Water quality modeling based on landscape analysis: importance of riparian hydrology

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Stockholm 2010

ISSN: 1653-7211
ISBN: 978-91-7447-135-9

Layout: Claudia Teutschbein (except Paper I and III)

Cover picture (by Thomas Grabs): Krycklan river as seen from the highway 363 bridge between Vindeln and Umeå (64°8’N, 19°56’E).

Printed by PrintCenter US-AB

Doctoral Dissertation 2010
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Abstract

Several studies in high-latitude catchments have demonstrated the importance of near-stream riparian zones as hydrogeochemical hotspots with a substantial influence on stream chemistry. An adequate representation of the spatial variability of riparian-zone processes and characteristics is the key for modeling spatio-temporal variations of stream-water quality. This thesis contributes to current knowledge by refining landscape-analysis techniques to describe riparian zones and by introducing a conceptual framework to quantify solute exports from riparian zones. The utility of the suggested concepts is evaluated based on an extensive set of hydrometric and chemical data comprising measurements of streamflow, groundwater levels, soil-water chemistry and stream chemistry.

Standard routines to analyze digital elevation models that are offered by current geographical information systems have been of very limited use for deriving hydrologically meaningful terrain indices for riparian zones. A model-based approach for hydrological landscape analysis is outlined, which, by explicitly simulating groundwater levels, allows better predictions of saturated areas compared to standard routines. Moreover, a novel algorithm is presented for distinguishing between left and right stream sides, which is a fundamental prerequisite for characterizing riparian zones through landscape analysis. The new algorithm was used to derive terrain indices from a high-resolution LiDAR digital elevation model. By combining these terrain indices with detailed hydrogeochemical measurements from a riparian observatory, it was possible to upscale the measured attributes and to subsequently characterize the variation of total organic-carbon exports from riparian zones in a boreal catchment in Northern Sweden. Riparian zones were recognized as highly heterogeneous landscape elements. Organic-rich riparian zones were found to be hotspots influencing temporal trends in stream-water organic carbon while spatial variations of organic carbon in streams were attributed to the arrangement of organic-poor and organic-rich riparian zones along the streams. These insights were integrated into a parsimonious modeling approach. An analytical solution of the model equations is presented, which provides a physical basis for commonly used power-law streamflow-load relations.

Key words: Water quality model, terrain analysis, geographical information system GIS, riparian zone, total organic carbon TOC, boreal catchments

Preface

Tuesday, 9th September 2008, Autumn snapshot campaign in Krycklan

"Why is the river so brown?" asked Steve L..

He said it with a mildly ironic tone in his voice while the two of us were looking down from the highway bridge into the slowly running Krycklan River beneath. I was perplex and babbled something meaningless in reply, while an overwhelming fountain of perceptual models, mathematical equations and chemical symbols popped into my mind. Still looking down into the brown waters of Krycklan, my thoughts were spinning: "What?! All these complex ideas were only there to answer such a simple question?" After my head had cleared, I pulled out my camera and took a picture (Cover photo). "Hopefully one day I will come up with a simple, yet elegant answer." I thought while Steve was returning from the car with some of the sampling equipment.

This thesis is an attempt to give a partial answer to why the Krycklan River was so brown that day. But it is not going to be as sweet and simple as I had thought in my daydream.

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This doctoral thesis consists of four appended papers and a synthesis. The synthesis summarizes the four papers and an additional fifth paper ('Development and test of the TRIM approach'), which is currently in progress.

List of Papers

I. **Grabs, T.**, Seibert J., Bishop K. and Laudon H. 2009: Modeling spatial patterns of the saturated areas: A comparison of the topographic wetness index and a distributed model. *Journal of Hydrology* 373, 15-23.

II. **Grabs, T. J.**, Jencso, K.G., McGlynn, B.L., Seibert J. 2010: Calculating terrain indices along streams - a new method for separating stream sides. *Water Resources Research* (in press, doi:10.1029/2010WR009296).

III. Seibert, J., **Grabs, T.**, Köhler S., Laudon, H., Winterdahl M. and Bishop K. 2009: Linking soil- and stream-water chemistry based on a Riparian Flow-Concentration Integration Model. *Hydrology and Earth System Sciences* 13, 2287-2297.

IV. **Grabs, T.**, Bishop K., Laudon, H., Lyon, S. W. and Seibert, J.: Riparian-zone processes and soil-water total organic carbon (TOC): Implications for spatial variability, upscaling and carbon exports. *Manuscript*.

Co-authorship

In all papers except Paper III, I developed, implemented and performed all numerical simulations, designed all figures (apart from Figure 9 in Paper II) and had the lead responsibility for the writing. Throughout the entire process I received highly valuable advice and feedback from my co-authors. I contributed to paper III by developing an analytical solution for a specific case of the presented model and by writing some of the related sections. I was also the main responsible for organizing and carrying out much of the fieldwork related to establishing and sampling a riparian observatory. I also helped conducting 'snapshot' stream-sampling campaigns in the Krycklan catchment. Most of the chemical analyses of water samples were done by staff at the Swedish University of Agricultural Sciences in Umeå.

Thomas Grabs
Stockholm, August 2010

Contents

Abbreviations

DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
GIS	Geographical information system
HBV	Lumped hydrological model by Bergström (1976)
HBV^{Raster}	Spatially distributed, raster-based version of HBV
HRS	Hillslope-riparian-stream
KCS	Krycklan catchment study
LiDAR	Light detection and ranging: An optical remote sensing technique
MD8	Multiple flow accumulation algorithm by Quinn et al. (1991)
MD∞	Multiple flow accumulation algorithm by Seibert and McGlynn (2007)
MWI	Model-based wetness index
MWI_{hbv}	MWI obtained from original HBV soil-moisture routine
MWI_{mod}	MWI obtained from modified HBV soil moisture routine
MWI_{steady}	MWI obtained from a steady-state run
POC	Particulate organic carbon
RIM	Riparian flow-concentration integration model
ROK	Riparian observatory in the Krycklan catchment
RZ	Riparian zone
S-transect	Riparian transect at Västrabäcken catchment
SIDE	Stream-index division-equations – a new algorithm for terrain analysis
TOC	Total organic carbon
TRIM	Topographic riparian flow-concentration integration model
TWI	Topographic wetness index: A proxy for shallow groundwater tables

Symbols

a_c	Specific contributing area: A proxy for shallow groundwater flow
a_c^H	Specific hillslope contributing area
a_c^R	Specific riparian contributing area
b	Transmissivity shape parameter
c_{TOC}	Soil-water TOC concentration
c_{TOC}^0	Soil-water TOC concentration at the soil surface
c_X	Soil-water concentration of a solute X
c_X^0	Soil-water solute concentration at the soil surface
Δz	Depth interval
f	Concentration shape parameter
k_0	Transmissivity parameter used in the RIM
$k_0^{'}$	Transmissivity parameter used in the TRIM
l_X	Load respectively mass flux per unit time of solute X
η	Power-law parameter
$\tan\beta$	Local terrain slope
q	Water flow rate
t	Parameter used to scale TOC concentrations in the TRIM
v_Q	Transformed flow-variable
w_R	Width (along the stream channel) of a riparian location
z	Depth
z_0	Offset parameter
z_{base}	Total soil profile depth
z_{gwt}	Groundwater-table position
$\bar{z}_{gwt}$	Median groundwater-table position
A	Catchment area
C_x	Stream-water solute concentration
$K(z)$	Hydraulic conductivity as a function of depth
L_X	Total load of a solute transported in a stream
Q	Streamflow at the catchment outlet
Q_R	Riparian groundwater discharge
R/H	Riparian-to-hillslope ratio

1 Introduction

Only if water is available in both sufficient quantity and quality, complex ecosystems can evolve to sustain human societies. The term 'water quality' refers to the collective physicochemical and biological properties of water and rates them based on the needs of a certain species or (eco) system. Since human welfare depends on 'good' water quality and well-functioning ecosystems, many scientific efforts have been made to improve the understanding of processes that regulate water quality. The number of potentially involved processes is, however, vast. This has often led to the development of water-quality models that are either highly complex and process-based or very simple and based on conceptual ideas. Complex process-based models are hardly applicable in practice because their enormous requirements of measured field data are difficult to fulfill. Simple conceptual models, on the other hand, are easy to apply but their internal conceptual components can hardly be verified, because they rarely correspond to quantities that are directly measurable by field experiments.

This thesis introduces a simple, yet process-based approach to partially solve this fundamental dilemma of water-quality models under certain assumptions. The principal assumption is that processes in near-stream riparian zones are dominant controls on many aspects of stream-water quality including concentrations of organic carbon, hydrogen cations (pH), heavy metals or organic pollutants. This assumption seems to be fulfilled at least in some high-latitude glacial-till environments as indicated by previous studies (Bishop et al., 1990; Fiebig et al., 1990; Dosskey and Bertsch, 1994; Bishop et al., 1995; Hinton et al., 1998; Cory et al., 2007). By focusing on dominant riparian-zone processes and in particular on organic carbon (Shafer et al., 1997; Hruška et al., 2003) it is possible to considerably reduce unnecessary complexities and, thus, create process-based water-quality models that can be used in practice.

1.1 Riparian zones

Almost all water flowing in natural streams and rivers has at one point in time travelled trough the riparian zone (RZ). Riparian zones (*ripa* is the Latin word for *streambank* or *shore*) are the strips of land that border every stream and river. Being located at the interface between land and water, RZs are therefore the last stage before groundwater mixes with water transported in the stream network.

From a hydrological landscape perspective, riparian zones are usually part of discharge areas located at valley bottoms or floodplains. Since discharge areas normally cover only a small fraction of the landscape (between 3-25% for the Krycklan and Tenderfoot creek catchments in this study), groundwater that flows from the remaining recharge areas towards the discharge areas converges strongly. Due to the resulting concentrated groundwater inflows, RZs are often characterized by high levels of groundwater and soil moisture compared to the surrounding landscape. Their proximity to the stream network as well as the combination of elevated groundwater tables and humid soils is what makes RZs key areas for fast generation of runoff (Hewlett and Hibbert, 1967; Dunne and Black, 1970; Sklash and Farvolden, 1979). The quick response of RZs to precipitation and snowmelt events is even more pronounced in the presence of soils with strongly increasing hydraulic conductivities towards the surface. Such conductivity profiles form the basis of the so-called 'transmissivity-feedback mechanism' in which small changes of the groundwater tables translate into disproportionally large changes of lateral groundwater flows to streams (Rodhe, 1989; Bishop, 1991; Kendall et al., 1999). As a consequence of the transmissivity-feedback mechanism, old pre-event groundwater from previously unsaturated horizons is mobilized and contributes considerably to the total streamflow even during periods of peak flow (Sklash and Farvolden, 1979; Rodhe, 1989; Bishop, 1991; Buttle, 1994). When accounting also for the water flowing as superficial flow on top of RZs, up to 100% of the observed streamflow originates in RZs. RZs are therefore crucial areas for the control of short term changes of stream-water quality (Cirmo and McDonnell, 1997).

Ecologically seen, riparian zones are transition zones or 'ecotones' at the land-water interface. The riparian ecotones can be imagined as semi-permeable membranes (i.e. selective barriers), which regulate the transfer of energy and matter from land to streams (Naiman and Decamps, 1997). Their distinct hydrological conditions are often reflected by riparian vegetation (Nilsson et al., 1991; Jansson et al., 2007b) and soils (Hill, 1990) that form the building blocks of the 'riparian membrane'. In particular the high degree of soil humidity coupled with anoxic conditions and the exposure to infrequent floods represent extreme conditions that are only tolerated by certain well-adapted plant species and microbes. Decomposition rates of plant debris and other organic matter are typically low in RZs due to the anoxic conditions that effectively limit microbial respiration. If production rates of organic matter by plant photosynthesis exceed (microbial) decomposition rates and fluvial erosion at certain RZs, organic matter starts to accumulate. In addition to producing organic matter, riparian vegetation can further structure soils and stream banks by reducing their erosivity and by intercepting lateral sediment transport from upslope areas. The formation of riparian soils and RZ structures eventually feeds back on the initial (hydrological) controls on riparian ecology by influencing hydrological conditions and flow pathways. The development of riparian peat and mosses (*sphagnum spp.*) in boreal regions (Moore and Bellamy, 1974; Rundgren, 2008) is an excellent example for the coevolutionary relationships between hydrological, ecological and pedological conditions in RZs. Living uncompressed mosses are highly permeable and can store considerable quantities of infiltrating water. Below the top layer of living plants, however, partially decomposed mosses form peat layers that are increasingly compressed over time. As a result, deep peats have high bulk densities, substantially reduced field capacities and permeabilities, which cause peats to hydrologically behave comparable to fine-grained glacial tills (Heinselman, 1975). The altered hydrological conditions eventually induce groundwater tables to rise. This mechanism underlies the process of paludification in which wet zones with anoxic conditions are extended vertically (and sometimes laterally) and, thus, allow further peat development. Since increasing soil moisture negatively affects forest production, it is very common to find artificially deepened streams in boreal regions where forestry is practiced, as is the case for many forests in Scandinavia. In other words, in many parts of the world RZs can often not be well understood without also considering the 'human factor' in addition to natural processes.

1.2 The role of riparian aqueous organic carbon for surface water quality

Riparian soils that emerged from the previously described long-term interplay of hydrological and biological processes can be seen as biogeochemical reactors that chemically modulate lateral groundwater flows and thereby control short-term variations of stream-water chemistry. A key component in the aqueous phase contained by the 'soil reactor' (i.e. the soil water) is aqueous organic carbon. Several studies (including this thesis) have highlighted RZs as dominant sources of aqueous organic carbon exported to surface-water systems in high-latitude landscapes (Bishop et al., 1990; Fiebig et al., 1990; Dosskey and Bertsch, 1994; Hinton et al., 1998). While still a subject of debate, the immediate origins of aqueous organic carbon are likely to be a combination of root exudates and solubilized soil organic matter. Technically, the total organic carbon (TOC) in the aquatic phase can be classified by size into particulate carbon (POC) and dissolved organic carbon (DOC). The distinction between POC and DOC is based on which fraction of TOC passes a filter of usually 0.45 µm. TOC, POC and DOC are collective terms comprising an immense variety of molecules and molecular composite structures (Thurman, 1985; Drever, 1997; vanLoon and Duffy 2005), which have in common that they contain carbon molecules and originate from a once living organism. Most aqueous carbon in boreal headwaters is allochthonous (i.e. produced outside the surface-water ecosystem) and has been exported to inland waters primarily as DOC, as dissolved inorganic carbon (DIC) and to a lesser degree as POC (Meybeck, 1993; Hope et al., 1994). The flux of carbon through inland waters (i.e. streams and lakes) plays an important role in the global carbon cycle, which has often been overlooked (Cole et al., 2007) even though inland waters receive up to $2.7 \cdot 10^{12}$ kg of carbon per year. This number corresponds approximately to terrestrial carbon sinks available for anthropogenic emissions (Battin et al., 2009). Aqueous organic carbon is not only an important component in the global carbon cycle but also directly affects water quality and ecosystem functions. In particular its DOC fraction plays an essential role by directly influencing the energy balance of surface-water ecosystems. This is because (1) DOC comprises carbon molecules that are a major source of energy (and carbon) for heterotrophic water microbes (Wetzel, 1992; Berggren et al., 2009) and because (2) DOC 'dyes' the water and thereby reduces the penetration of light as well as the productivity of autotrophic, photosynthetic organisms (Jansson et al., 2007a; Karlsson et al., 2009). In addition to affecting water quality by its physical properties, the chemical characteristics of DOC play an equally important role by acidifying surface waters (Kullberg et al., 1993; Laudon and Bishop, 1999; Hruška et al., 2001) and mobilizing a variety of pollutants including pesticides and metals (Thurman, 1985; Drever, 1997; vanLoon and Duffy 2005). In summary, aqueous organic carbon (and DOC in particular) is the 'chemical lever' by which RZs affect both water quality and carbon cycling.

1.3 Characterizing riparian zones through landscape analysis

River landscapes are rarely homogeneous, but consist of a heterogeneous mosaic of different RZs. Characteristic differences of RZs (such as the amount of accumulated soil organic matter) emerge from spatially variable environmental processes that originally formed the RZs. These processes are often related to the surrounding landscape and especially to topography (Moore et al., 1991). By analyzing spatial datasets such as digital elevation models (DEMs), it is often possible to locate and (partially) quantify a large variety of processes including solar radiation, (orographic) precipitation (Wilson and Gallant, 2000), snow dynamics (Hock, 2003), shallow groundwater flows (Beven and Kirkby, 1979; McGuire et al., 2005), hydrological connectivity (Jencso et al., 2009), solute transport (Darracq and Destouni, 2005), soil formation (Seibert et al., 2007), biological activity (Gardner and McGlynn, 2009; Riveros-Iregui and McGlynn, 2009) and vegetation (Zinko et al., 2005). Most current geographic information systems (GIS) provide the means to calculate terrain indices and to derive groundwater flow pathways from DEMs (O'Callaghan and Mark, 1984; Freeman, 1991; Quinn et al., 1991; Costa-Cabral and Burges, 1994; Tarboton, 1997; Seibert and McGlynn, 2007; Gruber and Peckham, 2009). The availability of GIS systems and DEMs has therefore led to a wide-spread and often successful use of terrain indices to quantify natural phenomena. In regard to RZs, terrain indices related to (ground-)water flows (Tarboton and Baker, 2008) show great promise since the formation and functioning of RZs is largely dictated by upslope hydrological controls (Vidon and Smith, 2007). Despite these opportunities, the calculation of (flow-related) terrain indices for RZs has been difficult because until recently no GIS routine was available to distinguish between RZs located on left and on right sides of streams. Improving and applying landscape-analysis methods to derive terrain-indices-related RZ characteristics was, thus, a central aspect of this thesis. One particular challenge was to develop a method to distinguish between RZs located on left and on right sides of streams when computing terrain indices.

1.4 Modeling stream-water quality with focus on riparian processes

A lot of models have been created to predict quantities related to stream-water quality (Srinivasan and Arnold, 1994; Arheimer and Brandt, 1998; Whitehead et al., 1998; de Wit, 2001; Lindström et al., 2010). These models can be generally categorized based on whether an empirical or mechanistic approach was chosen and on how the model represents space. Empirical models typically rely on an observed relation between the quantity that is to be predicted (predictant) and one or more (observed) quantities that are used for its prediction (predictors). A well-established empirical model to predict stream DOC concentrations is for instance the usage of wetlands percentage in a catchment as predictor (Dillon and Molot, 1997; Creed et al., 2003; Creed et al., 2008; Ågren et al., 2010). Mechanistic models, on the other hand, try to link predictors to predictants in the context of a theoretical, physically-motivated framework. An example for one of the most widely used 'mechanistic' modeling approaches in hydrology is the application of 'Darcy's Law' to link groundwater tables to groundwater flows. Most (if not all) hydrological models are, however, at best 'pseudo-mechanistic' approaches since they operate almost exclusively at scales that are far beyond those for which physical laws were originally developed.

Both empirical and mechanistic models can represent different spatial dimensions and, thus, can be accordingly classified as spatially distributed, semi-distributed or lumped models. Spatially distributed models are normally used to simulate processes in two or three dimensions, while lumped models do the same but for only one dimension and thereby implicitly assume spatially homogenous conditions. Semi-distributed models contain, as their name suggests, both spatially variable and spatially invariant (homogenous) components. Models that are truly spatially distributed are rare in reality because their immense data requirements are hardly ever met. If all models were strictly classified, chances are that there would be no spatially distributed model left. In the context of this study, the three model classes are, thus, defined based on the spatial dimension of their main outcome.

Modeling stream-water quality and water quality in general can be a daunting task because of the many physical and chemical processes that are potentially involved. Many water-quality models are therefore often complex and involve a large number of parameters in an attempt to capture as many of these processes as possible. If, however, the assumption holds that the RZ controls stream-water DOC together with a lot of other chemical parameters related to DOC, then much of the complexity found in other models is unnecessary. Firstly, since RZs cover only a relatively small and constrained area in a catchment, processes that occur in all other parts of the catchment do not need to be accounted for. Secondly, dynamic hydrological processes in RZs (such as groundwater-table variations) are closely linked to temporal variations in streamflow (Seibert et al., 2002) due to the proximity of RZs to streams. Hydrological processes in RZs can therefore be to some degree directly inferred from modeled or measured streamflow without requiring additional data such as precipitation or temperature. Finally, if DOC exports from RZs can be reasonably well modeled then several other water-quality parameters (e.g. heavy-metal concentrations) can be easily obtained because of their correlation to DOC.

2 Thesis objectives

The aim of this thesis was to contribute to the understanding of spatial and temporal variations of surface-water chemistry at the landscape scale. In particular, the functioning of riparian zones and their influence on stream-water chemistry was investigated. The thesis objective was divided into three principle stages:

1. Using and refining landscape-analysis techniques to describe the riparian zone and to extract hydrologically important information on spatial patterns and connections between different landscape units (Paper I and II).
2. Formulating the simplest set of conceptual model routines that adequately describes hydrological conditions and their impact on the flow of TOC from the riparian zone and other dominant landscape units in boreal catchments (Paper III, and V).
3. Testing the hypothesis that predictable differences in the riparian-zone soil chemistry and hydrology can explain stream-chemistry variability in the forested boreal landscape (Paper IV and V).

3 Overview of papers

3.1 Model-based landscape analysis

Paper I:

Grabs, T., Seibert J., Bishop K. and Laudon H. 2009: Modeling spatial patterns of the saturated areas: A comparison of the topographic wetness index and a distributed model. *Journal of Hydrology* 373, 15-23.

Most terrain indices calculated by standard landscape-analysis methods are derived from surface topography. This is well suited for geomorphological studies, which indeed aim at quantifying the topographic properties of land surfaces. In the context of hydrology, however, topography is normally used as a proxy for groundwater-table positions, which is not always appropriate (Haitjema and Mitchell-Bruker, 2005). Therefore this paper compares wetness indices derived from standard landscape analysis to wetness indices calculated by a distributed hydrological model, which explicitly simulates groundwater tables.

3.2 A new method to derive terrain indices for riparian zones

Paper II:

Grabs, T. J., Jencso, K.G., McGlynn, B.L., Seibert J. 2010: Calculating terrain indices along streams - a new method for separating stream sides. *Water Resources Research* (in press, doi:10.1029/2010WR009296).

A short-coming of landscape-analysis routines offered by current GIS programs has been the lack of a possibility to differentiate between left and right sides of a stream when calculating terrain indices. This is a severe limitation for assessing RZs, because RZs located on opposite stream sides can be fundamentally different. An innovative algorithm ('SIDE', **s**tream-**i**ndex **d**ivision-**e**quations) to efficiently calculate terrain indices separately for stream sides is therefore presented in this paper and its application to a mountainous catchment in Montana, USA, is demonstrated.

3.3 The riparian flow-concentration integration model (RIM)

Paper III:

Seibert, J., **Grabs, T.**, Köhler S., Laudon, H., Winterdahl M. and Bishop K. 2009: Linking soil- and stream-water chemistry based on a Riparian Flow-Concentration Integration Model. *Hydrology and Earth System Sciences* 13, 2287-2297.

This paper introduces a new mathematical framework to link soil-water chemistry of RZs to stream-water chemistry. The proposed **r**iparian flow-concentration **i**ntegration **m**odel (RIM) distinguishes itself from conventional box-models by explicitly accounting for the vertical distribution of lateral flows and solute concentrations in the soil profile. The general RIM concept was tested and validated against detailed field observations.

3.4 Heterogeneity and scaling of riparian-zone processes

Paper IV:

Grabs, T., Bishop K., Laudon, H., Lyon, S. W. and Seibert, J.: Riparian-zone processes and soil-water total organic carbon (TOC): Implications for spatial variability, upscaling and carbon exports. *Manuscript.*

Research on RZ functioning at the Krycklan and Svartberget catchments in Northern Sweden has been primarily focused on studying a single riparian transect for almost two decades. This paper investigates the catchment-scale variability of RZs based on hydrogeochemical observations from a riparian observatory. In particular the functioning of different RZ types is illuminated in regard to their hydrology and soil-water TOC characteristics. Moreover, a detailed landscape analysis of RZ characteristics is shown to provide the necessary means for upscaling riparian carbon-export processes from the plot scale to the catchment scale.

3.5 Development and test of the TRIM approach

Paper V:

Manuscript in progress.

The results presented in the four previous papers are complemented by the development and test of a **t**opographic **r**iparian flow-concentration **i**ntegration **m**odel (TRIM). The TRIM effectively combines results from Papers II-IV to extend the RIM approach presented in Paper III by scaling it to the stream network. This theoretically allows simulating temporal variations of stream chemistry at every point in the stream network based on simple metrics derived from a DEM. The manuscript also includes an evaluation of the TRIM against spatially distributed hydrogeochemical data from streams and RZs following a similar testing strategy as in Paper III.

4 Material and Methods

4.1 Study locations

The theoretical concepts in this study were developed and assessed based on data from two catchments located in northern Sweden (Papers I, III, IV and V) and in Montana, USA (Paper II).

4.1.1 Krycklan catchment, Sweden

The Krycklan catchment is a 67 km^2 basin situated partially within the Svartberget Experimental Forests (64° 14'N, 19° 46'E) about 50 km northwest of Umeå, Sweden (Figure 1). The catchment geology can be described as poorly weathered gneissic bedrock covered with sediment deposits at lower elevations and moraine (glacial till) deposits at higher elevations. The topography is gentle with elevations ranging from 126 to 372 m.a.s.l.. The annual air temperature averages 1.8°C and the mean annual precipitation is 623 mm of which approximately one third falls as snow (Ottosson Löfvenius et al., 2003). Forests cover most of the catchment area (88 %) followed by wetlands (8 %), agricultural land (3 %) and lakes (1%). Streamflow and stream chemistry have been monitored for three decades at the outlet of the nested 50 ha Svartberget catchment. The Svartberget catchment has also been sometimes (less accurately) referred to as 'Nyänget' or 'Kallkällsbäcken' catchment in other studies (e.g. Nyberg et al., 2001; Bishop et al., 2004; Paper I). In 1992, the so called S-transect was built to additionally monitor riparian soil-water chemistry at a single RZ in the Västrabäcken catchment (Bishop et al., 1994; Laudon et al., 2004; Cory et al., 2007). Ten years later the monitoring program was expanded from the nested Svartberget catchment to the mesoscale Krycklan catchment as part of the multidisciplinary Krycklan Catchment Study (KCS). At present, the KCS comprises several monitoring programs including monitoring of streamflow, water quality, snow depths and stream ecology. In autumn 2007 the riparian observatory

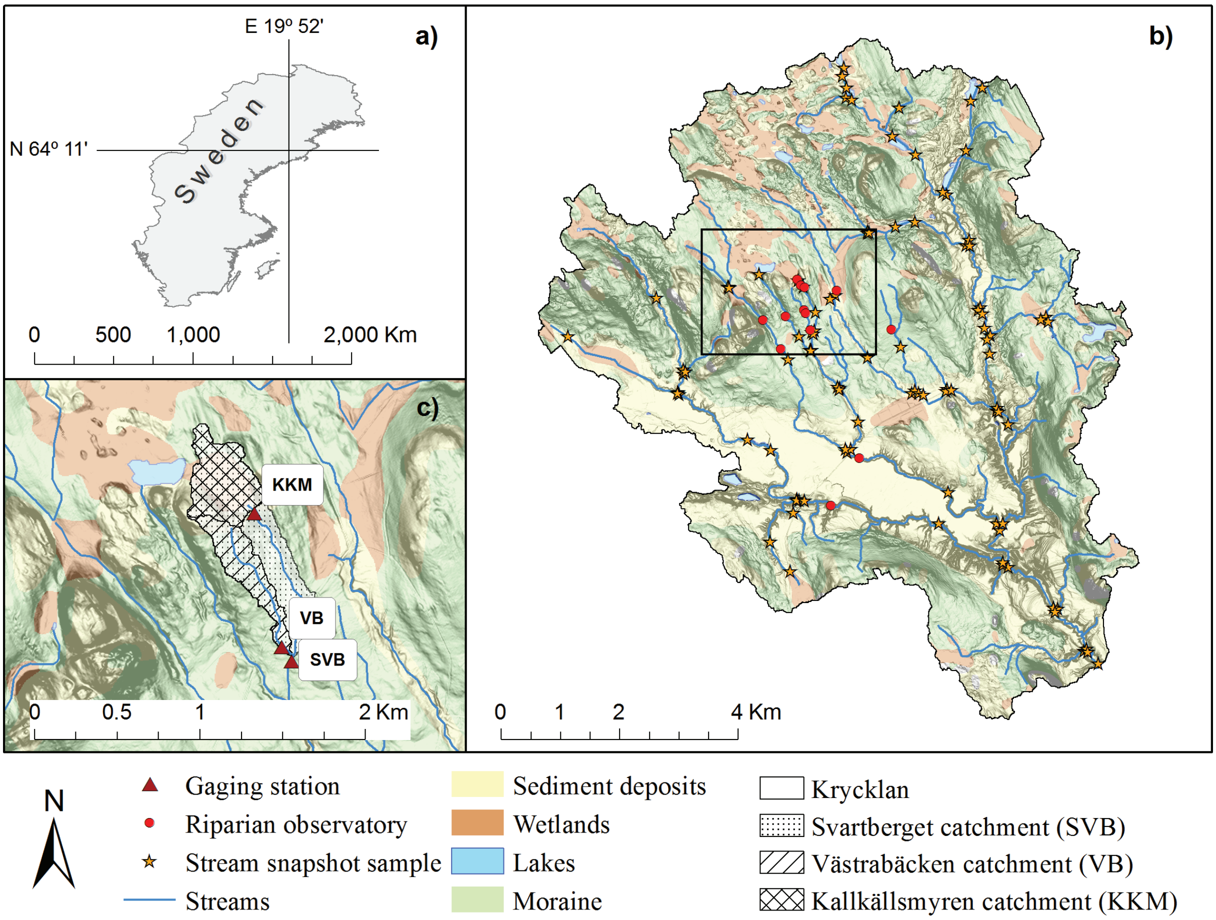

Figure 1: Overview of characteristics of the Krycklan catchment. Position of the Krycklan catchment (inset a), overview of surficial substrate, streams as well as locations of stream chemistry (snapshot-)sampling sites and riparian monitoring sites in the Krycklan catchment (inset b). The Svartberget catchment (SVB) including the Västrabäcken (VB) and Kallkällsmyren (KKM) subcatchments and gaging stations are shown in inset c). The rectangle in inset b) outlines the position of inset c).

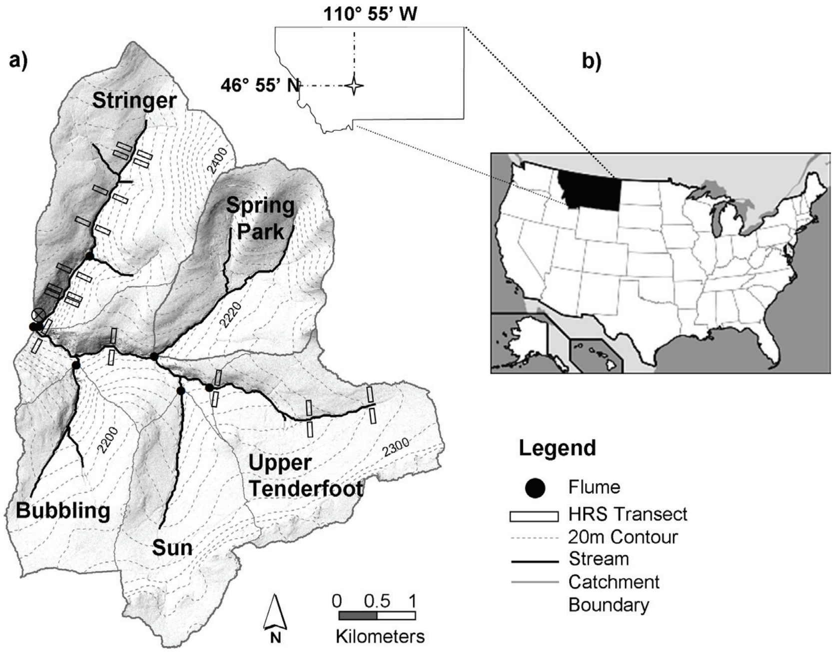

Figure 2: Overview of the Tenderfoot Creek catchment (a) and its position (b), modified from Jencso et al. (2009).

in the Krycklan catchment (ROK) was constructed (Paper IV) to allow monitoring RZs also at other locations in Krycklan (Figure 1), which was a central aspect for this thesis. The ROK consists of 13 sites (equipped with wells and suction lysimeter nests), which were selected based on a preliminary terrain analysis and several field visits.

4.1.2 Tenderfoot Creek catchment, USA

The 22.8 km^2 Tenderfoot Creek catchment is located in the Little Belt Mountains of the Lewis and Clark National Forest in Central Montana (46° 55'N, 110° 52'W), USA (Figure 2). Parent material consists of Flathead Sandstone and Wolsey shale in the upper elevations and of granite gneiss at lower elevations (Farnes et al., 1995). Elevation ranges from 1985 to 2426 m.a.s.l. with short, steep hillslopes along the main stem and long, moderately inclined hillslopes in headwater areas. The mean annual air temperature is 0°C and annual average precipitation amounts to 840 mm (averaged over period 1961-1990) of which 70 % falls as snow (Farnes et al., 1995) .

4.2 Field measurements

Field measurements provided the basis for all five papers. The used field data can be broadly classified into hydrometric data (comprising streamflow and groundwater-table records) and water-quality data (covering soil- and steam-water chemistry measurements). All field measurements, except those from Tenderfoot Creek (Paper II), were collected as part of the Krycklan catchment study (Table 1).

Table 1: Overview of field data and measurements used in the five papers of this thesis.

	Hydrometric data	Water-quality data
Paper I	· Streamflow (Västrabäcken, Kallkällsmyren and Svartberget)	· none
Paper II	· Groundwater levels (Tenderfoot Creek) · Streamflow (Tenderfoot Creek)	· none
Paper III	· Groundwater levels (S-Transect) · Streamflow (Västrabäcken)	· Soil-water chemistry (S-transect) · Stream-water chemistry (Svartberget)
Paper IV	· Groundwater levels (ROK) · Streamflow (Svartberget)	· Soil-water chemistry (ROK)
Paper V (in progress)	· Groundwater levels (ROK) · Streamflow (Svartberget)	· Soil-water chemistry (ROK) · Stream-water chemistry (Snapshot campaign, September 2008)

Streamflow was measured at the outlets of three nested subcatchments (Västrabäcken, Kallkällsmyren and Svartberget) in Krycklan (Figure 1) and seven subcatchments in the Tenderfoot Creek catchment (Figure 2). Stream-water levels were recorded continuously using automated loggers installed upstream of various gaging devices (three V-Notch weirs in Krycklan as well as four Parshall and three H-flumes in Tenderfoot). Between 20 to 50 direct streamflow measurements obtained from salt-slug injections (Moore, 2004) at each gaging site were used to derive rating curves, which were needed to convert stream water-level records to streamflow time series. Streamflow data was complemented by high-frequency records of groundwater tables measured at 13 wells at the riparian observatory in Krycklan (Paper IV), at one well installed at the S-transect in Krycklan (Paper III) and at 24 hillslope-riparian-stream (HRS) well-transects in the Tenderfoot creek catchment (Paper II). Hydrometric data of Tenderfoot creek was further used to determine hydrologic connectivity between HRS zones at each of the 24 transects. A HRS hydrologic connection was defined as a time interval during which streamflow occurred and both riparian and hillslope wells of a transect recorded water levels above bedrock (Jencso et al., 2009).

The water-quality data used in this study was derived from standard chemical analyses (Buffam, 2007) of stream-water and soil-water samples. Stream-water samples were taken frequently at the outlet of Västrabäcken (Paper III) as well as during a 'snapshot campaign' in September 2008 (Spans, 2010). The snapshot campaign was carried out over a period of four days of stable low-flow conditions (streamflow: ~0.55 mm·day^{-1}) during which 72 stream-water samples were collected at different locations along the stream network in Krycklan (Figure 1). Soil-water samples were regularly extracted from suction lysimeters at the S-Transect (Paper III) as well as during nine occasions (six in 2008, three in 2009) from suction lysimeters at the 13 ROK sites (Paper IV).

4.3 Landscape analysis

Landscape analysis in the Krycklan catchment was supported by a quaternary-deposits coverage map (1:100 000, Geological Survey of Sweden, Uppsala, Sweden) and a coarse, gridded DEM of 50 m resolution (Paper I) that was later replaced by a high-resolution (5 m grid-spacing) DEM derived from 1 m resolution airborne light detection and ranging (LiDAR) data (Paper IV). Terrain indices in the Tenderfoot catchment (Paper II) were calculated from a 10 m gridded DEM, which had also been derived from 1 m resolution airborne LiDAR data (Jencso et al., 2009). Key terrain indices used in this study include terrain slope $\tan\beta$, specific upslope contributing area a_c (a proxy for shallow groundwater flow), and the topographic wetness index $\mathrm{TWI} = \ln(a_c/\tan\beta)$ (Kirkby, 1975). Landscape analysis was not performed for the analyses in Paper III.

4.3.1 Model-based landscape analysis

Two variants of the TWI were calculated from a 50 m resolution DEM of Krycklan (Paper I). The first variant $\mathrm{TWI}_{\mathrm{MD8}}$ was derived from specific contributing area a_c and local slope $\tan\beta$ both calculated as described by Quinn et al. (1991). The second variant $\mathrm{TWI}_{\mathrm{MD}\infty}$ was based on the MD∞ flow-accumulation algorithm (Seibert and McGlynn, 2007) and on the downslope index method (Hjerdt et al., 2004) to compute values of a_c and $\tan\beta$. The performance of both TWI variants to predict saturated areas (represented by wetlands and lakes shown by the quaternary-deposits coverage map, Figure 1) was evaluated based on topological metrics (McGarigal and Marks, 1995), different similarity coefficients (Creed et al., 2003; Sheskin, 2003; Güntner et al., 2004; Dahlke et al., 2005) as well as on a statistical evaluation of the distributions of TWI values using the Kolmogorov-Smirnov measure (Rodhe and Seibert, 1999). Topological metrics were used to characterize the degree of fractionation of patches of saturated areas in the landscape while similarity coefficients quantified the degree of agreement (respectively disagreement) between mapped and predicted locations of saturated areas. The capacity of the TWI variants to separate saturated areas from surrounding land was expressed by the Kolmogorov-Smirnov measure. The performance of the TWI was subsequently compared against a distributed hydrological model.

4.3.2 A new method to derive terrain indices for riparian zones

A novel landscape-analysis algorithm was developed (Paper II) and implemented into the open-source SAGA GIS program (Conrad, 2007; Böhner et al., 2008). This so called stream-index division-equations (SIDE) algorithm relies on vector algebra to distinguish between left and right sides of a stream network, which is crucial for deriving terrain indices for RZs.

To analyze the effect of the SIDE algorithm, specific hillslope contributing-area values a_c^H were computed for left ($a_{c,L}^H$) and for right ($a_{c,R}^H$) sides along the entire stream network of the Tenderfoot Creek catchment by combining the SIDE algorithm with the MD∞ flow algorithm (Seibert and McGlynn, 2007). Hillslope contributing area refers to the area of a hillslope contributing laterally to a single stream segment and does not include any area entering from upstream stream segments. Specific hillslope contributing-area values are obtained by dividing hillslope contributing area by the length of the stream segment. All calculations of a_c^H values were performed on a 10 m resolution DEM and on a map showing the position and flow direction of streams in the Tenderfoot Creek catchment. This streamflow-direction map had been derived from the DEM by applying the 'Channel Network' module in SAGA GIS (Conrad, 2007; Böhner et al., 2008) using the DEM and a map of upslope area, where a threshold area of 40 ha defined stream initiation. Values of $a_{c,R}^H$ and $a_{c,L}^H$ values were compared by visual assessment of $a_{c,R}^H$ and $a_{c,L}^H$ maps and by plotting $a_{c,R}^H$ against $a_{c,L}^H$. Furthermore the use of the SIDE method to derive composite terrain indices was exemplified by calculating riparian-to-hillslope ratios (McGlynn and Seibert, 2003). The riparian-to-hillslope ratio R/H is a terrain metric that quantifies the capacity of RZs to biogeochemically or hydrologically modulate (or 'buffer') hillslope groundwater inflows. The R/H ratio was here defined as the ratio between the specific contributing riparian area a_c^R and the entire hillslope contribution a_c^H (Equation 1).

$$\frac{R}{H} = \frac{a_c^R}{a_c^H} \qquad 1$$

Values of a_c^R were calculated analogously to a_c^H after excluding parts of the DEM that did not belong to the mapped extend of RZs. The mapping of RZs in Tenderfoot is described in more detail in Paper II and by Jencso et al. (2009). R/H values calculated with the SIDE algorithm were compared to R/H values calculated without the SIDE algorithm by comparing the respective catchment-wide area-weighted distribution functions. The source code of the SIDE algorithm as well as a compiled SAGA-module along with usage instructions can be found at the author's website (http://thomasgrabs.com/side-algorithm).

4.3.3 Heterogeneity and scaling of riparian-zone processes

Soil-water TOC concentrations measured at sites of the riparian observatory in Krycklan were extrapolated along the entire stream network in the Krycklan catchment (Paper IV). The spatial information required for this extrapolation of the TOC concentrations was extracted from a quaternary-deposits map (1:100 000, Geological Survey of Sweden, Uppsala, Sweden) and from the TWI derived from a high-resolution (5 m) LiDAR DEM. More specifically, the TWI was calculated from terrain slope $\tan\beta$ (Zevenbergen and Thorne, 1987) and specific hillslope contributing areas a_c^H computed separately for each side of the stream using the MD∞ (Seibert and McGlynn, 2007) flow accumulation-algorithm coupled to the SIDE method developed in Paper II.

4.3.4 Development and test of the TRIM approach

The additional analyses presented in this thesis (Paper V in progress) rely exclusively on the three terrain indices derived in Paper IV (a_c^H, $\tan\beta$ and TWI).

4.4 Hydrological modeling

Three types of hydrological models were applied in this study, which together represent a full spectrum of spatial modeling approaches by ranging from a spatially distributed model (Paper I) to a semi-distributed model (Paper V) and several lumped models (Papers III and IV).

4.4.1 Model-based landscape analysis

Spatially distributed groundwater storages were simulated (Paper I) with the help of two spatially distributed variants of the HBV model (Bergström, 1976; Bergström, 1995; Seibert et al., 2003; Grabs et al., 2007). The original HBV model is a conceptual lumped model to efficiently simulate streamflow based on temperature and precipitation (Bergström, 1976). The structure of its soil-moisture routine was modified by Seibert et al. (2003) to allow for a dynamic coupling of saturated- and unsaturated-storage variables, which is important for modeling areas (including riparian zones) characterized by shallow groundwater tables. Grabs et al. (2007) implemented both HBV variants as spatially distributed raster-based models (HBV^{Raster}) using Darcy's Law to calculate flow and hydraulic gradients. The two distributed models were calibrated against streamflow measured at two internal subcatchments of the Svartberget catchment (Västrabäcken and Kallkällsmyren). By calibrating spatially homogenous model parameters, the spatial variability of simulated groundwater storages depended exclusively on topography represented by a 50×50 m DEM. Spatially distributed groundwater storages obtained from dynamic simulations of both model variants and from a steady-state model run were subsequently normalized and treated as model-based wetness indices (MWIs). MWIs were labeled according to the corresponding model concept (Table 2).

The performances of the three MWIs in predicting saturated areas (represented by wetlands and lakes) were evaluated in the same way as for the two previously introduced TWI variants, which had been determined by standard landscape analysis.

4.4.2 *A new method to derive terrain indices for riparian zones*

In Paper II, two linear regression models were fitted to observed durations of hillslope-riparian-stream hydrologic connections (predictants) and specific hillslope contributing area a_c^H (predictor). The first regression model relied on lumped a_c^H values (i.e. $a_c^H = a_{c,R}^H + a_{c,L}^H$) while the second regression model was based on side-separated $a_{c,R}^H$ and $a_{c,L}^H$ values (i.e. derived using the SIDE algorithm).

4.4.3 *The riparian flow-concentration integration model (RIM)*

The focus of Paper III was the development and testing of the so called riparian flow-concentration integration model (RIM). RIM is a parameter-parsimonious model concept designed to link stream-water chemistry to riparian soil-water chemistry. Although it is applied as lumped model (i.e. based on a single RZ 'representative' for all RZs) in Paper III, the general concept can also be applied in a distributed modeling approach. RIM is built on the idea that soil-water concentrations and lateral groundwater fluxes often vary systematically with depth. If both lateral flow q and the concentration of a certain solute c_X are known for a certain layer at depth z (and with a thickness of Δz), the mass of solute exported from that layer per unit of time, which is also referred to as solute load l_X , can be simply calculated by multiplying the lateral flow q with the solute concentration c_X and by the thickness Δz of the layer (Equation 2).

$$l_X(z) = c_X(z) \cdot q(z) \cdot \Delta z \qquad 2$$

In cases where the values of c_X and q are known across a whole soil profile of depth z_{base} the total load exported by a soil profile can be obtained by integrating flows and concentrations over depth (Equation 3).

$$l_X = \int_{z_{base}}^{0} l_X(z)\, dz = \int_{z_{base}}^{0} c_X(z) \cdot q(z)\, dz \qquad 3$$

The RIM model was tested by assuming that groundwater table and chemistry data collected at a single riparian soil profile at the S-transect would be representative for the entire RZ of the Västrabäcken catchment. This assumption implies that specific discharge (obtained by dividing streamflow by the corresponding catchment area), solute-concentration profiles $c_X(z)$ as well as lateral groundwater-flow profiles $q(z)$ are the same for all RZs in Västrabäcken. The RIM was further

Table 2: Overview of model concepts and corresponding MWI variants.

Model concept	Original HBV	Modified HBV	Steady-state run
MWI	MWI_{hbv}	MWI_{mod}	MWI_{steady}

simplified by additional assumptions motivated by field experience including:

- Exponential decreases of transmissivity with depth
- Negligible groundwater flows in the unsaturated zone and below one meter depth
- 100% soil-matrix groundwater flows and neither overland nor macropore flows
- Exponentially shaped soil solution profiles
- Time-invariant soil-solute concentrations at one meter depth

Based on these assumptions, streamflow Q measured at the outlet of Västrabäcken could be mathematically linked to groundwater-table positions z_{gwt} in the RZ. This was accomplished by fitting an exponential function involving a transmissivity shape parameter b and a transmissivity parameter k_0 (Equation 4; note that the parameter a used in Paper III is related to k_0 by $a = b \cdot k_0$).

$$Q(z_{gwt}) = k_0 \cdot e^{b \cdot z_{gwt}} \qquad 4$$

The lateral groundwater-flow profile needed for the RIM approach corresponds to the derivative over depth of the streamflow-groundwater-table relation (Equation 5).

$$q(z) = \frac{dQ}{dz} \qquad 5$$

Soil-solute concentration profiles were also expressed by an exponential function based on the solute concentration at the soil surface c_X^0 and a concentration shape parameter f (Equation 6).

$$c_x(z) = c_X^0 \cdot e^{f \cdot z} \qquad 6$$

By using exponential functions to describe groundwater flow and solute-concentration profiles and introducing a new parameter η (Equation 7), a simple analytical solution could be derived relating solute loads directly to streamflow (Equation 8).

$$\eta = \frac{b+f}{b} \qquad 7$$

$$l_X = \frac{c_0 \cdot k_0^{\,1-\eta}}{\eta} \cdot Q^{\eta} \qquad 8$$

Since both parameters of the streamflow-groundwater-table relation (k_0 and b) as well as the solute concentration at one meter depth could be directly inferred from field observations, calibration of the RIM was reduced to estimating the concentration shape parameter f . Values of f were subsequently estimated based on streamflow and stream solute concentrations measured at the catchment outlet. The back-calculation of the f parameter from stream observations allowed reconstructing theoretical concentration-depth profiles of solutes (TOC, Ca, Mg and Cl) in the RZ which could then be compared against independent soil-water chemistry measurements. In addition, seasonal changes of backwards calculated TOC-concentration profiles were examined.

4.4.4 Heterogeneity and scaling of riparian-zone processes

A slightly modified version of the RIM was used to calculate export rates of TOC per unit contributing area l^*_{TOC} (Equation 9) and TOC concentrations weighted by lateral groundwater flows $c_{TOC,q}$ at the 13 ROK sites (Paper IV). Important modifications included the use of TOC-concentration profiles constructed from soil-water TOC measurements, the addition of a groundwater-table offset parameter z_0 to the streamflow-groundwater-table relation (Equation 10) as well as the procedure used to estimate the parameter values of k_0, b and z_0.

$$l^*_{TOC} = \frac{l_{TOC}}{a_c^H \cdot w_R} \qquad 9$$

$$Q(z_{gwt}) = k_0 \cdot e^{b(z_{gwt} - z_0)} \qquad 10$$

Moreover, a robust linear-regression model was built to predict soil-water TOC-concentration profiles in the moraine part of the Krycklan catchment based on values of the topographic wetness index TWI (Equation 11).

$$c(z) = c_{TOC}^{0} \cdot e^{f \cdot z} \cdot TWI^{t} \qquad 11$$

4.4.5 Development and test of the TRIM approach

Finally some of the previously established landscape analysis methods were combined with the RIM approach (Paper III) to partly overcome its limitations as a lumped model (Paper V in progress). The fundamental principle of the new model conceptualization, hereafter referred to as **t**opographic **r**iparian flow-concentration **i**ntegration **m**odel (TRIM), is the topography-based scaling of RZ characteristics such as the scaling of lateral groundwater inflows and groundwater levels using specific lateral contributing area a_c and terrain slope $\tan\beta$ values calculated for each riparian position along a stream network.

The TRIM relies on the same assumptions as the RIM but further assumes that groundwater flows can be scaled with specific contributing area a_c and that hydraulic gradients can be approximated by terrain slope. Total groundwater discharge Q_R at a riparian location characterized by a soil depth z_{base} and a width w_R can be related to the groundwater-table position z_{gwt} based on Darcy's law and hydraulic conductivity $K(z)$ (Equation 12).

$$Q_R = w_R \cdot \tan\beta \cdot \int_{z_{base}}^{z_{gwt}} K(z)dz \qquad 12$$

Moreover, since spatially homogenous discharge rates are assumed to be scalable with a_c^H values derived from topography, the total groundwater discharge Q_R can be conveniently estimated based on streamflow Q measured at the catchment outlet and the catchment area A (Equation 13).

$$Q_R = w_R \cdot a_c^H \cdot \frac{Q}{A} \qquad 13$$

Riparian groundwater-table position z_{gwt} can be subsequently inferred from temporally variable streamflow by combining Equation 12 and Equation 13 and solving for z_{gwt} (Equation 14).

$$\tan\beta \cdot \int_{z_{base}}^{z_{gwt}} K(z)dz - a_c^H \cdot \frac{Q}{A} = 0 \qquad 14$$

Once z_{gwt} is determined, riparian solute load can be calculated in the same way as in the lumped RIM approach (Equation 15).

$$l_X = \int_{z_{base}}^{z_{gwt}} l_X(z)dz = \int_{z_{base}}^{z_{gwt}} \frac{\partial Q_R}{\partial z} \cdot c_X(z)dz \qquad 15$$

If the variations of riparian solute concentrations and hydraulic conductivity (respectively transmissivity) with depth can be expressed as exponential functions then an analytical solution can be derived for TRIM. In this case, Equation 12 can be rewritten using a simple exponential relation after introducing another transmissivity parameter $k_0^{'}$ comparable to k_0 in RIM (Equation 16).

$$Q_R = \tan\beta \cdot w_R \cdot k_0^{'} \cdot e^{b \cdot z_{gwt}} \qquad 16$$

The simple structure of Equation 16 makes it possible to solve Equation 14 directly for z_{gwt} (Equation 17).

$$z_{gwt} = b^{-1} \cdot \ln\left(\frac{a_c^H \cdot Q}{\tan\beta \cdot k_0^{'} \cdot A} \right) \qquad 17$$

Substituting z by Q and transforming the integration bounds (Equations 18-21) allows solving Equation 15 analytically (Equation 22). Note that setting the lower integration boundary to minus infinitely deep ($z \rightarrow -\infty$) does not affect the numerical result in practice given sufficiently large absolute values for b and z_{base} (see Paper III).

Transforming the integration bounds:

$$Q_R(z = z_{gwt}) = \frac{w_R \cdot a_c^H}{A} \cdot Q \qquad 18$$

$$Q_R(z \to -\infty) = 0 \qquad 19$$

Substituting z by Q:

$$\frac{\partial Q_R}{\partial z} = b \cdot w_R \cdot \tan\beta \cdot k_0^{'} \cdot e^{b \cdot z}$$
$$= b \cdot Q_R$$
$$\Rightarrow dz = Q_R^{-1} dQ_R = Q^{-1} dQ$$
$$\frac{\partial Q_R}{\partial z} \cdot c_X = \left(b \cdot w_R \cdot \tan\beta \cdot k_0^{'} \cdot e^{b \cdot z}\right) \cdot \left(c_X^0 \cdot e^{f \cdot z}\right)$$
$$= b \cdot w_R \cdot \tan\beta \cdot k_0^{'} \cdot c_X^0 \cdot e^{(b+f)z}$$
$$= b \cdot w_R \cdot \tan\beta \cdot k_0^{'} \cdot c_X^0 \cdot \left(\frac{a_c^H \cdot Q}{\tan\beta \cdot k_0^{'} \cdot A}\right)^{\frac{b+f}{b}} \qquad 20$$

Introducing the η-parameter (Equation 7):

$$\frac{\partial Q_R}{\partial z} \cdot c_X =$$
$$b \cdot c_X^0 \cdot w_R \cdot \left(\frac{a_c^H}{A}\right)^{\eta} \cdot \left(\tan\beta \cdot k_0^{'}\right)^{1-\eta} \cdot Q^{\eta} \qquad 21$$

And calculating the load l_X of a solute exported from the RZ:

$$l_X = \int_{z_{base}}^{z_{gwt}} \frac{\partial Q_R}{\partial z} \cdot c_X(z) dz$$
$$= \int_0^Q b \cdot c_X^0 \cdot w_R \cdot \left(\frac{a_c^H}{A}\right)^{\eta} \cdot \left(\tan\beta \cdot k_0^{'}\right)^{1-\eta} \cdot Q^{\eta} \cdot Q^{-1} dQ$$
$$= b \cdot c_X^0 \cdot w_R \cdot \left(\frac{a_c^H}{A}\right)^{\eta} \cdot \left(\tan\beta \cdot k_0^{'}\right)^{1-\eta} \cdot \int_0^Q Q^{\eta-1} \cdot dQ$$
$$= b \cdot c_X^0 \cdot w_R \cdot \left(\frac{a_c^H}{A}\right)^{\eta} \cdot \left(\tan\beta \cdot k_0^{'}\right)^{1-\eta} \cdot Q^{\eta} / \eta$$
$$= \frac{b \cdot c_X^0 \cdot w_R}{\eta} \cdot \left(\frac{a_c^H}{A}\right)^{\eta} \cdot \frac{\tan\beta \cdot k_0^{'}}{\left(\tan\beta \cdot k_0^{'}\right)^{\eta}} \cdot Q^{\eta}$$
$$= \tan\beta \cdot k_0^{'} \cdot \frac{b \cdot c_X^0 \cdot w_R}{\eta} \cdot \left(\frac{a_c^H}{\tan\beta} \cdot \frac{1}{k_0^{'} A}\right)^{\eta} \cdot Q^{\eta}$$

Thus:

$$l_X = \tan\beta \cdot k_0^{'} \cdot b \cdot c_X^0 \cdot w_R \cdot \frac{e^{\eta \cdot TWI}}{\eta \cdot \left(k_0^{'} A\right)^{\eta}} \cdot Q^{\eta} \qquad 22$$

During periods of high streamflow, groundwater tables can rise above the soil surface and thereby cause runoff on top of the RZs. The maximum flow threshold Q_0 at which surface flow begins varies with different RZs. The value of Q_0 can be determined by setting the groundwater-table position z_{gwt} in Equation 16 equal to zero and solving for Q_0 (Equation 23).

$$Q_0 = k_0^{'} \cdot \frac{\tan\beta}{a_c^H} \cdot A \qquad 23$$

Since surface flows bypassing the soil matrix are not captured by the RIM concept, its application is limited to loads transported by subsurface flows (Equation 24).

$$l_X = \tan\beta \cdot k_0^{'} \cdot b \cdot c_X^0 \cdot w_R \cdot \frac{e^{\eta \cdot TWI}}{\eta \cdot \left(k_0^{'} A\right)^{\eta}} \cdot \min\left\{Q^{\eta}, Q_0^{\eta}\right\} \qquad 24$$

Two alternative assumptions were tested to estimate loads carried by surface flows l_X^{surf}. In the first assumption, surface flow is equated to precipitation containing negligible amounts of TOC, $l_X^{surf} = 0$. In the second assumption surface flow is supposed to consist of return flow containing a mixture of TOC from all soil horizons, $l_X^{surf} = l_X / Q_0 \cdot (Q - Q_0)$.

As can be seen in Equation 24, solute exports from different RZ are scalable with catchment area A, terrain slope $\tan\beta$ and the topographic wetness index TWI, which are all standard terrain attributes. If in-stream sources and sinks are negligible, the total solute load L_x at an arbitrary point in the stream network equals the sum of all upstream riparian loads (Equation 25). Stream-water solute concentrations C_x can be calculated by dividing the total solute loads L_x by streamflow Q (Equation 26).

$$L_X = \sum_{upstream} \left(l_X + l_X^{surf} \right) \quad 25$$

$$C_X = \frac{L_X}{Q} \quad 26$$

The hydrological parameters $k_0^{'}$ and b of the TRIM model were inferred by applying a standard linear regression on observed groundwater-table positions z_{gwt} (predictors) and on a flow variable v_Q to be predicted (Equation 27).

$$v_Q = \ln\left(\frac{a_c^H \cdot Q}{\tan\beta \cdot A} \right) \quad 27$$

The linear relation used for the regression was obtained by introducing v_Q in Equation 26 and rearranging the terms (Equation 28). The value of v_Q was calculated for each ROK site separately depending on local values of $\tan\beta$ and a_c^H.

$$v_Q = b \cdot z_{gwt} + \ln\left(k_0^{'} \right) \quad 28$$

The chemical parameters (i.e. c_0 and f) of the TRIM model were determined separately for the different RZ types identified in Paper IV. The chemical parameters for organic-poor RZs were directly inferred from measured soil-water concentration profiles. For organic-rich RZs a constant TOC concentration was assumed at 75 cm depth whereas the concentration shape parameter f was calibrated to fit observed stream-water TOC concentrations. This is possible because the value of c_{TOC}^0 can be back-calculated by rearranging the terms in Equation 6. It is further worth noting that concentration-profile parameters are also scalable with terrain-indices (Paper IV). Since different scaling relations can affect the analytical solution of the TRIM equations, the presented analytical solution was, for simplicity, derived assuming spatially constant concentration-profile parameters.

5 Results

5.1 Model-based landscape analysis

In Paper I, performances for mapping saturated areas were assessed of three model-based wetness indices (MWI_{mod}, MWI_{steady} and MWI_{hbv}) and two variants of the topographic wetness index (TWI_{MD8} and $TWI_{MD\infty}$). Benchmarks based on three similarity coefficients were comparable with similar studies (Creed et al., 2003; Dahlke et al., 2005; Güntner et al., 2004; Rodhe and Seibert,

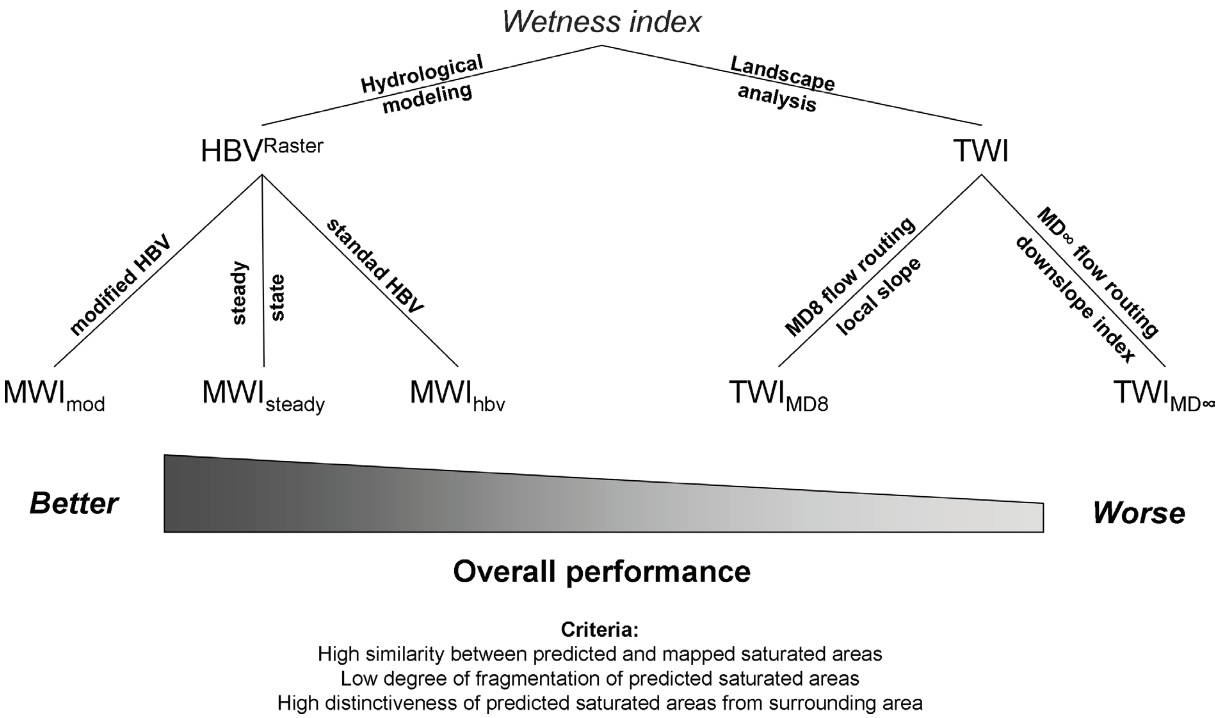

Figure 3: Overview of performances of model-based and terrain-based wetness indices.

1999) and demonstrated that the model-based wetness indices were in better agreement with mapped saturated areas than the TWI variants. Moreover, topologic metrics showed that both MWI and TWI variants predicted a higher degree of fragmentation of saturated areas than actually mapped. MWIs performed better than the TWI because they generated less fragmented and more contiguous patches of saturated areas. Evaluating the statistical distributions of the wetness indices with the Kolmogorov-Smirnov measure indicated that saturated areas simulated by MWIs were more distinct from surrounding areas than the saturated areas produced by the TWI variants. The MWI_{mod} index, which had been computed based on the modified HBV formulation suggested by Seibert et al. (2003), generally achieved the highest scores for all tested benchmarks (Figure 3).

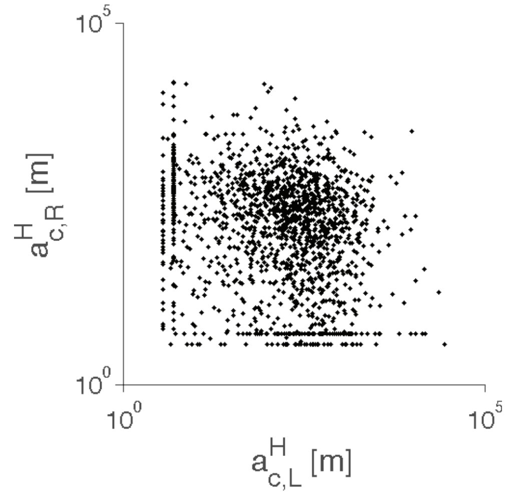

Figure 4: Scatter plot of left specific lateral contributing areas $a^H_{c,L}$ versus their counterpart on the right stream side $a^H_{c,R}$ along the stream network of the 23 km² Tenderfoot Creek catchment.

5.2 A new method to derive terrain indices for riparian zones

The $a^H_{c,L}$ values of specific hillslope contributing areas that were derived (based on the SIDE algorithm) for left sides of the stream network in Tenderfoot (Paper II) did not correlate with $a^H_{c,R}$ values derived for the right sides (Figure 4). Riparian-to-hillslope area ratios R/H derived using the SIDE algorithm resulted in substantially different area-weighted distributions compared to area-weighted distributions obtained when calculating R/H ratios without the SIDE algorithm. The utility of the SIDE method was further assessed for predicting local hydrologic observations from the Stringer catchment, a subcatchment of the Tenderfoot creek catchment (Figure 2). When comparing a^H_c values to HRS water-table connectivity the degree of correlation largely depended on whether total or side-separated a^H_c values were used (Figure 5). A poor relationship (r^2 = 0.42) was observed when comparing total a_c for each transect cross-section against HRS water-table connectivity (Figure 5a). When replacing total a^H_c with side-separated a^H_c values the correlation between specific lateral contributing area and HRS water-table connectivity improved considerably (r^2 = 0.91, Figure 5b).

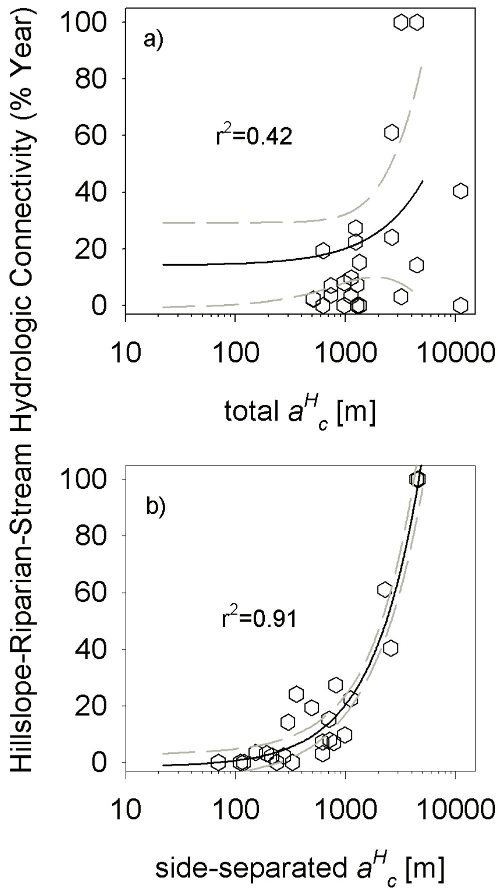

Figure 5: Hillslope a^H_c regressed against the percentage of the water year that a hillslope-riparian-stream water table connection existed for 24 well transects. a) Total a^H_c from both sides of a transect cross-section and b) a^H_c separated into left and right sides of the stream. A connection was defined as occasions when there was stream flow and water levels were recorded in both the riparian and hillslope wells (modified from Jencso et al. (2009)).

5.3 The riparian flow-concentration integration model (RIM)

The backwards calculated RIM-based soil-concentration profiles of all tested solutes (TOC, Ca, Mg and Cl) generally matched the profiles observed at the lysimeter nest located closest (four meters) to the stream (Paper III). The backwards calculated profiles, however, did not agree with soil-concentration profiles measured further away from the stream (12, 22 and 28 meters). Inferred values of the TOC-concentration shape parameter f showed some seasonal pattern when grouped by month and were generally higher for flow situations during summer and autumn compared to spring.

5.4 Heterogeneity and scaling of riparian-zone processes

Groundwater tables, soil-water TOC-concentration profiles, flow-weighted TOC concentrations and specific TOC-export rates from the 13 ROK sites exhibited considerable spatial variations (Paper IV). There were, however, certain patterns in these variations and all sites except two exhibited more TOC-enriched water at superficial soil horizons. At the same time, soil-water TOC-concentration profiles of RZs in the moraine parts tended to have higher TOC concentrations at locations with more superficial groundwater tables. On average, RZs located in the moraine part of Krycklan displayed higher soil-water TOC levels than organic-poor RZs that had evolved on alluvial sediments. Based on median groundwater-table positions $\bar{z}_{gwt}$ (determined for the period May 2008 to September 2009) RZs in the moraine part of the catchment were further subdivided on a relative scale into (1) dry and organic-poor RZs ($\bar{z}_{gwt} \leq$ -45 cm), (2) humid and organic-rich RZs (-45 cm $\leq \bar{z}_{gwt} \leq$ -15 cm) and (3) wet and organic-rich RZs ($\bar{z}_{gwt} \geq$ -15 cm). Wet organic-rich RZs were characterized by high carbon-export rates, strong temporal variations of soil-water TOC, high groundwater tables and relatively low groundwater-table fluctuations while the opposite was true for organic-poor (moraine) RZs. Humid organic-rich RZs formed an intermediate class characterized by pronounced water-table variations and elevated TOC concentrations. The RZs located in the sedimentary part of Krycklan were organic-poor and did not show a clear link between soil-water TOC and groundwater levels. Generally, organic-rich RZs were found to be the main drivers of temporal dynamics in TOC exported to streams, whereas organic-poor RZs showed very little temporal variations of soil-water TOC and released only comparatively small amounts of TOC to streams. The combined effects of variable groundwater tables and TOC concentrations on variable TOC-export rates are illustrated schematically in Figure 6.

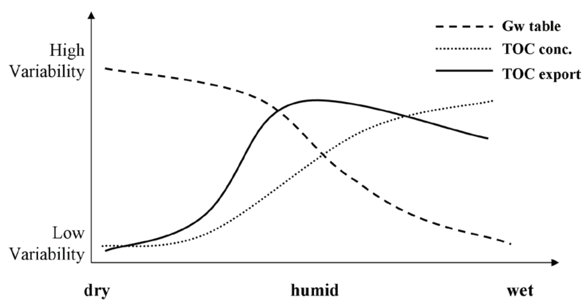

Figure 6: Temporal variability as function of riparian-zone wetness for different quantities (schematic figure).

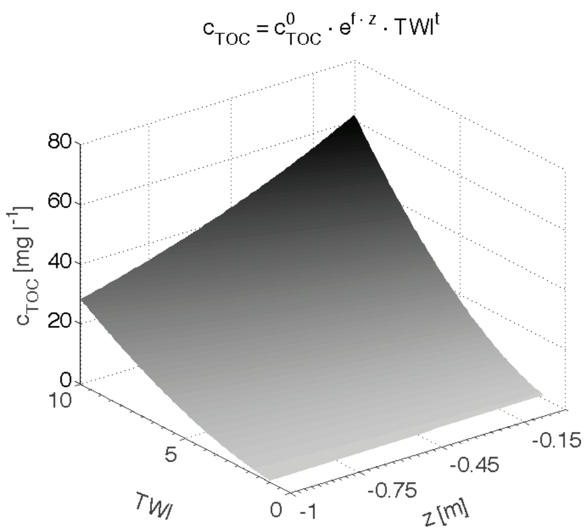

Figure 7: Empirical regression model relating TOC concentrations to depth z and values of the topographic wetness index TWI.

Since soil-water TOC concentration were related to groundwater-table positions, and the groundwater-table positions correlated with the TWI, it was further possible to establish an empirical model based on the TWI to predict spatially variable riparian TOC-concentration profiles for RZs located in the moraine parts of the Krycklan catchment (Figure 7).

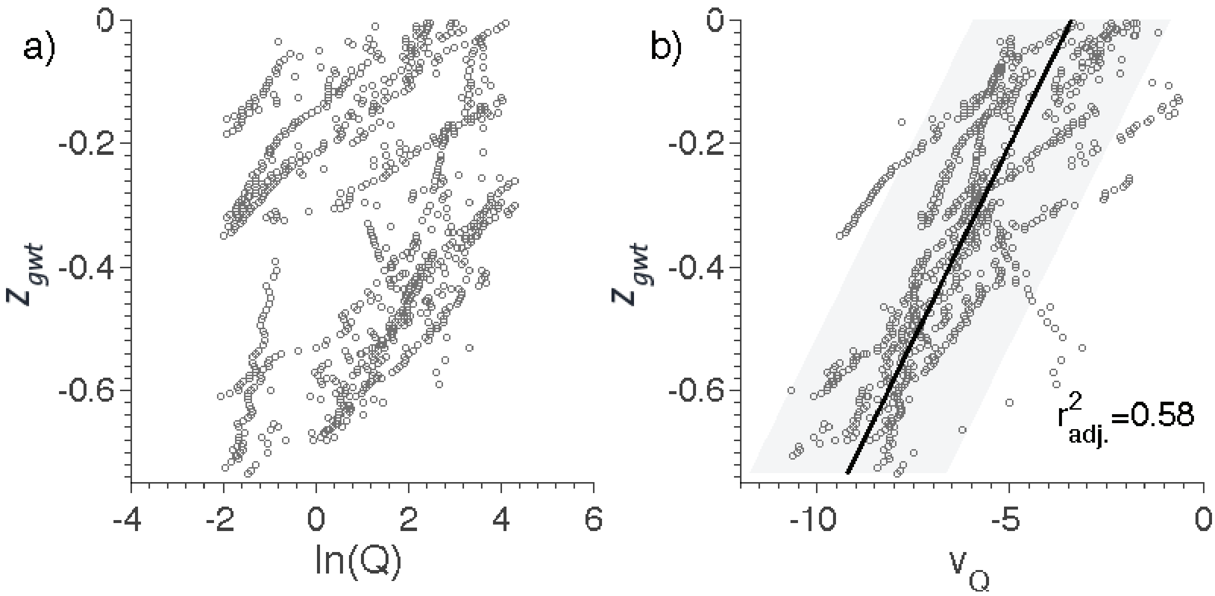

Figure 8: Scatter plots of groundwater levels z_{gwt} related to (a) logarithmically-transformed streamflow $\ln(Q)$ and (b) streamflow transformed based on terrain metrics v_Q. The black line in (b) corresponds to a linear regression (adjusted r^2= 0.58) and the grey-shaded area in (b) corresponds to the 95% prediction intervals of the regression.

5.5 Development and test of the TRIM approach

Hydrological parameters (Table 3) were successfully inferred from groundwater levels (Paper V in progress) and values of the streamflow variable v_Q (Equation 27). The use of terrain indices to calculate the variable v_Q resulted in a much better relation to streamflow compared to using only logarithmically transformed streamflow values (Figure 8). The chemical parameters of organic-poor soils (Table 3) generally matched the soil-water TOC-concentration profiles from which they were derived (not shown here). The two alternative conceptualizations of TOC loads transported by surface flows performed similarly after calibrating a single model parameter (i.e. the concentration shape parameter f of organic-rich RZs) to the observed distributed stream-water TOC concentrations. Plotting predicted against observed stream-water TOC concentrations revealed a slight trend but also a considerable amount of scatter (Figure 9). Calibrated f parameters inferred from stream TOC concentrations were compared against the histogram of f parameters calculated from soil-water TOC concentrations observed at RZs underlain by moraine deposits (Figure 10). This showed that the f parameters of the TRIM assuming no TOC transport by surface flows (Figure 10, b-label) lay at the upper end of the histogram of f parameters. Moreover, the f parameter of the TRIM assuming no TOC transport by surface flows resulted in values of c_0 that appear relatively high compared to measured TOC concentrations (Paper IV). Calibration of the f parameter under the assumption that surface flow consists of groundwater flow carrying a mixture of TOC from different horizons, however,

Table 3: TRIM parameters for different riparian zone (RZ) types characterized by different substrate (quaternary deposits) and values of the topographic wetness index TWI. Chemical parameters at organic-rich RZs differed depending on whether surface flows were assumed to transport TOC (values outside parentheses) or not (values in parentheses).

RZ Type	Substrate, TWI	Hydrological parameters		Chemical parameters	
		k_0'	b	c_0	f
No RZ	· Lakes & Sources	$3.4 \cdot 10^{-2}$	7.9	0	0
Organic-poor	· Sediment	$3.4 \cdot 10^{-2}$	7.9	11.5	0.8
	· Moraine, TWI <5	$3.4 \cdot 10^{-2}$	7.9	12	0.4
Organic-rich	· Moraine, TWI ≥ 5	$3.4 \cdot 10^{-2}$	7.9	61 (157)	1.5 (2.75)

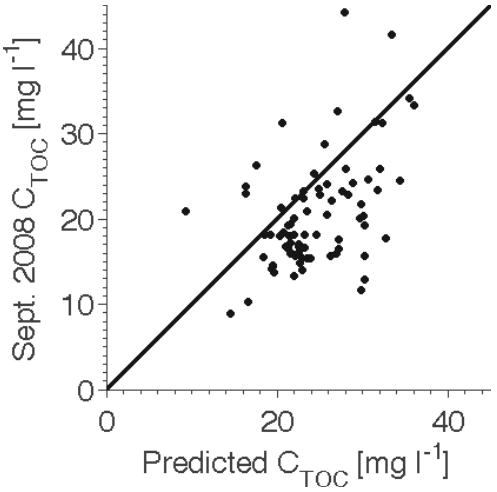

Figure 9: Stream TOC concentrations C_{TOC} [mg·l^{-1}] predicted by TRIM compared against TOC concentrations measured at different stream locations in September 2008 (black dots). The black line is a reference line of slope 1:1.

yielded a value located close to the center of the distribution (Figure 10 a-label). In other words, only the latter f parameter seems realistic whereas the f value obtained when assuming surface flows without TOC in TRIM (Figure 10 b-label) does not seem to be supported by the field measurements at the ROK.

6 Discussion

6.1 Refining current techniques for landscape analysis

Two approaches were presented which considerably refine current methods for hydrological landscape analysis of DEMs. These were a model-based wetness index (Paper I) and a new algorithm for the computation of side-separated contributing areas along stream networks (Paper II).

Model-based wetness indices provided better predictions of saturated areas than could be achieved by standard approaches based on the TWI (Paper I). The main reason for the better performance is that the magnitudes and directions of slopes derived from surface topography (standard approach) differ from those derived from simulated groundwater tables (model-based approach). This difference is particularly important in relatively flat areas of the landscape including areas occupied by RZs or wetlands. Here, hydraulic gradients are usually smaller than surface slopes. Moreover, even the most subtle differences of elevations can strongly influence flow directions if these are derived using standard approaches. This can lead to arbitrarily channelized flow pathways across flat areas and to neighboring locations (grid cells) having very different values of specific contributing area a_c and TWI. TWI-based predictions of saturated areas appear thus often highly fragmentized. Model-based predictions of saturated areas are, on the other hand, much less fragmentized, because they produce relatively smooth groundwater levels in flat areas. This 'smoothing effect' associated with simulated groundwater tables also reduces the influence of micro-topographic features on derived flow directions. Model-based landscape analysis is hence also a potentially attractive tool for hydrological assessments of high-resolution elevation data sets (like LiDAR DEMs), which often contain information that is too fine-grained for standard landscape-analysis methods.

In addition to refining the representation of hydraulic gradients, model-based landscape analysis can potentially also account for temporal variations (e.g. changing flow directions), which is not possible with standard (static) landscape analysis. Although model-based landscape analysis has several advantages over standard landscape analysis, it is computationally more demanding. In addition, results partially depend on the model

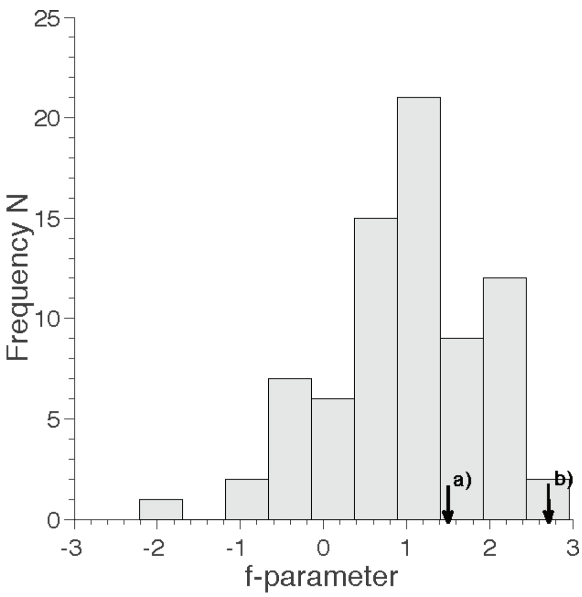

Figure 10: Histogram of concentration shape parameters calculated from 75 observed TOC concentration-depth profiles in the moraine part of Krycklan. Arrows indicate values of f obtained by using stream TOC to calibrate the TRIM a) assuming TOC in surface runoff and b) assuming no TOC in surface runoff.

structure which was illustrated by the fact that the process-oriented structure of the MWI_{mod} produced the best results compared to all tested indices.

The application of the SIDE algorithm to GIS data from Tenderfoot creek showed that values of specific hillslope area contributions to the left side of a stream segment differ considerably from contributions to the right side (Paper II). Total specific hillslope area contributions to a certain stream segment calculated by standard GIS techniques, however, cannot represent these differences, which highlights the importance of the SIDE algorithm. The substantial effect of the SIDE method on calculated *R/H* metrics further suggests that other flow-related terrain indices derived for RZs (e.g. the TWI) are probably also sensitive to the usage of the SIDE method. Applying the SIDE algorithm allowed Jencso et al. (2009) to extrapolate hydrologic-connectivity information observed at local hillslope-riparian-stream transects to the catchment scale. The SIDE method was further used to upscale TOC measured in riparian soil water to the catchment scale (Paper IV) as well as to derive the TRIM model (Paper V).

6.2 Developing simple models to simulate dynamic hydrogeochemical processes in riparian zones

The RIM approach (Paper III) is a parameter-parsimonious mathematical implementation of the qualitative concept outlined by Bishop et al. (2004) for linking solutes in riparian soil waters to solutes in streams. By accounting for the vertical distributions of solutes and flow in riparian soil profiles, the RIM approach is also clearly distinct from other concepts such as end-member-mixing-analysis (Hooper et al., 1990). The analytical solution of the RIM equations further provides a physically plausible explanation for the well-established use of power-law functions to relate concentrations and loads of certain solutes in streams to streamflow (Crawford, 1996; Smith et al., 1996; Cooper and Watts, 2002). The RIM approach was only suitable for back-calculating soil-water solute profiles in RZs but not in hillslopes, which seems to confirm the dominance of RZ processes for controlling stream chemistry for DOC and related constituents in many forested boreal headwaters.

Neither temporal nor spatial variations of hydrogeochemical characteristics of RZs are explicitly considered in the RIM concept. The observed seasonal patterns of the back-calculated f parameter, however, indicate that the TOC-related parameters of the RIM model are not time invariant, which could be considered more explicitly in future work (ongoing PhD thesis M. Winterdahl). The establishment of the ROK further allowed addressing some issues of spatial variability of RZ functioning (Paper IV) and motivated the extension of RIM towards TRIM (Paper V).

The TRIM concept efficiently extends the original RIM to a semi-distributed model. This is achieved by simulating the spatial variability of both chemical parameters and hydrological dynamics in RZs based on landscape-analysis techniques developed in Paper II and IV. If scaling relations can be derived and the RIM assumptions are fulfilled, then the TRIM approach allows estimating temporally variable concentrations and loads of certain solutes (e.g. TOC) at every point in a stream network. Moreover, in cases where the TRIM equations can be analytically solved, the resulting reduction of computational demands makes it possible to run TRIM even at very large scales. The mathematical formulation and the use of detailed hydrogeochemical measurements in RZs also set the TRIM approach clearly apart from distributed models suggested by other studies (e.g. Robson et al., 1992; Rassam et al., 2008). Especially the use of only four basic parameters per RZ class in combination with the wealth of field measurements almost eliminates the problem of 'parameter equifinality' (Beven, 1993) found in many other models.

The demonstration that, in addition to TOC characteristic, hydrological dynamics in RZs can be scaled using topographically derived metrics (Paper V) has been essential for the development of the TRIM model. It is further worth noting that the hydrological component of the TRIM model is similar to the approach taken by Robson et al. (1992) and, thus, relies also on similar assumptions as the TOPMODEL concept (Beven, 1997) introduced in the seminal work by Kirkby and Beven (1979). Like in TOPMODEL, the most critical assumptions in TRIM are the hypothesis of uniform specific-discharge rates and the use of terrain slope as proxy for hydraulic gradients. Although a possible way for relaxing both assumptions has been outlined in Paper I, the location of RZs next to streams can make the suggested calculation of model-based indices rather difficult. Most of the difficulty is probably related to simulating the lower-boundary condition represented by stream-water tables, which is hard to accomplish in practice due to the considerable heterogeneity of natural stream channels. Moreover, the fact that people have influenced the riparian geometry by deepening stream channels may also confound the underlying assumption in TRIM that topography has shaped the RZ over millennia. Some of the unexplained residual scatter seen after calibrating the hydrological (Figure 8) or geochemical (Figure 9) TRIM parameters based on topographic indices might therefore be a consequence of the simple structure of the hydrological component in TRIM and of the assumptions regarding topographic controls on RZs.

The inherent dependance of the TRIM on terrain indices creates another potential source of model-structure uncertainty. Differences in quality and resolution of DEMs as well as differences in the GIS algorithms for computing terrain indices (e.g. slope or contributing area) have been repeatedly shown to lead to different results (Wolock and McCabe, 1995; Sørensen et al., 2006; Seibert and McGlynn, 2007; Sørensen and Seibert, 2007; Kopecký and Čížková, 2010). Quantifying these and other uncertainties would require to incorporate the TRIM into a stochastic framework such as GLUE (Beven, 1993), which was, however, out of the scope of this thesis.

At its current stage of development, the TRIM concept only accounts for spatially variable TOC-related parameters and for topographic controls of hydrologic responses in RZs. Potential spatio-temporal variations of specific-discharge rates or hydrological parameters are neglected. An efficient integration of these factors into the TRIM approach would require developing scaling laws and/or pedotransfer functions (Pachepsky et al., 2006) in as similar way as for the TOC-TWI regression model presented in paper IV. Since the TRIM concept is, apart from its spatial dimension, identical to the original RIM concept, it also suffers from similar limitations. In particular, both concepts use time-invariant chemical parameters. This, however, seems highly problematic considering the strong effect of temporally varying TOC concentrations on carbon exports at organic-rich RZs. The possibility of modeling temporal variation of RZ function in TRIM is hence an important avenue that remains to be explored.

6.3 Relating spatially variable TOC exports from riparian zones to spatial variability of stream TOC

The considerable spatial variability of riparian soil-water TOC-concentration profile shapes at the ROK sites (Paper IV) clearly indicates that a single representative hillslope conceptualization of chemistry and flow pathways in the RZ (Boyer et al., 1996; Bishop et al., 2004; Köhler et al., 2009) is not appropriate for predicting the spatio-temporal variability of stream TOC concentrations when moving from headwater catchments to larger catchments.

The two principal patterns of TOC-concentration profiles (Paper IV) further underpin the development of scaling laws for use in a distributed model. The first identified pattern of increasingly TOC-enriched soil water at more superficial horizons, seen for most concentration profiles, is probably related to increases in soil organic matter storages, degradability of organic matter and biological activity. Deviations from this trend seen at two locations might indicate that some RZs have not been shaped exclusively by millennia of undisturbed riparian processes. Potential disturbances at these RZs could, for instance, be related to past ditching or farming activities. The other observed pattern of increasingly TOC-enriched soil water at sites with more superficial groundwater tables, seen for RZs in moraine parts of Krycklan, is possibly related to decreasing biological activity and increased accumulation of organic carbon (e.g. as peat) under more anoxic conditions. Both patterns are reflected in the mathematical structure of the fitted empirical TWI-TOC regression model (Equation 11). The smooth relation between TWI, depth and TOC concentrations implied in this model is most likely too simple to be true (Figure 7). In particular, it is questionable whether the observed patterns really belong to a continuous spectrum characterized by soils and soil-water TOC concentrations that gradually change with varying depths and wetness conditions. An alternative assumption would be that changes actually occur rather abruptly when environmental factors (such as groundwater-table positions) exceed certain 'thresholds'. Although indications for possible human disturbances (e.g. forestry or farming) that can cause abrupt changes were found in this study, other threshold relations might have been missed due to the limited number of monitored RZs. For instance, the rather crude classification of RZs based on their location in either the sedimentary or moraine parts of the catchment might in reality correspond to an undetected continuous transition of RZ types and vegetation caused by factors such as gradually changing substrate erodibility or ionic composition of groundwater inflows (Sjörs, 1950; Nilsson et al., 1991; Almendinger and Leete, 1998; Giesler et al., 1998).

The combined evaluation of hydrometric data and TOC measurements (Paper IV) generally suggests that organic-rich RZs are considerably involved in regulating temporal dynamics in stream-water TOC, whereas the mosaic of spatially distributed organic-rich and organic-poor RZs participates in generating the fundamental spatial patterns of TOC in stream networks. This further indicates that future (modeling) efforts for estimating spatio-temporal TOC exports from RZs would benefit considerably by focusing on a better understanding of dynamic processes in organic-rich RZs.

The importance of organic-rich RZs for TOC exports also motivated the choice of the f parameter of mapped organic-rich RZs as single calibration parameter when applying the TRIM to predict spatially distributed stream TOC concentrations in Krycklan (Paper V). Although the comparison of TOC concentration predicted by the TRIM and observed stream TOC concentrations indicates a substantial amount of unexplained variance, the obtained 'fit' is still quite remarkable considering that only one parameter was calibrated and that wetlands were not explicitly accounted for and instead treated as RZ end-members. Both conceptualizations of TOC transported by surface flows performed similarly when evaluated based on observed stream TOC concentrations, the calibrated f parameters, however, differed considerably (Figure 10), which indirectly shows that surface flows seem to play an important role in the application of TRIM. Moreover, the value of the f parameter obtained under the assumption that surface flows contain a mixture of TOC from lower soil horizons (which is supported by the field data) results in a quasi-linear relation between streamflow and solute loads (Figure 11). This is because the decrease of TOC concentrations with depth (indicated by the value of f) is small compared to the decrease of transmissivity with depth (quantified by the value of b), which results in a power-law parameter ($\eta = 1 + f/b$) that is close to one.

These preliminary results suggest that the simple perceptual model of the functioning of RZs that was established based on a RZ of a forested headwater catchment is not adequate for describing the more complex landscape of Krycklan. The sensitivity of the TRIM application in regard to the conceptualization of surface-flow TOC transports highlights the need for a more detailed investigation and evaluation of the spatial distribution of simulated groundwater levels to further validate the hydrological model component of TRIM. Under the premise that future investigations will not reject the current hydrological model component, one can speculate that the surface flows predicted by TRIM are most likely to occur for high values of the TWI. Moreover, streamflow was rather low and hardly any surface flow on top of RZs was visible at the time when the stream-water samples were collected for this study. The only places in this study that

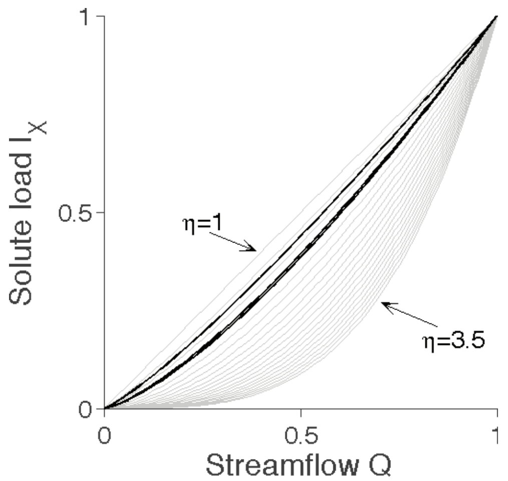

Figure 11: Schematic illustration of the effect of different power-law parameters on the relation between streamflow and solute loads exported from a RZ (assuming no surface flow on the RZ). Thick black lines correspond to curves resulting from the power-law parameters found in this study.

are characterized by high values of TWI and that had surface flows during the time of sampling are saturated areas (i.e. wetlands and lakes). The functioning of wetlands as sources of TOC appears to hold the key for going from a forested headwater catchment to the Krycklan landscape. In contrast to RZs, variable TOC contributions of wetlands might be also related to horizontal changes in connectivity (i.e. expansion and shrinking of regions with superficial flow) and not only to vertical changes in connectivity (i.e. varying groundwater levels in the soil profile). The implicit representation of wetlands in TRIM as 'RZ end-members' is thus probably not appropriate. Finally there are other important processes including in-stream attenuation, hyporheic exchanges or lake effects that might need to be addressed to be able to predict TOC and related aspects of water quality at larger scales.

7 Conclusions

The development of two new methods for landscape analysis and the formulation of an effective modeling framework for riparian zones allowed gaining new insights into linkages between dynamic riparian processes and spatio-temporal variations of stream chemistry. The main conclusions from this thesis are:

- Distributed hydrological models can be used for hydrological landscape analyses to overcome certain inherent limitations of standard terrain-analysis methods.
- The developed SIDE algorithm (Paper II) allows separating flow pathways to left and right stream sides, which is shown to be a fundamental pre-requisite for calculating meaningful flow-related terrain indices for riparian zones.
- The analysis of groundwater levels and total organic carbon (TOC) concentrations measured in the riparian observatory in the Krycklan catchment identified riparian zones as highly heterogeneous landscape elements.
- TOC concentrations in riparian soil waters as well as riparian carbon exports were found to correlate with groundwater levels which in turn correlated with the topographic wetness index. These findings allowed upscaling riparian-zone characteristics to the catchment scale.
- Riparian zones in Krycklan were characterized by different levels of TOC concentrations in their soil waters. Organic-rich riparian zones can be seen as hotspots regulating temporal trends in stream-water TOC, whereas the arrangement of organic-poor and organic-rich riparian zones forms the basic spatial patterns of TOC in stream networks.
- The riparian flow-concentration integration model (RIM) is a parameter-parsimonious approach to mathematically link stream-water chemistry to riparian soil-water chemistry.
- The RIM can be extended to a semi-distributed model (TRIM) by combining topography-based scaling relations with the original RIM equations.
- The analytical solutions of the (T)RIM equations are computationally efficient and provide a physically plausible explanation for commonly used power-law representations relating streamflow to stream solute loads.
- The exponent of the analytical power-law solution is related to the relative decreases of transmissivity and solute concentrations with depth. (In catchments dominated by riparian soils with strong vertical decreases of transmissivity and relatively weak vertical decreases of solute concentrations, stream solute loads are nearly linearly related to streamflow. In the opposite scenario, stream solute loads are proportional to streamflow to the power of an exponent greater than one. Exponents determined in this study varied approximately between 1.0 and 1.6).
- Preliminary results obtained from applying the TRIM to the entire Krycklan catchment seem to indicate that the conceptualization of superficial flows at wetlands is important for simulating the spatio-temporal variability of observed stream TOC concentrations.

8 Future Research

This thesis has led to the development of new landscape-analysis methods and has resulted in new insights regarding the functioning of riparian zones. These provide further research perspectives (some suggestions are listed below) which yet remain to be explored.

8.1 Landscape analysis

- Model-based landscape analysis could be used to derive terrain metrics linked to the temporal variation of a model variable (e.g. median groundwater-table positions in summer months as 'dryness indicators').

- Dynamic simulations of groundwater tables might also allow quantifying temporal variations of hydrological connections between streams and wetlands.

- Steady-state model-based analysis could be further applied to assess flow fields and travel times related to spatially variable values of hydraulic conductivity.

- The SIDE algorithm creates new possibilities to scale and predict riparian-zone characteristics based on landscape analysis. Such characteristics might include attributes related to vegetation, soils, microbiological activity or riparian buffering potential.

- The SIDE algorithm might be combined with model-based landscape analysis to derive model-based indices for riparian zones or to study potential effects of changing flow directions in the near-stream zone.

8.2 Field investigations

- Long-term monitoring of selected organic-rich sites of the riparian observatory in the Krycklan catchment could further elucidate processes involved in the temporal variations of soil-water TOC concentrations.

- The riparian-observatory approach could be used for studying other solutes than TOC such as total nitrogen, water isotopes, anions, cations, trace elements or carbon and sulfur isotopes.

- Data from a riparian observatory could be complemented by distributed soil samples collected at strategically selected riparian zones along the stream network.

- Processes in wetlands and lakes obviously need to be conceptualized in the TRIM modeling framework to make it applicable to larger scales.

Acknowledgements

"*By the way- do you know any nice places to go out in Uppsala?*"

I still can't think what came over me to boldly ask this, but I won't ever forget the confused look on Jan's face that I earned. I had just arrived in Sweden to start my diploma thesis, and it turned out that Jan couldn't give me much advice on this question. On the other hand, had anybody back then told me how my adventure in Sweden was going to evolve the next couple of years, I would have looked even more confused. Ever since, I received loads of good advices and outstanding support (not just limited to the scientific things in a researcher's life) from you, Jan. Thank you very much for giving me such a warm welcome to Sweden, lots of freedom in research and for being available almost 24/7 or whenever critical deadlines were coming up.

As if one enthusiastic supervisor wasn't enough, I had the luxury of having four (!) dynamic co-supervisors forming a true supervisor-dream-team. Kevin, you are an inspiring and energizing motivator who knows how to make research fun. Thanks for persuading me that sometimes even 'seemingly simple' things in research (e.g. streams having two sides) are worth the effort. Hjalmar, many times when I was about to get lost in the research jungle (and in all those bloody details) you were the 'magnetic compass' to help me getting back on track. Thank you also for your assistance to organize all field- and lab work, which otherwise would have been even trickier to accomplish with yet no good teleporting solution around. Brian, thank you for your hospitality and for giving me the opportunity to get in touch with some hydrology outside Europe/Krycklan. Although it later brought me in quite some trouble by my own undiplomatic clumsiness: Hjalmar, I really hope you have meanwhile forgiven me for delivering a non-Krycklan-related presentation at the Krycklan symposium. Such sacrilege shall never happen again! Gia, thank you for making my doctoral studies at Stockholm University a rather smooth process! Credit goes also to all my fellow doctoral students and colleagues who really made my time at INK so much nicer!

Being involved in quite some fieldwork gave me tons of unforgettable and pleasurable moments 'outside' and taught me a lot about being humble about what can be achieved when trying to squeeze a bit of nature into some overly optimistic and simplistic concepts. I am, however, most grateful for truly realizing how limited one is by oneself and how empowering it is to be in a team! Much of this thesis would not have been possible at all if I hadn't received so much help in doing the fieldwork. Thinking about the incredible number of persons who were involved and who helped me, really makes me feel dizzy and I hope I won't forget listing some of you here (I sincerely apologize if I miss someone, although this is quite likely): Peder, I bet most students in Umeå see you as some berserk-strong, kind-hearted University-Hagrid (which, I guess, would make Hjalmar Dumbledor if he had a bit more grey hair). Luckily, you are in a much better shape than J.K. Rowlings character and your drive along with your wicked jokes made every field trip not only effective but also hilarious…which reminds me that there are quite some things that I need to pay you back for, "mouhahaha" (=evil laughter). Thank you for being also a good friend and one more reason why I wish to have a 'teleporter' right at hand. Thank you also Maria J. and Lina for pioneering surveying riparian soils; Thorben and Dieuwke for more surveying, flume-building and tex-mex veggie-food; Eva and Maria B. for completing the riparian-survey work and getting more analyses done; 'Annne', Karin, Rasmus, Mattias, Martin and Maria I. - you were the digging-taskforce doing the fundamental work of installing the riparian observatory in Krycklan (ROK)…and introducing me to new culinary experiences (lost eggs, ultra-fast shrimp-tomato-spaghetti, last-minute korv stroganov, exotic veggi-curry and knäckebröd med långfil och ingefära). Ben, André, JF, Aline, Peng, Marcus N. for lots of fun during discharge and riparian-observatory measurements and brilliant 'music-like' sound effects (including heavy-metal, slow-motion-screaming, "Schnapp", opera singing, Chinese poems and the roaring of a motorbike). Julia and Maggie for their efforts and ingenious planning skills for sampling the ROK. Marcus W. and Ishi for providing me with their experience and help to partially organize stream-snapshot campaigns (you really saved my … back during the 2007 snapshot campaign…and I also didn't have to celebrate my birthday by myself!). Steve for many sharp-witted suggestions and comments, including the one in the preface of this thesis. Pia, Mahsa, Anneli, Lukas, Ida, Rémi, Victor and probably many more for analyzing hundreds of samples… - from doing the first ROK analyses I know how tedious it can be to wash sampling bottles, run the absorbance spectrometer, etc. … and, uhhmm, to clean-up after messy-me who had to run for the train/airplane (which, I guess, was one reason for putting up this infamous wheel-of-blame… I was happy to see that it vanished again!). Kelsey for the Tenderfoot data: You still haven't been in Krycklan but I am sure you'd also like it here and you are more than welcome. Finally, I want to thank the friendly staff at the Annexet and Vindelns fältstation for providing shelter for me and the equipment, an office, cool toy..ähh tools to work in the field and always some (ultra-black) coffee, knäckebröd and cheese when returning half starved from the forests.

There is also whole lot of life to enjoy when not being connected to a computer or scurrying through the woods. What's even more important is that I am blessed with stubborn friends who relentlessly keep reminding me this and who hardly give up trying to drag me out to the 'other good sides of life'. Especially the ones from Lappis FB-6, Lappis AV-8 and from Hammarby Rodd! I sure often gave you all a hard time, but I am really grateful for the good moments we had together (hopefully more are to come). Then there are the old friends who had it even harder reaching me due to distance - but who still managed to 'stay in touch'. It is really deeply reassuring to see the saying 'out-of-sight-out-of-mind' being disproved every time we manage to talk.

I also want to thank my parents Wolfgang and Margarete and my brother Konrad for their never ending love and support. Although you all had and still have a lot to do and deal with in your own lives, you still gave me so much mental and emotional strength over the last years. The same is true for you, Claudia. You always believed in me and you brought so much extra love and joy into my life. I seriously wonder how I could not have missed it before.

My four years of doctoral studies were generously funded by the Swedish Research Council (VR, grant no. 2005-4289) and by Stockholm University.

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