307	40.442612	4.374573	0.020947	0.524668	0.009524	8038800	24596910
ORST97-309	39.327103	4.646275	0.017917	0.468331	0.009364	8353800	27679752
ORST97-310	61.993992	5.347757	0.053816	0.808251	0.010425	12369600	49423684
ORST97-422	12.168064	2.798199	0.025168	0.131843	0.009168	2853000	4798016
ORST97-424	2.359455	0.43682	0.004265	0.048714	0.00209	293400	560080
ORST97-425	3.842595	1.380339	0.0433	0.049173	0.009698	685800	650548
ORST97-428	453.7612	6.495026	0.001729	5.815877	0.001054	93524384	2030616192
ORST97-431	790.918457	11.446281	0.00939	10.369838	0.007092	156690592	3532822016
ORST97-432	9.554836	2.576314	0.06925	0.124146	0.010921	1701900	1865497
ORST97-434	901.955322	24.490389	0.086865	11.10241	0.014819	194680992	3114374400
ORST97-435	3.357352	1.283095	0.031364	0.040876	0.008547	703800	714704
ORST97-436	2.834064	1.756191	0.135986	0.026122	0.010902	727200	515943
ORST97-440	42.017555	6.061381	0.067522	0.496035	0.012584	9221400	23873680
ORST97-441	1.590244	1.330628	0.145543	0.009412	0.008682	590400	438128
ORST97-443	30.229914	2.249706	0.035984	0.389382	0.008098	5661900	30365414
ORST97-444	1145.932007	17.767323	0.027758	14.814095	0.009869	228676128	6289888768
ORST97-445	16.421371	3.529486	0.097616	0.179814	0.010379	3735000	6751034
ORST97-448	7.438447	2.156627	0.025099	0.089582	0.009601	1500300	1843269
ORST97-451	170.741135	8.058847	0.054975	2.403961	0.010778	27648900	176548624
ORST97-460	1974.000244	28.520355	0.086228	24.407932	0.011701	434102048	11748742144
ORST97-461	13.877474	3.001842	0.629562	0.156338	0.01136	2993400	5945026
ORST97-462	7.784486	2.326098	0.034185	0.07868	0.009297	1934100	2771111
ORST97-464	52.520481	3.913703	0.02198	0.646872	0.002811	11859300	59886340
ORV01	1.424047	0.031139	0.000087	0.029585	0.000015	186300	2048062
ORV02	237.926483	4.579369	0.035839	3.196663	0.001713	43894800	689323648
ORV03	532.699646	9.121789	0.014795	6.685001	0.002834	111324320	2166657024
ORV04	127.644981	3.332399	0.007785	1.605061	0.006231	31306500	431764960
ORV08	88.02829	6.572261	0.057602	1.131612	0.011487	17156700	79791056
ORV20	0.779939	0.382246	0.00805	0.006372	0.00553	217800	272294
ORV21	7.514781	1.342453	0.003521	0.088699	0.001335	1672200	2964201
ORV22	0.747474	0.151797	0.000423	0.010572	0.000259	146700	258605
ORV25	19.179108	0.910078	0.00468	0.175569	0.000332	5648400	46901012
ORV27	125.708275	7.458584	0.026819	1.592586	0.009486	25183800	139120384
ORV28	244.681641	5.691973	0.014601	3.159039	0.006015	47045700	541906816
ORV30	3.703493	0.307799	0.004641	0.035726	0.004586	1362600	13124049
ORV31	2.159583	0.180091	0.001072	0.034635	0.000352	288000	1073664

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	A LU instream SUM(1/d)	A LU outstream SUM(1/d)	A LU total SUM(1/d)	A LU instream SUM(1/d2)	A LU outstream SUM(1/d2)	A LU total SUM(1/d2)
AR06	13.971877	205.148407	10.620805	0.069713	2.623649	0.00704
AR08	8.036767	129.841202	6.288532	0.04274	1.721146	0.01038
AR13	17.098824	85.764519	5.376448	0.663814	1.160468	0.005546
AR14	7.725984	101.234642	6.110409	0.034766	1.424735	0.003544
AR16	11.637462	132.962891	7.882822	0.026346	1.703607	0.008098
AR22	4.649443	81.220818	3.587484	0.028979	0.988449	0.008742
AR23	9.239553	96.311829	5.037683	0.083869	1.097062	0.008458
AR27	5.467101	113.703499	4.419543	0.022709	1.140234	0.005992
AR28	20.67487	214.350403	9.258715	0.630413	2.904454	0.011829
AR31	8.20049	87.746544	5.800507	0.031377	1.088625	0.003358
AR32	6.340415	116.199242	4.121513	0.089688	1.5494	0.003517
AR34	3.909557	63.676376	3.372374	0.002452	0.69775	0.001407
AR37	9.672623	151.178009	7.224508	0.032158	2.076288	0.009285
AR38	3.505853	74.08036	2.96007	0.008259	0.898205	0.006226
AR40	10.480569	202.019226	7.391773	0.111337	2.788798	0.004191
AR46	6.741691	65.602928	4.644691	0.075276	0.864516	0.0035
AR47	7.678098	93.369492	4.460641	0.157198	1.319133	0.003697
AR49	2.888251	15.584256	1.684664	0.014918	0.147926	0.001788
OR003S	1.078434	20.660332	0.808833	0.010364	0.325931	0.000329
OR005S	0	0	0	0	0	0
OR007S	0.065372	0.334758	0.052239	0.000154	0.001539	0.000073
OR009S	1.714108	2.868042	0.86111	0.006775	0.036622	0.001544
OR790S	0.859419	10.907464	0.68837	0.001539	0.195979	0.000733
OR823S	0.973655	2.570829	0.567412	0.003925	0.025715	0.000793
ORC01	0	0	0	0	0	0
ORC02	0	0	0	0	0	0
ORC03	0	0	0	0	0	0
ORC04	0	0	0	0	0	0
ORC05	0	0	0	0	0	0
ORC06	0	0	0	0	0	0
ORC07	0	0	0	0	0	0
ORC08	0	0	0	0	0	0
ORC09	0	0	0	0	0	0
ORC10	0	0	0	0	0	0
ORC28	0	0	0	0	0	0
ORC32	0	0	0	0	0	0
ORC36	0	0	0	0	0	0
ORC40	0	0	0	0	0	0
ORC43	0	0	0	0	0	0
ORC46A	0	0	0	0	0	0
ORST97-013	6.831793	30.835945	2.901445	0.2283	0.392745	0.003627
ORST97-023	0.183887	0.203507	0.11418	0.0094	0.002186	0.00502

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	A LU instream SUM(1/d)	A LU outstream SUM(1/d)	A LU total SUM(1/d)	A LU instream SUM(1/d2)	A LU outstream SUM(1/d2)	A LU total SUM(1/d2)
ORST97-032	0	0	0	0	0	0
ORST97-035	0	0	0	0	0	0
ORST97-304	0	0	0	0	0	0
ORST97-305	0	0	0	0	0	0
ORST97-307	0	0	0	0	0	0
ORST97-309	0	0	0	0	0	0
ORST97-310	0	0	0	0	0	0
ORST97-422	0	0	0	0	0	0
ORST97-424	3.908518	18.419592	2.390675	0.038342	0.273432	0.008547
ORST97-425	1.516299	1.590166	0.617931	0.0062	0.011049	0.000893
ORST97-428	15.361913	247.525009	9.680095	0.231856	3.125823	0.010953
ORST97-431	8.310455	221.916748	7.005923	0.006283	2.974455	0.002554
ORST97-432	0.002012	0.033333	0.001898	0.000004	0.001111	0.000004
ORST97-434	0.000702	0.05314	0.00069	0	0.00131	0
ORST97-435	0.066667	0.020769	0.015782	0.001481	0.000147	0.000084
ORST97-436	0	0	0	0	0	0
ORST97-440	0	0	0	0	0	0
ORST97-441	0	0	0	0	0	0
ORST97-443	0.000972	0.106421	0.00096	0	0.002607	0
ORST97-444	4.17649	14.75846	1.816997	0.066973	0.204188	0.003125
ORST97-445	0	0	0	0	0	0
ORST97-448	0.015012	0.036723	0.006336	0.000042	0.00079	0.000007
ORST97-451	0.056833	0.10716	0.047315	0.000482	0.002472	0.000444
ORST97-460	5.876566	107.474594	3.804836	0.064448	1.438603	0.002648
ORST97-461	0	0	0	0	0	0
ORST97-462	0	0	0	0	0	0
ORST97-464	0.933192	2.076406	0.48683	0.016893	0.028258	0.005505
ORV01	10.784227	112.219101	6.546759	0.088949	1.44405	0.01272
ORV02	12.029656	140.648788	6.771548	0.154707	1.710208	0.010987
ORV03	15.650928	205.620956	7.905671	0.329024	2.687204	0.009029
ORV04	4.787558	84.149208	3.950408	0.017898	1.070062	0.003034
ORV08	0.250811	3.475503	0.227212	0.000209	0.064545	0.00017
ORV20	2.347029	3.645399	0.985447	0.012407	0.048102	0.002108
ORV21	1.897687	3.35721	1.030222	0.022365	0.048827	0.007786
ORV22	0.877261	2.165215	0.600323	0.007872	0.033782	0.006322
ORV25	0.436821	5.230099	0.240739	0.009349	0.082014	0.000189
ORV27	1.168814	6.047187	0.768539	0.007092	0.076612	0.001035
ORV28	3.263353	41.129475	2.077379	0.024993	0.57517	0.001733
ORV30	0.339494	2.410886	0.260819	0.005787	0.031305	0.001657
ORV31	1.787928	31.864998	1.51414	0.000744	0.407129	0.000396

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	A LU area (m2)	A LU length(m)	U LU instream SUM(1/d)	U LU outstream SUM(1/d)	U LU total SUM(1/d)	U LU instream SUM(1/d2)	U LU outstream SUM(1/d2)
AR06	45265500	326383648	0.380177	4.748669	0.285277	0.001557	0.052993
AR08	26270100	199220832	0.186937	4.443627	0.159901	0.000093	0.05784
AR13	16342200	113189904	1.784649	27.552071	0.955902	0.049195	0.286487
AR14	16826400	76503672	0.963043	11.846248	0.79908	0.00084	0.136886
AR16	28665900	180035600	0.261638	4.306212	0.21398	0.000114	0.047348
AR22	16611300	179923936	0.178085	4.136244	0.162326	0.000042	0.041734
AR23	22598100	207078112	0.62199	5.278806	0.371921	0.005832	0.059612
AR27	28024200	324808480	0.175811	4.19244	0.127338	0.000271	0.051017
AR28	40005000	344688160	0.820739	16.42367	0.530048	0.013668	0.200182
AR31	20448900	124421976	0.421337	5.852782	0.338621	0.000388	0.064795
AR32	21267000	199360176	0.432484	6.560999	0.321323	0.001459	0.073698
AR34	14410800	101902560	0.691276	7.216035	0.482412	0.000877	0.090559
AR37	28215000	207981120	0.954848	12.421452	0.674716	0.001894	0.166591
AR38	15924600	154273840	0.618126	6.600455	0.346312	0.008771	0.070121
AR40	35179200	318732032	1.109082	37.358097	0.969346	0.000761	0.525838
AR46	11942100	53254632	0.963538	8.90826	0.685226	0.019199	0.127406
AR47	15883200	103183376	0.775334	10.588049	0.549314	0.005421	0.130683
AR49	4779900	26426900	0.655903	3.847349	0.317443	0.00403	0.054443
OR003S	2736000	14034016	0.742057	9.753315	0.350194	0.019922	0.186162
OR005S	0	0	0	0	0	0	0
OR007S	107100	347259	0.000378	0.002123	0.000321	0	0.000005
OR009S	714600	935407	0.282331	0.338326	0.125359	0.00173	0.002909
OR790S	1171800	4524350	0.227698	2.916596	0.174126	0.000483	0.048303
OR823S	612900	1051504	0.125914	0.67617	0.074168	0.000716	0.011078
ORC01	0	0	0.069048	0.126471	0.048552	0.000971	0.003707
ORC02	0	0	0	0	0	0	0
ORC03	0	0	0	0	0	0	0
ORC04	0	0	0	0	0	0	0
ORC05	0	0	0.001648	0.002993	0.001056	0.000002	0.000005
ORC06	0	0	0	0	0	0	0
ORC07	0	0	0	0	0	0	0
ORC08	0	0	0	0	0	0	0
ORC09	0	0	0	0	0	0	0
ORC10	0	0	0	0	0	0	0
ORC28	0	0	0	0	0	0	0
ORC32	0	0	0.155874	0.041802	0.109575	0.0092	0.001121
ORC36	0	0	0	0	0	0	0
ORC40	0	0	0	0	0	0	0
ORC43	0	0	0.773981	3.081576	0.25746	0.021691	0.026987
ORC46A	0	0	0	0	0	0	0
ORST97-013	6215400	25824474	1.927875	16.384033	1.187223	0.029808	0.22364
ORST97-023	33300	140043	0.602286	3.39722	0.203158	0.0207	0.063199

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	A LU area (m2)	A LU length(m)	U LU instream SUM(1/d)	U LU outstream SUM(1/d)	U LU total SUM(1/d)	U LU instream SUM(1/d2)	U LU outstream SUM(1/d2)
ORST97-032	0	0	0	0	0	0	0
ORST97-035	0	0	0	0	0	0	0
ORST97-304	0	0	0	0	0	0	0
ORST97-305	0	0	0	0	0	0	0
ORST97-307	0	0	0.000929	0.008333	0.000836	0.000001	0.000069
ORST97-309	0	0	0	0	0	0	0
ORST97-310	0	0	0	0	0	0	0
ORST97-422	0	0	0	0	0	0	0
ORST97-424	2986200	6990371	0.32119	1.707426	0.21325	0.000965	0.023782
ORST97-425	506700	607563	0.278831	0.580077	0.155817	0.000707	0.005316
ORST97-428	51489900	580284800	0.97988	13.78951	0.425519	0.035738	0.188058
ORST97-431	41383800	459323328	0.71418	33.707638	0.60561	0.0005	0.46206
ORST97-432	900	527	0	0	0	0	0
ORST97-434	2700	13111	0.230637	1.564981	0.127505	0.000872	0.018802
ORST97-435	2700	576	0	0	0	0	0
ORST97-436	0	0	0	0	0	0	0
ORST97-440	0	0	0	0	0	0	0
ORST97-441	0	0	0	0	0	0	0
ORST97-443	4500	26035	0.006766	0.323803	0.006441	0.000001	0.006856
ORST97-444	2347200	7184944	0.469038	7.932207	0.291339	0.00588	0.173775
ORST97-445	0	0	0	0	0	0	0
ORST97-448	5400	5965	0	0	0	0	0
ORST97-451	16200	19712	0.047926	0.241432	0.04046	0.000173	0.006244
ORST97-460	19530900	245831440	0.699954	21.401461	0.49516	0.005468	0.327034
ORST97-461	0	0	0	0	0	0	0
ORST97-462	0	0	0	0	0	0	0
ORST97-464	316800	447065	0.172257	0.195017	0.088144	0.005659	0.003114
ORV01	25866900	253691024	0.30326	2.853247	0.180894	0.001016	0.036375
ORV02	30042900	280048128	0.352209	10.035665	0.312602	0.000201	0.097075
ORV03	43773300	452633184	0.504264	18.917166	0.449366	0.00015	0.272328
ORV04	17866800	158808144	0.319005	7.016938	0.280587	0.000211	0.100649
ORV08	378900	1104099	0.039834	0.848286	0.037212	0.000028	0.017675
ORV20	786600	1100846	0.137008	0.154251	0.060688	0.000572	0.001678
ORV21	502200	494989	0.057001	0.050081	0.023608	0.000231	0.000207
ORV22	383400	565776	1.761074	8.782524	1.214237	0.002825	0.112525
ORV25	718200	3802176	11.47281	74.232552	5.440887	0.31576	0.935917
ORV27	877500	2002525	0.061988	0.787176	0.048417	0.000084	0.013253
ORV28	7641000	48446084	0.136734	4.981974	0.117617	0.000118	0.075663
ORV30	591300	3913307	4.320551	34.330811	2.859012	0.016249	0.421168
ORV31	7938000	60206896	5.887898	36.765663	4.132163	0.026729	0.471683

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	U LU total SUM(1/d2)	U LU area (m2)	U LU length(m)	TI instream SUM(1/d)	TI outstream SUM(1/d)	TI IS*OS 1/d	TI total SUM(1/d)	TI instream SUM(1/d2)
AR06	0.000461	1062000	7554824	6.544621562	7.041821135	46.0860544	6.820830559	4.474022885
AR08	0.000052	901800	8299687	9.491490289	9.478691842	89.9669116	9.504870877	9.83554737
AR13	0.000209	6853500	71251528	9.647195619	9.829881764	94.8307923	9.269816192	10.29359678
AR14	0.000404	2915100	15541250	8.118779893	9.060545953	73.5605783	8.36604165	6.420949492
AR16	0.000053	1129500	8890986	10.10153372	9.923925246	100.246866	9.9235924	12.4188622
AR22	0.000033	970200	9268781	9.307886438	9.849961257	91.6823208	9.621838807	9.114264376
AR23	0.000329	1018800	6814452	9.688611512	9.031678949	87.5044286	9.380309701	9.761180952
AR27	0.00006	777600	9810229	7.293609739	8.093535698	59.0310908	7.525820814	5.737532108
AR28	0.000164	3177900	31473572	8.628301428	8.922886669	76.9893558	9.33661882	7.43448467
AR31	0.000129	1325700	8344459	9.216044698	9.651649665	88.9500347	9.351598503	8.010382986
AR32	0.000207	1355400	11873075	8.367531903	9.620637347	80.5009899	9.144044537	6.998645364
AR34	0.000307	1571400	10531783	10.00977886	10.12162203	101.315198	10.109419	7.526715511
AR37	0.000367	2508300	17574596	9.297683269	9.596562629	89.2257998	9.484014944	8.354070177
AR38	0.000571	1432800	13427247	8.703061176	9.415524415	81.943885	9.182050107	7.161675431
AR40	0.000293	6362100	74147192	8.879083171	9.042422291	80.2884196	9.208956374	7.563317642
AR46	0.006185	1520100	7408919	7.846524277	8.152702956	63.9703817	7.830163686	8.674899722
AR47	0.000265	1998000	12937144	8.339990746	8.483771201	70.7545733	8.289322621	9.265181739
AR49	0.000249	659700	2588563	6.932137282	7.287894004	50.5206817	7.259477594	6.676890697
OR003S	0.000228	941400	4918751	7.071320854	7.58543889	53.6390722	7.306144453	6.122747404
OR005S	0	0	0	6.240789778	6.615907803	41.2884898	6.515161695	6.007932746
OR007S	0	900	3116	6.447702785	6.959543595	44.8730686	6.742189452	5.848393974
OR009S	0.000355	95400	129228	6.435609473	7.093006199	45.6478179	6.826726492	5.761551289
OR790S	0.000172	335700	1482428	6.979298377	7.721100229	53.8878623	7.142314716	7.25688498
OR823S	0.000087	91800	185240	7.622244784	7.628949821	58.149723	7.741523264	8.053279451
ORC01	0.000463	5400	903	6.105990589	7.211330541	44.0323164	6.431735118	5.831916244
ORC02	0	0	0	5.627742668	6.133128179	34.5156671	5.93833453	5.173390827
ORC03	0	0	0	6.285717576	8.540385339	53.6824502	7.141998144	5.103600978
ORC04	0	0	0	6.803048936	6.998371825	47.610266	6.937848543	7.514209856
ORC05	0.000001	1800	4371	5.960919958	6.397346839	38.1340724	6.172248734	5.76247649
ORC06	0	0	0	5.526240592	6.541634705	36.1506472	6.006536377	5.251007141
ORC07	0	0	0	5.563867717	5.906580496	32.8634325	5.813558713	5.291113796
ORC08	0	0	0	6.096091243	7.149920171	43.5865657	6.529225347	5.420586736
ORC09	0	0	0	5.841613381	7.326671921	42.7995847	6.596993628	5.468684847
ORC10	0	0	0	5.938854587	7.025068245	41.7208588	6.551131763	5.13044718
ORC28	0	0	0	6.656292779	7.237238152	48.1731761	6.914723132	6.709398154
ORC32	0.005247	9000	15713	7.599072603	8.800592915	66.8763445	7.813528569	5.853160322
ORC36	0	0	0	6.009417494	6.893108123	41.4235645	6.415794212	5.57870574
ORC40	0	0	0	5.886847047	6.683232369	39.3431667	6.177026183	4.581785211
ORC43	0.000345	518400	2764580	7.016703667	7.693012589	53.9795896	7.38035719	6.570569373
ORC46A	0	0	0	6.282965706	7.186145488	45.1503057	6.798636091	5.908434056
ORST97-013	0.001182	2736000	10492091	7.886119281	8.089373949	63.7937679	8.105159218	7.701861585
ORST97-023	0.000431	319500	1218141	6.942372688	7.508946147	52.1299026	7.128037716	7.171628273

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	U LU total SUM(1/d2)	U LU area (m2)	U LU length(m)	TI instream SUM(1/d)	TI outstream SUM(1/d)	TI IS*OS 1/d	TI total SUM(1/d)	TI instream SUM(1/d2)
ORST97-032	0	0	0	7.467340459	8.409498166	62.7965859	7.799176253	7.141683422
ORST97-035	0	0	0	7.113870318	7.963459902	56.651021	7.603934775	7.141520546
ORST97-304	0	0	0	6.279019173	6.961459125	43.7111353	6.55196728	5.823992319
ORST97-305	0	0	0	5.813393653	6.352419567	36.9291156	6.123989639	5.726980923
ORST97-307	0.000001	900	1196	7.159865135	7.833438004	56.0863597	7.350897041	6.057573877
ORST97-309	0	0	0	6.054795258	7.024748713	42.5334152	6.318135231	6.245255637
ORST97-310	0	0	0	6.442233916	7.226113785	46.5523153	6.773425325	5.551108386
ORST97-422	0	0	0	5.874576519	6.791639299	39.8980047	6.147958147	4.727070859
ORST97-424	0.000211	284400	584809	8.729864443	9.357671153	81.6912007	9.090462131	7.252885961
ORST97-425	0.000185	136800	169136	6.939528409	7.272379184	50.466882	7.257357413	6.553300402
ORST97-428	0.000337	2540700	29915350	8.735806232	8.342429127	72.8778444	8.62638957	8.762854406
ORST97-431	0.000122	5736600	84363344	8.359989563	8.244369668	68.9228444	8.39316223	8.78173499
ORST97-432	0	0	0	5.793183473	6.443446083	37.3280654	6.288542334	5.492866838
ORST97-434	0.000091	262800	1126659	6.838232267	7.003890763	47.8942318	6.832767296	7.947908355
ORST97-435	0	0	0	6.81316205	7.563697818	51.5326989	7.107985674	6.014552761
ORST97-436	0	0	0	6.218365014	6.887190894	42.8270669	6.726448163	5.91065191
ORST97-440	0	0	0	6.212408702	7.186197268	44.6435944	6.617649208	4.789686477
ORST97-441	0	0	0	5.674158552	6.23567815	35.3822265	5.971780242	5.633524113
ORST97-443	0.000001	31500	190494	6.429612023	6.666184551	42.8609803	6.694851959	6.674910384
ORST97-444	0.000778	616500	3929873	7.223096775	7.025131199	50.7432025	7.191946551	8.04407029
ORST97-445	0	0	0	6.040806272	7.265390463	43.8888163	6.540352136	4.797412336
ORST97-448	0	0	0	6.53097609	7.159945575	46.7614334	6.815684233	5.973469631
ORST97-451	0.000135	15300	10746	7.01747828	7.179072395	50.3789846	7.189384244	6.866364666
ORST97-460	0.000178	3401100	49601560	7.948570456	8.048391314	63.9732054	7.889067679	9.467716112
ORST97-461	0	0	0	6.550662714	7.761324799	50.841821	7.290927994	6.356560593
ORST97-462	0	0	0	6.439604109	7.299335397	47.0048302	6.738422252	5.713316171
ORST97-464	0.001086	21600	24745	7.363504187	7.641950464	56.2715342	7.505737114	6.64025869
ORV01	0.000129	640800	4998256	11.59263186	12.24072356	141.902202	11.83178468	10.73810105
ORV02	0.000059	2464200	28059332	9.101594383	9.007083031	81.9788163	9.181659684	8.741208296
ORV03	0.000072	3863700	50420344	11.23897216	8.967768758	100.788503	10.04123239	14.00855886
ORV04	0.000124	1565100	13854197	9.825567207	9.760905221	95.9064303	9.806833321	8.694176257
ORV08	0.000025	63900	171747	6.73504042	7.481341944	50.3871404	6.963779791	7.382621925
ORV20	0.000114	44100	63700	9.128392078	9.646358362	88.0557413	9.398122348	8.979361834
ORV21	0.000034	21600	30463	7.045075545	7.51434164	52.9391045	7.257532819	7.107163368
ORV22	0.001165	1629900	3344688	8.736808597	9.569756128	83.6091276	8.917422888	7.976261128
ORV25	0.010417	15642000	85633408	9.278317522	9.378802566	87.0195082	9.674991679	8.492382985
ORV27	0.00004	82800	258116	7.197410956	7.502501121	53.9985838	7.255590969	8.164078127
ORV28	0.000023	701100	5787447	8.845279752	8.262771028	73.0865213	8.303078136	13.45380606
ORV30	0.00243	9410400	59698036	11.61613639	11.74101976	136.385287	11.65191896	8.061443299
ORV31	0.010085	7776900	27376016	10.89697476	11.75811557	128.127889	11.18935308	8.731107452

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	TI outstream SUM(1/d2)	TI IS*OS 1/d2	TI total SUM(1/d2)	TI area (m2)	TI length(m)
AR06	7.6767231	34.34583483	6.098593907	6.841328796	6.58620526
AR08	10.67987317	105.0423985	12.35350554	8.487123302	7.86682268
AR13	11.20434048	115.3329631	11.5225812	8.676351933	8.23941909
AR14	10.15061479	65.17658489	8.124239351	8.131130383	7.92894779
AR16	11.26797014	139.9353685	14.52513202	8.875779286	8.29720174
AR22	10.91068967	99.44291018	12.83232481	9.168641346	8.77678386
AR23	9.930773346	96.93607563	9.827629666	8.517938891	8.19363666
AR27	8.970212328	51.46688125	6.604946398	7.574104287	7.47960302
AR28	10.07142919	74.8758859	15.90351395	8.104525248	7.57953838
AR31	10.5924526	84.84960208	7.895328103	8.978515625	8.50439792
AR32	10.04477525	70.29981971	7.837584147	9.614633967	9.64854403
AR34	9.862454285	74.23188764	7.472861744	10.89060902	11.0481865
AR37	9.911925865	82.80492427	8.79894476	9.615576027	9.67297172
AR38	9.385242322	67.21405935	9.121394581	10.07906321	10.4875004
AR40	9.350937379	70.72410965	12.02366448	9.150638691	9.00198449
AR46	9.338913262	81.01413605	8.681466005	7.283066735	7.05910353
AR47	9.661418594	89.51479913	13.96216998	7.499198212	7.2559902
AR49	8.119769958	54.2148165	11.46632875	6.872085308	6.79877831
OR003S	8.768497641	53.68729617	8.403566401	6.710005289	6.46156509
OR005S	7.294318051	43.82377228	6.754218953	6.335553705	6.27666623
OR007S	7.686502643	44.95369574	7.265201465	6.461896777	6.30661492
OR009S	8.048453998	46.37158051	6.787434289	6.501177702	6.31091227
OR790S	8.925033407	64.76794088	7.746592798	6.861990782	6.73421771
OR823S	8.514787409	68.57196247	13.1886939	7.036285585	6.93906817
ORC01	8.452499622	49.29426984	8.017521034	6.315069883	6.28032491
ORC02	6.817584496	35.27002909	6.78867543	5.696177359	5.56723297
ORC03	9.772280679	49.87382123	6.494290643	7.533967684	7.69055646
ORC04	7.625145997	57.2969472	11.09544901	6.657693564	6.58592234
ORC05	7.398694245	42.63480165	7.621130384	5.771259573	5.683995
ORC06	7.720806272	40.54200887	6.437654104	5.813246298	5.71205459
ORC07	6.522961511	34.51373164	5.927680184	5.501655131	5.31546614
ORC08	7.986425972	43.2911147	6.746041155	6.438558056	6.36783939
ORC09	8.539129763	46.69780955	5.935697173	6.531102733	6.41343105
ORC10	8.187641326	42.00626135	7.168496656	6.091460153	5.86586836
ORC28	8.364101904	56.11808987	7.292419989	6.401715266	6.16202358
ORC32	10.19173957	59.65388569	6.712593719	7.608494561	7.4793217
ORC36	7.93925518	44.29076845	6.377857063	6.2	6.03885722
ORC40	7.441679262	34.09617598	5.332564352	6.267969526	6.26962143
ORC43	9.135900612	60.02806875	9.105139379	6.580745846	6.31345077
ORC46A	8.228701519	48.61874029	7.261177474	6.494794736	6.36711196
ORST97-013	8.579663583	66.07938137	11.14964645	7.736536073	7.51896142
ORST97-023	8.592232694	61.62029891	7.679311419	6.769695465	6.57429646

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

STRM_ID	TI outstream SUM(1/d2)	TI IS*OS 1/d2	TI total SUM(1/d2)	TI area (m2)	TI length(m)
ORST97-032	9.640818272	68.85167203	7.167909653	7.416139241	7.12913846
ORST97-035	9.148207528	65.33211201	10.80771009	7.055018234	7.02661918
ORST97-304	7.821388715	45.5517078	7.415045396	6.462879383	6.45624503
ORST97-305	7.336015146	42.01321879	8.224928846	5.692183982	5.57918555
ORST97-307	8.846969792	53.5911731	6.582992126	6.988917497	6.66881921
ORST97-309	8.097308077	50.56975892	7.093102712	6.218702866	6.1396565
ORST97-310	8.111278136	45.02658408	8.192248657	6.590657742	6.53153587
ORST97-422	7.73329394	36.55582843	4.976599913	6.1499388	6.0677952
ORST97-424	9.587329119	69.53580477	8.092560155	9.583333333	9.74321534
ORST97-425	7.596762134	49.78386435	8.423348181	7.063642519	6.94118327
ORST97-428	9.488200363	83.14371835	10.17307848	7.54216515	7.1149355
ORST97-431	9.236505641	81.11254478	9.667724434	7.555071411	7.13781149
ORST97-432	7.081504427	38.89776083	6.684456243	5.958245243	5.79638901
ORST97-434	8.056073774	64.02893606	11.62813632	6.23933898	6.14582005
ORST97-435	8.422665334	50.65856504	6.044138091	7.113375796	7.06912118
ORST97-436	7.801921825	46.11444414	7.257635513	6.34529703	6.13564095
ORST97-440	8.096526031	38.77982124	6.127940242	6.528206129	6.4273945
ORST97-441	7.288142796	41.05792818	5.619815668	5.679878049	5.46375032
ORST97-443	7.707245403	51.44517237	8.3964687	5.964776497	5.65209641
ORST97-444	8.042953707	64.69808496	12.41641867	6.398062933	6.12652277
ORST97-445	8.496059795	40.75910207	5.942089034	6.324096386	6.12891003
ORST97-448	7.890718364	47.13496652	6.507181515	6.590555888	6.35572729
ORST97-451	7.949706926	54.58558674	8.098458829	6.82260372	6.63922173
ORST97-460	9.147914175	86.60985443	9.541669534	7.254501588	7.01565891
ORST97-461	8.973870382	57.04295083	7.857998063	6.930547204	6.70957251
ORST97-462	8.221670056	46.97300049	5.755216176	6.756630991	6.67635616
ORST97-464	8.517925649	56.56122982	7.53287234	7.001549472	6.8285765
ORV01	12.52995403	134.5479126	11.40819341	12.27677006	12.5017932
ORV02	10.0672381	87.9998252	10.42185296	8.331613481	7.78982624
ORV03	10.13155066	141.9284238	17.6999246	8.161648883	7.50143609
ORV04	10.6664827	92.73628059	8.941746539	8.911132397	8.35771616
ORV08	8.515521815	62.86687805	10.52636535	6.796880593	6.78960517
ORV20	10.45433013	93.873213	8.702179801	9.130472103	8.99819326
ORV21	8.460485142	60.13005007	7.43074066	6.81147541	6.53256957
ORV22	10.24876656	81.7468383	8.014584409	8.839166667	8.579367
ORV25	10.7030354	90.89427573	10.01554073	8.320601947	7.58645859
ORV27	8.660473703	70.70478393	8.615756084	6.632035526	6.38186406
ORV28	9.377746066	126.1663768	12.83953159	7.496744654	7.20180281
ORV30	11.93417835	96.20670205	9.081526753	11.77492674	11.2621724
ORV31	12.27343091	107.160644	9.655003693	11.51633766	11.5813879

Table A1 (con't). Western Oregon Dataset used for Linear Regressions.

SITE_ID	NO3 (ueq/L)	logNO3	Chloride	logCl	HLRU	SGW	WSAREAKM	POPDENKM	XELEV	SLOPMEAN	F LU instream SUM(1/d)
WCAP99-0505	3.6	0.5563025	12.4	1.093421685	20	1	152.3659	0.053	791	27.21	24.556307
WCAP99-0512	0.4	-0.39794	10928.4	4.038556583	18	1	62.4868	0	349	22.30	13.216895
WCAP99-0574	2.1	0.3222193	9.6	0.982271233	18	1	1.3095	0	935	22.79	10.143655
WCAP99-0581	0	-3	3.9	0.591064607	16	1	33.8794	0	967	15.19	14.418027
WCAP99-0583	2.9	0.462398	3.9	0.591064607	20	1	4.2639	0.235	1096	25.96	6.128001
WCAP99-0587	28.6	1.456366	107.2	2.030194785	18	1	6.5631	0.305	491	18.18	4.339839
WCAP99-0590	4.3	0.6334685	272.2	2.434888121	18	1	68.6855	0.611	62	20.08	17.436045
WCAP99-0592	2.1	0.3222193	59.5	1.774516966	20	1	4.4256	0.451	1750	18.09	26.598921
WCAP99-0593	7.9	0.8976271	40.3	1.605305046	18	1	9.4599	0	580	26.03	10.388885
WCAP99-0598	2.5	0.39794	16.9	1.227886705	18	1	6.9523	0.144	542	26.39	7.971765
WCAP99-0600	7.1	0.8512583	915.8	2.961800639	16	1	32.3653	2.533	24	19.18	9.144793
WCAP99-0604	2.1	0.3222193	8.2	0.913813852	mixed		6.119	0.163	1266	22.05	7.949508
WCAP99-0605	2.9	0.462398	88.6	1.947433722	16	1	167.386	0.305	281	17.46	29.169899
WCAP99-0606	2.9	0.462398	136.8	2.136086097	16	1	72.4902	0.221	325	21.64	16.281212
WCAP99-0613	4.3	0.6334685	24	1.380211242	19	1	46.8598	0.235	501	24.66	24.293234
WCAP99-0625	0.4	-0.39794	9.9	0.995635195	18	1	19.3754	0.413	552	23.44	16.272987
WCAP99-0750	0	-3	31.3	1.495544338	16	1	1.1674	0	1094	11.66	8.516262
WCAP99-0755	0.3	-0.5228787	5.9	0.770852012	20	1	5.0678	0.986	1508	18.12	2.982053
WCAP99-0756	0	-3	7.3	0.86332286	12	1	13.1348	0	1761	13.23	35.487469
WCAP99-0761	0	-3	15.8	1.198657087	20	1	1.119	0	1161	24.36	2.838225
WCAP99-0767	0.4	-0.39794	151.5	2.180412633	18	1	36.1047	0.111	647	18.92	11.726436
WCAP99-0768	2.9	0.462398	389.2	2.590172832	18	1	146.1422	0.718	539	22.21	16.694708
WCAP99-0810	4.3	0.6334685	3.9	0.591064607	18	1	2.5009	0	777	18.33	5.769464
WCAP99-0811	2.1	0.3222193	26.2	1.418301291	mixed		14.515	0.069	697	20.05	9.83277
WCAP99-0812	4.3	0.6334685	153.4	2.18582536	18	1	114.4613	30.526	548	18.31	19.134651
WCAP99-0816	0	-3	13.5	1.130333768	mixed		12.01	0	1630	12.31	7.426488
WCAP99-0824	2.9	0.462398	25.9	1.413299764	18	1	5.6975	0	1722	13.44	18.979359
WCAP99-0825	0	-3	28.8	1.459392488	mixed		196.2838	1.549	723	17.16	17.269705
WCAP99-0826	4.3	0.6334685	557.9	2.746556361	18	1	14.5404	0	352	21.68	8.085236
WCAP99-0831	5	0.69897	31	1.491361694	18	1	55.4937	0.162	815	19.11	21.825274
WCAP99-0832	2.1	0.3222193	277.3	2.44294987	16	1	12.7011	1.811	487	20.97	10.14365
WCAP99-0833	35.7	1.5526682	6.2	0.792391689	12	1	26.3048	0.304	1721	7.70	23.29459
WCAP99-0835	2.1	0.3222193	561.3	2.749195042	16	1	2.8794	13.542	91	13.35	9.065279
WCAP99-0837	1.4	0.146128	540.7	2.73295637	mixed		492.3019	0.055	568	19.11	45.472843
WCAP99-0838	4.3	0.6334685	505.7	2.703892954	16	1	2.7443	8.029	90	18.69	7.908451
WCAP99-0843	0	-3	20.6	1.31386722	18	1	16.5155	0	1058	20.33	12.781485
WCAP99-0844	1.4	0.146128	139.1	2.14332713	16	1	2.4735	0.81	362	23.36	5.900567
WCAP99-0862	0.2	-0.69897	115.9	2.064083436	20	1	9.7729	0.409	817	22.92	9.523422
WCAP99-1005	7.9	0.8976271	111.4	2.046885191	16	1	13.4942	0	550	23.86	8.894896
WCAP99-1051	20	1.30103	260.9	2.416474079	9	0	3.4499	48.696	43	8.37	2.693811
WCAP99-1055	0	-3	7.9	0.897627091	20	1	4.8422	0.207	1228	26.93	6.328371
WCAP99-1056	1.4	0.146128	31.3	1.495544338	18	1	22.963	1.35	932	13.86	27.579008

Table A2. Northern California Dataset used for Linear Regressions.

SITE_ID	NO3 (ueq/L)	logNO3	Chloride	logCl	HLRU	SGW	WSAREAKM	POPDENKM	XELEV	SLOPMEAN	F LU instream SUM(1/d)
WCAP99-0505	3.6	0.5563025	12.4	1.093421685	20	1	152.3659	0.053	791	27.21	24.556307
WCAP99-0512	0.4	-0.39794	10928.4	4.038556583	18	1	62.4868	0	349	22.30	13.216895
WCAP99-0574	2.1	0.3222193	9.6	0.982271233	18	1	1.3095	0	935	22.79	10.143655
WCAP99-0581	0	-3	3.9	0.591064607	16	1	33.8794	0	967	15.19	14.418027
WCAP99-0583	2.9	0.462398	3.9	0.591064607	20	1	4.2639	0.235	1096	25.96	6.128001
WCAP99-0587	28.6	1.456366	107.2	2.030194785	18	1	6.5631	0.305	491	18.18	4.339839
WCAP99-0590	4.3	0.6334685	272.2	2.434888121	18	1	68.6855	0.611	62	20.08	17.436045
WCAP99-0592	2.1	0.3222193	59.5	1.774516966	20	1	4.4256	0.451	1750	18.09	26.598921
WCAP99-0593	7.9	0.8976271	40.3	1.605305046	18	1	9.4599	0	580	26.03	10.388885
WCAP99-0598	2.5	0.39794	16.9	1.227886705	18	1	6.9523	0.144	542	26.39	7.971765
WCAP99-0600	7.1	0.8512583	915.8	2.961800639	16	1	32.3653	2.533	24	19.18	9.144793
WCAP99-0604	2.1	0.3222193	8.2	0.913813852	mixed		6.119	0.163	1266	22.05	7.949508
WCAP99-0605	2.9	0.462398	88.6	1.947433722	16	1	167.386	0.305	281	17.46	29.169899
WCAP99-0606	2.9	0.462398	136.8	2.136086097	16	1	72.4902	0.221	325	21.64	16.281212
WCAP99-0613	4.3	0.6334685	24	1.380211242	19	1	46.8598	0.235	501	24.66	24.293234
WCAP99-0625	0.4	-0.39794	9.9	0.995635195	18	1	19.3754	0.413	552	23.44	16.272987
WCAP99-0750	0	-3	31.3	1.495544338	16	1	1.1674	0	1094	11.66	8.516262
WCAP99-0755	0.3	-0.5228787	5.9	0.770852012	20	1	5.0678	0.986	1508	18.12	2.982053
WCAP99-0756	0	-3	7.3	0.86332286	12	1	13.1348	0	1761	13.23	35.487469
WCAP99-0761	0	-3	15.8	1.198657087	20	1	1.119	0	1161	24.36	2.838225
WCAP99-0767	0.4	-0.39794	151.5	2.180412633	18	1	36.1047	0.111	647	18.92	11.726436
WCAP99-0768	2.9	0.462398	389.2	2.590172832	18	1	146.1422	0.718	539	22.21	16.694708
WCAP99-0810	4.3	0.6334685	3.9	0.591064607	18	1	2.5009	0	777	18.33	5.769464
WCAP99-0811	2.1	0.3222193	26.2	1.418301291	mixed		14.515	0.069	697	20.05	9.83277
WCAP99-0812	4.3	0.6334685	153.4	2.18582536	18	1	114.4613	30.526	548	18.31	19.134651
WCAP99-0816	0	-3	13.5	1.130333768	mixed		12.01	0	1630	12.31	7.426488
WCAP99-0824	2.9	0.462398	25.9	1.413299764	18	1	5.6975	0	1722	13.44	18.979359
WCAP99-0825	0	-3	28.8	1.459392488	mixed		196.2838	1.549	723	17.16	17.269705
WCAP99-0826	4.3	0.6334685	557.9	2.746556361	18	1	14.5404	0	352	21.68	8.085236
WCAP99-0831	5	0.69897	31	1.491361694	18	1	55.4937	0.162	815	19.11	21.825274
WCAP99-0832	2.1	0.3222193	277.3	2.44294987	16	1	12.7011	1.811	487	20.97	10.14365
WCAP99-0833	35.7	1.5526682	6.2	0.792391689	12	1	26.3048	0.304	1721	7.70	23.29459
WCAP99-0835	2.1	0.3222193	561.3	2.749195042	16	1	2.8794	13.542	91	13.35	9.065279
WCAP99-0837	1.4	0.146128	540.7	2.73295637	mixed		492.3019	0.055	568	19.11	45.472843
WCAP99-0838	4.3	0.6334685	505.7	2.703892954	16	1	2.7443	8.029	90	18.69	7.908451
WCAP99-0843	0	-3	20.6	1.31386722	18	1	16.5155	0	1058	20.33	12.781485
WCAP99-0844	1.4	0.146128	139.1	2.14332713	16	1	2.4735	0.81	362	23.36	5.900567
WCAP99-0862	0.2	-0.69897	115.9	2.064083436	20	1	9.7729	0.409	817	22.92	9.523422
WCAP99-1005	7.9	0.8976271	111.4	2.046885191	16	1	13.4942	0	550	23.86	8.894896
WCAP99-1051	20	1.30103	260.9	2.416474079	9	0	3.4499	48.696	43	8.37	2.693811
WCAP99-1055	0	-3	7.9	0.897627091	20	1	4.8422	0.207	1228	26.93	6.328371
WCAP99-1056	1.4	0.146128	31.3	1.495544338	18	1	22.963	1.35	932	13.86	27.579008

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	NO3 (ueq/L)	logNO3	Chloride	logCl	HLRU	SGW	WSAREAKM	POPDENKM	XELEV	SLOPMEAN	F LU instream SUM(1/d)
WCAP99-1057	10.7	1.0293838	315.1	2.498448403	16	1	1.751	0	122	22.43	8.504705
WCAP99-1060	4.3	0.6334685	50.5	1.703291378	16	1	6.5379	0	742	18.69	9.116547
WCAP99-1062	2.8	0.447158	173	2.238046103	18	1	71.0914	0.253	588	22.99	19.445402
WCAP99-1063	5.4	0.7323938	145.1	2.161667412	16	1	6.7578	0.888	208	22.53	10.86406
WCAP99-1064	2.1	0.3222193	26.2	1.418301291	20	1	9.1154	0.219	747	21.36	7.279563
WCAP99-1065	0	-3	74.5	1.872156273	18	1	2.4948	0	1047	17.01	4.679248
WCAP99-1066	3.6	0.5563025	24.8	1.394451681	20	1	43.4265	0	540	19.75	12.716067
WCAP99-1074	1.4	0.146128	169.8	2.229937686	16	1	114.3091	0.14	249	15.57	26.392994
WCAP99-1075	0	-3	121	2.08278537	16	1	19.4452	2.057	205	20.34	8.476326
WCAP99-1079	2.1	0.3222193	253.9	2.404662701	20	1	25.2628	0.198	1307	19.78	20.632061
WCAP99-1080	0	-3	74.5	1.872156273	mixed		622.9885	0.197	541	18.71	36.560589
WCAP99-1090	12.9	1.1105897	243.1	2.385784959	16	1	8.7215	1.147	143	19.28	6.144124
WCAP99-1091	5.7	0.7558749	81.5	1.911157609	20	1	24.1369	0.041	1518	24.63	14.301015
WCAP99-1092	2.1	0.3222193	80.1	1.903632516	16	1	3.7191	1.075	285	19.75	5.428753
WCAP99-1100	0.6	-0.2218487	77	1.886490725	20	1	236.6612	7.348	614	14.69	29.32523
WCAP99-1106	2.1	0.3222193	192.6	2.284656283	16	1	1.3934	0	551	12.23	7.582013
WCAP99-1111	0.3	-0.5228787	121.6	2.084933575	19	1	23.5093	0.17	14	16.72	13.737751

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	F LU outstream SUM(1/d)	F LU total SUM(1/d)	F LU instream SUM(1/d2)	F LU outstream SUM(1/d2)	F LU total SUM(1/d2)	F LU area (m2)	F LU length(m)
WCAP99-0505	717.557739	20.19615	0.058032	9.255743	0.009743	151454368	2216432128
WCAP99-0512	327.692902	9.2175	0.08548	4.650969	0.011864	61981200	848876608
WCAP99-0574	5.131122	2.419974	0.433489	0.056054	0.013477	1278900	1227999
WCAP99-0581	196.41394	10.747202	0.145587	2.902133	0.014136	33702300	184516000
WCAP99-0583	18.367146	3.353256	0.042484	0.213521	0.005371	4188600	9378513
WCAP99-0587	32.462349	3.030567	0.03693	0.44041	0.00871	6435900	22972076
WCAP99-0590	409.906464	14.671091	0.039895	6.03974	0.013822	67621024	609162368
WCAP99-0592	15.686559	3.895918	1.277563	0.16635	0.006575	4304700	8316196
WCAP99-0593	50.290123	6.131779	0.207207	0.679297	0.014652	9365400	24850756
WCAP99-0598	34.07777	4.746593	0.078415	0.445591	0.006725	6898500	17374860
WCAP99-0600	165.40657	6.173704	0.149204	2.295762	0.003639	31127400	263701984
WCAP99-0604	30.165867	3.479279	0.190546	0.378505	0.011607	6079500	22661978
WCAP99-0605	837.827881	18.328251	0.364453	11.727662	0.015217	166181024	2998569728
WCAP99-0606	459.943359	12.368301	0.095773	7.024144	0.008502	72375200	860588160
WCAP99-0613	213.651718	10.466741	0.716525	2.887686	0.01452	42423300	407562976
WCAP99-0625	94.133209	8.840229	0.151245	1.253784	0.015329	19317600	78384496
WCAP99-0750	3.447958	1.787673	0.238438	0.03011	0.009358	1141200	1298557
WCAP99-0755	29.263893	2.286926	0.030872	0.42475	0.008499	5019300	20126280
WCAP99-0756	40.099464	7.988915	1.541304	0.39906	0.01634	13080600	38415616
WCAP99-0761	6.736747	1.514714	0.065784	0.099683	0.010001	1097100	1707710
WCAP99-0767	198.381897	9.179946	0.059293	2.952411	0.012929	35893800	286560640
WCAP99-0768	821.337036	12.390074	0.192145	11.483339	0.007436	145556000	3226057472
WCAP99-0810	14.118027	2.320416	0.199081	0.203815	0.005608	2452500	4941377
WCAP99-0811	76.395264	6.23645	0.16381	1.063611	0.013237	14465700	58223776
WCAP99-0812	624.054016	15.964433	0.073341	8.899658	0.013255	111632544	1383805568
WCAP99-0816	64.896889	5.529357	0.027587	0.981823	0.01059	11207700	41424184
WCAP99-0824	18.871063	5.127685	0.558535	0.184953	0.009451	5673600	10868965
WCAP99-0825	1021.471497	14.487651	0.061601	14.31524	0.011674	195208736	5519374848
WCAP99-0826	76.484657	5.467095	0.106162	1.044675	0.011521	14493600	71861008
WCAP99-0831	341.607025	14.586878	0.306701	5.05259	0.016769	55377000	369378496
WCAP99-0832	67.57412	5.756613	0.135458	0.964771	0.013273	12651300	55877388
WCAP99-0833	90.324318	9.170288	0.406729	0.98673	0.014026	25958700	130394480
WCAP99-0835	16.534279	2.971679	0.366078	0.227896	0.01256	2862900	5855365
WCAP99-0837	2770.999023	32.818413	0.657379	40.576019	0.016919	489953312	15555813376
WCAP99-0838	14.272577	3.083045	0.21938	0.198002	0.011516	2723400	4320271
WCAP99-0843	90.73806	7.770032	0.139587	1.200598	0.016444	16442100	74779112
WCAP99-0844	11.462631	2.741268	0.111933	0.14467	0.006166	2425500	3585890
WCAP99-0862	50.846783	6.924881	0.0174	0.69113	0.008283	9509400	21504740
WCAP99-1005	75.40258	6.246486	0.072899	1.088137	0.012933	13419000	50547900
WCAP99-1051	18.246384	2.016073	0.012043	0.270864	0.00205	3294900	8090096
WCAP99-1055	24.982025	3.677898	0.117922	0.344912	0.011784	4671000	10313167
WCAP99-1056	116.267548	7.561249	1.213357	1.595008	0.014832	22699800	142986720

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	F LU outstream SUM(1/d)	F LU total SUM(1/d)	F LU instream SUM(1/d2)	F LU outstream SUM(1/d2)	F LU total SUM(1/d2)	F LU area (m2)	F LU length(m)
WCAP99-1057	8.169106	2.658314	0.103104	0.102478	0.011761	1700100	1942136
WCAP99-1060	37.317726	4.882252	0.196361	0.51927	0.013077	6512400	14781079
WCAP99-1062	392.053314	13.67295	0.204591	5.435622	0.014168	70731040	785277696
WCAP99-1063	42.193523	3.906269	0.296649	0.636225	0.012471	6726600	24414076
WCAP99-1064	49.10371	5.164703	0.032804	0.649615	0.011754	9053100	30186500
WCAP99-1065	10.536941	2.146123	0.035592	0.134073	0.008755	2444400	5130462
WCAP99-1066	231.04747	9.664991	0.070913	3.150563	0.012449	43170300	356988320
WCAP99-1074	612.718079	14.565256	0.261463	8.788116	0.009559	113775776	1795455360
WCAP99-1075	120.32917	6.301993	0.049953	1.820838	0.011509	19387800	103921920
WCAP99-1079	124.888512	9.141257	0.413433	1.645818	0.014635	25196400	128812304
WCAP99-1080	3394.034424	30.724625	0.08444	48.264801	0.014781	619311552	26003937280
WCAP99-1090	52.252613	4.233206	0.077932	0.778256	0.011057	8682300	33188896
WCAP99-1091	101.010857	8.616355	0.146338	1.228095	0.014615	24071400	118658272
WCAP99-1092	16.631718	3.444496	0.028934	0.221815	0.010706	3647700	6475518
WCAP99-1100	1212.283325	21.607786	0.422749	17.009701	0.014909	232087200	5939475456
WCAP99-1106	5.612476	2.091594	0.1603	0.072587	0.010969	1367100	1673684
WCAP99-1111	122.367348	7.44733	0.122091	1.71946	0.013752	22953600	149049440

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	A LU instream SUM(1/d)	A LU outstream SUM(1/d)	A LU total SUM(1/d)	A LU instream SUM(1/d2)	A LU outstream SUM(1/d2)	A LU total SUM(1/d2)
WCAP99-0505	0	0	0	0	0	0
WCAP99-0512	0	0	0	0	0	0
WCAP99-0574	0	0	0	0	0	0
WCAP99-0581	0	0	0	0	0	0
WCAP99-0583	0	0	0	0	0	0
WCAP99-0587	0	0	0	0	0	0
WCAP99-0590	0.124315	3.891607	0.117151	0.00002	0.047449	0.000018
WCAP99-0592	0	0	0	0	0	0
WCAP99-0593	0	0	0	0	0	0
WCAP99-0598	0	0	0	0	0	0
WCAP99-0600	3.930752	7.275245	1.299853	0.161657	0.109014	0.003871
WCAP99-0604	0	0	0	0	0	0
WCAP99-0605	0	0	0	0	0	0
WCAP99-0606	0	0	0	0	0	0
WCAP99-0613	0	0	0	0	0	0
WCAP99-0625	0	0	0	0	0	0
WCAP99-0750	0	0	0	0	0	0
WCAP99-0755	0	0	0	0	0	0
WCAP99-0756	0	0	0	0	0	0
WCAP99-0761	0	0	0	0	0	0
WCAP99-0767	0	0	0	0	0	0
WCAP99-0768	0	0	0	0	0	0
WCAP99-0810	0	0	0	0	0	0
WCAP99-0811	0	0	0	0	0	0
WCAP99-0812	0.000276	0.003861	0.000258	0	0.000015	0
WCAP99-0816	0	0	0	0	0	0
WCAP99-0824	0	0	0	0	0	0
WCAP99-0825	0.005361	0.029259	0.00447	0.000004	0.000116	0.000003
WCAP99-0826	0	0	0	0	0	0
WCAP99-0831	0	0	0	0	0	0
WCAP99-0832	0	0	0	0	0	0
WCAP99-0833	0	0	0	0	0	0
WCAP99-0835	0	0	0	0	0	0
WCAP99-0837	0	0	0	0	0	0
WCAP99-0838	0	0	0	0	0	0
WCAP99-0843	0	0	0	0	0	0
WCAP99-0844	0	0	0	0	0	0
WCAP99-0862	0	0	0	0	0	0
WCAP99-1005	0.004691	0.036827	0.004154	0.000001	0.000069	0.000001
WCAP99-1051	0.60887	1.140574	0.345931	0.015388	0.021245	0.002134
WCAP99-1055	0	0	0	0	0	0
WCAP99-1056	0	0	0	0	0	0

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	A LU instream SUM(1/d)	A LU outstream SUM(1/d)	A LU total SUM(1/d)	A LU instream SUM(1/d2)	A LU outstream SUM(1/d2)	A LU total SUM(1/d2)
WCAP99-1057	0	0	0	0	0	0
WCAP99-1060	0	0	0	0	0	0
WCAP99-1062	0	0	0	0	0	0
WCAP99-1063	0	0	0	0	0	0
WCAP99-1064	0	0	0	0	0	0
WCAP99-1065	0	0	0	0	0	0
WCAP99-1066	0	0	0	0	0	0
WCAP99-1074	0	0	0	0	0	0
WCAP99-1075	0	0	0	0	0	0
WCAP99-1079	0	0	0	0	0	0
WCAP99-1080	0	0	0	0	0	0
WCAP99-1090	0	0	0	0	0	0
WCAP99-1091	0	0	0	0	0	0
WCAP99-1092	0	0	0	0	0	0
WCAP99-1100	0.039498	7.23799	0.038985	0.000002	0.128318	0.000002
WCAP99-1106	0	0	0	0	0	0
WCAP99-1111	0	0	0	0	0	0

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	A LU area (m2)	A LU length(m)	U LU instream SUM(1/d)	U LU outstream SUM(1/d)	U LU total SUM(1/d)	U LU instream SUM(1/d2)	U LU outstream SUM(1/d2)
WCAP99-0505	0	0	0.002409	0.166636	0.002363	0.000001	0.003573
WCAP99-0512	0	0	0	0	0	0	0
WCAP99-0574	0	0	0	0	0	0	0
WCAP99-0581	0	0	0	0	0	0	0
WCAP99-0583	0	0	0	0	0	0	0
WCAP99-0587	0	0	0.109304	0.273847	0.068503	0.000213	0.002481
WCAP99-0590	686700	4979525	0	0	0	0	0
WCAP99-0592	0	0	0	0	0	0	0
WCAP99-0593	0	0	0	0	0	0	0
WCAP99-0598	0	0	0	0	0	0	0
WCAP99-0600	990900	1422822	0	0	0	0	0
WCAP99-0604	0	0	0	0	0	0	0
WCAP99-0605	0	0	0	0	0	0	0
WCAP99-0606	0	0	0	0	0	0	0
WCAP99-0613	0	0	0	0	0	0	0
WCAP99-0625	0	0	0	0	0	0	0
WCAP99-0750	0	0	0	0	0	0	0
WCAP99-0755	0	0	0	0	0	0	0
WCAP99-0756	0	0	0.001588	0.001567	0.000789	0.000001	0.000001
WCAP99-0761	0	0	0	0	0	0	0
WCAP99-0767	0	0	0	0	0	0	0
WCAP99-0768	0	0	0	0	0	0	0
WCAP99-0810	0	0	0	0	0	0	0
WCAP99-0811	0	0	0	0	0	0	0
WCAP99-0812	900	3883	0.487525	18.871506	0.45668	0.000096	0.296267
WCAP99-0816	0	0	0	0	0	0	0
WCAP99-0824	0	0	0	0	0	0	0
WCAP99-0825	7200	14457	0.028716	0.520112	0.026209	0.000015	0.004323
WCAP99-0826	0	0	0	0	0	0	0
WCAP99-0831	0	0	0	0	0	0	0
WCAP99-0832	0	0	0	0	0	0	0
WCAP99-0833	0	0	0	0	0	0	0
WCAP99-0835	0	0	0	0	0	0	0
WCAP99-0837	0	0	0.000103	0.003563	0.0001	0	0.000006
WCAP99-0838	0	0	0	0	0	0	0
WCAP99-0843	0	0	0	0	0	0	0
WCAP99-0844	0	0	0	0	0	0	0
WCAP99-0862	0	0	0	0	0	0	0
WCAP99-1005	18000	96581	0	0	0	0	0
WCAP99-1051	121500	83127	0.135543	0.161818	0.113193	0.00479	0.002805
WCAP99-1055	0	0	0	0	0	0	0
WCAP99-1056	0	0	0.002332	0.074861	0.00223	0	0.001269

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	A LU area (m2)	A LU length(m)	U LU instream SUM(1/d)	U LU outstream SUM(1/d)	U LU total SUM(1/d)	U LU instream SUM(1/d2)	U LU outstream SUM(1/d2)
WCAP99-1057	0	0	0	0	0	0	0
WCAP99-1060	0	0	0.000405	0.001277	0.000307	0	0.000002
WCAP99-1062	0	0	0	0	0	0	0
WCAP99-1063	0	0	0	0	0	0	0
WCAP99-1064	0	0	0	0	0	0	0
WCAP99-1065	0	0	0	0	0	0	0
WCAP99-1066	0	0	0	0	0	0	0
WCAP99-1074	0	0	0	0	0	0	0
WCAP99-1075	0	0	0	0	0	0	0
WCAP99-1079	0	0	0	0	0	0	0
WCAP99-1080	0	0	0.000042	0.001013	0.00004	0	0.000001
WCAP99-1090	0	0	0.000192	0.0012	0.000165	0	0.000001
WCAP99-1091	0	0	0.001693	0.021369	0.001378	0.000001	0.000283
WCAP99-1092	0	0	0	0	0	0	0
WCAP99-1100	814500	21981272	0.093508	10.804241	0.091914	0.000004	0.140713
WCAP99-1106	0	0	0	0	0	0	0
WCAP99-1111	0	0	0	0	0	0	0

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	U LU total SUM(1/d2)	U LU area (m2)	U LU length(m)	TI instream SUM(1/d)	TI outstream SUM(1/d)	TI IS*OS 1/d	TI total SUM(1/d)	TI instream SUM(1/d2)
WCAP99-0505	0.000001	13500	163763	5.98091725	6.744800642	40.3400945	6.113060497	6.970930362
WCAP99-0512	0	0	0	5.780645872	6.117834554	35.3650351	6.006279242	5.876118611
WCAP99-0574	0	0	0	5.227515308	6.359296734	33.243321	5.837157012	5.064306613
WCAP99-0581	0	0	0	6.025759036	6.464737309	38.9549493	6.337455876	5.107760023
WCAP99-0583	0	0	0	5.460505091	6.413587172	35.0214254	5.836189638	5.205042825
WCAP99-0587	0.000083	52200	50547	6.895038112	6.709063708	46.25925	6.823512363	10.61669222
WCAP99-0590	0	0	0	5.834516736	6.085677843	35.5069892	5.972297396	5.374938836
WCAP99-0592	0	0	0	6.022626396	7.86617667	47.3750432	7.123559884	5.742321417
WCAP99-0593	0	0	0	5.289743608	5.919012782	31.31006	5.736306595	4.656339415
WCAP99-0598	0	0	0	5.514322533	6.287770213	34.672793	5.848953526	4.51944944
WCAP99-0600	0	0	0	6.13454762	6.507622075	39.9213175	6.443596378	5.937342703
WCAP99-0604	0	0	0	6.03615837	7.503217152	45.290607	6.677666051	5.852643266
WCAP99-0605	0	0	0	6.169060056	6.928889271	42.744734	6.653407868	5.178786129
WCAP99-0606	0	0	0	5.622194407	5.948709832	33.4448031	5.755322318	5.320524373
WCAP99-0613	0	0	0	6.011776021	6.552399816	39.3915601	6.213146637	5.923158617
WCAP99-0625	0	0	0	5.782130954	6.234203495	36.046981	6.153481782	5.67409386
WCAP99-0750	0	0	0	6.779683663	7.613835296	51.6193948	7.118605235	6.585724573
WCAP99-0755	0	0	0	6.896868471	7.374338977	50.859846	6.844285698	7.850456875
WCAP99-0756	0	2700	11412	7.99231277	9.076801379	72.5446356	8.770906532	7.87514979
WCAP99-0761	0	0	0	5.643259621	6.049409503	34.1383884	6.051086803	5.896386968
WCAP99-0767	0	0	0	6.10902642	6.637115034	40.5463111	6.322544864	5.751505135
WCAP99-0768	0	0	0	5.822521615	6.409185332	37.3176201	6.151564934	4.992974466
WCAP99-0810	0	0	0	5.133501403	5.566593068	28.5761133	5.569082415	4.931178132
WCAP99-0811	0	0	0	5.757338332	6.63085481	38.1760746	6.221561993	5.044558171
WCAP99-0812	0.000083	2433600	17077332	6.227457317	6.784801247	42.2520602	6.450907552	5.137671019
WCAP99-0816	0	0	0	6.399741782	6.451437141	41.2875318	6.576181295	8.551069228
WCAP99-0824	0	0	0	7.259081525	8.316454709	60.3698227	8.021072153	7.240703513
WCAP99-0825	0.000012	86400	909144	6.986114086	7.201480492	50.3103643	7.016728876	9.31188971
WCAP99-0826	0	0	0	5.578456516	6.281798291	35.0427386	5.92759268	4.765673766
WCAP99-0831	0	0	0	6.146937535	6.459084392	39.7035883	6.351555716	5.910266667
WCAP99-0832	0	0	0	5.937150746	6.105574612	36.2497169	6.25165992	6.030689424
WCAP99-0833	0	0	0	7.228234428	8.736787345	63.1515471	8.046159564	6.115054974
WCAP99-0835	0	0	0	5.75848472	6.499102751	37.4249839	6.315514787	5.559707715
WCAP99-0837	0	1800	40403	6.119616061	6.433437735	39.3701689	6.319756549	5.68730863
WCAP99-0838	0	0	0	5.214468443	5.677678326	29.6060745	5.760935524	4.957374894
WCAP99-0843	0	0	0	6.015883757	6.520775905	39.2282298	6.322002504	6.041922227
WCAP99-0844	0	0	0	5.433695646	6.441581237	35.0015919	5.849253543	4.871396524
WCAP99-0862	0	0	0	6.067958669	6.555658129	39.7794626	6.242852757	6.157984088
WCAP99-1005	0	0	0	5.534907333	5.668991867	31.3773447	5.760901596	5.371804527
WCAP99-1051	0.004603	15300	6436	7.113755729	6.924242556	49.2573702	7.258329225	7.606949807
WCAP99-1055	0	0	0	5.400556926	6.428173117	34.7157149	5.87684023	4.971885916
WCAP99-1056	0	12600	91495	5.972539091	7.696598341	45.9682345	6.962705149	5.657113126

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	U LU total SUM(1/d2)	U LU area (m2)	U LU length(m)	TI instream SUM(1/d)	TI outstream SUM(1/d)	TI IS*OS 1/d	TI total SUM(1/d)	TI instream SUM(1/d2)
WCAP99-1057	0	0	0	5.411375699	6.231450459	33.7207196	5.733676236	5.17066292
WCAP99-1060	0	900	3255	5.469436922	6.104437973	33.3878384	6.020188794	4.83050951
WCAP99-1062	0	0	0	6.038088164	6.39360989	38.6051802	6.351854003	5.079108853
WCAP99-1063	0	0	0	5.408772103	5.68841256	30.7673272	5.873432287	5.260743772
WCAP99-1064	0	0	0	6.088886285	6.506630103	39.6181308	6.33318644	6.607973165
WCAP99-1065	0	0	0	6.752300812	7.241617791	48.8975817	7.021680569	7.608545521
WCAP99-1066	0	0	0	5.765928198	6.471110168	37.3119566	6.035608896	4.79756634
WCAP99-1074	0	0	0	6.193942834	6.713754809	41.5846135	6.492309605	6.090800871
WCAP99-1075	0	0	0	5.656072221	5.849691479	33.0862775	5.87963889	5.606706035
WCAP99-1079	0	0	0	5.791577898	7.010705293	40.6030458	6.416440674	5.18715634
WCAP99-1080	0	900	24726	6.490719946	6.585430352	42.7441841	6.573342922	7.328761449
WCAP99-1090	0	900	6044	5.471835748	5.868990192	32.1141503	5.873308178	5.05087986
WCAP99-1091	0	4500	18241	6.623027963	7.522485019	49.8216286	6.912676445	6.642960165
WCAP99-1092	0	0	0	5.998946884	6.923896074	41.5360848	6.220565672	6.38255287
WCAP99-1100	0.000003	2254500	68697200	5.992250656	7.523218734	45.0810124	6.36536609	5.068109456
WCAP99-1106	0	0	0	6.023609311	6.06941571	36.559789	6.422034463	5.730348735
WCAP99-1111	0	0	0	6.127739875	6.448060673	39.5120385	6.554490867	5.938729783

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	TI outstream SUM(1/d2)	TI IS*OS 1/d2	TI total SUM(1/d2)	TI area (m2)	TI length(m)
WCAP99-0505	7.642393804	53.27459501	10.5126189	6.01936741	6.06081999
WCAP99-0512	6.586484502	38.70296416	10.18908885	5.8714706	5.86178762
WCAP99-0574	7.860306978	39.80700461	6.565039744	5.40677966	5.20815089
WCAP99-0581	6.930926718	35.40151042	9.242885849	6.21272186	6.15978604
WCAP99-0583	7.40190973	38.52725713	6.516503873	5.75843542	5.73671961
WCAP99-0587	7.477754907	79.38902233	11.825	6.14597431	6.05007903
WCAP99-0590	6.550978735	35.21111002	9.347843886	5.85858824	5.8136955
WCAP99-0592	9.346350146	53.66974661	8.055025066	6.63602561	6.36722539
WCAP99-0593	6.471895742	30.13534323	8.527133897	5.5439597	5.47159757
WCAP99-0598	7.019126566	31.72258763	6.163924619	5.77876569	5.7769103
WCAP99-0600	7.060224449	41.91897211	11.20243484	6.12007792	6.1422564
WCAP99-0604	8.904449076	52.11456392	10.04336603	6.23339714	6.12896174
WCAP99-0605	7.452678141	38.59582618	10.315717	6.63380875	6.62265007
WCAP99-0606	6.444589312	34.28859451	6.755896496	5.63286284	5.57899861
WCAP99-0613	7.28310646	43.13899479	7.406780827	5.96124582	5.97904118
WCAP99-0625	6.713049623	38.09047365	9.551924331	5.94000929	5.92813946
WCAP99-0750	8.800198535	57.95568374	6.209713771	6.87440382	6.64738989
WCAP99-0755	8.287092775	65.05746446	6.582467608	6.51208674	6.34542673
WCAP99-0756	10.63287341	83.73547081	12.07982302	7.57717615	6.94860755
WCAP99-0761	6.661889114	39.28107615	8.070858283	5.63045267	5.52371929
WCAP99-0767	7.184402484	41.32112778	9.654033568	6.24709593	6.23336673
WCAP99-0768	7.084575443	35.37310429	11.01160073	5.98381512	5.95047667
WCAP99-0810	5.994530303	29.56009674	6.212506969	5.46013833	5.47109306
WCAP99-0811	7.303044318	36.84063189	8.255943007	6.18371142	6.21806544
WCAP99-0812	7.451283896	38.28224533	10.15813118	6.295928	6.14004746
WCAP99-0816	6.835800846	58.45340626	11.44750755	6.28725875	6.21504326
WCAP99-0824	9.846565215	71.29605934	10.90139154	6.97852858	6.48890493
WCAP99-0825	8.005225834	74.54378007	12.88522531	6.53167485	6.42030734
WCAP99-0826	6.799641365	32.40487247	6.313208498	5.91461829	5.95208155
WCAP99-0831	7.069893988	41.78495878	10.05701859	6.00477211	5.88522203
WCAP99-0832	6.657641569	40.1501686	10.72490398	5.70720114	5.56057732
WCAP99-0833	9.782692089	59.82169992	10.23734854	7.93156208	7.75060608
WCAP99-0835	7.127903814	39.62906183	8.645992366	6.03776529	5.95198097
WCAP99-0837	6.90119138	39.24920529	10.01282806	6.18496705	6.10447518
WCAP99-0838	6.088654625	30.18374357	8.52614042	5.42312746	5.33961683
WCAP99-0843	7.299812287	44.10489811	10.07997332	6.02144611	5.99975203
WCAP99-0844	7.41207641	36.10716326	5.664543326	5.60885746	5.42870124
WCAP99-0862	7.443424984	45.83649262	7.625706785	5.86064347	5.70968283
WCAP99-1005	5.930203945	31.85589639	8.721651078	5.59122479	5.55546831
WCAP99-1051	7.38787924	56.19922656	8.958961569	6.59510799	6.37149744
WCAP99-1055	7.337988284	36.4836406	8.493046133	5.69345972	5.57413351
WCAP99-1056	8.458501646	47.85070068	10.42751054	6.82206673	6.71168655

Table A2 (con't). Northern California Dataset used for Linear Regressions.

SITE_ID	TI outstream SUM(1/d2)	TI IS*OS 1/d2	TI total SUM(1/d2)	TI area (m2)	TI length(m)
WCAP99-1057	7.402606913	38.27638507	6.378771256	5.42209178	5.26608149
WCAP99-1060	6.563024854	31.70275397	8.8340598	5.90558236	5.92333884
WCAP99-1062	7.106359772	36.09397483	10.51656796	5.91909828	5.77980119
WCAP99-1063	5.972126365	31.41782658	9.087580596	5.55049294	5.47926241
WCAP99-1064	7.429098322	49.09128235	10.18125051	5.79038652	5.59550609
WCAP99-1065	8.151219076	62.01892139	9.006853227	6.39424142	6.11417472
WCAP99-1066	6.968673704	33.4326744	6.196483058	6.24229413	6.3321615
WCAP99-1074	7.203421958	43.87460873	10.46907668	6.44534263	6.45766799
WCAP99-1075	6.174182972	34.61682894	9.750195499	5.67781535	5.66061562
WCAP99-1079	7.821751174	40.57264619	9.548435579	6.3208314	6.31198016
WCAP99-1080	7.107119488	52.08638332	12.25924924	6.32222469	6.26393208
WCAP99-1090	6.30006911	31.82089218	10.14004121	5.67985537	5.69111191
WCAP99-1091	8.781142233	58.33277805	10.74020073	6.34358917	6.11641189
WCAP99-1092	7.831407599	49.98437305	6.301212611	6.10869565	5.97884077
WCAP99-1100	8.289021211	42.00966678	10.10484682	6.97749533	7.30888803
WCAP99-1106	6.403422545	36.69384428	8.942799891	5.953125	5.81683134
WCAP99-1111	6.860494744	40.74262446	10.77469286	6.27920162	6.25508524

Table A2 (con't). Northern California Dataset used for Linear Regressions.