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Stream Restoration in Dynamic Fluvial Systems: Scientific Approaches, Analyses, and Tools

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Cover Image: Time series photographs (1997 and 2009) of a meander bend on Goodwin Creek, Mississippi, before and 2 years after restoration. This successful project is described in the book. Photographs by Andrew Simon and David Derrick.

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PREFACE

Stream restoration is a catchall term for modifications to streams and adjacent riparian zones undertaken to improve geomorphic and/or ecologic function, structure, and integrity of river corridors, and it has become a multibillion dollar industry worldwide. A vigorous debate currently exists in research and professional communities regarding the approaches, applications, and tools most effective in designing, implementing, and assessing stream restoration strategies given a multitude of goals, objectives, stakeholders, and boundary conditions. More importantly, stream restoration as a research-oriented academic discipline is, at present, lagging stream restoration as a rapidly evolving, practitioner-centric endeavor.

Our initial discussions for an edited volume on stream restoration led to a preliminary list of potential contributors assembled by the editors and Colin Thorne. Our approach for soliciting contributions to the volume was simple: we extended invitations to as many leading stream restoration scholars and practitioners as possible (though initially limited to 25). In addition, we made a concerted effort to have a diversified group of contributors. On the basis of the comments from the proposal peer reviewers, the editors altered a few of the contributions in consultation with select authors and solicited a few additional papers to achieve parity in both scope and content as suggested.

The final product of these efforts is a volume that brings together leading experts in both the science and practice of stream restoration, providing a comprehensive, integrative, and interdisciplinary synthesis of process-based approaches, tools, and techniques currently in use, as well as their philosophical foundations. Here nearly 70 researchers from North America, Europe, and Australia contribute papers divided into six broad categories: (1) general approaches, (2) stream hydrology and hydraulics, (3) habitat essentials, (4) sediment transport issues, (5) structural approaches, and (6) model applications. The result is a concise, up-to-date treatise addressing key issues in stream restoration, stressing scientifically defensible approaches and applications from a wide range of perspectives and geographic regions. Most importantly, the volume furthers the ongoing dialogue among researchers and practitioners.

We should like to extend our appreciation to those who made this publication possible. We thank the authors who contributed to the volume, and those individuals who provided constructive and timely reviews of these papers (listed below). We thank Colin Thorne for offering many helpful suggestions in preparing the book proposal. Finally, we gratefully acknowledge the continued support of the University at Buffalo, the U.S. Fish and Wildlife Service, and the Agricultural Research Service of the U.S. Department of Agriculture.

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The Evolving Science of Stream Restoration

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Stream restoration is a general term used for the wide range of actions undertaken to improve the geomorphic and ecologic function, structure, and integrity of river corridors. While the practice of stream restoration is not new to geomorphic, ecologic, or engineering communities, the number of restoration activities and their associated costs has increased dramatically over the last few decades because of government policies intended to protect and restore water quality and aquatic species and their habitats. The goals and objectives, tools and technologies, approaches and applications, and assessment and monitoring standards promoted and employed in stream restoration are rapidly evolving in response to this increased focus and funding. Because technology transfer is an important activity in scientific discourse, this volume provides a comprehensive, integrative, and interdisciplinary synthesis of process-based approaches, tools, and techniques currently used in stream restoration, as well as their philosophical and conceptual foundations. This introductory paper provides a brief summary of the history and evolving science of stream restoration and emerging areas relevant to the stream restoration community.

1. INTRODUCTION

Stream restoration is a catchall term used to describe a wide range of management actions and as such is difficult to define. The definition of stream restoration can vary with the perspective or discipline of the practitioner or with the temporal and spatial scale under consideration. For example, to environmental engineers, stream restoration could mean the return of a degraded ecosystem to a close approximation of its remaining natural potential [*Shields et al.*, 2003], while geomorphologists and hydrologists might define restoration as improving hydrologic, geomorphic, and ecological processes in degraded watershed systems and replacing lost, damaged, or compromised elements of those natural systems [*Wohl et al.*, 2005]. Ecologists further note that restoration of rivers should result in a watershed's improved capacity to provide clean water, consumable fish, wildlife habitat, and healthier coastal waters [*Palmer and Bernhardt*, 2006]. Any of these definitions could include a spectrum of management activities, from replanting riparian trees to full-scale redesign of river channels [*Bernhardt et al.*, 2007]. The wide range of definitions used for stream restoration, and its variation in time, is summarized by *Dufour and Piégay* [2009].

The primary focus of stream restoration has, not surprisingly, been on corridors impaired or degraded by anthropogenic activities. These activities include channelization and

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hydromodification, alteration of land use and land cover, the discharge of pollutants and contaminants into surface and ground waters, and the introduction of new aquatic species [*Wohl et al.*, 2005; *Palmer and Bernhardt*, 2006]. On the basis of recent reports, leading causes of water quality impairment in U.S. rivers include water quality, habitat alterations, impaired biota, nutrients, and sediment [*U.S. Environmental Protection Agency (U.S. EPA)*, 2009]. The majority of low-order U.S. streams, which constitute 90% of all stream miles, have some level of biological impairment, and the most frequent stressors include nutrient loadings, riparian disturbance, and streambed sediment [*U.S. EPA*, 2006]. The most commonly stated goals for river restoration in the United States are to enhance water quality, to manage riparian zones, to improve in-stream habitat, to provide for fish passage, and for bank stabilization [*Bernhardt et al.*, 2005].

The objectives of this introductory paper are to provide a brief history of stream management, to summarize the evolving science of stream restoration, and to identify emerging areas relevant to the stream restoration community. While the emerging areas identified here are not intended to be all inclusive, they do represent the continually changing issues and challenges surrounding stream restoration research and practice and include the following: (1) conflicts within the stream restoration community, (2) the communication of "failure" or lack of success, (3) policy, uncertainty, and practice, (4) landscape trajectories and rise of the social dimension, (5) the future of flow redirection techniques, and (6) the role of models. Finally, the intended goals and thematic focus of this edited volume are presented and contextualized.

2. A BRIEF HISTORY

While "stream restoration" has been vigorously debated from theoretical and philosophical bases over the past few decades, the implementation of stream restoration projects has grown into a multibillion dollar industry. The term "stream restoration" is fairly recent in our river management lexicon, yet the practice of modifying channels for benefit is not.

Early stream management efforts were aimed at bringing water to settlements, reducing the ravages of floods, and irrigating croplands [*Hodge*, 2000, 2002]. The oldest known artificial watercourses were irrigation canals, built in Mesopotamia circa 4000 B.C., in the area of modern day Iraq and Syria. In what is now Jordan and Egypt, the earliest known dams were constructed between 3000 and 2600 B.C. The Indus Valley civilization in Pakistan and north India (circa 2600 B.C.) developed sophisticated irrigation and storage systems, including the reservoirs built at Girnar in 3000 B.C. [*Rodda and Ubertini*, 2004]. In Egypt, canals date back to 2300 B.C. when one was built to bypass the cataract on the Nile near Aswan [*Hadfield*, 1986], while construction of embankments and drainage ditches took place in Italy and Britain 2000 years ago during Roman rule [*Brookes*, 1988; *Billi et al.*, 1997]. Greek engineers were the first to use canal locks, which regulated water flow in the ancient Suez Canal as early as the third century B.C. [*Moore*, 1950; *Froriep*, 1986; *Schörner*, 2000].

By the nineteenth century, large-scale agricultural development associated with European settlement in North America, Australia, and India led to the clearing of large tracts of land and alteration of rainfall-runoff relations. Poor soil conservation practices led to massive erosion of fields and upland areas [*Ireland et al.*, 1939], causing infilling of channels and increasing the magnitude and extent of flooding [*Hidinger and Morgan*, 1912]. To alleviate this, programs were undertaken to dredge and straighten channels particularly in low-gradient valleys [*Moore*, 1917]. Such "channel improvements" were conducted during the first half of the twentieth century in the United States [*Simon*, 1994]; almost 98% of the Denmark's watercourses have been straightened [*Brookes*, 1988].

Given the cycles of intense, deliberative stream management through history, it is not surprising that a new cycle has emerged: "stream restoration." The expansion and popularity of stream restoration today is a societal response to protect water and aquatic habitat. Legislative measures in the mid to late twentieth century, such as the Clean Water Act in the United States and the Water Framework Directive in Europe, continue to be major drivers for the rapid development of stream restoration practice. The concept that streams are the "information superhighway of watersheds," transporting energy and mass from the system as a whole, has taken root in academic institutions and in the psyche of the general public.

3. CONFLICTS WITHIN THE STREAM RESTORATION COMMUNITY

Within the stream restoration community, including practitioners and researchers, there continues to be a wide divergence of what is considered an acceptable stream restoration approach. These differences often are expressed in terms of form-based versus process-based approaches to design and analyses [e.g., *Rosgen*, 2008; *Simon et al.*, 2007, 2008]. Although these differences may be due to the divergent perspectives of the stream restoration practitioner and scholar [*Gillian et al.*, 2005; *Lave*, 2009], this simplistic view is not advocated here. The stream restoration practitioner, no doubt, learns primarily through direct experience and networking with other practitioners, but virtually no written record of these activities exists [*Bernhardt et al.*, 2007]. Moreover, while stream restoration practitioners may produce design reports and engineering drawings, few practitioners provide adequate technology transfer of their methods and procedures. This lack of technology transfer is partially due to the competitive nature of the private sector and a reluctance to share such details, and there is often a lack of critical peer review of these practices. While stream restoration scholars recognize the need to include well-vetted scientific principles into the design and implementation of such activities [*Wohl et al.*, 2005], no such mechanism for the practitioner (scientific, policy, regulatory, etc.) currently exists, and there actually may be a disincentive to do so. Professional journals and panel discussions at technical meetings have, on occasion, aired this tension [e.g., *Rosgen*, 2008; *Simon et al.*, 2007, 2008] but without any significant resolution [*Lave*, 2009].

Recognizing the diversity of stream restoration theory and practice, numerous agencies and scholars have proposed guidance for successful stream restoration in the form of design manuals [*Doll et al.*, 2003; *Natural Resources Conservation Service (NRCS)*, 2007], professional short courses [*Marr*, 2009], journal articles advocating standards and protocols [*Palmer et al.*, 2005; *Woolsey et al.*, 2007], and authored and edited textbooks attempting to compile relevant literature and case studies [*Brookes and Shields*, 1996; *Watson et al.*, 2005; *Brierley and Fryirs*, 2008; *Darby and Sear*, 2008; *Thorp et al.*, 2008]. Most efforts recognize that diverse perspectives shape stream restoration projects, but the emphases for goal setting and evaluation typically reflect the dominant technical disciplines and perspectives within their institution, vocation, or agency. In some cases, government agencies have mandated a specific stream restoration approach, which has intensified conflicts across professional disciplines [*Lave*, 2009; *Lave et al.*, 2010].

Conflicts also can occur across scientific disciplinary boundaries. Hydraulic engineers and geomorphologists often view stream restoration as primarily concerned with producing dynamically stable (not static) channels that do not markedly change their dimensions over periods of years. Ecologists often argue that such practices should focus more explicitly on improving habitat [*Palmer et al.*, 2005] and dispute the use of physical indicators to assess ecological integrity [*Palmer et al.*, 2010]. Differences such as these are shaped by group membership, conflicting values (economic versus ecologic), and different underlying philosophies of science [*Reiners and Lockwood*, 2010]. While many of these conflicts will remain unresolved in the near future, the evolving practice of stream restoration is placing greater emphasis on interdisciplinary, scientifically based approaches well vetted by critical peer review [*Simon et al.*, 2007].

4. THE COMMUNICATION OF "FAILURE" OR LACK OF SUCCESS

Practitioners often refer to "success" or "failure" of individual projects in terms that contradict formally established goals and objectives. Unfortunately, "failure" is often equated with the displacement or loss of a structure, thus promulgating the perception that stream restoration is synonymous with "stability" and is essentially an engineering practice. Anecdotal accounts of "failure" are common components of "in-stream" discussions held during professional development workshops, but very few publications define failure or offer diagnoses or lessons learned from such projects [*Smith and Prestegaard*, 2005; *Shields et al.*, 2007]. Furthermore, the multidisciplinary compositions of project teams, whose members may have very different perceptions of the value of stream restoration, challenge the development of a consistent evaluation protocol. That is, stream restoration evaluations can be highly dependent on the individual reviewer and chosen methodology [*Whitacre et al.*, 2007]. Thus, it is common for stream restoration projects to demonstrate "success" for an incomplete subset of the project objectives [*Palmer et al.*, 2005].

Results from stream restoration projects often are not well communicated, even when project objectives and evaluation criteria have been formalized [*Palmer and Bernhardt*, 2006]. Improved communication between stream restoration practitioners and scholars must occur if advancements in the field are to be made and current design methods more fully understood [*Nagle*, 2007]. In particular, outcomes of both successful and failed stream restoration projects, and the criteria used in these determinations, should be shared more widely in a language understood by all interested parties.

5. POLICY, UNCERTAINTY, AND PRACTICE

Policy clearly has affected the practice of stream restoration. From the U.S. Clean Water Act of 1972 and Endangered Species Act of 1973 [*U.S. EPA*, 2006] to the recent European Union Water Framework Directive and the ongoing debate over stream mitigation credits, legislation provides both the motivation and funding for stream restoration. The Clean Water Act required the U.S. Environmental Protection Agency to regulate water quality and to report on the success or failure of efforts to protect and restore U.S. waterways [*U.S. EPA*, 2006], while the European Union Water Framework Directive requires that streams be restored to "good surface water status."

Current discussion of mitigation credits [*Lave et al.*, 2010] reveals the policy implications of not evaluating projects and their risks clearly. This includes quantifying and accepting, where necessary, the uncertainties within each phase of the stream restoration process [*Wheaton et al.*, 2008; see *Darby and Sear*, 2008]. Moreover, the discussion with policy makers of uncertainty in stream restoration design and practice is not trivial [*Stewardson and Rutherford*, 2008]. The reduction of uncertainty through advancing the science and application of process-based tools and technology will help address many of the issues raised by policy makers.

The social and political dimensions of stream restoration also can be affected by uncertainty. Sites selected for restoration may not be prioritized by their likelihood of success but rather by socioeconomic constraints, perceived ecological condition, geographic location, land ownership, or the community's perspective on project benefits [*Miller and Kochel*, 2010]. Moreover, the social and economic aspects of restoration projects often are not mentioned in the literature or considered in evaluation protocols, even though these aspects may be the impetus behind a stream restoration project [*Eden and Tunstall*, 2006]. At present, there are few established methods for assessing social values in stream restoration, as many rely on questionnaires [*Bernhardt et al.*, 2005] and interviews [*Bernhardt et al.*, 2007; *Lave*, 2009].

6. LANDSCAPE TRAJECTORIES AND RISE OF THE SOCIAL DIMENSION

Because it is an evolving science, the conceptual framework of stream restoration projects, as well as the goals and expectations of such activities, also are changing with time. Stream restoration's formative years as a developing science were focused on water quality issues [*Dufour and Piégay*, 2009]. Over the last few decades, this emphasis shifted to riverine ecosystems adversely affected by anthropogenic activities and the use of reference conditions and then to ecosystem goods and services. As the definition of stream restoration has evolved, so too have the expectations of such projects.

Two important shifts in this evolving science have occurred recently, which will continue to shape future restorations activities. The first is the recognition that fluvial landscapes follow a complex trajectory with time and that naturalness of river corridors has significant value for ecosystems and society [*Dufour and Piégay*, 2009]. This concept, while not new to geomorphologists, does challenge the practitioner to consider stream restoration activities more holistically. That is, localized fixes of rivers at the stream bank or reach scale generally are just symptomatic palliatives, not genuine restoration actions [*Booth*, 2007], and the reliance on concepts such as "reference conditions" should be reduced significantly. Moreover, large financial investments for localized fixes should not be made when stream restoration and ecological targets may be unattainable or unrealized [*Booth*, 2005].

The second important shift in this evolving science is the recognition and promotion of human, societal, or cultural requirements for stream restoration [*Wohl et al.*, 2005; *Kondolf and Yang*, 2008]. While stakeholder participation is recognized universally as an integral component of stream restoration practices, especially in the design, funding, and authorization of such projects, the weight now placed on human requirements offers new complexity to this evolving science and prompts new questions. One may wonder if human or societal valuation of river corridors is wholly concordant with ecosystem services and river function and form. Moreover, such emphasis on human requirements may place even greater emphasis on urban stream projects, presumably at the expense of river corridors in less populated regions.

7. THE FUTURE OF FLOW REDIRECTION TECHNIQUES

The dominant paradigm in stream restoration today is one of creating stability and increasing habitat heterogeneity [*Hey*, 1996; *Palmer et al.*, 2010], and the installation of structures to redirect flow, to protect vulnerable stream banks, and to create such habitat is a popular approach amongst practitioners [*NRCS*, 2007]. While these in-stream structures can produce aquatic habitat such as scour pools [*Kuhnle et al.*, 2002; *Shields et al.*, 2005], the linkages between channel changes induced by these in-stream structures and ecological function are now under new scrutiny. There is growing empirical evidence to suggest that hydraulic structures for flow redirection may not provide sustained or long-lived positive benefits to biota such as macroinvertebrates and fish, in part because habitat heterogeneity alone does not solve the issues of ecologic impairment occurring at larger spatial scales [*Shields et al.*, 2007; *Baldigo et al.*, 2010; *Palmer et al.*, 2010].

While flow redirection techniques clearly provide hydraulic benefits to river corridors, the positive effects on stream ecology and biota must be examined further. The simple creation of habitat heterogeneity by hydraulic structures should no longer be used as conclusive evidence for or demonstration of ecologic restoration.

8. ROLE OF MODELS

Both physical and numerical models have emerged as important tools for transformative research in stream restoration. Physical models include a wide range of experimental apparatuses used to explore various aspects of open-channel flow. Numerical models can span from simple analytic formulations to multidimensional algorithms predicting turbulent flow, mass flux, and biological agents and indices in rivers.

Physical models provide unrivalled opportunities to examine key attributes of river restoration design and their relation to ecologic indices. Such models have examined, for example, the effects of large wood or riparian vegetation on river form and process [*Wallerstein et al.*, 2001; *Bennett et al.*, 2008], the habitat potential of hydraulic structures [*Kuhnle et al.*, 2002], alluvial response to dam removal [*Cantelli et al.*, 2004], and hyporheic flow exchange in heterogeneous sediments [*Salehin et al.*, 2004]. Experimental facilities also can be used to examine biological responses to hydrologic events and channel complexity [*Kemp and Williams*, 2008; *Rice et al.*, 2008; *Merten et al.*, 2010]. Experimental programs such as these ensure that data quality is high and parameters critical for stream restoration designs are included explicitly.

Numerical models, once validated and verified, provide the opportunity to examine the efficacy of stream restoration projects, assessing those already in existence and facilitating the design of planned installations. Such models have examined, for example, stream bank stability [*Simon et al.*, 2000], the effects of stream restoration installations [*Wu et al.*, 2005; *Langendoen*, this volume], turbulent flow around spur dikes [*Kuhnle et al.*, 2008], and fish movement through riverine bypass structures [*Goodwin et al.*, 2006].

The future practice of river restoration will further embrace the use of models for project design and assessment. Moreover, numerical models will become more commonplace in designing stream restoration projects. By default, stakeholders also will expect that models be used to demonstrate that proposed stream restoration projects will be resilient and sustainable and that water quality and ecologic goals will be met. As such, there will be a growing demand for user-friendly, scientifically robust tools and technology to meet these challenges.

9. FOCUS OF THIS EDITED VOLUME

Technology transfer is an important activity in scientific discourse. Because it is a rapidly evolving science, few treatises today concisely summarize scientifically defensible approaches and applications in stream restoration from a wide range of perspectives and geographic regions. The goal of this edited volume is to bring together leading experts in both the science and practice of stream restoration and to provide a comprehensive, integrative, and interdisciplinary synthesis of process-based approaches, tools, and techniques currently in use, as well as their philosophical and conceptual foundations. Here nearly 70 researchers from North America, Europe, and Australia have contributed papers presenting, discussing, and reviewing current and emerging trends critical to the evolving science of stream restoration. These contributions can be divided into six broad categories.

9.1. General Approaches

In this section, conceptual frameworks and systematic strategies for stream restoration are presented and discussed. The strength of this collection of papers is its richness of diversity, as it offers differing perspectives on stream restoration from both practitioners and scholars from a range of geographic regions.

9.2. Stream Hydrology and Hydraulics

Success in stream restoration design depends heavily on a fundamental understanding of hydrology and channel hydraulics. Here critical aspects of these topics, including the geomorphic significance of design discharge and fluid and mass exchange with the hyporheic zone, are presented.

9.3. Habitat Essentials

As many restoration projects address biological indices, this section focuses on critical aspects of stream channel and floodplain habitat, and it reviews approaches to improve these important ecologic attributes.

9.4. Sediment Transport Issues

This section highlights the important relationship between sediment transport and stream restoration, including the role sediment plays in conditioning channel stability, water quality and ecologic indices, and project design.

9.5. Structural Approaches

The use of structures is nearly ubiquitous in stream restoration. This section reviews the efficacy of some commonly used structures in rivers as well as the design criteria for hydraulically stable pool-riffle sequences.

9.6. Model Applications

As noted above, there is growing demand for stream restoration assessment tools, and this section presents a wide range of technology currently available to design river channels, to assess channel stability, and to determine the impacts of restoration projects on channel hydraulics and sediment transport.

10. CONCLUSIONS

Stream restoration is a rapidly evolving science for the wide range of activities enacted to improve the function, form, and water quality and ecologic indices of river corridors. The focus of these activities has been those streams impaired or degraded as a result of anthropogenic activities. Several emerging areas relevant to the stream restoration community include the following.

10.1. Conflicts Within the Stream Restoration Community

There continues to be a wide divergence of what is considered an acceptable design and analysis approach within the stream restoration community. While diverse perspectives shape stream restoration projects, the goals and evaluation of projects typically reflect dominant technical disciplines.

10.2. The Communication of "Failure" or Lack of Success

There is little formal presentation of restoration projects that fail to meet their project's goals, and the valuation of such projects can be highly variable. Both successful and failed stream restoration projects, and the criteria used in these determinations, should be more widely shared in a language understood by all interested parties.

10.3. Policy, Uncertainty, and Practice

Government policy clearly has affected the practice of stream restoration, yet there is much uncertainty in the formulation and implementation of this policy, as well as in the social and political dimensions of these activities.

10.4. Landscape Trajectories and Rise of the Social Dimension

Because fluvial landscapes follow a complex trajectory with time, stream restoration practitioners are challenged to consider the design, implementation, and evaluation of these activities in more holistic rather than local terms. Moreover, the recognition and promotion of human, societal, and cultural requirements further complicates the practice of stream restoration.

10.5. The Future of Flow Redirection Techniques

In-stream hydraulic structures can produce potential aquatic habitat such as scour pools, but empirical evidence now suggests that these structures may not provide sustained positive benefits to biota. The use of flow redirection techniques in ecologic stream restoration deserves further attention.

10.6. Role of Models

The future practice of river restoration will further embrace the use of physical and numerical models for project design and assessment. As such, there will be a growing demand for user-friendly, scientifically robust tools and technology to meet these challenges.

Edited volumes often capture the essence and immediacy of a scientific topic, and the collection of papers assembled here have achieved this goal. More importantly, it was the intent of the editors to participate positively in the discourse of stream restoration using scientifically defensible approaches and to provide important foundations for the continued success and evolution of the practice of restoration.

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Conceptualizing and Communicating Ecological River Restoration

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We present a general conceptual model for communicating aspects of river restoration and management. The model is generic and adaptable to most riverine settings, independent of size. The model has separate categories of natural and social-economic drivers, and management actions are envisioned as modifiers of naturally dynamic systems. The model includes a decision-making structure in which managers, stakeholders, and scientists interact to define management objectives and performance evaluation. The model depicts a stress to the riverine ecosystem as either (1) deviation in the regimes (flow, sediment, temperature, light, biogeochemical, and genetic) by altering the frequency, magnitude, duration, timing, or rate of change of the fluxes or (2) imposition of a hard structural constraint on channel form. Restoration is depicted as naturalization of those regimes or removal of the constraint. The model recognizes the importance of river history in conditioning future responses. Three hierarchical tiers of essential ecosystem characteristics (EECs) illustrate how management actions typically propagate through physical/chemical processes to habitat to biotic responses. Uncertainty and expense in modeling or measuring responses increase in moving from tiers 1 to 3. Social-economic characteristics are shown in a parallel structure that emphasizes the need to quantify trade-offs between ecological and social-economic systems. Performance measures for EECs are also hierarchical, showing that selection of measures depend on participants' willingness to accept uncertainty. The general form is of an adaptive management loop in which the performance measures are compared to reference conditions or success criteria and the information is fed back into the decision-making process.

1. INTRODUCTION

As rivers integrate water, energy, and material fluxes in watersheds, they also integrate human values and interests related to the goods and services they provide. As a result, river restoration can involve many people, institutions, diverse backgrounds, and interests. Interested groups of people (stakeholders) include political entities (countries, tribal groups, states, and municipalities), agencies that regulate commerce or environmental quality, commercial entities with interests in water quantity and quality, nongovernmental

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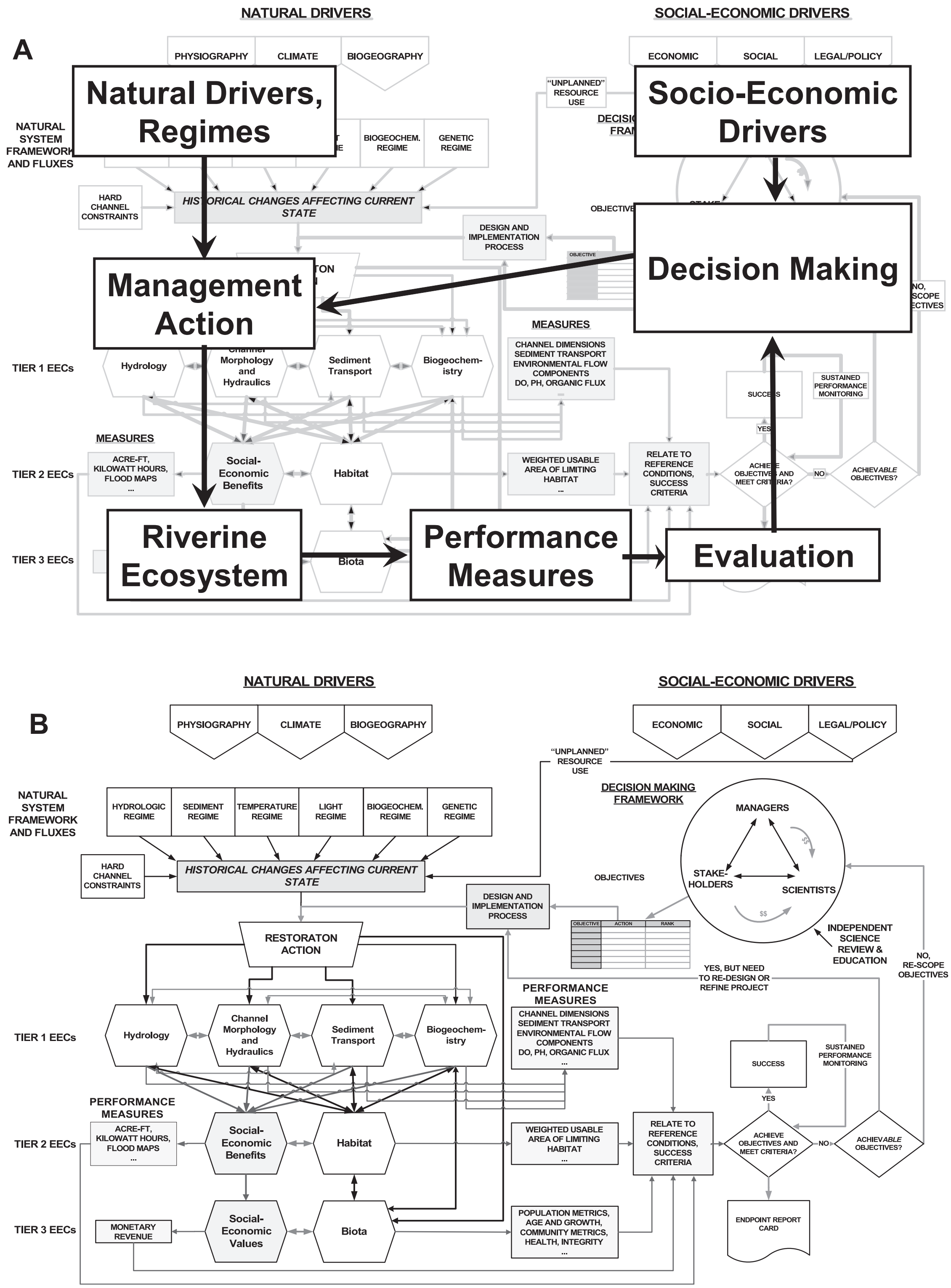

Figure 1. (a) Simplified view of the conceptual model, illustrating the adaptive management loop structure. (b) Detailed view of the conceptual model.

organizations that may represent coalitions of commercial, environmental, or civic interests, and individual members of the public including owners of riparian lands and those who live far from the river but enjoy the river's cultural, recreational, or aesthetic values [*Klubnikin et al.*, 2000].

Interest in river restoration is growing rapidly, and large quantities of money are being committed annually to the practice [*Palmer et al.*, 2007]. Three trends are increasingly apparent. The first and most fundamental trend is the emphasis on restoration and management for ecological objectives. These objectives are institutionalized in the United States by the Endangered Species Act and the Clean Water Act [*Adler*, 2003; *Karr*, 1990] and in the European Union by the Water Framework Directive [*European Parliament, Council*, 2000]. These types of legislation reflect the shared social values of restoring ecological functioning to river systems. Such restoration is challenging, however, because of substantial uncertainties in understanding complex riverine ecosystems [*Christensen et al.*, 1996; *Frissel and Bayles*, 1996; *Palmer et al.*, 2007].

The second trend is increased use of adaptive management: a strategy that specifically addresses uncertainties in management actions [*Lee*, 1993; *Walters*, 1986]. Adaptive management embraces uncertainties in how restoration actions propagate through a river ecosystem by formulating actions as experiments and explicitly including learning in the management process. Adaptive management has become a key strategy for natural resource management in the United States [*Williams et al.*, 2007].

The third trend, increased participation of stakeholders in the river restoration and management process, is linked to the first two trends. Stakeholder involvement is considered a prerequisite to successful implementation of adaptive management because the political realities of many natural resource management decisions require the intentional buy in of stakeholders [*Williams et al.*, 2007]. Social learning that occurs within adaptive management is thought to provide a robust basis for implementing resource-management decisions [*Buijse et al.*, 2002; *Lee*, 1993; *Pahl-Wostl*, 2006; *Pahl-Wostl et al.*, 2007; *Rogers*, 2006]. Stakeholders may also bring specific and important local information to a restoration planning process based on their experiences with a river and its biota [*Jacobson and Primm*, 1997; *McDonald et al.*, 2004; *Robertson and McGee*, 2003].

The sum of these trends has produced, for many restoration projects, a complex planning environment characterized by participation of people and institutions representing disparate technical understanding and diverse values. Although the trends are most apparent in large restoration projects involving many governmental and nongovernmental institutions, diverse values, and large sums of public money (Sacramento-San Joaquin Delta, Chesapeake Bay, Florida Everglades, Colorado River, Platte River, Upper Mississippi River, for example), the social drivers promoting these trends are present in any project when ecological outcomes are uncertain and when there is a perceived accountability for public funds or to off-site stakeholders. The thesis of this chapter is that river restoration planning in a multidisciplinary and stakeholder-driven environment will be aided by conceptual models that encourage effective communication of complex systems and enforce systematic thinking. Conceptual models have been used in this role in other restoration projects, notably the Kissimmee River, Florida [*Trexler*, 1995], the Sacramento-San Joaquin Delta [*Taylor and Short*, 2009], and the Elwha River, Washington [*Woodward et al.*, 2008].

The conceptual model presented here (Figure 1) is intended to provide a framework for understanding river restoration and many of the decisions common to river restoration processes. The salient parts of the model are (1) recognition of multiple drivers of the decision-making process and ecosystem characteristics; (2) implementation of an adaptive decision process incorporating managers, stakeholders, and independent scientists; (3) recognition of the role of historical legacy in shaping present-day river responses to management; (4) a three-tiered hierarchical conceptualization of ecosystem response; (5) an explicit incorporation of social-economic responses in parallel with ecosystem responses; and (6) an adaptive management feedback loop based on response measures, explicit reference conditions, and learning.

The model has evolved from an initial conceptualization used in understanding ecosystem restoration in the Everglades [*Harwell et al.*, 1999]. The Everglades example was used subsequently to craft a hierarchical response model to illustrate river restoration on the Upper Mississippi River [*Lubinski and Barko*, 2003]. While working with adaptive management of river restoration projects on the Lower Missouri River, the first author continued to elaborate the hierarchical model and place it within a broader framework that includes decision making and learning. An intermediate version of the hierarchical response model was used to illustrate concepts in flow-regime restoration on the Lower Missouri River [*Jacobson and Galat*, 2008]. While the model has evolved toward generality, it has inevitably grown in complexity. In the form presented here, it is intended to be generally applicable to river restoration processes where ecological uncertainties are acknowledged and the restoration process incorporates stakeholders with a diversity of backgrounds and values.

Each river restoration project may ultimately develop one or many conceptual models refined to communicate the specific characteristics of its project, its river, and its decision framework. The model presented here is intended to illustrate the general usefulness of conceptual modeling in the river restoration process and to introduce some specific characteristics of conceptual models that may increase their utility.

2. CONCEPTUAL MODELS

A conceptual model is simply an abstract mental image of important parts of a system and how they are related. In an ecosystem context, conceptual models are defined as "graphical representations of interactions among key ecosystems components, processes, and drivers" [*Woodward et al.*, 2008]. A conceptual model is usually displayed graphically for increased understanding.

Conceptual models vary broadly in their structure and complexity [*Gentile et al.*, 2001]. Those for ecosystems can get very complicated and often evolve into complex process-based [*Walters et al.*, 2000] or probabilistic [*Reiman et al.*, 2001; *Stewart-Koster et al.*, 2010] computational models. Conceptual models may also vary depending on perspective. For example, many conceptual models are focused on specific biota and may be structured to support population models [*Wildhaber et al.*, 2007]. The emphasis in such a model is to illustrate the influence of factors that determine probabilities of passing from one life stage to another. In contrast, the Grand Canyon Ecosystem conceptual model is focused on illustrating general ecosystem productivity with less focus on particular species [*Walters et al.*, 2000].

The model presented here is intended to illustrate the broad effects of management or restoration actions. As such, it has a bias toward management actions and how they propagate through a riverine ecosystem. Unlike the models cited above, this model is considerably more generic because it does not specify an endpoint but allows users to define their own biotic or abiotic interests.

Conceptual models are frequently cited as a necessary step in formal adaptive management in which stakeholders and scientists jointly develop a shared understanding of the river system and then apply the model to predictions of system behavior (hypotheses) under management scenarios [*Walters*, 1986]. Eventually, hypotheses are identified that are worthy of implementation as management experiments. While there is value in starting the conceptualizing process with a blank piece of paper so that no ideas are left out of consideration, provision of a general framework serves to increase the efficiency of discussion and to assure that essential structural components of restoration are included. The framework can be generic and flexible for adaptations yet still convey the relational interactions that should be addressed in most restoration projects.

The conceptual model also functions as a teaching tool for participants who may lack technical background or who are uncertain about the adaptive management process. The ecological relations illustrated in the model serve to convey a general understanding of the various factors associated with river restoration and management, including consideration of external factors that are not manageable like historical events and geologic context. The general structure additionally serves to show participants their role in decision making, how management actions are evaluated against reference conditions, and how learning is fed back into the decision-making process.

Conceptual models can also be used to communicate to an external audience for purposes of greater understanding and transparency of process. Graphical documentation of structural components of restoration projects supports the logic of restoration decisions and monitoring designs. Well-constructed conceptual models also support the credibility of a project and help justify the restoration investment.

Another key role of a conceptual model is to focus discussion among scientists who typically represent diverse disciplines in river restoration projects. These disciplines may bring different understandings of what is considered salient, credible, and legitimate information to the restoration process. The conceptual model can aid scientists in the development and negotiation of their roles through the common visualization of relations among their disciplinary perspectives and the placement of their science within the overall restoration management process.

3. MODEL FRAMEWORK

3.1. Simplified Version

Our experience supports the value of conceptual models in communicating restoration goals and strategies in a multidisciplinary and stakeholder-driven environment. The framework is intended to act as a guide for the design and development of river restoration actions across multiple disciplines and varying degrees of participant's knowledge about the ecosystem or decision-making process. In practice, the model framework would likely be introduced to participants incrementally, starting with broad overviews of the restoration and adaptive management process (Figure 1a).

The simplified version of the model (Figure 1a) has a circular form familiar in adaptive management models [*Lee*, 1993; *Walters*, 1986]. The broad components of the model are (1) the natural framework that determines the nature of the river, including flow, sediment, and chemical regimes and geologic constraints; (2) social-economic drivers that influence the ecosystem directly and also influence the decision-making process; (3) a restoration or management action that arises from the decision-making process and acts as a filter on the natural system; (4) the riverine ecosystem affected by both natural processes and constraints as well as the restoration action; (5) performance measures for how well restoration activity functions relevant to the objectives set in the decision-making process; and (6) an evaluation step in which performance measures are evaluated and learning is fed back to decision making.

This rendering of the conceptual model is useful as an introduction to the more complex model and serves to emphasize several key points:

1. The restoration framework is dominated by an adaptive management loop of implementation, response, evaluation, and learning.

2. Restoration actions occur as filters that mediate naturally dynamic processes or regimes. In this sense, reconfiguring a channel or changing reservoir operating rules are examples of actions that change the spatial or temporal distributions of river characteristics but that operate within a natural system with the capability of altering or overwhelming the restoration activity.

3. Social-economic changes may be imposed on the riverine ecosystem outside of the restoration decision-making process and are thereby uncontrolled in the design. An example of uncontrolled social-economic change might be emergence of a biofuel economy that increases land values for crop production, reduces land availability for wetland restoration, and increases competition for in-stream flows.

The overview version of the model helps to communicate concepts incrementally to participants and to emphasize the point that restoration occurs in an open system framework in which results can be altered by uncontrollable natural forces and unplanned social-economic forces.

3.2. Building Blocks of the Detailed Model

A more detailed version of the model is used to develop understanding of typical relations in river restoration (Figure 1b). The detailed version is based on the general idea of illustrating drivers, stressors, and effects on an hierarchically structured ecosystem [*Gentile et al.*, 2001; *Henderson and O'Neil*, 2004] (Figures 1b and 2–4).

3.2.1. Drivers. Drivers are natural and social-economic forces that operate to provide the background context within which restoration occurs. For the purposes of this model, drivers are treated as boundary conditions or factors that are input to the model and not affected by model dynamics. Natural drivers are climatic, physiographic, land cover, and biogeographic factors that control natural fluxes of water, mass,

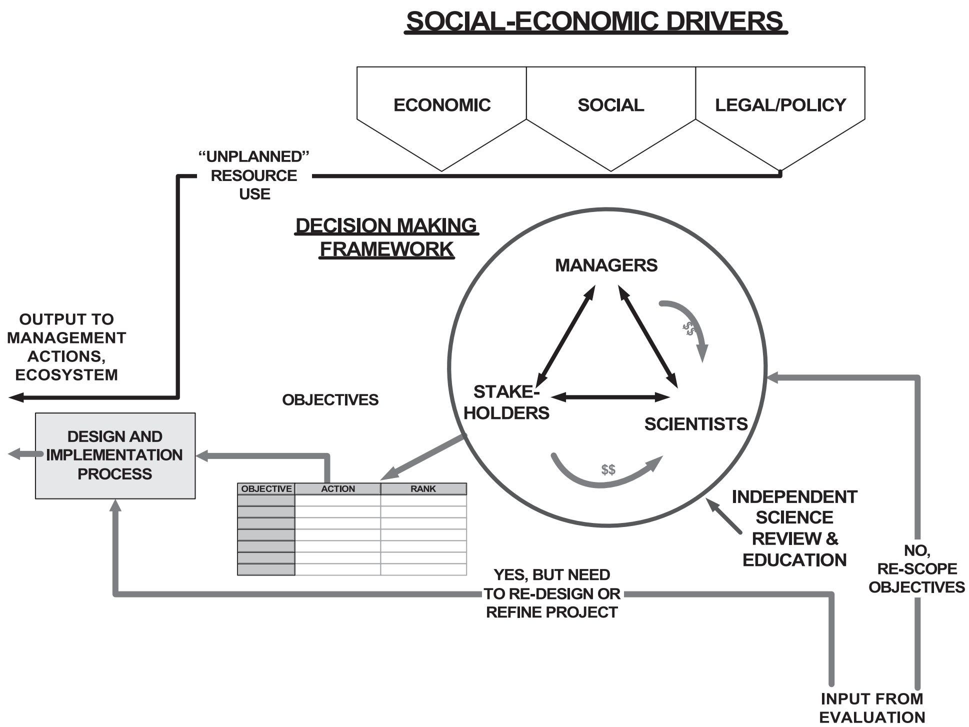

Figure 2. Upper right-hand quadrant of the conceptual model, showing the social-economic drivers and decision-making process.

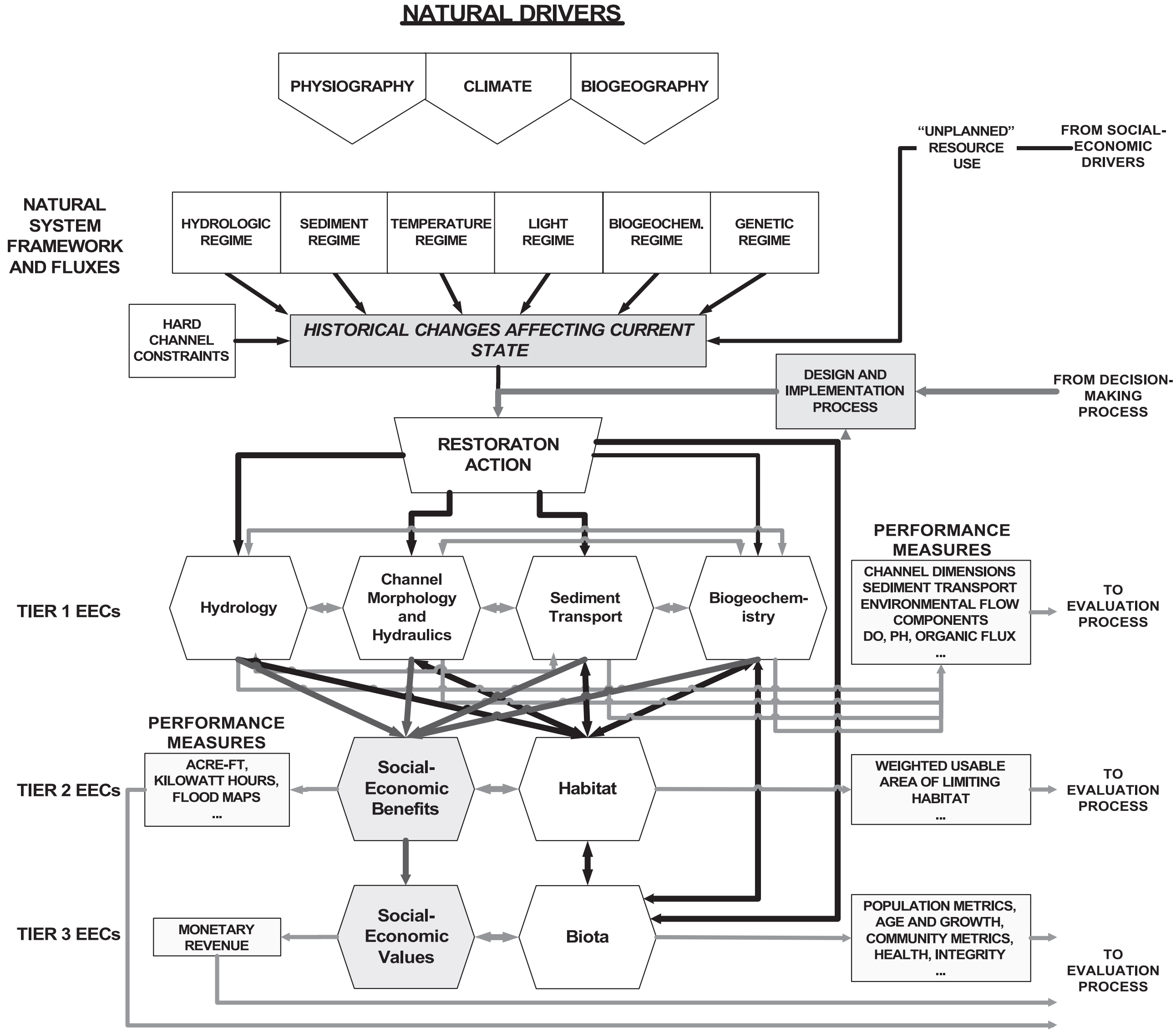

Figure 3. Left side of conceptual model showing natural system framework and regimes, filter of historical changes, and the three-tiered riverine ecosystem consisting of ecologic and social-economic essential ecosystem characteristics (EECs).

energy, and genetic information in a watershed. Social-economic drivers are economic, social, and legal/policy factors that influence human decisions about river restoration, including factors that act to limit restoration actions, for example, costs, laws, or prevailing management philosophies.

Social-economic drivers are depicted separately from the natural drivers and are treated as boundary conditions to the model as they impose constraints on ecosystem performance and the decision-making process (Figure 2). Economic benefits, social learning, and new policies may be generated internal to the system in the decision-making part of the model, but the drivers shown are considered external. The economic driver includes external market-driven forces that would alter monetary valuation of goods and services provided by a river ecosystem. The social driver includes social movements that may change values recognized in goods and services provided by rivers. The legal/policy driver is the framework of laws within which restoration occurs.

The physiography driver (Figure 3) includes geology, soils, and topography of the watershed, factors that exert controls on water, sediment, and geochemical fluxes into the river corridor. In large watersheds, physiography generally can be considered invariant over planning time frames of decades to centuries. However, as smaller watersheds are considered, or

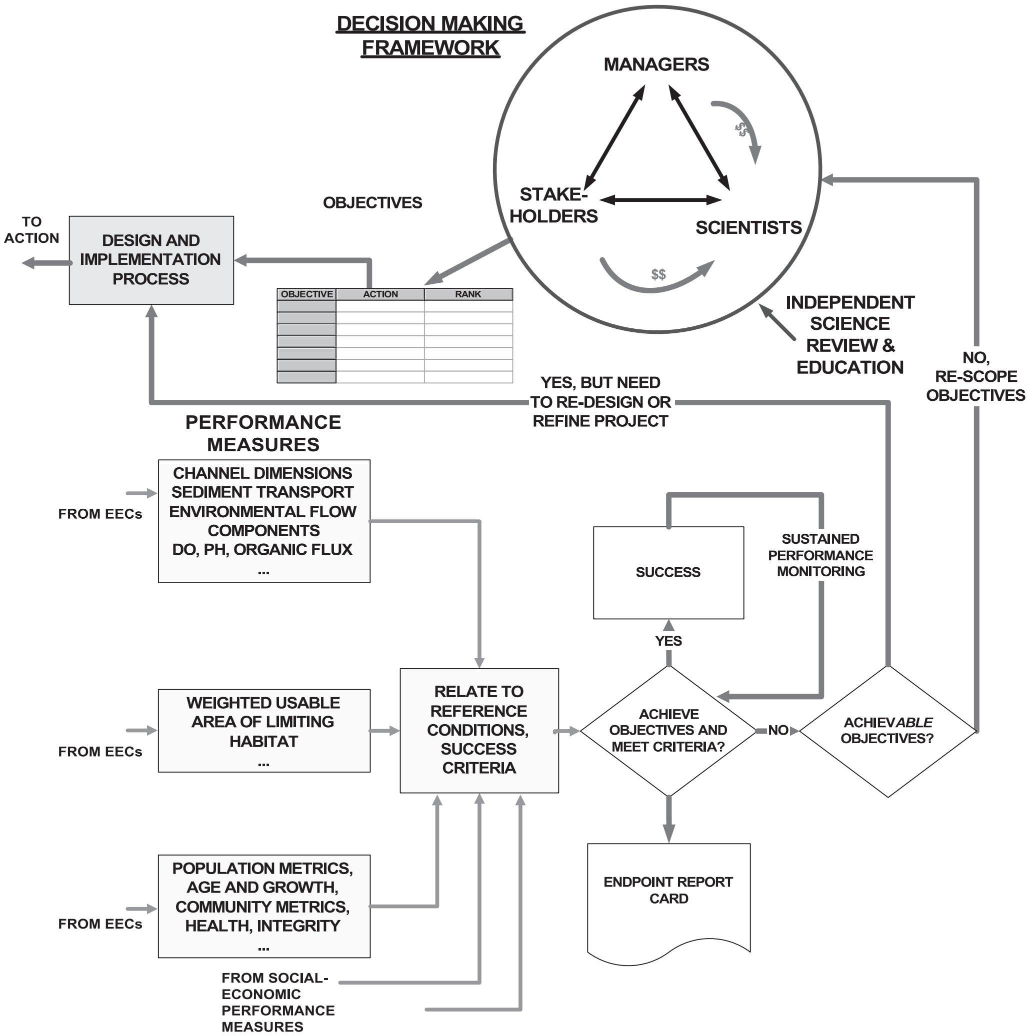

Figure 4. Lower right quadrant of conceptual model showing evaluation of metrics against reference conditions or other success criteria, decisions, and adaptive feedback to redesign or to the decision-making process.

tectonism increases, topography and surficial materials may change considerably over a time frame of years, especially with urbanization. The climate driver is the broadscale climatic context of a watershed that controls fluxes of atmospheric energy and moisture into the watershed. Unlike physiography, climate is more likely to vary dynamically within a planning time frame, for example, due to multidecadal climatic shifts. The biogeography driver describes the pool of organisms available in the watershed, and the natural flux of genetic information due to immigrations, emigrations, mutations, and extinctions. The biogeography driver includes the spatial distribution of organisms within the watershed, which may influence fluxes into the river corridor. For example, natural variation of the type and distribution of vegetation can affect the time series of runoff events.

3.2.2. Decision making. The upper right-hand corner of the model depicts the decision-making process, in this case, symbolized as the interaction among action agency (managers), stakeholders, and scientists (Figure 2). These three roles are generic to decision-making processes in river restoration and management, although the venue for interaction and the engagement in roles certainly varies among projects. In large, multipurpose river systems, restoration decisions typically involve institutions and agencies that are fully engaged in these three roles. In a small project, for example, a reach-scale restoration of a low-order stream, the main participants may be limited to a funding agency (manager) and a landowner (a stakeholder). To the extent that controversy arises, however, other stakeholders (downstream neighbors, regulatory agencies, and watershed councils) may become involved. Conflict often engenders additional independent scientific input to decision making.

The three roles can be conceptually distinguished, although in practice, their boundaries may be somewhat blurred. The action agency is primarily responsible for funding, planning, and carrying out the restoration action. Although labeled as an agency, this role can also be carried out by a partnership, a nongovernmental organization, or a private entity.

The role of stakeholder in river restoration is more fluid. The definition of stakeholder is a person or entity that is affected by or can affect the action [*Williams et al.*, 2007]; in the case of river restoration decisions, the influence can extend far beyond the piece of property or river reach involved. Transmission of restoration effects to downstream areas potentially involves large numbers of the public. For example, river restoration in the Midwestern United States that is effective in diminishing nitrogen loading to local streams may ultimately affect hypoxia conditions in the Gulf of Mexico [*O'Donnell and Galat*, 2007], hence shrimp fishermen in Louisiana may believe that they are stakeholders in small upland projects hundreds of miles away in Illinois. In some cases, there can be indeterminacy between the roles of manager and stakeholder. For example, an agency with a legislated mandate to protect endangered species may consider itself in an action agency role, whereas other participants may consider its role to be as a stakeholder, albeit a particularly powerful one.

The role envisioned for science emphasizes the need for credible and salient science information as the foundation of restoration, and the legitimate participation of scientists in decision making. "Credibility" [*Cash et al.*, 2003] refers to technical adequacy of scientific information, "salience" refers to relevance of the information to decision making, and "legitimacy" refers to perception that the science has been unbiased and respectful of stakeholders' divergent values [*Cash et al.*, 2003]. It has been argued that this role should be limited to individuals, institutions, and commercial interests that agree to participate under terms of policy-neutrality, transparency, peer review, and equal access to information [*Lackey*, 2007]. Working under these terms minimizes opportunities for bias and creates the best opportunity for independence from agency missions and stakeholder influences. This role differs from that of scientists who participate in decision making under the auspices of a management agency or a stakeholder group to advocate specific management objectives. Scientists who forego policy-neutrality, transparency, peer review, and equal access to information are best classified in the role of manager or stakeholder. A similar distinction between "research scientists" and "management scientists" has been proposed in the context of the CALFED program [*Taylor and Short*, 2009].

As rendered in the decision-making portion of the model, decisions are determined through an open, three-way interaction among these roles. Although governance and power sharing can take many different forms, in the ideal situation, individuals or institutions in the role of independent scientists will provide information but will not vote on objectives or policy, thus maintaining policy neutrality. In many cases, scientists involved with the decision-making process, or monitoring and evaluation of the process, are funded wholly or in part by action or stakeholder entities. As an additional check against bias, another layer of outside, independent science review may be justified (Figure 2).

The interaction of managers, stakeholders, and independent scientists determines and prioritizes restoration objectives within the context of the prevailing social-economic drivers and some form of analysis relating presumed restoration benefits to costs. The participants in the decision-making process would work closely with technical staff from the management agencies to design and implement the restoration, design the monitoring and evaluation process, determine reference conditions or other criteria for success, and institutionalize learning and feedback to the decision-making process. The conceptual model indicates how performance measures feed into evaluation and back to the decision-making process where the decision can be made to act, or not, on the generated information.

3.2.3. Stressors, regimes, and filters. In previous application of conceptual models to ecological risk assessments and ecosystem management, stressors have been identified as the physical, chemical, or biological changes that link drivers to ecological effects; the effects are usually considered deleterious [*Gentile et al.*, 2001; *Henderson and O'Neil*, 2004; *Rodier and Norton*, 1992]. For chemical contamination, a stressor is a harmful chemical introduced to the environment; for physical characteristics, a stressor is a harmful extreme of a physical process; a biological stressor might be a native or nonnative population that is out of balance with its resources. In this formulation, a driver (for example, a human action like draining of a wetland) produces a stressor (a change in hydroperiod) which is linked to an ecosystem effect (change in the composition of the plant community).

An alternative formulation emphasizes the natural background dynamics of riverine systems (Figure 3). Continuing with the wetland example, this formulation identifies the drivers as those that determined the wetland plant community in the natural system (climate, physiography, and biogeography). These natural drivers produce regimes, that is, time series of fluxes of water, energy, sediment, and other dissolved and transported materials characterized by their magnitude, duration, timing frequency, and rate of change. A natural wetland community is adjusted to the range of dynamic variation that regulates ecological processes and disturbances. Alteration of one or more of the regimes can be conceptualized as a filtering process that may dampen variability, remove some frequencies, or amplify others. For example, dams tend to decrease magnitude and frequency of floods, which combined with changes to the sediment regime, result in channel adjustment and alteration of habitat availability [*Schmidt and Wilcock*, 2008]. A restoration action then can be understood as a change to the filter, resulting in naturalization in the magnitude, frequency, duration, timing, or rate of change of the regime. The regimes identified in this conceptual model are flow, sediment, temperature, light, biogeochemistry, and genetics (Figure 3). The regimes are symbolized in separate boxes to emphasize that they may vary independently from one another. For example, water temperature, water quality, and sediment regime downstream of a dam may be decoupled from the flow regime, depending on how the system is engineered. In other cases, sediment, temperature, light, and biogeochemical regimes may be strongly controlled by the flow regime, and in these cases, flow regime could be considered the master restoration variable [*Poff et al.*, 1997]. The genetic regime refers to processes and rates of movement of genetic information in a river basin due to immigration, emigration, mutation, and extinction. The genetic regime may also be influenced by the other regimes, for example, in the case where flow-regime alteration is associated with competitive advantages for exotic species [*Olden and Poff*, 2006].

3.2.4. Hard channel constraints. Self-formed alluvial rivers adjust to flow and sediment regimes to attain quasi-equilibrium channel morphology and associated physical habitat characteristics [*Langbein and Leopold*, 1964]. Many natural river channels, however, are also affected by what can be considered hard constraints, that is, geologic or engineering features that are resistant to erosion over decadal or longer time frames (also known as fixed local controls [*Schumm*, 2005]). Some features, like bedrock bluffs abutting a channel, are permanent natural influences on channel morphology. Other features, like debris fans, can be seen as externally imposed geologic features, but because they have some degree of erodibility, their effect on channel morphology is less permanent. For the purposes of this model, all geologic features that impinge directly on the channel and persist over a multiyear time frame are considered hard channel constraints (Figure 3). In addition, because engineering structures are persistent features that affect the channel in a similar way, we add engineering structures to the category of hard channel constraints. Hence, additions or removals of hard channel constraints are considered another type of management action that can transmit or diminish a stress to the river ecosystem. Bank stabilization is probably the most common example of engineered, hard channel constraint.

3.2.5. History, thresholds, and lags. The present state of a river can be strongly conditioned by its history, including alterations in the watershed and at the channel scale. Some alterations may be reversible and may therefore be candidates for restoration. Other alterations, like large dams or urban infrastructure, may not be practically reversible because of their presently perceived social-economic value; these values can change with time, but, as seen with dam removal [*Graf*, 2005], larger infrastructure is generally more permanent. Still other alterations of the watershed and channel will be persistent and resistant to reversal because they have surpassed biologic or geomorphic thresholds, that is, a state of disturbance beyond which the system has difficulty recovering to its predisturbance state. The box "Historical Changes Affecting Current State" communicates the need to understand how the history of river change constrains present-day restoration options (Figure 3).

Examples of threshold historical changes include accelerated erosion of upland soils, an alteration that will have a practically permanent effect on infiltration and runoff rates in some landscapes [*Trimble*, 1974]. A related example is accumulation of eroded soil in floodplain deposits, resulting in floodplain aggradation and disconnection of the floodplain from its channel [*Costa*, 1975; *Jacobson and Coleman*, 1986; *Walter and Merritts*, 2008]. Although the effects of floodplain aggradation can be reversed by extensive excavation, doing so may require efforts that outweigh the benefits [*Bain et al.*, 2008]. Yet another example is choking of stream channels with riparian vegetation when peak flows are diminished because of upstream dam operations [*Williams*, 1978]. Established woody riparian vegetation can impart threshold erosional resistance that requires greater energy to remove in order to restore channel dynamics than would have been necessary before the vegetation became established [*Johnson*, 2000; *Tal et al.*, 2004].

3.2.6. The riverine ecosystem: Essential ecosystem characteristics. The centerpiece of the conceptual model is a hierarchical arrangement of ecosystem components (Figure 3). The individual components are essential ecosystem characteristics (EECs), a unit that was developed originally in conceptual modeling in the Florida Everglades [*Harwell et al.*, 1999]. EECs are groupings of ecosystem characteristics that are ecologically meaningful, that facilitate communication to a broad audience, and that can be linked to management and to measurable endpoints. EECs may be useful in simulation modeling; however, because they are groupings of characteristics, it is more likely that simulation modeling would focus on specific characteristics within an EEC.

The arrangement of EECs is intended to convey several ideas that are important to river restoration. First, the EECs are linked by arrows that signify some level of causal influence. When the conceptual model is applied to a specific restoration action, these arrows can be rendered in weights or colors to show their hypothesized importance. Most of the arrows are double-ended, indicating that causal influence can move in two directions as an interaction among the EECs. In implementation, some arrows may be neglected, singled-ended, or double-ended depending on hypothesized system dynamics.

The EECs are arranged in tiers indicating a general hierarchy of groups of characteristics that are affected by restoration actions. The tiered structure communicates the idea that many, if not most, management actions propagate through a riverine ecosystem, from initial physical/chemical effects (tier 1), to an integrated habitat effect (tier 2), and then to a biotic effect (tier 3), following the right side of tiered arrangements. Hence, the structure of tiers reflects a cascade of measurable effects; placement does not necessarily denote importance or rank. Because of interactions among EECs, the ability to measure and predict the effects of management actions generally decreases from tier 1 to tier 3 (that is, uncertainty increases). Tiers could certainly be subdivided and increased in number, as would be appropriate for more complex models intended to describe complex cascades of cause and effect. For simplicity in this model, we have limited the cascade to three tiers.

Tier 1 EECs are fundamental measures of process and directly affected by restoration actions that involve altering watershed characteristics or dam operations or reconfiguring a channel. These characteristics are usually fairly easy to measure or predict with some confidence, although interactions can create uncertainty. For example, the Channel Morphology and Hydraulics EEC is intimately linked to the Flow Regime and Sediment Regime EECs; Channel Morphology and Hydraulics will adjust dynamically to flow and sediment management in a somewhat predictable way, although the details of adjustment are not always straightforward [*Sear et al.*, 1998].

At tier 2, EECs are split into those associated with conventional social-economic characteristics (left side) and those associated with ecological characteristics (right side). This split is somewhat arbitrary, as all EECs could be considered to measure ecosystem services as broadly defined [*de Groot et al.*, 2002]. Nevertheless, the split is useful in comparing the economic goods and services that arise from direct exploitation of a river to those that arise dominantly from ecological processes.

The tier 2 biotic EEC integrates effects from tier 1 EECs into Habitat, a broad category that encompasses temporal and spatial variation of the physical, chemical, and biological components of the environment that influence reproduction, growth, and survival of biotic communities. This definition assumes biological understanding informs what portion of the environment qualifies as functional habitat. That is, restoration could target habitat for fish spawning, habitat for benthic invertebrate growth, or habitat for shorebird nesting. This EEC is particularly important because of the large number of restoration projects that are intended to restore habitat [*Bernhardt et al.*, 2005]. The relation of the Habitat EEC as an intermediary between Physical and Chemical EECs and Biota EEC draws attention to the need to consider carefully what qualifies as functional habitat.

Habitat has a strong connection to the Biota tier 3 EEC indicating the potential role of habitat in creating bottlenecks for populations of many species. There is also a strong feedback from Biota to Habitat because of the role of some species in altering habitat for other species (and by extension, channel morphology, sediment transport, and biogeochemistry characteristics). Examples include (1) the role of vegetation in providing cover and shading and altering sediment transport; (2) alteration of substrate particle size distributions and sediment transport characteristics by nest-building activities of some fish species; and (3) alteration of hydraulics, sediment transport, nutrient cycling, and organic retention in beaver-dammed ponds. The Biota EEC is disarming in its size as it could conceivably contain a very wide range of biotic characteristics, including life stages of various species and community interactions. In practice, one of the challenges facing river restoration planning is to articulate practical biotic objectives and performance measures (see details in the Performance Measures section).

The left side of the diagram depicts Social-Economic EECs parallel to the Habitat and Biota EECs. The Social-Economic Functions EEC integrates tier 2 EECs into the functions that managed or unmanaged ecosystems provide, including, for example, water supply, flood control benefits, and denitrification of river water. The tier 3 EEC translates those functions into monetary Social-Economic Costs and Benefits. Similar to the right side of the diagram, uncertainty increases from tier 2 to tier 3. Inclusion of the Social-Economic EECs on the left side of the diagram, and connections to the right side of the diagram, emphasizes that humans are part of riverine ecosystems [*Rhoads et al.*, 1999] and that most restoration planning processes eventually need to confront the trade-off between ecological costs/benefits of a restoration project and its social-economic costs/benefits [*Jacobson and Galat*, 2008]. In some cases, benefits may exist for both sides because restoration for ecological goals results in increased ecosystem services that are valued by society, for example, when restoration of connectivity to floodplains provides flood-mitigation benefits [*Nienhuis and Leuven*, 2001]. Unanticipated social-economic values emerging in tier 3, possibly either additional costs or additional benefits, may be important in determining the viability of a restoration action.

Some restoration actions for rivers may bypass tier 1 or 2. For example, some restoration actions may manage biota directly, such as stocking of endangered fishes or eradication of exotic fishes. Direct management of riparian vegetation skips tier 1 and goes directly to tier 2, when vegetation is valued for its function in creating habitat.

The advantage of depicting the ecosystem using the three tiers and two columns is in the ability to communicate two generic issues in river restoration. The first is increasing uncertainty in ability to measure and predict effects moving from tier 1 to tier 3. At the same time, relevance to restoration objectives typically increases from tier 1 to tier 3. That is, the motivation to restore rivers ultimately arises from societal interest in increasing biotic or social-economic values, not from the inherent value of characteristics measurable at tier 1. The structure allows stakeholders to consider strategies to diversify investments in modeling and monitoring among tiers. For example, although the main objective for a restoration project might be to recover the population of an at-risk species, monitoring of population measures at tier 3 alone would fail to compile cause/effect understanding from tiers 1 and 2 that would help document how and why a project succeeded or failed.

Second, the direct comparison between the Social-Economic EECs and the Habitat and Biota EECs focuses attention on the challenges of evaluating trade-offs between ecological restoration objectives and other river-management benefits. Ecological characteristics on the right side are more difficult to quantify than those on the social-economic side, thereby typically leading to information disparities in comparing costs and benefit. Most social-economic benefits of river management (hydropower, flood control, and water supply, for example) can be evaluated very precisely in monetary terms, whereas it is difficult to assign economic values to typical ecological benefits of restoration (species richness, populations, and trophic structure, for example). The conceptual model does nothing to reconcile this problem but it does serve to make it explicit.

3.2.7. Performance measures. Performance measures are depicted as separate entities from the associated EECs to emphasize that the measures are an abstraction of the complexity that characterizes an EEC (Figure 3). This also serves to emphasize that a wide range of measures, with a range of cost and information content, are available for monitoring restoration performance. Generic, illustrative measures are indicated in Figures 3 and 4; the ellipsis (. . .) indicates that many more could be defined for application to specific projects.

The choice of measures has a substantial bearing on costs of project monitoring and therefore on the extent and quality of monitoring accomplished. In a recent nationwide survey, it was found that only 10% of river restoration projects had any monitoring and assessment [*Bernhardt et al.*, 2005]. This low percentage may reflect, in part, difficulties in determining cost-effective performance measures. The decision of how to invest scarce monitoring funds can be informed by consideration of the conceptual model and a focus on stakeholders' comfort with uncertainty or risk. For example, changes in depth and velocity distributions associated with a channel reconfiguration can be measured and modeled with precision and accuracy at relatively low cost. If stakeholders in the restoration are willing to risk the inference that a change in depth and velocity will improve habitat conditions, and therefore increase biodiversity of the river reach, the monitoring budget could be focused on simple hydraulic variables. If, however, the consensus does not support accepting that inference, monitoring may have to invest in more costly, integrative measures of habitat and the biotic community. A typical restoration evaluation may employ a variety of measures in tiers 1, 2, and 3 that, in combination, provide a level of certainty or "weight of evidence" acceptable to all participants.

3.2.8. Reference conditions and evaluation. Performance evaluation involves comparing measures to criteria developed for a successful restoration, usually framed in terms of performance that conforms to or exceeds a reference condition. Reference conditions can range from a fully natural, least disturbed condition, defined by well-documented historical information, to a reference defined by stakeholders as the best attainable condition, which may have little resemblance to a natural reference [*Rhoads et al.*, 1999; *Stoddard et al.*, 2006]. Reference conditions or other criteria for success need to be definable in the same measures associated with the EECs. Reference conditions may also be defined for social-economic values. Such reference conditions would indicate willingness to pay for restoration or thresholds of cost beyond which the project would not be considered viable.

In a formalized adaptive management process, performance results are fed back into the design or decision-making process. At least two decisions can be defined in the feedback (Figure 4). The first is based on whether performance indicates that the restoration is a success, with success being defined as achieving performance objectives relative to reference conditions. Objectives are typically formulated as a range of quantitative scores or a detailed qualitative condition. If the objectives are achieved, the project could be declared completed, or more prudently, monitoring could enter a low-intensity phase to assure that results do not drift. If performance does not indicate success, two additional possibilities exist. In the first case, performance measures may indicate that the project has not achieved success, but measures do indicate that success is possible if the design is altered or if more time is allowed for the design to equilibrate. A separate criterion would be defined for conditions requiring design alteration, for example, after n years of implementation, only x% of the reference condition has been achieved. If this criterion is met, learning developed from the performance monitoring would be fed back into the design process to determine design changes. The second possibility, however, is that the assessment indicates that the objectives are not practical or not achievable, either in terms of restoration benefits or in social-economic costs. This result would indicate that the objectives need to be reconsidered by the decision-making process.

Information gleaned from the performance measures, cause/effect understanding, evaluations relative to reference conditions, and understanding of whether objectives are appropriate, all contribute to learning, a key element in applications of adaptive management [*Lee*, 1993; *Walters*, 1986]. Learning includes assimilation of information by managers, stakeholders, and scientists, formalization of understanding in published literature, and dissemination to the public. Learning may or may not lead to action (changed designs, re-scoped objectives, or altered policies) depending on how scientific information is reconciled with social, economic, and legal constraints [*Pahl-Wostl et al.*, 2007].

4. EXAMPLES AND DISCUSSION

Our conceptual model is meant to be used in an interactive restoration planning process where it contributes to understanding among managers, stakeholders, and scientists. The following two examples illustrate how the conceptual model has been used to facilitate planning and communication for two different types of restoration projects on the Lower Missouri River, Midwestern United States.

4.1. Pulsed-Flow Modifications

The objective of instituting pulsed spring flows (spring rise) on the Missouri River was set by the Missouri River Biological Opinion [*U.S. Fish and Wildlife Service*, 2000, 2003]. This conceptual model was applied to some aspects of the collaborative design of the flow modification [*Jacobson and Galat*, 2008] and to aspects of the monitoring program [*Korschgen*, 2007]. The main objective of the flow modification was to support reproduction and survival of the endangered pallid sturgeon (*Scaphirhynchus albus*). The dominant hypothesis was that a spring flow pulse would cue spawning as a direct influence on fish reproductive physiology and behavior. Although the detailed mechanisms were unknown, it was hypothesized that a flow pulse would result in spawning when water temperatures were conducive and possibly related in unknown ways to turbidity or water chemistry [*Jacobson and Galat*, 2008]. The dominant hypothesis was accompanied by secondary hypotheses that a flow pulse might produce an episode of spawning habitat availability or serve to flush spawning substrate of fine sediment.

A simple application of the conceptual model involves exploration of how the flow modification might propagate through the ecosystem and the best ways to measure performance. The dominant hypothesis (Figure 5) is shown with a heavy black arrow connecting the management action (pulsed flows from Gavins Point Dam, South Dakota), to the Hydrology EEC and then to the Biota EEC, where it was hypothesized to act as an environmental cue for spawning. In drawing hypothetical connections, the lines can be annotated with the presumed processes, as shown. The two secondary hypotheses are depicted as slightly narrower black arrows addressing habitat availability and spawning substrate conditioning. The first of these connects the Hydrology and Channel Morphology EECs to the Habitat EEC to depict a change in habitat availability over time and space (that is, the riverine hydroscape). The second connects the Hydrology, Channel Morphology, and Sediment EECs to the Habitat EEC to depict the interaction among these factors in determining sediment transport conditions.

The diagram also enforces consideration of other potential links and interactions. The interaction between flow regime, channel morphology, and sediment regime, for example, indicates a potential for adjustment of reach-scale hydraulics during a flow pulse that would be considerably more complex than simply evaluating habitat availability or sediment transport in isolation. Moreover, although substantive biogeochemical interactions were considered unlikely in initial planning, the diagram enforces consideration of the potential of the pulsed flows to alter water-quality characteristics. The collective sense of participants about the relative strength of these relationships can be shown by width or color of the lines, and this prioritization can help drive allocation of effort in monitoring and evaluation.

The tier 1 measures for the spawning cue hypothesis relate to the Hydrology EEC. These measures might be peak discharge, duration, timing, and rate of change of the individual flow pulses. Taken near the dam, these measures simply

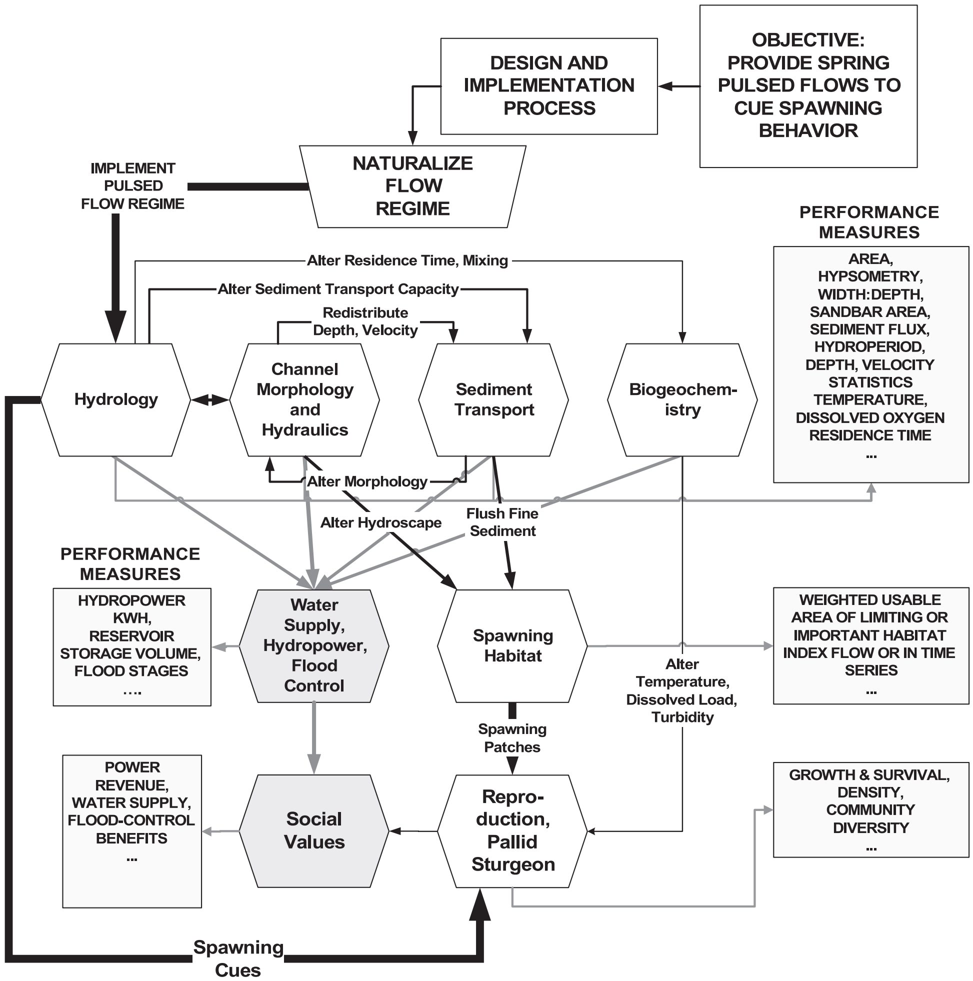

Figure 5. Conceptual model applied to pulsed flow modifications on the Lower Missouri River. Pulsed flows (or spring rises) were designed to restore some characteristics of the natural flow regime to promote reproduction of the endangered pallid sturgeon.

address compliance: How well did dam operations conform to the intended restoration action? Downstream, the hydrologic measures address how effectively a pulse is transmitted through the intended length of the river. Flow regime measures are easily compared to the historical flow regime reference condition to evaluate success at tier 1. The spawning cue hypothesis skips tier 2 and goes directly to tier 3. Here decisions need to address what biotic measures are most useful in guiding restoration. The ultimate objective of the flow modification was to increase reproduction and survival of a long-lived and very rare fish, a difficult and costly assessment. A key decision in this case was how to allocate assessment resources between direct measurement of population responses (requiring multiple replications of the pulses over years to decades and a long-term monitoring program) and short-term measures of effects that can be documented with fish movement, egg deposition, or larval drift [*DeLonay et al.*, 2009].

The secondary hypotheses can be evaluated at tier 1 by assessing sediment transport data (do the flows significantly increase sediment fluxes?), hydraulic data (do flows increase diversity of depth and velocity?), and water-quality data (do the flows significantly alter key water-quality parameters?). At tier 2, these hypotheses require investment in an integrated understanding of how flow, channel morphology, and sediment transport combine to affect availability and quality of spawning habitat. Measures in this EEC presume a robust, biologically based understanding of what defines preferred spawning habitat. To the extent that this is not known, investment in monitoring at this level would be risky. Inference that ecosystem functions will be restored if the proper physical habitat is provided has been described as the "field of dreams" myth of restoration ecology [*Hilderbrand et al.*, 2005]. Although it may be labeled a myth, it is not necessarily an incorrect inference. One function of this conceptual model is to communicate the strength of that inference to participants.

Allocation of resources for performance monitoring is another critical question for many restoration projects. Possibly, all participants would be comfortable with the inference that attainment of flow pulses that match some proportion of the magnitude and duration of the pulses in the natural flow regime, at the correct time of year, would be sufficient to meet the biological objective. Determining success under this assumption would be relatively inexpensive. Some participants, however, may not be comfortable with that inference and, instead, support investment in more directly relevant measures in tiers 2 and 3. Choice of performance measures may also be influenced by the value of information to the adaptive management process [*Lee*, 1993]. For example, assessments that are limited to tier 3 address the fundamental need to document performance in terms of biotic responses, but addition of measurements in tiers 1 and 2 may provide better understanding of cause and effect, and thereby improve future designs and management.

The left-hand side of Figure 5 enforces consideration of the potential social-economic costs and benefits associated with the flow-regime restoration. In this case, prime concerns were drafting of water from the reservoirs, passing of water through hydroelectric generation facilities at a time of year when hydropower revenue rates are low, and increased floods downstream, factors that could be measured at tier 2 with storage volumes, generation data, and water levels [*Jacobson and Galat*, 2008]. Tier 2 effects propagate directly to social-economic benefits in tier 3, measured (usually very precisely) by water supply costs, power revenue, and flood-control benefits. Recognition of the social-economic side also helps frame the value of the information investment on the biota side. Most restoration decisions implicitly or explicitly address trade-offs in costs and benefits. The parallel structure of the conceptual model serves to help participants acknowledge all costs and benefits and envision the information needed to support their decisions.

Pulsed-flow modifications are an example of a restoration action that takes place within natural background hydrologic variation. The structure of the model (Figure 1) emphasizes the idea that historical flow modification by dams and restoration flow modifications by naturalizing release schedules act as filters on the natural flow regime, indicating that extreme natural events have the potential to overwhelm the restoration action. In the actual implementation of flow normalization illustrated in this model, the size of the negotiated flow pulses were modest, amounting to about the 10% of the magnitude of natural flow pulses [*Jacobson and Galat*, 2008]; nevertheless, flooding from tributaries 2 of the 4 years of implementation (2006–2009) produced uncontrolled flow pulses as much as two times the magnitude of the planned pulses [*DeLonay et al.*, 2009]. The structure of the conceptual model encourages stakeholders to acknowledge and prepare for the effects of such natural events.

4.2. Shallow Water Habitat Construction

Another restoration objective on the Lower Missouri River has been to restore approximately 20% of the shallow water habitat (SWH) that existed before channelization of the river [*U.S. Fish and Wildlife Service*, 2000, 2003]. SWH was defined as 0–1.5 m depth and 0–0.6 m s^{-1} current velocity, a class that was historically much more abundant than it is today [*Jacobson and Galat*, 2006] and which is thought to be important for rearing of larval and juvenile pallid sturgeon [*U.S. Army Corps of Engineers*, 2004; *U.S. Fish and Wildlife Service*, 2003]. The conceptual model was not used in planning objectives or design of SWH restoration, but has been used in subsequent planning for monitoring and assessment (Figure 6).

SWH is constructed by removing sediment from the floodplain to widen the channel or by excavating side-channel chutes through the floodplain. This is shown in the conceptual model as a direct change to the Channel Morphology EEC. The dominant hypothesis holds that alteration of the channel morphology will create habitat (integrated effects of flow regime and Channel Morphology, denoted as hydroscape) for larval and juvenile rearing and feeding, which will then propagate to survival, recruitment, and reproduction of the endangered fish. Consideration of all the EECs and their relations enforces recognition of other connections that were not emphasized in the original objective. For example, creation of accommodation space in the floodplain will necessarily change local hydraulics and sediment transport conditions, an adjustment that can be seen as a feedback between Channel Morphology and Sediment EECs. Moreover, SWH can be expected to change water temperatures and residence times, potentially creating new biogeochemical conditions in the restored areas and perhaps producing acute effects directly on fish. A key unknown in this process is the response of vegetation (Biota EEC) in the new habitat, which can produce a feedback to hydraulics, channel morphology, sediment transport, and biogeochemistry. While the results of these complex feedback may not be predictable with confidence, the identification of sources of uncertainties is an important contribution of the conceptual model to the planning process [*Sear et al.*, 2008].

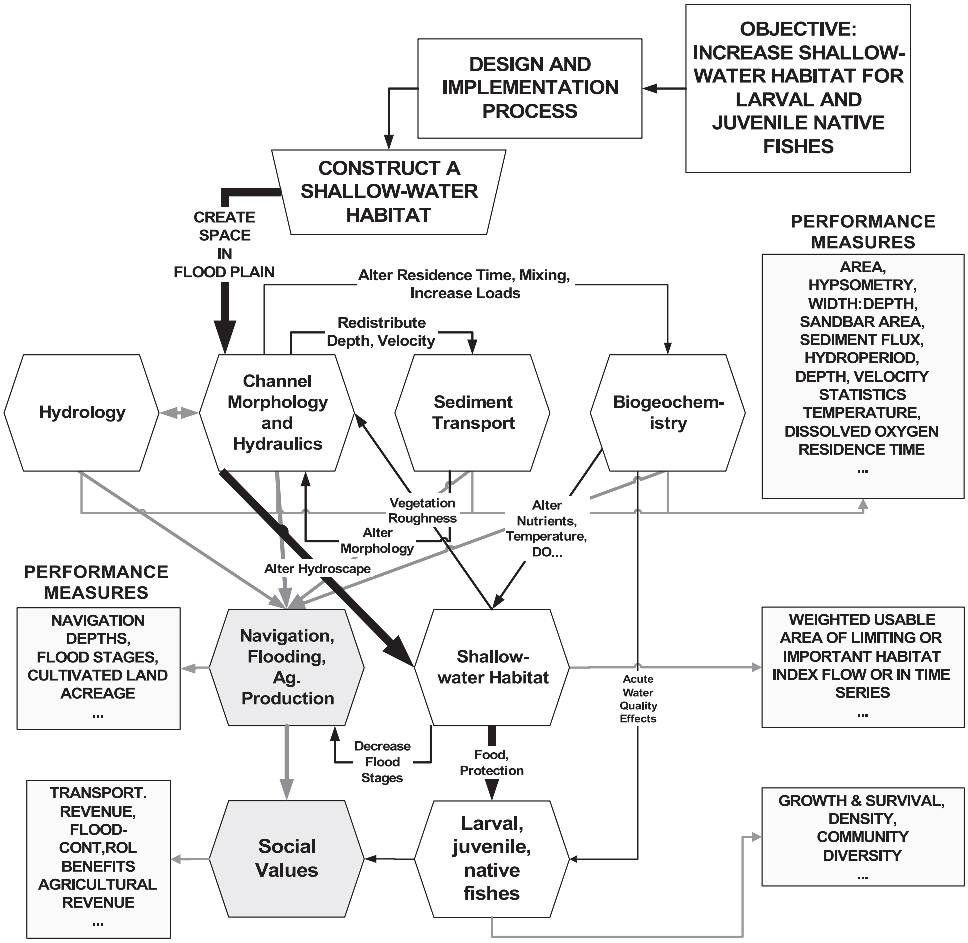

Figure 6. Conceptual model applied to shallow water habitat restoration on the Lower Missouri River. Shallow water habitat construction is intended to restore rearing habitat for larval and juvenile pallid sturgeon.

Similar to the preceding example, performance measures vary markedly in complexity from tier 1 to tier 3. Area of excavation is a simple measure of performance of the restoration action, but does not provide much understanding of functional habitat quality or quantity. Habitat quantity, quality, and temporal distribution can be assessed in the Habitat EEC using standard habitat assessment methods, but recognition of the potentially complex interactions and adjustments among flow regime, channel morphology, and sediment regime indicates that habitat development could be a complex and prolonged process; rigorous performance evaluation may need to wait for an uncertain time interval for the channel morphology to equilibrate to the new hydraulic conditions. Measures in the Biota EEC are more directly relevant to project objectives, but involve technological and statistical challenges in enumerating larval fish abundance, growth, and survival. On the social-economic side, creation of SWH takes land out of existing floodplain land uses (principally agricultural production), may diminish sediment transport capacity and channel depths for navigation adjacent to SWH projects, and may increase flood conveyance (decrease flood peaks). Removing land from existing land uses and diminishing the navigation channel would be considered social-economic costs, but the potential decrease in flood peaks could be considered a social-economic benefit. These effects can be translated into social-economic values measured as agricultural revenue, transportation costs, and flood-control benefits.

Systematic thinking encouraged by conceptual models may help avoid implementation problems. For example, initial planning of SWH restoration on the Lower Missouri River neglected to consider that sediment reintroduced to the river might increase nutrient loads in the river (Biogeochemical EEC) that could potentially be transported downstream to contribute to hypoxia in Gulf of Mexico habitats [*Jacobson et al.*, 2009]. Lack of adequate consideration of offsite biogeochemical effects, and public reactions to those effects, resulted in suspension of SWH restoration activities on the Missouri River in Missouri in 2008 [*Harmon*, 2007; *National Research Council*, 2011].

5. SUMMARY AND CONCLUSIONS

The conceptual model framework described in this article is one of very many ways that river restoration can be visualized and communicated among managers, stakeholders, and scientists. Our goal in this article is to illustrate some important points about conceptual models in river restoration rather than to promote this particular model. This model illustrates the following:

1. The model defines natural and social-economic drivers as groups of forces. The natural forces produce background dynamics to the system, and the social-economic forces directly affect governance, decision making, and "uncontrolled" exploitation of the river outside the restoration decision-making process.

2. The model explicitly symbolizes a decision-making structure as the idealized interactions of managers, stakeholders, and independent scientists. Although specific governance structures may apply to specific projects, we believe these three roles are generic to river restoration and management. It has been argued that open and civil interaction among these three roles is essential for successful river management [*Rogers*, 2006].

3. The model modifies the typical driver-stressor-response framework of ecological risk assessment by depicting a stress as (1) deviation from the natural regimes (flow, sediment, temperature, light, biogeochemical, and genetic) by altering the frequency, magnitude, duration, timing, or rate of change of the fluxes or (2) imposition of hard channel constraints. Conversely, restoration can be seen as an alleviation of these stresses.

4. The model recognizes the importance of river history in conditioning future responses and communicates the need to determine whether past alterations are reversible.

5. The model structures the riverine ecosystem as a three-tiered system of EECs in order to communicate the typical propagation of management actions (or stressors) through physical/chemical processes, to habitat, to biotic responses. Uncertainty and expense in modeling or measuring responses typically increase in moving from tier 1 to tier 3.

6. The model represents social-economic characteristics in a parallel column, indicating that restoration projects typically have associated social-economic costs and benefits. The parallel structure emphasizes that planning often produces a need to quantify trade-offs between ecological and social-economic benefits and costs.

7. Performance measures are associated with each EEC. The performance measures are maintained as separate entities to emphasize that they are abstractions of the actual ecosystem, and choice of measures depends on their perceived costs, information content, and relevance to management decisions.

8. The model includes completion of the adaptive management loop by illustrating performance evaluation relative to reference conditions or success criteria and showing the movement of that information back into the decision-making process. Whether learning is sufficient to trigger change depends on the organizational framework and the social, economic, and legal constraints of the decision-making process.

Because of the inherent social nature of river restoration, conceptual models have particular importance as a medium to promote communication among participants. For managers, a conceptual model for a restoration project can serve to communicate expectations and the nature of the restoration process to stakeholders, while also serving to document that planning has considered a broad suite of ecosystem characteristics. For stakeholders, the conceptual model can be important for learning about the riverine ecosystem while also serving as a structure for communicating their understanding of the ecosystem to managers and scientists. Because scientists involved in river restoration often represent diverse disciplines, the conceptual model also serves as a mechanism to promote necessary cross-disciplinary communication and to indicate the roles of scientific disciplines within the overall restoration process.

Conceptual models like the one illustrated here are probably best employed as a tool in an interactive, collaborative process. In a collaborative exercise, the generic framework of drivers, regimes, history, and EECs can be used as a structure to organize understanding while still allowing participants to explore processes they believe to be important by connecting lines among EECs and selecting performance measures. The lines represent hypotheses about how actions propagate through the ecosystem, and the model can accommodate many lines to indicate many hypotheses, rendered to depict participants' beliefs in their relative importance. The framework of EECs enforces consideration of a broad view of riverine ecosystems, helping to assure that key components are not overlooked while the tiered structure of EECs indicates the need to select measures based on their information content.

Broadly defined models like the one presented here can guide development of more detailed conceptual and simulation models. Spatial and temporal variability can be addressed by expanding the number of conceptual models for specific river reaches or specific time periods in evolution of a project [*Woodward et al.*, 2008]. Detailed conceptual models can be constructed by extracting specific pathways and EEC components for further development as process-based simulations [*Walters et al.*, 2000] or probabilistic models like Bayesian networks [*Stewart-Koster et al.*, 2010]. The more complex models are likely to achieve greater buy-in from all participants if they are introduced through a conceptual modeling process that encourages systematic thinking and effectively communicates the scope and complexity of ecological restoration.

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Setting Goals in River Restoration: When and Where Can the River "Heal Itself"?

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Ecological research demonstrates that the most diverse, ecologically valuable river habitats are those associated with dynamically migrating, flooding river channels. Thus, allowing the river channel to "heal itself" through setting aside a channel migration zone, or erodible corridor, is the most sustainable strategy for ecological restoration. The width and extent of channel can be set from historical channel migration and model predictions of future migration. However, the approach is not universally applicable because not all rivers have sufficient stream power and sediment load to reestablish channel complexity on a time scale of decades to years, and many are restricted by levees and infrastructure on floodplains that preclude allowing the river a wide corridor. A bivariate plot of stream power/sediment load (*y* axis) and degree of encroachment (urban, agricultural, etc.) (*x* axis) is proposed as a framework for evaluating the suitability of various restoration approaches. Erodible corridors are most appropriate where both the potential for channel dynamics and available space are high. In highly modified, urban channels, runoff patterns are altered, and bottomlands are usually encroached by development, making a wide corridor infeasible. There, restoration projects can still feature deliberately installed components such as riparian trees and trails with the social benefits of public education and providing recreation to underserved families. Intermediate approaches include partial restoration of flow and sediment load below dams and "anticipatory management": sites of bank erosion are anticipated, and infrastructure is set back in advance of floods, to prevent "emergency" dumping of concrete rubble down eroding banks during high water.

1. INTRODUCTION

Ecological research demonstrates that the most diverse, ecologically valuable river habitats are those associated with dynamically migrating, flooding river channels [*Ward and Stanford*, 1995; *Ward et al.*, 1999, *Naiman et al.*, 2005]. Yet eroding banks may create conflicts with human uses, and there is a long tradition of measures to protect riverbanks from erosion. Ironically, many of the projects funded as "restoration" in North America have been oriented toward "stabilizing" banks, i.e., arresting bank erosion, which is implicitly assumed to be negative. The most common projects involve use of large logs, root wads, and boulders to stabilize eroding banks, along with planting of willow (*Salix* spp.) and other woody riparian plants to stabilize banks [*Bernhardt et al.*, 2005]. The underlying conflict with habitat needs for fish and other organisms is commonly ignored.

There is increasing recognition that the most effective and sustainable approach to restoring the ecological value of rivers is to let them "heal themselves" by facilitating or restoring the physical processes of flooding, sediment transport, erosion, deposition, and channel change that create and maintain complex river forms [*Beechie et al.*, 2010]. This requires both room for the river to move and flood and a

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sufficiently dynamic flow regime and sediment load to permit the channel to move and change in response to floods. In rivers whose flow regimes and sediment loads are still reasonably intact, self-healing by rivers can often be achieved by giving the river room to erode and flood, setting human infrastructure back to avoid conflicts with active channel movement [*Piégay et al.*, 2005]. This approach usually has the added virtue of reducing maintenance costs that result from conflicts between infrastructure and dynamic river processes.

Where the flow and sediment regimes have been substantially altered, simply setting back from the river will not suffice. In such cases, it may be possible to restore (at least partially) some of the natural processes, e.g., to adjust reservoir operations to restore a more natural flow regime (including seasonally appropriate high flows) and to add sediment below dams to compensate for loss of sediment load to trapping in the reservoir. Downstream of large, important reservoirs, it is usually possible to return the river flow regime only partially to its natural seasonal and interannual flow pattern. In highly urbanized settings, it may be impossible to restore process to any significant degree because space is lacking to expand the stream corridor, and the runoff patterns from the urbanized catchment have been so altered that scouring floods occur frequently, resulting in simplification of channel form.

Thus, the question is posed: When can we allow rivers to the freedom to move and develop their own complex habitats, and when is this approach impossible? This chapter provides an overview of the role of active channel migration and flooding in creating and maintaining aquatic and riparian habitat in rivers, and reviews a range of restoration approaches, from allowing the river a wide corridor in which to develop complex channel morphology to active channel reconstruction, as a function of stream power and sediment load, and availability of space for the river. The illustrations draw upon studies from many rivers and use the Sacramento River, California, as a recurring example.

2. ECOLOGICAL VALUE OF DYNAMIC RIVER CHANNELS

2.1. Channel Complexity

The process of bank erosion creates channel complexity in many river systems [*Florsheim et al.*, 2008] (Figure 1). As the outside bank erodes, the deep pools and undercut banks at its base provide cover, holding habitat for large fish, and thermal refugia during hot weather. Bank erosion also recruits large wood, as (often mature, late-successional stage species) trees fall into the channel, providing important complexity to many river systems [*Gurnell et al.*, 2002]. On many North American rivers, including the Sacramento, bare vertical banks of cohesive silt provide habitat for bank swallows (*Riparia riparia*) and other bird species, for which the banks offer a refuge inaccessible to land-based predators. Maintaining the verticality of the banks requires active bank erosion; no-longer actively eroding banks evolve from vertical to sloping profiles, along which predators can access nests.

As channels laterally migrate, scour and deposition produce bare sand and gravel bars, providing surfaces for colonization by pioneer woody riparian vegetation species. In the meantime, older established surfaces evolve through vegetative succession into later-successional-stage woodlands. The young plants of pioneer species that establish on newly deposited bars provide a marked contrast in vegetative structure to the mature, late-successional trees established on older, higher surfaces, and thus provide a range of habitats for birds and other riparian-dependent animals [*California State Lands Commission*, 1993]. The result is a palimpsest of diverse habitat types, a pattern that is constantly shifting from year to year, but which always retains a diverse mixture of vegetative structures and open bars, and which thus provides habitat for a wide range of faunal species and life stages [*Stanford et al.*, 2005].

Geomorphically produced channel complexity is also expressed, in part, as shallow water, seasonally inundated habitats on channel margins. These habitats form as a function of overbank flows (e.g., floodplains) and point bar dynamics (e.g., scour channels on point bars and edge habitat). Shallow water areas provide important rearing habitat for juvenile salmon [*Lister and Genoe*, 1970; *Bjornn and Reiser*, 1991] and have been documented to provide the best juvenile rearing habitat for Chinook salmon (*Oncorhynchus tshawytscha*) in the Sacramento River basin [*Sommer et al.*, 2001].

2.2. Former Channels and Other Floodplain Water Bodies

Oxbow lakes, sloughs, and side channels and other off-channel water bodies are created by channel cutoff or channel change and typically go through an evolutionary sequence in which sedimentation gradually converts them from aquatic to terrestrial environments [*Piégay et al.*, 2002]. The initial creation of an abandoned channel occurs through geomorphic processes such as development of tortuous meander bends leading to neck cutoff, overbank flood flows shortcutting bends and leading to chute cutoff, or avulsion caused by debris jams or by sedimentation and abandonment of braid channels. In one of many such examples, a meander bend along the Sacramento River near Hamilton City was cut off during a high flow in 1970 as a chute channel across the

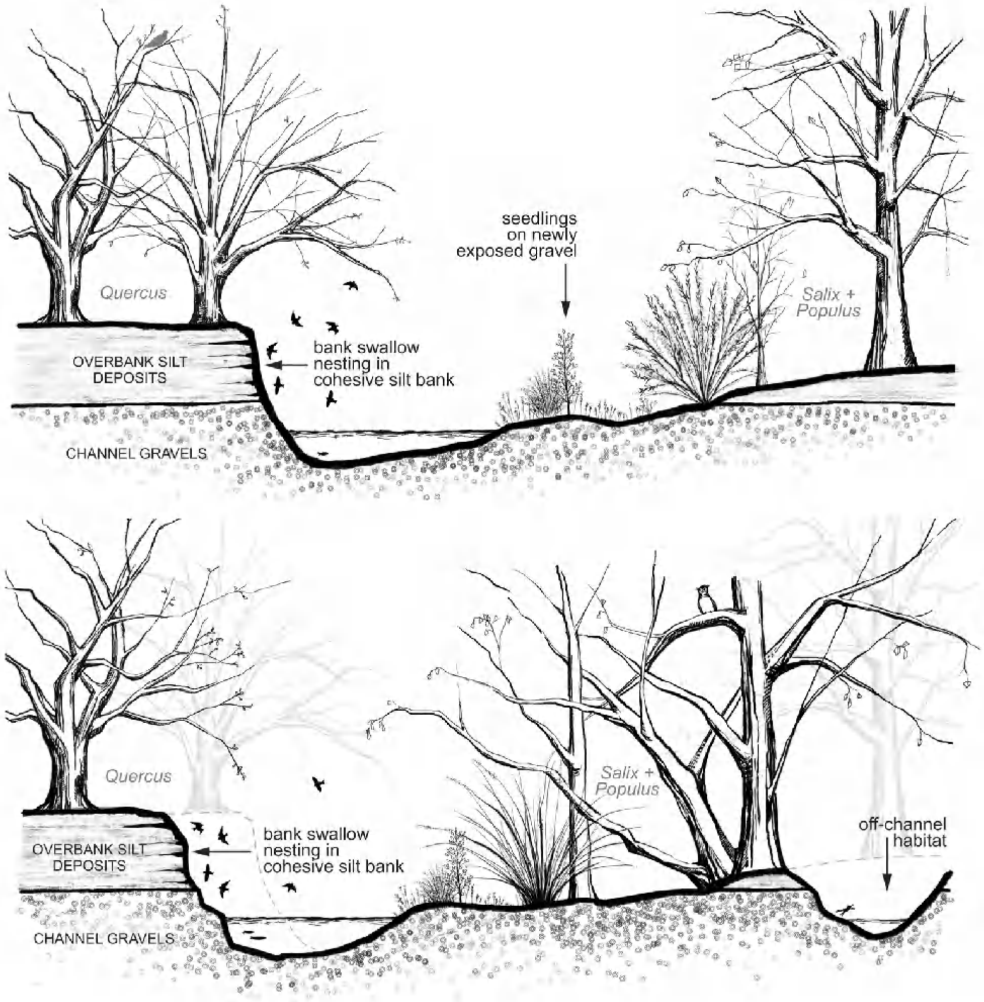

Figure 1. Lateral channel migration and its relation to riparian and aquatic habitats, Sacramento River (generalized relations).

floodplain grew in dimensions, and by the time the flood receded, the main flow of the river had been captured by this cutoff channel (Figure 2). Meander cutoffs on the Sacramento are dominantly chute cutoffs, probably owing to extensive clearing of riparian forests from floodplains, which has decreased hydraulic roughness and increased overbank flow velocities, accelerating erosion and expansion of chute channels [*Brice*, 1977]. The sinuosity of the Sacramento River has not measurably changed since the late nineteenth century, but the size of cutoffs after about 1962 was significantly smaller, probably reflecting changes in flow regime and sediment supply due to dam construction and extensive bank revetments [*Constantine and Dunne*, 2008; *Michalková et al.*, 2011].

Thus, abandoned channels owe their origins to dynamic channel migration and change. Once created, they evolve through sedimentation, vegetation colonization and succession, and the buildup of organic detritus from aquatic vegetation into progressively more terrestrial environments. The evolution of oxbow lakes is illustrated in Figure 3, which begins with the flowing river channel at the bottom of the diagram. During the initiation of a meander bend cutoff, the original main channel transitions to a side channel that is hydrologically connected at both ends. The upstream inlet to the side channel usually plugs with sediment first, creating an oxbow slough. When the downstream outlet of the side channel plugs as well, the feature becomes an oxbow lake, which begins as a fully aquatic feature. As the oxbow lake

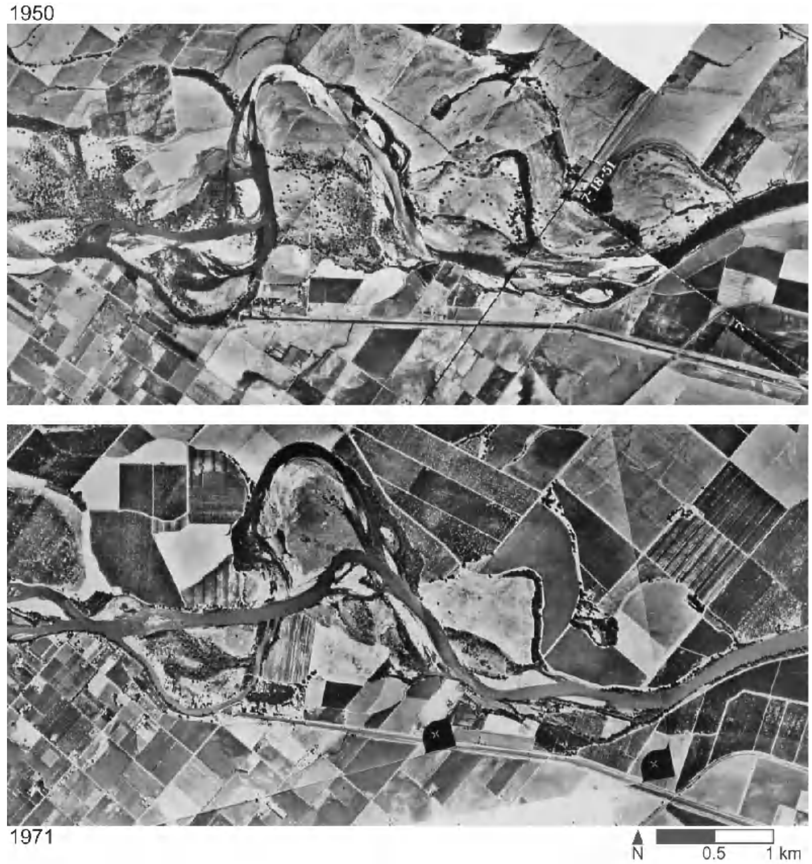

Figure 2. Meander cutoff on the Sacramento River near km 323–328, as shown on historical aerial photographs. The well-developed leftward (eastward) meander bend in the top (1951) aerial photograph cut off in the flood of 1970, leaving the former bend as an oxbow lake in the bottom (1970) photograph. The 1951 photography is by U.S. Bureau of Reclamation; 1970 photography is by U.S. Army Corps of Engineers.

fills with sediment and vegetation establishes and undergoes succession, the oxbow lake evolves from fully aquatic to progressively more terrestrial habitat, with each stage providing distinct habitats (e.g., in vegetative structure, soil conditions, frequency, and duration of inundation) that meet habitat needs for different faunal species and life stages.

The rate at which a former channel evolves from fully aquatic to terrestrial determines its persistence as aquatic habitat and its value to different species. Within the Sacramento River corridor, some oxbow lakes (such as Packer Lake) have persisted as open-water habitat for over a century, while others (such as Hartley Island) completely filled within decades. Oxbow lakes and other off-channel water bodies provide important (and diverse) habitats, and can be regarded as ecological "hot spots" on the landscape [*Amoros et al.*, 2005]. On the Sacramento River, California, off-channel water bodies provide critical habitat for a variety of native species, such as western pond turtle (*Clemmys marmorata*), Sacramento sucker (*Catostomus occidentalis*), Sacramento pikeminnow (*Ptychochelilus grandis*), California roach (*Hesperoleucus symmetricus*), and Chinook salmon (*O. tshawytscha*) [*Kondolf and Stillwater Sciences*, 2007].

2.3. Effects of Reduced Channel Dynamics on Habitat Complexity

The complex in-channel features and floodplain water bodies form, persist, and evolve as a function of flow and sediment dynamics. In many rivers, these have been altered dramatically by the emplacement of upstream reservoirs and rock revetment along the banks. Reservoir regulation typically reduces the frequency and magnitude of high flows that drive bank erosion and meander migration. Even more important are bank revetments, designed specifically to halt bank erosion and meander migration, which thus prevent creation of new cutoffs. However, other human actions may

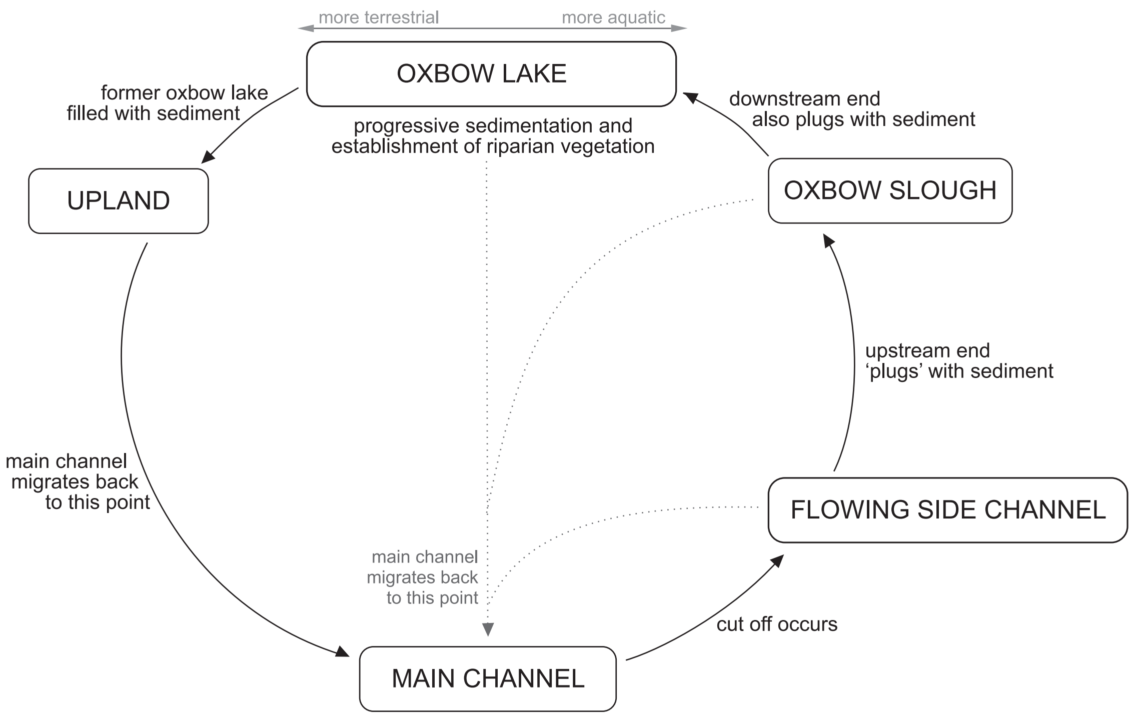

Figure 3. Conceptual model of oxbow lake evolution. A given reach of channel goes from being part of the main channel to a flowing side channel (when the new, shorter channel has been cut but some of the river water still flows through the meander bend). Because the slope is lower through the old channel than the new cutoff channel, velocities are lower, and the abandoned channel starts to fill with sediment. Usually the upstream end plugs with sediment first, creating an oxbow slough, whose downstream end is still connected hydrologically with the river. Next, the downstream end typically fills with sediment, producing an oxbow lake. Over time, the oxbow lake fills with (mostly fine-grained) sediment suspended in overbank flows, eventually reaching the elevation of the surrounding floodplain. As the oxbow lake silts up further with each overbank flow, its habitats transition from fully aquatic to more terrestrial. At any point in the cycle, the reach in question may transition abruptly back to "main channel" if the river channel erodes back to the point in question. Given that existing oxbow lakes are always undergoing the process of filling, to sustain the complex mix of habitats in river-floodplain systems requires that new oxbow lakes be frequently cut off by active channel migration.

promote meander migration and concomitant channel cutoff, such as clearing of riparian vegetation from the floodplain, which reduces hydraulic roughness of overbank flow and encourages formation of chute channels, which can lead to chute cutoffs [*Brice*, 1977].

The seasonal inundation of shallow water habitat is also affected, as flow regulation typically reduces the magnitude and frequency of flows large enough to produce overbank flooding, and levees have isolated channels from floodplains. Both factors reduce the frequency, extent, and duration of floodplain inundation.

When periodic flood scour is eliminated, as commonly occurs downstream of large storage reservoirs, riparian vegetation can encroach into the active channel, eliminating open sandbars. On the Platte River in Nebraska, these geomorphic features provide essential habitat for three species of threatened or endangered birds: whooping crane (*Grus americana*), piping plover (*Charadrius melodus*), and interior least tern (*Sterna antillarum athalassos*). Dam-induced reductions in flow regime (and artificially raised water tables) have resulted in encroachment of vegetation onto sandbars that would formerly have been scoured biannually [*Johnson*, 1994, 1997; *Murphy and Randle*, 2003]. To maintain some habitat for these important bird species, large areas of the channel are mechanically cleared of vegetation [*National Research Council (NRC)*, 2004].

On the Missouri River below Garrison Dam, reduced flood flows and sediment load have resulted in loss of open sandbar habitat and gradual conversion of young and early-successional-stage vegetation to late-successional-stage vegetation. *Johnson* [1992] documented the reduced rate of channel erosion and deposition after construction of

Garrison Dam in 1953 and the resultant loss of diversity in vegetative structure and habitat (Figure 4). Postdam, the ratio of different vegetation types changes, with the percentage of early-to-midsuccessional-stage vegetation decreasing, as later successional stages establish, and open sandbars disappear.

In sum, actively migrating meandering rivers create the greatest floodplain habitat diversity [*Ward and Stanford*, 1995], when meanders migrate across the bottomland, eroding outside banks, depositing fresh point bars, and cut off to create oxbow lakes (Figure 5). Rivers that are more dynamic, such as braided channels, have lower diversity because floods rework the bottomland so often that vegetative succession is arrested, and the landscape is dominated by bare bars and supports only early-successional-stage vegetation. Rivers, whose frequent floods have been eliminated by upstream regulation (or whose bank erosion is arrested by revetments), have lower diversity because migration is slowed or stopped, and the attendant habitat creation is thus eliminated [*Johnson*, 1992].

2.4. Implications for Restoration

The ecological literature suggests that actively migrating, flooding rivers support the greatest habitat diversity and that these habitats are constantly being renewed. They are not static features, but ever evolving in response to geomorphic processes. These insights suggest that restoration of the ecosystem is best accomplished by the geomorphic processes that create and renew habitats and thus, where processes have been impaired, by restoration of those processes [*Beechie et al.*, 2010; *Kondolf*, 2000]. While this is the preferred approach in most European countries (where restoration has become more widespread in response to requirements of the European Union (EU) Water Framework Directive), it is in stark contrast to the most common, conventional restoration approaches in North America, which have emphasized building of structural elements (or rebuilding entire channels) to create desired forms.

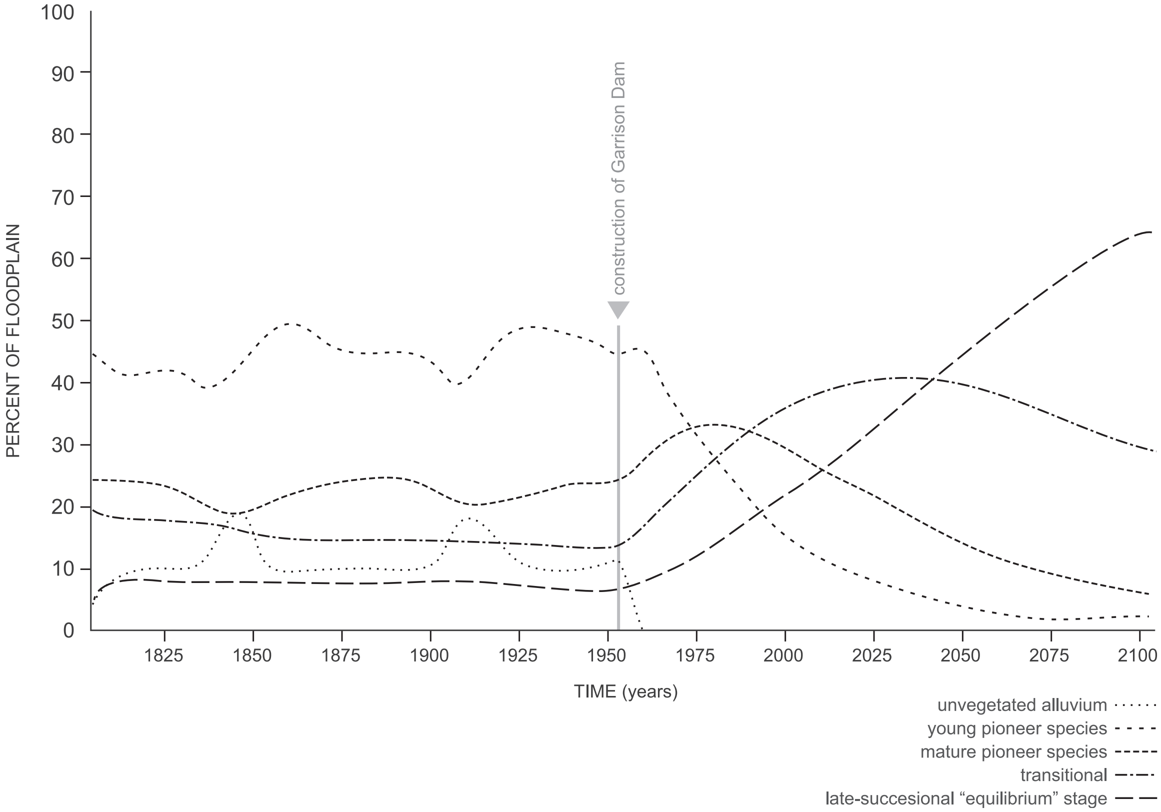

Figure 4. Floodplain habitat diversity over time for the Missouri River before and after construction of Garrison Dam in 1953, based on observations through the 1990s and model predictions thereafter by *Johnson* [1992]. The dynamic predam regime maintained a mosaic of diverse vegetative communities, dominated by juveniles of pioneer species such as *Salix* and *Populus*. After the dam cut off sediment supply and reduced flood peaks, the process of creating new surfaces for colonization by vegetation essentially stopped, but the process of vegetative succession continued, so there is a progressive shift to dominance by later successional stage vegetation. Adapted from *Johnson* [1992], reprinted with permission from S.E.L. & Associates.

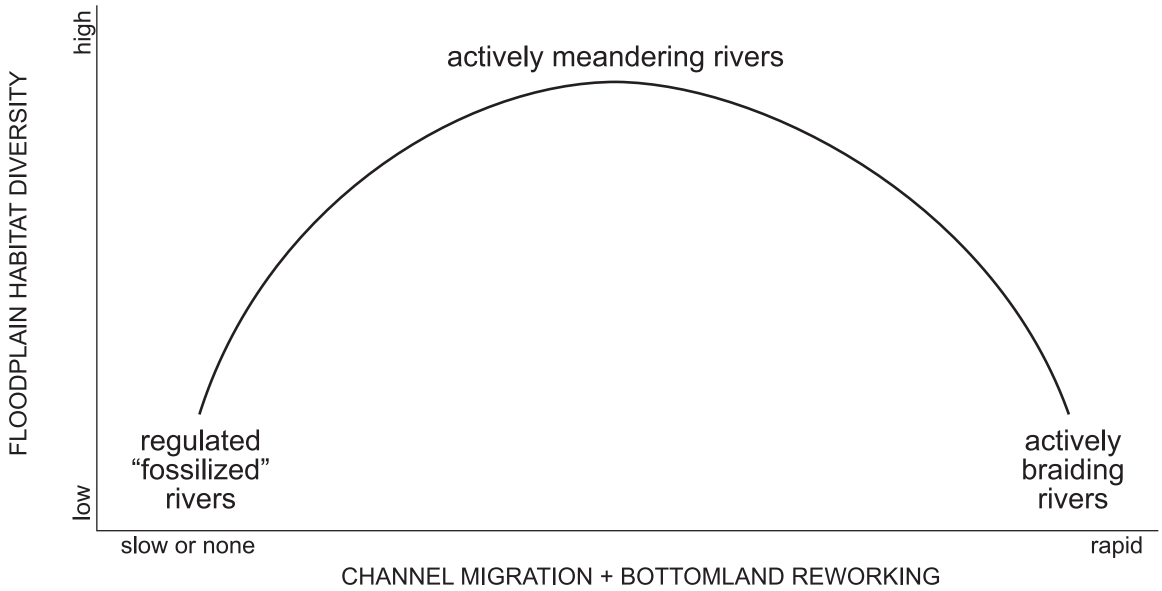

Figure 5. Floodplain habitat diversity as a function of channel migration rates. Habitat diversity is greatest when the river channel migrates actively. Braided channels are so active that they are able to support only juvenile and some adult pioneer plants, whose seedlings establish on freshly scoured or deposited bar surfaces. Formerly dynamic channels whose high flow regime and sediment supply has been reduced by upstream dams become less active, and in extreme cases, the bed forms are "fossilized." Later successional stage vegetation increasingly dominates. While there is nothing wrong with the mature later successional stages trees, the diverse mosaic is lacking. Adapted from *Ward and Stanford* [1995], reprinted with permission from John Wiley and Sons Ltd.

The form-based restoration projects so common in North America have mostly been based on templates derived from the popular Rosgen channel classification scheme and inevitably include revetment of outside banks with boulders, large logs, and basal root wads, designed to stabilize the channel (prevent migration and bank erosion) and also to provide some complexity to the static bank [*Kondolf*, 2006]. Well-documented examples of this kind of project include single-thread meandering channels built on Cuneo and Uvas creeks, California, in the mid-1990s. Despite the log and boulder revetments on their outside meander bends, both of these projects washed out, so they are widely seen as "failures" [*Kondolf*, 2006]. Consultants involved in the design of these projects have argued that they failed because the construction did not follow their specifications regarding length of revetments, etc., but these channels did not fail by erosion of revetments; rather, the streams simply cut down the middle, ignoring the revetments. In both cases, the appropriateness of attempting to build meandering channels in these high-energy, episodic streams can be questioned. But more fundamentally, what if the channels had not washed out, but remained stable. Would they have been "successful"? Perhaps they would have met their objectives of stabilizing the channels, but at a more fundamental level, would they have constituted real ecological restoration [*Palmer et al.*, 2005]? Would they have created diverse habitats for native species? Without the renewal of habitats by active migration, erosion, and deposition, the ecological value of such restoration projects that make static habitats is questionable, at least in the long term.

Ironically, one of the most significant barriers to letting rivers heal themselves is that "action agencies" need to be seen by the community (and especially those in power) to be "doing something," whether or not that something is the "right thing" in the long term. With the media saturation, short attention spans, and rapid feedback provided by new technology, there is an expectation of quick results, which tends to discount longer-term goals (sustainability and planning for future generations). Unfortunately, letting the river to do the work may be seen as "doing nothing" and may not be acceptable under these constraints, at least without significant public education.

3. THE ERODIBLE CORRIDOR OR CHANNEL MIGRATION ZONE

Setting infrastructure back from the active channel to give the river a zone in which to freely erode and deposit has been advocated by several authors in different countries, including France (the "erodible corridor" or "espace de liberté" [*Piégay* *et al.*, 2005]), Spain (the "fluvial territory" [*Ollero*, 2010]), the Netherlands ("Room for the River" [*Nijland*, 2005]), and the United States in the Pacific Northwest (the channel migration zone [*Rapp and Abbe*, 2003]), and in California (the "conservation area" of the Sacramento River). This approach has the virtues of reducing conflicts with human infrastructure and allows the river to accomplish the work of building habitats itself through dynamic channel processes [*Piégay et al.*, 1997].

Piégay et al. [2005] identified three scales at which the instability (or potential instability) of a river channel can be assessed: the river basin scale, the longitudinal reach scale (discrete reaches of 10–100 km in length), and the scale of the unstable reach, each with its own utility to management agencies and stakeholders (Table 1). *Piégay et al.* [2005] reviewed various approaches to delimit the erodible corridor width, noting that attempts to develop simple rules of thumb (such as 10 times the active channel width) had not been easily exported to other river systems. A historical overlay of past channels can be based on mapping from historical maps (typically going back about a century for accurate topographic maps, longer for manuscript maps) and aerial photographs (typically back to the 1940s). Simulation modeling can be used to predict future directions of channel erosion, but "models are frequently restricted to artificial morphologies tied to idealized representations of the river planform, such as uniform width. . . . Meander models do not account for all the degrees of freedom involved in planform adjustment" [*Piégay et al.*, 2005, p. 784].

Along the Sacramento River, mapping of historical channel courses was supplanted by predictions of channel erosion over the coming 50 years to develop the limits of the "inner river zone" [*Larsen et al.*, 2007; *Greco et al.*, 2007], in which the river was (eventually) to be allowed to migrate freely (Figure 6).

Rapp and Abbe [2003] identified four components of the channel migration zone: (1) The "historical migration zone" was the collective area occupied by the channel in the historical record, which for the Pacific Northwest of the United States encompassed roughly a century; this zone is essentially the same as the overlay of channel positions described by *Piégay et al.* [2005] and used along the Sacramento River (Figure 6) [*Larsen et al.*, 2007]. (2) The "avulsion hazard zone" is the area vulnerable to avulsion that lies outside the historical migration zone. (3) The "erosion hazard area" consists of additional areas at risk from future stream bank erosion or mass wasting of terraces. (4) The "disconnected migration area" is bottomland where channel migration is now physically prohibited by artificial structures. *Rapp and Abbe* [2003] therefore recommended the channel migration zone be delimited as the sum of the first three areas, with the fourth (artificially protected) area subtracted.

Given the advantages of the erodible corridor concept, why has the concept not been more widely applied? In part, the problem probably lies in a lack of understanding of fluvial systems by the general public and many decision makers. Rivers are commonly seen as permanent, static features, and when they flood or erode a bank, it is seen as a natural disaster, rather than an expected event linked to normal fluvial behavior. In the face of strong pressure to develop housing and other human uses, local jurisdictions with land use authority find it difficult to keep development away from the channel and off riverbanks. In addition, there are places where the concept is simply not appropriate because preexisting development restricts options, or the current flow and sediment transport regimes are inadequate for the river to rebuild its natural channel forms. *Piégay et al.* [2005, p. 775] observed that the erodible corridor concept "is perhaps most usefully applied to free-moving meandering and braided rivers in alluvial plains that can reasonably be expected to remain within a defined corridor on the time scale of interest (several decades). The [concept] therefore has most potential to be a helpful management tool in cases where there is generalized movement of the bank (e.g., a few meters of bank erosion a year along a significant length of river) and where human activities within the corridor are insufficiently developed to conflict strongly with other

Table 1. Nested Approach to Identify Potential Locations of Erodible Corridors[a]

Approach	Specific Steps	Application
River basin or network scale	At river basin scale, identify reaches with greatest divergence from reference state or with greatest mobility or potential for mobility.	Agencies responsible for meeting ecological goals can select reaches with potential to reactivate fluvial processes to restore habitat.
Longitudinal targeting (10–100 km long reaches)	Within given reach, identify locations of greater instability.	Agencies locating large-scale channel works can avoid zones of high mobility.
Unstable reach scale	Define erodible corridor width based on historical movements (from maps, air photos), vegetation patterns, sedimentology, modeling, etc.	Corridor is defined such that infrastructure is set back and channel permitted to migrate.

[a]Adapted from the work of *Piégay et al*. [2005].

Figure 6. The "inner river zone" of the Sacramento River Conservation Area from approximately km 215 to km 255. This zone (in which the river is proposed to be allowed to migrate freely except at infrastructure) was determined from channel migrations over the preceding century (based on analysis of historical maps back to the 1890s and aerial photographs) and projected channel migration for the coming 50 years (based on modeling). Unpublished GIS data layers courtesy of the California Department of Water Resources, Red Bluff.

management goals." These conditions are best met in rural areas on rivers with sufficient stream power and sediment load, as illustrated in Figure 7, a bivariate plot in which the erodible corridor approach appears as "Espace de Liberté" in the upper right, corresponding to a bottomland unencroached by urbanization (i.e., with space available adjacent to the channel), relatively undisturbed catchment conditions ("wilderness") (to the right along the *x* axis), and to high stream power and sediment supply (toward the top along the *y* axis).

4. RESTORING FLOW AND SEDIMENT LOAD

When flow or sediment load is inadequate to do the geomorphic work needed to create and maintain complex channel forms, as is frequently the case below dams, simply giving the river lateral room may not recreate the desired channel complexity. In such cases, it may be necessary to find ways to reoperate the reservoir to let out higher flows capable of supporting a dynamic meandering channel. Such reservoir reoperation schemes have successfully led to reestablishment of riparian vegetation through mimicking natural hydrographs, including postflood or wet season recession rates [*Rood et al.*, 2005]. Another specific goal of such deliberate reservoir releases is often mobilization of the channel bed, to flush fine sediment from spawning gravels or to prevent encroachment of riparian vegetation in the active channel [*Kondolf and Wilcock*, 1996].

Even if reservoirs have relatively small effects on flow regime, they still trap all of the coarser bed load sediment, and some fraction of the finer suspended load, with the effect of causing sediment starvation downstream. To compensate for this "hungry water," especially the lack of desirable sediment size fractions such as the gravels needed for salmonid spawning, sediment (commonly gravel) is added below many dams [*Kondolf*, 1997].

For this approach to work, the released high flows must be capable of mobilizing the bed, eroding banks, depositing point bars, etc. In some cases, adequate releases are not economically/politically possible, such as on the Platte River, where encroached vegetation is instead removed mechanically [*NRC*, 2004]. In the Central Valley of California, the idea of a scaled-down river is being explored, partly by adding smaller gravels than characterize the channel at present, as well as specifying flow releases that are high enough to move sediment, but considerably lower than floods that would naturally occur.

5. ANTICIPATORY MANAGEMENT

Occupying a similar position on the *x* axis of the bivariate plot (Figure 7) as "Flow + SedimentRestoration" is

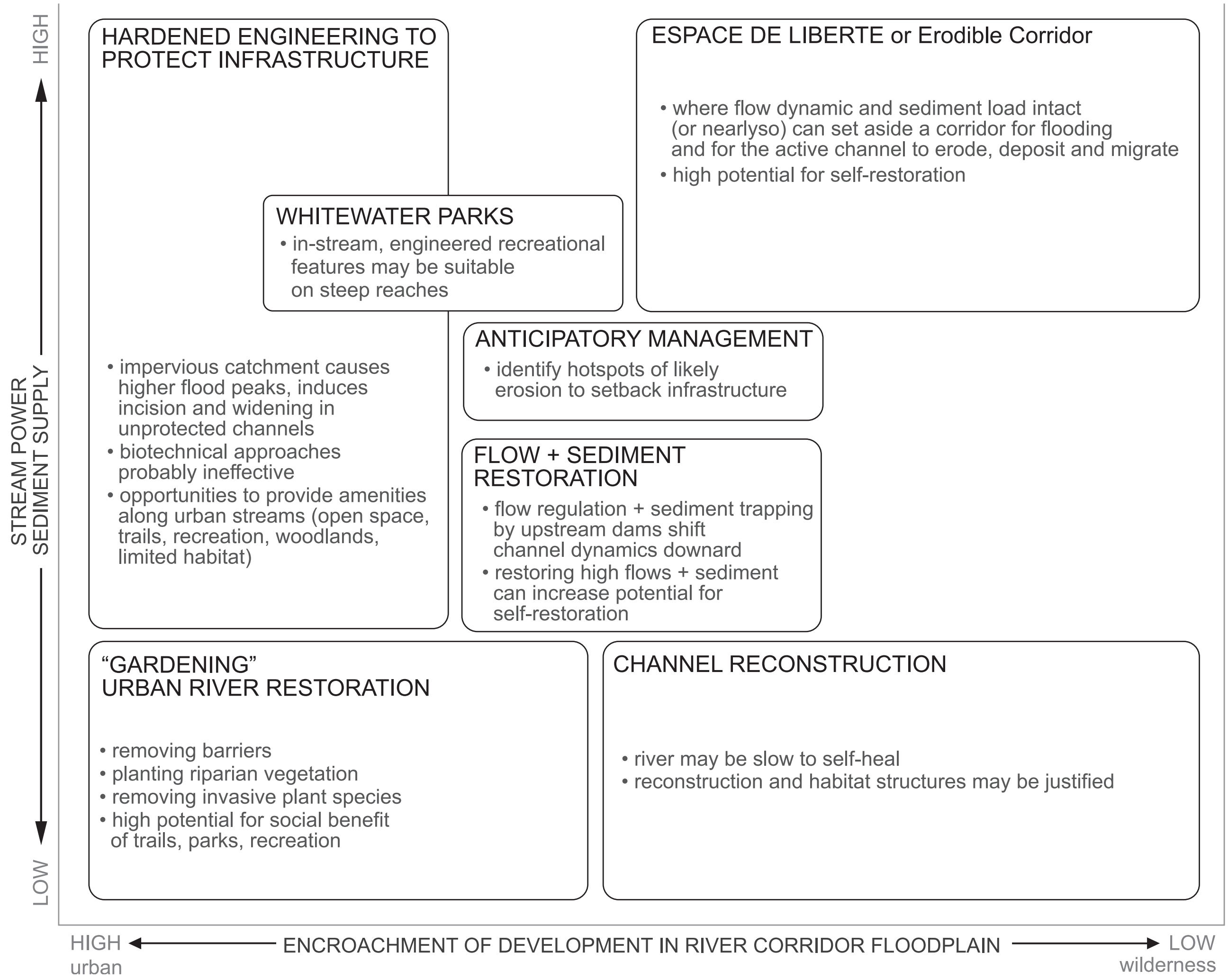

Figure 7. Suitability of self-healing approaches to restoration, such as the erodible corridor concept, depend upon the degree to which the river still retains its dynamic flow regime and sediment supply and the degree to which it is not constrained by land uses and infrastructure. The greatest potential is found in rivers with high stream power and whose sediment loads have not been reduced by upstream dams and which are located away from dense settlement or infrastructure constraints (upper right corner of diagram). Low stream power reaches are unlikely to restore themselves, so channel reconstruction is more justified (lower right). Below dams, it may be possible to partially restore flow dynamics and sediment loads through reservoir reoperation and sediment augmentation (center). Channels with adjacent high-value land uses, but which are not highly dynamic, are good candidates for anticipatory management, wherein the zones most vulnerable to bank erosion are identified, and infrastructure is set back from these banks in advance of high flows that would cause erosion. Where urban encroachment is severe, stream restoration can be likened to gardening, where individual elements are chosen for inclusion and where social benefits may outweigh ecological (left side diagram).

"Anticipatory Management." This is an approach suitable for rivers whose channels would (under current climatic and geological conditions) not migrate across the entire valley floor and where agriculture or urban developments encroach up to the channel edge so that a broad, uniform setback would entail significant economic impacts [*Beagle*, 2010]. Under anticipatory management, flood damage is treated as an inevitable, expected event, and landowners and agency staff work out a postflood response that meets the landowners' needs while protecting the integrity of aquatic habitat.

The approach is illustrated on Carneros Creek, a tributary to the Napa River, California. The catchment was largely cleared to harvest timber and create pasture in the late nineteenth century, and in the second half of the twentieth century, vineyards (and some rural residences) became the dominant land use in the catchment. The creek still supports native, anadromous steelhead trout (*Oncorhynchus mykiss*). Carneros Creek is deeply incised, and as a consequence, it experiences high shear stresses during floods. Much of the channel is simple in form and offers little habitat for fish. The best fish habitats (and most observed fish) occur at sites of active bank erosion, with undercut banks, large wood in the channel, and greater channel complexity than observed along most of the incised channel [*Beagle*, 2010]. However, the ecological functions of these eroding bank sites may be lost immediately after floods, when landowners commonly respond to bank erosion by dumping concrete rubble, boulders, even old automobiles onto the bank, under "emergency" authorities that allows them to bypass environmental permit requirements.

To protect complex habitats and prevent dumping of debris for bank protection, *Beagle* [2010] proposed an anticipatory management plan that identified where bank erosion was likely to occur (based on an analysis of bank height, bank material, channel orientation, and field evidence of recent active erosion). At sites most vulnerable to bank erosion, farmers would set back their vineyards, roads, and other infrastructure a distance equivalent to about three channel widths from the creek. They would also plant riparian trees along these setback areas, to potentially provide large wood to the channel in the future (Figure 8). Most of the large landowners along Carneros Creek already participate in the "Fish Friendly Farming" program, a voluntary program under which farmers develop a plan for their entire property and implement best management practices to reduce impacts of farming operations upon stream channels. The vineyards produce very high quality, expensive wines, so giving up

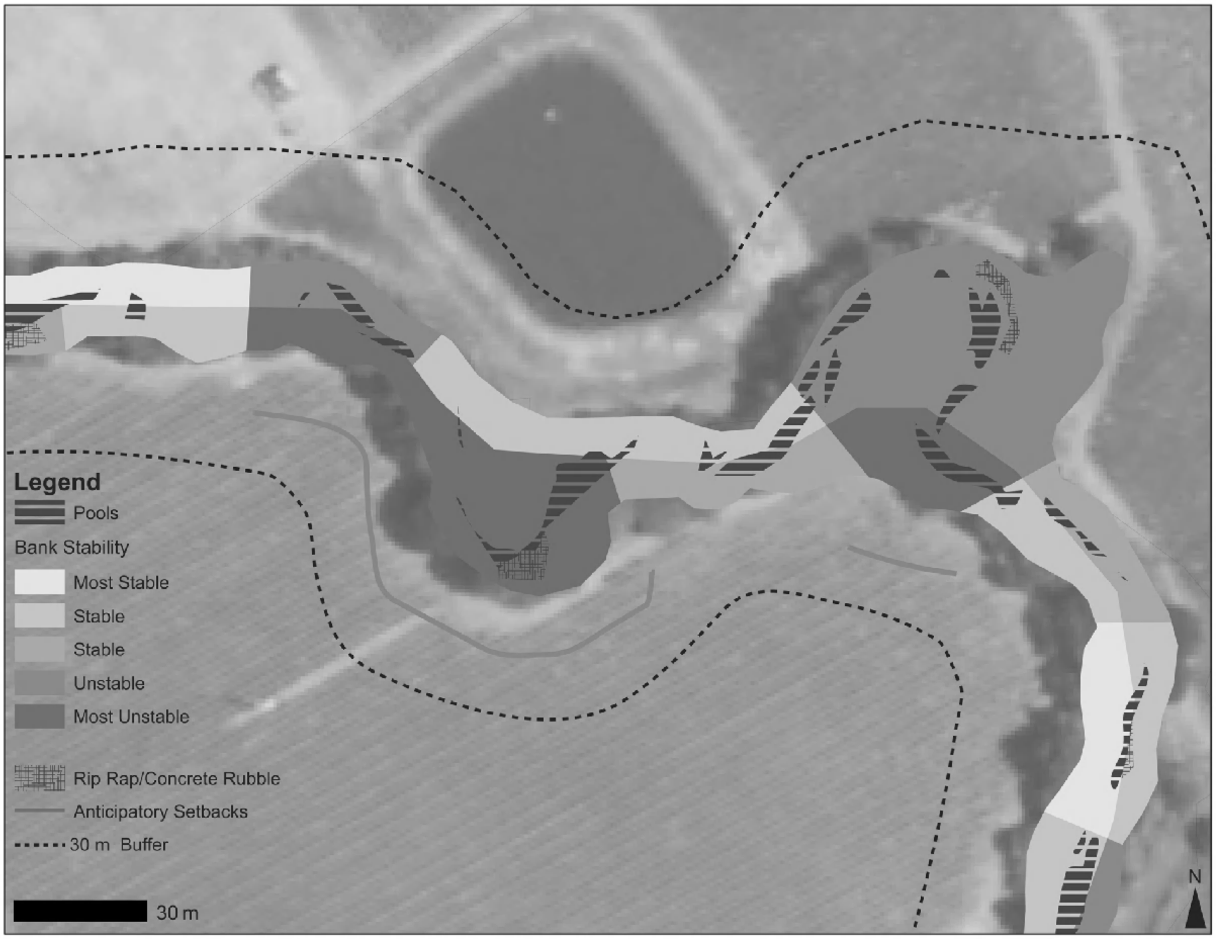

Figure 8. Map of current conditions and proposed anticipatory management for a 600 m long reach of Carneros Creek, California. At sites most likely to experience bank erosion, infrastructure is to be set back from the stream channel so that bank erosion will not create serious conflicts with farming operations. Adapted from *Beagle* [2010].

land to the creek is not a trivial matter. However, many of the landowners are environmentally aware, and largely thanks to their positive experiences with Fish Friendly Farming, initial reception to the anticipatory management approach has been positive.

6. CHANNEL RECONSTRUCTION IN LOWLAND RIVERS

Many formerly sinuous lowland rivers have been straightened to improve agricultural drainage, urban flood control, or to improve navigation. To reverse the loss of channel complexity in such rivers, reestablishing the meander beds (often termed renaturalization or remeandering) is an obvious restoration approach.

One could ask whether such rivers could reestablish their meander bends on their own, without the need for direct intervention in the form of channel reconstruction. Well-known examples of straightened rivers reasserting their former meandering nature include the Walla Walla River in southeastern Washington state, United States, which broke through its straightened channel levees in a flood in the 1960s, as captured in a well-known aerial view [*Kondolf*, 2009]. However, in low-energy, low-sediment-load rivers, it is unclear how long this kind of self-recovery from channelization might take. In some rivers, it may be centuries, if indeed it were to occur at all. However, on the River Idle in the United Kingdom, in-channel structures installed in the 1990s to encourage the channel to meander have increased channel complexity over a 15 year period, changes that are now being quantified (P. Downs, University of Plymouth, personal communication, September 2010), so at least on a decadal time scale, even low-energy channels may be capable of self-healing. Nonetheless, on the shorter-term time scales expected by the public, active intervention in the form of channel reconstruction may be justified in lowland streams. Well-known recent examples include the Kissimmee River, Florida [*Toth*, 1993; *Koebel*, 1995] and the Brede River, Denmark [*Neilsen*, 2002], both low-energy systems whose meander bends have been successfully restored, with measurable improvements in aquatic habitat and populations of valued species. In both cases, the restored channels are not fixed by hardened banks but are allowed to have natural banks, even if that means they experience some erosion.

As exemplified by the Kissimmee and Brede rivers, channel reconstruction is most appropriate on rivers with low stream power and sediment load but which have not been intensely encroached by development, so there is room to reestablish former meander patterns (illustrated by the lower right corner of Figure 7).

7. HIGHLY MODIFIED URBAN RIVERS

On urban rivers whose catchments have been rendered largely impermeable and whose bottomlands have been encroached by urban settlement, allowing the river to "heal itself" or even to restore fluvial processes, is unlikely to succeed unless there is sufficient land available to set aside a fluvial corridor. However, such a corridor would require the purchase of multiple properties, usually at high cost, and there are inevitably some property owners who resist being moved, so this approach is inevitably more difficult to implement in most already urbanized settings. The current, posturbanization flood regime is usually not well suited to restoring complex channel forms because the exaggerated peak flows tend to scour bars and vegetation from constricted urban channels, eliminating the features that could impart some complexity. Thus, urban channels must be constructed to withstand intense flows without failing. Viewed holistically at a catchment scale, restoration of urban streams should involve upstream storm water infiltration to address the underlying hydrologic distortions that cause the channel degradation. In the absence of solutions that address the underlying causes, restoration of urban channels can be seen as treating symptoms, a form of "gardening." In the design, one can include desired elements such as riparian trees, bicycle trails, picnic areas, swimming, and wading access points, but these elements are all artificially implemented and maintained, the opposite to the erodible corridor concept, in which we leave the river alone so that its natural processes can create the habitats on its own. This space is illustrated along the left side of Figure 7.

Moreover, the ecological potential of such urban streams will always be limited, so that in seeking a balance between ecological goals and human uses, the relative benefits of designing for human enjoyment will often outweigh the potential wildlife benefits of habitat creation [*Kondolf and Yang*, 2008]. Thus, restoration projects on highly urban streams in Oakland, California, have often pitted advocates for riparian habitat against local residents: the former seek to establish dense stands of willow (*Salix* spp.), while the latter oppose them because the thick vegetation may hide illicit activities.

This is not to say that we should reject outright the option of using the river to do the work or healing itself in urban settings. However, the potential benefits and limitations of each approach need to be evaluated carefully, so that preconceived ideas of "restoration" are not inappropriately applied.

8. WHITEWATER PARKS

A special case of active human use of rivers is whitewater parks, increasingly popular in cities in the United States and

EU (also often referred to as "slalom courses"). These are reaches of river designed with drop structures to create standing waves on which kayakers and boogie-boarders can surf, with shallow, protected marginal waters suitable for wading by toddlers, etc. During higher spring flows, many of these artificial courses are used for kayak competitions, while during the base flows of summer, they attract families with children. Wingfield Park on the Truckee River in Reno is a particularly successful example, attracting thousands of users on hot summer afternoons. User surveys indicate that over 80% of users come from the immediate urban area, and many are low-income families for whom escape to more distant and expensive recreational sites would be difficult (K. Podolak, University of California, Berkeley, unpublished data, 2010). Because they require sufficient slope to create multiple drops (typically 0.30–0.40 m), these features are most appropriate toward the higher end of the *y* axis in Figure 7 and, because the demand for these features is within urban areas, they would usually plot toward the left side of the *x* axis, although this is not always the case as some such parks have been built in rural areas.

9. CONCLUSION

Where possible, allowing the river channel to "heal itself" through setting aside a channel migration zone is the most sustainable strategy for ecological restoration. The width and extent of this zone can be set based on mapping of historical channel migration and model predictions of future migration. However, the approach is not universally applicable because not all rivers will naturally have sufficient stream power and sediment to reestablish channel complexity on the management time scale of years to decades. Some rivers have had their stream power and sediment load reduced by upstream dam regulation and have become inactive. For this approach to work, in addition to requiring stream power and sediment, rivers require space. Many rivers are restricted by levees and infrastructure on floodplains that preclude allowing the river a wide corridor in which to move. Thus, in a bivariate plot of stream power/sediment load (*y* axis) and degree of urban encroachment (*x* axis), the space in which such erodible corridors are most appropriate lies in the upper right, with both channel dynamics and space for the channel to move (Figure 7).

Highly modified, urban channels are typically unsuited to self-restoration by rivers because the fluvial processes that might accomplish this restoration would typically be so altered that they would no longer produce the desired channel complexity, but might instead "blow out" bars and other complex features. Moreover, urban encroachment has usually foreclosed opportunities to expand the width of the river corridor. In such cases, "gardening" may be an appropriate analogy because such urban projects can include many worthwhile features such as riparian woodlands, trails, and swimming access points, but these components are deliberately chosen and installed, rather than created by the river itself. Such projects plot along the left side of the bivariate plot (Figure 7). In such highly urban settings, the potential for real ecological restoration is limited, so the social benefits of providing recreation to disadvantaged families, and the increased potential for public education, may ultimately be more important.

Intermediate approaches include partial restoration of flow and sediment load below dams, and anticipatory management, in which sites of bank erosion are anticipated, and infrastructure is set back in advance of the erosion itself, to prevent the common "emergency" response of dumping concrete rubble down an eroding bank during high water.

River restoration can mean many things to different people. In North America, channel reconstruction and bank stabilization are among the most popular activities undertaken in the name of (and funded by) river restoration programs, but by any scientifically credible measure, they are not real ecological restoration. In urban areas and where infrastructure is threatened, active intervention and hardened bed and banks may be unavoidable given constraints of urban encroachments and altered hydrology. But wherever possible, river restoration should embrace channel dynamics and allow the river room to move and develop channel complexity through natural fluvial processes. Viewing the opportunities and potential actions along a bivariate plot can provide a framework within which to evaluate different options.

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Stream Restoration Benefits

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More than 1 billion dollars is spent annually restoring degraded streams and rivers in the United States alone because of the perceived value that healthy streams and rivers provide. Despite this immense investment, quantifying the benefits from these projects is often neglected. Without this step, it is difficult to compare restoration alternatives, prioritize projects, and determine the real returns on investment. While there are many factors that make quantification difficult, a more rigid adherence to and acceptance of the benefits assessments process will improve the ability of practitioners and sponsors to assess the value of their investment. Further, current practice can be improved with the explicit use of conceptual models, establishment of clear objectives and associated metrics, better predictive tools, quantification of uncertainty, more structured decision methods, and adaptive management. This chapter provides both a theoretical foundation and a practical framework for the vital process of assessing the benefits of stream restoration projects.

1. STATE OF THE PRACTICE

Recent studies and the development of a comprehensive database of more than 37,000 projects show that although over 1 billion dollars is spent on restoration projects each year [*Bernhardt et al.*, 2005; *Wohl et al.*, 2005], the overwhelming majority of these projects do not have explicit success criteria, and even fewer projects have postconstruction validation to ensure that the intended project goals are being achieved [*Kondolf*, 1995; *Kondolf and Micheli*, 1995; *Thompson*, 2006; *Brooks and Lake*, 2007; *Palmer et al.*, 2007]. In the few cases where systematic project assessment and monitoring were performed, it was found that half or more of the projects failed to meet the intended goals and design criteria [*Kondolf and Downs*, 2004]. Reviews of habitat restoration efforts focusing on the emplacement of in-stream structures have generally found little evidence that these techniques are effective or sustainable over a significant period of time [*Frissell and Nawa*, 1992; *Roper et al.*, 1997; *Pretty et al.*, 2003; *Roni et al.*, 2005].

In light of the above findings, it is not surprising that we have yet to fully account for the return on investment for completed projects. However, several studies have been completed that provide an indication of some of the economic benefits that can be derived from stream restoration and stewardship. Valuation methods have been used to quantify the value of fisheries as a way of estimating restoration benefits [*Dalton et al.*, 1998; *Stevens et al.*, 2000; *Morey et al.*, 2002]. Studies have shown that urban stream restoration, riparian corridors, and storm water best management practices improve nearby property values [*Wiegand et al.*, 1986; *Paterson et al.*, 1993; *U.S. Environment Protection Agency (U.S. EPA)*, 1995; *Streiner and Loomis*, 1996; *Center for Watershed Protection*, 1997], while willingness to pay surveys shed light on the broader value of stream restoration [*McDonald and Johns*, 1999; *Basnyat et al.*, 2000; *Collins et al.*, 2005; *Weber and Stewart*, 2009].

Within the private sector, no standard of practice has emerged, and there are few requirements to identify, quantify, and present the benefits of stream restoration projects. Studies consistently demonstrate that most projects fail to

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articulate clear objectives [*Kondolf and Downs*, 2004; *Palmer et al.*, 2007], so it should come as no surprise that they also fail to quantify the anticipated benefits. Indeed, the nature of project formulation generally precludes the need for evaluating benefits; a funding entity decides a stream reach should be restored for whatever reason and engages a professional to develop and implement a design. There is little incentive for the professional to further justify the effort.

Current stream restoration practice usually proceeds with the identification of problem reaches of streams that can be "fixed" by applying methods that have demonstrated success in the past. Streams and riparian corridors are generally viewed as consisting of "good" sections interspersed with "poor" segments, and it is often believed that the system can be improved by making the poor segments good. Determining how best to stabilize a stream reach while concurrently affording the greatest habitat for the species of interest, and even the desired age cohort of the species of interest, has become the focus of most conventional restoration efforts.

Federal, state, and other public water resource projects are developed under a variety of laws, policies, and institutional directives that sometimes stipulate the application of certain methods for the quantification of benefits (or impacts). The principles and guidelines (P&G) of the *U.S. Water Resources Council* [1983] provide the main basis for evaluating potential federal water resource projects and their alternatives. The P&G has guided the U.S. Army Corps of Engineers (USACE), Bureau of Reclamation, Natural Resource Conservation Service, and Tennessee Valley Authority (TVA) in project formulation since 1983. The analyses of government-funded stream restoration projects depends upon the agency and program, but generally centers upon the manipulation of habitat or, occasionally, changes in water quality. In the case of restoration actions associated with mitigation, an assessment of the quantity and quality of habitat produced is usually required.

Habitat-based approaches generally have roots in the Habitat Evaluation Procedure (HEP). HEP was developed in 1980 in response to the need to document nonmonetary values of fish and wildlife resources. It is based on the fundamental assumption that habitat quality and quantity can be numerically described using Habitat Suitability Index (HSI) models. HSI models summarize the conceptual understanding of habitat preferences of a target species scaled between 0.0 (no habitat) and 1.0 (ideal habitat) as functions of selected environmental variables, based on various sources of information [*Storch*, 2002]. In-stream flow methods and tools (e.g., Instream Flow Incremental Methodology (IFIM) and Physical Habitat Simulation System (PHABSIM) [*Bovee*, 1982]) developed by biologists and hydrologists working for regulatory agencies quantify changes in habitat as a function of discharge, utilizing HSIs as a basis for determining habitat quality [*Annear et al.*, 2002].

HSI-based methods have received much criticism because they use arbitrary classification and narrow habitat preference schemes, are rarely validated with independent data, are not readily transferable across systems due to scale and behavioral issues, involve species of dubious relevance or importance, assume that populations respond in lockstep with habitat availability, or cause complicated trade-offs [*Roloff and Kernohan*, 1999; *Ferrier*, 2002; *Gurnell et al.*, 2002]. Two major flaws exist in the assumptions of HSI models [*Railsback et al.*, 2003]: first that a species uses the selected habitat type, even if other habitats were available and second that the selected habitat provides the resources for a population to reach a sustainable carrying capacity. Despite widespread use, controversy has also accompanied the IFIM, in particular, the hydraulic and habitat models (PHABSIM) [*Mathur et al.*, 1985; *Scott and Shirvell*, 1997; *Kondolf et al.*, 2000; *Hudson et al.*, 2003]. A multiauthored review produced divergent opinions regarding the scientific defensibility of PHABSIM [*Castleberry et al.*, 1996].

Methods for benefits analysis providing alternatives to the habitat-based tools described above have been developed, and others are emerging. Improvements have also been made to the habitat-based methods, especially in the use of community- rather than species-based index models and in applications that recognize serially changing needs in complex life histories or settings with distinct seasonality. Several federal agencies have invested heavily in research to develop tools and methods for valuing ecosystems and for conducting benefits analyses for aquatic ecosystem restoration projects. The gulf between the state of the science and practice in this area is indicative of the recent growth of the field and the interest in the topic.

2. ECOSYSTEM ORGANIZATION AND THE ASSIGNMENT OF VALUE

The *National Research Council (NRC)* [1992, p. 18] defined restoration as "the return of the form and function of an ecosystem to its pre-disturbance condition." While other definitions have been advanced that capture various nuances of restoration, the reference to form and function is a common theme and is useful for conceptually organizing ecosystems. Ecosystem form, or structure, refers to both the composition of the ecosystem and to its physical and biological organization [*NRC*, 2005]. Structural characteristics vary in time and space, are unique to each system, and include, for example, stream morphology, size and distribution of bed sediments, composition of the riparian vegetation community, and the stream's hydrodynamic signature.

Ecosystem functions are the physical, chemical, and biological processes that create and sustain an ecosystem [*Fischenich*, 2005]. Functions include, for example, movement of water and sediment, decay of organic matter and cycling of nutrients, and growth and development of the organisms utilizing the ecosystem. Functions are largely responsible for the "self-organizing" and dynamic characteristics of ecosystems. Structure and function are closely linked in river corridors such that change to one is likely to affect the other.

The term ecosystem services emerged in the early 1980s to describe human-valued uses of ecosystems [*Mooney and Ehrlich*, 1997]. These uses are a derivative of the system's functions and structural characteristics and can be direct (e.g., recreational fishing, potable water, and transportation) or indirect (e.g., nutrient retention, flood control, and habitat provision). Several efforts have been made to define ecosystem services for streams and other aquatic ecosystems, but a consensus has yet to emerge.

Values are an estimate, usually subjective, of worth, merit, quality, or importance. Values can be expressed in economic (monetary) terms or using other (generally qualitative) means. Ecosystem values can be related to directly consumed outputs, such as water, food, recreation, or timber; or indirect uses that arise from the functions occurring within the ecosystem, such as habitat, water quality, and flood control. Thus, values are derived from certain ecosystem characteristics that, in turn, are determined by the underlying functions. Values can thus be applied to the ecosystem itself, to one or more of its structural elements or functions, or to any of the derived uses (services).

Farber et al. [2002, p. 387] state, "As humans are only one of many species in an ecosystem, the values they place on ecosystem functions, structures and processes may differ significantly from the values of those ecosystem characteristics to species or the maintenance (health) of the ecosystem itself." The basis for those values can be instrumental, subjective, or intrinsic [*Sagoff*, 1996]. The instrumental value of streams stems from the fact that they provide products and services necessary for human well-being. Streams also have subjective value insofar as people happen to want, like, and enjoy them; at least this is the case for healthy streams.

The intrinsic value of streams lies in the belief that they have value for their own sake, beyond that which can be ascribed to anthropocentric needs. This latter view has a cultural basis for Americans who, regardless of religious faith, tend to consider nature sacred and deserving protection [*Kempton et al.*, 1995]. Intrinsic values also have a pragmatic foundation; they promote ecological sustainability because they implicitly value future ecosystem uses that may not be highly valued in the present, but prove critical in time. Potential future values, spiritual qualities, aesthetics, the ability of exposure to natural settings to attenuate stress, inspire art, or catalyze maturation, as well as other, related roles cannot be easily monetized or quantified, but they are important and discussed by growing literatures [e.g., *Freeman*, 1993; *NRC*, 2005].

Figure 1 provides a schematic representation of the relationships among ecosystem structure and functions, ecosystem services, and the ways in which systems can be valued. The figure also introduces three fundamental strategies for organizing metrics used in benefits analyses. These include an approach based upon an assessment of the functional condition of the overall ecosystem, one based on an assessment of services, and an objective-based approach that focuses on functions and conditions directly related to the project objectives. These strategies are developed further in the following sections.

3. BENEFITS ASSESSMENT FRAMEWORK

Quantification of stream restoration benefits requires a prediction of changes in the state or condition of streams over time and assignment of a value to those changes. Motivation for assessing the benefits is generally one or more of the following: (1) to justify spending on restoration initiatives, (2) to prioritize restoration projects in the face of limited budgets; (3) to compare the benefits of different alternatives, projects, or programs; (4) to maximize the environmental benefits per dollar spent; and (5) to ensure that mitigation requirements are met or to calculate banking credits.

Results that emerge from a benefits assessment are fundamentally influenced by the way in which the benefits question is framed. To provide meaningful input to decision makers, it is important that computed benefits and costs reasonably reflect important changes that occur to the ecosystem as a consequence of the restoration actions. The general strategy best suited to characterizing the benefits and

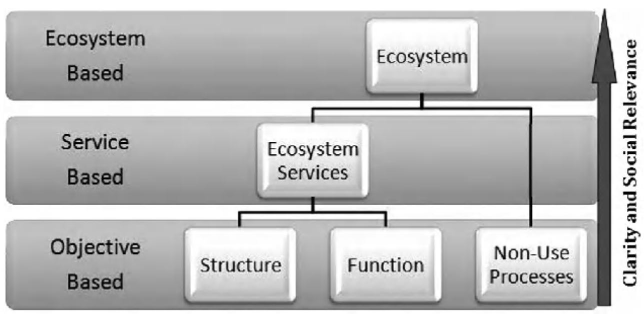

Figure 1. Organization of potential valuation metric sets and characterization strategies.

selection of the appropriate analysis scales are also important considerations that must be addressed for all projects.

3.1. Benefits Measure Change

Restoration does not create new ecosystems, but rather causes a change or changes in the condition or character of ecosystems over time. It is important to note also that ecosystems are not static; their condition changes over time in response to both natural and anthropocentric influences. Consequently, the appropriate basis for evaluating project benefits is the changes over time in the "state" of the ecosystem, as reflected by key metrics. Figure 2 shows the basis for comparison that serves as a benchmark for discussions in this chapter. The baseline is referred to as the future without-project (FWOP) condition and is represented by the projected system benefits over the planning time frame (50 years in this example) in the absence of any action. The incremental benefit afforded by each of the alternatives is the area between the benefit curve for a given alternative and the curve for the FWOP condition.

In cases for which benefits are monetized, the area under the curve in Figure 2 is a net economic benefit that can be expressed in terms of total dollars and can be converted to a present value, average annual value, etc., by applying basic economic formulae. In those instances, relative ranking of the alternatives is clear-cut, and determination of overall project worth can be made by dividing the project benefits by the costs, yielding a benefit/cost ratio or by calculating the net difference between benefits and costs. The latter approach is used for federal projects.

Difficulties in assignment of monetary values to ecosystems have limited the application of benefit cost methods for ecosystem restoration projects. When the units for metrics are not dollars, other decision support methods may be needed to evaluate alternatives. Techniques such as cost effectiveness evaluations and incremental cost assessments are often used as a way of comparing alternatives for which the benefits are described using a nonmonetary metric. Various Multi Criteria Decision Analysis (MCDA) methods can be helpful when there is not a common metric set for all alternatives.

An example of a nonmonetary metric commonly used for stream restoration projects is the expression of output in terms of the associated "habitat" created or restored. More specifically, the output is the product of the quantity of desired habitat (in acres or miles of stream) multiplied by a modifier (usually indexed from 0 to 1.0) representing the "quality" of the habitat. This habitat-quality metric is often referred to in terms of "habitat units." The same comparison strategy as shown in Figure 2 applies except that benefits are expressed as habitat units rather than dollars.

3.2. Metric Assessment Strategies

Figure 1 presents three alternative metrics strategies that can be used for benefits assessment. The lower two alternatives, objective-based and ecosystem service-based, are similar in that they rely upon identification and quantification of

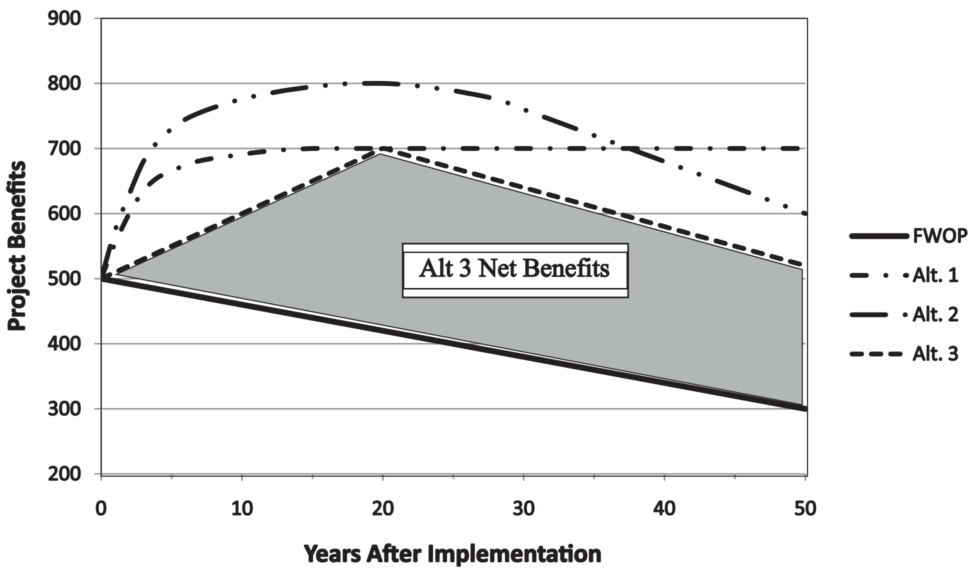

Figure 2. Schematic representation of benefit curves for restoration alternatives. The shaded area represents the net benefits for alternative 3.

key ecosystem functions or services as the basis for assessing benefits. These approaches are consistent with some existing practices for benefits quantification, and outputs can be expressed in monetary terms or in other nonmonetary units that convey ecosystem value or benefit. The third strategy, ecosystem-based, has its origins in mitigation practice and seeks to value changes from restoration in terms of overall ecosystem quality.

An important consideration for the objective- and service-based methods is identification of the ecosystem functions or services that are to be included in the analysis and those that are to be excluded. The valuation exercise, particularly when used to compare alternatives as opposed to broader analyses (such as the documentation of a program's value), may focus on only a subset of these factors, for example, habitat and water quality improvement, while ignoring all other factors. The ideal solution is to limit the considered factors to those that have a clear effect on decision making while omitting all others.

There has been a growing advocacy for the use of hydrologic and geomorphic metrics as a fundamental basis for evaluating aquatic ecosystem restoration projects. The concept stems from the realization that hydrology and geomorphic processes are overriding forces that influence almost all other functions. The Nature Conservancy (TNC), for example, advocates reestablishing or replicating the natural hydrologic variability in river systems as a necessary means to restore native biodiversity [*Richter et al.*, 2003]. The objective-based strategy is geared toward this basic approach, while acknowledging that specific objectives for each project might suggest the inclusion of additional metrics. While there is no consensus as to which specific hydrogeomorphic metrics are most ecologically relevant, methods exist to quantify many hydrologic and geomorphic parameters with reasonable certainty and replicability.

Both the objective-based and ecosystem service strategies can utilize biological metrics. Examples include community composition, species populations, provision of habitat, and maintenance of biodiversity. Most biological metrics are correlated to physical changes caused by restoration, requiring an understanding of the associated hydrogeomorphic processes but imposing additional data assessment, modeling, or other predictive techniques to translate these abiotic changes into the biological metric of interest. Furthermore, they are subject to many independent drivers outside the arena of restoration tools. This adds analytical complexity, uncertainty, and costs to most benefit evaluations. Biologically based metrics may be more socially or ecologically relevant and meaningful to decision makers in many cases, potentially justifying the added costs and uncertainties.

The use of service-based concepts for assessing ecosystems has gained considerable policy support in recent years. The Millennium Assessment, a formal effort by an international group of economists and ecologists to promote the consideration of services in decision making, illustrated the wide-ranging importance of ecosystem services [*Millennium Ecosystem Assessment*, 2005]. Most services can be monetized, providing consistent units for valuation. Services also tend to have more meaning to the general public, and decision makers then do basic ecosystem functions. In practice, however, services require the prediction of the supporting hydrologic, geomorphic, and biological processes, as well as analyses to impart a social value to those functions. Monetization adds yet an additional level of analysis and associated uncertainty. Significant advances are needed in relevant social, economic, and policy science for ecosystem services to move from a conceptual to an operational framework for decision making [*Brauman et al.*, 2007; *Daily et al.*, 2009].

The ecosystem-based strategy is founded on the notion that ecosystems form a convenient scale of organization that is understandable by the scientific community, decision makers, and the public. Under this strategy, restoration benefits can be expressed in terms of the type of the ecosystem and the degree to which its potential functionality is restored. In the simplest terms, a system's health or functionality can be expressed as a percentage of some reference condition, for example, the restoration action might improve a stream condition from 70% to 90% functional. One basis for determining functionality would be to evaluate key structure or process metrics or ecosystem services, in much the same way that the hydrogeomorphic (HGM) approach is applied to wetlands for mitigation [*Smith et al.*, 1995].

An additional modifier can be applied to assign a value to various ecosystems allowing for an easier comparison of benefits across diverse project settings (e.g., a stream, a wetland, and an estuary). The value modifier can be based upon the regional or national significance of the resource and might be established as a matter of policy. For example, recent studies emphasizing the value of headwater streams might suggest that they receive a higher significance rating as a matter of national policy than third- to fifth-order urban streams. Some classification scheme(s) sensitive to scale hierarchies would be necessary to apply this approach. Significance ratings for various ecosystems do not presently exist, although the USACE does have a method for considering significance when evaluating ecosystem restoration projects.

Figure 3 provides systematic representation of these various metric strategies and relative analytical complexity, uncertainty, and study costs for each. It demonstrates that almost all ecosystem restoration projects build from assessment of geomorphic and hydrologic conditions and that additional uncertainty, complexity, and cost is associated with metric sets that become

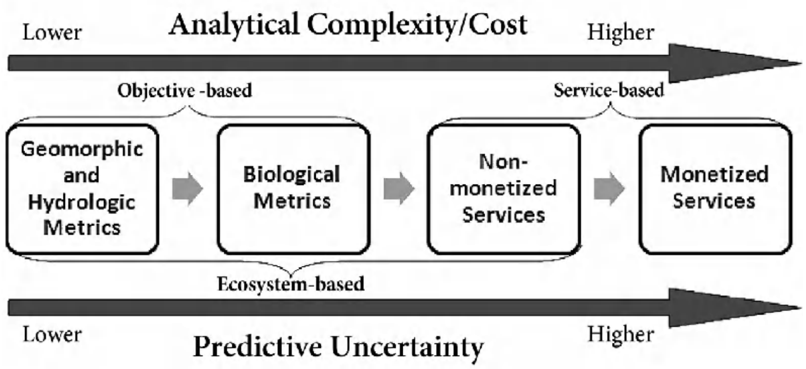

Figure 3. Metric formulation strategies and associated uncertainty, complexity, and cost.

further removed from these foundational factors. Exceptions exist: the restoration of riparian systems as a means of addressing energy, nutrient, or other water quality problems being a notable example. The evaluation of a biological metric typically involves assessing hydrologic or geomorphic consequences of restoration actions (e.g., depth, velocity, and substrate size), then converting these to some biologically relevant metric (e.g., habitat quality, diversity, and community structure). Conversion of these factors into services (e.g., recreational fishing) involves yet another level of effort with associated uncertainty, complexity, cost, and independent variables intrinsic to resource utilization. Monetizing goods and services represents yet a higher level of complexity.

Comprehensive valuation of aquatic ecosystems should be viewed as a practical improbability. The recognition that our knowledge is imperfect is at the root of issues with aggregation of assessments to higher scales and composite valuation of whole ecosystems. Multiplying one range of uncertain values by another, perhaps iteratively, let alone critical interdependencies and unforeseen behaviors of processes, services, and valuations, create the need for caveats regarding the state of the science. This does not imply no ecosystem valuation can be accomplished, simply that comprehensive valuation and summation of ecosystem goods and services to arrive at a total value is both unlikely and unnecessary.

3.3. Scalar Considerations

Identification of the spatial scale of the restoration effects is a key factor in the analysis independent from the metric set used for the analysis. The direct footprint of the project is obviously included, but projects affect ecosystem processes such that both direct and indirect impacts may extend beyond this footprint. Consideration of these impacts will yield a more inclusive analysis, but may be more difficult to accurately quantify. The study limits should extend beyond areas of direct impact to incorporate areas with indirect or secondary effects likely to affect management decisions.

The temporal scale of the analysis (the period of time over which benefits and costs are distributed) can play a crucial role in determining the results. Most restoration measures cause long-term (and potentially irreversible) changes to the ecosystem such that the project "life" is effectively indefinite. However, both benefits and costs become more uncertain and less meaningful with time from the present, suggesting practical limits for the analysis period. For federal water resource projects, 50 years has become the norm. Twenty years may be a reasonable time frame for some stream restoration projects, but the long time required for riparian system development and the equally slow response to some disturbances suggest that longer periods might provide better estimates of benefit.

Costs and benefits from stream restoration projects are unlikely to be constant over time. In order to accurately calculate benefits, the annual time streams of estimated benefits and costs must be translated into total values at a common point in time. A common and accepted practice is to establish a "base year" (usually when a project becomes operational) then use appropriate methods to convert future benefits and costs to a "present value" for the base year. If projects or alternatives with different project lives must be compared, values are often amortized over the project time horizon, yielding annualized benefits and costs.

Empirical evidence suggests that humans value immediate or near-term resources at higher levels than those acquired in the distant future [*NOAA*, 1999]. Thus, discounting has been introduced to address this time preference. The present value of a future benefit or cost is computed from:

$$PV = FV/(1+i)^n, \tag{1}$$

where PV is the present value of a benefit or cost, FV is its future value, i is the discount rate, and n is the number of periods (generally years) between the base year and when the benefit or cost occurs. For example, assume that a future benefit of a stream restoration project is an expanded catch of salmon valued at $1,000,000 in year 10. The present value of that benefit, assuming a 4% discount rate, is

$$PV = \$1{,}000{,}000/(1 + 0.04)^{10}$$

$$PV = \$675{,}584.$$

Discounting is mechanically easy, but is not without its critics. No agreement exists on the correct discount rate, and some object to the application of discounting to nonmonetary metrics. Discount rate selection can profoundly influence benefit-cost analyses. The Congressional Budget Office recommends a 2% rate based on the long-term cost of borrowing for the federal government. Since 1992, the Office of Management and Budget has recommended 7%, based on the marginal pretax rate of return on an average investment in the private sector in recent years. These figures roughly bound prevailing opinions regarding appropriate rates.

4. CONDUCTING BENEFIT ANALYSES

Benefits analysis involves multiple steps, many of which are common to all assessments and some that depend upon the specific project characteristics, metric set, and valuation techniques that are applied. These steps are summarized here:

1. Determine the purpose of the assessment. The assessment scope depends upon the potential use of the results. Common applications include the following: (1) relative comparison of different alternatives, (2) meeting mitigation requirements, and (3) determining if the benefits warrant overall costs.
2. Ensure a sound qualitative understanding of the ecosystem. This may require the development of a conceptual model representing a clear understanding of the causal mechanisms for degradation and the means to achieve restoration objectives.
3. Characterize the restoration alternatives under consideration. Specifically, define (1) how the various actions influence the ecosystem processes or condition to yield desired improvements, (2) adaptive management opportunities and how they may affect outcomes, and (3) life cycle costs for each alternative (including any adverse impacts).
4. Determine the general metric strategy and select specific metrics. This decision is based on results of previous steps, an understanding of the advantages and limitations of each strategy, available resources, policies, and so on.
5. Determine the spatial and temporal scopes of the analysis.
6. Forecast the parameters of interest. This may be the most complex and critical step in the process, potentially involving different models and analytical tools as well as the application of professional judgment.
7. Conduct any needed sensitivity and uncertainty analyses.
8. Apply any additional valuation approaches, if necessary (e.g., monetization of outputs, application of significance modifiers, etc.).
9. Make any needed comparisons and carefully document the process and results.
10. Monitor and adaptively manage the project.

4.1. Metric Selection Factors

Metrics can be (1) measurable system properties that quantify the degree of objective achievement [*Reichert et al.*, 2007], (2) mathematical functions developed for the purpose of assigning a value, as in the case of the ecosystem-based approach, or (3) ecological indicators. Metrics that can be directly measured relate to the physical, chemical, biological, or even social system attributes needed to affect the desired system response. The *U.S. EPA* [1991] distinguishes indicators on the basis of whether they best measure stresses, exposures, or responses. An accurate portrayal of the condition of a system when using indirect measures requires the use of suites of indicators, each in their appropriate role [*Schulze*, 1999].

No universally applicable metric set has been developed for stream restoration projects. Appropriate metrics for restoration projects heavily depend upon the project objectives, benefits assessment strategy, and other factors unique to the individual project. Direct measures are preferred to indicators for the purpose of quantifying benefits because the direct measures are more specific and more easily correlated to restoration actions. However, multiple metrics including both direct measures and indicators are often needed to characterize benefits, especially given the long response and recovery times for some systems. Table 1 provides examples of indicators and direct measures for a few ecosystem services and processes.

Good metrics should measure the level of performance, raise awareness and understanding, measure progress toward programmatic goals and objectives, and support decision making. The best metrics possess the following attributes:

1. They are *scientifically verifiable*. Two independent assessments would yield similar results.
2. They are *cost-effective*. The technology required to generate data for the metrics is economically feasible and does not require an intensive deployment of labor.

Table 1. Example Indicators and Measures for Select Functions[a]

Function	Description	Indicators	Measures
Maintain water quality	Water quality parameters are directly tied to support of biologic community. Riparian communities trap, retain, and remove constituents of surface and overland flow, improving water quality. Water quality influences potential use for consumption, irrigation, and other purposes.	watershed conditions (% impervious surface) stream order presence/absence/abundance of key indicator biota abnormal forms or behaviors; unusual mortalities of indicator species plant, fish, and invertebrate density, diversity, distribution, and health riparian buffer condition	conventional water quality measures (e.g., d.o., ph, conductivity, turbidity, tds, salinity, temperature, suspended sediment) bacterial counts metals and trace element sampling nutrient (n, p) tests rates of sediment deposition in channel and riparian corridor
Quality and quantity of sediments	Organisms often evolve under specific sediment regimes, and these must be preserved for the ecological health of the system. Sediment yield and character are primary variables in determining the physical character of the system.	change in banks, pools, and bars acceptable relative to other similar streams distribution, abundance, health, and diversity of aquatic biota presence of indicator species macroinvertebrate survey Redd counts Secchi depth	sediment grain size distribution embeddedness sediment yield sediment concentration and load by type/fraction armor layer size and thickness depth to bedrock sediment mineralogy
Maintain surface/ subsurface water connections and processes	Provides bidirectional flow pathways from open channel to subsurface soils, mitigating flood and draught impacts, maintaining base flow. Allows exchange of chemicals and nutrients. Provides habitat and pathways for organisms. Maintains subsurface capacity to store water.	invertebrates found in the hyporheic zone moist soil conditions, hydrophytic vegetation adjacent wetlands, hydric soil indicators groundwater elevation fluctuations watershed % impervious surface soil porosity	flux in groundwater levels stream base flow hyporheic macroinvertebrate distribution, density, and diversity isotope dating water chemistry profiles temperature recording texture, structure, moisture, redox, and porosity of adjacent soils
Regulate chemical processes and nutrient cycles	Provides for complex chemical reactions to maintain equilibrium and supply required elements to biota. Provides for acquisition, breakdown, storage, conversion, and transformation of nutrients within recurrent patterns.	presence of seasonal debris in riparian area presence/absence of indicator species and their health presence/absence of photosynthesis, fecal matter, biofilms, and decomposition products riparian vegetation composition and vigor changes in algae, periphyton, or macrophyte communities changes in trophic indicators	BOD (CBOD and NBOD) and DOC. stable carbon isotope analyses cell counts, atp concentration, respiration rates, uptake of labeled substances redox potential ion exchange capacity adsorption capacity dissolution/precipitation rates decomposition rates plant growth rates, biomass production

[a]From *Fischenich* [2005].

3. They are *easy to communicate to a wide audience*. The public would understand the scale and context and be able to interpret the metric with little additional explanation.

4. They are *changeable by human intervention*. The metric would have a causal relationship between the state of the system and the variables that are under a decision maker's control. Metrics that are independent of human action do not inform a management, policy-making, or design process.

5. They are *credible*. It would be perceived by most of the stakeholders as accurately measuring what it is intended to measure.

6. They are *scalable*. It would be directional, whether qualitative (best, good, or worst) or quantitative, as appropriate.

7. They are *relevant*. It would reflect the priorities of the public and other stakeholders and enhance the ability of managers and/or regulators to faithfully execute their stewardship responsibilities. There is no point assembling a metric no one cares about.

8. They are *sensitive* enough to capture the minimum meaningful level of change or make the smallest distinctions that are still significant, and it would have uncertainty bounds that are easy to communicate.

9. They are *minimally redundant* in that what it measures is not essentially reflected by another metric in the set being used.

10. They are *transparent* such that use of the metric avoids "readily unapparent and/or known agendas."

4.2. Ecosystem-Based Approach

The ecosystem-based approach is intended to provide a mechanism for assigning benefits that allows for comparisons across ecosystem types, facilitating prioritization and trade-off decisions in the face of limited budgets. It also offers the advantage of presenting benefits in terms that are relevant to and easily understood by scientists and the general public alike: the ecosystem itself. People generally understand the intrinsic value and importance of streams, wetlands, lakes, and estuaries. By scaling the system based upon the degree to which it functions or its overall integrity, and further delineating ecosystem types by more refined classifications, this method can integrate a variety of factors that contribute to decisions regarding the benefits or value of restoration actions.

The ecosystem-based approach requires three steps: (1) classification of the stream, (2) assignment of a value to each stream class, and (3) determination of the functionality of the stream relative to reference standards for the range of conditions to be evaluated. The ecosystem-based approach is not presently developed for stream systems and is presented herein as a concept that can serve as a basis for conducting benefits analyses, recognizing that considerable work is needed before it can be practically implemented. Many of the concepts draw upon the HGM approach to assessing wetland function [*Smith et al.*, 1995].

The development phase is carried out by an interdisciplinary team of experts (Team) and begins with the classification of streams into regional subclasses. Alternatively, an existing classification scheme [e.g., *Rosgen*, 1994] can be used provided it adequately delineates streams by function and value. The Team then develops a functional profile that describes the physical, chemical, and biological characteristics (functions) of the regional subclass, identifies which functions are most important, and determines ecosystem and landscape attributes and processes that influence each function. The functional profile is based on the experience and expertise of the Team and information collected from reference streams. Reference streams are selected from a reference domain (a defined geographic area) and represent sites that exhibit a range of variation within a particular stream type including sites that have been degraded or disturbed as well as those sites that have had little disturbance.

The Team next develops assessment models and calibrates them based on data collected from the reference streams. These models define the relationship between critical attributes and processes of the ecosystem and surrounding landscape and the capacity of a stream to perform a function. The assessment model results in a functional capacity index (FCI) (0–1.0), which estimates the capacity of a stream to perform a function relative to other streams from the same regional subclass in the reference domain. The standards used to scale functional indices are reference standards or the conditions under which the highest, sustainable level of function is achieved across the suite of functions performed by reference streams in a regional subclass.

In the implementation of this method, the assessment model is applied to the FWOP as well as to each restoration alternative to determine the FCI at various points in time over the planning period. The frequency of computation depends upon the anticipated change in the condition of the system and the need to accurately portray the changes in the quality of the system over time. If changes are linear, calculating an FCI at the beginning and end of the study period is adequate. Nonlinear response, thresholds, and variable implementation schedules may demand calculation of time steps on the order of decades or years. If the quantity of stream length or area differs among the alternatives and the FWOP, then it should be calculated at each time step as well.

Calculation of overall benefit proceeds as described for Figure 2. The stream length or ecosystem area is multiplied by the FCI and by the value for that particular system for both the alternatives and the FWOP. The benefits are the difference between the computed product for the alternative and FWOP. The value term could be expressed any number of ways ranging from an overall monetary value determined from detailed data collection and analysis to a simple semi-quantitative scale based upon factors related to ecosystem significance, public utilization, production of services, etc. The value term can be eliminated in circumstances where it does not affect decisions, for example, when simply comparing alternatives in the same ecosystem type.

4.3. Objective-Based Approach

The objective-based approach to assessing benefits of ecosystem restoration is very closely linked to the restoration process itself. Specifically, metrics that have ecological significance and are closely related to restoration objectives are used to assess project effectiveness or as a proxy for the benefits. The method relies upon a careful assessment of the conditions and processes for the ecosystem in order to evaluate causal mechanisms for degradation, critical limiting factors, and likely effects of management actions relative to the physical, chemical, and biological condition of the system. Without this sound theoretical understanding, it would be difficult, if not impossible, to develop performance criteria and meaningful measures of ecological condition.

Selected metrics should meet the criteria presented in section 4.1 and should be the most efficient way of reflecting the ecological effects of the proposed restoration work. They should be geared toward measuring change, generally in terms of both quantity and quality of some key physical, chemical, or biological condition or process. The objective-based strategy is generally consistent with the current state of practice in that it promotes identification of specific metrics related to ecological quality. These include, for example, (1) natural processes and dynamic properties that drive ecosystem self-design (i.e., hydrology and geomorphology) and (2) desired ecological end points (e.g., wildlife habitat).

Scientists have increasingly emphasized the need to focus upon processes rather than structure or form when developing stream restoration designs [*Kondolf*, 1998; *Bain et al.*, 2000; *Bennett et al.*, 2009]. The concept stems from recognition that habitat restoration will not be effective in the long term unless the ecological processes that sustain habitats are also maintained. Because habitat and biological health are closely aligned with watershed hydrology and geomorphology, proxy metrics for the effects of alternatives on these ecological services can be based on predicted hydrologic and geomorphic changes. Changes in these attributes are more directly linked to typical stream restoration actions and thus can be more readily and accurately predicted with an acceptable degree of uncertainty within study budget and time constraints. Metrics based on hydrologic and geomorphic outcomes must be ecologically meaningful, however, and thus would necessarily be place-specific and based on the central issues of concern.

Hydrologic metrics include measures of frequency, duration, magnitude, timing, and rate of change of flow. Each of these aspects of flow is an important determinant of the chemical and biological features and functions of stream ecosystems. The magnitude of flow is important for channel formation, sediment transport, and solute flux [*Doyle et al.*, 2005]. Flow duration is critical to biological processes and communities, while the timing of high and low flows exerts strong influence on biological community structure [*Poff and Ward*, 1989]. It must be recognized explicitly that rivers may respond to disturbance in episodic, complex, and unpredictable ways, especially if certain threshold conditions are crossed.

Potentially relevant geomorphic metrics include those ecologically relevant processes and structural characteristics affected by restoration measures. Examples of geomorphic processes included erosion, sediment transport and deposition, evolution of channel form, and changes in the channel morphology. Structural metrics include composition of bed material, presence of important floodplain features, riparian zone organization, channel cross section, planform, and slope. It is important that the selected metrics relate directly to relevant degradation and restoration processes as well as the ecological health of the system.

4.4. Service-Based Approach

Ecosystem services have been defined as "the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" [*Daily*, 1997, p. 3]. As this definition implies, ecosystem services can be viewed as the link between the natural properties of ecosystems and human welfare. That is, the service concept connects an ecological focus on "what ecosystems do" with an economic focus on how ecosystems satisfy human needs. As such, the concept embodies both an ecological and human dimensions. Table 2 provides a list of example ecosystem services and the various ways in which they can benefit society. Information in Table 2 is extracted from more comprehensive listings given by *Daily et al.* [2000], *Stakhiv et al.* [2003], *Fischenich* [2005], and the *Millennium Ecosystem Assessment* [2005].

The concept of using ecosystem services as a basis for decision making, especially within the public sector, has gained considerable momentum in recent years. Significant investment in service research by the U.S. EPA, USACE, and Department of Agriculture demonstrate both interest in the topic and the need to advance scientific understanding and

Table 2. Examples of Ecosystem Services Relevant to Streams[a]

Services	Comments and Examples
Provisioning	
Food	production of fish, wild game, and nuts and grains
Freshwater	storage and delivery of water for domestic, industrial, and agricultural use
Fiber and fuel	production of logs, fuel wood, peat, fodder
Transport	waterborne movement of goods and people, animal movement, etc.
Power	Hydroelectrical supply
Regulating	
Flow regulation	groundwater recharge/discharge; surface storage
Water purification	retention, recovery, and removal of excess nutrients and pollutants
Sediment processes	erosion, transport, sorting and retention of soils and sediments
Natural hazard regulation	mitigation of droughts, flood attenuation
Climate regulation	influence local and regional temperature, precipitation
Cultural	
Recreation	fishing, hunting, birding, swimming, boating, etc.
Aesthetic	subjective value associated with pleasure derived from viewscapes
Educational	opportunities for formal and informal education and training
Spiritual	inspirational or religious values
Supporting	
Soil formation	sediment retention and accumulation of organic matter
Nutrient cycling	storage, recycling, processing, and acquisition of nutrients

[a]Adapted from *Daily et al.* [2000], *Stakhiv et al.* [2003], *Fischenich* [2005], and *Millennium Ecosystem Assessment* [2005].

develop tools before it can be operational. The Council on Environmental Quality (CEQ) in its draft revision of the principles and standards for Federal water resource development (P&S) says that "consideration of ecosystem services can play a key role in evaluating water resource alternatives" [*Council on Environmental Quality*, 2009]. Accordingly, it advises that planning studies identify ecosystem services associated with the study area and account for any changes in the quantity or quality of those services in plan formulation, evaluation, and selection.

Despite considerable interest in utilizing service-based approaches for characterizing restoration benefits, several key challenges remain. First, there is no consensus regarding the scope of ecosystem services and little agreement upon which services are most significant for streams. Second, service-based approaches face the same challenge as objective-based approaches with regard to the integration of multiple metrics; the issues of interdependencies, double counting, and variable units must somehow be addressed. Third, in addition to these challenges, tools to quantify ecosystem service production functions are lacking, and analyses must build upon predictions of the structural and functional conditions of the system. This adds to the uncertainty of the predictions as well as the cost and complexity of the analysis.

The process for conducting a benefits analysis using service-based approaches essentially mirrors the objective-based approach. The primary differences lie in the added effort of linking the structural and functional changes to the service outputs, computing those outputs, and the fixing an economic value. The additional step of monetizing service benefits is optional, but provides the convenience of common units for the cost of the benefits, and consistency among the services. This facilitates the trade-off and overall investment decision making, but it can lead to the compromise of overall ecosystem integrity or sustainability if individual services are optimized at the expense of other important ecosystem functions only because they are more easily monetized or have more immediate value. Thus, the application of the service-based approach should include additional analyses as necessary to ensure ecosystem integrity.

5. TECHNIQUES FOR PREDICTING AND VALUING ECOSYSTEM OUTPUTS

Methods for characterizing the benefits of ecosystem restoration efforts can be classified in numerous ways. One division is to separate those benefits that can be monetized from those that cannot or should not. The distinction is not always clear because an economic value can theoretically be placed upon any benefit, although practical limits exist in available methods and acceptable uncertainty. In this section, approaches for predicting outputs are described in terms of the types of models typically used. Model outputs sometimes have sufficient meaning for decision making, and no further action is needed. In other cases, the outputs require valuation, usually in monetary terms. Monetizing benefits facilitates trade-offs and other difficult decisions, but the techniques for monetization of ecological outputs are often contentious.

5.1. Predictive Models

Many types of quantitative models have been developed to indicate ecological response (outputs) to natural and managed changes in ecosystem conditions. They vary widely in structure, assumptions, data and expertise requirements, and utility. While the emphasis here is on numerical models, ecological models useful for this purpose can also include statistical models, which develop relationships between and among variables based on sampled-data distributions. Statistical models can be particularly useful in close conjunction with natural reference conditions, which can be regarded as a form of physical model often useful in restoration.

Numerical models fall into two basically different output categories: index models and actual output estimation models [*Stakhiv et al.*, 2003]. Index models typically use species habitat, community habitat, biotic integrity, and functional capacity indexes to reflect relative quality of a system anchored in some optimal condition of maximum quality and varying downward toward zero as conditions change from optimum. Quality indices and geographical area are typically "integrated" by multiplying unit area (e.g., 1 acre) by the unit quality index and summing the multiples. One example of the product of this multiplication is the habitat unit of HEP [*U.S. Fish and Wildlife Service (U.S. FWS)*, 1981], which in ideal circumstances can be compared directly to other habitat units of different spatial quantities and quality index values. Alternatively, Index of Biotic Integrity (IBI) [*Karr*, 1981] and some other multimetric index models scale over a broader range and are intended to reflect biological health relative to unimpacted reference conditions independent of stream length or area. Examples of index models are listed in Table 3.

Actual output estimation models include statistical and process simulation models that are typically developed from theoretical mathematical descriptors of process and form but may be hybrid models including both theoretical and empirical elements (statistical equations). Their common intent is to simulate natural process rates and output amounts as closely as needed for the model purpose. They generate model outputs in physical units matching the actual ecosystem output measured in the field. Examples include number of days per year of floodplain inundation, numbers of fish per mile, or average input of organic matter per acre of riparian habitat per year. Of the model types, the physical process models are most common and useful for predicting restoration benefits, while statistical models may be most robust.

The number of process-based models with potential application to a stream restoration projects is far too great to permit a summary in this document. Included are various hydrologic, hydraulic, water quality, sediment transport, and geomorphic models that are useful in predicting relevant physical and chemical characteristics over time. A number of biological process models have relevance including those focusing on trophic structure, community composition and interaction, species populations, nutrient and energy utilization, growth and succession, and similar important processes.

Determining the "best" models to use for evaluating restoration of stream ecosystems is situational, depending on a number of factors including the specific processes or conditions needing evaluation, required accuracy, available resources (expertise, time, funding), needed data, and institutional acceptability. In many cases, the "correct" model does not exist, and a model must be developed or adapted to meet the needs of the specific project and circumstances. An examination of existing models by *Stakhiv et al.* [2003] yielded the following conclusions:

1. Species-habitat models are sensitive to significant effects at the species level but are not inclusive enough to formulate for restored natural ecosystem integrity.
2. Community-habitat models are inclusive enough to formulate for more natural ecosystem integrity but may be insensitive to significant effects at the species level.
3. Index models (e.g., HEP/HSI, IBI, and HGM) are most widely available but tend to exclude important systems context, require greater planner and stakeholder interpretation, and may require both community and species level index models for analysis.
4. Process simulation models (e.g., Hydrologic Engineering Center (HEC) River Analysis System (RAS) and Comprehensive Aquatic Systems Model (CASM)) are less available but more output and process explicit. They can incorporate complete systems contexts, can provide simultaneous output for conditions of naturalness and significant resources, and are superior for organizing lessons learned into improved model structure.
5. As ecosystem planning conditions grow more complicated and the science improves, the advantages of process simulation models outweigh the expediency and lower-cost advantages of index models.

5.2. Economic Valuation

The concept of economic value rests squarely on the "utilitarian" premise that human welfare derives from the satisfaction of preferences. For the purposes of assessing the economic value of ecosystem functions or services, it is important to note that measuring the value of something using dollars does not require its purchase and selling in markets. It

Table 3. Example Index Models and Methods

Method	Description	Applicability
Habitat Evaluation Procedures (HEP) *U.S. FWS* [1980, 1981]	Procedure for assessing habitat based upon habitat quality as reflected by suitability indices multiplied by habitat quantity.	Broadly used for a variety of ecosystems, but widely criticized as overly simplistic. Results are not transferrable across systems or scales.
Hydrogeomorphic Approach for Assessing Wetland Functions (HGM) *Smith et al.* [1995]	Functional capacity determined by size of wetland. Capacity of a wetland to perform a function relative to other wetlands within a regional wetland subclass in a reference domain.	Developed for wetlands and questions remain regarding the applicability to other systems and across different classifications. Sound statistical basis but requires significant investment in time to develop models.
Index of Biotic Integrity (IBI) *Karr* [1981]	Determination of integrity of a particular reach compared to a reference site based upon multiple metrics. Several variants have been developed for regional applications.	Has been applied to various systems. Scores can be compared with similar habitat types in the same region, with regions defined as part of the assessment process. Simplicity is a benefit and a limitation. May be less robust in simple, species-poor or guild-poor contexts.
Instream Flow Incremental Methodology (IFIM) *Bovee* [1982]	Index model that calculates the amount of microhabitat available for different fish life stages at varying flow levels for selected fish species.	Primarily applicable to situations involving changes clearly related to discharge or stage. Results are theoretically comparable across classes. Most widely used method for streams despite numerous criticisms.
Riverine Community Habitat Assessment and Restoration Concept (RCHARC) *Nestler et al.* [1995]	Measures habitat based upon velocity-depth distributions as compared to a reference condition standard. Variants of the method include other parameters.	Underlying concept is broadly applicable to streams, but existing models are limited to situations where model variables are applicable. Results not transferable across ecosystems or scales.
Rapid Bioassessment Protocols (RBP) *Plafkin et al.* [1989]	Subjective score of the quality of conditions for taxonomic groups.	RBP is applied within the classification of low or high gradient streams and not for comparison across stream types. Extremely subjective, but quick and easy to apply.
Wildlife Community Habitat Evaluation (WCHE) *Schroeder* [1996]	Index based on the relationship of native vertebrate species richness to several habitat variables including habitat edge and isolation	WCHE is applied to forested wetland types and is not intended for comparison across systems. Applicability for streams may be limited regionally and topically.

can be measured by estimating how much purchasing power (dollars) people would be willing to give up to obtain it (or would need to be paid to give it up), if they were forced to make a choice. Thus, economic value defined in strict economic terms is the aggregate willingness to pay (WTP) in dollars for services expected from an ecosystem or the willingness to accept the loss of those services [*NRC*, 2005].

Methods for assigning economic value to environmental outputs can be classified in terms of the way in which preferences are expressed by an individual and by the availability of supporting markets (see Figure 4). Preferences that serve as the basis for economic valuation can be revealed (e.g., in purchasing decisions) or stated (e.g., through surveys). Revealed and stated preference methods within surrogate and hypothetical markets are used to capture values of ecosystem goods and services that are not incorporated in existing market values. Table 4 provides a summary of the more common valuation methods used for ecosystem restoration.

Conducting site-specific valuation studies using these valuation approaches can be time consuming and expensive [*McComb et al.*, 2006]. Benefits transfer techniques are methods used to infer a value for an ecosystem or service based upon data collected from another similar ecosystem [*Wilson and Hoehn*, 2006]. Benefit transfer offers an economical approach to assess ecosystem services values in decision making. Although problems with the method persist and criticisms are common, benefit transfer techniques have become more accurate for estimating ecosystem services values as valuation studies have grown over the years and through the application of simple guidelines, developed by economists, for improving validity and accuracy [*Wilson and Hoehn*, 2006; *Plummer*, 2009].

Adequate data, let alone complete data, are often not available when making decisions. In these cases, more informed decisions are promoted by using alternative analytical strategies. Qualitative discussions of the benefits could be

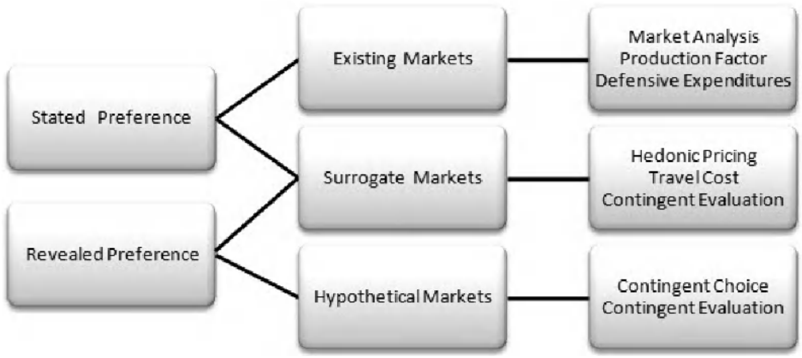

Figure 4. Economic valuation methods.

included in cases where quantitative analysis is not possible. Such discussions should address specifically why such quantitative analysis is not feasible and the reasons why the qualitative data is relevant. Breakeven analysis can be used in cases where risk or valuation data is lacking to estimate the number of units affected or willingness-to-pay value required to "break even" on a given project. Decision makers can determine whether the breakeven estimate is reasonable or not. Bounded analysis could be used when values are available for high-end and low-end scenarios for ecosystem services and environmental quality to create upper and lower bounds for the value [*University of Washington*, 2009].

There are many challenges to ecosystem valuation. Those who affirm the intrinsic value of ecosystems often object to the very idea of quantifying the value of environmental goods and services, comparing this to trying to value human life [*NRC*, 2005]. Environmental resources are particularly hard to quantify due to their broad range of intangible benefits and multiple value options [*Hussen*, 2001]. Accounting for the full range of values from aquatic ecosystems without "double counting" can be difficult, especially when multiple valuation methods are used [*Randall*, 1991]. The lack of markets make valuation in economic terms reliant upon methods that are often criticized [*Freeman*, 1993]. The selection of appropriate metrics for nonmonetary benefits is difficult and contentious, and there are no generally accepted standards.

6. OTHER ANALYTICAL METHODS

6.1. Benefit Cost Analysis

The key to all complex decisions is a skillful evaluation of trade-offs, in this case between various restoration alternatives and doing nothing. No existing decision-making protocol will establish, by itself, which of these choices to make, although protocols can certainly help organize the information [*Cairns*, 2006]. One common strategy is to evaluate the potential return on the investment in terms of benefits (monetary or otherwise) relative to the costs.

The formal process for this evaluation when the investment is a public expenditure is often referred to as benefit-cost analysis (BCA). The 1936 U.S. Flood Control Act, which required that the benefits of flood-control projects exceed their costs, caused the USACE to develop and adopt BCA as a basis for evaluating projects. Since then, cost-benefit techniques have gradually developed to the extent that substantial guidance now exists on how public projects should be appraised, and BCA methods are employed by agencies in many countries around the world [*Tevfik*, 1996].

Economic valuation plays a central role in the application of BCA, since BCA requires an estimate of the benefits and costs of each alternative using a common method (economic valuation) and metric (dollars) so that the two can be compared [*NRC*, 2005]. Comparison of costs and benefits allows an explicit consideration of trade-offs that are almost inevitably involved in restoration projects. These evaluations are particularly useful for (1) comparing the relative benefits and costs of different alternatives to select the preferred alternative and (2) determining whether the benefits are "worth" the costs.

Ideally, BCA provides objective information to a decision maker about quantifiable costs and benefits in common terms (dollars). The decision maker may then compare the costs and benefits of the decision and make a more informed decision than possible without them. In practice, the application of BCA is quite complicated. Benefits and costs are often difficult to identify, difficult to measure or monetize, and highly uncertain [*NRC*, 1999]. Additionally, although the BCA process aims for objectivity, analysts must make many subjective decisions and assumptions. These might include the choice of discount rate, whether and how to value environmental amenities (which are not traded in a marketplace), and what categories of benefits and costs to use. For

Table 4. Methods for Economic Valuation[a]

Method	Applicable To	Description and Importance	Constraints and Limitations
		Market Techniques	
Market price	Direct use values, especially wetland products.	The value is estimated from the price in commercial markets (law of supply and demand)	Market imperfections (subsidies, lack of transparency) and policy distort the market price.
Damage cost avoided, replacement cost or substitute cost	Indirect use values: flood protection, avoided erosion, pollution control, water retention, etc.,	Value of organic pollutant's removal estimated from the cost of building\ running treatment plant (substitute cost). Value of flood control derived from damage if flooding would occur (damage cost avoided).	Assumes that cost of avoided damage or substitutes match the original benefit. External circumstances may change the value of the original expected benefit and the method may therefore lead to under- or overestimates. Insurance companies interested in this method.
Productivity method	For specific wetland goods and services: water, soils, humidity in the air . . .	Estimates economic values for wetland products\services that contribute to the production of commercially marketed goods	Although methodology is straightforward and data requirements are limited, the method only works for some goods or services.
		Nonmarket Techniques	
Travel cost	Recreation and tourism	The recreational value of a site is estimated from the amount of money that people spend on reaching the site.	Only provides an estimate. Overestimates stem from other reasons for traveling to that area. Requires a large amount of quantitative data.
Hedonic pricing	Some aspects of indirect use, future use and nonuse values	Used when wetland values influence the price of marketed goods. Clean air, large surface of water or aesthetic views increase price of houses or land.	Captures people's willingness to pay for perceived benefits. Requires awareness of the link between the environmental attributes and benefits, else value not reflected in price. Very data intensive.
Contingent valuation	Recreation, tourism and nonuse values	Asks people directly how much they are willing to pay for specific services. It is often the only way to estimate nonuse values. Also referred to as a "stated preference method."	Possible bias in interview techniques. The most controversial of the nonmarket methods but one of the only ways to assign monetary values to nonuse values of ecosystems that do not involve market purchases.
Contingent choice method	For all wetland goods and services	Estimate values based on asking people to make trade-offs among sets of ecosystem or environmental services	Willingness to pay is inferred from trade-offs that include cost attribute. This is a very good method to help decision makers to rank policy options.
Benefit transfer method	For ecosystem services in general and recreational uses in particular	Estimates economic values by transferring existing benefit estimates from studies already completed for another location or context.	Used if it is too expensive to conduct a new full economic valuation for a specific site. Only as accurate as the initial study. Extrapolation limited to sites with the same characteristics.

[a]Adapted from *U.S. Army Corps of Engineers* [2004].

federal water resource projects, guidance like the P&G is used to ensure that subjective decisions are made as consistently as possible across projects and agencies.

6.2. Cost Effectiveness and Incremental Cost Analysis (CE/ICA)

CE/ICA is a form of efficiency analysis that serves to refine and illustrate trade-offs among a set of alternatives for which the benefits are expressed in a single or aggregated nonmonetary metric. The combined use of CE/ICA allows the optimization of ecosystem restoration outputs, supply side (outputs) without consideration for the demand (measured by WTP). The approach is widely used on federal water resource development projects, and tools exist to help in its implementation (Institute for Water Resources Planning Suite (IWRPLAN), downloadable public domain model for conducting CE/ICA analyses, U.S. Army Corps of Engineers Institute for Water Resources, Washington, D. C., available at http://www.pmcl.com/iwrplan/GenInfoOverview.asp IWRPLAN 2010, site accessed 1 August 2010). Cost effectiveness (CE) analysis weighs the costs of each project plan against its nonmonetary measure of output. The CE analysis screens out plans that are not cost effective from further consideration to ensure that the least cost alternative plan is identified for each possible level of output. Any particular plan is not cost effective if the same or a larger output level could be produced by another plan at less cost, or if a larger output level could be produced by another plan at the same cost. The plans that remain after this screening process is performed define the "CE frontier," or the set of cost-effective (or "nondominated") plans associated with successively higher possible levels of ecosystem outputs.

Once all cost-effective plans have been identified, incremental cost (IC) analysis can be used to help answer "What level of restoration output is worth it?" The IC analysis identifies incremental costs per unit output gained from moving from one plan to the next higher-output plan. This information helps to identify plans that capture production efficiencies with respect to the predicted output along different segments of the CE frontier (i.e., output ranges). The technique may not identify a single "best" plan, but it does eliminate those plans that are demonstrably inferior to others, and it provides useful information to support decision making.

6.3. Techniques for Comparing Dissimilar Metrics

Given the multitude and diversity of ecosystem functions and services that could serve as a basis for evaluating restoration benefits, situations involving multiple metrics with different units of measure are not uncommon. Techniques facilitating comparisons and trade-offs have been well-studied and may be coarsely divided into four categories: (1) For simpler decision problems, direct comparison of dissimilar metrics may be straightforward, rapid, and require little or no analysis beyond a qualitative comparison and evaluation. (2) Dissimilar metrics may be converted into consistent units (e.g., dollars/acre, habitat units, etc.) for direct comparison [*Daily et al.*, 2000]. (3) Transformation or normalization of metrics to an equivalent scale represents a third option for metric comparison [*Yoe*, 2002]. (4) MCDA provides a useful framework for comparing dissimilar metrics to inform environmental decision making [*Gregory and Keeney*, 2002], where normalized metrics are combined with value judgments of those involved in the decision to create an alternative metric for decision making.

7. CRITICAL CONSIDERATIONS IN CONDUCTING BENEFITS ANALYSES

7.1. Conceptual Models

One of the greatest deficiencies in our endeavors to realize the potential benefits of stream restoration lies in the lack of quality and coherency of available data and our capacity to effectively communicate our understanding as a basis for informed decision making [*Hillman and Brierley*, 2005]. The range of responses of river systems to disturbance events, whether natural or man-made, induces an inordinate degree of complexity and uncertainty in our interpretation of trends and rates of change and likely future states/conditions. Such phenomena cannot be effectively appraised through black-box exercises. Rather, system-specific insights of the causal mechanisms for degradation and likely restoration trajectories over time are required. These must be communicated appropriately to key decision makers and stakeholders in the stream restoration process.

Conceptual models are descriptions of the general functional relationships among essential components of a system. They tell the story of "how the system works" with respect to key processes and attributes and, in the case of ecosystem restoration, how the proposed alternatives aim to alter those processes or attributes to benefit the system [*Fischenich*, 2008]. Conceptual models should be required as a first step in the planning process, as they provide a key link between early planning (e.g., an effective statement of problem, need, opportunity, and constraint) and later evaluation and implementation.

Conceptual models can be invaluable in supporting benefits analyses because they provide key linkages among ecosystem components and processes and help identify appropriate metrics for the measurement of project outcomes. They provide feedback to, and help formulate, goals and objectives, indicators, and management strategies. Conceptual models also play an important role in determining indicators for monitoring and are an invaluable tool to help interpret monitoring results and explore alternative courses of management. Detailed guidance on the development of conceptual models is given by *Fischenich* [2008], and a tool to assist the preparation of conceptual models is publicly available [*Dalyander and Fischenich*, 2010].

7.2. Nonlinearity and Thresholds

Natural processes tend to vary over time and space, as well as between species, communities, and geologic, physiographic, or ecological settings. The ecosystem services these natural processes provide are therefore also highly variable. Ecosystem services are also affected by thresholds and limiting functions that influence natural processes as well as changes in the values that might be applied as opinions and needs change over time. Improvements in the understanding and quantification of nonlinearities in ecosystem functions are likely to provide more realistic ecosystem service values.

Many ecological functions are likely to be characterized by a tendency to level off (i.e., asymptotic relationship) or change dramatically (i.e., ecological thresholds) over time and space, as is the case with certain ecological processes such as population growth, predator-prey interactions, and species-area relationships [*Cain et al.*, 2008]. However, such nonlinear relationships between ecological traits and ecosystem function, and ecosystem function and service delivery, have not been explored in depth, quantitatively or conceptually.

Stream and riparian habitats and conditions are highly variable and patchy. Efforts to restore riverine systems should seek to reinstate processes that create the variability in temporal regimes and spatial diversity that characterize healthy systems. Insofar as these characteristics are important to ecosystem function and health, they should be accounted for in the calculation of benefits. This might suggest the selection of metrics that quantify or at least capture the presence or absence of dynamism and key thresholds. Additionally, the "resolution" of forecasting efforts may need to be sufficiently fine that they capture important variability in benefit streams and certainly must capture the effects of thresholds.

7.3. Uncertainty

The natural variability of river systems, and the range of spatial and temporal scales over which processes interact, introduce complexity into ecosystem-based approaches to stream rehabilitation [*Everard and Powell*, 2002]. The emerging approach is essentially probabilistic rather than deterministic, recognizing the central place of disturbance-driven temporal and spatial variability in a nonequilibrium or multi-equilibrium view of ecosystem functioning [*Landres et al.*, 1999].

All stream restoration projects face uncertainties, with the principal sources including (1) incomplete description and understanding of relevant ecosystem structure and function, (2) imprecise relationships between restoration actions and corresponding outcomes, (3) variable opinions and weightings regarding the values of ecosystem services, and (4) unpredictable and highly stochastic events and interactions affecting key processes (e.g., flooding, fire, regional climate change, etc.).

Most components within benefit-cost analysis do not have one value but are best captured as being within a range of values (see Figure 5 for example). With enough information, benefits and costs can be expressed as probability distribution functions. Analytical tools can be used to provide benefit-cost information as probabilities to better account for uncertainty. There are a number of ways in which uncertainty and associated risks can be identified and addressed for steam restoration, providing decision makers with important information that can influence the selected alternative as well as expectations for the project's benefits: (1) identify and document study elements contributing to significant uncertainty, (2) employ scenario analyses to bound possible outcomes and assess the sensitivity of outcomes to judgments regarding key inputs, (3) use Monte Carlo analysis to provide probability estimates of outcomes when feasible, and (4) use confidence intervals or probability distributions as opposed to point estimates to describe uncertainty whenever possible.

A widely used criterion for decision making is to choose the alternative that yields the greatest net benefit. Using Figure 5 as an example, Alt 1 yields the maximum predicted net benefit. Decisions might change when uncertainty is quantified, however. For example, Alt 4 may be preferred over Alt 1 in Figure 5 because, although it has a lower predicted outcome, the

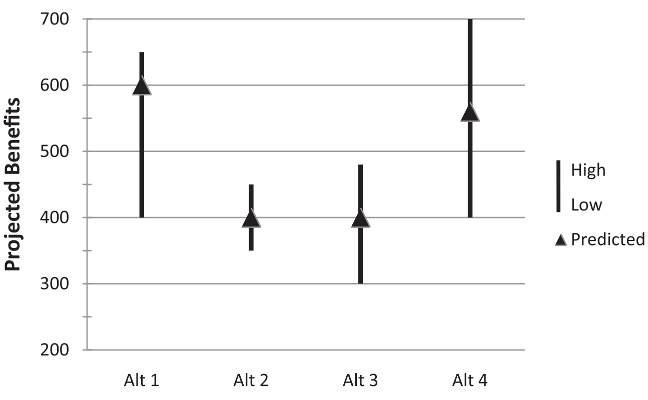

Figure 5. Influence of uncertainty on decision making.

range of likely outcomes may be regarded as more attractive. The uncertainty of the outcome of an alternative means that while the benefits could be excellent, they also have a chance of being poor. In general, faced with the choice between alternatives that generate the same expected value but with different ranges of outcomes, most people would choose the alternative with the lowest variability, implying that they are "risk averse" [*NRC*, 2005]. Alt 2 would be preferred to Alt 3 in Figure 5 following this logic.

Although considerable uncertainty exists regarding the value of ecosystem services, there is often the possibility of reducing this uncertainty over time through learning. An adaptive management program can increase the likelihood of achieving desired project outcomes in the face of uncertainty. When adaptive management is employed, alternatives with a greater range of uncertainty in outcome may be attractive to decision makers because, in theory, the more poorly performing outcomes will be eliminated through the adaptive management actions, increasing the likelihood of attaining the maximum result. Thus, if either Alt 1 or Alt 4 in Figure 5 includes adaptive management, it would likely be preferred because of the elimination of the lower part of the uncertainty bar.

7.4. Monitoring and Adaptive Management

Adaptive management recognizes that decisions are based on the best available, yet often incomplete and imperfect scientific data, information, and understanding [*Walters*, 1997]. Importantly, adaptive management provides a decision-making framework that can adjust management actions based on newly acquired information and monitored outcomes of previous decisions. This adaptive decision-making process can increase the chances that management goals and objectives (e.g., ecosystem restoration or sustainability) will be achieved despite uncertainties.

There are many benefits to the development and implementation of an adaptive management program for stream restoration projects, virtually assuring a reasonable return on investment. For the purpose of benefits analyses, greatest return is an increased probability of achieving the maximum benefits from the ecosystem restoration action. From a probabilistic standpoint, these potential benefits can be described using Figure 6.

Each of the lines shown on the graph represents a potential project outcome over a given period, and each of these outcomes has an associated probability set shown to the right of each line. The first probability is for the full set of outcomes, while the second is for only the solid lines. The dashed lines represent outcomes that adaptive management practices will prevent. Thus, the total benefits can be regarded as the sum of the products of the benefits for each possible trajectory multiplied by their probability. By eliminating the poorly performing trajectories (dashed lines), the overall probabilistic project benefits will increase due to the elimination of poorly scoring outcomes as well as the restructuring of the probabilities for the higher scoring outcomes.

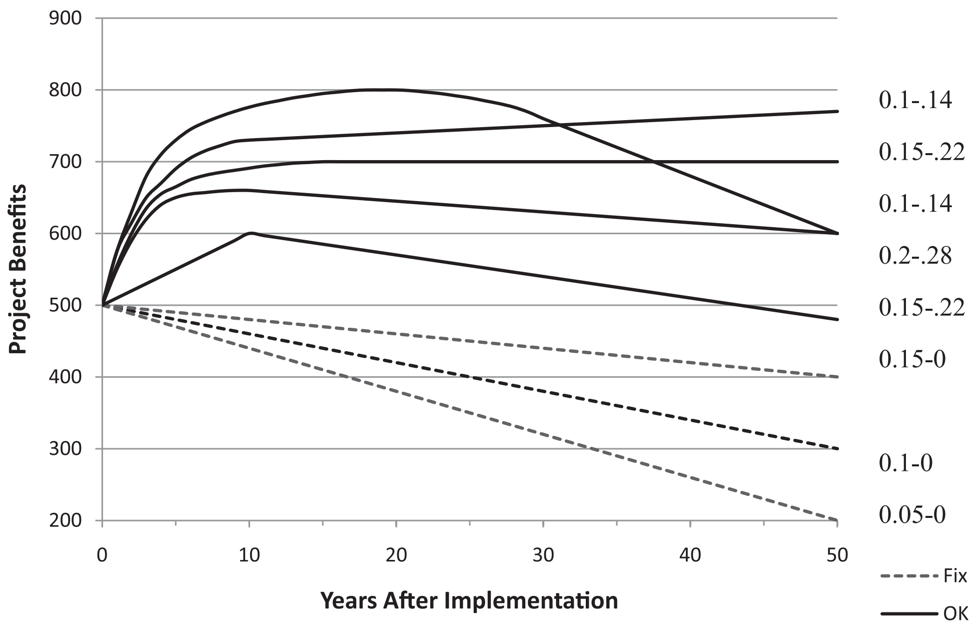

Figure 6. Quantification of the benefits of adaptive management.

Monitoring is a significant component of an adaptive management program. Additionally, project-level monitoring can (1) confirm that a project was implemented as intended, (2) provide feedback regarding the effects of the project relative to expectations, and (3) support management decisions based on trends and outcomes. Metascale monitoring can be used to document or increase program effectiveness in both ecosystem restoration (where multiple restoration actions or projects have occurred) and regulatory arenas (such as mitigation programs). To the extent practical, monitoring programs should be geared toward maximizing these benefits and contributing to a better understanding of the benefits of stream restoration.

8. DISCUSSION

Despite annual investments of over 1 billion U.S. dollars in aquatic habitat rehabilitation activities, very little is spent on monitoring or on evaluating these projects. Consequently, little information exists with which to assess project outcomes or determine if the benefits are worth the costs. Retrospective investigations of completed projects would provide useful information regarding the efficacy of various restoration strategies and possibly some indications as to the benefits derived from investments.

On a go forward basis, estimating benefits from stream restoration projects provides a useful means for comparing alternatives, prioritizing projects, and assessing overall return on investment. Critical factors in the estimation of benefits include identification of an appropriate strategy, selection of the most effective metrics, and a determination whether or not to monetize the benefits. Use of ecosystem service-based concepts for calculating benefits has gained considerable interest in recent years, and efforts are underway to develop new tools and data to support these approaches.

Interest has also grown in using more direct hydrologic and geomorphic metrics as indicators for ecological and service-based benefits sought from most stream restoration projects. The basis for this interest stems from the fact that management measures to achieve restoration objectives typically involve manipulation of hydrology or geomorphology, and tools to quantify and predict related metrics are much more developed than those for evaluating biological or service-based outputs. Decreased study cost and complexity along with the reduced uncertainty offset possible ambiguities due to the proxy nature of the metrics.

Table 5 presents a matrix that considers different classes of metrics for measuring the effects of alternatives on ecosystem support services and includes an assessment of how they compare relative to time and cost of implementation, associated uncertainty, and overall credibility. Of course, specific judgments made in any planning case would necessarily consider place- and situation-specific circumstances when selecting metrics.

Some economists argue that use and nonuse preferences for changes in ecosystem services can be directly estimated using stated preferences techniques such as "contingent valuation," which essentially involves sophisticated public surveys. These surveys elicit the choices that survey respondents would make if they had to pay for alternative states of nature. However,

Table 5. Options for Measuring Alternative Effects on Ecosystem Benefits

Basis for Evaluation	Example Performance Metrics	Time and Cost of Analysis	Uncertainty in Estimates	Scientific Credibility
Hydrologic and geomorphic structure and processes	Hydrograph shape; frequency of floodplain inundation; physical habitat distribution; sediment transport capacity	low	low to moderate	high
Biological structure and function	Index of Biotic Integrity; habitat suitability for a species or community; species richness; population estimates	low to moderate	moderate to high	moderate to high
Services rendered (non-monetized)	Recreation use-days; number of catchable fish; tons of cargo; tons of nitrogen removed	moderate to high	high	moderate to high
Ecosystem functional capacity	A classified ecosystem scaled by the functionality relative to reference conditions and (optionally) public significance	low to high	moderate to high	unknown
Economic value	Increase in property values adjacent to restored streams; commercial fishery yield; WTP for recreational opportunities	high	high	low to high

many objections to this approach have been articulated outside as well as within the economics profession. One important conceptual criticism argues that people do not view ecological services as individual consumers; instead, people view and express their preferences for such services collectively through environmental laws and other public policies.

Fischenich [2005] outlined principles necessary for the effective restoration of streams. A common theme of these principles is the understanding of key processes that occur within a system to create conditions important to the ecosystem's character and maintenance. In other words, we must know how the system operates, or functions, in order to make good management decisions. This concept is fundamental to restoring and managing ecosystems. It is equally fundamental to the assessment of the benefits from restoration projects. Selection of an appropriate method and associated metrics are largely influenced by this level of understanding.

Several measures can be employed to improve benefit analyses, independently of the method or metrics that are utilized. Quantifying and documenting the uncertainty associated with predicted conditions provides decision makers with valuable information. The considered development and use of specific conceptual ecological models guide not only decisions regarding ecosystem restoration process, but also metric selection and benefits quantification efforts. Monitoring and adaptive management programs are important not only as follow-ups to the project implementation, but also during the formulation process. Decisions regarding the potential for adaptive management actions can influence decisions and affect overall project benefits.

The principal and overarching output of ecosystem restoration should be improvement to the natural integrity of the system. In the narrow sense defined by *Karr* [1981], ecosystem integrity is the relative completeness of natural ecosystem function, structure, and associated complexity, which reflects the system's resilience and sustainability. Measuring this important and integrative characteristic would provide the best means by which to assess stream and other ecosystem restoration efforts. Some suggest that this can be accomplished by assessing the ecosystem structure and functions [e.g., *Schneider and Kay*, 1995], while others argue for a more socially based perspective such as critical ecosystem services [e.g., *Daily et al.*, 2000]. Considerable research is underway to evaluate the alternative existing methods and develop new approaches for assessing benefits. Ecosystem integrity might be a useful concept for assessing the various models and methods that are developed from these efforts.

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Natural Channel Design: Fundamental Concepts, Assumptions, and Methods

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The natural channel design (NCD) approach to river restoration emulates natural river systems and was initially developed to help redirect the manner in which past traditional river works have impacted natural river systems. The NCD approach integrates fluvial processes over temporal and spatial scales of self-formed and self-maintained natural rivers. Landscapes and stream systems must be observed in light of their evolution or successional states through various stages of adjustment. In doing so, the processes that produce a stable reference reach morphology can be inferred through time trends of river change. To understand the cause and consequence of change becomes a formidable yet essential phase in this NCD process; thus, rigorous protocols are necessary to document field observations and complete a consistent, quantitative, comparative assessment. NCD requires an understanding of process and form relations that must be formally quantified, tested, designed, and monitored. Over 67 form variables must be predicted in NCD that cannot be accurately predicted using current analytical models, which currently contain an incomplete system of equations. However, analog, empirical, and analytical methods are applied in NCD to determine and test the design variables. This chapter explains the underlying fundamental principles and concepts of NCD, definitions, assumptions, ecological integration, prediction methodologies, and minimum application requirements required for a sustainable design that strives to meet multiple objectives.

1. INTRODUCTION

To restore an impaired river is an admirable and rewarding venture; it also is one of the most challenging undertakings due to the inherent complexity, uncertainty, and risk. These circumstances should discourage most, but the cumulative anthropogenic impacts of impaired stream systems often makes the "do nothing" alternative unacceptable. Traditional river works have created unexpected major instability and environmental problems because of the unnatural conditions imposed on river systems by modifying the bankfull channel morphology associated with various streamflow and sediment regimes [*Hey*, 1997a]. The river engineering works carried out for single-purpose objectives, such as navigation, flood control, flood alleviation, and channel stabilization, have destroyed the conservation and amenity value of riverine areas [*Brookes*, 1988; *Purseglove*, 1988; *Hey*, 1997a]. Benthic and in-stream habitats and associated aquatic plant and invertebrate communities have consequently been destroyed [*Hey*, 1997a; *Brookes*, 1988]. Further consequences include downstream flooding, poor aesthetics, reduced recreation, slow natural recovery, and unsustainable maintenance [*Soar and Thorne*, 2001]. Rigid materials and methods (such as rock riprap, concrete, and gabion baskets) have also been widely applied to stabilize stream banks with limited opportunity to soften the environmental and aesthetic impacts [*Hemphill and Bramley*, 1989; *Hey et al.*, 1991]. These works have been driven by economic, social, and

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political pressures rather than the ecological health of the river.

However, people want their rivers back . . . therein lies a challenge . . . back to what? The boundary conditions and the driving variables (flow and sediment regimes) that influence channel morphology have changed from the pristine and undisturbed "pre-white settlement" conditions; thus, it is generally impractical and unsustainable to recreate the ideal pristine river channel. What can be done practically, however, is to emulate natural stable rivers that exist under the present boundary conditions and driving variables reflected in their watersheds. By designing with nature rather than against it, such approaches are more likely to be cost-effective, require less maintenance and minimize environmental impacts compared to traditional engineering solutions [*Hey*, 1997a; *Soar and Thorne*, 2001].

The natural channel design (NCD) approach to river restoration emulates natural river systems and was initially developed to help redirect the manner in which past traditional river works have impacted natural river systems [*Rosgen*, 2007]. The NCD approach integrates fluvial processes over temporal and spatial scales of self-formed and self-maintained natural rivers. Landscapes and stream systems must be observed in light of their evolution or successional states through various stages of adjustment. In doing so, the processes that produce a stable "reference reach" morphology can be inferred through time trends of river change. To understand the cause and consequence of change becomes a formidable, yet essential phase in this NCD process; thus, rigorous protocols are necessary to document field observations and complete a consistent, quantitative, and comparative watershed and channel stability assessment.

NCD involves procedures for three different reaches throughout the methodology: the "existing reach," the "reference reach," and the "proposed design reach." The "existing reach" represents the current impaired condition of the stream reach identified for potential restoration. The "reference reach" is a stable stream that represents the same "potential" stream type, valley type, flow regime, sediment regime, stream bank type, and riparian vegetation community as the existing reach. Reference reaches do not necessarily represent pristine systems [*Hughes et al.*, 1986] but have adjusted to the driving variables and boundary conditions in such a way as to be self-maintaining. The reference reach is used to establish dimensionless relations that represent the stable dimension, pattern, and profile (morphology) for a given stream type and valley type. Ranges of values are determined for each morphologic variable to represent the natural variability inherent in streams. These ranges are determined by surveying numerous cross sections and taking multiple pattern and profile measurements for each variable at the reference reach site. The values are converted to a dimensionless form by dividing by a normalization parameter, such as bankfull width, bankfull mean depth, or bankfull slope. The dimensionless relations are then extrapolated to the existing reach for scale comparisons. The dimensionless values are converted to dimensional values once the bankfull conditions are determined to obtain the "scaled" morphological characteristics for the proposed design reach. The "proposed design reach" is intended to emulate a natural stable channel that has the same stream type and valley type as the reference reach. Selection criteria and assessment procedures are described in subsequent sections.

Overall, the NCD procedure strives to put scientific principles into practice and involves detailed field measurements of the morphological, hydraulic, sedimentological, and biological characteristics of river channels. NCD requires an understanding of process and form relations that must be formally quantified, tested, designed, and monitored. Over 67 form variables must be predicted in NCD that cannot be accurately predicted using current analytical models, which currently contain an incomplete system of equations [*Hey*, 1978, 1988, 1997b, 2006; *Soar and Thorne*, 2001]. However, analog, empirical, and analytical methods are applied in NCD to establish and test the design variables. This chapter explains the underlying fundamental principles and concepts of NCD, definitions, assumptions, ecological integration, prediction methodologies, and minimum application requirements required for a sustainable design that strives to meet multiple objectives.

2. DEFINITIONS

"River restoration," as defined in this NCD approach, is to establish the physical, chemical, and biological functions of the river system that are self-regulating and emulate the natural stable form within the constraints imposed by the larger landscape conditions. A "river system" includes not only the river channel but also its related components, including adjacent floodplains, flood-prone areas (low terrace plus active floodplain), wetlands, and associated riparian communities. The "natural stable form" involves reestablishing a physical stability that integrates the processes responsible for creating and maintaining the dimension, pattern, and profile of river channels. Such form variables are based on the driving variables of flow and sediment as well as the boundary conditions of channel materials, riparian vegetation, boundary roughness, and the slope, width, and sinuosity of its valley. "River stability" is defined as a river or stream's ability in the present climate to transport the streamflows and sediment of its watershed, over time, in such a manner that the channel maintains its dimension, pattern, and profile without either aggrading or degrading [*Rosgen*, 1996, 2001b, 2006b, 2007].

The term "dynamic equilibrium" is defined by *Leopold et al.* [1964, p. 6], from the work by *Hack* [1960] extended from the work of *Gilbert* [1877], as a postulation "that there is at all times an approximate balance between the work done and imposed load and that as the landscape is lowered by erosion and solution, or is uplifted, or as processes alter with changing climate, adjustments occur that maintain this approximate balance." Dynamic equilibrium is synonymous with river stability as used in NCD. River stability is predicted and validated by field measurement and protocols presented in the assessment phase of NCD based on specific methods documented by *Rosgen* [2006b].

River stability in NCD does not mean that a river is "fixed" in place; "hardening" of the channel boundary including the streambed and stream banks is *not* an objective related to the NCD approach to river restoration. The NCD method assumes that there will be some postrestoration adjustment of the form variables over time and following floods. The allowable departure of dimension, pattern, and profile data within the range of the proposed design variables is determined by reference reach data sets that prescribe the allowable criteria. A certain amount of deposition is acceptable unless it leads to a raise of the local base level through aggradation processes. Conversely, channel scour is acceptable in a natural stable river; however, scour that over time leads to degradation or abandonment of floodplain surfaces through channel incision is not acceptable. Stream bank erosion is also expected in natural stable rivers, but concern exists when the stream bank erosion rates become accelerated.

3. NCD FUNDAMENTAL PRINCIPLES AND CONCEPTS

3.1. The Independent and Dependent Variables Related to Form and Process

Following disturbance, rivers have a central tendency to reestablish their stable form [*Mackin*, 1948; *Leopold*, 1994]. A stable channel's role is to transport the flows and sediment produced by its watershed. Underlying the complexities of river processes is an assortment of interrelated variables that determine the morphology of the present-day river. "The shape of the cross section of any river channel is a function of the flow, the quantity and character of the sediment in motion through the section, and the character or composition of the materials (including vegetation) that make up the bed and banks of the channel" [*Leopold*, 1994]. "Links between channel form and process have been the foundation of our understanding of fluvial geomorphology" [*Simon et al.*, 2007, p. 1119]. Thus, the mutual interdependence between channel process and form has been demonstrated in numerous works [e.g., *Leopold et al.*, 1964; *Schumm*, 1977; *Leopold*, 1994; *Knighton*, 1998; *Hey*, 1982]. It is a key assumption in NCD that river form and fluvial processes evolve simultaneously and operate through mutual adjustments toward self-stabilization [*Rosgen*, 1994].

Figure 1 depicts the independent driving variables of streamflow and sediment regime as the key controlling variables affecting the dependent variables of channel form. The independent controlling variables also include the boundary conditions that are associated with the form and processes of natural rivers (Figure 1). The riparian vegetation community, for example, is a boundary condition developed and maintained naturally through the integration of various valley features, soil types, soil moisture, and microclimate. Bank strength, flow resistance, and channel roughness elements (such as large woody debris) are influenced by the riparian community and are important to many of the form variables. Many of these independent variables cannot be changed (e.g., valley dimensions), and others may not practically be changed (e.g., the streambed and stream bank materials, the delivered bed load and suspended sediment, and streamflow regime). Although streamflow regime can change over time with climate or watershed recovery, NCD must facilitate a range of flows within the river system.

A total of 67 dependent form variables are obtained in NCD that relate to the driving variables and boundary conditions (Figure 1). These morphological variables are measured and analyzed to represent the range and mean values of the dimension, pattern, and profile variables for the existing and reference reach conditions. Typical dimension variables are associated with the bankfull discharge stage. Bankfull channel width and mean depth are used as normalization parameters for the morphological variables in NCD for extrapolation and comparison among rivers of various sizes. The various dimensions of bed features, including riffles, pools, runs, and glides, are measured for their unique morphology. Runs are transition features from riffles into pools, and glides are transition features from pools to riffles. Glides are typical spawning bed features where "redds" are found for salmonids associated with gentle slopes, shallow depths, and natural sorting of bed materials. The glides, being associated with adverse slopes, create a hyporheic exchange and upwelling forces. The channel dimensions also include the inner berm feature associated with the low-flow channel. Such river data is required to directly incorporate these various features into NCD.

The pattern variables reflect the boundary conditions and, similar to channel dimensions, are also related to the bankfull channel width. Pattern variables include the meander geometry relations of stream meander length, radius of curvature,

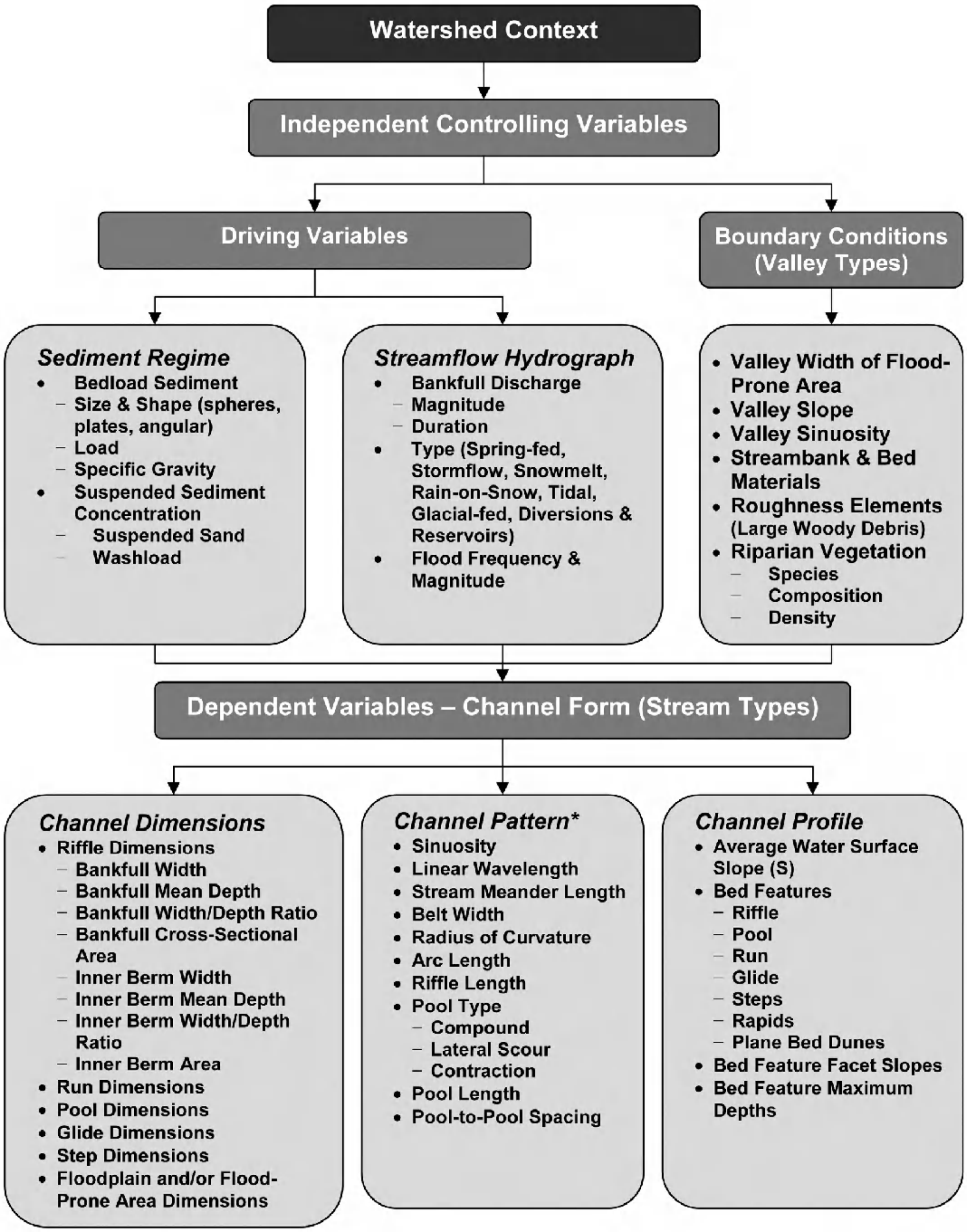

Figure 1. Independent and dependent variables that link the controlling variables and boundary conditions to the channel dimensions, pattern, and profile. *These channel pattern variables are representative of single-thread, meandering stream types; thus additional pattern recognition and description is required for bar-braided (D) and anastomosed (DA) stream types.

sinuosity, belt width, arc length, riffle and pool lengths, and pool-to-pool spacing (Figure 1). The channel profile includes slope measurements and an assortment of thalweg depths for the various bed features in addition to the depths measured at cross sections. Floodplain and/or flood-prone area dimensions and elevations are also measured.

The controlling variables for the existing and reference reaches are stratified (organized) by stream type and valley type with specific variables collected during the geomorphic characterization and assessment phases in the NCD methodology. Within each valley type is a unique characterization of flow regime, sediment regime, roughness elements, such as large woody debris, and riparian vegetation that influences the morphological character of the stream types contained in a valley. It is important to describe the flow regime (e.g., snowmelt, stormflow, spring fed, tidal influence, glacial fed, reservoir/diversion outflows, urban stormflow or rain-on-snow) to imply certain morphological conditions for a series of given channel form features. For example, spring-fed stream systems are associated with lower width/depth ratios due to flow resistance from dense riparian vegetation and low bed load sediment, compared to a snowmelt or rain-on-snow dominated flow regime. The sediment regime (size, type, and load or supply) that influences channel morphology is reflective of the depositional history of the valley type (e.g., terraced alluvial valley fills, glacial trough, lacustrine, alluvial fans, colluvial valleys, or deltas), including bar samples and stream bank and bed material inventories. The riparian vegetation type (overstory/understory, rooting character and ground cover type and density) also integrates the boundary conditions that influence the channel morphology.

Overall, an intimate relationship exists between process and form (Figures 1 and 2). Rivers having similar boundary conditions and driving variables of flow and sediment regime processes will have similar morphology, whereas any change in the controlling variables will alter channel morphology [*Schumm*, 2005]. Any sustainable solution in river restoration must properly replicate the form variables that represent the process integration of the independent, controlling variables with the dependent, form variables to maintain natural stability.

3.2. Applications of Form and Process Interrelations

The study of streams for any purpose involves form measurements of channel dimensions, pattern, profile, and materials. For any erosional, depositional, and equilibrium processes to be inferred, predicted, and validated, direct observations of river morphology are essential to obtain the stream's hydraulic, sedimentological, and biological character. From this information, the process interpretations are derived. For example, when a form variable changes due to imposed conditions, the corresponding hydraulic and sedimentological process relations are also influenced that result in "process changes" (e.g., aggradation, degradation, and lateral migration) and "channel consequences" (e.g., land loss, habitat changes, and shifts in stability) (Figure 2). An increase in the form variable of width/depth ratio from disturbance, for instance, without a change in the driving variable of bankfull discharge, results in an increase in flow resistance due to changes in relative roughness and friction factor because of reduced hydraulic mean depth. This results in a decrease of mean velocity and shear stress. The increase in width/depth ratio also creates a corresponding decrease in

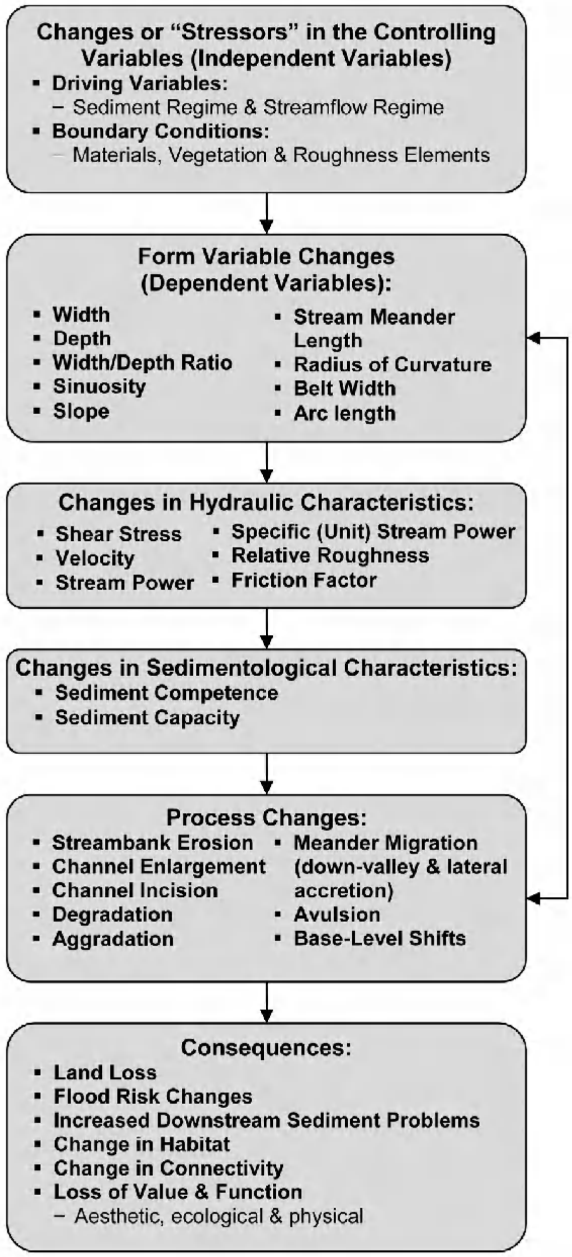

Figure 2. Linkage between form and process variable changes and the consequences due to changes in the controlling variables.

total stream power. Consequently, sediment transport competence and capacity are also decreased. Aggradation, accelerated stream bank erosion, chute cutoffs, and channel enlargement processes occur as a result of changes in the form variables. Additional form variables are subsequently adjusted including decreased sinuosity and increased slope. This form change, whether induced directly or indirectly, often results in a change in stream type from a meandering, riffle/pool, single-thread system to a multiple-thread, convergence/divergence bed-featured, bar/braided system. A change in both form and process can induce shifts in the geomorphic character of the river or a "threshold stream type" change.

3.3. Assumptions in Natural Channel Design

The primary assumptions in the NCD approach for river restoration are the following:

1. Form and process are interrelated.
2. Channel width is related to the bankfull discharge (normal high flow).
3. Assessments of river stability can be conducted to determine departure from a stable, reference condition.
4. Spatial and temporal changes of stream systems can be evaluated in watershed and river stability assessments through time trend studies and local validation using space for time substitution to select the appropriate stream succession scenarios and states.
5. Regional bankfull discharge and cross-sectional area can be determined from stream gauge sites and can be expressed as a function of drainage area within a hydrophysiographic area and can be extrapolated to ungauged sites within the same province; exceptions are associated with changes in streamflow and drainage area relations by diversions, reservoirs, and land use and must be determined from analysis and field studies from a watershed assessment rather than regional curves. The bankfull channel width and depth are not used for design from regional curves or hydraulic geometry unless such empirical relations are stratified by stream type and valley type.
6. A "reference reach" can be used to extrapolate dimensionless relations to determine the departure of the existing reach and for natural channel design. This assumption is based on the similarities in the boundary conditions and driving variables of the impaired existing reach and its potential stable stream type.
7. The dimensionless relations of the reference reach can be used to develop detailed dimensional values of dimension, pattern, and profile for the proposed design reach (e.g., bankfull maximum depths and facet slopes for riffles, runs, pools, glides, and steps).
8. Bar and bed samples and channel slope can be obtained to establish ratios to calculate critical dimensionless shear stress for the bankfull stage condition.
9. An entrainment relation using the Shields (or modified Shields [*Rosgen*, 2006b, 2007]) relation can be used to test for sediment competence for the existing, reference, and proposed design reaches.
10. Bankfull stage measurements of discharge, bed load, suspended sediment, and suspended sand sediment can be used to convert dimensionless relations of sediment rating curves to actual values of sediment rating curves (FLOWSED model [*Rosgen*, 2006a, 2006b, 2007]).
11. Regional bankfull bed load and suspended sediment curves can be established by major geology, stream stability, and drainage area in the interim absence of bankfull sediment data [*Rosgen*, 2006b, 2010].
12. Bankfull mean daily discharge can be obtained to develop dimensionless flow-duration curves at gauge stations. Mean daily bankfull discharge is then computed at ungauged sites and used to convert the dimensionless flow-duration curve to dimensional.
13. A sediment transport capacity model can be used to test for sediment continuity and channel stability for the existing, reference, and proposed design reaches.
14. Postrestoration stream adjustment of the dimension, pattern, and profile can appropriately occur within the range of natural variability of the reference reach data.

3.4. The Ten Phases of Natural Channel Design

Any river restoration design must first identify the multiple specific objectives, goals, and anticipated benefits of the proposed restoration. Analytical calculations, regionalized validated relationships, and analogy are combined in a precise series of computational sequences [*Rosgen*, 2007]. The conceptual layout for the 10 phases of the NCD approach is shown in Figure 3. The flowchart is indicative of the full extent and complexity associated with this approach. The NCD approach is divided into 10 major sequential phases (Figure 3) that act as a fundamental design framework and guide users through the minimum requirements and specific design procedures that must be incorporated: phase I, define restoration objectives; phase II, develop local and regional relations; phase III, conduct watershed, river, and biological assessments; phase IV, consider passive recommendations for restoration; phase V, develop conceptual design plan; phase VI, develop and evaluate the preliminary natural channel design; phase VII, design stabilization and enhancement structures; phase VIII, finalize natural channel design; phase IX, implement natural channel design; and phase X, conduct monitoring and maintenance

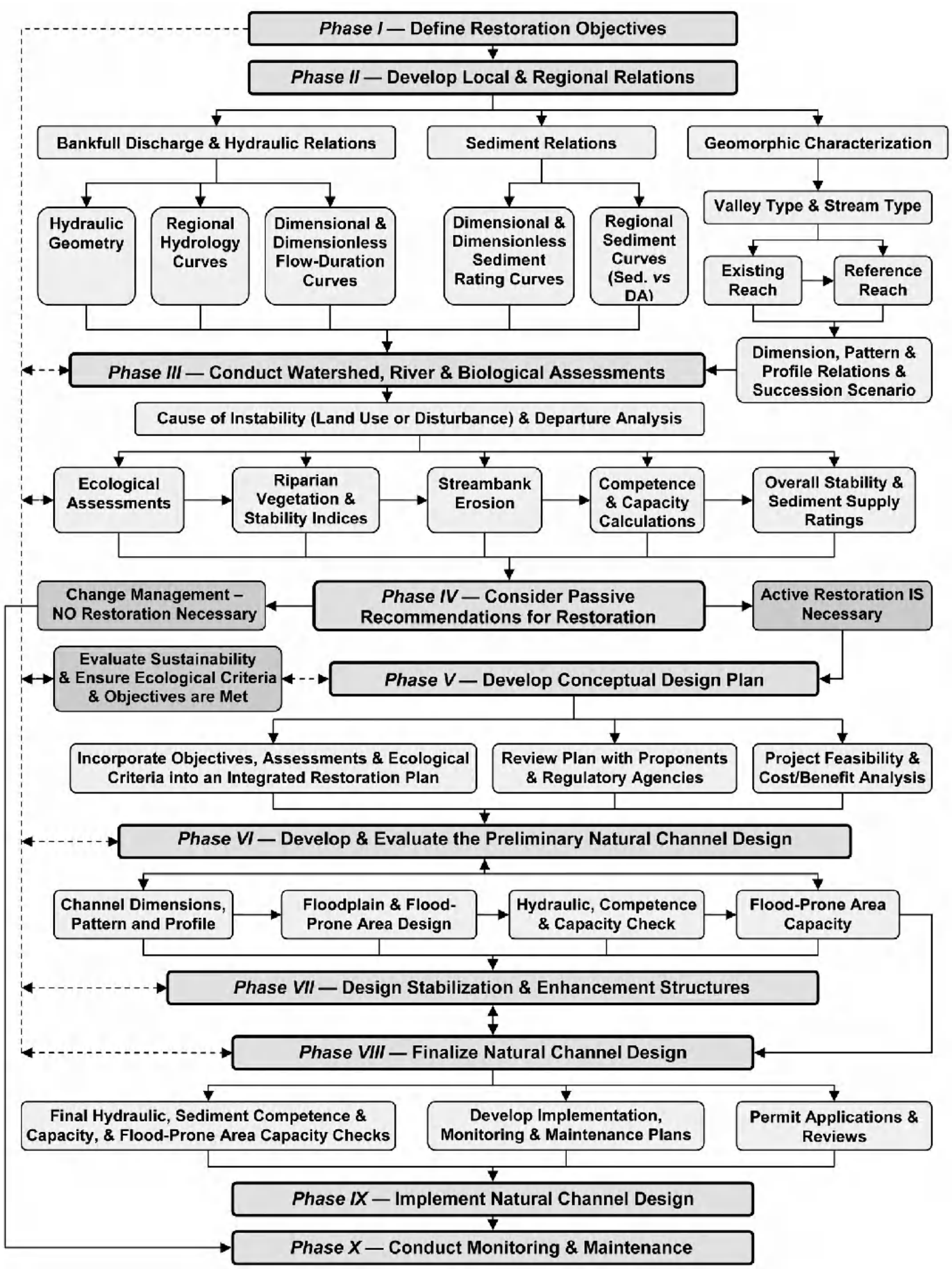

Figure 3. Ten phases in the natural channel design (NCD) approach to river restoration.

1. Phase I defines specific restoration objectives associated with physical, biological, and chemical processes. The restoration objectives must be stated clearly and concisely to appropriately design the solutions. It is essential to fully describe and understand all objectives, which also must be achievable and measureable. The goals or objectives of a river restoration design are often driven by an observed or perceived change over time resulting from impairment of uses and values. Common goals and objectives include enhancing water quality, managing riparian zones, improving in-stream habitat, allowing for fish passage, and stabilizing stream banks [*Bernhardt et al.*, 2005]. Creating terrestrial and off-channel aquatic habitats for mammals, birds, amphibians, and beaver; reducing flood levels, sediment supply, land loss, and attached nutrients; improving aesthetics (both visual and sound), recreational opportunities (e.g., trails, picnicking, camping, boating, fishing, and hunting), and wetlands; and allowing for self-maintenance and cost-effectiveness are also common objectives.

2. Phase II develops regional and localized specific information on the geomorphic characterization, sedimentology, hydrology, and hydraulics. Field data for the existing and reference reaches are collected and analyzed to define sedimentological, hydraulic, and morphological parameters in addition to biological and ecological studies, water quality data, and the riparian plant community. Phase II establishes the fundamental relations to determine the bankfull discharge and sediment supply (both bed load and suspended sediment) of the watershed and the reach in question.

Stream classification and valley type are also determined for the existing and reference reaches. The stable form and corresponding stream type from stream succession data must be determined for the existing reach to assist in selecting the correct reference reach to establish dimensionless relations of dimension, pattern, and profile data. Additionally, the recognition and matching of similar controlling variables and boundary conditions of the reference reach stream type and valley type with the impaired riparian ecosystem is crucial at this phase. Testing and evaluating the stability of the reference reach is conducted in phase III.

3. Phase III includes the watershed, river, and biological assessments to identify and understand causes of impairment and the nature, magnitude, direction, duration, and consequences of change. A cumulative watershed assessment is implemented utilizing the procedures given by *Rosgen* [2006b]. The relations among hillslope, hydrology, and channel processes are evaluated by location, land use, and erosional or depositional processes to help ascertain river impairment. The land use history and time trend analysis of river change are studied to provide insight into the cause of change. The morphological changes resulting in geomorphic thresholds that change stream types are documented.

The primary causes of instability or loss of physical and biological function must also be isolated and understood. Concurrent biological data (analysis of limiting factors) is obtained on a parallel track with the physical data. Ecological assessments compared to the potential state within the riparian ecosystem are necessary to establish criteria to integrate into the physical system for an appropriate assessment and design. Without such assessments and established criteria, the "vision" of ecological restoration objectives could be missed. The river and biological assessments are also conducted on the reference reach to ensure stability and to understand the physical and biological departure of the existing condition from the stable form.

4. Phase IV considers passive recommendations based on land use change in lieu of mechanical restoration. The causes of impairment should be understood from the assessment in phase III, and a passive restoration can be effective by influencing the drivers of the instability in a direction toward self-recovery. For example, riparian vegetation alterations of the boundary conditions can be reversed by better riparian management under high recovery potential. Changes in grazing strategy, land clearing, riparian management zone changes, flow regime changes from reservoirs or diversions, and changes in sediment budgets may be considered to initiate natural recovery of impaired rivers. If passive methods are reasonable to meet objectives, the procedure advances to the monitoring phase (phase X); otherwise, it is necessary to proceed with the subsequent phases in NCD for active restoration.

5. Phase V incorporates the objectives, assessments, and physical and ecological criteria into a resource-integrated conceptual natural channel design. The conceptual plan must address the multiple objectives and strive to meet the specific criteria identified in the assessments. True ecological restoration can only be accomplished if the conceptual design incorporates the limiting factors and critical criteria previously established in both phases I and III. The conceptual design provides a preliminary opportunity to properly integrate ecological criteria rather than "after the fact" add-ons. The conceptual design, however, must also be physically compatible with the fundamental central tendencies of the stable river form.

Project feasibility including physical and economic analyses are also conducted and discussed with the restoration sponsors in this phase. Following the sponsor review, the conceptual design is reviewed in the field with the regulatory agencies to share information, investigate various alternatives, and conduct an initial environmental evaluation. This provides the opportunity to include regulatory personnel at these early stages to investigate problem solving, resource enhancement, and how to direct mitigation to offset adverse effects.

6. Phase VI quantitatively develops and evaluates the preliminary natural channel design. The dimension, pattern, and profile variables of the proposed design reach are established and evaluated with subsequent analytical testing of hydraulic and sediment transport relations (competence and capacity). Also the floodplain and/or flood-prone area are designed and evaluated for flood discharge capacity along with the diversity, appropriateness, and compatibility of the proposed riparian habitats. The variability in natural rivers is incorporated into the design derived from the range of channel form features of the reference reach; this allows for an array of possible design solutions that incorporate multiple goals rather than a single, uniform design.

The multiple objectives are also reviewed and evaluated again in this phase for compatibility of both physical and ecological criteria. Water rights issues, diversions, habitat diversity (such as side channels, oxbow systems, rearing habitats, and wetlands), riparian plant assemblages (planned understory, midstory, and overstory composition and density), and specific aquatic and terrestrial habitats are tested against desired outcomes within existing or perceived physical, economic, and sociological constraints. Review of the preliminary design by professionals representing multiple disciplines and the restoration sponsors will help formulate and modify a potentially feasible, compatible, and sustainable design.

7. Phase VII incorporates stabilization and enhancement structures. River structures are designed to meet specific project requirements, such as energy dissipation, grade control, and lateral stability to buy time to establish the riparian plant community. A diversity of structures is required for fish habitat enhancement, recreational boating features, irrigation diversion structures, and specific habitat features. Common materials used in NCD structures include logs, root wads, woody debris, native boulders, and riparian vegetation, such as vegetation transplants and sod mats.

8. Phase VIII revises any preliminary design specifications following detailed computations (including final hydraulic, sediment competence and capacity, and flood-prone area capacity checks) and reviews by the planning team, sponsors, and regulatory agencies to finalize the natural channel design. Implementation, monitoring, and maintenance plans are also developed in this phase along with reviewing and incorporating regional requirements and submitting the necessary permit applications. Submitted plans for final review and approval should include the results of the previous phases including the watershed and ecological assessment tasks.

9. Phase IX is the implementation phase. The proposed design and stabilization measures are described and constructed. These measures involve contracting criteria, design layout, water quality control, field supervision, field methods, appropriate equipment recommendations, and construction staging.

10. Phase X is the final phase incorporating monitoring and maintenance. Implementation, validation, and effectiveness monitoring are required to evaluate project success. "Implementation monitoring" documents how well the design is actually constructed. "As-built" monitoring is often required to help ensure proper implementation and provide timely corrections for deficiencies identified during daily construction inspections. "Validation monitoring" evaluates the predicted versus observed system response related to river stability (e.g., lateral and vertical stability, channel enlargement, and lateral migration or bank erosion rates) where the prediction models are compared to observed response. "Effectiveness monitoring" evaluates the nature and extent of restoration response to meet stated objectives. The physical, biological, and chemical responses of the restoration, including terrestrial and aquatic habitat responses, are evaluated. Success criteria are documented to test and compare with postrestoration data. The acceptable post runoff departure from the "as built" data is based on the natural variability of the same parameters from the reference reach relations reflected in the ranges utilized in the design.

A maintenance plan is also implemented with established criteria that document when the nature and extent of change requires maintenance; reentry following restoration is recommended only if the morphological variables depart from the natural variability of the reference reach (stable river) used for design.

3.5. The Stream Classification System

An integral part of the NCD methodology involves the use of a stream classification system, which serves as the foundation of the assessment and design procedures. Due to the great variability in the fluvial landscape, various valley and stream types occur and represent a diverse range of morphologies. Their character and behavior is the result of past and present changes in the watershed: some are geologic or natural and some anthropogenic. Not all stream systems respond similarly to imposed change nor offer consistent interpretations. As a result, it becomes imperative that the various fluvial forms that represent river and valley types are described.

Because of the great diversity of morphological features among rivers, a stream classification system was developed to stratify and describe various river types [*Rosgen*, 1994, 1996]. The nature and range of the dependent form variables of river channels were delineated to help describe the variety of morphological stream types that do occur in nature. These types were not determined arbitrarily but rather were organized by measured data representing hundreds of rivers between 1969 and 1994 [*Rosgen*, 1994, 1996]. Resultant stream types are a reflection of mutually adjusting variables that describe their unique sedimentological, hydraulic, morphological, and biological characteristics. "The classification is based on parameters of form and pattern but has the advantage of implying channel behavior [*Leopold*, 1994, p. 20]."

Stream classification is based primarily on the measured bankfull stage morphology of the river because it is the bankfull stage that is responsible for shaping and maintaining the channel dimensions over time. Channel widths and other dimensions of alluvial river systems are more consistent with the more frequent, but lower magnitude (bankfull) discharge [*Wolman and Miller*, 1960; *Leopold*, 1994; *Rosgen*, 1994]. The bankfull discharge is also responsible for the long-term cumulative sediment transport, which also influences the channel boundary [*Wolman and Miller*, 1960; *Dunne and Leopold*, 1978]. However, rather than using the measured values of dimension, pattern, and profile to define a stream type, the classification system is based on dimensionless morphological parameters required for scaling purposes (Table 1). Study streams are seldom located immediately upstream or downstream of reference stream types; thus, scaling of the morphological relations is necessary.

Specific objectives of the stream classification system [*Rosgen*, 1994, 1996, 2003] are to (1) predict a river's behavior from its morphological appearance based on documentation of similar response from similar types for imposed conditions; (2) stratify empirical hydraulic, sedimentological, and biological relations by stream type by state (condition) to minimize variance; (3) provide a mechanism to extrapolate site-specific morphological data; (4) describe physical stream relations to complement biological and riparian ecosystem inventories and assist in establishing potential and departure states; and (5) provide a consistent, reproducible frame of reference for communicating stream morphology and condition among a variety of professional disciplines.

The stream classification system consists of a hierarchical assessment of channel morphology that includes four levels of assessment [*Rosgen*, 1994, 1996]. The four levels provide the physical, hydrologic, sedimentological, and geomorphic context for linking the driving forces and response variables at all scales of inquiry. The detail required at each level of assessment varies with the degree of resolution necessary to achieve the specific objectives previously stated.

Level I of the hierarchical assessment is the geomorphic characterization where streams are classified at a broad level on the basis of valley landforms and observable channel dimensions. Eight major morphological stream types can be identified (A, B, C, D, DA, E, F, and G) using five initial definitive criteria: channel pattern (multiple-thread versus single-thread channels), entrenchment ratio, width/depth ratio, sinuosity, and slope (Table 1) [*Rosgen*, 1994, 1996]. "Entrenchment ratio" is a measure of vertical containment described as the ratio of the flood-prone area width to bankfull width. The flood-prone area width is obtained at an elevation at two times the maximum bankfull depth. If the entrenchment ratio is less than 1.4 (±0.2 to allow for the continuum of channel form), the stream is classified as entrenched or vertically contained (A, G, and F stream types) (Table 1). If the entrenchment ratio is between 1.4 and 2.2, (+ or −0.2), the stream is moderately entrenched (B stream types). If the ratio is greater than 2.2, the stream is not entrenched (C, E, and DA stream types). Additionally, some stream types are associated with valley types that have well-developed floodplains (C, D, E, and DA stream types), while other stream types are associated with valley types with no floodplains (A, B, certain D, G, and F stream types). Table 1 describes the additional criteria (channel pattern, width/depth ratio, sinuosity, and slope) for each major stream type.

Because stream morphology is invariably fixed to the landscape position, prior to the broad-level stream classification, level I also identifies valley types that integrate structural controls, fluvial process, depositional history, climate, and broad life zones. Valley types are stratified into 11 broad geologic categories that reflect their origin and represent the independent boundary conditions that influence channel morphology [*Rosgen*, 1994, 1996]. Table 2 summarizes the valley types and their associated characteristics, separated by historic erosional or depositional processes, and corresponding differences in valley slope, channel materials, and width. Valley types and related landforms are the initial stratification of stream types (Table 2). For example, highly dissected fluvial slopes (valley type VII) are indicative of steep, narrow, deeply incised, erosional A and G stream types. Narrow, low-gradient streams in confined canyons and deep gorges (valley type IV) are characteristic of the entrenched F stream types.

In addition to valley types, stream types must also be stratified by the driving process variables of flow and sediment regime to help minimize the variance of the integrated form variables. For example, stable C4 stream types (gravel-dominated C type) in terraced alluvial fill valleys (valley type VIII) with river widths between 3 and 15 m characteristically average width/depth ratios between 12 and 14. However, the width/depth ratios average between 18 and 24 for C4 stream types in U-shaped, glacial trough valleys (valley type V). The width/depth ratios for the C4 stream type in valley type V are larger because of higher ratios of bed load to total sediment

Table 1. General Stream Type Descriptions and Definitive Criteria for Broad-Level Classification[a]

Stream Type	General Description	Entrenchment Ratio	*W*/*d* Ratio	Sinuosity	Slope	Landform/Soils/Features
Aa+	Very steep, deeply entrenched, debris transport, torrent streams.	<1.4	<12	1.0 to 1.1	>0.10	Very high relief. Erosional, bedrock, or depositional features; debris flow potential. Deeply entrenched streams. Vertical steps with deep scour pools; waterfalls.
A	Steep, entrenched, cascading, step/pool streams. High energy/debris transport associated with depositional soils. Very stable if bedrock- or boulder-dominated channel.	<1.4	<12	1.0 to 1.2	0.04 to 0.10	High relief. Erosional or depositional and bedrock forms. Entrenched and confined streams with cascading reaches. Frequently spaced, deep pools in associated step/pool bed morphology.
B	Moderately entrenched, moderate gradient, riffle-dominated channel, with infrequently spaced pools. Very stable plan and profile. Stable banks.	1.4 to 2.2	>12	>1.2	0.02 to 0.039	Moderate relief, colluvial deposition and/or structural. Moderate entrenchment and width/depth ratio. Narrow, gently sloping valleys. Rapids predominate with scour pools.
C	Low gradient, meandering, point bar, riffle/pool, alluvial channels with broad, well-defined floodplains.	>2.2	>12	>1.2	<0.02	Broad valleys with terraces in association with floodplains, alluvial soils. Slightly entrenched with well-defined meandering channels. Riffle/pool bed morphology.
D	Braided channel with longitudinal and transverse bars. Very wide channel with eroding banks.	NA	>40	NA	<0.04	Broad valleys with alluvium, steeper fans. Glacial debris and depositional features. Active lateral adjustment with abundance of sediment supply. Convergence/divergence of bed features, aggradational processes, high bed load and bank erosion.
DA	Anastomosing (multiple channels) narrow and deep with extensive, well-vegetated floodplains and associated wetlands. Very gentle relief with highly variable sinuosities and width/depth ratios. Very stable stream banks.	>2.2	highly variable	highly variable	<0.005	Broad, low-gradient valleys with fine alluvium and/or lacustrine soils. Anastomosed (multiple channel) geologic control creating fine deposition with well-vegetated bars that are laterally stable with broad wetland floodplains. Very low bed load, high wash load sediment.
E	Low gradient, meandering riffle/pool stream with low width/depth ratio and little deposition. Very efficient and stable. High meander width ratio.	>2.2	<12	>1.5	<0.02	Broad valley/meadows. Alluvial materials with floodplains. Highly sinuous with stable, well-vegetated banks. Riffle/pool morphology with very low width/depth ratios.
F	Entrenched meandering riffle/pool channel on low gradients with high width/depth ratio.	<1.4	>12	>1.2	<0.02	Entrenched in highly weathered material. Gentle gradients with a high width/depth ratio. Meandering, laterally unstable with high bank erosion rates. Riffle/pool morphology.
G	Entrenched "gully" step/pool and low width/depth ratio on moderate gradients.	<1.4	<12	>1.2	<0.039	Gullies, step/pool morphology with moderate slopes and low width/depth ratio. Narrow valleys or deeply incised in alluvial or colluvial materials, i.e., fans or deltas. Unstable, with grade control problems and high bank erosion rates.

[a]See *Rosgen* [1994, 1996, 2006b] for more information. From *Rosgen* [2006b].

Table 2. Valley Types Used in the Geomorphic Characterization and Their Associated Stream Types[a]

Valley Types	Summary Description of Valley Types	Stream Types
I	Steep, confined, V-notched canyons, rejuvenated side slopes	Aa+, A, G
II	Moderately steep, gentle-sloping side slopes often in colluvial valleys	B, G
III	Alluvial fans and debris cones	A, B, F, G, D
IV	Canyons, gorges, and confined alluvial and bedrock-controlled valleys with gentle valley slopes	C, F
V	Moderately steep, U-shaped glacial-trough valleys	C, D, F, G
VI	Moderately steep, fault-, joint-, or bedrock-controlled valleys	Aa+, A, B, C, F, G
VII	Steep, fluvial dissected, high-drainage density alluvial slopes	Aa+, A, G
VIII	Alluvial valley fills either narrow or wide with moderate to gentle valley slope with well-developed floodplain adjacent to river, and river terraces, glacial terraces, or colluvial slopes adjacent to the alluvial valley	C, D, E, F, G
IX	Broad, moderate to gentle slopes associated with glacial outwash or Eolian sand dunes	C, D, F
X	Very broad and gentle valley slopes associated with glacio- and nonglaciolacustrine deposits	C, DA, D, E, F, G
XI	Deltas	C, D, DA, E

[a]See *Rosgen* [1996, 2006b] for more information. From *Rosgen* [2006b].

load, steeper valley slopes than the valley type VIII, higher sediment supply, and unconsolidated, noncohesive bank material. Pattern and profile variables also differ, such as sinuosity (greater in valley type VIII) and radius of curvature (larger in valley type V). Regardless of valley type, these are still C stream types with meanders, riffle/pool bed features on slopes less than 0.02 with floodplain connectivity. When developing "reference reach" relations, it is essential to stratify stream types by valley type and the corresponding flow and sediment regimes [*Rosgen*, 1998, 2006b, 2007].

Level II is the morphological description that classifies stream types within certain valley types using field measurements of the same criteria necessary for the broad-level classification from specific channel reaches and fluvial features [*Rosgen*, 1994, 1996]. In addition, the initial stream type is further subdivided by its dominant channel material size: 1, bedrock; 2, boulder; 3, cobble; 4, gravel; 5, sand; and 6, silt/clay. In total, 41 primary stream types exist. Subcategories of slope are also utilized along a slope continuum where the combined morphological variables are consistent for a stream type. However, for a particular stream reach that is steeper or flatter than the normal range of that type, a small letter subcategory is used to best reflect actual variables [*Rosgen*, 1994, p. 181]: a+ (steeper than 0.10), a (0.04–0.10; slopes typical of A stream types), b (0.02–0.04; slopes typical of B stream types), c (0.001–0.02; slopes typical of C stream types), and c− (less than 0.001).

The various categories and threshold ranges were obtained from field data representing over 800 rivers using frequency distributions from each major stream type grouping to establish the interrelations of morphological data. The parameter ranges are described by the frequency distributions summarized by *Rosgen* [1996, chapter 5]. In addition, Rosgen also describes the process-integration and interrelated morphologic, hydraulic, and sedimentological characteristics of each primary stream type.

Due to the continuum of channel form and shifts in stream types along river reaches, the definitive criteria values can depart from the typical ranges for a given stream type. These instances are indicative of (1) a transition between stream types and valley types that occurs when changing from an upstream reach into a downstream reach (spatial variability), (2) a shift in stability or condition influenced by variables described in level III (temporal variability), and/or (3) an equilibrium threshold shift trending toward a new stream type (temporal and spatial variability). In these instances, the variables that best represent the dominant morphological type must be determined.

Level III assesses stream condition to predict river stability (e.g., aggradation, degradation, sediment supply, stream bank erosion, and channel enlargement). The stream classification system was developed with an understanding that a stability evaluation must be conducted at a higher degree of resolution (level III assessment) than morphological groupings (level II). Channel stability assessments, however, must be stratified by stream type and valley type for extrapolation purposes. Additional form variables are identified by stream type and their definitive criteria to determine a state or condition. Various processes and stream channel response to imposed changes in the controlling variables can then be inferred using time trend aerial photo analysis and detailed field measurements [*Rosgen*, 1994, 1996, 2006b]. Variables assessed and introduced in this level include bank-height ratio (a measure of degree of channel incision determined as the lowest bank height divided by the bankfull maximum depth), meander width ratio (lateral containment or confinement measured by channel belt width divided by bankfull width), shear stress, shear velocity, and total stream power. Prediction of stream bank erosion (BANCS model

[*Rosgen*, 1996, 2001a, 2006b]), hydraulic analysis [*Rosgen*, 1996, 2006b], sediment competence and transport capacity [*Rosgen*, 2006a, 2006b], and quantitative indices for river stability are also collected at this level [*Rosgen*, 1996, 2001b, 2006b].

Critical, but often difficult, in the stability assessments and interpretations is an understanding of what constitutes a natural process versus an acceleration of a natural process as streams can be stable, yet dynamic. It is essential to distinguish if the methods used in the river stability assessment predict the differences between natural, stable rates versus accelerated rates that may exceed a geomorphic threshold. The assessment phase in NCD requires a departure analysis of the existing reach from the reference reach condition to assist with these interpretations. Without such stability assessments for the reference and existing reaches, it is often difficult to understand the cause and consequence of change related to certain land uses that are the agents of disequilibrium.

Level IV is conducted to validate process-based assessments of stream condition, potential, and stability as predicted from levels I–III. Prediction of river system process is complex and uncertain; thus, validation of the procedure is essential, since restoration designs are based upon such predictions. Validation procedures include annual dimension, pattern, profile, and material resurveys; annual stream bank erosion studies; sediment competence validation; hydraulic relations using gauging stations or current meter measurements; and direct measurements of bed load and suspended sediment for the accurate estimate of sediment transport capacity. After reach conditions are verified, the validation data are used to establish empirical relationships for testing, validating, and improving the prediction methods. In fact, the basic foundation of the stream classification system was developed from the author's level IV field data collected over many years that were used to develop the prediction methodologies and for the interpretation and extrapolation of the basic relations. The field data involve sediment transport, stream bank erosion rates, hydraulics, and corresponding changes in the channel form variables, all of which are time-consuming and expensive to collect. It is necessary to validate the procedures for both the existing and reference reaches. In this manner, it is possible to measure natural stream bank erosion rates and to obtain a wide range of natural variability of the dimensions, pattern, and profile to determine acceptable rates and tolerances.

Levels III and IV of the stream classification system are often overlooked in the published literature when discussing how stream classification can be used to infer process and how it applies to river restoration [e.g., *Miller and Ritter*, 1996; *Simon et al.*, 2007; *Juracek and Fitzpatrick*, 2003]. The importance of conducting a watershed and river stability assessment should not be underestimated. Level III is performed specifically to assess the processes occurring in river systems, and the process predictions are followed by validation procedures in level IV. The following stream succession scenarios are used as part of the level III analysis to infer channel succession over time and space using historical evidence and current geomorphic conditions to predict future response.

3.6. Stream Channel Succession

Predicting a river's behavioral response to geologic and anthropogenic disturbances is necessary for those working with river systems. The observations of the past and an understanding of form and process interactions create the basis to predict future channel response and erosional or depositional processes associated with similar impacts. It is paramount to first look back in time using time trend aerial photographs, historic records, dendrochronology, paleochannel analysis, carbon dating, and other methods to understand channel change over time and space. Parallel with such analysis is an understanding of the change in the controlling process variables that influence river morphology.

Rivers do not always change instantaneously under a geomorphic exceedance or "threshold." Rather, they undergo a series of channel adjustments over time to accommodate change in the driving variables. Their dimensions, pattern, and profile reflect on these adjustment processes that are presently responsible for the form of the river. The nature, rate, and direction of channel adjustments are unique to the stream type involved. Some streams change very rapidly, while others are slow in their response [*Rosgen*, 1994, 1996].

Understanding the central tendency and the characteristics of the stable form and the processes of river adjustment that shape the landscapes and river systems over time lends the observer an insight into the processes of the past. These processes can then be projected to interpret future conditions under similar boundary conditions or driving variables. Furthermore, landforms and rivers equilibriate with different endpoint features of their morphology due to the variation in the erosional or depositional processes under a wide spectrum and great variation of the independent variables. Due to changes in the driving variables and boundary conditions, not every stream returns to its original or predisturbance form.

Stream succession is a central element to predict a river's behavior from its morphological characteristics, which are directly related to the stream type's corresponding hydraulic and sedimentological relations. Stream channel succession is the result of adverse consequences of excess sediment supply;

accelerated bank erosion rates; degradation, aggradation, and channel enlargement from channel disturbance; streamflow changes; and/or sediment budget changes that lead to channel change. These changes result in stability shifts and adjustments leading to channel morphological changes and eventual stream type changes over time. Classification of stream type [*Rosgen*, 1994, 1996] is used to establish the links between channel process, form, and stability [*Thorne*, 1997]. It is essential that the field observer ascertain the cause, direction, and trend of river change as well as the stable equilibrium form in NCD.

Twelve various scenarios are illustrated in Figure 4 representing successional scenarios of stream type shifts, each representing various sequences from actual rivers. These scenarios represent morphological shifts and their tendencies toward stable endpoints (additional scenarios are possible). Each stage of the individual scenarios is associated with unique relations of morphological, hydrological, sedimentological, and biological functions. Adverse adjustments due to disequilibrium can result in accelerated sediment yields, loss of land, lowering of the water table, decreased land productivity, loss of aquatic habitat, and diminished recreational and visual values.

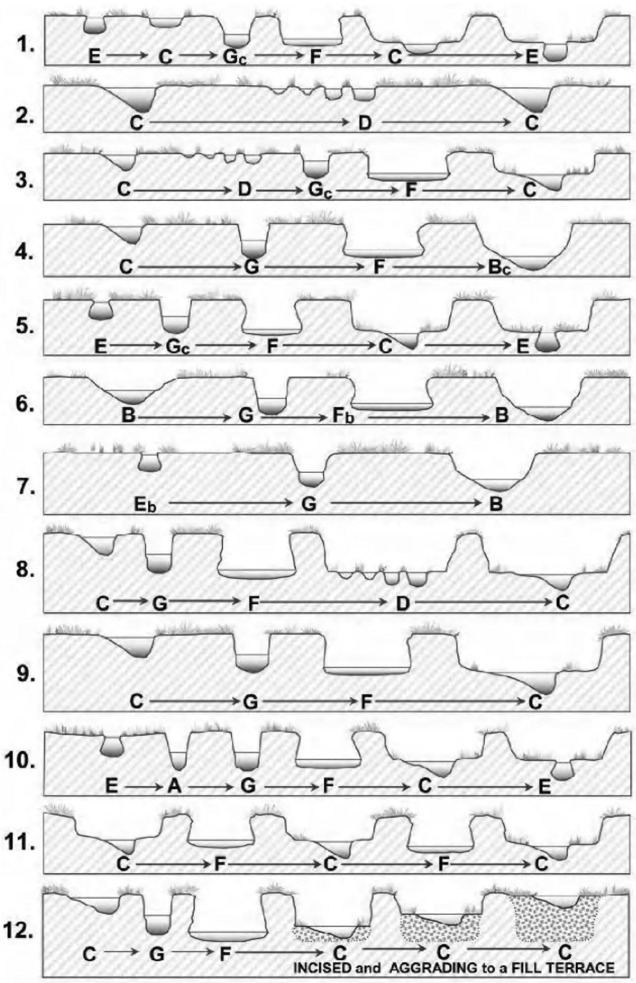

Figure 4. Various stream succession scenarios of stream type shifts over time (for letter codes see Table 1). Note that these various scenarios represent actual rivers (i.e., they are not hypothetical) and do not represent the only possible scenarios. Adapted from *Rosgen* [1999, 2001b, 2006b, 2007].

The "existing reach" in NCD is often associated with a stream type that is not stable or is in disequilibrium. Referring to Figure 4, these stream types represent the intermediate or transitional stages of each succession scenario. The following must be determined for the existing reach: (1) the appropriate morphological scenario (scenarios 1–12 in Figure 4), (2) within a scenario, the current successional stage of the existing stream type, (3) the various stages leading up to a succession endpoint, (4) the series of natural changes that occur prior to reaching stability, and (5) the potential stable form of the channel type. Selecting the appropriate stream succession scenario and sequence is aided by time trend aerial photography, dendrochronology, paleochannel evaluation, and other historical evidence. The potential stream type of the existing reach is an important criterion necessary to select the appropriate reference reach.

Restoration direction is aided by understanding the present successional stage within a specific scenario and the starting and endpoints. In some cases, restoration involves returning the stream to its predisturbance state on previously abandoned surfaces (priority IV [*Rosgen*, 1997]). Knowing the direction and rate of change and recovery potential also assists to prescribe management changes for potential passive restoration recommendations. Boundary condition changes from predisturbance, such as channel confinement (lateral containment), for example, promote stream types with low meander width ratios (stream belt width divided by bankfull width) typical of Bc stream types [*Rosgen*, 1996].

3.7. The Reference Reach and Proposed Design Reach

The reference reach selection is a critical step in NCD. The reference reach must be stratified based on identified geomorphic characteristics, boundary conditions, and driving variables of the existing and proposed design reaches (Figure 5). A reference reach is required for each identified existing reach that has a different valley type or potential stream type. As stated previously, a geomorphic characterization is then completed for the reference reach followed by an assessment to ensure stability and to determine the departure of the existing stream stability from the reference reach

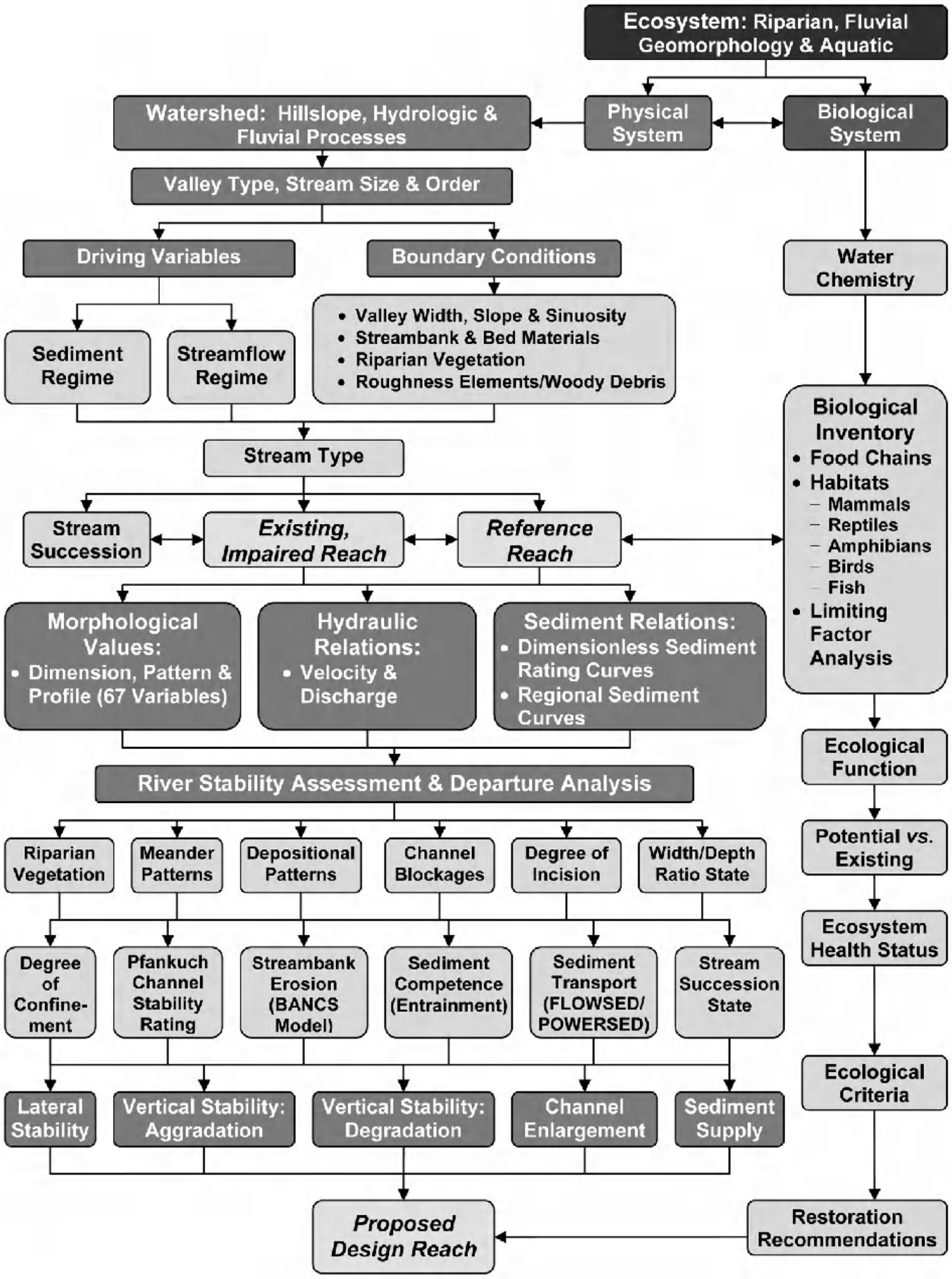

Figure 5. Watershed variables integrated into the development of physical and biological relations in NCD.

condition. Table 3 lists the major criteria to select a reference reach that must match or be similar to the proposed design reach. This table also identifies the range of variability and scaling criteria for extrapolation purposes. Table 4 is a priority list of reference reach selection scenarios in relation to the proposed design reach.

The "proposed design reach" enters the NCD methodology after the existing and reference reaches have been identified, the geomorphic characterization conducted, and the watershed, river, and biological assessments are completed (phases II and III) (Figure 5). If passive recommendations (phase IV) are insufficient to address the cause of impairment and active restoration is necessary, a conceptual channel design is developed (phase V) to emulate a natural stable channel for the proposed design reach followed by the preliminary natural channel design (phase VI) with the proposed

Table 3. Reference Reach Selection Criteria

Reference Reach Selection Criteria	Relation to Proposed Design Reach
Valley type	same
Stream type	same
Scaling (bankfull width)	within one order of magnitude for bankfull widths less than 50 ft within one-half order of magnitude for bankfull widths greater than 50 ft
Stream order	within one stream order
Boundary conditions	similar
Valley slope	
Valley sinuosity	
Valley width of flood-prone area	
Stream bank and bed material	
Riparian vegetation	
Driving variables	similar
Sediment regime and sediment sizes	
Flow regime	

dimension, pattern, profile, and floodplain/flood-prone area relations. A proposed design reach is required for each existing reach identified. The procedures must be completed for each proposed design reach utilizing the appropriate reference reach data. The required restoration variables for the existing, reference, and proposed design reaches are organized and recorded in an extensive multipage master table [*Rosgen*, 2007].

While designing the physical variables of the proposed design reach, the concurrent integration of the physical and biological components is necessary to help meet the design objectives and work toward sustainability. Figure 5 illustrates the integration of the biological and ecological objectives and functions into the natural channel design, which is not solely limited to stream channels. Floodplains, terraces, riparian community types, wetlands, oxbow channels, and off-channel ponds are all part of river systems and are important to restoring the physical, chemical, and biological functions. Ecology includes the organism and its associated habitats; thus, physical alterations of river systems are essential habitat components for various species, age classes, and functions. Changes to habitats should be designed with an understanding of the benefits from specific criteria that create the needed conditions to offset the limiting factors. Overall, the ecosystem complexity and diversity must satisfy site- and community-specific objectives involving the interactions between animal and plant communities for mammals, reptiles, amphibians, birds, and fish. To accomplish these ecological objectives, a multidisciplinary team is required to provide: (1) specific objectives; (2) an assessment of the existing conditions, including limiting factors for specific animal communities, age classes, life stages, and food chains, in relation to their habitats; (3) guidance criteria to the restoration effort; (4) an integration and assessment of conflict resolution due to potential conflicting and competing uses and objectives; (5) evaluation and monitoring criteria; (6) advice on project implementation and critical seasons to reduce conflict with existing and proposed habitats; and (7) reasonable alternatives to accommodate multiple plant and animal communities.

Table 4. Priorities of Reference Reach Locations in Relation to the Proposed Design Reach[a]

Priority	Reference Reach Locations in Relation to Proposed Design Reach
First	immediately upstream (carbon copy)
Second	immediately downstream (carbon copy)
Third	same stream but not immediately upstream or downstream (scale variation)
Fourth	within the same watershed
Fifth	outside of watershed and similar in size and scale
Sixth	outside of watershed and much smaller or larger in size and scale[b]

[a]Assuming similar valley type, stream type, boundary conditions, and driving variables.

[b]Must be tested against a smaller or larger reference condition to determine variability of dimensionless relations.

Ecological restoration is currently seen as a top priority for society and as a good investment [*Aronson et al.*, 2010; *Rey Benayas et al.*, 2009]. However, criteria for ecological restoration are noticeably absent in the published literature and in practice and must be established. Currently, site-, species- and habitat-specific criteria must be developed for each project.

Figure 5 is the culmination of the physical and biological assessments that help identify specific reaches and proposed actions based on the ecological and physical limitations.

4. THE NATURAL CHANNEL DESIGN APPROACH

4.1. NCD Prediction Methodologies

NCD incorporates analog, empirical, and analytical methods for assessment and design (Figure 6) to predict the channel morphology for natural river systems [*Rosgen*, 2007]. There are 67 form variables representing the dimension, pattern, and profile of natural, stable channels required for NCD prediction and implementation. The current analytical, numeric, rational, and empirical models used in non-

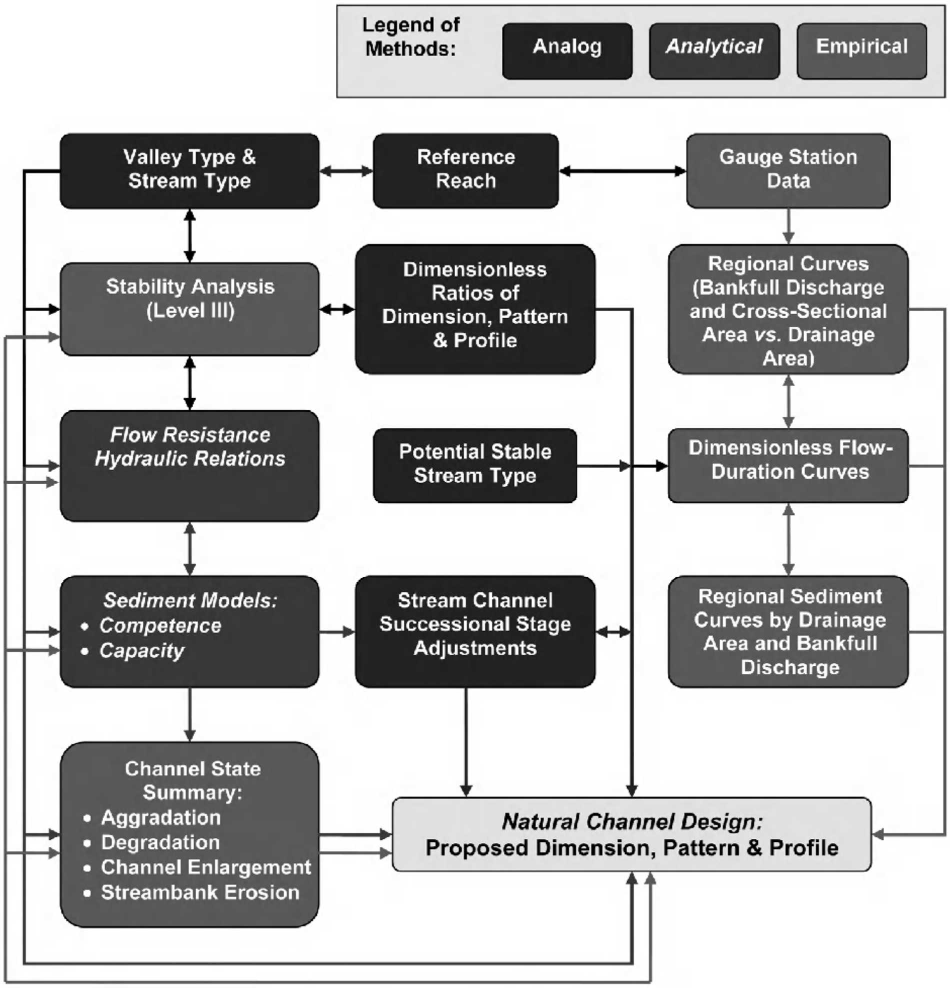

Figure 6. Generalized NCD flowchart utilizing analog, analytical, and empirical approaches [*Rosgen*, 2007].

NCD approaches to river restoration cannot provide this required output. For example, there are no known analytical or process-based models that predict the depth and slope of runs and glides, point bar slopes, meander geometry, and other features of riffle/pool, meandering stream types. To design and construct such features, form-based calculations using analog methods from reference reach data by stream type and valley type, and integrated by the driving variables and boundary conditions, have proven to be an appropriate method to consistently provide the morphology of the restored river [*Hey*, 2006; *Kondolf and Downs*, 1996].

Soar and Thorne [2001, p. 112] discuss that the analog approach "is preferred over more analytical methods based on the application of sediment transport equations which often yield significant errors in estimates of the design discharge and supply load that could affect the design specification." As so many unknown variables are involved to describe the channel configuration, "the river is the best model of itself" [*Shields*, 1996, p. 26] and "is ultimately the best channel restoration designer" [*Soar and Thorne*, 2001, p. 49]. Reference reaches can also serve to estimate attainable conditions, to evaluate temporal and spatial changes in ecological integrity, to classify attainable uses of streams, and to set biological and environmental criteria [*Hughes et al.*, 1986].

The empirical approach in NCD uses equations associated with various similar basins and channel boundary characteristics derived from regionalized or universal data. Empirical relations are used in the hydraulic and sedimentological evaluations for the existing, reference, and proposed design reaches [*Rosgen*, 2007]. Empirical relations for relative roughness and friction factor relations are used for velocity prediction [*Rosgen*, 2006b, 2007]. Tractive force relations including dimensional and dimensionless shear stress relations for particle entrainment and sediment competence calculations are used as well as dimensionless sediment rating curves for both suspended sediment and bed load [*Rosgen*, 1998, 2006b, 2007]. Empirical relations are also developed for regional bankfull discharge and cross-sectional area versus drainage area by hydrophysiographic provinces [*Rosgen*, 2006b, 2007]; these values are validated using the velocity calculations and requirements. Regional bankfull suspended and bed load sediment relations by dominant geologic type and river stability versus drainage area or bankfull discharge are also useful [*Rosgen*, 2010].

The analytical approach makes use of hydraulic and sediment transport models to derive relations for the existing and proposed stability conditions. The POWERSED model utilizes flow resistance, unit stream power, and sediment transport relations by flow stage to simulate sediment transport capacity computations for various dimension, pattern, and profile relations [*Rosgen*, 2006a, 2006b, 2007]. This model is run on the existing, reference, and proposed design reaches. The FLOWSED and POWERSED models are programmed and available in the RIVERMorph™ software program. Validation and applications of these models in restoration and engineering are described by *Rosgen* [2006a, 2010] and *Athanasakes and Rosgen* [2010].

4.2. The Multistage Channel Design for Specified Streamflows

NCD incorporates a multistage channel design as displayed in natural rivers to accommodate a wide range of streamflows, including base flow and bankfull discharge, and the floods are designed at a stage above the stream channel in floodplains and flood-prone areas to accommodate the frequent and the infrequent or rare floods. Rather than overwiden the active channel to accommodate flood flows, NCD generally designs toward the minimum width/depth ratio values of the active bankfull channel. However, the floodplain and flood-prone area features are commonly overwidened to accommodate the large floods. Setback terraces outside of the floodplain can be used to protect certain critical areas from flooding while providing river system function. Such stream restoration involving interconnection of stream channels and floodplains add to ecological function and species richness [*Paillex et al.*, 2009].

The multistage channel provides the alternative of design complexity under a changing flow regime, typical of expanding urban development, operational hydrology of reservoirs and diversions, and climate change. The multistage channel also allows for the greatest diversity and complexity of both aquatic and terrestrial habitats and appropriate riparian systems. Extreme flows of both floods and droughts are common and are best accommodated in the multiple-stage scenarios. The wide range of streamflows can be accommodated in four stages (Figure 7) (most common in C stream types (Table 1) in a terraced, alluvial valley type VIII (Table 2)): stage 1, the low-flow or "inner-berm" channel; stage 2, the bankfull stage channel; stage 3, the active floodplain at the incipient point of flooding; and stage 4, the infrequent but highest flood-level stage.

The multistage channel allows for a range of shifts in flows but an option of placing these flows on various levels. This design concept, which is found in natural reference reach systems, is superior to the overwidened, trapezoidal-shaped channel prevalent in many traditional river designs. The advantages of the four-stage channel, as compared to the "one-size-fits-all flows" channel, include the following:

1. Vegetation is established on the banks of stages 2, 3, and 4 (Figure 7) due to favorable soil moisture.

2. Stream bank erosion rates are decreased, and rooting depth and density are increased due to lower bank heights and favorable riparian vegetation conditions at the various benches and flats.

3. Stream bank erosion is also reduced due to reductions in near-bank stress as the flows onto the next highest level are

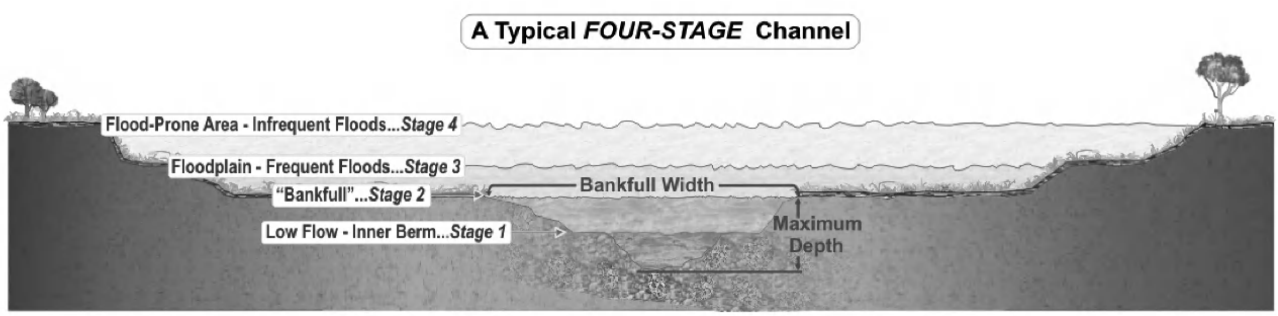

Figure 7. A four-stage channel design typical for a C4 stream type in a valley type VIII.

spread out rather than being vertically and laterally constrained by a greater stream bank height.

4. During drought, the low-flow channel (stage 1) can provide sufficient depth for fish habitat.

5. During high flows, the low-flow channel (stage 1) helps maintain the sediment transport capacity.

6. Increases in the magnitude and frequency of flood peaks due to watershed development or climate changes can be dispersed out of channel and onto a floodplain or flood-prone area.

7. Recreational activities and trails can be created on the floodplain (stage 3) and flood-prone area (stage 4).

8. There is a more natural, visually pleasing river setting.

9. There is a decrease in flood stages for the same magnitude flood due to improved hydraulic and sediment transport efficiency.

10. Habitat is improved, and ecological diversity is increased.

In some situations involving colluvial valley type II (Table 2) or for confined, laterally contained streams in alluvial valleys with meander width ratios (belt width divided by bankfull width) less than 2.0, a flood-prone area exists, which includes the area above the bankfull stage (e.g., B stream types (Table 1) in a colluvial valley type II). Under these conditions, a three-stage channel (Figure 8) exists and is associated with stage 1, the low-flow or "inner-berm" channel; stage 2, the bankfull stage channel; and stage 3, the flood-prone area.

A two-stage channel exists in E stream types (Table 1) in a lacustrine or glaciolacustrine valley type X (Table 2) due to the absence of an inner berm (low flow) channel and a low terrace. The stages involve the bankfull channel and the floodplain/flood-prone area. The two-stage channel is also associated with A stream types in a V-notched valley type I and also with A, B, C, F, and G stream types that are bedrock- or boulder-dominated in a bedrock-controlled valley type VI.

4.3. Channel Dimension, Pattern, and Profile Design

The dimensions and profile of the design channel in traditional river works are often derived from relations developed for clear water discharge, uniform flow, rigid boundary theory, uniform channel materials, and regime relations not stratified by distinct, identifiable river types. Unfortunately, the assumptions are not appropriate for most natural stream channels that are self-formed and self-maintained under much different controlling variables. Hence, traditional river works have typically designed single-thread, "one-size-fits-all flows" in a trapezoidal, flat-bottomed channel [*Soar and Thorne*, 2001]. These channels are often relatively straight and often "hardened" to prevent channel erosion and to increase velocity for major flood stage reduction. Many of these channels have required frequent and expensive dredging as the design did not account for sediment transport capacity. If empirical or regime equations are used to derive channel dimensions (with the understanding of the river types and conditions used to develop the relations), the values should be checked against reference reach data. Accordingly, *Shields* [1996, p. 37] states that "after initial selection of average channel width and depth, designers should consider the compatibility of these dimensions with other factors using guidance provided by *Rosgen* [1994] or their own experience with nearby stable reaches."

In NCD, the cross section involves a multiple-stage channel design as described in the previous section that is required to transport sediment and to provide aquatic habitats and address water quality issues during a range of flows. The design bankfull discharge and the corresponding cross-sectional area are obtained first when developing the proposed channel dimensions by using validated regional curves [*Rosgen*, 2007]. Regional curves of bankfull cross-sectional area versus drainage area generally have an excellent correlation coefficient and low variance making it acceptable to determine the

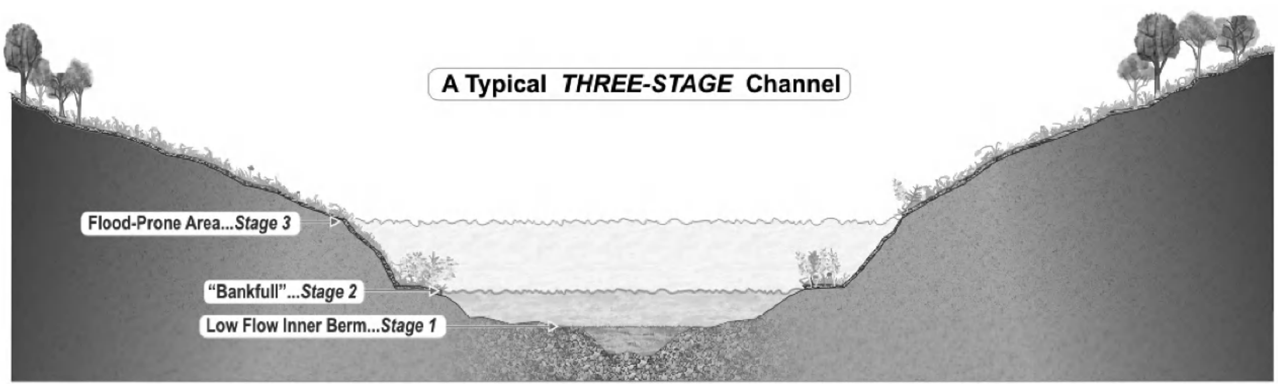

Figure 8. A three-stage channel design typical for B stream types in a colluvial valley type II.

proposed channel's cross-sectional area. However, predicting bankfull width and bankfull depth from regional curves is discouraged due to the consistent higher error term in the relation and because the regional curves are not stratified by stream type (reflecting the variation in width/depth ratio). In scenarios where regional curves are not available or cannot be used (e.g., when project location is below a reservoir), bankfull cross-sectional area can also be calculated from continuity ($A_{bkf} = Q_{bkf} / u_{bkf}$) by knowing bankfull discharge and either knowing or estimating the bankfull mean velocity (u_{bkf}). The bankfull width is then calculated as:

$$W_{bkf} = (A_{bkf} * W/d_{ref})^{1/2},$$

where W_{bkf} = bankfull width, A_{bkf} = bankfull cross-sectional area from regional curves or continuity, W/d_{ref} = bankfull width/depth ratio from the reference reach.

Bankfull mean depth can then be computed by $d_{bkf} = A_{bkf} / W_{bkf}$. Bankfull maximum depth and inner berm channel dimensions are then calculated using dimensionless data from the reference reach and scaled using the bankfull width of the proposed design reach. The mean, minimum, and maximum values for all dimensions must be computed from the ranges specified in the reference reach data. Dimensions are required for all bed features (e.g., riffles, runs, pools, glides, and steps) and also for the floodplain, low terrace, and/or flood-prone areas.

The typical longitudinal profile for NCD involves a range of depths, slopes, and bed feature shapes designed specifically to quantitatively describe bed features. A range of pattern data is also obtained from the dimensionless ratios from a reference reach. Sinuosity is not simply a ratio of valley slope to channel slope but rather is generated from a channel layout incorporating the range of multiple pattern variables that represent natural planform variability, including linear wavelength, stream meander length, belt width, arc length, radius of curvature, riffle length, and pool length ratios. The resulting sinuosity is then determined by dividing the proposed design stream length by the valley length. The meandering pattern determined in NCD (as opposed to straightened, channelized rivers) and the heterogeneity of bed features (e.g., riffles, pools, and glides) are important to dissipate energy and to promote a hyporheic exchange function [*Kasahara and Wondzell*, 2003; *Boulton*, 2007; *Cardenas*, 2009].

The initial channel slope of the proposed design reach is determined by dividing the valley slope by the design sinuosity. This analog method does not rely on an empirical equation but requires compatibility among valley and stream types of the reference reach dimensionless relations and the proposed bankfull width (used as the normalization parameter for pattern). This approach also accounts for any boundary constraints (e.g., terrain and vegetation) within the valley. The final design slope and dimensions are determined following verification of sediment transport capacity and competence.

5. MINIMUM NATURAL CHANNEL DESIGN REQUIREMENTS

Proper implementation of the NCD approach to river restoration must encompass all phases and procedures as outlined in Figure 3. It is also strongly advised that the practitioner be involved in *all* phases. Completing only partial phases or skipping a phase in the NCD method is not an acceptable river restoration practice and will add to the risk of failure and potentially may not meet stated objectives. NCD involves, as a minimum, experience, knowledge, and unique abilities to carry out the following 20 requirements:

1. Be observant and respectful of the complexity of the assignment.
2. Clearly understand and incorporate multiple objectives, including physical, biological, chemical, aesthetic, social, and economical considerations, into restoration designs.
3. Integrate multiple disciplines into the design schemes, including plant science, fisheries, soils, fluvial geomorphology, hydrology, engineering, terrestrial and aquatic biology, and ecology to provide a sustainable design solution that meets the multiple objectives.
4. Seek out ecological criteria and require an analysis of limiting factors for various organisms and their habitats.
5. Obtain and verify the "bankfull discharge" for assessment and design purposes; this includes developing and calibrating regional curves of bankfull discharge versus drainage area. (Note that it is critical that the design discharge not be a flood flow; however, flood flows must be designed and accommodated.) Avoid a "one-size-fits-all flows" and design multistage channels for specific flows including base flow, bankfull, and floods.
6. Identify the driving variables and boundary conditions that influence the channel dimensions, pattern and profile (Figures 1 and 2).
7. Identify the stream succession sequence and the current state of a given river reach (Figure 4) and study and verify the potential, natural stable stream type for the proposed design reach for the given valley type incorporating space for time substitution and recovery potential and direction.
8. Select the appropriate reference reach that meets the controlling variable criteria to establish a range of dimensionless ratios and morphological relations to calculate the

stable dimension, pattern, and profile variables for the natural channel design; do *not* rely on stream structures to create the morphological features over time.

9. Collect and inventory the geomorphic characterization and stream morphology data for the existing and reference reaches.

10. Conduct watershed, river stability, and biological assessments on both the existing reach and reference reach to understand the cause and consequence of past actions that led to river impairment and loss of physical and biological function; this includes time trend assessments, streamflow changes, and erosional or depositional process relations of aggradation, degradation, channel migration, stream bank erosion rates, down-valley channel migration rates, channel enlargement, and sediment supply.

11. Document the exact cause, nature, and extent of the erosional or depositional processes related to instability or disequilibrium (e.g., base-level change due to aggradation, degradation, incision, channel enlargement, accelerated lateral erosion and/or down-valley meander migration).

12. Incorporate hydraulic relations using resistance relations or appropriate prediction methods.

13. Calculate and validate sediment competence and sediment transport capacity for bed load, suspended sand sediment, and total suspended sediment.

14. Maintain consistency for assessment, design, implementation, and monitoring to meet stated objectives, offset the cause of the problems and incorporate the natural variability determined by the reference reach data for layout and the criteria for postrestoration monitoring.

15. Understand the uncertainty of prediction, validate all models, and place controls that document the various process responses from detailed postrestoration monitoring.

16. Recognize the economic and social constraints, prepare reasonable budgets, and present design alternatives to the public and restoration sponsors.

17. Communicate all phases of design to contractors, the public, restoration sponsors, and regulatory personnel.

18. Provide field supervision and training of construction personnel to ensure proper implementation of the design, staging, water quality control, and specification of appropriate equipment and materials needed.

19. Establish success criteria that incorporate meeting specific objectives within the natural variability and dynamic nature of river systems and their ecological function.

20. Monitor to determine the consequence of on-site implementation, evaluate effectiveness of design, validate predictions, assess how well the design met stated objectives, and determine if the stream is self-maintaining within the acceptable range of natural variability; utilize data for future restorations.

This list was developed from field experience over time based on reviews of implemented NCD projects and should alert the stream restoration practitioner to the extensive requirements and challenges involved in the design and implementation of river restoration projects. This is not a complete or exhaustive list. It does indicate, however, that unique skills and experience are required. It is strongly advised not to undertake river restoration without the following: (1) field experience, (2) a strong academic and practical applied science background, (3) incorporating multiple disciplines as necessary, and (4) specific training, mentoring, and peer review.

5.1. Increasing the Risk of Failure

The highest risk of failure comes from not correctly implementing all 10 phases of the NCD methodology and the corresponding 20 minimum NCD requirements. It has been this author's experience that risks are needlessly increased by shortcutting river restoration details and implementation. In addition to not meeting the 20 minimum NCD requirements, the following list documents reasons that increase the likelihood of project failure: (1) insufficient project funding where, unfortunately, completion of projects is encouraged by taking "shortcuts"; (2) implementing designs during poor weather conditions, such as saturated soils, moderate to high flow stages, snow, ice, and frozen ground; (3) utilizing inappropriate materials and stabilization methods, including rock sizes, gabions, fabrics, wrong plant materials, concrete, riprap, and "Jacks"; (4) political and social constraints, such as boundaries of construction limits that are not compatible with minimum river boundaries; (5) using equipment not matched to site conditions or that is inefficient to properly complete the design; (6) not providing irrigation or methods to establish riparian vegetation in a timely manner; (7) not designing floodplain grading of meandering, riffle/pool channels (C stream types, Table 1) in terraced alluvial fill valleys (valley type VIII, Table 2) to ensure that the "flood wave" is opposite of the sine wave of the meander to prevent erosion and gully development in the newly created floodplain surfaces (accomplished by grading from the floodplain height on the inside bend to the low terrace height on the outside bend to allow flood flows to shift opposite of the sine wave of the channel meanders); (8) field supervision during construction is not consistently provided resulting in poorly implemented design; (9) construction given to the lowest bidder regardless of experience in river restoration projects; and (10) disconnects among the individuals doing assessment, design, implementation, and monitoring; the same individuals should be involved in all stages.

5.2. Case Examples

Many projects have failed as the result of problems stemming from the aforementioned list as well as not adequately completing the 10 phases and the 20 minimum requirements. The following are case examples where project failures and nonsustainable project designs increased the risk for failure and where specific minimum requirements (MR) were not met.

The first example is a project in Maryland (White Marsh Run) that failed because the bankfull discharge was not validated (MR 5), and no sediment transport capacity computation was conducted (MR 13). The river was designed and constructed with too high of a bankfull discharge resulting in a high width/depth ratio. The first runoff caused major stream aggradation, although multiple rock structures were used. A misguided concept is that designing and implementing large, dominant stabilization structures will offset the need to correctly design the bankfull discharge and the associated dimension, pattern, and profile of the river (MR 8). This common misconception has led to multiple, yet predictable and preventable failures.

Another common oversight that can lead to failures is designing the wrong stream type for the given valley type (MR 6). This occurred in Virginia following a major hurricane-driven flood where the postflood rehabilitation created a single-thread, straight trapezoidal channel with levees (F3 stream type, Table 1) on an actively building, steep alluvial fan (valley type III, Table 2). The fan was located below a debris flow/debris torrent stream type (A3a+). This transported small boulders, large cobble, gravel, and sand from the A3a+ stream type directly into the Rapidan River resulting in aggradation of the main stem river reach with subsequent hurricane floods. The stable stream type for such actively building alluvial fans is a bar-braided, D3 stream type. This stream type's function is to naturally deposit the coarse erosional debris on the fan surface rather than route it to the main stem reach of the valley floor. The constructed stream type did not follow the geomorphically stable form for this fluvial landform and caused accelerated disequilibrium of the receiving stream.

Projects that are proposed that do not control the cause of instability (MR 10 and MR 13) are often rejected (or should be rejected) for restoration design. One example was on the Swift Current River in Montana where the regulated main stem below a reservoir reduced the flow release during the snowmelt runoff season. This change in the timing and flow reduction caused downstream aggradation and braiding due to the unregulated tributary of Boulder Creek that transported large quantities and sizes of bed load into the regulated main stem reach. The proponent's design was to convert the braided reach (D3 stream type) to a meandering pattern (C3 stream type) reach. However, the cause of the braiding was due to flow regulation and the high bed load that came from an undisturbed watershed; a C3 stream type conversion would have promoted both a high risk and a probability of failure without addressing the flow releases. If the operational hydrology of the dam had been modified to release a bankfull discharge timed with the sediment transport flows of the unregulated tributary, the designed C3 stream type would potentially be sustainable.

Other common project failures have occurred due to constructing "incised" river channels. Degree of incision is a measure of a local reduction in base level and abandonment of an active floodplain as determined by bank-height ratio (the lowest bank height divided by the maximum depth at bankfull stage). If the bankfull discharge or depth is incorrect for the designed dimensions, an incised channel results (MR 5 and MR 8). Flood flows greater than the bankfull stage create excess shear stress and unit stream power in incised channels resulting in accelerated streambed and stream bank erosion. Bankfull discharge, slope, and width/depth ratio are critical design requirements in NCD.

Furthermore, traditional computations for river design are often not appropriate for natural channels and are conservative in nature. The tendency to design a "one-size-fits-all-flows" channel creates oversized widths of stream channels to increase channel capacity to handle floods, reduce velocities within the "minimum" allowable velocities, and reduce shear stress for critical depth computations so as not to entrain D_{50} bed particles (MR 5). Such traditional designs promote high width/depth ratios and sediment deposition or channel aggradation. If validated sediment transport models were applied, these high width/depth ratio channels would indicate the channel process of aggradation (MR 13). Aggrading channels are not only unstable but require high maintenance, add to flood stage problems, and contribute to poor aquatic habitat.

6. DISCUSSION AND SUMMARY

NCD is based on the fundamental principles of form and process integration. Selection of the appropriate form is based on recognition of the controlling processes. In the absence of reasonable time periods to validate prediction methodologies required for design, the reference reach is required to represent the channel process and form relations. There are 67 dependent variables developed from the reference reach and extrapolated to existing impaired reaches for NCD. Critical for proper extrapolation is the inherent stratification of such morphological variables by valley type and stream type. In addition, each stream type within its valley type must further be described by the controlling variables representing the boundary conditions and driving variables. For example, high bed load streams in glacial trough valleys (valley type V, Table 2) with rain-on-snow-dominated hydrographs for their attendant forcing condition will exhibit unique morphology. In contrast, spring-fed systems in lacustrine valley types (valley type X) have cohesive banks, lower bed load, and lower gradients and are associated with meandering, low width/depth ratio, riffle/pool channels with floodplain connectivity (E and C stream types, Table 1).

In addition to the reference reach approach, the NCD method also uses analytical and empirical methods to develop the proposed channel design. Hydraulic and sedimentological relations are predicted and validated. This approach is utilized for river restoration rather than applying an incomplete system of equations prevalent in traditional river design approaches. The major differences between the NCD approach to river restoration and traditional river design works are that NCD (1) integrates multiple disciplines; (2) assumes a higher risk as the design allows for channel adjustment within a stable range and does not "fix" a river in place; (3) generally uses "softer" stabilization materials, such as native materials that include wood and riparian vegetation; (4) often requires a larger watershed perspective to identify the cause of impairment beyond the reach scale; (5) designs a multiple-stage channel to match a range of flows including floods that create floodplain connectivity and function compared to traditional river works that often involve the calculation of flood discharge and trapezoidal channels that accommodate the design flood; and (6) derives the dimension, pattern, and profile variables based on an analog method that integrates process and form relations associated with the controlling variables rather than using analytical models.

The NCD approach, if implemented correctly, will offset many of the adverse consequences and problems identified from past traditional river works. The incorporation of NCD procedures provides for more sustainable designs that are intended to work in harmony with the river. The method requires rigor in field observations. The NCD method has been successfully implemented on hundreds of river restoration projects by this author and many others since its inception [e.g., *Berger*, 1992; *National Research Council*, 1992, pp. 217–228; *Klein et al.*, 2007; *Hammersmark et al.*, 2008, 2010; *Ernst et al.*, 2010; *Pierce et al.*, 2008; *Baldigo et al.*, 2008, 2010].

Less than 5% of the multiple and large-scale restoration projects constructed by the author have required any maintenance. A 5 year postrestoration monitoring project was conducted by the Colorado State University on a 19 mi restoration project designed and constructed by the author (Little Snake River, Three Forks Ranch in Colorado, and Wyoming). The results of this monitoring verified that very little maintenance was required and that the project met the restoration objectives [*Bledsoe and Meyer*, 2005; *Meyer*, 2007]. This project involved channel relocation and reconstruction of the dimension, pattern, and profile that incorporated a variety of river structures and reestablishment of the riparian vegetation community. The design also reconnected the floodplain and involved a rise in the water table with oxbow lakes, an improvement in aquatic, terrestrial, and waterfowl habitat, as well as a change in the livestock grazing system (the cause of impairment). "NCD has proven to have enormous practical and economic utility for the growing stream restoration field [*Lave*, 2009, p. 1529]." Success or failure of this method is closely linked to the 10 phases and the 20 minimum requirements in addition to the experience of the restoration practitioner and the required attention to detail.

As in any science, river restoration involves multiple processes and forms whose predictions are not only complex but require extensive field validation over time. Integrating the combined experience from river studies to develop classifications and fundamental relations form the basis of the NCD method. Due to the recognized uncertainty of prediction, continued validation is not only encouraged but essential to provide confidence in the method. It has been this validation and testing that has modified and improved the NCD approach over four decades. As restoration objectives continue to expand, the tools required to meet such demands will continue to be updated. Regardless, the basic tenet for this work should be to continue to monitor in a manner that helps us direct our future work: for the answers are to be found in the river.

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Geomorphological Approaches for River Management and Restoration in Italian and French Rivers

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River management and restoration in Italy and France are increasingly considering physical processes and trends of channel adjustment as a basic knowledge for enhancing river conditions and promoting channel recovery. Italian and French rivers are characterized by a long history of human disturbances and land use changes. As a consequence, trends of channel adjustments and related management problems are similar, with a historical phase of aggradation followed by a period (last century) of intense channel incision and narrowing. A general overview on recent progress in using geomorphic approaches to river management in Italy and France is presented here by illustrating a series of examples of studies and management applications. A synthetic state of the art on the recent morphological changes of Italian and French rivers is first reported. Some examples of quantification of bed load are also illustrated, providing a necessary quantitative knowledge for possible interventions or strategies for promoting bed load recovery. Finally, examples are provided to illustrate how an understanding of geomorphic processes is used to define regional visions and associated tools for planning and targeting actions and to promote sustainable actions from local to catchment scale.

1. INTRODUCTION

Nowadays, it is widely recognized that physical processes, including those of sediment production, transfer, and storage, are fundamental to the ecological functioning of fluvial systems [see, for example, *Boon et al.*, 1992; *Goodson et al.*, 2002; *Kondolf et al.*, 2003; *Wohl et al.*, 2005; *Brierley and Fryirs*, 2005; *Florsheim et al.*, 2008; *Habersack and Piégay*, 2008]. The geomorphic dynamics of rivers is increasingly seen as vital for creating and maintaining the physical habitats that underpins the survival of aquatic and riparian flora and fauna. Sediment transport, bank erosion, and associated channel mobility represent key physical processes, and their understanding is of crucial importance for defining management strategies and river restoration measures.

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As a consequence, river management and restoration in Italy and France, as in other parts of the world, are increasingly taking fluvial processes into account as a necessary condition for enhancing river conditions to improve ecosystem value or provide other benefits for society. Geomorphic problems of fluvial systems and their management are, to some extent, similar in these two countries, with both having a long history of human impacts and similar trends of land use changes and channel adjustments during the last few centuries. In both cases, a progressive reduction of sediment delivery at a regional scale, due to dam constructions, steep upland stream regulation, and land use change, has affected many regions, with the effects particularly evident in the piedmont areas of mountain regions [*Liébault and Piégay*, 2001, 2002; *Surian and Rinaldi*, 2003]. Recent (last century) intensive sediment mining has had one of the most important impacts, inducing further channel changes, with severe bed incision (maximum of 12 to 14 m) especially significant [*Peiry*, 1987; *Bravard*, 1991; *Landon et al.*, 1998; *Rinaldi and Simon*, 1998; *Surian and Rinaldi*, 2003; *Rinaldi et al.*, 2005]. Such morphological changes have had abiotic and biotic consequences followed by ecological and economical impacts [*Bravard et al.*, 1999]. This critical situation in terms of management (channel instability problems, limitations of flood regulation works, and biodiversity decrease) has led to an increasing need for sustainable management of the bed load and other physical processes, but also to undertake measures for mitigating the impacts of channel incision [*Bravard et al.*, 1999; *Habersack and Piégay*, 2008; *Liébault bault et al.*, 2008].

The aim of this chapter is to provide a general overview of recent progress in the application of geomorphic approaches and concepts to river management and restoration, in the specific context of Italian and French rivers. This will be achieved by illustrating a series of case studies and examples, with a particular focus on some rivers of central and northern Italy and of southeastern France. The case studies mainly include alluvial, mobile gravel bed rivers, most of them with a present or previous braided pattern associated with relatively high bed load. In these environments, the energy available due to steep slopes is important, and channel features are sensitive to changes in control parameters (peak flow regime and bed load input). The geomorphic diagnosis established to design actions combined much knowledge from the different disciplines of this research field, involving engineers, geologists, and geographers. A wide space-time framework is considered to understand variability and provide a long term and sustainable vision of river functioning. This approach is combined with traditional local-scale engineering to provide quantitative assessments and physical process understanding. The structure of the chapter reflects how this geomorphic knowledge can be achieved and applied for sustainable river management, by illustrating the following points: (1) the need for historical analysis to highlight the temporal trajectory and sensitivity to changes, (2) the need to evaluate the sediment budget so as to understand the balance between sediment transport and delivery, and (3) use of all this knowledge for designing regional visions, strategies, and targeting actions.

2. MORPHOLOGICAL CHANGES OF RIVER CHANNELS

A detailed retrospective study, including an analysis of morphological channel changes and trends of adjustment and relations with potential causes, represents the first fundamental step in the definition of appropriate strategies for river management and restoration. It is important to identify channel changes and the types of adjustments that have generated the present channel morphology at a time scale of the order of the last 100–200 years and to identify the trajectory of channel evolution [*Brierley and Fryirs*, 2005; *Dufour and Piégay*, 2009], so as to understand present process-form interactions and the response of the river system to human impact or other natural factors. Understanding this trajectory is important for predicting whether proposed restoration actions have a chance to be successful or not, depending on the capacity of the river to react to them and to understand the present river behavior. Independent of any restoration actions, the river is changing, as it is not static but dynamic, and any intervention must anticipate this change so that these corrective measures can be sustained.

During the last decade, many studies have been addressed to understand and clarify the past and recent evolution of fluvial systems in Italy and France. In France, several studies have been focused on the piedmont Alpine area, and they have been able to identify the main types of channel adjustments and associated causes [e.g., *Liébault and Piégay*, 2001, 2002]. Sediment delivery has decreased at a regional scale due to dam construction, steep upland stream regulation, locally reducing or interrupting sediment transport, and in-channel mining, which removes and stores sediment locally sometimes over decades. Moreover, this mountain area has undergone another significant change, a major afforestation due to human depopulation and agricultural decline, which reduced land sensitivity to erosion and decreased sediment delivery. This factor is then exacerbating the sediment deficit downstream resulting from damming, torrent controls, and in-channel mining, since the rivers are not yet adjusted to this factor and whose effects are still propagating downstream.

This significant land use change has not only been observed in the catchment areas but also along the major river corridors. Afforestation within the river corridors, in areas which were actively used for grazing, also explains the generation of new natural processes such as the introduction of wood and its transfer within the hydrographic network (Figure 1). All these phenomena have been responsible for channel metamorphosis at a network scale with narrowing and incision as the common response, the former usually occurring slightly before the incision as it was mainly associated with floodplain/catchment abandonment (circa 1930s to the late 1960s), whereas incision reached a peak in the 1970s in relation to intense mining activity.

In Italy, systematic studies on channel adjustments were carried out from the end of the 1990s [*Rinaldi and Simon*, 1998; *Surian*, 1999; *Rinaldi*, 2003; *Surian and Rinaldi*, 2003]. More recent studies [i.e., *Surian et al.*, 2009a] have involved a systematic analysis of channel changes over the last 200 years, on a larger number of study cases, with the aims of reconstructing the channel changes and understanding the relationship between channel adjustments and various human interventions. Twelve rivers in northern and central Italy have experienced almost the same processes in terms of temporal trends; however, the magnitude of adjustment varies on a case-by-case basis, according to several factors such as the original channel morphology [*Surian et al.*, 2009a]. After a historical phase of dominant floodplains and in-channel aggradation (in some cases since Etruscan-Roman times until the nineteenth century [*Billi et al.*, 1997; *Caporali et al.*, 2005]), river channels underwent two phases of narrowing (up to 80 %) and incision (up to 8 to 10 m), which started at the end of the nineteenth century (phase I), and was very intense from the 1950s to the 1980s (phase II). A series of human impacts have been recognized as responsible for these channel changes. Referring to a series of representative case studies in northern and central Italy reported by *Surian et al.* [2009a], the main human impacts and relative periods and duration include (1) various types of

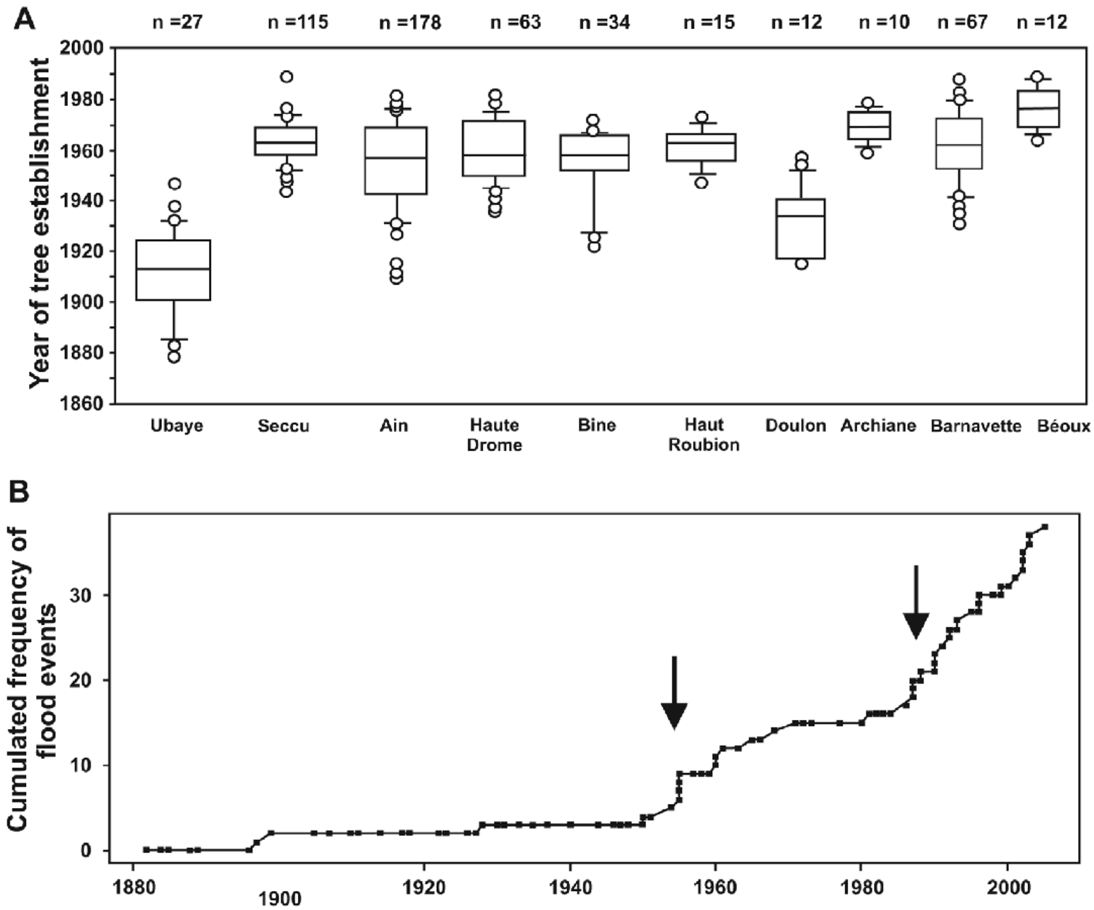

Figure 1. (a) Date of tree establishment in mature units of the riparian forests of a few French southeast rivers. (b) Cumulative frequency of wood jams during floods that occurred in rivers of the northern French Alps between 1880 and 2005 based on press articles. Modified from *Le Lay* [2007].

river training works, such as groins, levees, and bank protections (nineteenth to twentieth century); (2) afforestation, slope stabilization, and construction of weirs along tributaries in the upper portions of the catchments (from 1920s to 1930s); (3) dams (mainly 1930s–1960s); and (4) sediment mining (mainly from 1960s to 1980s). Such interventions have caused a dramatic alteration of the sediment regime, whereas effects on channel-forming discharges have seldom been observed. Then, over the last 15–20 years, channel widening and sedimentation have been observed along parts of the reaches (phase III), while a continuation of the previous phase of incision and narrowing was also observed in other cases. Among 24 subreaches analyzed in the study of *Surian et al.* [2009a], six cases have shown a continuation of channel narrowing, while in 18 cases, an inversion of the trend of channel width adjustments was observed. Among these 18 cases, 14 subreaches have revealed an increase up to 20% and in only four cases, an increase higher than 20% of the original width in the nineteenth century. Therefore, the magnitude of widening and aggradation has generally been much lower compared to those of the previous phases of narrowing and incision.

Various factors can be considered to explain this inversion of trend for part of the study rivers. First, the cessation of the intense period of gravel mining (since the end of the 1980s) can be considered the most important factor, implying higher in-channel sediment supply [*Rinaldi et al.*, 2009; *Surian et al.*, 2009a]. Second, the occurrence of some large flood may have been the driving factor during the most recent phase of adjustment, as evidenced by the evolution of the Orco River, where a very large flood (the largest recorded in the twentieth century) occurred in October 2000 [*Surian et al.*, 2009a], and by the Magra River and Vara (its main tributary), where two flood events with estimated return period of about 20–30 years occurred on 2000 and 1999, respectively [*Rinaldi et al.*, 2009].

A schematic summary of the main types and phases of channel adjustments is illustrated in Figure 2, relative to a series of case studies of the central and northern Apennines (Cecina, Magra, Vara, and Panaro rivers [*Rinaldi et al.*, 2008]). Similar trends have been observed along rivers of the piedmont Alpine area [*Pellegrini et al.*, 2008; *Surian et al.*, 2008, 2009a].

Based on the studies previously mentioned, Italian and French rivers exhibit many common characteristics and trends of evolution, both in terms of channel adjustments and human disturbances. They also possess some differences in terms of controlling factors and evolution scenarios. In both countries, channel incision and narrowing have been identified as the two main types of adjustments, although channel narrowing in Italian rivers appears less associated with land use changes in the river corridor and more related to sediment mining. Besides, bed incision generally also reaches higher amounts in the Italian rivers also as a consequence of intense sediment mining.

Cases of recent widening are also observed along some of the French rivers, notably on the lower Ain River [*Rollet*, 2007] or along the Drôme [*Liébault*, 2003], and appear to be associated with more intense floods that occurred in the 1990s. This recent trend does not counteract the long-term narrowing evolution observed, and it can be interpreted as a short-term fluctuation of channel width associated with the

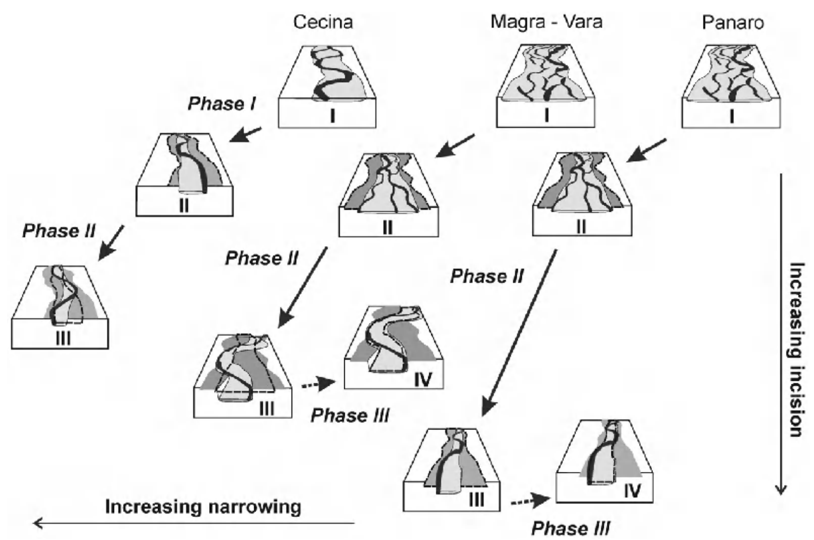

Figure 2. Summary of the types of channel adjustment for three study cases (Cecina, Magra-Vara, Panaro rivers) in the Apennines of central northern Italy. Modified from the work of *Rinaldi et al.* [2008].

intense floods. This period, characterized by floods with a similar intensity, mirrors those observed at the end of the nineteenth century, notably on the Rhine and the Rhône, for which we have long hydrological series, showing that these events are not unusual within this period of time. As shown by *Piégay et al.* [2009], however, the magnitude of this widening is low if compared to the magnitude of narrowing observed at the contemporary time scale and concurs with the intensity of channel width fluctuations related to flood series, as we have observed on different rivers and notably on the Drôme. Moreover, a new narrowing trend is observed following these 1990s/early 2000s events, notably on the Drôme and other braided rivers located in the French Alps [*Hervouet*, 2010], and interpreted as a recovery process acting in a fairly stationary trend.

These geomorphic changes observed at a regional scale in both France and Italy are also observed in the Pyrenees [i.e., *Garcia-Ruiz et al.*, 1997; *Rovira et al.*, 2005] and can be considered a general response to the pressures of human society across Europe, with remarkable differences to what is observed in other parts of the world where deforestation and its impact is more commonly described.

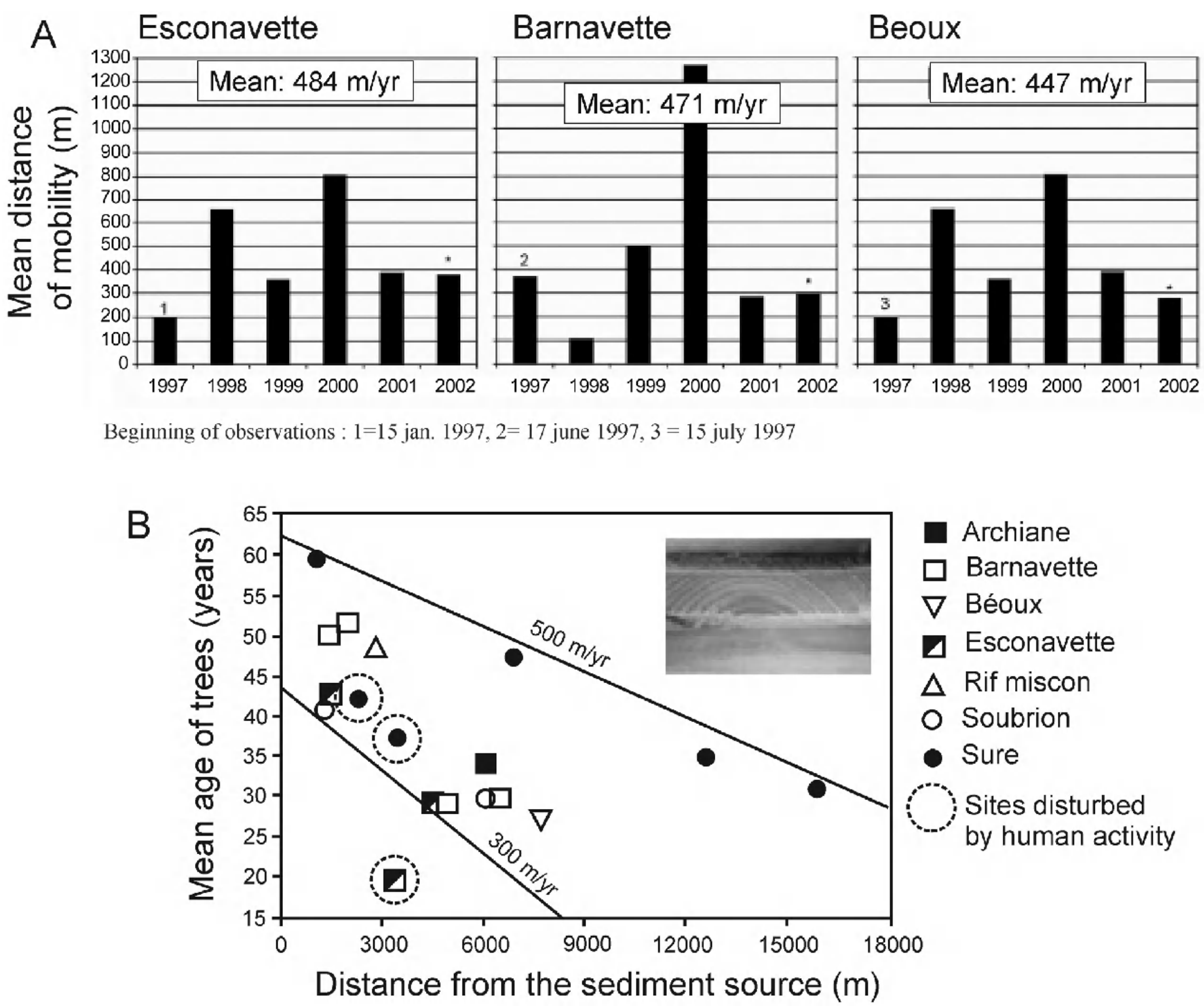

Figure 3. Evidence of sediment deficit propagating downstream along the Drôme River. (a) Mean annual distance of bed load propagation along three tributaries of the Drôme surveyed between 1997 and 2002. Modified from the work of *Liébault and Clement* [2007], reprinted with permission from Copibec. (b) Mean age of mature trees established along tributaries of the Drôme according to the distance from the sediment sources. Modified from the work of *Liébault et al.* [2005], reprinted with permission from John Wiley and Sons.

The research undertaken in the Drôme catchment has demonstrated that the afforestation has generated a deficit of sediment that is propagating downstream [*Liébault*, 2005; *Liébault et al.*, 2008]. This deficit is observed 15 km downstream from the sources using dendrochronological evidence, and the mean annual distance has been estimated to 500 m yr^{-1} using both longitudinal trend fitted on the dendrochronological data but also observations of bed load migration using tracers (Figure 3). Similar observations have been made downstream of dams along the Ain River. Aerial photo analysis combined with longitudinal survey of the coarsest grain size of bar heads showed that a sediment deficit is observed with a mean annual propagation of ~500 m as well [*Rollet*, 2007; *Rollet et al.*, 2008]. These two examples of propagation of sediment deficit resulting from human pressures that occurred between the 1940s and the

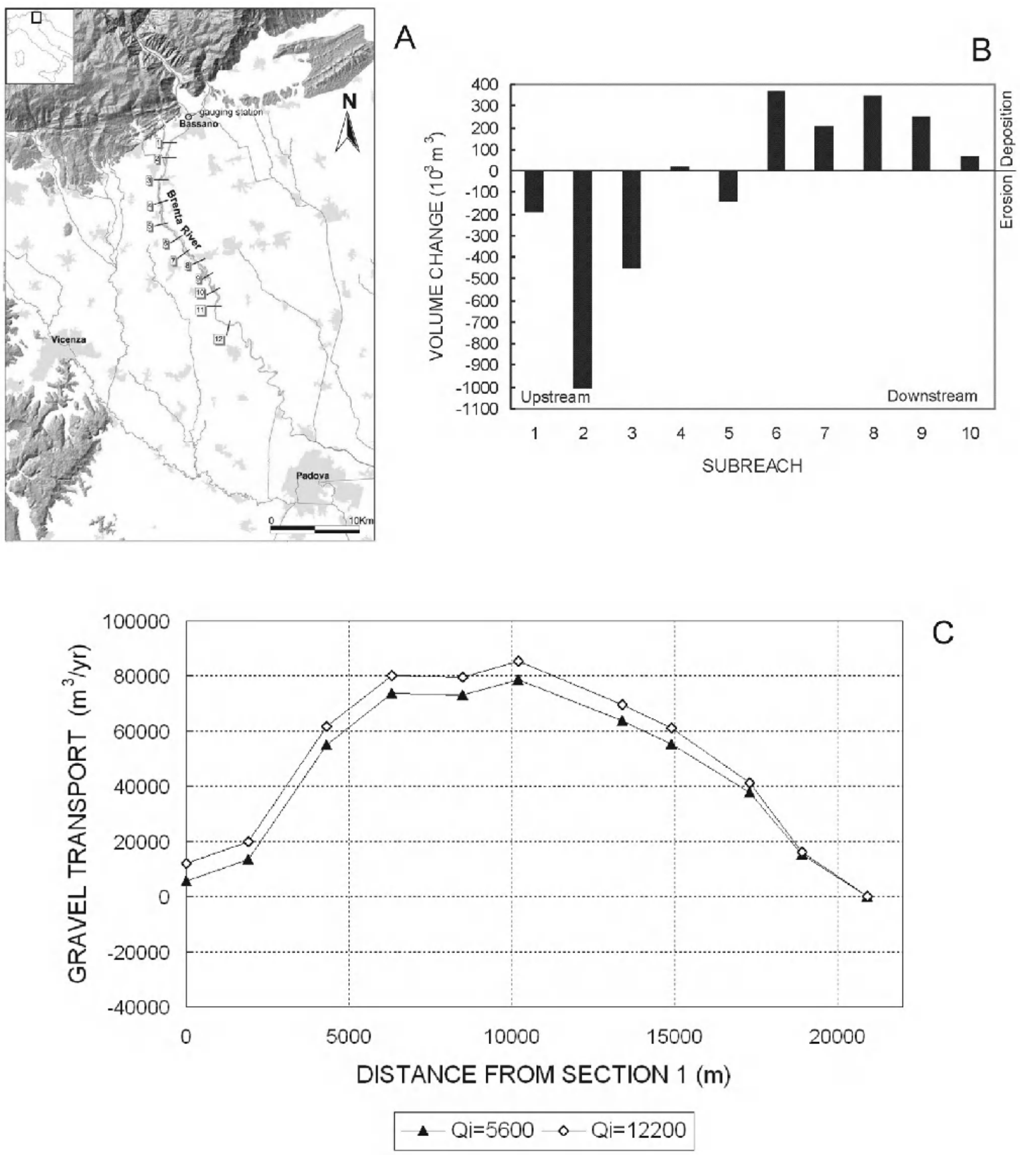

Figure 4. Bed load estimate using a morphological approach along the Brenta River. (a) Location of the 11 monumented cross sections used for the sediment budget. (b) Erosion and deposition volumes computed by subreaches for the period 1984 to 1997. (c) Gravel budget represented as sediment transport at subreach boundaries. The budget was computed using two possible conditions, i.e., sediment transport equal to 5600 and 12,200 m^3 yr^{-1} at the upper section of the study reach. Modified from the work of *Surian and Cisotto* [2007], reprinted with permission from John Wiley and Sons.

1960s clearly demonstrate that the adjustment process at the regional scale is just beginning. The adjustment occurred only 15 km downstream from the sediment sources on the Drôme and 12 km downstream from the Allemand Dam along the Ain, suggesting that the adjustment will probably continue downstream for years.

3. QUANTIFICATION OF BED LOAD AND SEDIMENT BUDGETS

In recent years, much progress has been made in the measurement and prediction of bed load transport on gravel bed rivers. In the last decades, sediment budgeting based on morphological approaches has been studied in different geographical contexts based on various sets of data [i.e., *McLean and Church*, 1999; *Ham and Church*, 2000; *Brewer and Passmore*, 2002]. The technique involves the evaluation of bed load by assessing the morphological changes that reflect erosion, transport, and deposition of sediment over a given period. The morphological method is based on the continuity principle applied to bed material in the channel zone and involves the quantification of sediment inputs, outputs, and storage changes in a defined reach:

$$S_o = S_i - \Delta S, \tag{1}$$

where S_o is the bed material output, S_i is the bed material input, and ΔS is the change in storage. If these variables are measured over a period of time, then the equation becomes

$$Q_o = Q_i - (1-p)\, \partial S/\partial t, \tag{2}$$

where Q_o is the volumetric transport out of the reach per unit time ∂t, Q_i is the volumetric transport into the reach per unit time, and p is the porosity of the sediments.

Sediment budgeting using the morphological approach in gravel bed rivers has been applied over a variety of spatial and temporal scales, from discrete bars and single flow events, to extended channel reaches of several kilometers and decadal intervals or longer. Depending on the scale, methods and data sources used to estimate channel changes can also vary, from detailed direct field surveys to combinations of archived channel long profiles, cross sections, and remote sensing sources for characterizing topographical and planimetric changes.

Examples of application of the morphological approach for sediment budgeting for some Italian and French rivers include the works of *Rollet* [2007], *Surian and Cisotto* [2007], and *Simoncini* [2008].

The first case reported here is the Brenta River [*Surian and Cisotto*, 2007], which is one of the largest rivers draining the Dolomites (Southern Alps, Italy), having a length of 174 km and a drainage basin of 1567 km^2. The morphological sediment budget was constructed to analyze the present condition of river in terms of bed load transport and sediment sources. The morphological method was applied to a 21 km reach, using 11 monumented cross sections surveyed in 1984 and 1997 (Figure 4a). The construction of the gravel budget involved the following steps: (1) estimating the net change in area for each cross section, (2) estimating the net change in volume for each subreach, (3) analyzing grain size to estimate the proportion of material transported on the bed and its porosity, and (4) identifying a cross section where sediment transport is known. Once a net change in area was estimated for each cross-section, values of adjacent sections were averaged and multiplied by the distance between them to obtain the net change in volume for each subreach [*Griffiths*, 1979]. The estimate of change in volume shows clearly that, in the upper part of the study reach, erosion has been the dominant process, whereas deposition has occurred predominantly in the lower part (Figure 4b). The overall gravel budget, represented as gravel transport at subreach boundaries, is shown in Figure 4c for two possible conditions of bed load at the upper section of the study reach. These two conditions were based on (1) available measurements of suspended load and (2) possible contributions of bed load to the total load based on bed load-to-suspended load ratios of 0.33 and 0.15. The sediment budget obtained, although affected by some approximations, is useful to investigate the spatial variations of bed load along the reach. The budget shows that gravel transport increases significantly in the upper part of the study reach, remaining relatively constant in the middle part and then decreasing to zero in the last 11 km of the study reach. Such spatial variations in gravel transport highlight that there are major contributions of local erosion to bed load transport and, therefore, to the total sediment deposited in the study reach.

Having determined that most of the material available for transport is sourced from local erosion with only a small proportion derived from upstream (i.e., from the drainage basin), investigations can be concentrated on local erosion to determine the proportion between the channel bottom and banks. The analysis of cross-sections revealed that bank erosion was the dominant process in the period 1984 to 1997, contributing 83% of the total erosion in the study reach. The next step was to make an estimate of the material eroded from the banks. Areal changes along the banks were estimated using aerial photographs, whereas bank heights were estimated using lidar data and field observations. After correcting for porosity and fine sediment, the total amount of gravel coming from the banks was 110,000 m^3 yr^{-1}. This implies that the amount of gravel coming from the banks was 10 to 20 times larger than the amount coming from the drainage basin (5600 to 12200 m^3 yr^{-1}, see Figure 4c).

The budget was calculated for the period 1984–1997, and over that period, the channel mainly experienced widening. Over this period, gravel deposition occurred mainly within the channel, and very small portions of the channel were stabilized by vegetation.

Another approach in sediment budgeting involves the use of sediment transport equations for each of a series of discrete, relatively homogeneous, subreaches within the study river. Based on the sediment continuity equation, a mean annual sediment budget can be obtained for each subreach by estimating the difference between the input of bed load from the upstream subreach (assuming the flow is at its transport capacity), plus the input from any major tributaries and the output from the given subreach:

$$\Delta Q_{s\,(i)} = Q_{s\,\mathrm{IN}\,(i-1)} + Q_{s\,\mathrm{IN}\,(t)} - Q_{s\,\mathrm{OUT}\,(i)}, \tag{3}$$

where $\Delta Q_{s\,(i)}$ is the mean annual sediment budget (m^3 yr^{-1}) for the subreach (i), $Q_{s\,\mathrm{IN}\,(i-1)}$ is the mean annual sediment input (m^3 yr^{-1}) from the upstream subreach ($i-1$), $Q_{s\,\mathrm{IN}\,(t)}$ is the mean annual sediment input (m^3 yr^{-1}) from tributaries in the subreach (i), and $Q_{s\,\mathrm{OUT}\,(i)}$ is the mean annual sediment output (m^3 yr^{-1}) from the subreach (i). In contrast to geomorphological sediment budget, this approach does not account for observed changes of the river channel in a given time interval, but rather expresses the tendency of each subreach to aggrade (positive values) or degrade (negative values) given its hydraulic and sedimentary characteristics.

This approach has been applied to the Magra and Vara rivers (central and northern Italy) in a study aimed at defining a program for future sediment management [*Rinaldi et al.*, 2009]. In this specific case, four bed load equations were used (Shields, Schoklitsch, Parker, and Meyer-Peter and Müller, in the form corrected by *Wong and Parker* [2006]), as they were considered the most suitable for the characteristics of the study rivers. The estimated mean annual bed load sediment budgets are summarized in Figure 5. Sediment budgets derived from the four bed load equations differed by 1 or 2 orders of magnitude but gave consistent results in relation to overall incision/aggradation tendencies for 22 out of 23 subreaches. These tendencies were then used, in combination with other parameters, to define a classification of the river aimed at aiding sediment management (see following sections).

An attempt to calculate the gravel budget by including the contribution of the river banks has also been undertaken. Mean annual rates of bank retreat were obtained by comparison of recent aerial photographs, while bank height and composition were obtained by field measurements. The results have shown that, on a total of nine subreaches that were interpreted to have a tendency to incision without consideration of the banks, six of them changed their tendency to

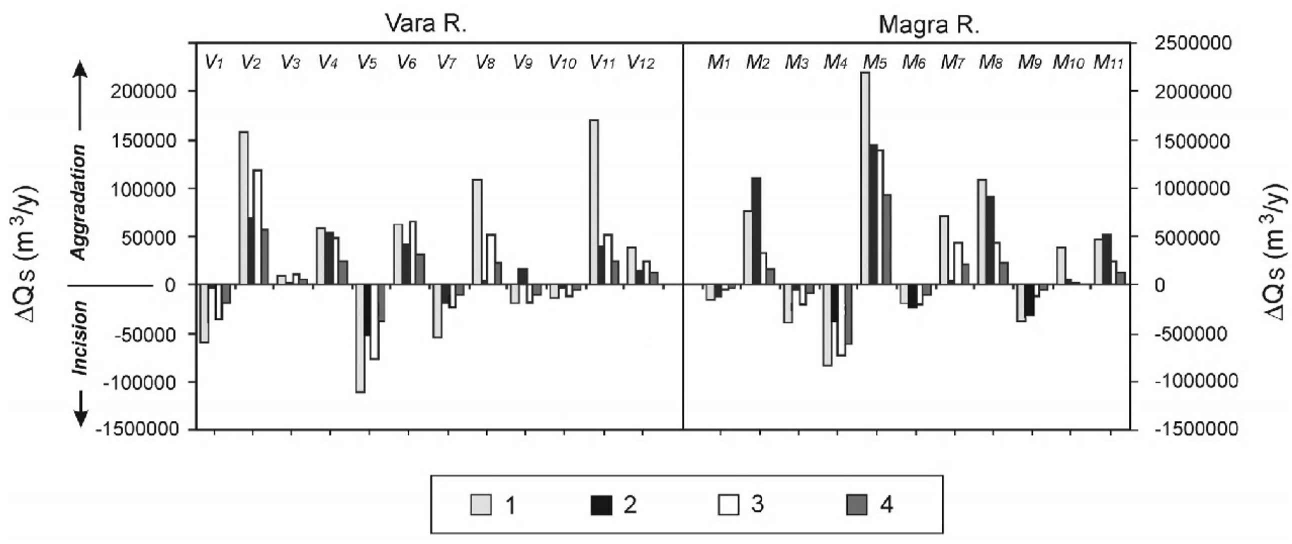

Figure 5. Sediment budget using bed load equations for the Magra and Vara rivers. ΔQ_s represents the mean annual sediment budget for a given reach. (left) Values calculated by the Schoklitsch formula are shown. (right) Values calculated by the other formulas are shown. Positive values indicate a tendency of the reach to aggrade; negative values indicate tendency to incision. Shading is defined as follows: 1, Shields equation; 2, Schoklitsch equation; 3, Parker equation; 4, Meyer-Peter and Muller equation. *V*1 to *V*12 and *M*1 to *M*11 indicate reaches of the Vara and Magra rivers, respectively. Modified from the work of *Rinaldi et al.* [2009], reprinted with permission from John Wiley and Sons.

aggradation once the banks were included in the calculations. Both the sediment budgets on the Brenta and on the Magra and Vara rivers, although with different approaches, have demonstrated in a quantitative way the role of the banks as a significant source of bed load sediment in the pluri-decadal adjustment context that we observed in this study.

These reach-scale approaches can also be combined with an estimate of bed load delivery from subcatchments, to facilitate comparison with independent calculation methods and to assess the relative contributions of valley reaches and upstream basins. Such approaches can be based on detailed field measurements and extrapolations for establishing potential scenarios for sediment management. Field data on annual bed load transport can therefore provide a gross assessment of bed load transport conditions in neighboring catchments for which no data are available. A key-contribution of this type of study comes from the works of *Liébault* [2005] and *Liébault et al.* [2008] in France reporting bed load yields for a series of rivers and streams of the Southern French Prealps, which were used to establish a regional law linking catchment size and the mean annual bed load transport (Figure 6).

Following these examples, some critical points can be discussed as follows.

1. The major incision and narrowing that occurred along Italian and French rivers has created a new floodplain that has not yet adjusted to the new sediment supply and transport conditions. As a consequence, when large magnitude floods occur as in the 1990s, then channel widening can introduce significant amount of gravel to the river, which, in turn, smoothes the impact of mining activity. The question is now to define the period of time required for the floodplain to adjust to the new conditions and to see if the bank erosion can be a long-term process of sediment delivery or only a transitional one. The example of the Ain River provides additional elements to this discussion. Along the Ain River downstream from the Allemand Dam, the historical analysis of aerial photos was combined with field survey of bank height and estimates of overbank sedimentation to develop a sediment budget over the 1980 to 2000 period [*Rollet*, 2007]. It appears that bank erosion re-introduced 205,000 m^3 of sediment per year, but 210,500 m^3 yr^{-1} were stored following bar encroachment by vegetation. As a consequence, this river is not significantly incised, notably in the reaches where the channel is actively mobile is not recharged by bank erosion, and sediment is therefore thought to be sourced mainly from upstream. When damming occurs, the channel begins to transport sediment stored in the bed providing a winnowing process and associated channel incision. Once incision occurs in the channel, it seems that it is much less mobile, so that sediment loading from bank erosion is not significant.

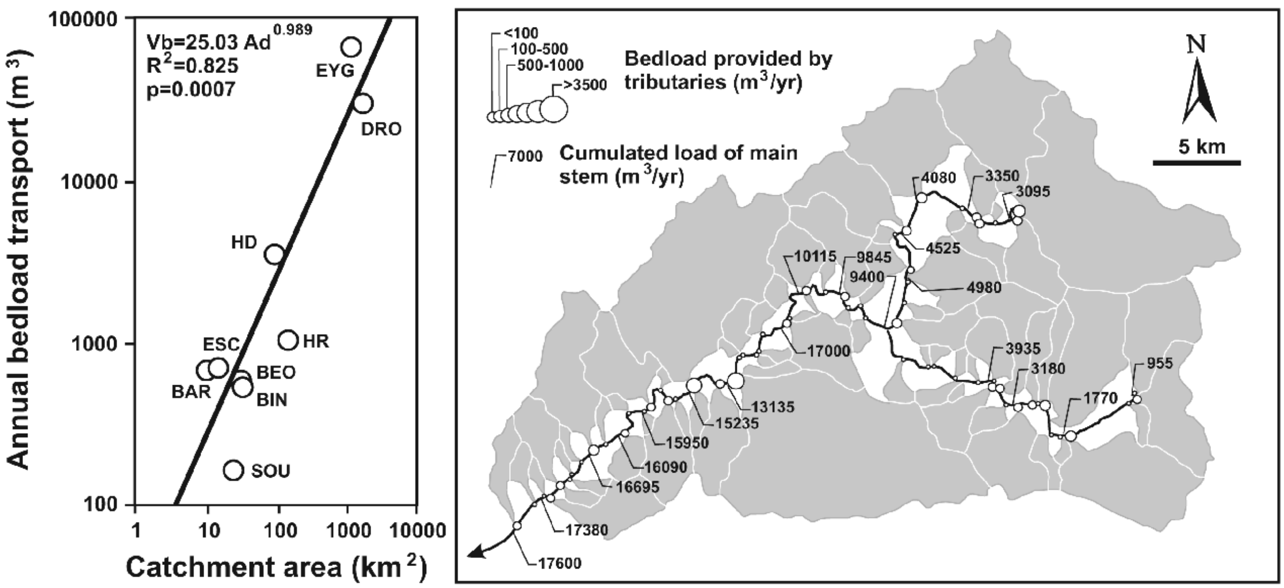

Figure 6. Bed load contributions at catchment scale: (left) calibration of the statistical law from field surveys and archived data performed on three catchments of the Drôme, the Eygues, and the Roubion and (right) its application to the Eygues River network. Modified from the work of *Liébault et al.* [2008], reprinted with permission from Wiley-Blackwell.

2. The bed load contribution from the uplands must be better known where it seems that sediment delivery is decreasing and the rivers are using the valley sources during the widening process. Preliminary work was done to extrapolate local known sediment transport rates to better understand basin contribution, but the inner catchment complexity must be better characterized for targeting potential actions on sediment transport restoration and preservation. It is important to determine where the most active tributaries are located and their contributions to the bed load transported by the main stream, as this can help in sustainable sediment management.

3. Over the last decade, geomorphologists often consider processes rather than forms when promoting restoration actions or mitigations, since these are more sustainable, long-term efficient measures for channel adjustments. Nevertheless, the question of "sustainable solutions" is still underexplored. It is important to consider the appropriate time scale when promoting sustainable solutions. Even if we have a good understanding of past adjustments, it is still difficult to provide a clear scenario of the future adjustments because the process is not completely ended, and there still are floodplain properties that can slow down the adjustment time. The use of local known physically based processes in the extrapolations that we can do at catchment scale is therefore a new scientific frontier to provide data for sustainable scenarios. Prospective geomorphology is a new application, where important uncertainties must be identified and time scales of channel adjustments need careful consideration if we really want to move from a local-scale traditional expertise to the promotion of long-term sustainable options [*Pont et al.*, 2009]. Thinking process rather than form is easy to say, but it is difficult to apply.

4. APPLICATION OF GEOMORPHIC APPROACHES TO RIVER MANAGEMENT AND RESTORATION

Even if important uncertainties must be considered, the low level of understanding that we have of rivers has been used for improving river management and restoration in Italy and France. As it is promoted in other parts of the world, applied initiatives are increasingly including considerations on physical processes, such as bank erosion, sediment transport, channel incision, and water flows, among others, as a necessary condition for enhancing river conditions and promoting channel recovery.

Channel incision and other related adjustments have had dramatic environmental and societal effects [*Bravard et al.*, 1999], and there is an increasing effort to mitigate its consequences and promote channel recovery. The following possible scientific approaches and management strategies can be identified, depending on the decision makers involved, the spatial scale that are considered (local, drainage basin, and national scale), and the geomorphic conditions of the river on which management actions are promoted. This could include (1) promoting a regional vision, based on a geomorphic diagnosis, and identifying associated tools for planning, targeting, and regulating actions and (2) promoting sustainable actions at the local/catchment scale, aimed at preservation, mitigation, or restoration of physical processes and associated channel features. Some examples of application of these possible approaches are provided in the following sections.

4.1. Promoting a Regional Vision and Associated Tools for Planning and Targeting Actions

Braided rivers are generally characterized by high energy, low bank resistance, and a large amount of transported sediment, creating highly dynamic channels, bars, and islands that provide valuable in-channel and riparian habitats. They can alternate expansion and contraction phases, depending on various factors such as sediment supply and the occurrence of large floods. Figure 7 provides a conceptual framework on how various management measures can be associated with different evolutionary phases [*Piégay et al.*, 2006]. Braiding intensity (dotted gray line) decreases with decreasing sediment supply: t_1 and t_2 represent time-points along a trajectory of evolution from high sediment loads, high braiding intensity (expansion phase) to low sediment supply, low braiding intensity (contraction phase). Social and economic benefits of braided rivers (solid black line) describe a bell-shaped curve, with intermediate levels of braiding providing the most benefits. Ecological benefits (dashed black lines) of braided rivers differ from one system to another according to other parameters (low flow conditions, turbidity, versus diversity). Following trend A, ecological diversity has a peak at an intermediate level of braiding, mostly in riparian environments. Some rivers can also follow trend B (e.g., Fraser River and Platte River) providing ecological value not only at the intermediate levels of braiding but all along the braiding stage; their ecological interest diminishes once the gravel surface area decreases, and the associated optimal habitat conditions are no longer present.

In practical terms, historical geomorphology of rivers must be known at a regional level to understand where they are located on the socioeconomic curve and what the most appropriate management strategies integrating channel evolution at a decadal scale are. Process understanding is important to consider the sensitivity of systems to change and the associated time scale of changes. According to the energy, the distance of the concerned reach to the sediment sources,

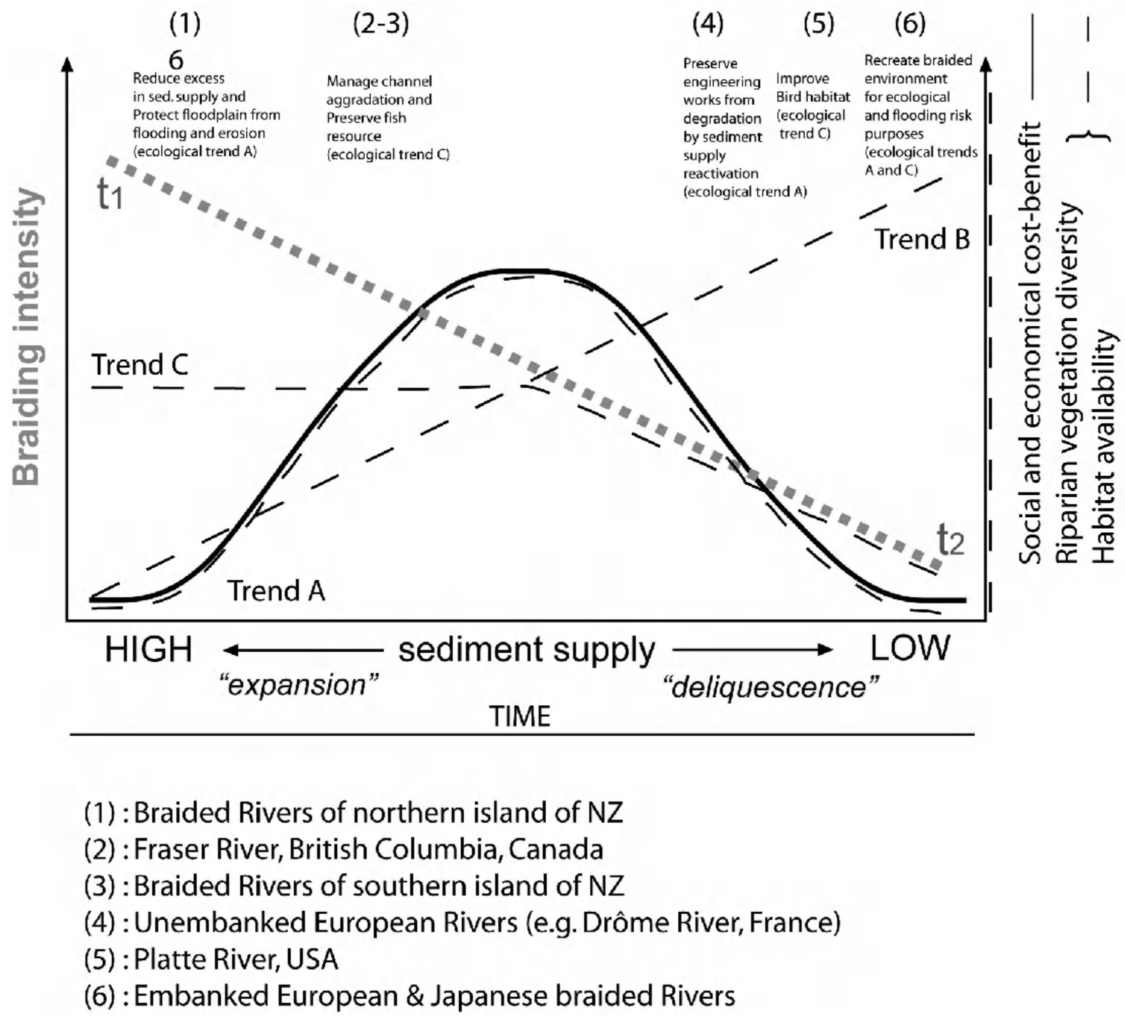

Figure 7. Conceptual framework for understanding the relationship between braiding intensity as a function of sediment supply and the ecological and human benefits derived from braided rivers. From the work of *Piégay et al.* [2006], reprinted with permission from John Wiley and Sons.

the capacity of the riparian vegetation to establish, and the time scale of changes all can vary.

Braided rivers were common in Alpine regions during the last century, even if they have undergone dramatic changes due to human activities. Few braided rivers still exist in northeastern Italy (for example the Tagliamento River) and in southeastern France [*Piégay et al.*, 2009], and there is a need to promote their preservation. In a recent study, *Surian et al.* [2009b] have made an attempt to discuss systematically the possible future scenarios of channel changes, starting from past and present trends and considering potential sediment yield and connectivity. The five selected gravel bed rivers in northeastern Italy have undergone notable channel adjustments in the last 100 years, specifically narrowing by up to 76%, incision by up to 8.5 m, and changing from braided to wandering or single-threaded rivers. Alteration of sediment fluxes has been the main factor driving such channel adjustments and has been due to in-channel mining, dams, or other upstream factors (e.g., torrent control works in the drainage basin). Evolutionary channel trends show that channel recovery is ongoing in several of the selected reaches, since widening and aggradation have occurred over the last 15 to 20 years. Channel recovery has been possible because sediment mining has significantly decreased or ceased along the study reaches, but several constraints on sediment fluxes remain. Notably, dams and other transverse structures reduce connectivity with upstream sediment sources to various degrees.

To set restoration goals, it is worth avoiding the identification of a reference state [*Kondolf et al.*, 2007], which hardly can be defined in fluvial systems with a long history of human impact. Instead, it would be useful to restore geomorphic processes that do not imply restoring the channel morphology of the beginning of the twentieth century, i.e., before the major adjustments took place. Pragmatically, the two main goals could be (1) stopping channel incision where it is still occurring and (2) promoting channel widening and shifting, to enhance the natural corridor mosaics and the associated flora and fauna species.

To assess the potentials and limitations of channel recovery, the analysis proceeded in two steps [*Surian et al.*, 2009b]: (1) the identification of the recent trajectory of channel change and (2) the definition of restoration goals for three different scenarios, basin- and reach-scale interventions, reach scale only, and no interventions. Four categories of channel were defined taking into account recent channel

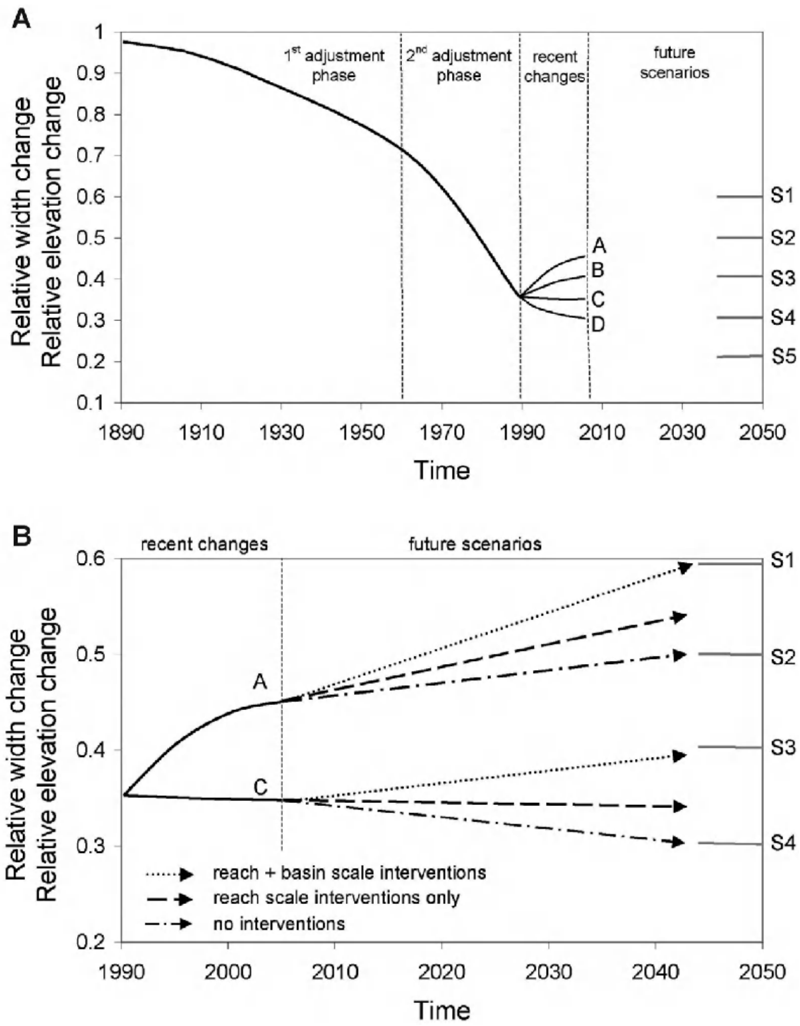

Figure 8. (a) Channel adjustments and possible future scenarios of channel recovery in selected rivers of northeastern Italy. The relative magnitude of width and bed-level changes over the twentieth century and four different trajectories of the last 15 to 20 years (A, high recovery; B, moderate recovery; C, slight recovery or no significant changes in channel morphology; D, no channel recovery) are shown. (b) Future scenarios of channel changes according to different strategies of sediment management. The trajectories of possible future evolution are shown for two categories (A and C) out of the four categories shown in Figure 8a. Modified from the work of *Surian et al.* [2009b], reprinted with permission from John Wiley and Sons.

evolution (Figure 8a): A, high recovery; B, moderate recovery; C, slight recovery or no significant changes in channel morphology; and D, no channel recovery. Restoration goals are then defined in the context of an intermediate time scale (40 to 50 years), assuming simplified boundary conditions. It is assumed that there will be no dramatic changes in land use and human activities within the fluvial system in the next 40 to 50 years (e.g., dams will not be removed). Also the morphological effects of very large flood events (e.g., >100 years return period) are not taken into account because of the great uncertainty in assessing such effects using a simple conceptual model. Five future states are shown in Figure 8a, but the entire range is possible as a set of future endpoints. The ways in which different sediment management strategies (reach- and basin-scale interventions) could affect future channel dynamics were analyzed (Figure 8b). Without any intervention, channel recovery would be possible in those reaches that have a relatively high degree of connectivity with upstream sediment sources or tributaries (e.g., category A in Figure 8b). However, further incision and narrowing could be expected in those reaches where connectivity is low or very low. Reach-scale interventions, such as the definition of an erodible corridor and removal of some bank protections, are the most feasible interventions to allow an increase in the supply of coarse sediment. This should help those reaches suffering from reduced upstream connectivity reach an equilibrium condition (e.g., category C in Figure 8b), while it could lead to significant channel recovery in those reaches where bed load transport has been altered to a minor extent (e.g., category A in Figure 8b). A more substantial channel recovery could be obtained through interventions at the basin scale (e.g., adoption of open check dams and sediment transfer downstream of dams). Even though both reach- and basin-scale interventions may be carried out, it is likely that channels will not recover to the morphology they exhibited in the first half of the twentieth century, since sediment yield and connectivity will remain less than during the nineteenth centuries and the first half of the twentieth century. A further step of this analysis could be the use of numerical models. An example of this approach is the application of a model such as Cellular Automaton Evolutionary Slope And River (CAESAR) [*Coulthard et al.*, 2007], which has the capability to reproduce the main features of the braiding morphology and its evolution to long river reaches and over some decades [*Ziliani and Surian*, 2009]. Numerical modeling also allows the exploration of different scenarios from those analyzed using the conceptual model, which relies on simplified assumptions. For instance, the effects of climate change, very large floods, or remarkable changes in land use at catchment scale all can be assessed through a multiscenario approach that includes a wide range of flow and sediment regimes. This would eventually allow the prediction of more complex evolutionary trajectories than those identified by the conceptual model (Figure 8b).

4.2. Promoting Sustainable Actions by Implementing Management Strategy Designs

A first option for promoting sustainable actions is to develop and implement legislative recommendations and management strategy designs. The identification of a "free space," "functional mobility corridor," "streamway" [*Malavoi et al.*, 1998] or of an erodible corridor [*Piégay et al.*, 2005] is now a procedure that is recommended in French legislation, with clear constraints for implementing bank protections or authorizing mining in floodplains of shifting rivers. The Management Master plan for the Rhône district states that gravel transport and bank erosion are positive processes to be preserved. In such a legislative context, there is a clear expectation in terms of planning tools in order to define actions on reaches where problems are identified. The use of GIS and orthophotographs to characterize the physical character of rivers to locate specific geomorphic features and reaches sensitive to changes or human pressures is one way to target actions at a network-based scale [*Alber and Piégay*, 2011; *Wiederkehr et al.*, 2010]. In Italy, the definition of such a "streamway" has been required for the Basin Authority Plans, but this was generally identified only in terms of flooding hazard, with few examples that used a geomorphological approach. However, the importance of the erodible corridor concept is increasingly recognized, and some examples of its mapping exist (Tagliamento and Magra rivers).

The case of the Magra River (central northern Italy) is an important example developing an overall strategy for promoting sustainable sediment management [*Rinaldi et al.*, 2009], where the identification of a functional mobility corridor was included in a wider management context. The procedure is composed of four main aspects that are described below.

4.2.1. Synthesis step. This is the simplification of diagnostic data to establish a single indicator of geomorphic health. Four indicators were used to summarize diagnostic information: (1) secular bed-level changes (at the scale of about 100 years, i.e., from 1900 to 2006), (2) decennial trend of bed-level adjustments (from 1989 to 2006), (3) bed-level recovery since 1950, and (4) hydraulic sediment budget. From the combination of these four basic indicators, six classes of a geomorphic health index were defined, and three macroclasses were used ((1) prevailing stability-aggradation conditions, (2) intermediate conditions, and (3) past incision, present tendency to incision, and low recovery) to which specific management actions were assigned (see next step).

4.2.2. Strategic step. In the strategic step, proposed actions are based on the previous synthesis. A series of management actions to promote sediment recovery at the scale of the main alluvial channels (Magra and Vara) were defined and associated with each of the macroclasses of channel conditions defined above. These include the following: M1, move sediments trapped upstream of weirs; M2, move in-stream sediments; M3, move sediments accumulated on the floodplain into the channel; M4, carry out a bed load release downstream of dams; M5, move sediments in situations of hydraulic risk (for aggradation); M6, introduce sediments deriving from other reaches; and M7, introduce sediments in situations of risk (for local scour).

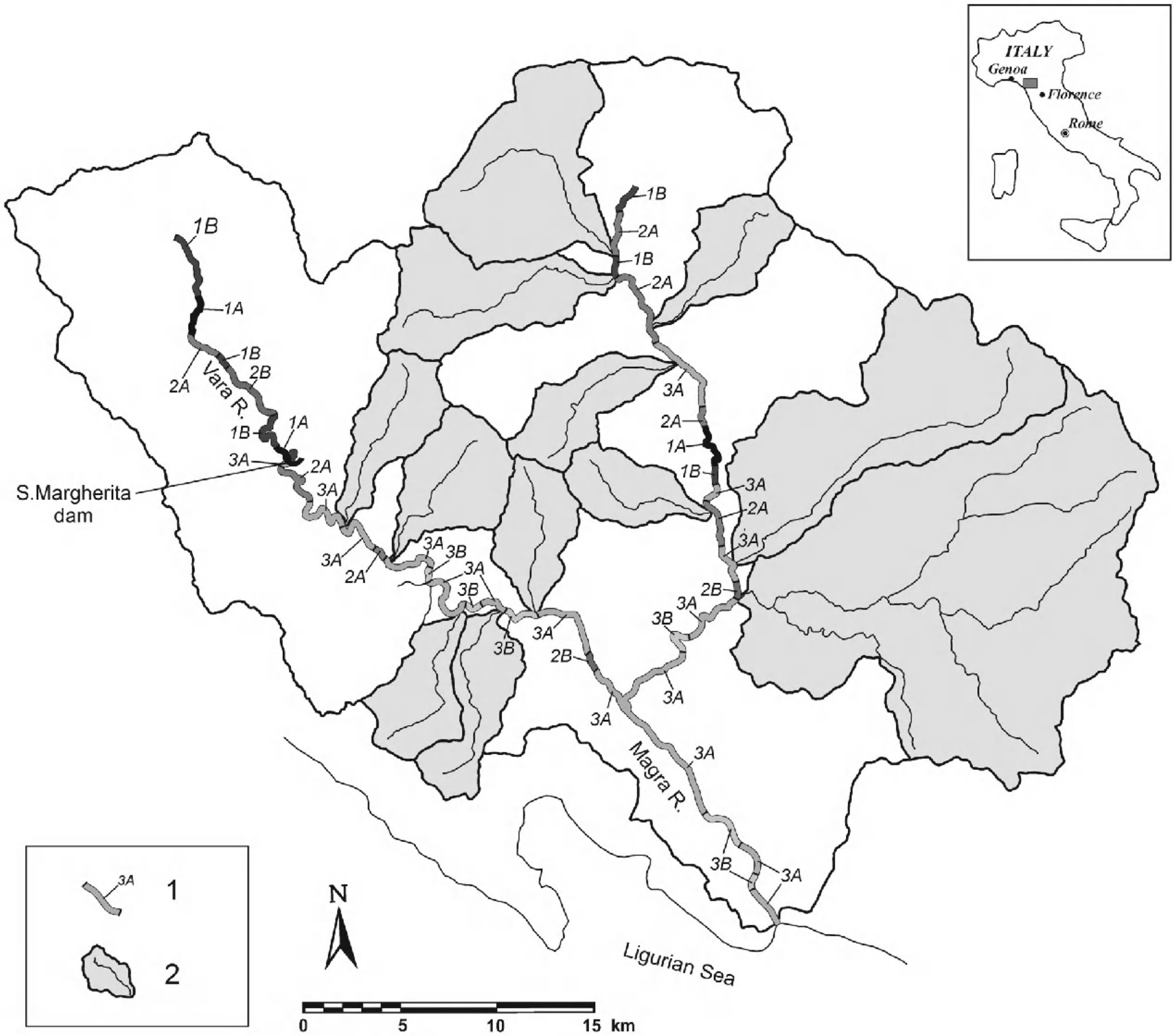

Figure 9. Schematic representation of the main elements included in the "map of strategies for sediment management." Numeral 1 indicates classification of the river segments based on the geomorphic health index. Codes from 1A to 3B, follow a gray scale from black, corresponding to class 1A (highest geomorphic health), to light gray, corresponding to class 3B (lowest geomorphic health). Numeral 2 shows subcatchments selected for potential sediment recharge. The center point of the catchment has approximately the following coordinates: latitude 45°15′N and longitude 9°53′E. Modified from the work of *Rinaldi et al.* [2009], reprinted with permission from John Wiley and Sons.

A series of reaches where the functional mobility corridor encouraged additional sediment supply from eroding banks were identified. Finally, a series of conservation measures (for example, do not stabilize landslides or hillslopes in direct connection with the river channel network) were associated to the catchment areas and features identified as zones of natural sediment recharge. These include the following: C1, do not stabilize landslides; C2, do not stabilize hillslopes in direct connection with the river channel network; C3, do not stabilize eroding stream banks; C4, do not build new transverse hydraulic structures; C5, do not build new longitudinal hydraulic structures; and C6, avoid maintenance of existing hydraulic structures. Obviously, these actions are not applicable in erosion or flood-risk sensitive reaches, such as urbanized areas or areas with particular high-risk elements (single buildings or infrastructure elements).

4.2.3. Practical methodology to promote sediment delivery. This step included two different aspects: (1) management of channel mobility and (2) identification of suitable areas for potential sediment recharge. Regarding the first aspect, this was based on the definition of the functional mobility corridor (or erodible corridor) and identification of the reaches where lateral channel mobility can be allowed and/or promoted. Second, to define an overall plan for sediment management at a catchment scale, some basic evaluation of potential recharge at the subcatchment scale was carried out. A semiquantitative approach was used, similar to that recently applied to the catchment of the Drôme River, France [*Liébault et al.*, 2008], to obtain a classification of the basin areas with relative potential for sediment recharge.

4.2.4. Mapping strategies for a sustainable management of bed sediment at the catchment scale. A "map of strategies for sediment management" was carried out for the Basin Authority of Magra River as a technical tool to be used for future river management. This map synthesized aspects of morphological evolution, sediment budget assessment, and areas of potential sediment recharge as described above, it reports river segments and associated sediment management recommendations, and it identifies suitable areas for potential bed load recharge and associated management actions and/or measures at both network and catchment scales (Figure 9).

4.3. Promoting Restoration Actions

Beyond legislative recommendations and/or management strategy designs, another possible option is to implement restoration actions to repair or improve existing conditions so that the ecological benefits can be increased. In fact, the current critical management situation (i.e., problems related to channel incision, sediment deficit, and associated biodiversity decrease) has progressively made river restoration a challenging issue in France and Italy, and this trend is reinforced by the European Water Framework Directive, which aims to ensure that rivers attain a good ecological status by 2015.

As in other parts of the world, a progressive shift from small-scale interventions toward "process-based restoration" is observed, where there is an aim to restore natural geomorphic processes to promote conditions of self-sustaining physical diversity. Restoration actions have also shifted toward higher energy, bed load-transport-dominated channels (e.g., in the piedmont Alpine areas). In such environments, successful restoration must include the full spectrum of scales and consider the related natural processes and human boundary conditions [*Habersack and Piégay*, 2008].

In France, a significant number of restoration actions have been carried out during the last few decades, mainly focusing on local scales to enhance fish habitats with artificial structures complicating flow velocity, water depth, and grain size conditions, but also at reach scales to restore processes (reflooding, remeandering, recovering of sediment transport, etc.) (Figure 10). Some of these interventions involve specific consideration of morphological forms and processes, including the following [*Habersack and Piégay*, 2008]:

1. Sediment reintroduction or promotion of bed load supply from floodplains, tributaries, and hillslopes is considered, including the removal of bank protection to recreate natural banks and promote sediment recharge. Relevant applications of this type of approach are relative to the Ain River [*Rollet et al.*, 2008], on the Drôme catchment [*Liébault et al.*, 2008], and a feasibility study is actually being conducted on the Rhine downstream from the Kembs Dam on a 45 km long bypassed section.

2. Former channel reconstruction and reconnection are considered. Old channels were typical features along many peri-alpine rivers, but most of them have been disconnected because of channel narrowing and incision. Their restoration is a well-accepted strategy by local authorities, and significant cases of promoting lateral reconnection by dredging former channels have been implemented in the Rhône restoration plan [*Habersack and Piégay*, 2008].

3. Enlarging river space is also a common practice in central Europe. In rivers of Switzerland and Austria with similar characteristics to the Alpine rivers of northern Italy and southeastern France, much channel widening of previously embanked reaches have already been performed for safety purposes as well as for ecological improvement.

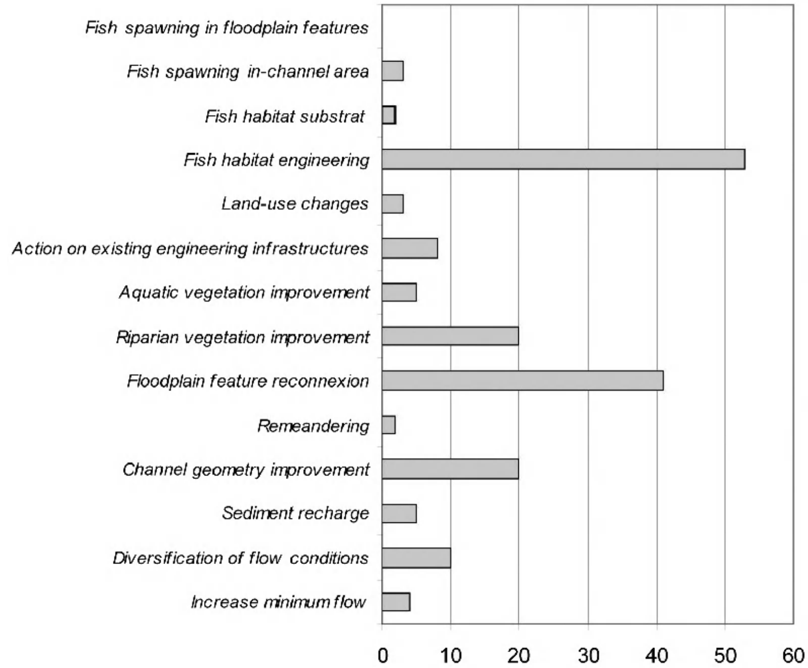

Figure 10. Summary of restoration actions censed on the web for the southeastern France: Rhône-Alpes and PACA regions. Data from *Morandi* [2009]. $N = 176$ actions.

Floodplain lowering is also becoming an issue promoted in a win-win perspective, aiming to improve riparian ecosystems along incised river channels with a dry and disconnected floodplain and to cut peak flows downstream by increasing the flood retention capacity of such corridors.

In Italy, notwithstanding increasing experience and evolving approaches, river restoration using geomorphic principles is still in its infancy, emphasizing analysis (i.e., assessment of problems, proposition of strategies, and interventions based on comprehension of processes, etc.) rather than implementing specific interventions. Recently, *Rinaldi and Gumiero* [2008] carried out a brief overview of a series of research projects (Ombrone and Pesa in the Arno River catchment; Vara, Magra, and Panaro rivers) that propose strategies to reconcile flood risk and restoration objectives. All the case studies are similar in that they involve a common morphological channel evolution (channel incision, narrowing, and sediment deficit). The proposed strategies of management and restoration take into account the past channel evolution and are compatible with present trends of channel adjustments. In particular, the option of promoting channel recovery, by allowing natural channel adjustments, rather than morphological reconstruction, is identified as the best strategy, being that the sediment load and stream power of these rivers is sufficiently high.

Several challenging issues and open points still remain for a wider application of process-based morphological restoration, including (1) feasibility of such interventions, in relation to possible negative consequences in terms of other uses and risk conditions; (2) sustainability and self-maintenance of reactivated processes or recreated forms in the long term; and (3) ecological benefits of such measures are still under investigation, and there is an urgent need to link ecological to geomorphological processes.

5. CONCLUSIONS

This chapter presented an overview on recent progress in using geomorphic approaches to river management and restoration by providing a series of case studies and applications to some Italian and French rivers.

Knowledge of channel evolution, temporal trajectory of adjustments, and causes and sensitivity to changes provides the basis for defining channel and sediment management strategies. Most Italian and French alluvial rivers considered here are characterized by a similar history of human disturbance and trends of channel adjustment, with a historical phase of aggradation followed by a period of intense channel incision and narrowing, yet these river systems differed in terms of their controlling factors and evolution scenarios.

Cases of recent widening are also observed along some of the Italian and French rivers, and this widening appears to be associated with the cessation of intense gravel mining (Italian rivers) and/or with more intense floods that occurred in the 1990s. The sediment deficit resulting from human pressures that occurred between the 1940s and the 1960s within the Drôme and Ain catchments (France) demonstrates that the adjustment process at the regional scale is just beginning, and will probably continue downstream for many years to come. These types of channel adjustments have led to an increasing need for sustainable management of bed load transport and other physical processes.

The quantification of bed load transport and bed sediment budget, therefore, is fundamental to understanding the balance between sediment transport and delivery. Various examples of a morphological approach for sediment budgeting have been provided. For the Brenta River (northeastern Italy), where the spatial variations of bed load along the reach were analyzed, most of the material available for transport was sourced from local erosion. and only a small proportion was derived from the drainage basin. The analysis also showed that bank erosion was the dominant process from 1984 to 1997, contributing 83% of the total erosion in the study reach. For the Magra and Vara rivers (central and northern Italy), sediment budgets were constructed using sediment transport equations for each of a series of discrete subreaches to classify their tendency to incision or aggradation. Examples of bed load yields for select rivers of the Southern French Prealps were used to establish a regional law linking catchment size and the mean annual bed load transport.

Finally, previous knowledge for designing regional visions, strategies, and targeting actions were used to assess these rivers. Examples were presented to illustrate how scientific approaches and management strategies can be used to promote a regional vision, based on a geomorphic diagnosis, and identifying associated tools for planning, targeting, and regulating actions, and to promote sustainable actions at the local/catchment scale, aimed at preservation, mitigation, or restoration of physical processes and associated channel features.

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Hydraulic Modeling of Large Roughness Elements With Computational Fluid Dynamics for Improved Realism in Stream Restoration Planning

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Many stream restoration design procedures are based on user experience in distributing standard stream design features into stream channel types based on a stream classification scheme. Computational fluid dynamics (CFD) models, increasingly used to represent stream flow fields, offer a more quantitative path forward. However, CFD models, in practice, parameterize roughness on too large a scale and therefore do not explicitly represent discrete features such as large rocks and large woody material whose placement is the focus of stream restoration activities. The Stream Habitat Assessment Package (SHAPE), made possible by rapid advances and availability of high-performance computing resources and increased sophistication of both in-house and commercial software, overcomes barriers that prevent the routine use of CFD modeling in stream restoration planning. Capabilities of SHAPE that improve stream restoration planning include (1) realistically representing natural streambeds from potentially coarse sets of field measurements, (2) easily deforming the streambed surface with a virtual excavator, (3) selecting complex objects from a library and embedding them within the surface (e.g., rocks and fallen trees), (4) successfully meshing the channel surface and its surrounding volume in accordance with established mesh quality criteria, and (5) sufficiently resolving flow field solutions. We illustrate these capabilities of SHAPE using a coarse set of field data taken from one of four study sites along a 1.5 mile stretch along the Robinson Restoration project of the Merced River, California, along with respective challenges, solution strategies, and resulting outcomes. Flow field solutions are conducted using parallelized finite element/volume solvers.

1. INTRODUCTION

Effective stream restoration design is important because restoration projects are expensive, publicly financed, and therefore the focus of extensive stakeholder involvement. Many stream restoration design procedures are, at best, semi-quantitative. That is, they are based on user experience in distributing standard stream design features into stream channel types based on a stream classification scheme. The present prescriptive nature of present stream restoration procedures is accompanied by little or no postconstruction monitoring, lack of agreement on what constitutes success, and ambiguity over the time scale that restoration benefit should be calculated. In addition to performance over a limited flow

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range, effective stream restoration should also include effects on biogeochemical cycling and coupled effects on stream biota. Commonly, stream restoration projects are developed with a conceptual understanding of larger scale issues but often are implemented on local (reach scales). This ecological myopia cannot be addressed without considering the physical effects of stream restoration on material (e.g., sediments, organic matter, and nutrients) mobilization, transport, and deposition. These physical effects can be addressed by computational fluid dynamics (CFD) modeling.

Historically, standard stream channel design relied on one-dimensional (1-D) models that represent the river as a series of cross sections. As computational resources have grown, examples of 2- and 3-D models have increased [*Crowder and Diplas*, 2006; *Tayefi et al.*, 2007]. These more recent CFD models solve the Navier-Stokes equations or derivatives thereof and provide detailed flow field information from which physical and biological design outcomes may be evaluated. However, there are at least two practical problems with using multidimensional models for stream restoration design. First, such models require extensive data on channel and bank topography and desired future topography resulting from channel alternations. Such data, especially in early design stages when multiple alternative designs may be under consideration, are rare. More importantly, suitable biological models through which CFD output can be interpreted are not common.

One successful approach used to convert the voluminous output of CFD models into a biologically relevant context is the Eulerian-Lagrangian-agent Method (ELAM) [*Goodwin et al.*, 2006]. An ELAM is based on the premise that animals have a sensory system and neural processing capability to separate the hydrodynamic signatures of different sources of flow resistance and thus infer the characteristics of the solid features of a stream channel from their hydrodynamic signatures [*Nestler et al.*, 2008]. Moving water in natural channels is slowed by flow resistance arising from several sources including skin friction or wall-based resistance and form drag [*Yen*, 2002]. Skin friction characteristically exhibits velocities at the boundary equal to zero that increase as distance from the boundary increases. Alternatively, form drag is associated with large roughness elements that extend from the bottom up into the flow. Form drag is related to pressure drag on the surface and is an important component of the overall flow resistance in rivers [*Buffington and Montgomery*, 1999; *Leopold et al.*, 1964]. For example, form drag from large wood can account for the majority of overall flow resistance, while physically occurring in a small portion of the wetted channel [*Manga and Kirchner*, 2000]. Similarly, large rock, such as in step pool streams, can also dominate the overall flow resistance budget [*Wohl and Thompson*, 2000]. An ELAM can separate form and friction drag and is very sensitive to the flow field produced by large roughness elements. Thus, methods for accounting for large roughness elements are important.

In stream restoration, roughness elements such as large wood and large rock that produce substantial flow resistance are routinely placed in streams as important habitat features. In CFD models, large rock and particularly large wood are not routinely explicitly represented so that their contribution to the overall flow resistance budget is captured indirectly through a Manning's n or other roughness parameter. The importance of accurate form drag representation from hydraulic [*Nicholas*, 2005] and biological [*Goodwin et al.*, 2006; *Nestler et al.*, 2008] perspectives highlights the need for techniques that allow stream restoration design to explicitly capture form drag in CFD models. These roughness elements must be explicitly represented by CFD models if the model is to be used to evaluate the effectiveness of alternative stream restoration designs to increase habitat value.

2. FLOW PATTERN AND DISCRETE ELEMENT MODELING

The importance of accurately representing discrete roughness elements in flow field simulation is well established. For example, CFD models have been used in discrete element modeling of geometric shapes including cylinders and half spheres [*Christoph and Pletcher*, 1983; *Taylor et al.*, 1985]. It is evident that discrete roughness elements cause a hydraulic pattern that is substantial in size relative to the spatial domain of the model. We illustrate this using a 1 m square times 4 m tall column in a rectangular channel (8 m wide × 4 m deep by 120 m long) with a mean velocity of 1 m s^{-1}. The velocity gradient is the absolute value of the change in velocity over three dimensions. Upstream from the column, velocity is dominated by wall-based drag meaning that the velocity gradient is steep proximate to the channel margin, and velocity increases with distance from the margin. On approach to the column, form drag increases and is evident as an increasing velocity gradient. Simultaneously, velocities increase on approach to the column (Plates 1a and 1b). Thus, information on the velocity and velocity gradient provide a pattern that signals the location of the channel margin and locations of large roughness elements within the channel. In turn, this pattern suggests a hypothesis whereby fish can determine its position proximate to the channel margin or large roughness elements and then choose an appropriate reaction based on that information [*Smith*, 2003; *Goodwin et al.*, 2006; *Nestler et al.*, 2008].

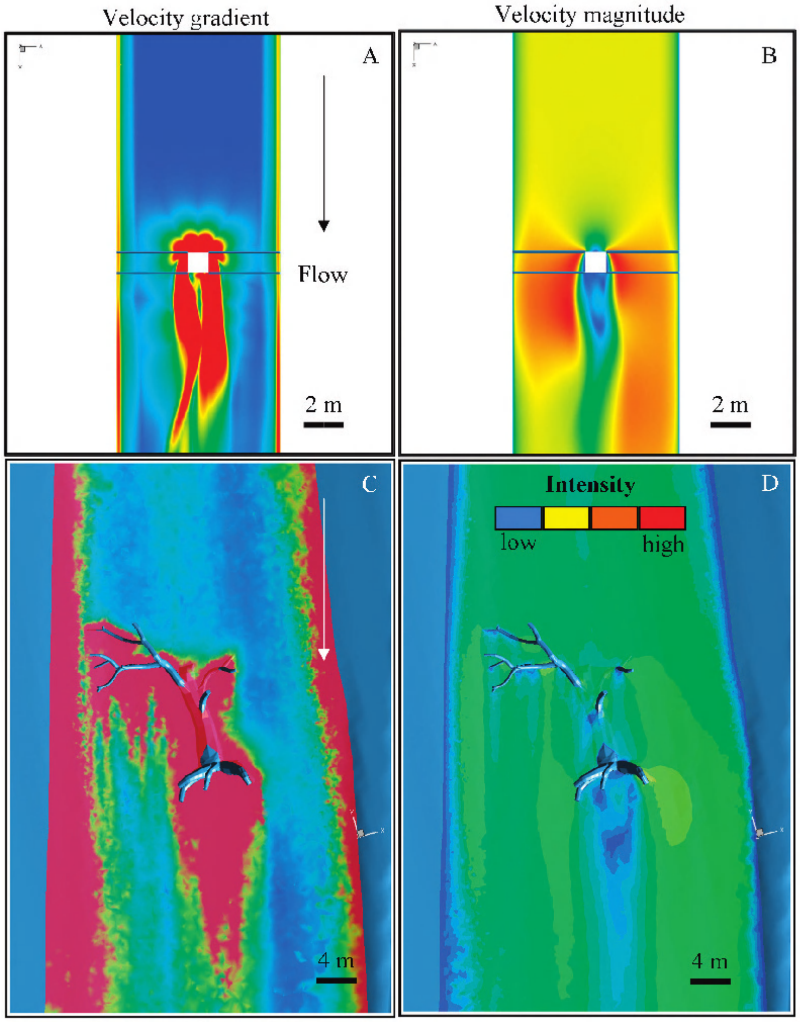

Plate 1. Comparison of velocity gradient and velocity magnitude for (a and b) a 1 m square column and (c and d) large wood.

Discrete element modeling in rivers is an emerging field, made possible by lower computational cost, and higher data resolution, that allows modeling of fine scale in rivers. Discrete element models of gravel bed roughness have been completed [*Olsen and Stoksteth*, 1995; *Lane et al.*, 2002], but discrete element modeling of complex shapes in rivers is not yet common. The size and distortion of the flow pattern that results from a simple discrete element, such as a column, should also be indicative of the impact of more geometrically complex objects such as large wood or rock added as part of stream restoration activities (Plates 1c and 1d). We also anticipate that complex forms in a channel have important ecological implications through local erosion, transport, and deposition of material.

Developing a computational mesh that includes a detailed mesh representation of large roughness elements is challenging. We developed an approach, termed the Stream Habitat Analysis Package (SHAPE) that facilitates stream restoration design and alternative analysis using CFD models coupled to numerical representations of animal response to hydraulic pattern. We envision that this approach will be useful to engineers and biologists involved in planning stream restoration strategies. In addition, new approaches for representing large roughness will assist modelers in representing restoration designs. Coupled with biological models, these methods will produce information that assist hypothesis generation and testing about alternative stream restoration designs.

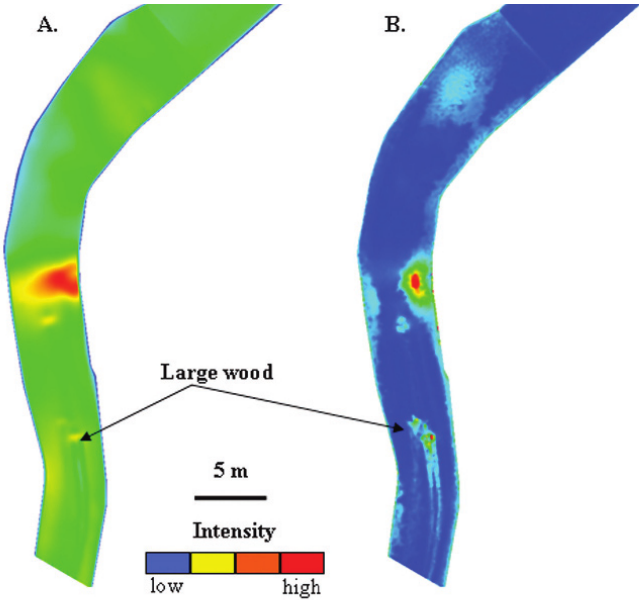

Plate 2. (a) Resulting velocity and (b) velocity gradient based on the S1 reach geometry and containing large woody debris. Colors represent levels of velocity magnitude and velocity gradient.

3. APPROACH

For a given ecosystem analysis, the temporal and physical scales applicable to fluid dynamics, ecology, and fluvial geomorphology should ideally converge. The challenge associated with determining this scale range, as well as the requisite information exchange among the related scientists and engineers can be, unsurprisingly, overwhelming. This challenge is particularly evident in the field of stream restoration projects wherein engineers, ecologists, and fluvial geomorphologists all must interact during project planning and execution. This interaction is vital for the increased success rate of individual stream restoration projects. Within the United States alone, approximately 15 billion dollars, or an average of $100,000.00 per restoration kilometer [*Malakoff*, 2004], have been spent on stream restoration in the last decade, and the pace of spending is expected to increase [*Bernhardt et al.*, 2005]. Unfortunately, due to the lack of proper coordination, planning, and monitoring, many of these projects do not fully meet their goals. Predictably, the engineering and biological disciplines are deeply divided on what better planning and implementation actually entails.

In conjunction with better planning and coordination among disciplines, a representative stream restoration procedure must also operate at an appropriate range of temporal and physical resolutions. The computational fluid dynamicist, for example, must ensure that the computational grid over which the equations of motion are solved are sufficiently resolved in areas of high flow gradients and that grid independent solutions result. The ecologist, interested in better understanding the strategy employed by fish to move within the fluid environment and its likely dependence on flow resistance, will likely desire sufficient model resolution to adequately describe the signatures of both form and friction drag. The fluvial geomorphologist interested in landform evolution may have a particular interest in flow field patterns associated with increased vegetation or topological variations. Temporal interest may concern climate variation or changes in water diversion resulting from increased water demand.

SHAPE represents the combined, collaborative efforts from a team of authors having backgrounds in the diverse fields of engineering, computational science, and ecology. Team diversity was critical to deriving realistic, computational models of naturally occurring stream and river systems appropriate to the aforementioned scales required of multidisciplinary, numerical, stream restoration modeling. Several of the key technical challenges include the ability to (1) create a sufficiently resolved surface mesh from potentially coarse sets of field data, (2) freely deform the streambed surface geometry to describe channel features not described in channel cross-section data, (3) embed large roughness elements and mesh the resulting surface/volume in accordance with representative length scales and mesh quality requirements, and (4) obtain a sufficiently resolved flow field solution.

In the remainder of this chapter, we describe how we address each of these challenges. To demonstrate the pragmatic nature of the current capabilities, and for environmental and geographical context, the authors use field measurements taken from one of four study sites (S1) along a 1.5 mile stretch of the Robinson Restoration project of the Merced River, California [*U.S. Fish and Wildlife Service*, 2005]. The goal of this restoration effort was to reverse the decade-long decline of fall-run Chinook salmon by creating both spawning and rearing habitat.

3.1. Creation of Resolved Surface Mesh From Coarse Field Data

The S1 site has a mean length and width of 311 and 46 m, respectively, producing a surface area of about 14,306 m^2, with a mean site bed slope of approximately 0.44% (Figure 1a). Data collected at the site in April 2003 included the following: water surface elevations (measured to the nearest 0.003 m), wetted streambed elevations, dry ground elevations at points above bankfull discharge (nearest 0.009 m), and mean water column velocities at 0.6 the local

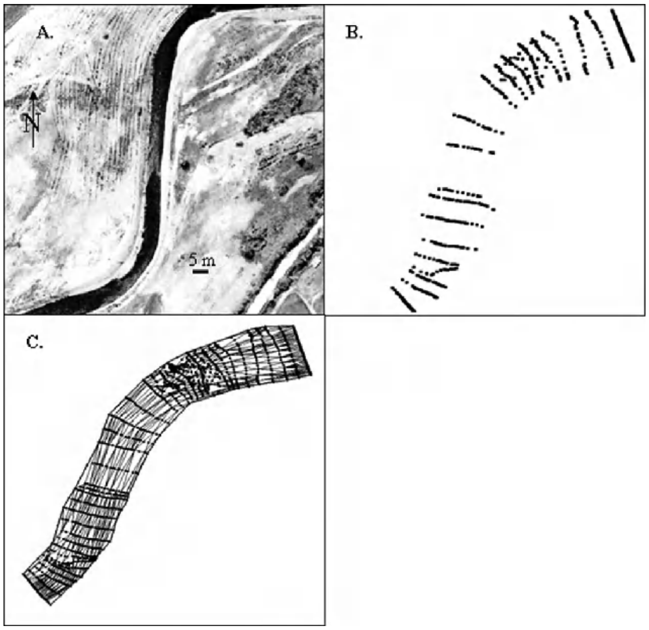

Figure 1. (a) Aerial photography, (b) coarse cross-sectional data, and (c) the resulting triangulation of the Merced River S1 reach.

depth. A total of 448 topological measurements were collected within the site area to produce a Cartesian point cloud having a density of about one point per 32.5 m^2 to represent the model domain (Figure 1b). A Cartesian point density of about 1 per 32.5 m^2 is adequate to represent only the most dominant topological features. The spatial coarseness of the data is inadequate for the meshing requirements appropriate to a CFD model to adequately describe either spawning or rearing habitat, particularly in areas of high flow gradients. Topological field data (shown in the *x-y* plane with depth corresponding to the *z* coordinate) and the corresponding, 2-D Delaunay triangulation (Figure 1c) are displayed using 3ds Max (Autodesk, Inc., 3ds Max, available at www.autodesk.com/3ds Max, 22 February 2008).

To address this inadequacy, a new grid was created using the "conform" utility of 3ds Max. The unique functionality of this utility allows the user to generate a new mesh that retains the geometric contours of the original by overlaying a planar surface over the initial contoured surface and "fitting" or conforming it to match the desired contours of the original geometry (Figure 2a). The new mesh duplicates the geometry of the original, but with the advantage of being completely independent of the original coarse set of data so that the grid resolution of the original mesh can be improved. This technique is not equivalent to obtaining new measured field bathymetric data and thus introduces error.

3.2. Free-Form Deformation of Streambed Surfaces

The increased resolution of the mesh produced from the previous step allows the user to render features at a resolution finer than in the original mesh using free-form deformation (FFD) software. This software avoids the time and expense associated with gathering new field data that could be prohibitive. The following text outlines the FFD method of *Sederberg and Parry* [1986].

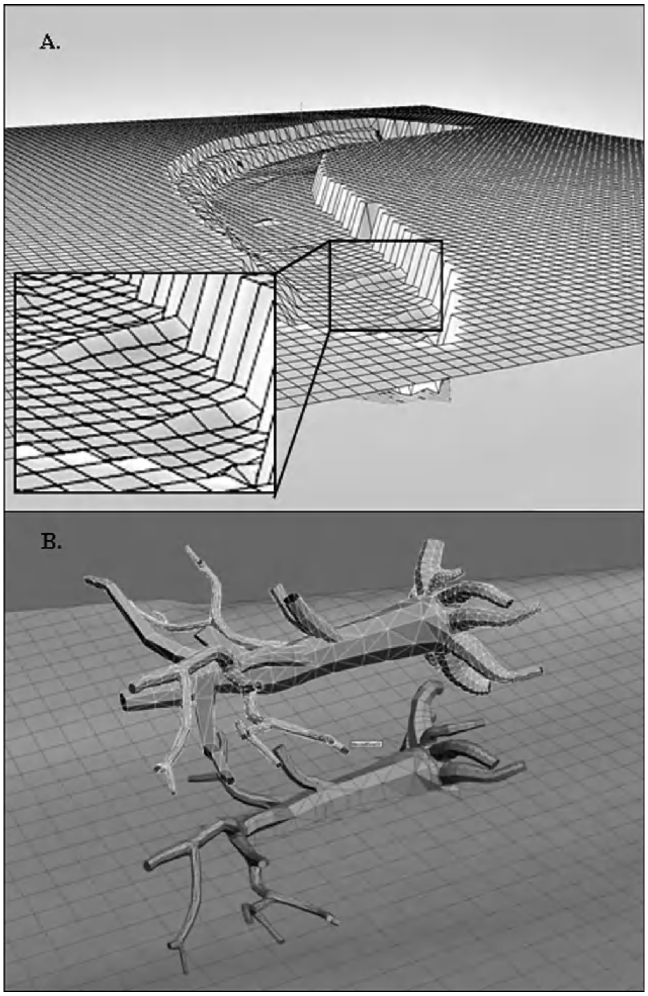

Figure 2. (a) The conform utility in 3ds Max generated the bathymetry with more elevation points while retaining the original geometry. Free-form deformation (inset) was used to add large roughness along the channel bottom. (b) Addition of discrete large roughness, in this case a root wad into the computational mesh. Mirror image is the reflection of the large wood, which helps to visualize the interface between the stream bed bathymetry and the large wood mesh.

The original, 3-D (x,y,z) coordinates of an object are mapped ($R^3 \rightarrow R^3$) to a local parametric coordinate system (s,t,u) described as

$$\mathbf{X}(s,t,u) = \mathbf{X}_0 + s\mathbf{S} + t\mathbf{T} + u\mathbf{U}, \tag{1}$$

where $\mathbf{X}_0$ is a prescribed origin, and $\mathbf{S}$, $\mathbf{T}$, and $\mathbf{U}$ are vectors lying along the object volume with magnitudes corresponding to volume edge dimensions. The lattice space volume is defined by an array of control points ($\mathbf{CP}_{ijk}$) with locations:

$$\mathbf{CP}_{ijk} = \mathbf{X}_0 + \left(\frac{i}{l}\right)\mathbf{S} + \left(\frac{j}{m}\right)\mathbf{T} + \left(\frac{k}{n}\right)\mathbf{U}, \tag{2}$$

where l, m, and n represent the number of control points along the S, T, and U axes, respectively, and $0 \leq i \leq l$, $0 \leq j \leq m$, and $0 \leq k \leq n$.

The lattice space is deformed by first moving the control points from their undisplaced lattice positions. Next, a tricubic, Bezier hyperpatch (a 3-D extension of a simple Bezier curve) is applied to each control point of the lattice space as defined by the tensor product:

$$\mathbf{Q}(s,t,u) = \sum_{i=0}^{l}\sum_{j=0}^{m}\sum_{k=0}^{n} \mathbf{CP}_{ijk} B_{i,l}(s) B_{j,m}(t) B_{k,n}(u), \tag{3}$$

where $B_{i,l}(s)$, $B_{j,m}(t)$, and $B_{k,n}(u)$ are the Bezier Basis functions of degree l, m, and n, respectively, and found from

$$B_{\nu,n}(x) = \binom{n}{\nu} x^{\nu}(1-x)^{n-\nu}, \quad \nu = 0, \ldots, n. \tag{4}$$

Figure 2a (inset) highlights the results of applying the above procedure to a point cloud of 28,869 points to produce large roughness along the channel bed. A total of 64 control points were used to dictate the relative amount of surface displacement.

Owing to the inherent value associated with object deformation, FFD functionality, as well as several other deformation techniques, including nonuniform B-splines and the direct manipulation of object vertices, are widely available in a variety of softwares such as 3ds Max (Autodesk, 3ds Max, available at www.autodesk.com/3ds Max, 22 February 2008) and Maya (Autodesk, Maya, available at www.autodesk.com/maya, 22 February 2008). Its utility allows for the creation of any number of global or localized streambed deformations, including depressions, elevations, stream embankments, and many other potential topological formations.

3.3. Addition of Large Roughness Elements and Remeshing

The utility of 3ds Max also allows for the creation and embedment of large roughness elements (e.g., rocks, large woody debris, root wads, etc.) into the newly refined surface geometry. The roughness elements may come from any number of sources including commercial or in-house database libraries, 3-D laser scan images, or manual derivations. These imported objects may be further deformed and manipulated as desired using FFD. The Boolean embedding process consists first of subtracting out streambed surface elements occupying (either completely or partially) locations in which the roughness element(s) will be embedded. Next, the roughness element is joined to the underlying surface through the manual manipulation of nodes corresponding to both the roughness element and the underlying surface (occasionally the creation of additional nodes is required). Finally, the transition elements are refined and smoothed as necessary to create the appearance of a natural transition (Figure 2b).

Unfortunately, the resulting surface mesh generated in this manner with 3ds Max remains unsuitable for most CFD applications. The absence of mesh grading options, mesh quality checks, and diversity of 3-D element types, requires that the new surface be remeshed using appropriate grid generation software. From 3ds Max, the surface geometry is exported in one of several industry standard geometry formats (IGES, STEP, Parasolid, ACIS, and STL) to GAMBIT (ANSYS, Inc., GAMBIT user's guide, Lebanon, New Hampshire, 2007). Once imported, a set of "cleanup" tools are used to correct defects that may exist in the geometry. Here, the surface and surrounding volume (as applicable) are meshed with hexagonal, tetrahedral, or hybrid elements. We used a fixed lid approach and specified the water surface elevation. GAMBIT's size function application allows the user to specify localized regions of mesh refinement, such that surrounding cells gradually coarsen through a functional approach. In this manner, face geometries having very small length scales govern the initial size function parameter.

3.4. Flow Field Solution

The fourth and final challenge, that of obtaining a sufficiently resolved, 3-D flow field solution, is clearly dependent upon several factors including the successful outcome of the three preceding steps. In addition, a successful CFD simulation will necessarily require proper boundary and initial condition assignment, material property assignment, grid independence, sufficient discretization accuracy of convective, diffusive and temporal terms, turbulence considerations, surface roughness, and numerous other solver-related inputs. Finally, for quantitative accuracy and completeness, the numerical simulation should be properly validated against experimental results. No field data were available for this study, so our results are intended to highlight our methods' applicability to modeling large roughness elements. The above considerations, particularly the grid resolution required to adequately resolve the new geometry and flow field quantities, may require several millions or 10's of millions of grid elements making the use of high performance computer (HPC) resources essential. A Cray XT3 supercomputer using upwards of 16 nodes (32 processors) and runtimes of up to 12 h were typical.

For the S1 application, we use two different parallel solvers, Fluent (ANSYS, Inc., FLUENT 6.3 user's guide, Lebanon, New Hampshire, 2006) and the Adaptive Hydraulics Model (ADH) [*Howington et al.*, 1999]. Plates 2a and 2b show velocity and velocity gradient contours for the S1 geometry along a top planar surface (approximately 6/10 depth) subsequent to the addition of two large roughness elements composed of a root wad and boulder. The simulation was conducted using a Reynolds Averaged Navier Stokes, *k*-epsilon turbulence closure model with a fully developed inlet velocity condition (using a maximum inlet velocity of 1 m s^{-1}). Fully developed flow at the inlet was obtained through the use of an upstream, constant area channel of length approximated as: 5*Average stream width. Model output of the mean flow shows expected features such as stagnations, wakes, and, in the case of the embedded boulder, an increased velocity along the lateral edges.

After mesh quality checks of face skewness, aspect ratios, stretch, and cell volumes, especially surrounding embedded features (Figure 3a), the resulting mesh consists of approximately 1.5 million tetrahedral surface elements. Inspection of the vector results of Figures 3b and 3c reveals several localized vortices generated directly downstream of the cylindrical branches of the root wad. This detail suggests that indirect estimates of the impact of form drag on local hydraulic features may not capture the detail needed for realistic ecological and biological simulations.

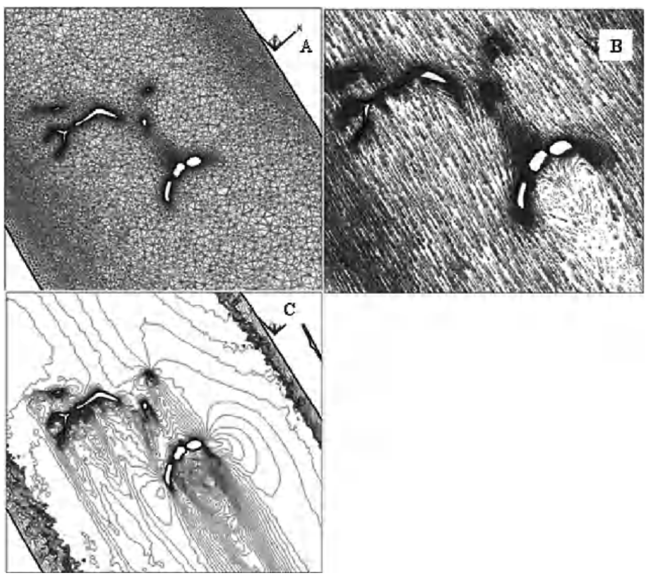

Figure 3. (a) Computational mesh around large wood, (b) velocity magnitude vectors illustrating recirculation region downstream of large wood, and (c) velocity isopleths.

Adding complex objects to the CFD model mesh was challenging and required use of recently developed numerical approaches. However, the virtual environment that was created depicted the flow field at a level of resolution consistent with the scales likely important to understand spawning or rearing habitats, provide great detail about the flow field, and cost less than direct field measurement.

4. CONCLUSIONS

We present a procedure that decreases the effort required to model discrete elements whether added to the computational mesh directly or through FFD. This capability is important to more fully evaluate alternative stream restoration strategies at the scale of their effects on aquatic biota. This level of resolution allows CFD model output to be coupled to biological response methods like the ELAM that could be used for stream restoration evaluation.

One of the primary challenges is that stream restoration design often occurs with little actual measured channel bathymetry [*Merwade et al.*, 2008]. In addition, what channel bathymetry may be available does not routinely include large roughness as part of the data set. Thus, CFD models developed from field data often have to overcome coarse resolution data and account for some of the large roughness elements through calibration. However, since there is a link between large roughness elements and the resulting flow pattern and biological response, not explicitly accounting for sources of form drag means that the resulting CFD model may not adequately reproduce flow pattern of interest to biological analyses.

Incorporating discrete elements, particularly complex representations of large wood and rock is an evolving technology. We present methods that ease (1) the creation of a sufficiently resolved surface mesh from a coarse set of field data, (2) the creation of stream features not described in cross-sectional data using free deform software, (3) embedding large roughness elements and remeshing the resulting surface/volume in accordance with representative length scales and mesh quality requirements, and (4) obtaining sufficiently resolved flow field solutions using HPC resources. Aiding in the solution process, the authors were greatly assisted by a variety of different software platforms including 3ds Max, Gambit, Fluent, and ADH.

Geometric refinement, manipulation, and embedding tools used in conjunction with established meshing practices will be applicable to more than just these particular applications. Indeed, the procedures outlined in this chapter may be applied to any number of modeling efforts containing coarseness in field data and/or the need to freely manipulate or alter existing geometries in some manner.

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Design Discharge for River Restoration

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Selecting a design discharge is a critical stage in a wide range of river restoration approaches and tasks but is not straightforward in practice and rarely involves following any of the several standardized procedures suggested in the literature on stable channel design because the data required are simply not readily available for most project sites. This chapter reviews the scientific bases of three popular candidates for representing the geomorphologically important dominant discharge that can be adopted as a design discharge for channel restoration: the bankfull discharge, a discharge of specified recurrence interval, and the effective discharge. The chapter goes on to assess how the strengths and weaknesses, inherent to their derivation and application, play out in practice. Experience shows that effective discharge analysis has considerable potential for further advances in computational methods that could provide improved insights into the morphological significance of an effective range of flows, enabling restorers to incorporate not one but a series of nested design discharges into their restoration plan, enhancing both geomorphological sustainability and ecological integrity. It is increasingly recognized that the primary challenge in selecting a suitable design discharge for river restoration lies in accounting for uncertainty in future flow and sediment regimes, associated with global warming and ongoing changes in watershed land use, by making sufficient allowance for restored channels to adjust within their restored, functional floodplains, while maintaining the dynamic equilibrium necessary to conserve key species and ecosystems.

1. INTRODUCTION: DESIGN DISCHARGE IN THE RIVER RESTORATION PROCESS

Designing dynamically stable channels with mobile bed materials and adjustable banklines requires that a range of complex scientific and technical issues are addressed by the project design team and is recognized as being one of the most difficult challenges in river restoration [*Shields*, 1996]. Additionally, the requirement for many restored rivers to support high biodiversity and good aesthetics, while simultaneously meeting objectives for flood control, land drainage, and navigation often imposes constraints on the design outcomes [*Brookes*, 1987; *Natural Resources Conservation Service (NRCS)*, 2007].

In naturally stable, alluvial rivers, the dimensions, geometry, and sediment features of the channel are not designed but evolve over time in response to complex interactions between the sequence of flow and sediment transport events

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that actually occurs and the boundary sediments and vegetation that resist morphological change. It is therefore the magnitude, duration, and sequencing of flows that entrain, transport, and deposit boundary sediments that are the primary driving parameters responsible for molding channel morphology and sediment features over time [*Lane*, 1955]. It follows that it is the diversity of flow and sediment transport events that ultimately provides a broad and dynamic assemblage of physical habitats.

Recognizing the multifunctional objectives of channel restoration projects, and the importance of channel evolution, best practice guidance recommends that careful consideration of the hydrological and sediment regimes should be central to channel restoration design [*Soar and Thorne*, 2001]. In theory, it would seem appropriate to apply deterministic equations to predict the stable geometry of a self-formed, alluvial channel as a function of the full spectrum of sediment-transporting flows for design purposes, but in practice, the assumptions required to overcome mathematical indeterminacy and uncertainty in modeling sediment transport processes remain major concerns [*Petts*, 1995]. To counter this, channel restoration design methods have been developed that embrace concepts of dynamic equilibrium and channel stability by attempting to match the sediment supply from upstream to the transport capacity of the restored channel [e.g., *Shields*, 1996; *Soar and Thorne*, 2001; *NRCS*, 2007; *Shields et al.*, 2003, 2008]. These approaches employ a single design discharge in the initial design specification but, importantly, recommend testing the performance of the proposed channel geometry against the full range of sediment-transporting flows as a closure loop in the design process [*Soar and Thorne*, 2001].

Despite the known limitations of using a single flow to represent the geomorphic effects of the range of flows actually experienced by a channel, the fact is that selecting an appropriate design discharge currently remains an essential step in an increasingly wide range of channel restoration design approaches and tasks. Examples include application of (1) downstream hydraulic geometry or "regime" type equations for stable channel geometry [e.g., *Hey*, 1997; *Federal Interagency Stream Restoration Working Group*, 1998; *Soar and Thorne*, 2001], (2) the "Natural Channel Design" method [*Rosgen*, 1998, 2006a, 2006b; *Hey*, 2006; *NRCS*, 2007], (3) analytical techniques based on simultaneous solution of flow resistance and sediment transport equations [e.g., *Copeland*, 1994; *Shields*, 1996; *Shields et al.*, 2008; *Soar and Thorne*, 2001], (4) empirical methods for laying out restored meanders [e.g., *Dury*, 1976; *Schumm*, 1967, 1968], (5) sizing riffle sediments in restored channels based on tractive force analysis [e.g., *Newbury and Gaboury*, 1993], and (6) conducting postproject channel stability assessments based on stream power screening [*Brookes*, 1987] or hydraulic geometry analysis [e.g., *Thorne et al.*, 1996].

The task of identifying the appropriate design discharge is not straightforward. The textbook scenario of utilizing a record of measured discharges from a nearby gauging station as the basis for deriving a design discharge is seldom possible in practice, as hydrometric stations are sparsely distributed along main stem rivers, and many tributaries are entirely ungauged. The task of specifying a design discharge therefore rarely involves following a standard procedure as the data required are simply unavailable. While it is possible to "synthesize" a flow distribution for an ungauged site by transferring data from other gauged sites, this inevitably introduces further uncertainty concerning the reliability of the resulting design discharge and confidence in the suitability and sustainability of the restored channel morphology.

Common to most of these approaches is the adoption of the "dominant discharge" concept that the spectrum of sediment-transporting flows in a river's flow regime may be represented by a single, "channel-forming flow."

2. DOMINANT DISCHARGE CONCEPT

The concept of there being a single discharge to which the form of the channel adjusts stems from regime theory and empirical research into the relationships between discharge and channel geometry performed to support the design of stable ("in regime") irrigation canals with granular beds, initially, in the Indian subcontinent during the first half of the twentieth century [e.g., *Inglis*, 1941, 1947, 1949] and, later, in North America [e.g., *Blench*, 1952, 1957; *Simons and Albertson*, 1960]. The regime theory revealed that stable channel width, depth, and slope may be expressed as power functions of the supply discharge. Subsequent laboratory studies undertaken at the Hydraulics Research Station, Wallingford, United Kingdom, validated the form of these regime equations [*Ackers and Charlton*, 1970a, 1970b].

Unlike canals, in rivers the discharge varies seasonally, annually, and interannually depending on the occurrence and duration of precipitation events. In alluvial rivers, all discharges competent to mobilize sediment from the channel boundaries influence the channel form, rendering canal-based, regime equations inapplicable to channel design in rivers. The concept of the dominant discharge or channel-forming flow seeks to overcome this problem by proposing that there is a single discharge which, if held constant over a prolonged period, would produce the same channel morphology (width, depth, and slope), planform pattern, and hydraulic roughness as that generated by the actual distribution of flows experienced by the river. Despite being criticized by prominent academics [e.g., *Richards*, 1982], the dominant discharge concept remains a device attractive to practitioners of river restoration.

According to *Inglis* [1947], the dominant discharge is associated with the condition at which equilibrium is most closely approached and the tendency for channel change is at a minimum. This condition can be regarded as the integrated effect of all varying conditions over a long period. The link between the dominant discharge in rivers and the downstream hydraulic geometry was first investigated by *Leopold and Maddock* [1953] and *Leopold et al.* [1964] and later expanded through the collection of data sets for different types of stable rivers [e.g., *Hey and Thorne*, 1986]. In a further development, *Soar and Thorne* [2001] used confidence bands applied to "typed" hydraulic geometry equations as a mechanism through which natural rivers can be used as analogs for channel restoration design.

Application of the dominant discharge concept is best suited to river systems in which flow regimes are sufficiently steady to allow their morphologies to adjust to prevailing conditions. In such cases, most geomorphic work is performed by events that do not significantly overtop the banks, typically having low to moderate recurrence intervals of less than 2 or 3 years in the annual maximum series (AMS) (the series of single highest discharge in each year of interest, ideally derived from the record of gauged flows averaged over 15 min or hourly intervals).

These conditions pertain in humid, temperate environments where the morphology of perennial rivers recovers relatively quickly following major events that perturb the channel, due to the geomorphic effectiveness of short to medium return-period events, coupled with rapid vegetation growth that helps limit flood-driven erosion and encourage sedimentation [*Hack and Goodlett*, 1960; *Gupta and Fox*, 1974]. In contrast, streams in semiarid environments have flood dominated, flashy regimes. They exhibit morphologies that reflect the recent sequence of floods and which are frequently reshaped [*Macklin and Lewin*, 2003]. Morphological recovery in these ephemeral channels tends to take much longer than in more temperate regions, partly reflecting the stress placed on vegetation growth during long dry periods [*Schumm and Lichty*, 1963; *Burkham*, 1972]. In truly arid areas, infrequent floods of very high magnitude leave long lasting imprints on the channel morphology as intermediate flows occurring between floods lack the energy necessary to drive adjustment toward a regime condition [*Schick*, 1974]. It follows that, where these highly variable flow regimes prevail, the notion that there may be a single discharge that can explain channel form is barely tenable [*Stevens et al.*, 1975; *Baker*, 1977].

The "dominant discharge" is a geomorphic concept rather than a measurable parameter. However, there are three popular candidate discharges that could be taken to represent the dominant flow, based on the application of geomorphic and hydrologic principles: (1) bankfull discharge, (2) a discharge of specified recurrence interval, and (3) effective discharge.

Each can be adopted as the design discharge for channel restoration, but each rests on a different set of assumptions, has different data requirements, and is associated with particular problems and challenges. The next section of this chapter introduces the scientific basis for each of these potential design discharges, evaluates their scientific validity and assesses how the strengths and weaknesses inherent to their derivation and application play out in practice.

3. APPROACHES TO CALCULATING THE DESIGN DISCHARGE

3.1. Bankfull Discharge

3.1.1. Science base. The bankfull discharge is essentially the largest flow that can be conveyed by a channel without overtopping its banks. Based on extensive field data, *Inglis* [1947] first considered that flows at or near the bankfull stage might approximate to the dominant discharge, and the link he proposed between the bankfull and dominant discharges has been supported by a wealth of subsequent research findings demonstrating that flows around bankfull exhibit a strong relationship with stable channel dimensions [*Wolman and Leopold*, 1957; *Nixon*, 1959; *Simons and Albertson*, 1960; *Leopold et al.*, 1964; *Kellerhalls*, 1967; *Hey*, 1975, 1982; *Charlton et al.*, 1978; *Hey and Thorne*, 1986; and many others]. Based on these findings, the bankfull discharge in river systems appears to be of comparable morphological significance to the supply discharge in canals that are in regime.

Hey [1997] highlighted that the bankfull elevation often marks a significant discontinuity in the stage-discharge curve. As water spills onto the floodplain, the greater depth of flow and lower hydraulic roughness of the main channel together can result in appreciably higher velocities in the main channel than those occurring on the floodplain. The difference in velocity between in-bank and over-bank flows can then lead to a lateral transfer of momentum and a reduced channel discharge capacity [*Knight and Shiono*, 1990; *Shiono and Knight*, 1991; *Ackers*, 1993; *Ervine et al.*, 1993; *Shiono et al.*, 1999; and many others]. As a result, floodplain flows rarely impose appreciable increases in bed shear stress in the channel, and so high in-bank stages also tend to represent the condition under which the availability of energy to drive in-channel processes of sediment erosion, transport, and deposition is greatest. Above the bankfull level, experimental studies have demonstrated that concentrations of sediment transported as bed load actually decline with further increase in discharge or floodplain roughness, even dropping below the value at bankfull in some cases [*Atabay et al.*, 2005; *Tang and Knight*, 2006]. It is the strong "morphogenetic" significance of bankfull discharge that led *Hey* [1978] to stress the utility of bankfull discharge for stable channel design purposes.

3.1.2. Science into practice. In practice, the challenge is less that of estimating the bankfull discharge, per se, and more that of identifying the correct bankfull reference level and measuring the corresponding bankfull elevation. As *Leopold et al.* [1964] pointed out, this is not a simple matter, and small differences in the selected bankfull elevation can lead to large differences in bankfull discharge. *Williams* [1978] presented a detailed review of how to identify the bankfull stage, including a range of definitions based on sedimentary features, cross-sectional morphology, and changes in bank vegetation (Table 1). In a natural river, an appropriate definition is "the discharge conveyed at the elevation of the active floodplain" [after *Wolman and Leopold*, 1957; *Dury*, 1961; *Emmett*, 1972, 1975; *Williams*, 1978; *Andrews*, 1980, 1984; *Nolan et al.*, 1987; *Hey and Thorne*, 1986; and others]. Recently, *Pike and Skatena* [2010] demonstrated that the first occurrence of soil and woody vegetation can be a reliable indicator of the bankfull level.

However, accurate location of bankfull indicators is not a routine procedure with a precise analytical method [*Radecki-Pawlik*, 2002]; it is fraught with difficulty and uncertainty [*Williams*, 1978; *Johnson and Heil*, 1996], with most methods being highly subjective. For individual cross sections, the erosional and depositional forms associated with bank processes, and the presence and character of vegetation interact to yield an indistinct boundary between the channel bank and its floodplain, resulting in a transitional range of bankfull elevations, rather than a single value [*Navratil et al.*, 2006].

A more objective method is to identify the level corresponding to the minimum width-to-depth ratio within the cross section [*Wolman*, 1955]; although *Navratil et al.* [2006] found geometric criteria to be less reliable in locating the bankfull level than identifying geomorphic features. However, despite these documented methods, and the availability of instructions intended to minimize uncertainty and encourage consistency [e.g., *Harrelson et al.*, 1994; *Leopold*, 1994; *Forest Service, U.S. Department of Agriculture*, 1995, 2003], there is no method for defining the bankfull reference level that is universally applicable, comparison between reaches remains difficult [*Richards*, 1982], and accurately locating field indicators continues to remain a major hurdle. In seeking to reduce uncertainty in the identification of the bankfull level, the experience of a fluvial geomorphologist is critical, and several bankfull criteria should be adopted and applied to more than one cross section in order to produce a reliable result [*Harman et al.*, 2008].

Once the bankfull elevation has been identified, the method applied to derive a value for the bankfull discharge is dependent on whether there is a gauging station close to the project reach. If there is a gauging station, the recommended procedure is to survey a long profile of bankfull elevations within the reach of interest, extrapolate this to the gauging station, and then read the corresponding discharge from the gauging station's stage-discharge curve [*Leopold et al.*, 1964]. This approach has been used successfully in many studies [e.g., *Hey and Thorne*, 1986], though, in practice, it is subject to many assumptions and difficulties, particularly regarding the reliability of the gauged flow record and the impacts of any channel or floodplain modifications or structures that complicate extrapolation of the bankfull profile. The accuracy of this approach decreases as the distance to the nearest gauging station increases, especially if channel conditions change significantly en route.

A number of methods may be considered for application to ungauged rivers, including (1) stream gauging, (2) synthesizing a stage-discharge curve using either a flow resistance equation (typically the Manning formula) or a hydraulic model, such as Hydrologic Engineering Center River Analysis System (HEC-RAS) [*Brunner*, 2010], (3) applying a "channel geometry" equation to predict discharge from bankfull width or cross-sectional area [e.g., *Wharton et al.*, 1989; *Wharton*, 1992, 1995a, 1995b; *Osterkamp and Hedman*, 1982], or (4) applying a regional curve relating bankfull discharge to drainage basin area (see discussion below).

Attempts have also been made to estimate bankfull discharge solely from remotely measured data [e.g., *Bjerklie*, 2007], with reasonable success. Table 2 presents the options available to calculate bankfull discharge for gauged and ungauged sites, together with the possible limitations, sources of uncertainty, and constraints.

The association between bankfull and the dominant discharges rests on the assumption that the project reach is dynamically stable; that is, that the reach-averaged channel dimensions and planform are adjusted to the prevailing flow and sediment regimes. If the river is unstable, its channel is likely to reflect either the trend of morphological evolution toward a new, equilibrium condition or the degree of morphological recovery following destabilization [*Wolman and Gerson*, 1978], rather than the magnitude of the channel-forming flow. This is an issue because channel instability is often the reason that a reach is a candidate for restoration.

Table 1. Variable Criteria for Identifying the Bankfull Reference Level

Bankfull Indicator Reference	Source
Geomorphic/Sediment Criteria	
Elevation of active floodplain	*Wolman and Leopold* [1957]
	Nixon [1959]
	Leopold and Skibitzke [1967]
	Emmett [1972, 1975]
Highest elevation of channel bars	*Wolman and Leopold* [1957]
	Hickin [1968]
Elevation of the most prominent bench	*Kilpatrick and Barnes* [1964]
Elevation of the "middle bench" in rivers with several overflow surfaces	*Woodyer* [1968]
Elevation of low bench	*Schumm* [1960]
	Bray [1972]
Elevation of upper limit of sand-sized particles in boundary sediment	*Nunally* [1967]
	Leopold and Skibitzke [1967]
Geometric Criteria	
Minimum width-to-depth ratio	*Wolman* [1955]
	Harvey [1969]
	Pickup and Warner [1976]
Minimum width-to-depth ratio plus a vegetative and or physical discontinuity in the channel boundary	*Wolman* [1955]
Maximum of the bench index (developed from the width-to-depth ratio)	*Riley* [1972]
Change in relation of cross-sectional area to top width	*Williams* [1978]
Vegetative Criteria	
Channelward limit of perennial vegetation (normally trees or tall grasses)	*Schumm* [1960]
	Speight [1965]
	Nunally [1967]
	Bray [1972]
Change in vegetation type (herbs, grass, shrubs)	*Woodyer* [1968]
	Leopold [1994]

Under these circumstances, the bankfull condition in the project reach is unlikely to be a reliable indicator of the channel-forming discharge [*Doyle et al.*, 1999], and it may, therefore, be unsuitable as the design discharge for restoration to a stable condition. This precludes use of bankfull as the design discharge for restoration unless a suitably stable "reference" reach can be identified in relatively close proximity. However, finding a stable reference reach presents a particular challenge in watersheds exhibiting system-wide instability in the drainage network, and nonimpacted neighboring reaches provide bankfull discharge estimates suitable for restoration designs only in situations where channel instability in the project reach can be clearly attributed to a local disturbance [*NRCS*, 2007].

It follows that adoption of bankfull discharge to represent the channel-forming flow relies on geomorphic reconnaissance of the project and adjacent reaches, coupled with accurate interpretation of channel forms and processes within the context of adjustments in the fluvial system, and some knowledge of sediment dynamics at the watershed scale [see *Downs and Thorne*, 1996; *Thorne et al.*, 1996; *Thorne*, 1998; *Sear et al.*, 2010]. A watershed assessment (see *NRCS* [2007] for methodologies) or fluvial audit [*Sear*, 1994; *Sear et al.*, 2009, 2010] provides the ideal baseline from which to establish the catchment context for restoration and is a prerequisite to locating reference reaches from which a bankfull discharge suitable for design purposes can be derived. However, such comprehensive watershed assessments require extensive project resources, which are seldom available. Where project resources constrain background investigations, assessment of the river in the sediment supply reach immediately upstream of the project reach is recommended as the minimum necessary to support channel restoration design [*Soar and Thorne*, 2001].

Given the difficulties involved in determining the bankfull discharge based on field observation, it is unsurprising that application of generalized, regional regression curves is gaining popularity as an alternative approach to estimating

Table 2. Practical Methods for Calculating the Bankfull Discharge at Gauged and Ungauged Sites

Methods	Data Requirements	Limitations, Uncertainties, and Constraints
	Gauged Sites	
Stage-discharge analysis	Surveyed bankfull elevation profile extrapolated from the project reach to the gauging station.	Defining bankfull stage based on field indicators or morphological criteria can be problematic and misleading. Bankfull stages can be highly variable over short distances. Channel modifications and structures can prevent accurate extrapolation of the bankfull level profile. Geomorphic skills and experience are essential.
	Stage-discharge curve generated from the gauged record.	Requires a reliable flow record ideally for the past 10 years or more. Potential unreliable rating at high flows. Nonstationarity in flows through the period of record could indicate that the restored channel might not be sustainable in the future. Time base (mean daily, hourly or 15 min data) can influence the shape of the curve.
	Ungauged Sites	
Direct stream gauging	Velocity-area method (cross section; velocity distribution).	Channel inaccessible at high stages.
Stage-discharge analysis	Bankfull elevation measured over at least 10 channel widths.	Difficulty locating a stable and unmodified reach in the vicinity of the site, often in watersheds with system-wide instability. Defining bankfull stage based on field indicators or morphological criteria can be problematic and misleading. Bankfull stages can be highly variable over short distances. Geomorphic skills and experience are essential.
	Synthesized discharge corresponding to bankfull elevation, based on either	
	(1) flow resistance equation (cross section; roughness coefficient; slope).	Equations assume uniform flow conditions and are widely reported to generate errors. Experience is required to select an appropriate roughness equation for the type of watercourse and assign an appropriate roughness coefficient. Different measures of slope (bed, water surface) can significantly influence discharge calculation.
	(2) computer model (e.g., HEC-RAS) (geo-referenced channel survey extending through reach; roughness coefficients).	Model assumptions for generating water surface profile. Channel surveys are costly and can be problematic. Calibration data are required. Modeling experience is essential.
Channel geometry analysis	Existing relationship predicting bankfull discharge from bankfull width for similar type of region.	Issues related to identification of bankfull stage (see above, for stage-discharge analysis).
	Bankfull elevation measured over at least 10 channel widths to derive an average bankfull width.	Issues related to application of regression equation (see below, for regional curve application).
Regional curve application	Existing relationship predicting bankfull discharge from drainage basin area for similar type of region or new relationship developed for the study watershed.	Often considerable variability of points around the regression lines.
	Drainage basin area at site.	Other variables that influence stream flow are not accounted for. Restored reach must have similar physiography, geologic and hydrologic conditions to sites used to develop regional curve. Limited equations available.

bankfull discharge for restoration design purposes [*Rosgen*, 1998, 2006a, 2006b; *Hey*, 2006; *NRCS*, 2007].

Regional curves are based on regression analysis using a power law of the form,

$$Q = aA^b, \tag{1}$$

where Q is bankfull discharge (typically in $\mathrm{ft}^3\ \mathrm{s}^{-1}$), and A is drainage basin area (typically in square miles). The regression coefficient "a" and exponent "b" depend on regional physiography, hydrology, geology, and vegetation cover. The exponent "b" is typically between 0.7 and 0.75 [*Leopold et al.*, 1964], although considerable variation is found across regions. Early work by *Emmett* [1975] and *Dunne and Leopold* [1978] established that a clear relationship between bankfull discharge and drainage area exists in most watersheds, and regional relationships are available for several areas of the United States (Figure 1 provides an example) and elsewhere [e.g., *Petit and Pauquet*, 1997]. Regional analyses usually also derive downstream hydraulic geometry relationships, expressing bankfull width, depth, and cross-sectional area as functions of drainage basin area (see *Faustini et al.* [2009] for a review and *Johnson and Fecko* [2008] for a statistical comparison between data sets). Where available, regional hydraulic geometry equations may be applied to design stable channels directly, obviating the need for a design discharge.

A comprehensive overview of the "regional curve" method is provided by *NRCS* [2007], while the National Water Management Center (NWMC) of the Natural Resources Conservation Service (NRCS) hosts a dedicated archive of regional curve studies on their web site (http://wmc.ar.nrcs.usda.gov/technical/HHSWR/Geomorphic/). The NWMC is currently partnering other federal, state, and local agencies in a mission to develop regional curves for the entire country, based on the 25 physiographic provinces previously identified by *Fenneman and Johnson* [1946]. Numerous studies reported in the academic literature [e.g., *Castro and Jackson*, 2001; *Doll et al.*, 2002; *Sweet and Geratz*, 2003; *Metcalf et al.*, 2009] and in technical reports [e.g., *McCandless*, 2003; *Metcalf*, 2003; *Chaplin*, 2005; *Dudley*, 2005; *Keaton et al.*, 2005; *Sherwood and Huitger*, 2005; *Mulvihill et al.*, 2007] support the utility of the regional curve approach. However, *Wilkerson* [2008] found that bankfull discharge could be more reliably predicted through regression against the 2 year flow than the drainage area. The *Wilkerson* [2008] approach facilitates the integration of geologic, climatic, and hydrologic factors (in addition to drainage area) into relations for predicting bankfull discharge, and its application is thus not restricted to watersheds with reliable records of gauged flows.

The regional curve method clearly has merit for estimating bankfull discharges, validating field estimates of bankfull stage, and/or establishing stable channel dimensions for river restoration projects in ungauged watersheds. The regional curves produced by federal and state agencies are freely available, and users can be confident that they have been derived with a high degree of care, adhering to best practice

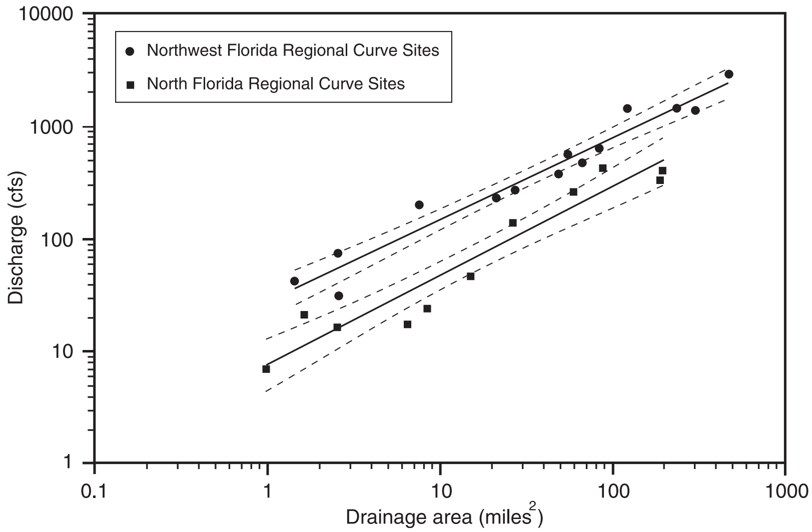

Figure 1. Regional curves for bankfull discharge estimated from drainage basin area for coastal plain streams in Florida. Northwest Florida is represented by the top solid line and north Florida by the bottom solid line with 95% confidence intervals (dashed lines). From *Metcalf et al.* [2009], reprinted with permission from John Wiley and Sons, Ltd.

in data collection and processing. Most studies also examine and report on uncertainties and include useful discussion of the methods' limitations.

However, the development of regional curves still cannot avoid the well-documented problems associated with accurately identifying the bankfull stage and the approach can be criticized as it lacks a basis in physical processes and fails to take into account the multitude of catchment variables that actually influence the flow and sediment regimes responsible for driving channel-forming processes. In practice, many regional curves exhibit considerable data scatter, making the derivation of a single value for the bankfull discharge associated with a given drainage basin area questionable statistically. This issue is also pertinent to the channel geometry method proposed by *Wharton* [1992, 1995a, 1995b] as well as downstream hydraulic geometry relationships in general. Finally, not all regions of the United States currently have regional curves, and uptake of the approach outside the United States has been patchy (however, see *Davidson and North* [2009]).

3.2. Discharge of Specified Recurrence Interval

3.2.1. Science base. The frequencies and durations of candidate channel-forming discharges have been investigated widely since the 1950s. Based on the premise that the dominant discharge must occur often enough to permit alluvial river channels to display a regime condition most of the time [*Nixon*, 1959], numerous studies have revealed a remarkable similarity in the recurrence interval of the bankfull discharge in a variety of rivers, based on the AMS of measured peak flows. Measurements in different regions in the United States by *Wolman and Leopold* [1957] showed that the recurrence interval for bankfull flow in undisturbed rivers with well-developed floodplains ranged between 1 and 5 years. Later, *Leopold et al.* [1964] evaluated 19 river reaches "where the recurrence interval of the incipient flood stage could be accurately fixed" from reliable, nearby gauging stations and found that the frequency of bankfull discharge ranged between 1.07 years and 4.0 years, although the frequency only exceeded 1.9 years at 4 of the 19 sites. While there is no consensus concerning the modal recurrence interval for the bankfull discharge, it is generally considered among practitioners that the bankfull event for perennial rivers in temperate-humid environments will occur, in most cases, every 1 to 2 years, following the findings of *Leopold et al.* [1964] and others, including *Kilpatrick and Barnes* [1964] and *Carlston* [1965].

A recurrence interval of 1.5 years was considered by *Leopold et al.* [1964] to be a representative average frequency for bankfull discharge, a figure that was later corroborated for gravel bed rivers in the United Kingdom by *Hey* [1975], and linked to the "most-probable" (modal) annual flood (with a recurrence interval of 1.58 years) by *Dury* [1973, 1976]. More recently, use of the 1.5 year flood to represent the bankfull discharge has been supported by the results of numerous regional studies in the United States [e.g., *Castro and Jackson*, 2001]. On the basis of observations in a range of hydrophysiographic regions in the United States, *Rosgen* [1998] concluded that the average recurrence interval of the bankfull discharge is 1.1 to 1.8 years, which is remarkably close to the earlier findings.

3.2.2. Science into practice. Adoption of the flood with a recurrence interval of 1 to 2 years in the AMS as a channel-forming flow equivalent to the bankfull discharge has become something of an orthodoxy in applied fluvial geomorphology and river restoration practice, and its use as a design discharge has been actively promoted in situations where use of the morphologically defined bankfull discharge is inappropriate due to past channel modifications or channel instability [*Hey*, 1997]. There is also practical evidence that the 1.5 year flood provides a viable alternative to the bankfull discharge in restoration design [*Hey*, 1994].

While using an objective measure of channel-forming discharge based on measured flows is attractive, especially in light of the potential subjectivity and challenges in deriving a value for the bankfull discharge, numerous cases have been reported where the recurrence interval of bankfull discharge has been found to lie outside the expected range of 1 to 2 years. For example, *Pickup and Warner* [1976] demonstrated that the recurrence interval for bankfull discharge may range from 4 to 10 years in the AMS, while *Williams* [1978] found that bankfull discharge corresponded to a recurrence interval of about 1.5 years in only one third of 36 cases examined, the range being 1.01 to 32 years.

Deviation from a 1.5 year recurrence interval is also supported by *Andrews* [1980], who found that the bankfull discharge for half the sites investigated in the Yampa River basin in Colorado and Wyoming had recurrence intervals that were greater than 1.75 years or less than 1.25 years, the range being from 1.18 to 3.26 years. He attributed this variability to climatic, geological, and physiographic factors. The widely applied U.S. Army Corps of Engineers manual on channel stability assessment [*U.S. Army Corps of Engineers (USACE)*, 1994] recommends, for engineering analysis, a recurrence interval of approximately 2 years for the channel-forming discharge (a frequency given significance by the findings of *Bray* [1973, 1975, 1982] for gravel bed rivers in Alberta and recently by *De Rose et al.* [2008] for rivers in Victoria, Australia), but also acknowledges that this frequency may vary between the 1 and 10 year flood flows.

Table 3 lists some of the ranges of recurrence interval for bankfull discharge reported in the literature, although the list is not exhaustive and does not include the findings of U.S. regional studies documented in a wealth of technical reports. Importantly, small differences in recurrence interval can correspond to marked differences in flow magnitude, which could translate into significant differences in designed channel dimensions for a river restoration scheme. Based on the tabulated data reported by *Crowder and Knapp* [2005] for sites on Illinois streams, the average ratio of the 2 year to 1.5 year flow is 1.27, and the average ratio of the 2 year to 1.25 year flow is 1.62. For example, the 2 year flow for Silver Creek near Freeburg, Illinois, is reported to be 148 m^3 s^{-1}; almost twice the 1.25 year flow of 76 m^3 s^{-1}. In summary, there is a growing recognition that the bankfull discharge of stable, alluvial rivers may be associated with a range of flows of varying magnitude and frequency [e.g., *Petit and Pauquet*, 1997; *Radecki-Pawlik*, 2002], which challenges the utility of an event with a unique recurrence interval as a design discharge.

A problem associated with use of the AMS to identify the recurrence interval for bankfull discharge is the potential introduction of bias due to its asymptotic lower limit of 1 year as the shortest recurrence interval event that can be identified [*Navratil et al.*, 2006]. For example, *Castro and Jackson* [2001] found the modal recurrence interval for streams in the American Pacific Northwest to be 1.0, based on the AMS. However, studies based on a partial duration analysis that considers all the independent peak discharges that exceed a specified threshold discharge, rather than just the annual maxima, have revealed frequencies of bankfull discharge considerably shorter than 1 year.

Despite methodological difficulties in defining the threshold discharge for the partial duration series of peak flows [*Petit and Pauquet*, 1997], *Hey and Heritage* [1988] discovered a range of recurrence intervals for bankfull discharge between 0.56 and 3.44 years for 14 gravel bed rivers in England and Wales, with a modal value of 0.9 years. This frequency was corroborated by *Carling* [1988] for two gravel bed rivers in northern England. These findings are unsurprising given that *Nixon* [1959] had previously analyzed flow duration data from 29 rivers in England and Wales and demonstrated that the bankfull discharge was equaled or exceeded on average 0.6% of the time; that is, slightly more than 2 days per year. Interestingly, this corresponds to a bank overtopping frequency of 2.2 times per year, which is equivalent to a recurrence interval of approximately 0.5 years (based on the reanalysis by *Leopold et al.* [1964]). In light of this, *Hey* [1998] recommended the use of exceedance durations, rather than annual recurrence intervals, to describe the frequency of the channel-forming discharge for river restoration applications.

These findings indicate that adoption of the flood with a recurrence interval of 1 to 2 years as the design discharge for river restoration cannot be assumed, but should be corroborated by information from other sources and analyses.

An emerging body of evidence suggests that variability in the flow regime might be responsible for the observed variation in the frequencies of bankfull flow. Flows that tend to be more effective in performing geomorphologic work, through transporting sediment and shaping the channel boundary, are often more frequent than average in base flow-dominated streams and, conversely, less frequent than average in streams with flashy hydrographs. This hydrologic influence is examined further in the discussion of effective discharge, below. It is worthy of note though that published "ranges" of frequencies tend to highlight the low populated tails of the sample distributions and conceal the more significant modal values and central portions. For example, *Soar and Thorne* [2001] used data from 58 stable sand bed rivers in the United States to conclude that, although a wide-range of recurrence intervals are possible for the bankfull condition, 86% of the sites studied fell within the 1 to 2 year range.

Studies that have highlighted inconsistencies in the recurrence intervals for bankfull discharge have variously attributed this to the influence of discharge variability, catchment size, bed material type, and other influences. For example, *Petit and Pauquet* [1997] identified that the recurrence interval for bankfull discharge was 0.5 years for small gravel bed rivers in Belgium, rising to 1.5 years for larger catchments, exceeding 2 years for rivers with base flow-dominated regimes and longer still for rivers with fine-grained beds. Despite this, to date, such investigations have failed to provide any generalized guidance to practitioners on predicting

Table 3. Variable Ranges for the Recurrence Interval of Bankfull Discharge Reported in the Literature

Discharge Frequency (years)	Source of Research or Recommendation
1 to 1.23	*Crowder and Knapp* [2005]
0.3 to 1.4	*Powell et al.* [2006]
1 to 2.5	*Leopold* [1994]; *Simon et al.* [2004]
1.02 to 2.69	*Woodyer* [1968]
1 to 3.1	*Castro and Jackson* [2001]
1.18 to 3.26	*Andrews* [1980]
1.07 to 4	*Leopold et al.* [1964]
1.1 to 4.8	*Whiting et al.* [1999]
1.01 to 5	*Wolman and Leopold* [1957]
0.7 to 5.3	*Petit and Pauquet* [1997]
1 to 10	*Brush* [1961]; *USACE* [1994]
1.01 to 32	*Williams* [1978]

the likely range of recurrence intervals for bankfull discharge on the basis of the characteristics of the study stream or its flow regime.

The statistical treatment of gauged peak flows for flood frequency analysis is a long established practice in applied hydrology and is widely documented in the technical literature [e.g., *Robson and Reed*, 1999; *NRCS*, 1999, 2007]. The United States Geological Survey (USGS) operate and maintain a large network of gauging stations across the United States, with historical peak flow data archived and readily available from their website (http://nwis.waterdata.usgs.gov/usa/nwis/peak/). Additional data sets are also available for thousands of discontinued gauging stations. Currently, peak stream flow data from 27,500 sites can be obtained from the USGS National Water Information System. In the United Kingdom, the HiFlows-UK website (http://www.environment-agency.gov.uk/hiflowsuk/) hosts the hydrometric data archives from the various gauging authorities (the Environment Agency in England and Wales, Scottish Environmental Protection Agency in Scotland and the Rivers Agency in Northern Ireland) and includes updated flood peak data for almost 1000 stations, together with the supporting information necessary to enable hydrologists to make informed judgments concerning the utility of the data.

However, despite the existence of large networks of gauging stations in more economically developed countries like the United States and the United Kingdom, it is rare for a restoration project reach to be sufficiently close to a hydrometric station for the flow record to be applied to the project site without some adjustment to account for the difference in drainage areas. This is particularly the case for small watersheds, remote areas, and headwater streams, where gauging networks tend to be sparse and data availability limited [*Juracek and Fitzpatrick*, 2009]. The fact is that inadequate availability of raw flow data continues to represent a serious impediment to river analysis.

Even where gauge data are available, data quality issues can preclude use of historical peak discharge records for flow frequency analysis [*Juracek and Fitzpatrick*, 2009]. Issues include the following: (1) stage-discharge rating curves that are unreliable for out of bank flows due to flow bypassing the gauged section, (2) gaps and/or spurious records caused by equipment failures, (3) inadequate length of flow record, (4) inadequate representation of recent events if contemporary data are unavailable or the gauge is discontinued, (5) nonstationarity in the record reflecting historical changes to the catchment or drainage system, (6) underestimation of the true peaks if mean daily discharges are recorded/reported rather than 15 min values.

These issues are most problematic when analyzing discharges toward the extremes of the discharge record. In practice, gauge data are usually accurate for relatively frequent, in-bank flows close to bankfull.

Given the sparsity of gauging networks, restoration designers usually have to estimate the discharge of a specific recurrence interval, such as the 2 year event, for ungauged project sites. Most of the approaches they adopt involve translating data from a gauging station elsewhere in the river system or from an analog watershed.

The simplest transfer method, requiring least amount of data, is development of a regional relationship for predicting discharge of specified recurrence intervals as a power function of drainage basin area, in a manner similar to the popular regional curve method for estimating bankfull discharge. A number of relationships are available to do this, though development of a simple regression curve specific to the study watershed or parent region is often preferred [*NRCS*, 1999]. More advanced analyses use multiple regression relationships that account not only for the influence of drainage area, but also watershed climate, slope, and flood storage capacity. The U.S. Geological Survey, together with state and local agencies, has applied this type of approach to gauged watersheds within every American state [*Jennings et al.*, 1994], and the results of these advanced hydrological investigations are available in the National Streamflow Statistics database (http://water.usgs.gov/osw/programs/nss/), which includes regional regression relations for estimating peak discharges at ungauged sites in 289 flood regions nationwide [*Ries*, 2006; *Turnipseed and Ries*, 2007].

As with the regional curves used to predict bankfull discharge, peak flow relationships exhibit varying degrees of reliability, with standard errors of estimate commonly between 30% and 60%, particularly for western areas of the United States, where high flow variability, the sparsity of gauging stations, and the comparatively short duration of available flow records often combine to produce significant uncertainty [*NRCS*, 2007]. The fact is that regional regression equations are not as accurate as frequency analyses applied to the flow series from a single gauging station, and they should be applied with caution, especially when estimating recurrence intervals for flood flows in watersheds whose characteristics lie outside the ranges of values used in the development of the regression equations.

In the United Kingdom, the Flood Estimation Handbook (FEH) and associated hydrologic software [*Institute of Hydrology*, 1999; *Centre for Ecology and Hydrology*, 2007] comprise the nationally applied standard approach for flood magnitude and frequency estimation, and these tools include a number of techniques for dealing with ungauged sites and sites with short periods of record. In such cases, data are "pooled" from a group of gauging stations identified using the standard software as exhibiting similar "catchment descriptors" and assumed to share a common flow regime [*Robson and Reed*, 1999]. This pooling approach offers an alternative to conventional, regional methods that can be unreliable in geographical areas that include watersheds with contrasting hydrologic characteristics.

The FEH also supports rapid estimation of discharges for any selected recurrence interval using a multiple regression model for the median annual maximum flood, which is the standard "index" flood event used by FEH at ungauged sites in the United Kingdom, based on catchment descriptors, and then scaling this value to less frequent events according to a dimensionless growth curve. The median annual maximum flood is a good estimator of the peak flow with a 2 year recurrence interval provided that more than 15 years of AMS data are available [*Reed*, 2002].

Although advanced applications of FEH methods require the attention of an experienced hydrologist, individuals can use the FEH rapid technique to estimate the 2 year flow routinely, with just some basic training. Hence, the rapid technique is an attractive option for generating restoration design discharges for the United Kingdom when limited project resources preclude the use of more detailed analyses.

Flood-frequency estimation is inherently uncertain, and all the approaches outlined above require sound insight and judgment on the part of the individual performing the analysis. In practice, the estimates can only be considered to be reliable if they are consistent with the flood frequency behavior of the river and the characteristics of the parent watershed.

Given the number of factors that influence flood frequency, it would be surprising if the hypothesis that a discharge with a particular recurrence interval equates to the bankfull discharge went unchallenged [*Doyle et al.*, 1999, 2007; *Shields et al.*, 2003, 2008]. In essence, adoption of the flow associated with a selected recurrence interval as the design discharge for a restoration project involves a trade-off between its strengths (ease of application, speed of calculation, and apparent objectivity) and its weaknesses (high uncertainty, inability to account for the influence of fluvial processes, and considerable reliance on gauged data). In light of this, it is recommended that a design discharge based on a specified recurrence interval and derived from flood frequency analysis is only taken to be indicative of the channel-forming flow and that practitioners are encouraged, wherever possible, to validate the reliability of the design discharge using one or more of the other approaches described in this chapter.

3.3. Effective Discharge

3.3.1. Science base. Effective discharge theory is based on the premise that the stable channel morphology is intrinsically linked to the prevailing sediment transport regime. This is argued to be the case because disturbance of a stable (or graded) river generates imbalance in the transfer of sediment along its course, which initiates morphological responses (driven by erosive and/or depositional processes) that cause the channel either to adjust toward a new condition of dynamic equilibrium or recover its predisturbance morphology. According to this reasoning, the bankfull channel geometry of a stable alluvial channel is shaped by the delicate balance between sediment supply and sediment transport so that, over a period of years, sediment inputs and outputs are balanced [*Mackin*, 1948].

Wolman and Miller [1960] built on the concept of the dynamically stable river, with its "graded profile," by proposing that the geomorphic effectiveness of discharges making up the flow regime depends not only the magnitude of a flow event but also its frequency of occurrence. They argued that an alluvial river with a mobile bed will tend to adjust its bankfull capacity to the flow that transports the greatest quantity of sediment over a number of years; that is, the flow doing most geomorphic work on the channel through transporting sediment. The notion that the flow doing most work could be considered to be the dominant discharge was alluded to by *Wolman and Miller* [1960] and later by *Wolman and Gerson* [1978], though it was *Andrews* [1980] who first described this flow as the "effective discharge."

Magnitude-frequency analysis, as described by *Wolman and Miller* [1960], requires integration of the flow duration (the cumulative distribution of gauged discharges) with a sediment rating curve (the relationship between discharge and sediment transport rate) to derive a sediment load histogram (which can be expressed as the percentage of the average annual sediment yield for the range of discharge classes). The effective discharge is then defined by the peak in the sediment load histogram (Figure 2). The specific stages in computation of the effective discharge are described more fully in section 3.3.3.

Generally, the effective discharge corresponds to a moderate discharge of intermediate frequency, as demonstrated by *Costa and O'Connor* [1995] using stream power concepts. *Wolman and Miller* [1960] showed that 90% of the sediment transported in suspension (the suspended load) in the alluvial rivers they studied in the west of the United States is transported by flows with recurrence intervals of less than 5 years. It follows that, according to magnitude-frequency analysis, both low discharges with high frequencies and large, rare events with long recurrence intervals play relatively minor roles in forming the channel. This is the case because high frequency flows smaller than the effective discharge are capable of transporting little sediment and, hence, are

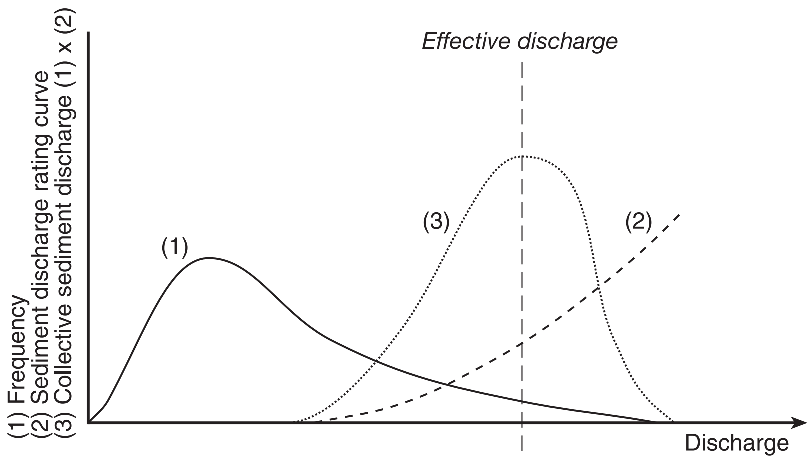

Figure 2. Calculation of the effective discharge from magnitude-frequency analysis, showing the derivation of bed material load-discharge histogram (labeled 3) from flow frequency (labeled 1) and bed material load rating curves (labeled 2).

ineffectual as channel-forming agents. Conversely, flow events significantly greater in magnitude than the effective discharge, while having the capacity to transport sediment at very high rates, occur too infrequently to have a marked, long-term influence in shaping the channel boundary.

It follows that the morphological impacts of long recurrence interval events tend to be significant only for relatively short time periods, whereas the intermediate events that occur multiple times between these extreme events cause the channel to recover its stable, or regime, morphology [*Wolman and Gerson*, 1978] by adjusting its bankfull dimensions to accommodate the effective discharge [*Hey*, 1975]. However, it should be noted that in river systems where the length of time required for full morphological recovery following disturbance from high magnitude floods is long, the recovery driven by lesser intermediate size flows is likely to be interrupted by other high magnitude events, and the hypothesis that channel dimensions are "adjusted" to the flow doing most work through sediment transport becomes less tenable. Hence, in highly responsive systems, such as those found in semiarid and arid regions, the effective discharge would not be a good representation of a channel-forming flow [*Hey*, 1975].

Despite this limitation, the effective discharge is widely regarded as the preferred choice for representing the channel-forming or dominant discharge and therefore the best candidate for acting as a design discharge for river restoration. The effective discharge has also been shown to be useful when analyzing stream ecosystems [*Doyle et al.*, 2005].

A strong case has been made that the effective discharge should equate to the bankfull discharge in dynamically stable rivers with mobile beds [*Knighton*, 1984]. This argument rests on the argument that the bankfull condition maximizes energy efficiency by minimizing the impact of in-bank boundary roughness, while avoiding energy losses to floodplain vegetation resistance and lateral momentum exchange, so maximizing the amount of energy available to be expended in performing geomorphic work through sediment transport. This theoretical argument is supported by the results of empirical studies that have described how sediment transport rate increases rapidly during high in-bank flows approaching the bankfull level [e.g., *Parker et al.*, 1982; *Andrews*, 1984; *Carling*, 1988; *Ashworth and Ferguson*, 1989; *Warburton*, 1992; *Andrews and Nankervis*, 1995; *Whiting et al.*, 1999; *Ryan et al.*, 2005].

Equivalence between the effective and bankfull flows has been demonstrated in a wide range of river types and settings [*Wolman and Miller*, 1960; *Leopold et al.*, 1964; *Andrews*, 1980; *Knighton*, 1984; *Carling*, 1988; *Andrews and Nankervis*, 1995; *Batalla and Sala*, 1995; *Pitlick and Van Steeter*, 1998; *Torizzo and Pitlick*, 2004; *Powell et al.*, 2006]. Magnitude-frequency analyses of the Lower Mississippi and Pearl Rivers reported by *Biedenharn et al.* [1987] revealed that the effective discharge had a recurrence interval close to 2 years at many sites, while *Watson et al.* [1997] suggested an upper frequency bound of 5 years for streams in north Mississippi and *Whiting et al.* [1999] calculated an average effective discharge recurrence interval of 1.4 years for headwater, gravel bed streams in Idaho, which is remarkably close to the 1.5 years modal value found by *Leopold et al.* [1964], *Hey* [1975], and others. *Simon et al.* [2004] considered the 1.5 year peak flow to be a fair representation of the effective discharge for rivers dominated by suspended sediment transport across the United States.

Since its conception in 1960, magnitude-frequency analysis has proven to be a popular geomorphic technique with a wide range of applications. Examples include detailed investigations of sediment transport [e.g., *Ashmore and Day*, 1988; *Lyons et al.*, 1992; *Biedenharn and Thorne*, 1994] and studies reexamining the magnitude-frequency methodology itself [e.g., *Sichingabula*, 1999; *Orndorff and Whiting*, 1999; *Biedenharn et al.*, 2000, 2001]. The technique has also been employed to facilitate prediction of the trend and magnitude of channel response to hydrological change [*Tilleard*, 1999] and has been used as a mechanism to assess the restorative potential of rehabilitation schemes by comparing observed channel response (a function of flow events since project implementation) with the potential for morphological change, inferred from the full spectrum and range of flows in the long-term record [*Downs et al.*, 1999].

3.3.2. Science into practice. Numerous studies have advocated use of the effective discharge as the design discharge for river restoration [e.g., *Orndorff and Whiting*, 1999; *Shields et al.*, 2003, 2008; *Goodwin*, 2004], with reliance on bankfull discharge or a recurrence interval flow considered "risky and unwise" [*Doyle et al.*, 2007]. In addition, magnitude-frequency analysis of sediment transporting flows allows quantification of the total sediment yield and enables sediment continuity objectives to be tested as part of the restoration process, so providing the best chance of achieving dynamically stable channel morphology.

However, prior to calculating and using the effective discharge as a design discharge for river restoration, three issues should be considered:

1. Selecting a single design discharge of intermediate magnitude implies that the morphological impacts of all other flows may be ignored, which has been shown in numerous studies not to be the case.

2. There is a body of evidence that suggests a degree of discordance between the effective and bankfull discharges, yet currently there are no generally accepted deterministic or probabilistic methods for relating the two.

3. Of the available approaches to specifying the design discharge for river restoration, the effective discharge requires the most effort and data. In light of this, practitioners desire a standardized procedure for calculating the effective discharge with practical guidance for data collection and processing.

These issues represent real challenges to restorers wishing to use the effective discharge as a design flow and impose potential constraints on the use of magnitude-frequency analysis in practice.

The magnitude and recurrence interval of the effective discharge are functions of the flow frequency distribution (usually represented by a histogram of measured discharges), the sediment rating curve and, most importantly, how the flow and sediment regimes represented by these two relationships interact.

The most significant influence on the effective discharge is often the degree and type of skewness in the flow frequency distribution. Negatively skewed distributions indicate a highly variable, flashy regime. In a flashy flow regime, a greater proportion of the sediment load is likely to be transported by infrequent, high magnitude flows. This explains why major flood flows are channel-forming events in semiarid and arid regions [*Wolman and Miller*, 1960; *Werrity*, 1997], particularly for streams with resistant boundaries that render more frequent, in-bank flows ineffective in shaping the channel [*Harvey*, 1969; *Baker*, 1977].

Wolman and Miller [1960], *Baker* [1977], *Andrews* [1980], and *Andrews and Nankervis* [1995] reported that negative skewness in flow frequency increases as drainage basin area decreases, so that, in very small catchments, the effective discharge is likely to correspond to a low frequency event. However, the influence of watershed area was found to be insignificant by *Whiting et al.* [1999] and *Torrizo and Pitlick* [2004]. It may be the case that the lower frequency of effective discharges observed in smaller watersheds may stem simply from the associated increase in the discharge variance, with some evidence linking flow variability to bankfull depth [*Pizzuto*, 1986], possibly due to a greater number of events capable of exporting sediment onto the floodplain.

In streams that exhibit positively skewed flow frequency distributions (base flow dominated) but rarely experience discharges capable of overtopping their banks, high frequency, in-bank flows with relatively low stages may be the most effective in terms of sediment transport over a period of years, especially when the river bed material is highly mobile. Where this is the case, the overall form of the channel is related to events less frequent than the effective discharge based on magnitude-frequency analysis [*Harvey*, 1969]. The geomorphological significance of flows below bankfull, resulting in the effective discharge being smaller than the bankfull discharge, is supported by a number of field studies [e.g., *Benson and Thomas*, 1966; *Pickup and Warner*, 1976; *Webb and Walling*, 1982; *Nolan et al.*, 1987; *Lyons et al.*, 1992; *Whiting et al.*, 1999; *Orndorff and Glonek*, 2004; and others].

The significance of the sediment transport threshold and mobility of bed sediments was addressed by *Werrity* [1997], who noted that the streams studied by *Wolman and Miller* [1960] were predominantly sand bedded and that the effective discharge concept is most valid in these streams because the threshold discharge for sediment entrainment is low. Indeed, in channels with beds comprising easily mobilized, fine sands and positively skewed flow distributions, it is conceivable that base flow is the most effective discharge in terms of long-term sediment transport, especially where there is an abundant sediment supply [*Hey*, 1975]. However, as the entrainment threshold increases, the frequency of the effective discharge tends to decrease and, in gravel bed rivers, a significant proportion of the low flow distribution may be argued to be entirely ineffective. Extending this line of reasoning to cobble and boulder-bed streams with low stream powers and negligible sediment loads indicates that the effective discharge concept is inapplicable to such watercourses.

There is a tendency for the effective discharge to have a high magnitude and low frequency of occurrence if the sediment-rating curve has a steep gradient (a high exponent in the power relationship of sediment transport rate as a function of water discharge). As a result, in streams that transport fine sediment, predominantly in suspension, the effective discharge is lower and more frequently occurring (possibly less than the bankfull discharge) than in streams predominantly transporting coarse sediments as bed load [*Hey*, 1975].

Emmett and Wolman [2001] revealed that the ratio of effective to bankfull discharges in gravel bed streams ranged from 0.98 to 1.31 (representing a doubling of the recurrence interval), the ratio correlating significantly with the exponent of the bed load rating curve, which was shown to increase with bed surface particle size. They demonstrated that, in very coarse bed streams, flows above bankfull appear to be the most effective, in terms of transporting sediment. In line with these findings, *Whiting et al.* [1999] demonstrated that up to 37% of the bed load can be transported by flows above the bankfull discharge. The exponent in the bed load rating curve was found by *Emmett and Wolman* [2001] to be 2.5 when bankfull and effective discharges were equivalent. Similarly, *Quadar and Guo* [2009] discovered that the effective discharge has a recurrence interval of 1.5 years when the exponent was 3.5. Interestingly, if sediment transport rate only increases weakly with discharge, this could counter the influence of a high entrainment threshold on the frequency of the effective discharge [*Wolman and Miller*, 1960; *Andrews*, 1980].

In streams where the gradient of the sediment-rating curve is mild, there may be no discernible peak in the sediment load histogram derived through the magnitude-frequency analysis, indicating the existence of a range of geomorphic effective flows that, cumulatively, are responsible for shaping the channel and maintaining its morphological forms and features [e.g., *Biedenharn and Thorne*, 1994]. It is not surprising then that *Ashmore and Day* [1988], for streams in the Saskatchewan basin, Alberta, and *Nash* [1994], for American streams in a range of physiographic regions, concluded that that no generalization can be made regarding the recurrence interval of the effective discharge.

This discussion indicates that subtle changes in the character of the flow distribution and/or the shape of the sediment transport rating curve can have marked impacts on the magnitude and frequency of the effective discharge and its relationship to bankfull discharge. Owing to the combined influence of climatic, geologic, and physiographic factors, the frequency of the effective discharge can also vary along length of a watercourse as well as between streams [*Andrews*, 1980]. Application of the effective discharge concept may be inappropriate in very small catchments featuring very highly variable or strongly skewed flow distributions and streams with boulder or cobble beds and very low sediment transport rates at all discharges below bankfull.

Summarizing, the effective discharge methodology is the most advanced and scientific representation of the channel-forming or dominant discharge, but it is also the most demanding in terms of data and is subject to considerable uncertainty when the input data are synthesized for ungauged sites. Magnitude-frequency analysis involves subjective decision making [*Crowder and Knapp*, 2005; *Lenzi et al.*, 2006], and despite wide support among river restoration practitioners, application of the effective discharge theory for stable channel design remains problematic in many situations. However, it is encouraging that 50 years after Wolman and Miller's groundbreaking paper on the geomorphic effectiveness of floods, research on this important topic continues, with new representations of the effective discharge forthcoming.

For example, *Emmett and Wolman* [2001], *Vogel et al.* [2003], and *Klonsky and Vogel* [2011] have found close agreement between the effective discharge and the half-load discharge, which is defined as the discharge above and below which 50% of the overall sediment load has been transported over time, while *Copeland et al.* [2005] found that the 75th percentile flow on the cumulative sediment transport curve provides an improvement in the relationship with bankfull discharge compared to that for the conventionally calculated, effective discharge (Figure 3).

Finally, *Doyle and Shields* [2008] recently introduced the "functionally equivalent discharge" as the single flow that would produce the same sediment yield as that generated by the entire range of discharges actually experienced by the river. This approach is commendable for attempting to account, albeit indirectly, for all the flows capable of performing geomorphic work through transporting bed material; an aspiration for channel restoration design that continues to elude any of the currently employed, rational, scientific methods.

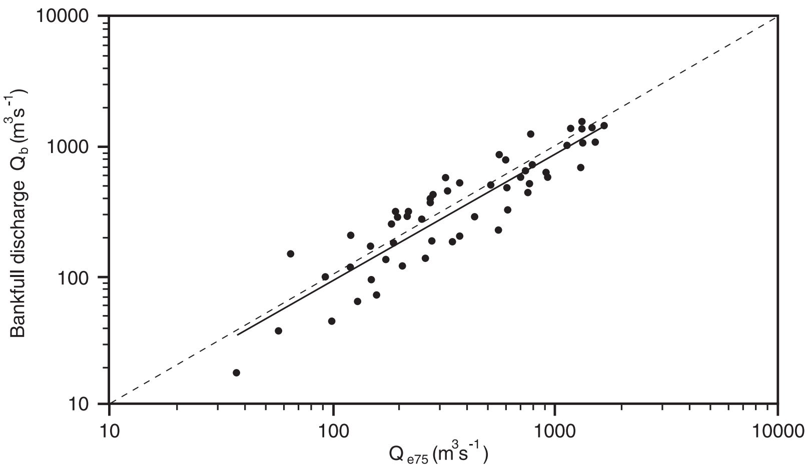

Figure 3. Relationship between the discharge marking the upper limit of the range of discharges that cumulatively transport 75% of the average annual bed material load, Q_{e75}, and the bankfull discharge, Q_b, for 57 American sand bed rivers. Solid line is the best fit power relationship. Dotted line marks equality [*Copeland et al.*, 2005].

3.3.3. Standardized procedure for calculation. The challenges of deriving an effective discharge value in practice have been highlighted by *Orndorff and Whiting* [1999] and *Shields et al.* [2008] who demonstrated the use of statistical software to facilitate the calculations. A standardized procedure for calculating the effective discharge has been proposed and described in detail by *Biedenharn et al.* [2000, 2001] who argued that a procedure is required in order that effective discharges for different sites may be compared and so that practitioners know how to avoid common data collection and processing pitfalls that introduce uncertainty into a magnitude-frequency analysis. Also, while the procedure appears relatively straightforward, in practice, there are a number of potential difficulties with the assimilation, processing, and interpretation of data; meaning that the effective discharge is not only sensitive to the availability and caliber of data, but also influenced by decision making during the analysis.

Biedenharn et al.'s [2000, 2001] procedure adheres to the approach of *Wolman and Miller* [1960] and involves three stages: (1) constructing a frequency distribution of discharges, (2) constructing a sediment transport rating from measured data or using an appropriate sediment transport equation, and (3) integrating the two relationships by calculating the sediment transport rate (in units of mass per year) for the median value of each discharge class and then multiplying that rate by the frequency of occurrence of that discharge to yield a histogram of average annual sediment yields for the range of discharge classes. The effective discharge is then defined as the median discharge of the modal class in the sediment load histogram.

Two obvious constraints on deriving the effective discharge stem from the limited availability of gauged flow records and measured sediment transport rates for the great majority of candidate river restoration project sites. However, if the restoration site is close to a gauging station, a flow frequency distribution can be derived from the record of measured discharges. The quality of the distribution depends upon the reliability of the gauged discharges (particularly measurements at very low and high stages), the period of record and the time-base of the recorded discharges. Ideally, the period of record should be at least 10 years, though for long periods of record, care should be taken to ensure that the record is representative of the prevailing hydrology by checking for nonstationarity in the flow regime. To accurately capture the magnitude of the peak flows, hourly, or better still, 15 min data (as collected by the USGS) are essential, as the more readily available, mean daily discharge values can significantly underestimate instantaneous peak flows and sediment transport associated with high magnitude events in small- and medium-sized catchments. These recommendations for assimilating gauged flow data are also valid for calculating the dominant discharge as an event with a specified recurrence interval (see section 3.2.2).

The effective discharge can be sensitive to the number of discharge classes used to generate the flow frequency distribution [*Orndorff and Whiting*, 1999; *Sichingabula*, 1999; *Crowder and Knapp*, 2005; *Lenzi et al.*, 2006]. *Biedenharn et al.* [2000, 2001] and *Soar and Thorne* [2001] recommend starting with 25 discharge classes and then applying an iterative process of adjusting the number to achieve intervals that are as small as possible while maintaining a "smooth" frequency distribution. The minimum discharge should be set to zero in streams transporting fine sediment in suspension and to the critical discharge for the threshold of bed load motion for coarse-bedded channels. Uniform, arithmetic discharge classes must be used to prevent bias [*Soar and Thorne*, 2001], which means that the effective discharge might fall within the first class when the flow distribution is base flow-dominated and highly skewed toward the smaller flows. Increasing the number of discharge classes may provide improved resolution of the effective discharge when this is the case.

Recognizing that discretizing the discharge series can influence the magnitude and frequency of the effective discharge by masking the true variability and episodic nature of sediment transport events, several researchers have sought alternatives to conventional "class-based" calculation of the effective discharge. *Sichingabula* [1999] recommended calculating an "event-based" effective discharge, defined as that with the maximum sediment load considering all of the individual events, and *Goodwin* [2004] and *Klonsky and Vogel* [2011] experimented with analytical solutions using theoretical probability density functions to represent the distributions of discharges and sediment loads. A different approach was outlined by *Soar and Thorne* [2001] in which very small discharge intervals (potentially hundreds or thousands of classes) could be employed by discretizing an event-based flow duration curve (cumulative flow distribution), rather than developing a flow frequency histogram directly from the raw discharge series, and then identifying a quasi-event-based effective discharge through repeated, moving average, smoothing of the resultant sediment load histogram.

These methods can more accurately describe the empirical distribution of sediment transport effectiveness and so potentially overcome some of the common limitations of the methodology. However, further research, testing, and standardization would be required before these innovative approaches could be considered for routine application to river restoration design based on gauge records from hydrologic stations.

At ungauged sites, and gauged sites where the flow record is considered unreliable or unrepresentative, it is necessary to "synthesize" a flow distribution. There are two approaches to achieving this, both involving the transfer of gauged flows from nearby gauging stations within the same watershed or analog sites in watersheds with similar physiographic and hydrologic characteristics. The approaches are (1) drainage area-flow duration curve method and (2) regionalized flow duration curve method.

The first approach involves fitting power relationships to data sets linking discharge exceedance duration to upstream drainage basin area, ideally based on data from several gauging stations [see *Hey*, 1975]. The second approach involves scaling flow duration according to a nondimensional discharge index such as the ratio of discharge to the 2 year flow, as proposed by *Watson et al.* [1997]. Reference should be made to the works of *Biedenharn et al.* [2000, 2001] for further details on these methods.

Extrapolation of flow duration curves from gauged to ungauged sites can also be achieved by using bankfull discharge as the normalization parameter, although this can introduce additional uncertainty as calculating the bankfull discharge is itself subject to error and, in any case, been shown to have an inconsistent recurrence interval. An alternative approach is available that derives dimensionless flow duration statistics scaled on the mean flow for a suite of regions identified as sharing similar watershed characteristics [*Holmes et al.*, 2002]. This approach has been adopted in a component of the Low Flows 2000 suite of hydrologic models for use in England and Wales [*Young et al.*, 2003].

In developing the sediment rating curve for an effective discharge calculation, it is the bed material load that should be used, rather than the total load, as this excludes the wash load component. The bed material load is the proportion of the total sediment load composed of grain sizes found in appreciable quantities in the stream bed. It should be noted that in gravel bed rivers, the bed material load moves as bed load, but in sand bed streams, significant quantities of bed material load are distributed through the water column as suspended load. The wash load is the portion of the total sediment load composed of grain sizes finer than those found in appreciable quantities in the stream bed, with the 10th percentile in the bed sediment particle size distribution often taken as the boundary between the wash load and bed material load components of the total load. It is usually assumed that wash load plays no significant role in shaping the channel, passing through the reach, but not long residing there.

When measured, suspended load data are available from a gauged site; particles finer than sand (that is less than 0.062 mm) should be excluded when deriving the sediment rating curve as these are likely to constitute wash load only. Routine bed load measurements are rare, but if a data set does exist, it can be combined with the coarse fraction of the measured suspended load to produce a better representation of the bed material load. Typically, sediment rating curves have the form of a power relationship expressing sediment concentration or transport rate as a function of discharge, although in some cases, two or even three log-log segments are necessary to describe the relationship adequately [see *Simon et al.*, 2004; *Shields et al.*, 2008].

When measured sediment transport data are unavailable, a suitable sediment transport equation can be used to synthesize a rating curve. If the bed material load moves predominantly as bed load (as in gravel bed rivers), then a dedicated bed load transport equation should be used. Alternatively, other equations are available that account for both the bed load and suspended load components of the bed material load. While there are a range of equations available to the practitioner [e.g., see *Yang*, 1996], selecting the equation best-suited to the type of river and bed material is critical to minimize uncertainty in calculated sediment loads. In this context, the Stable channel Analytical Method (SAM) hydraulic design package [*Raphelt*, 1990; *Thomas et al.*, 2002] provides useful guidance on matching the equation selected to the scale of stream and type of sediment involved. However, it should be remembered that uncalibrated calculations of bed material load are prone to substantive uncertainty. In practice, the absolute magnitudes of calculated sediment loads will vary markedly depending on the sediment transport equation selected, and experience shows that calculated loads are unlikely be within ±50% of the actual load more than 70% of the time. Recognizing this, it is fortunate that prediction of the effective discharge based on the modal class in the bed material load histogram has been shown to be insensitive to both the choice of sediment transport relationship [*Barry et al.*, 2008] and the coefficient in the sediment transport rating curve [*Goodwin*, 2004].

The bed material load histogram should display a continuous distribution with a single modal discharge class, and conventionally, the effective discharge corresponds to the median discharge of the modal class. Alternatively, the effective discharge can be estimated by drawing a smooth curve through the tops of the histogram bars and inferring the effective discharge from the peak of that curve.

As checks on the reliability of the calculation, the magnitude of the effective discharge should be compared to that of the bankfull discharge, where available, and predicted effective discharges with recurrence intervals outside the range of 1 to 3 years, based on the AMS, should be queried and possibly reexamined. Finally, it is recommended that a cumulative frequency curve be plotted from the bed material load data to identify other potentially important flows and the possible existence of a range of effective discharges, as indicated by breakpoints in the gradient of the curve [after *Biedenharn and Thorne*, 1994].

4. PROGRESS AND PROSPECTS: TOWARD THE USE OF MULTIPLE DESIGN DISCHARGES

While the effective discharge is clearly important geomorphologically, unless it relates closely to the bankfull discharge, its utility as a design discharge for river restoration may be limited. There is clearly a need for further, concerted research to provide improved guidance on the application of magnitude-frequency analysis and to develop objective methods of predicting and accounting for differences between the effective and bankfull flows.

As a rule of thumb for meandering sand bed rivers in the United States, *Soar and Thorne* [2001] found that mean annual and bankfull discharges, respectively, formed the lower and upper bounds to a range of effective flows. The effective discharge was found to be less than bankfull at 86% of the sites studied. *Biedenharn and Thorne* [1994] also demonstrated for the lower Mississippi River that the longitudinal water surface profile at the upper limit of the range of effective flows (with a recurrence interval of 5 years) coincided with the upper boundary of the range of top of bank elevations.

These findings challenge the existence of a single channel-forming flow, in that the effective discharge in sand bed streams only appears to correspond to the bankfull discharge in certain cases. In fact, based simply on numerical analysis, *Soar and Thorne* [2001] hypothesized that equivalence was unlikely because the effective discharge corresponds to the inflection point (point of steepest gradient) in the cumulative sediment load frequency curve (the cumulative distribution of sediment yield as a function of discharge, derived through the magnitude-frequency analysis), whereas the bankfull discharge tends to coincide with the upper breakpoint in the curve, which is associated with the transition from in-channel to overbank flow, and a marked discontinuity in the sediment rating curve due to the break in bank slope, rapid increase in width, increased flow resistance on the floodplain, and exchange of momentum between in-bank and overbank flows (Figure 4).

Research on large rivers with both single-thread and multi-thread planforms has provided some support for this hypothesis, whereby the effective discharge has been shown to correspond to an elevation at the top of channel bars (i.e., barfull discharge), at a stage well below bankfull [see *Latrubesse*, 2008]. This phenomenon is demonstrated in the results of magnitude-frequency analysis of the Brahmaputra River, Bangladesh [*Thorne et al.*, 1993], confirming also that the effective discharge has morphological significance in braided as well as meandering rivers and suggesting that it might provide a useful design flow in the restoration of multithread as well as single-thread channels.

Further analysis of the data set of American sand bed rivers compiled by *Soar and Thorne* [2001] revealed that the variance in discharges appears to exert an important influence on the magnitude and variability of the ratio between bankfull discharge, Q_b, and effective discharge, Q_e. Specifically, this ratio appears to be largest when the flow distribution is skewed toward small discharges, as represented by the ratio of the 2 year peak flow, Q_2, to the mean annual (time averaged) discharge, Q_m. The best fit relationship is a power function (Figure 5) which explains 73% of the variance in Q_b/Q_e, and is given by

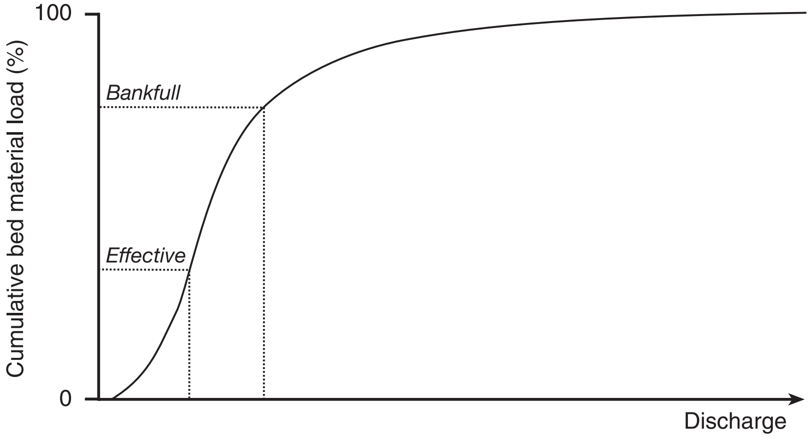

Figure 4. Hypothetical curve of cumulative bed material load as a function of discharge, derived from magnitude-frequency analysis, showing the locations of the effective discharge at the inflection point and the bankfull discharge at the upper break point [*Soar and Thorne*, 2001].

$$\frac{Q_b}{Q_e} = 0 \cdot 16 \left(\frac{Q_2}{Q_m} \right)^{1 \cdot 35}. \qquad (2)$$

In base flow–dominated streams with infrequent flood flows, it appears that the small to intermediate floods that occur frequently in between the high-magnitude events are highly effective in transporting sediment over a period of years. In such cases, the bankfull flow, rather than the effective discharge, might be a better representation of the dominant discharge, in that it exerts a stronger influence on and corresponds more closely with the channel morphology.

The impact of flow variability on discordance between the bankfull and effective discharges is illustrated in Figure 6 for three of the American sand bed streams analyzed by *Soar and Thorne* [2001]. However, while equation (2) provides initial guidance for channel restoration design in sand bed rivers, further research is strongly recommended to verify and develop this approach.

In addition to more deterministic understanding of the discordance between effective and bankfull discharges, further development of regional curves used for predicting bankfull discharges should be encouraged, with research focused on the evaluation of uncertainties, broadening of the databases from which regional curves are developed within

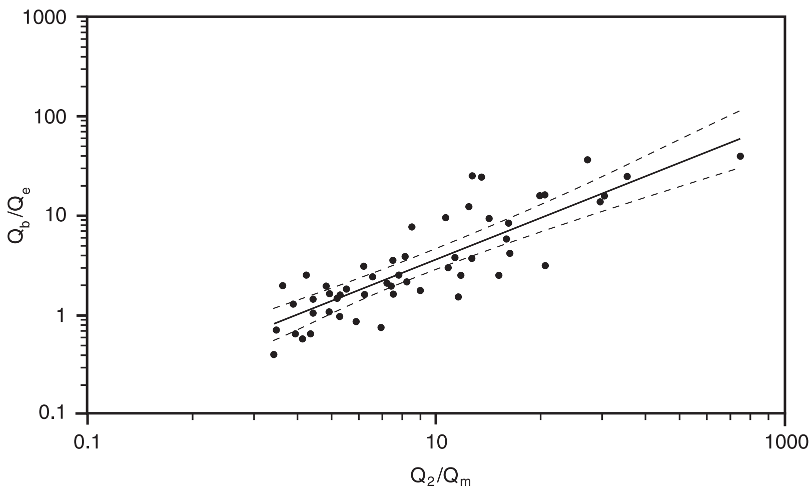

Figure 5. Ratio between bankfull discharge, Q_b, and effective discharge, Q_e, for American sand bed rivers expressed as a function of flow variability, defined as the ratio between the 2 year recurrence interval flow, Q_2, and the mean annual (time averaged) discharge, Q_m [*Soar and Thorne*, 2001].

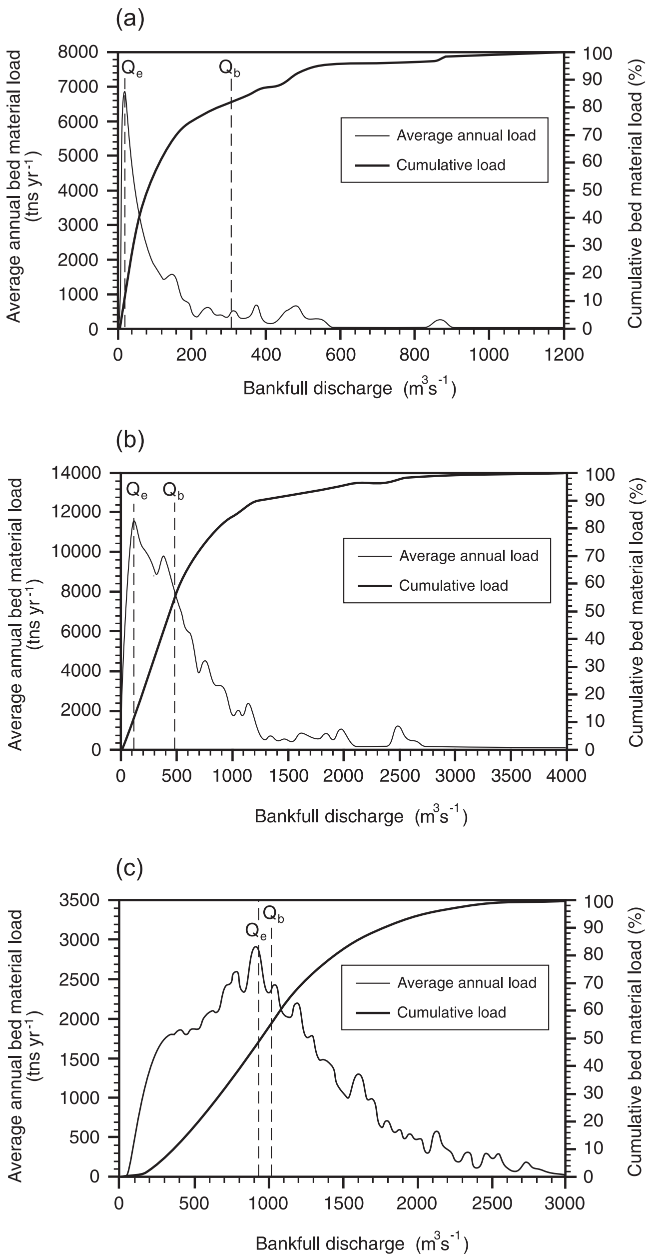

Figure 6. Bed material load histograms and cumulative sediment curves for: (a) the East Nishnabotna River at Red Oak, Iowa (Q_2/Q_m = 20.3), (b) the Tombigbee River near Amory, Mississippi (Q_2/Q_m = 11.4), and (c) the Wabash River at Riverton, Indiana (Q_2/Q_m = 4.5) [*Soar and Thorne*, 2001].

and between regions and clear identification of limitations to applicability of the concept in restoration design.

The three design approaches outlined and discussed here offer river restorers flexibility in specifying a design discharge, but cannot account objectively for the constructive, destructive, and restorative impacts of the "range" of flows which are actually likely to occur in nature and which are recognized as important in shaping the plethora of morphological features found in alluvial stream channels [*Wolman and Gerson*, 1978; *Yu and Wolman*, 1987].

For mountain streams, both *Phillips* [2002] and *Lenzi et al.* [2006] described the occurrence of two potentially dominant discharges that can exert significant geomorphic impacts: a relatively frequent discharge responsible for maintaining the channel, shaping in-stream sediment features, and preventing significant accumulations of fine sediment, and a second, less frequently occurring discharge responsible for shaping the channel's banks, controlling its width and configuring its planform. The existence of multiple formative discharges was corroborated in research on the Tagliamento River, Italy, by *Surian et al.* [2009], where flows less than half bankfull discharge appear to be formative for the channel bed sediment, the bankfull discharge (with just over a 1 year recurrence interval) is formative with respect to low elevation bars, while larger events (with recurrence intervals of up to 5 years) are the most effective for gravel transport on the high bar features and are responsible for morphological changes to the islands.

In their original treatise on magnitude-frequency analysis, *Wolman and Miller* [1960] clearly stressed that the channel shape is affected by a *range* of flows rather than a single, formative flow. It follows that channel reconstruction should be based on the precept that every competent flow event that occurs exerts some influence on channel form and that the shape and dimensions of the channel at any time are the weighted sum of the effects of all the preceding discharges [*Pickup and Reiger*, 1979]. However, at present, the science base underpinning channel restoration design is insufficiently advanced to support morphological modeling of the complex process-form interactions involved in the semicontinuous evolution of channel morphology that occurs in natural, alluvial streams.

Recognizing this, a feasible first step in accounting for the significant impacts of competent flows other than the single, effective discharge would be careful inspection of the cumulative sediment transport curve to identify the *range of effective flows* responsible for transporting the great majority (say 70–80%) of the sediment load and, importantly, break points in the cumulative curve associated with sedimentary and morphological features in the cross section that have particular ecohydraulic and hydromorphological significance (e.g., the base flow channel, low and high bar tops, bankfull stage) in dynamically stable, alluvial channels [*Biedenharn and Thorne*, 1994; *Surian et al.*, 2009]. This would allow restorers to incorporate not one but a series of nested design discharges into their restoration plan.

A more ambitious approach to accounting, first, for the full spectrum of competent flows and, second, for the fact that the future occurrence, timing, and sequencing of events cannot be predicted or even known, is to perform a series of future channel stability assessments to check that probable design outcomes are acceptable with respect to erosion, sedimentation, and channel evolution [see *Soar and Thorne*, 2001; *Shields et al.*, 2003, 2008].

In this context, the concept of "total restoration potential" may be useful [*Downs et al.*, 1999]. This approach has been used to assess the anticipated performance of in-stream river rehabilitation structures and is based on the ability of the natural sequence of flows to transport the quantity of sediment necessary to modify the channel morphology significantly. By adopting this approach, the potential for geomorphic success of a restored channel can be assessed according to whether the average annual bed material load in the restored channel (transport capacity) matches the mean annual bed material load input from upstream (sediment supply). As the annualized computations can be performed quickly once the supply-capacity model has been developed, it is possible to run them for a large number of possible hydrologic futures, featuring selected frequencies and sequences of transport events, in effect, to model multiple scenarios for the hydrologic and sediment loadings imposed on the restored reach.

Use of this design closure loop would not only validate the efficiency and resilience of the restored channel geometry but also identify particular events or combinations of events likely to destabilize the channel and, so, facilitate the design modifications necessary to reduce the risk of a loss of dynamic stability in the medium- to long-term to a tolerable level. Testing the sensitivity of a restoration design to future sediment impacts based on analysis of a range of realistic possible scenarios seems likely to become an essential component of restoration design as it becomes clearer that future flow and sediment regimes will be different from those of the past, an inevitable response to global warming and ongoing changes in watershed land use.

Consideration of the ecological significance of the design flows alongside their morphological significance is further emphasized by emerging evidence that a suite of flows must be considered for successful restoration of the diversity of physical habitats necessary to support sustainable ecological functioning in a restored river [e.g., *Kondolf et al.*, 2001; *Doyle et al.*, 2005; *Smith and Prestegaard*, 2005]. The significance of both low and flood flows to riverine ecosystems is now well established [*Poff et al.*, 1997; *Postel and Richter*, 2003] and is manifest in the emerging field of "ecohydrology" [*Hannah et al.*, 2007]. Clearly, channel restoration designs will, in future, have to account fully for the diverse ecological roles of flows other than the channel-forming discharge. In this context, it is significant that new guidance for federal and state services staff responsible for permitting river restoration proposals in rivers draining to the west coast of the United States [*Skidmore et al.*, 2011] stresses that restoration designers must demonstrate a thorough understanding of the entire flow regime before being permitted to proceed to construction.

The importance of in-channel fluvial features is widely recognized, particularly during summer low flows (often measured by the 95th percentile flow or the mean annual minimum 7 day flow) that may limit the combinations of depth and velocity necessary for particular species or life stages. Habitat diversity at low flows is often created through the construction of in-stream structures, such as weirs and flow deflectors that are sited significantly below the bankfull level. Interactions between these structures, the flow field and sediment dynamics at discharges well below the conventional design flow are responsible for generating the desired patterns of velocity, depth, scour, and fill. However, in-stream rehabilitation structures can adversely impact channel conveyance capacity, and this aspect of their functioning must also be addressed in their design. Clearly, it is essential to consider multiple discharges in the design, testing, and appraisal of restoration schemes that employ in-stream structures [*Downs and Thorne*, 1998].

This is not to underestimate the significance of flood flows with recurrence intervals considerably longer than that of bankfull discharge, which impart numerous advantages to riverine ecology, at multiple scales within the fluvial hydrosystem [*Petts and Amoros*, 1996]. These "environmental maintenance flows" [*Whiting*, 2002] are crucial in several ways, including "power washing" coarse bed materials to remove suffocating blankets of fines, removing overly mature bank and riparian vegetation, depositing sediment, plant seeds, and propagules on floodplains, recharging floodplain aquifers, improving the productivity of floodplain habitats and driving wetland dynamics.

Many restoration schemes are implemented in channels with multiple functions, requiring designs that balance targets for ecology and biodiversity with those for flood control, land drainage, and channel stability. Restoration designs for such multifunctional restorations commonly employ multistage channels comprising a "regime" channel, sized to convey the channel-forming (design) discharge, within a wider floodway, sized to convey a much larger flood event with a designated recurrence interval. As noted above, in such situations, the need to promote habitat diversity and sustain fish passage during critical low flows is often addressed through the construction of in-stream structures within the "regime'" channel.

Designing these complex channel configurations requires optimization of fluvial conditions at multiple flow stages based on design discharges that cannot be derived using the conventional, regime-based methods described herein. This is the case because regime approaches cannot account for the significant energy exchanges that take place at the interfaces between the inner channel, regime channel, and the high stage floodway, and therefore cannot properly mitigate against the risk of lesser channels being infilled during sediment transporting events that overtop them or channel scour due to elevated boundary shear stresses when flood flows are contained between levees bounding the floodway. Recognizing this, there is a strong research need for improved understanding and modeling capability in the design of multistage channels.

5. CONCLUSIONS

Each of the approaches to defining a design discharge for river restoration described here employs different arguments to support the case that it can adequately represent the "dominant discharge" or channel-forming flow for restoration design purposes.

The case for the bankfull discharge rests on its clear morphological association with the capacity and dimensions of stable channels that are "in regime." Selection of a flow with a recurrence interval of 1 to 2 years is supported by the premise that the dominant discharge must occur sufficiently often for alluvial river channels to maintain regime dimensions most of the time, coupled with the widely established similarity in the recurrence intervals for bankfull flows in a variety of river types. The effective discharge concept seeks to integrate the sediment transport processes responsible for doing work on the channel and so forming its dimensions.

In practice, each of these approaches has been demonstrated to have some utility in the river restoration design process. However, as discussed in this chapter, relationships between the design flows produced by the different prescribed methods remain deeply equivocal, and none of them can be applied routinely or universally.

Recognizing this, it is recommended that river restoration projects employ all applicable methods, so that the results can be cross-checked against each other to improve confidence that the selected design discharge does adequately represent the channel-forming flow.

Looking ahead, effective discharge analysis has considerable potential for further advances in computational methods that could provide improved insights into the morphological significance of different discharges within the effective range of flows and so increase their utility in restoration design. In addition, further development of regional curves used for predicting bankfull discharges should be encouraged, with research focused on the evaluation of uncertainties, broadening of the databases from which regional curves are developed within and between regions, and clear identification of limitations to applicabilily of the concept for river restoration.

In conclusion, river restoration design must work toward improving the biological integrity and sustainability of degraded riverine ecosystems by mimicking not only the morphological diversity that is appropriate to the type of restored channel within what is usually a modified watershed setting [*Dufour and Piégay*, 2009] but also restoring the fluvial processes that sustain the ecological functionality of the stream. This requires restoration goals that center on the creation of an allied distribution of patches and ecological spaces within the fluvial hydrosystem rather than focusing on target species or habitats that may or may not be sustainable geomorphologically.

Such goals will continue to prove elusive until the scope of restoration expands from the channel to the riparian corridor and, ideally, the "functional floodplain." Adoption of corridor and floodplain templates for restoration will inevitably lead designers away from the use of single-value design discharges, generating demand for new approaches that account for ranges and suites of design discharges that are not only morphologically effective but also ecologically appropriate. In short, restoring not only heterogeneity but also the capacity for dynamic adjustment of the river channel's boundaries, sedimentary features, planform configurations, and floodplain environments will provide a near-term research impetus that will require improved design discharges capable of simultaneously supporting goals for morphological reconstruction and ecological restoration.

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Scale-Dependent Effects of Bank Vegetation on Channel Processes: Field Data, Computational Fluid Dynamics Modeling, and Restoration Design

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Bank vegetation substantially influences flow resistance, velocity, shear stress distributions, and geomorphic stability in many natural river settings. We analyze field data from gravel bed streams with typed bank vegetation characteristics and employ three-dimensional computational fluid dynamics (CFD) modeling to examine whether the effects of bank vegetation on channel form and processes are scale dependent. Field data from the United States and United Kingdom indicate that mean bankfull dimensionless shear stresses are significantly higher in channels with thick woody vegetation but only for channel widths less than ~20 m. Because specific mechanisms controlling the apparent scale dependency are difficult to isolate in natural channels, we develop CFD models of streams with coarse beds and bank vegetation to investigate physical processes in channels with variable bed and bank roughness. The CFD models are applied in two sets of simulations to improve mechanistic understanding of patterns in the field data and to examine (1) spatial scale dependency between channel width and vegetation effects and (2) the coevolution of flow hydraulics, channel form, and vegetation establishment. The scale-dependent bank vegetation effects on shear stress distributions in the CFD representations are consistent with field data from gravel bed streams and suggest that the length scale of bank vegetation protrusion relative to channel width is an important factor that could improve shear stress partitioning models. In general, the field data and CFD simulations indicate a significant scale-dependent effect of bank vegetation with important implications for stream restoration designs based on tractive force, regime, and analytical approaches.

1. INTRODUCTION

Bank vegetation along streams and rivers performs important ecological and geomorphic functions by influencing flow hydraulics, channel form and stability, and habitat diversity. Stream restoration plans frequently include reestablishment of bank vegetation to increase geotechnical stability of banks due to root reinforcement [*Abernethy and Rutherfurd*, 2000; *Simon and Collison*, 2002], influence flow patterns in streams and decrease near-bank velocities [*Thorne and Furbish*, 1995]. Decreased flow velocities in the vicinity of vegetated banks can significantly alter distributions of shear stress and sediment transport across the entire channel

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and particularly in the near-bank region. Many researchers have found riparian vegetation significantly influences channel form, including hydraulic geometry [*Charlton et al.*, 1978; *Andrews*, 1984; *Hey and Thorne*, 1986; *Soar and Thorne*, 2001], planform characteristics [*Millar*, 2000], and scour pool characteristics [*Gran and Paola*, 2001]. Bank and floodplain vegetation also affect stage-discharge relationships in rivers, and specific methods for estimating flow resistance due to riparian vegetation have been previously developed [*Darby and Thorne*, 1996; *Darby*, 1999].

Previous work on the influence of bank vegetation on flow hydraulics has typically focused on empirically estimating increases in channel resistance due to vegetation [*Coon*, 1998], partitioning resistance between vegetated and nonvegetated portions of the channel [*Darby and Thorne*, 1996; *Darby*, 1999; *Yen*, 2002], or computing average flow profiles over or through vegetation [*Fischenich*, 1996]. *Fischenich* [1997] and *Lopez and Garcia* [1997] provide reviews of many of the basic qualitative and quantitative methods of accounting for channel vegetation.

Previous research also suggests that the influence of bank vegetation is scale dependent [*Anderson et al.*, 2004]. For example, *Coon* [1998] investigated the influence of bank vegetation on Manning's roughness coefficient and concluded that the influence was minimal in channels wider than about 20 m and could not be discerned in channels wider than approximately 30 m. Likewise, modeling by *Masterman and Thorne* [1992] indicates that the effects of riparian vegetation on flow hydraulics is limited when width-to-depth ratios increase beyond about 15. A better understanding of the fundamental processes controlling channel form and stability in terms of scale-dependent influence of vegetation on flow hydraulics would benefit channel evolution modeling, characterization of material fluxes and instream habitats, and stream restoration design.

Although field investigations could further clarify these processes, field measurements of detailed flow fields in the vicinity of vegetation are difficult to collect due to the infrequency of high-flow events that inundate vegetated banks and the associated measurement difficulties during high flows. Furthermore, finding multiple stream reaches with similar characteristics is challenging due to differences in vegetation characteristics, nonvegetative form roughness, sediment supply, flow regime, anthropogenic disturbance, and other influences. Physical modeling provides another tool for analyzing the influence of vegetation, but can be costly and time consuming. Selecting appropriate variable test ranges and scaling factors for vegetation and channel dimensions can also be challenging. Numerical modeling is an attractive alternative, provided that vegetation can be appropriately represented.

Computational fluid dynamic (CFD) modeling offers an increasingly viable means of analyzing the influence of bank vegetation on channel hydraulics and form in a variety of applications. With a growing number of CFD software packages and advances in computational efficiency, two-dimensional (2-D) and three-dimensional (3-D) modeling of river flow problems have become more feasible and widely utilized. CFD has been utilized to investigate many complex flow situations typical to natural channels [e.g., *Bradbrook et al.*, 1998; *Nicholas and Sambrook-Smith*, 1999; *Booker et al.*, 2001; *Morvan et al.*, 2002; *Rameshwaran and Naden*, 2003; *Nicholas and McLelland*, 2004], model erosion and sedimentation patterns [e.g., *Wu et al.*, 2000; *Shams et al.*, 2002], and study habitat suitability for fish species [e.g., *Crowder and Diplas*, 2000; *Booker*, 2003]. In CFD modeling, detailed 3-D velocity fields and shear stress distributions can be resolved for a given channel geometry. CFD modeling also offers the advantage of allowing the investigation of a variety of cases and scenarios with less labor and expenses than would be required to complete comparable physical experiments and can be used to design and optimize physical modeling efforts. The commercially available CFD package FLUENT [*Fluent Inc.*, 2003] has been widely applied to model open-channel flows [e.g., *Nicholas and Sambrook-Smith*, 1999; *Nicholas*, 2001; *Gessler and Meroney*, 2002; *Ma et al.*, 2002; *Shams et al.*, 2002; *Prinos et al.*, 2003; *Nicholas and McLelland*, 2004].

In this study, we (1) analyze field data from gravel bed streams and rivers in the United States and United Kingdom with various bank vegetation characteristics and (2) employ 3-D CFD modeling in FLUENT to examine the influence of bank vegetation in gravel bed channels and whether the effects of bank vegetation on key hydraulic parameters used in restoration design of gravel bed streams are scale dependent. We hypothesize that in relatively narrow channels, dense woody vegetation protruding appreciably into the bankfull flow field results in significantly higher values of bankfull dimensionless shear stress (τ_*) and slope (S). In the CFD modeling experiments, a drag force representation of vegetation [e.g., *Fischer-Antze et al.*, 2001] is coupled with the porous treatment of bed roughness described by *Carney et al.* [2006] to investigate the influence of bank vegetation on flow hydraulics in trapezoidal channels. The resulting model is used to elucidate processes responsible for patterns observed in the field data and to specifically examine (1) scale dependency of vegetation influence in channels with similar characteristics but different widths and (2) changes in flow hydraulics following a succession of vegetation establishment along a channel. CFD modeling results are compared with data from natural channels of various scales with typed bank vegetation to examine the magnitude and scale dependency of hydraulic effects. Finally, implications for geomorphic analysis, CFD modeling, and restoration design of natural gravel channels with successional bank vegetation are discussed.

2. METHODS

2.1. Analysis of Field Data

We compared mean values of bankfull dimensionless shear stress (τ_*) for channels with different bank vegetation conditions using data from field studies of gravel bed streams and rivers in the western United States by *Andrews* [1984] and in the United Kingdom by *Charlton et al.* [1978] and *Hey and Thorne* [1986]. These studies were selected because bank vegetation was typed for each study site, and the data sets contain the necessary geomorphic and hydraulic data for examining τ_* as it varies with channel width and vegetation type. Bankfull dimensionless shear stress is referenced to the median grain size (D_{50}) and defined as

$$\tau_* = \frac{RS}{(G-1)D_{50}}, \tag{1}$$

where R is hydraulic radius, S is slope, and G is the specific gravity of sediment assumed to equal 2.65.

Throughout this chapter, bank vegetation conditions are referred to as "thick" or "thin." If percent coverage data were available, "thick" vegetation refers to bank vegetation qualitatively described by the researchers as forested, heavy, or thick vegetated bank conditions with greater than 5% tree/shrub cover. "Thin" vegetation refers to grass-covered banks, nonforested channels, or channels where tree/shrub coverage is less than 5%. It should be noted that thick does not equate to density, as grasses may be much denser than woody vegetation on a stem per area basis. Thus, the term thick is best described as a qualitative index of woody vegetation dominance (density, basal area, and coverage) that is directly related to the stiffness and length scale of bank roughness elements [*Anderson et al.*, 2004].

Mean comparisons of τ_*, relative submergence (R/D_{90}, R/D_{84}, and h/D_{84} for the *Andrews* [1984], *Charlton et al.* [1978], and *Hey and Thorne* [1986] data sets, respectively), substrate gradation D_{84}/D_{50} (D_{90}/D_{50} for *Andrews* [1984]), and S between channels of differing sizes and bank vegetation types were performed using Student's t tests [*Snedecor and Cochran*, 1989]. Following *Coon* [1998], channels with top widths <20 m were considered as one group and those ≥20 m as another. In each group, mean values of dimensionless shear stress were compared between channels with thick versus thin bank vegetation. In all cases, a test for equal variance was conducted using an F test, and a modified t test was performed when the variances were not equal. We used one-tailed tests for τ_* and S of channels <20 m wide, and two-tailed tests for all others.

To test for scale-dependent effects of thick bank vegetation on stable channel slope, we developed multivariate power function models of the form $S = f(Q, D_{84})$ where Q is discharge (m^3 s^{-1}), and D_{84} is the 84th percentile of grain size (m) for the *Hey and Thorne* [1986] and *Charlton et al.* [1978] data sets using standard multiple regression techniques. Models were fit with and without a "toggle" variable that shifted the model intercept for observations with both thick vegetation and a top width <20 m to test for a significant scale-dependent effect of vegetation. Sediment transport capacity was not included as a predictor variable as in the *Hey and Thorne* [1986] analysis because it could potentially confound the results due to a lack of shear stress partitioning that accounts for vegetation influences. The *Andrews* [1984] data were not included in the regressions because D_{84} values were not reported.

2.2. CFD Simulations of Bank Vegetation Influence

Two types of CFD modeling simulations were performed to examine some of the fundamental vegetative controls on channel hydraulics and to demonstrate the utility of the modeling approach developed in this study. The first set of simulations addresses the spatial scale dependence of the influence of bank vegetation in channels of varying width. The second set of simulations examines the evolving hydraulics of vegetated streams due to either natural regeneration of vegetation following a disturbance event (e.g., postflood) or accelerated vegetation establishment following rehabilitation or restoration activities.

Previous studies have not included a channel width versus lateral vegetation roughness length scale term to quantify potential scale-dependent effects on hydraulic behavior. The first set of simulations focused on our hypothesis that width relative to lateral protrusion of bank vegetation could be an important factor in channels <20 m wide. In these simulations, channel bank angle, slope, depth, grain-size distribution, and vegetation characteristics (shape extending into the channel and inertial loss coefficient) were all held constant for each run. The porous zone representing vegetation was given a C_DA_v value of 11.5 m^{-1} reported for natural channels by *Fischenich* [1996], where C_D is a drag coefficient, and A_v is a measure of vegetation density (L^{-1}). Using a periodic boundary condition, the discharge was adjusted to maintain a constant depth of flow, and channel slope was held constant for each run. The width of the channel was then varied between each configuration to simulate different channel

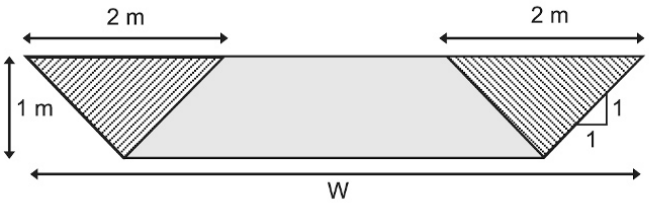

Figure 1. Channel configurations for the scale-dependent simulations. Width is the only channel characteristic changing among simulations (S_o = 0.001; D_{84} = 50 mm; C_DA_v = 11.5 m^{-1} for the vegetation on both banks). The hashed region on both banks represents the shape of the vegetation zone extending into the channel. The depth and slope were held constant, and discharge was varied.

sizes (Figure 1). For each configuration, one realization was performed with bank vegetation, and another was made without bank vegetation.

Following disturbances such as large floods, fires, grazing, or clearing, vegetation will often regenerate in stream riparian areas. Reestablishing vegetation on stream banks is also a common strategy of stream restoration. Whether natural regeneration or accelerated reestablishment due to rehabilitation efforts, bank vegetation typically is sparse immediately following disturbance or implementation of a restoration plan. Thus, the influence of bank vegetation on flow hydraulics evolves as density, stiffness, and protrusion change over time.

In the second set of model simulations, we simulated vegetative succession in a channel of fixed characteristics (width, side slope, bed roughness, and channel slope) with four vegetation treatments. In the first case, no vegetation was simulated, and channel roughness was limited to the porous treatment of the bed. In each subsequent scenario, simulated vegetation is included on the banks of the channel with incrementally greater amounts of protrusion. The protrusion of the simulated vegetation into the channel is increased in each scenario by lowering the minimum elevation of vegetation on the bank. The inertial loss coefficient of the vegetation zones is increased to simulate increasing flow resistance as vegetation matures and becomes denser and stiffer. Channel slope was set through the pressure gradient in the periodic boundary condition. Flow depth was then varied by iteratively regridding the domain and rerunning the solver until an equal discharge (±1%) was achieved for all scenarios.

2.3. Background on CFD Modeling Approach

Numerous researchers have recognized the potential CFD modeling provides for geomorphic analyses involving vegetation. In CFD, roughness is typically parameterized using a roughness length in some form of logarithmic velocity equation [e.g., *Hey*, 1979]. The representation of boundary roughness using a roughness length inadequately represents the physical processes creating flow profiles over and through vegetation. Alternative modeling methods have been proposed based on drag force representations of vegetation. *Shimizu and Tsujimoto* [1994] proposed the addition of a sink term in the governing momentum equations based on the drag due to vegetation. The drag force was parameterized as a function of vegetation density and drag coefficients. In the *Shimizu and Tsujimoto* [1994] model, two additional terms are included in the turbulent kinetic energy and turbulence dissipation rate equations of the k-ε turbulence model to account for the additional effects of vegetation on turbulence. These terms were also a function of the vegetation density and drag coefficients, but were treated as calibration parameters. Detailed flow observations through vertical rods in a laboratory flume were used to calibrate and verify their model.

Lopez and Garcia [1997] developed a similar vegetation representation based on atmospheric science studies [*Raupach and Shaw*, 1982]. Their model also used vegetation density and associated drag coefficients to represent vegetation, although they gave a theoretical basis for two additional terms included in the turbulent kinetic energy and turbulence dissipation rate equations. Their model was also compared to detailed velocity measurements in a flume setting, with vegetation represented as either stiff or flexible submerged dowels.

Fischer-Antze et al. [2001] noted differences among model coefficients used in the *Shimizu and Tsujimoto* [1994] and *Lopez and Garcia* [1997] studies, and postulated that the turbulent diffusive terms introduced by both modelers were of minor importance relative to the drag term in the momentum equations. In the *Fischer-Antze et al.* [2001] model, the same vegetation representation is utilized to include a drag force in the momentum equations without modifications to the turbulence equations of the k-ε model. Using the same data sets as *Shimizu and Tsujimoto* [1994] and *Lopez and Garcia* [1997], *Fischer-Antze et al.* [2001] added a third flume data set [*Pasche*, 1984] and demonstrated that velocity profiles of similar accuracy could be captured without turbulence model modifications. We also use these data sets to verify model simulations as described below.

Each of the above studies presented different vegetation modeling methods and compared the results with laboratory flume data, but did not extend those studies to natural channels or use the models to investigate the geomorphic influence of vegetation in open-channel flow. *Kean and Smith* [2004] present a simplified model based on an algebraic turbulence closure with vegetation represented using similar drag force concepts. After developing their model and presenting a verification based on the *Pasche* [1984] flume data set, *Kean and Smith* [2004] use the model to investigate the impacts of vegetation on shear stress distributions in straight, clay-bed, prismatic channels. Although the model may be simpler to implement, its applicability is limited to relatively simple channel geometries. In a different application, *Nicholas and McLelland* [2004] used CFD to model overbank flow through vegetation by implementing the *Fischer-Antze et al.* [2001] representation of vegetation, a roughness length representation of boundary roughness, and a renormalized group theory (RNG) k-ε turbulence closure model. *Nicholas and McLelland* [2004] recognized the potential to utilize a simple drag force representation of vegetation to reproduce velocity and turbulence profiles in the vicinity of vegetation, yet they noted the inherent challenges associated with determining appropriate model parameters to represent vegetation.

In the present study, the vegetation representation of *Fischer-Antze et al.* [2001] is implemented. Vegetative resistance is parameterized in terms of a drag coefficient and a measure of vegetation density:

$$\frac{F_{D,i}}{V_{\text{fluid}}} = \frac{1}{2}\rho C_D A_v u_{\text{mag}} u_i, \tag{2}$$

where $F_{D,i}$ is the drag force in the ith (x, y, or z) direction, V_{fluid} is the fluid volume over which the drag force is applied (equal to unity), ρ is density of the fluid-sediment mixture (assumed equal to 1000 kg m^{-3}), u_{mag} is the resultant reference velocity magnitude, and u_i is the reference velocity in the ith direction (the velocity which would be present if the stem being acted upon were removed from the flow [*Kean and Smith*, 2004]). If the resistance due to vegetation is due primarily to rigid stems that can be modeled as rigid cylinders, the value of A_v may be determined from:

$$A_v = nD_s = \frac{D_s}{\lambda^2}, \tag{3}$$

where n is number of stems per unit area, D_s is average stem diameter, and λ is average stem spacing (L). *Shimizu and Tsujimoto* [1994], *Lopez and Garcia* [1997], *Fischer-Antze et al.* [2001], and *Kean and Smith* [2004] all represent vegetative resistance in the form presented in equations (2) and (3). Alternatively, *Fischenich* [1996] and *Fischenich and Dudley* [2000] compiled numerous data sets (primarily *Rahmeyer et al.* [1995]) and presented methods for computing $C_D A_v$ for different riparian vegetation species. Their data sets are based on flume studies involving actual vegetation species assemblages. These sources provide an alternative means of estimating drag coefficients and representative areas for typical riparian vegetation without representing vegetation as a field of rigid cylinders. The range of representative $C_D A_v$ values used in the present study was extracted from *Fischenich* [1996].

The vegetative resistance computed according to equation (2) is included as a source term in the governing momentum equations. Assuming the turbulent diffusive terms due to vegetation are dominated by the vegetative drag term per *Fischer-Antze et al.* [2001], the drag force may be applied through the use of a porous media zone in FLUENT without modification to the turbulence models. *Carney* [2004] demonstrated that the vegetation representation used in this study could be coupled with the RNG k-ε turbulence model to reasonably reproduce the laboratory velocity profiles of *Tsujimoto et al.* [1991], *Dunn et al.* [1996], and *Pasche* [1984]. *Carney* [2004] also conducted grid-dependency tests to quantify the uncertainty in simulated results following *Hardy et al.* [2003] for the flume study verification model runs. The mean absolute percentage error in simulated downstream velocities among different resolution grids was less than 5% for all tested cases. The grids used for the case studies were of a similar resolution to those generated for the grid-dependency tests and contained 25 to 30 vertical cells with similar resolution horizontally. Cells were more closely concentrated in regions where higher velocity gradients were expected. A typical grid used in this simulation is shown in Figure 2.

2.3.1. Bed roughness representation. CFD modeling of coarse-grained channels can be challenging due to difficulties with roughness parameterization [*Nicholas*, 2001] as the maximum roughness length is limited to half the thickness of the near-bed cells [*Fluent Inc.*, 2003]. *Carney et al.* [2006] adapted the model of *Wiberg and Smith* [1991] to represent coarse-bedded channels in CFD. In this approach, the drag force per unit volume acting over the height of the D_{84} grain size is:

$$\frac{F_{D,\text{total}}(z)}{V_{\text{total}}} = c_b \frac{\frac{\rho}{2} C_D \frac{\pi}{4} D_{\text{my}} D_{\text{mz}} u(z)^2}{\frac{\pi}{6} D_{\text{mx}} D_{\text{my}} D_{\text{mz}}} = \frac{3}{4}\rho \frac{c_b C_D}{D_{84x}} u(z)^2, \tag{4}$$

where c_b represents the inverse of the average porosity of the bed, D_{mx}, D_{my}, and D_{mz} are the grain dimensions in

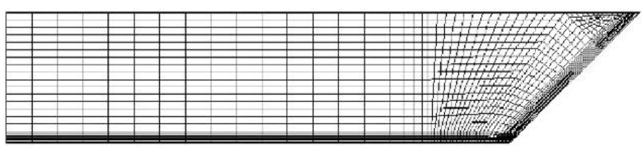

Figure 2. Computational grid (cross-section view, half channel) for the second set of simulations.

the downstream, cross-sectional, and vertical directions, respectively, and the drag force at a level z between the top and bottom of a grain, $F_D(z)$, is computed by replacing the average velocity with the velocity at the height z, $u(z)$. A zone is created adjacent to the bed the height of the D_{84} grain size, and a momentum sink term defined according to equation (2) is assigned to this zone. In the FLUENT application, the above may be accomplished using a porous media zone [*Carney et al.*, 2006]. Field observations indicate that c_b typically takes a value of about 0.6, which is used in this model [*Wiberg and Smith*, 1991]. The value of C_D is set to 0.45 based on drag relationships for spheres [*Coleman*, 1967].

Shear stress at a channel boundary is typically computed in CFD using a wall function. However, when bed roughness is represented using the above approach, the channel wall is located below the grains composing the bed. Therefore, shear stresses were computed at the top of the D_{84} particle size for comparisons among model runs based on the Reynolds stress and molecular shear stress at a point:

$$\tau_{ij} = (\mu + \mu_e)\left(\frac{\partial u_i}{\partial x_j} + \frac{\partial u_j}{\partial x_i}\right), \tag{5}$$

where μ is the molecular viscosity, μ_e is the eddy viscosity (computed by the turbulence model), and u_i and u_j are the velocities in the x_i and x_j directions, respectively. The resultant of the applied forces was taken to determine the total shear stress at a point. The average bed shear ($\tau_{bed,avg}$) could then be computed over the bed of the cross-section according to:

$$\tau_{bed,avg} = \frac{\sum(\tau_{ij}d)_w}{W}, \tag{6}$$

where d is the width of a cell in the cross-stream direction over which the shear stress τ_{ij} is computed, the summation is computed over all cells across the cross-section, and W is the bottom width of the channel. These estimates of average bed shear stress are contrasted with cross-section averaged shear stresses (τ_o) defined as $\tau_o = \rho gRS$.

2.3.2. Additional modeling details. We used the RNG k-ε turbulence model with standard equilibrium wall functions [*Yakhot and Orszag*, 1986]. The RNG k-ε turbulence model has been shown to perform better than the standard k-ε model in natural streams involving complex flow geometry [e.g., *Bradbrook et al.*, 1998] and was utilized by *Nicholas and McLelland* [2004] to model flow through natural vegetation using the same vegetation treatment.

FLUENT discretizes the governing conservation of mass and momentum equations using a finite volume approach [*Fluent Inc.*, 2003]. The momentum equations and turbulence equations were discretized using second-order upwind differencing. SIMPLEC pressure-velocity coupling was utilized, which permitted higher under-relaxation factors (0.8–1.0). The water surface was modeled using a fixed lid approach where the free surface was simulated as a symmetry plane with normal velocity components and normal gradients of all variables equal to zero.

The banks of natural channels exhibit substantial variability as channel meandering characteristics, bed topography such as pool-riffle sequences, or other channel variability influences shear stress, turbulence, and velocity characteristics [*Buffington and Montgomery*, 1999; *Wohl*, 2000; *Nicholas and McLelland*, 2004]. In this study, the objective was to specifically isolate the impact of vegetation on flow hydraulics, and only straight prismatic channels were modeled.

Vegetation characteristics (density, species, maturity, and extent of protrusion into the channel) can vary significantly in a given channel reach and contributes to flow complexity. However, *Carney* [2004] demonstrated that by representing the vegetation in the model with a constant width and density (drag characteristics), the essential flow characteristics could be captured, providing a substantial simplification for modeling.

Modeling prismatic channels with constant vegetation characteristics permitted analysis with a periodic inlet/outlet boundary condition to achieve a fully developed flow profile through the flow domain. With the periodic boundary condition, discharge Q through the domain was adjusted until the slope computed based on the downstream pressure gradient $S_{o,P}$, matches the desired channel slope [*Nicholas*, 2001], where

$$S_{o,P} = \frac{1}{\rho g}\frac{dP}{dx} \tag{7}$$

and g, P, and x are gravitational acceleration, pressure, and downstream distance, respectively. The periodic boundary condition also permits modeling a short reach, reducing computational costs.

3. RESULTS

3.1. Analysis of Field Data Results

Analysis of the three field data sets indicates vegetative effects on shear stress magnitudes and scale dependency in dimensionless shear stress values. Mean values of bankfull dimensionless shear stresses in channels <20 m wide are contrasted with values from wider channels in Table 1. For narrower streams with width <20 m, channels with thick

Table 1. Mean Values of Bankfull Dimensionless Shear Stress (τ_*) Stratified by Bank Vegetation and Channel Size[a]

	Channels With Width <20 m			Channels With Width >20 m		
	Thin	Thick	p Value	Thin	Thick	p Value[b]
Andrews [1984]	0.034	0.058	0.0003	0.038	0.030	0.310
Charlton et al. [1978]	0.032	0.073	0.0040	0.038	0.047	0.397
Hey and Thorne [1986][c]	0.045	0.094	0.0002	0.048	0.050	0.849

[a]Differences in τ_* by vegetation type are significant only for channels <20 m wide.

[b]The p value equals the probability that τ_* for thin vegetation is less than τ_* for thick vegetation in channels <20 m and the probability that τ_* for thin vegetation is different than τ_* for thick vegetation in channel widths >20 m.

[c]Thin indicates Hey and Thorne types 1 and 2; thick indicates Hey and Thorne types 3 and 4.

bank vegetation exhibit significantly higher dimensionless shear stresses than those with thin bank vegetation in each of the data sets investigated. Although the bankfull dimensionless shear stresses in channels with thick vegetation are significantly higher in all three studies ($p < 0.003$), there is no significant difference between dimensionless shear stresses for channels with thick or thin bank vegetation at channel widths >20 m (Figure 3).

Relative submergence in channels with widths <20 m was significantly greater ($p = 0.091$) for channels with thick compared to thin vegetation in the *Hey and Thorne* [1986] data. In all three data sets, slopes of channels <20 m wide with thick vegetation were significantly greater than those of channels <20 m wide with thin vegetation ($p \leq 0.053$). All other t tests were nonsignificant. Multiple regression modeling results were very consistent for the *Charlton et al.* [1978] and *Hey and Thorne* [1986] data sets. The toggle variable representing an effect of thick vegetation on channel slope solely for channels <20 m wide was highly significant for the individual data sets ($p < 0.0012$) and both data sets combined ($p < 0.00002$). Models including the toggle variable for small channels with thick vegetation explained 10% to 15% more variance in slope than models with only Q and D_{84}. The slopes of channels <20 m wide with thick vegetation were on

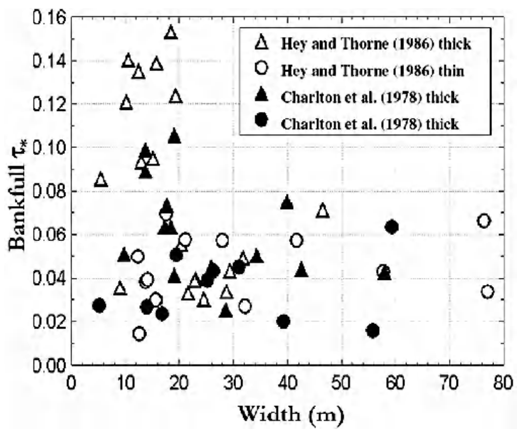

Figure 3. Bankfull dimensionless shear stress referenced to D_{50} for channels with thick and thin bank vegetation. Data are from *Hey and Thorne* [1986] and *Charlton et al.* [1978].

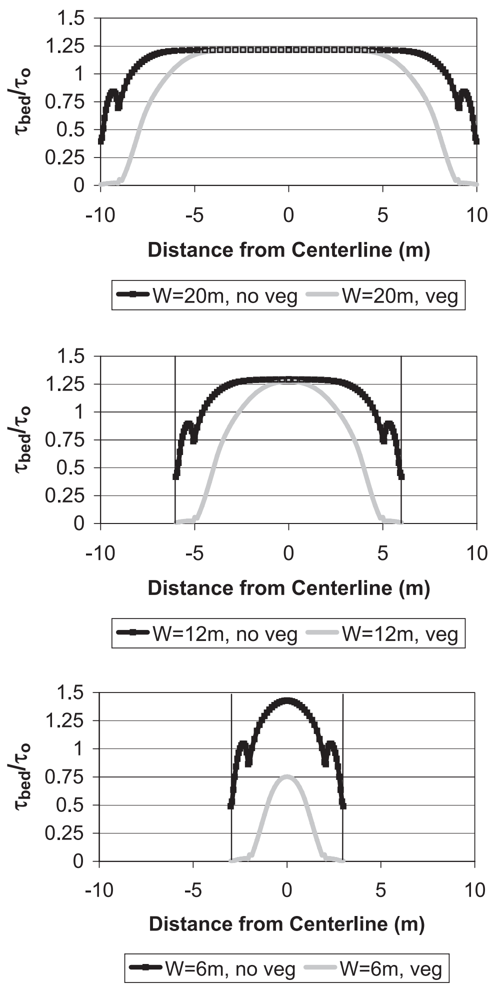

Figure 4. Cross-section boundary shear stress distributions for channels of different widths.

Table 2. Comparison of $\tau_{bed,avg}/\tau_o$ and the Portion of Shear Stress Consumed by the Vegetation for Channels of Different Widths[a]

Top Width (m)	Bed Shear Stress Fraction $\tau_{bed,avg}/\tau_o$	Vegetation Shear Stress Fraction $1-\tau_{bed,avg}/\tau_o$
6	0.37	0.63
12	0.74	0.26
20	0.86	0.14
30	0.91	0.09

[a]In this case, for consistency of comparison between computed shear stresses, τ_o is computed as the average shear stress over the entire boundary in an unvegetated channel according to equations (5) and (6).

average 60% and 105% steeper for a given combination of Q and D_{84} in the *Hey and Thorne* [1986] and *Charlton et al.* [1978] data sets, respectively.

3.2. Modeling Study Results

3.2.1. Scale-dependent influence of vegetation. For the first modeling scenario, Figure 4 depicts the resulting shear stress distributions along the bed of the channel for selected channel widths. For each channel width, τ_o is equal for the vegetated and unvegetated cases because hydraulic radius and channel slope were held constant for each realization. For a given width, the actual shear stress on the bed of the vegetated channel ($\tau_{bed,avg}$) is less than the shear stress on the bed of the unvegetated channel and reflects the additional energy attenuated by the vegetation. Thus, although τ_o is the same for vegetated and unvegetated channels, shear stress distributions on channel beds differ markedly and depend on channel width.

The largest channel shown in Figure 4 is 20 m wide. The drop in shear stress at the bank toe for the unvegetated channels is due to the trapezoidal channel shape. Comparing the shear stress along the channel bed in vegetated and unvegetated channel scenarios, vegetation affects bed shear stress approximately 6 m into the channel from each bank or approximately 60% of the total width. In the channel center beyond the zone of vegetation influence, shear stresses are virtually identical, regardless of vegetation conditions. For channels wider than 20 m, the same pattern is observed. Simulated vegetation affects the shear stress across the entire perimeter of a 12 m channel (Figure 4), with a much smaller zone in the channel center that remains unaffected by the simulated vegetation. Finally, in channels narrower than about 12 m, a considerable drop in shear stress on the bed of the channel is present under the vegetated case. The ratio of average boundary shear stress to cross-section averaged shear stress for the vegetated channels of different widths are presented in Table 2. In this table, the average shear stress over the bed of the channel is computed using equations (5) and (6). To provide a consistent comparison between the shear stress computed using this method and the cross-section averaged shear stress, the shear stress over the entire boundary in an unvegetated channel is computed using equations (5) and (6). If no other factors contribute to the shear stress, this can be assumed equivalent to τ_o. The vegetation fraction of the total shear stress is over four times greater with the decrement of channel width from 20 to 6 m.

3.2.2. Vegetation reestablishment following disturbance or restoration. For the second set of model simulations focusing on bank vegetation succession, channel characteristics and resulting flow depths with average velocities for each scenario are presented in Table 3. In these scenarios,

Table 3. Channel Characteristics for Scenarios Simulating Establishment of Bank Vegetation[a]

	Scenario 1 (No Vegetation)	2	3	4 (Full Vegetation)
Distance from bed vegetation begins (m)	–	0.75	0.5	0.25
C_DA_v (m^{-1})	–	0.4	0.8	1.2
Depth, z above the D_{84} grain height (m)	1	1.01	1.06	1.17
$\bar{u}$ (m s^{-1})	1.28	1.27	1.20	1.07
u_{max} (m s^{-1})	1.80	1.81	1.87	1.95
τ_o(Pa)	24.4	24.5	25.6	27.8
$\tau_{bed,avg}$ (Pa)	25.6	25.6	25.3	23.7
$\tau_{bed,max}$ (Pa)	28.5	28.7	29.9	31.4

[a]Bottom width is 4 m; side slopes equal 1:1; $S_o = 0.003$; $D_{84} = 50$ mm; $Q = 5.75$ m^3 s^{-1}. Depth is measured from the top of the D_{84} grains. Here $\bar{u}$ and u_{max} are the average and maximum downstream velocities for the cross-section, respectively; $\tau_{bed,avg}$ and $\tau_{bed,max}$ are the average and maximum bed shear stress, respectively, computed using equations (5) and (6) at the D_{84} grain height across the channel boundary.

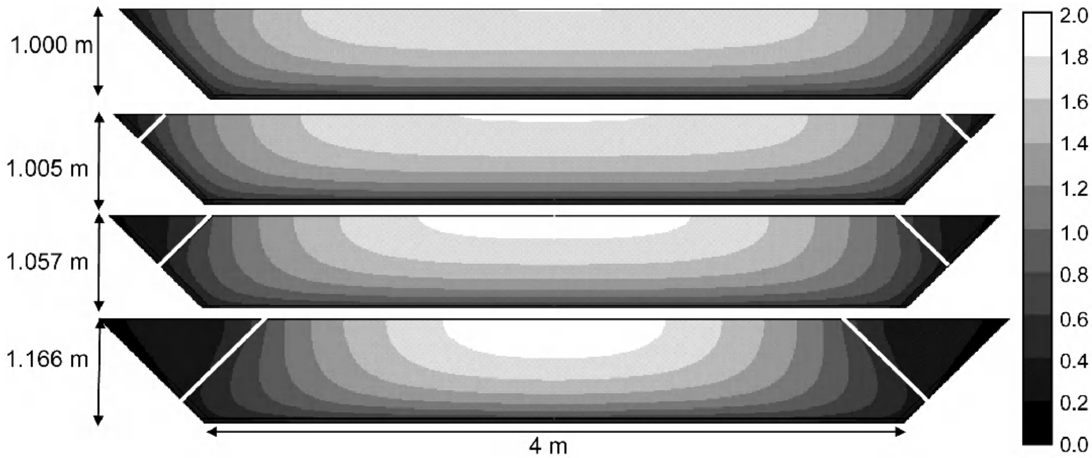

Figure 5. Velocity contours (m s^{-1}) for four increments of simulated vegetation establishment. The white line indicates the edge of the simulated vegetation.

increases in width and depth were necessary to convey a constant discharge. Figures 5 and 6 depict velocity contours and contours of the fluid shear stress computed according to equation (8), respectively, for each of the four scenarios. These simulations indicate that as vegetation establishes on channel banks, near-bank velocities are reduced. Lower velocities in near-bank regions result in an increase in the depth required to convey a constant flow. Accordingly, from the unvegetated scenario (scenario 1) to the scenario with fully established vegetation (scenario 4), there is a 17% increase in the flow depth and a 16% reduction in average velocity. Bank vegetation concentrates flow away from the bank toe and into the channel center. Although there is a large reduction in near-bank velocities due to the vegetation, maximum velocity in the channel center increases 8% from scenarios 1 to 4.

Examination of shear stresses reveals similar patterns (Table 3 and Figure 7). Cross-section averaged shear stress, which was not held constant in these scenarios, increases 14% from scenario 1 to scenario 4 due to increases in flow depth. Average bed shear stress ($\tau_{bed,avg}$) computed according to equations (5) and (6), however, follows an opposite trend and decreases 7% from scenario 1 to scenario 4. The

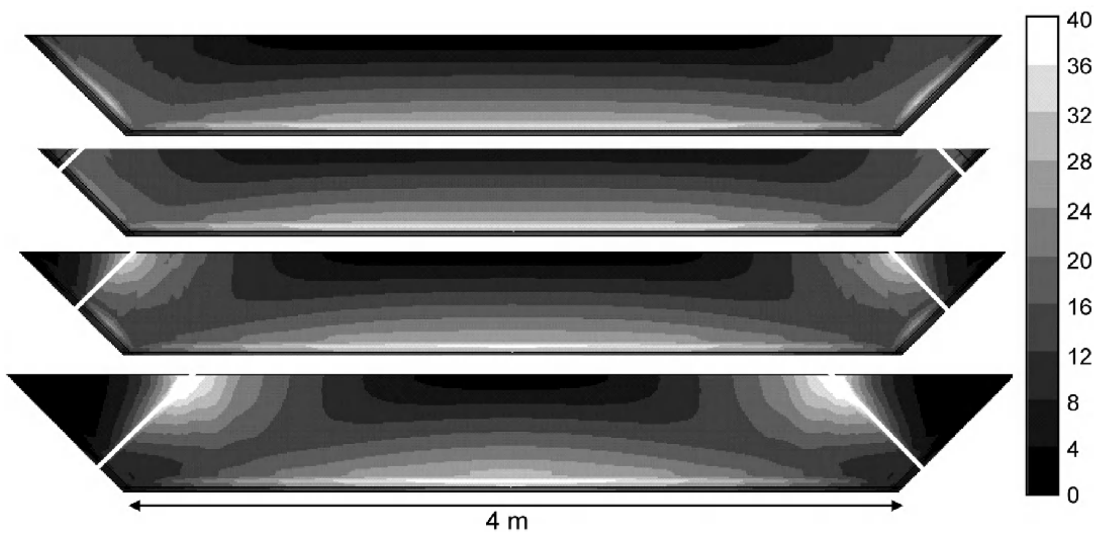

Figure 6. Contours of fluid shear stress (Pa) for four increments of simulated vegetation establishment. The white line indicates the edge of the simulated vegetation. Simulated vegetation density and protrusion increase from top to bottom. A maximum shear stress of 123 Pa occurs in the bottom plot near the water surface at the interface between porous zone and main channel flow.

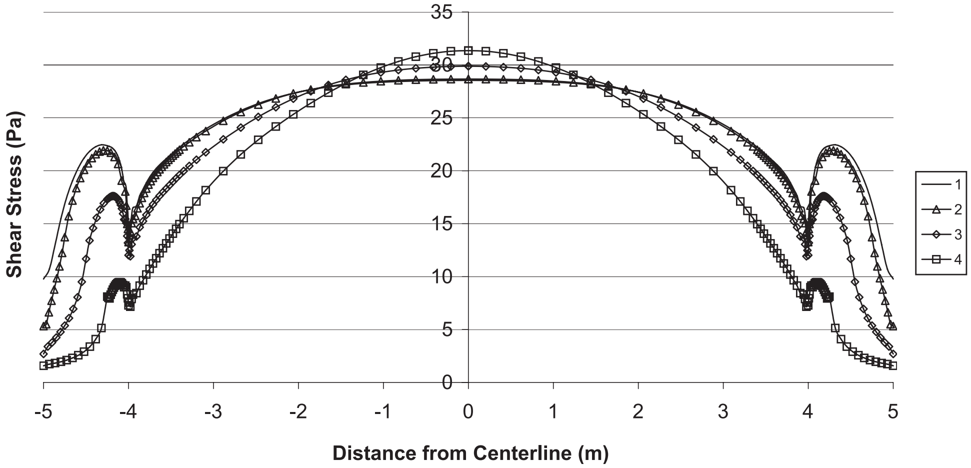

Figure 7. Boundary shear stress profiles for four increments of simulated vegetation establishment.

reduction in boundary shear stress is most significant on the channel banks, as with the velocities. The lowered shear stresses are not limited to flows within the vegetation, but extend beyond the bank toe. This reduction can be attributed in part to the large amount of energy lost in and adjacent to the simulated vegetation zone, which is reflected in the high shear stresses at the vegetation edge near the water surface. In scenario 4, the shear stress at the point of greatest vegetation protrusion is 123 Pa, significantly higher than at any other point in the channel. The magnitude of this high shear stress at the edge of the vegetation may be an artifact of modeling assumptions and limitations to turbulence closure, as this zone typically exhibits highly anisotropic behavior [*Nezu and Onitsuka*, 2001]. However, this zone of shear is consistent with visual observations of vortex shedding in the field, and suggests that this location could be an important source of energy dissipation in vegetated channels. Although the vegetation consumes a large amount of energy and reduces shear stresses in the vicinity of banks, shear stress in the channel center actually increases 9% from scenarios 1 to 4.

A trapezoidal channel was modeled in each of the scenarios presented above. Channels with vertical banks were also modeled in sensitivity tests, yet the resulting velocity profiles varied little from those using a trapezoidal channel with 1:1 slopes when there was thick simulated vegetation on the banks. Similar results were found in simulations involving milder side slopes. Alternatively, with no or minor bank vegetation, there was a more pronounced effect of bank angle on the resulting velocity profiles. With respect to hydraulics, these results suggest that as channels develop thicker bank vegetation, bank morphologic characteristics become less important and are overshadowed by the characteristics of the vegetation on the bank.

4. DISCUSSION

Field data from the United Kingdom and the U.S. Rocky Mountain region indicate a significant scale-dependent effect of bank vegetation that has not been previously accounted for in downstream hydraulic geometry analyses, regime slope models, and shear stress partitioning schemes for gravel bed rivers. The small channels with thick bank vegetation examined in this study have bankfull τ_* and τ values that are roughly twofold those of their thinly vegetated counterparts. The field data and results of the CFD modeling simulations suggest that, as opposed to the traditional focus on relative submergence of vegetation in the vertical dimension, the lateral dimension of channel size relative to the length scale of vegetative roughness is a key missing parameter in understanding shear stress behavior in small streams.

Although there have been numerous physical modeling studies [e.g., *Flintham and Carling*, 1988] examining the effects of channel width-to-depth ratio on shear stress behavior, to our knowledge, no flume study has systematically examined (1) shear stress partitioning using models with banks rougher than the beds and (2) scale-dependent interactions between channel size and vegetation characteristics and their effects on the hydraulic parameters controlling sediment transport and other fluxes in small streams. The CFD simulation results demonstrate the utility and flexibility of using porous media to model various combinations of bed and bank roughness. The CFD modeling approach employed in this study, as well as that of *Kean and Smith* [2004] based on the work of *Houjou et al.* [1990], provide promising computational methods for modeling differentially rough beds and banks that warrant further investigation.

Advances in our knowledge of the scale-dependent influence of bank vegetation have important implications for restoration design of streams based on "tractive force" (dimensionless shear stress criterion), regime, and analytical approaches [e.g., *Copeland and McComas*, 2001]. For example, *Millar* [2005] presented theoretical regime equations for mobile gravel bed rivers with stable banks that include terms representing relative bank strength as affected by vegetation. However, in selecting a method for shear stress partitioning, Millar was forced to rely on relationships developed in flumes with bed roughness equaling or exceeding bank roughness in all cases [*Knight*, 1981; *Knight et al.*, 1984; *Flintham and Carling*, 1988]. The lack of a shear stress partitioning method that accounts for the scale-dependent effects of bank vegetation introduces uncertainty into estimates of the sediment transport capacity and sediment continuity in small, differentially rough channels designed using these methods.

In analytical approaches to stream restoration, designers often generate a "family" of stable channel designs (Figure 8) that all theoretically convey inflowing water and sediment loads without morphologic change. Uncertainty regarding the coevolution of bank vegetation, hydraulics, and channel form currently confounds design in that shear stress distributions, roughness, and continuity of water and sediment are changing with time (Figure 8). A better understanding of shear stress/vegetation interactions would increase the likelihood of placing channels on a self-organizing trajectory that maintains stability as vegetation reestablishes. This could potentially reduce the reliance on hard structures that "lock in" channels at the desired future width and thereby increase the cost effectiveness of stream restoration in many instances.

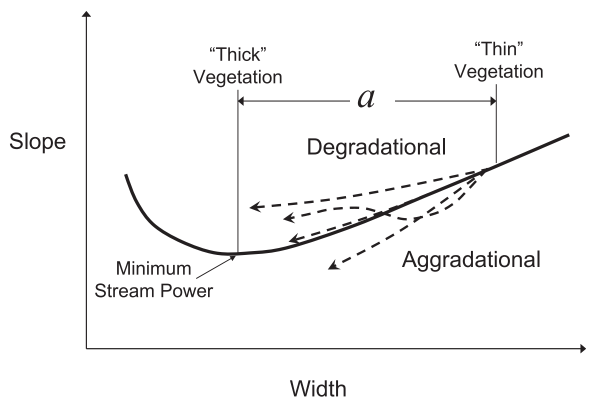

Figure 8. Schematic of theoretical stable channel design solutions for specified inflows of water and sediment with arrows depicting uncertainty in the trajectory and stability of a restoration design as shear stress and conveyance change with vegetation reestablishment along small streams. The range of possible channel widths is also strongly dependent on streamside vegetation as depicted by the range of a, where channel width equals a(bankfull discharge)$^{0.5}$.

Hey [1997] argued that channel slope is not influenced by bank vegetation, and the absence of such an influence can be interpreted as a "problem" for extremal hypotheses such as minimum stream power and maximum sediment transport efficiency. The results of this study indicate that slope is significantly influenced by bank vegetation in channels <20-m wide ($p < 10^{-4}$) and that inferences regarding the realism of extremal hypotheses are spurious in the absence of stream power and shear stress partitioning schemes that account for scale-dependent vegetation effects.

Another aspect of the "problems" associated with extremal hypotheses involves width and slope having different degrees and temporal scales of adjustability [*Hey*, 1997]. The field data used in this study suggest that an increase in slope may be the predominant mode of adjustment in small, thickly vegetated channels, and that differences in depth and grain size account for <20% of the observed twofold increase in bankfull τ_*. If total sediment transport scales roughly with $\tau_*^{1.5}w$, then the fact that widths of the small, thick channels average 60% of their thin counterparts would require τ_* to increase approximately 40% to 45% to maintain an equivalent sediment transport capacity in densely vegetated small channels, all else being equal. The approximately 20% to 25% decrease in bed shear predicted in the CFD simulations for channels of average width in the <20 m wide group (Table 2), when combined with the effect of width reduction would predict a 60% to 70% increase in τ_* to achieve a comparable sediment transport capacity. Thus, the relative influence of bank vegetation on shear stress distributions appears to be potentially greater in natural channels than in the CFD-simulated channels.

Discrepancies in the relative influence of vegetation in natural channels compared with the modeled channels could be explained by a number of factors related to the modeling methods. The channels that were modeled were straight, prismatic channels with no variation in the vegetation characteristics. The existence of topographic complexity, both in the channel characteristics as well as the vegetation characteristics, would seem to influence the results, although the nature of these influences is not clear at present. In the CFD scheme used in this study, modeling the vegetation with varying protrusion characteristics in the downstream direction has minor effects [*Carney*, 2004]. The model was shown to reasonably represent the velocity profiles over and through simulated vegetation in a number of different scenarios; however, in each of the flume experiments used to verify the CFD models and in the simulations described here, the simulated vegetation had regular, homogeneous patterns. Natural vegetation exhibits highly heterogeneous characteristics, causing expansions, contractions, eddies, and other losses as the flow moves around and through the vegetation. This suggests that future studies should consider the importance of heterogeneous vegetation. While the CFD vegetation representations account for the influence of the vegetation on the momentum equations, they do not account for flow blockage effects caused by vegetation. Particularly in the vicinity of woody vegetation, flow blockage effects would seem to have a dominant effect on the resulting flow fields. Future research could focus on determining appropriate means of accounting for these effects in CFD modeling.

The scale-dependent influence of vegetation on channel hydraulics observed in the CFD simulations appears to support the field observations of *Coon* [1998], who identified an upper limit for the influence of bank vegetation around 20 to 30 m. Similarly, the modeling conducted by *Masterman and Thorne* [1992] suggests at width-to-depth ratios greater than about 15, the effects of riparian vegetation are negligible. In our CFD simulations, the shear stress attributed to the vegetation was 14% and 9% of the total shear stress for 20 and 30 m wide channels, respectively. The *Coon* [1998] channels contained rougher bed material than those used for the case study, so the vegetation probably had less relative influence in these channels than what is suggested by our simulations.

The CFD simulations also point to physical processes that underlie observed differences in channel form in the field studies from the United Kingdom and the western United States, which found that channels with thick, woody bank vegetation were generally narrower and deeper than channels with banks comprised of grasses and thin vegetation. In the simulation of vegetation succession, bank vegetation concentrates flows in the channel center, causing an increase in flow depths for equivalent discharges. Bank vegetation reduces shear stresses in the near-bank zone and increases shear stresses in the channel center. These effects could be more pronounced if erosional and depositional processes were considered. Given time and sufficient supply, aggradation could occur in the low shear stress region along the bank, causing the channel to narrow. In addition, higher shear stresses in the channel center could erode the bed, resulting in deeper channels. These effects would be combined with the increases in bank strength due to the root structure of the vegetation [*Simon and Collison*, 2002], which would provide additional resistance to erosion in narrower channels, especially in smaller channels where rooting depth is on the order of bank height. Thus, vegetation can affect hydraulics in a manner that creates a tendency toward cross-sections that are narrower and deeper than unvegetated channels (but see *Montgomery* [1997] and *Anderson et al.* [2004] for disparate effects due to wood debris and other factors).

Sediment transport capacity in complex channels like those examined in this study is frequently computed using cross-section average shear stress, τ_o [*Julien*, 1995]. In the simulated vegetation succession, however, it was shown that although τ_o increased with successively thicker simulated vegetation due to flow depth increases, the average bed shear stress decreased. Thus, using τ_o to compute sediment transport in these channels may yield inaccurate results without partitioning the shear stress between the bed and vegetation; however, the specific response is complex. The differing shear stress distributions would have nonlinear effects on sediment transport rates depending on the bed characteristics, and the nonuniform distribution of shear stresses could be further magnified in nonprismatic channels [*Lisle et al.*, 2000; *Ferguson*, 2003]. Variations on the CFD modeling approach presented here could be used to provide additional insights into morphologic influences on sediment transport phenomena and assist in the further development and verification of shear stress partitioning schemes.

In summary, these analyses underscore a fundamental gap in our understanding of fluvial processes and hydraulics in wadeable streams with variable bank vegetation. Most previous CFD studies have modeled channels with relatively smooth boundaries for which roughness can be represented using a roughness height and have ignored the impact of bank vegetation. This study demonstrated the need to account for both forms of roughness in CFD modeling of streams less than approximately 30 m wide. Previous research has demonstrated the important influence of vegetation on planform characteristics and meander bend hydraulics [*Thorne and Furbish*, 1995; *Millar*, 2000; *Gran and Paola*, 2001]. CFD modeling focusing on more complex geometries could provide additional insights on the important effects of bank vegetation on planform characteristics of streams. Improved and transferable quantitative tools for predicting shear stress behavior in small streams of different scales and bank conditions could improve the physical basis and effectiveness of stream restoration and simultaneously advance understanding of the inverse problem of stream vulnerability to vegetation disturbance and removal.

5. CONCLUSIONS

The effect of bank vegetation roughness on the hydraulics of natural channels is scale dependent with process shifts evident at widths less than approximately 20 m. This finding has important implications for improvement of shear stress partitioning models used in stable channel design. Because specific mechanisms controlling the apparent scale dependency are difficult to isolate in natural channels, the method of representing vegetation in CFD applications combined with a porous treatment of rough beds based on the work of *Wiberg and Smith* [1991] and *Carney et al.* [2006] presented in this study can be used to simulate the mechanisms controlling channel form and reveal patterns and scale dependency in hydraulic parameters consistent with field data from channels of different scales and bank vegetation types. Although additional research could improve the parameterization of vegetation in the CFD models, the use of these methods provides a means to better understand the influence of vegetation, on material fluxes, channel evolution, and geomorphic processes in wadeable streams and thereby improve the scientific basis of restoration designs.

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Hyporheic Restoration in Streams and Rivers

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The hyporheic zone is the area of mixing of surface and groundwater beneath and adjacent to streams and rivers. The unique physical, chemical, and biological properties of the hyporheic zone, often different from both surface water and deeper groundwater, create unique habitat for organisms. Exchange of water between surface water and the hyporheic zone additionally creates hyporheic functions such as nutrient processing, toxin mineralization, and thermal buffering, which benefit surface water ecosystems and humans downstream. Human activities have reduced hyporheic exchange through impacts such as channel simplification and introduction of fine sediment that clogs the bed. Efforts to improve ecological conditions in impaired streams and rivers have increased dramatically in recent decades. Nevertheless, the value of restoring hyporheic exchange, where it has been lost due to human actions (hyporheic restoration) as a component of stream and river restoration, is only beginning to be acknowledged. Further, guidance for accomplishing hyporheic restoration is scarce. Nevertheless, due to considerable recent interest in the hyporheic zone and its functions, data that could inform hyporheic restoration efforts are already fairly common. Here we lay out possible goals for hyporheic restoration and summarize engineering data that already exist in the scientific literature. We also lay out the hyporheic restoration process and set that within the largest context of stream and river restoration and watershed planning. Finally, we present our future vision for future research, creating design and management guidance, and government leadership.

1. INTRODUCTION

The hyporheic zone is the area of mixing of surface and groundwater beneath and adjacent to streams and rivers [*Triska et al.*, 1989], particularly that region where hydrologic flow paths leave and return to the surface stream many times along its length (Figure 1) [*Harvey and Wagner*, 2000]. The hyporheic zone is biogeochemically unique relative to surface water and deeper groundwater, containing entrained detrital carbon and exhibiting intermediate levels of temperature, oxygen, and other solutes. Such conditions, existing in combination with the high surface area and large residence time of sediment grains, create a unique and important area for biogeochemical reactions [*Brunke and Gonser*, 1997] (Figure 2). Exchange of stream water through the hyporheic zone (hyporheic exchange) facilitates important exchanges of heat, chemical solutes, and biota between surface stream

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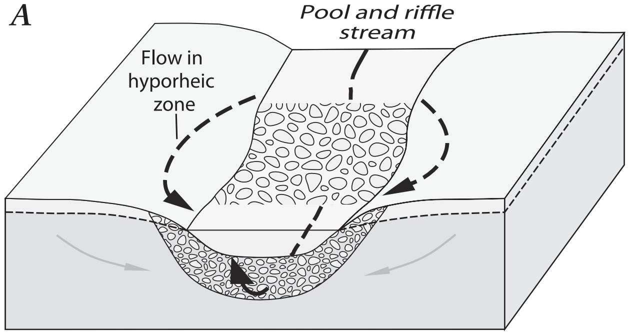

Figure 1. Hyporheic zone showing vertical and horizontal flow paths due to drops in (a) channel bed and (b) meander bends. From *Winter et al.* [1998, Figure 14].

and subsurface water [*Jones and Mulholland*, 2000]. Exchange processes then affect the distribution and abundance of organisms in streams and the hyporheic zone, ecosystem level processes such as nutrient cycling and carbon flux, and water quality [*Boulton et al.*, 1998; *Jones and Mulholland*, 2000; *Groffman et al.*, 2005]. The hyporheic zone thus functions as both important subsurface habitat and an important influence on surface stream ecology and chemistry.

Human activities, both directly in stream and river channels and within contributing watersheds, have impaired ecological conditions in streams and rivers, including hyporheic exchange and function [*Hancock*, 2002; *Palmer and Allan*, 2006]. Efforts to improve ecological conditions in impaired streams and rivers in the United States stretch back to the early 20th century [*Thompson*, 2006] and have increased dramatically in recent decades [*Bernhardt et al.*, 2005]. Nevertheless, the value of restoring lost hyporheic exchange and associated benefits (hyporheic restoration) as a component of stream and river restoration [*Kasahara et al.*, 2009; *Boulton et al.*, 2010; *Hester and Gooseff*, 2010], and the impact of stream and river restoration on hyporheic exchange are only beginning to be acknowledged [*Kasahara and Hill*, 2007a; *Crispell and Endreny*, 2009; *Knust and Warwick*, 2009] in the research community. To our knowledge, restoring hyporheic exchange or function is currently even less common as a goal in the practice world, with some notable exceptions, for example, in the Willamette River basin [*Grant et al.*, 2006]. Further, while conceptual guidance for accomplishing hyporheic restoration is becoming more common [*Boulton*, 2007; *Smith et al.*, 2008; *Hester and Gooseff*,

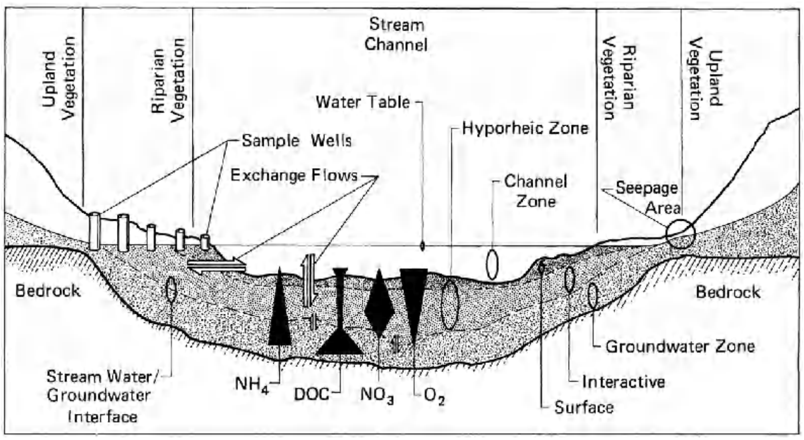

Figure 2. Conceptual model of groundwater-surface water linkage at Little Lost Man Creek. Waters are divided into three zones: a channel zone containing surface water, a hyporheic zone, and a groundwater zone. The hyporheic zone is divided into a surface hyporheos with virtually identical chemistry to channel waters that contains >98% advected channel water and into an interactive hyporheos that contains 10–98% channel water and is characterized by physical-chemical gradients (chemical and temperature). The general solute concentration profiles for several biogeochemical parameters are presented. From *Triska et al.* [1989, Figure 6], reprinted with permission.

2010], detailed guidance is still lacking. Nevertheless, due to considerable recent interest in the hyporheic zone and its functions, data that could inform hyporheic restoration efforts are already fairly common. The goal of this chapter is to review such data and demonstrate its potential use in increasing hyporheic exchange via restoration projects. We address the following questions: (1) Should hyporheic restoration be included as a component of a particular stream project? (2) What are the possible goals of hyporheic restoration? (3) How can the restoration potential of a given stream reach be assessed? (4) What project elements can be included in restoration design or as part of channel management to increase hyporheic exchange? (5) What data are available to determine specifications for such features? (6) What is the design process for hyporheic restoration?

We also present our vision for future hyporheic restoration research, guidance, and policy. While our focus is hyporheic restoration, such practice fits within a broader stream and river restoration context [*Boon*, 1998]. Many techniques for hyporheic restoration have additional nonhyporheic benefits, and while we do not discuss those other benefits here, we acknowledge that, like other restoration techniques (e.g., bank stabilization), hyporheic restoration should generally not occur in isolation. Furthermore, in keeping with the scope of this volume, we focus here on current typical practice of stream restoration, which entails direct intervention in stream or river channels and floodplains, typically based on engineering design drawings or detailed channel management plans. Nevertheless, we emphasize that the ultimate solution to aquatic impairment is reconfiguring human activities and consequently land use planning on a watershed scale [*Palmer et al.*, 2010]. This broader context for stream restoration is discussed at relevant points in the chapter.

2. RESTORATION GOALS (EXCHANGE AND ITS BENEFITS)

Establishing clear restoration goals is critical for the success of any stream or river restoration project [*Wohl et al.*, 2005]. This is applicable to hyporheic restoration as well. Hyporheic restoration goals may be stated as simply restoring hyporheic exchange (bidirectional flow of water across the streambed) or may be more specific, such as specifying restoration of hyporheic habitat for particular species or restoring particular hyporheic functions. We define hyporheic functions as functions performed by the hyporheic zone that benefit some ecological or human health aspect of surface water [*Dent et al.*, 2000]. In this section, we present an overview of the various possible goals that might motivate hyporheic restoration. These goals are discussed in the context of the benefits of hyporheic exchange and function that can be lost due to human activities. We accordingly also outline the human activities that tend to cause impairment in each of the given categories, creating opportunities for restoration. Discussion of how channel and watershed characteristics control hyporheic exchange, function, and benefits, including what actions might mitigate lost exchange, can be found in section 3.

2.1. Hydraulics

The physical flow of water through the hyporheic zone (i.e., hyporheic exchange hydraulics) is fundamental to any chemical or biological or processes that may occur therein. Restoration of hyporheic hydraulics is therefore required for all of the other hyporheic restoration goals, such as hyporheic habitat restoration or hyporheic function enhancement. Enhancing hyporheic exchange of water accordingly will be a goal of all hyporheic restoration efforts. Hyporheic flow is driven by a variety of mechanisms, including turbulent exchange, movement of sediment, and heterogeneity of sediment texture [*Elliott and Brooks*, 1997; *Salehin et al.*, 2004; *Cardenas and Wilson*, 2007a]. However, the primary mechanism in most cases is local variation in head gradients between the stream and groundwater induced by in-channel geomorphic complexity/features (e.g., steps, debris dams, pool-riffle sequences, and large wood) that drive Darcy flux of water in the subsurface [*Harvey and Bencala*, 1993; *Hester and Doyle*, 2008]. Darcy flux also increases with channel planform complexity, defined here as sinuosity or branching of the channel (i.e., features such as meander bends, side channels, and islands), which increases the length or area of channel engaged in hyporheic exchange [*Boano et al.*, 2006; *Poole et al.*, 2008; *Cardenas*, 2009]. A critical factor in all such exchange mechanisms is the hydraulic conductivity (K) of hyporheic sediments, which directly controls Darcy flux. Substantial low permeability deposits will significantly modify flow path tortuosity [*Salehin et al., 2004*; *Sawyer and Cardenas*, 2009] and/or substantially reduce exchange flux.

Human activities decrease hyporheic exchange mainly through two basic mechanisms. First, humans reduce both in-channel and planform complexity both directly via channel straightening, dredging, diking, and disconnection of floodplains, and indirectly via land use change such as urbanization, agriculture, and forestry [*Poole and Berman*, 2001; *Hancock*, 2002; *Allan and Castillo*, 2007]. Second, humans increase the loading of fine sediments to waterways through activities such as urbanization, agriculture, and mining, which clogs surface sediment pores [*Wood and Armitage*, 1997], reduces K, and cuts off hyporheic exchange [*Hancock*, 2002]. Reversing any of these anthropogenic impacts represents opportunity for restoration of hyporheic hydraulics. Such human degradation of hyporheic hydraulics is also the root cause of most degradation of habitat and hyporheic function, as described below.

2.2. Temperature

Hyporheic zones can benefit a stream or river by impacting water column temperatures above. Deep groundwater maintains a constant temperature approximately equal to the annual average air temperature, while stream and river temperatures warm and cool over the annual cycle [*Caissie*, 2006]. As a result, hyporheic exchange, which circulates surface water through the sediments and thereby enhances the exchange of heat between the surface stream and groundwater, moderates surface water temperatures by providing cooling in summer and warming in winter. This effect tends to be strongest in summer, when base flow rates are generally less than in winter, and is superimposed by diel cycles of lesser magnitude. Such thermal effects can benefit organisms by reducing temperature extremes [*Poole and Berman*, 2001]. Heat exchange across the streambed can in turn increase thermal heterogeneity within the sediments [*Hester et al.*, 2009]. Temperature impacts can then affect water quality by impacting oxygen solubility [*Allan and Castillo*, 2007] and biochemical reaction rates [*Butturini and Sabater*, 1998; *Hedin et al.*, 1998; *Nimick et al.*, 2003]. Both degradation and restoration of thermal moderation and heterogeneity is directly tied to the fate of the hyporheic hydraulics themselves.

2.3. Nutrients

Hyporheic zones can benefit a stream or river through increased nutrient processing [*Craig et al.*, 2008; *Smith et al.*, 2008]. The unique biogeochemical conditions of the hyporheic zone (high interstitial surface area, carbon sources, intermediate flow velocities, etc.) allow abiotic and biotic chemical transformations to occur often to a much greater degree than in surface water or deeper groundwater. In nitrogen-limited areas (i.e., areas minimally impacted by humans), ammonification (conversion of organic nitrogen to ammonia) and nitrification (generation of nitrate from ammonium) can occur in hyporheic zones, which can be important for autotrophic communities [*Coleman and Dahm*, 1990; *Jones et al.*, 1995]. On the other hand, human activities, such as agriculture, industry, and combustion of fossils fuels, increasingly create a nitrogen excess harmful to humans and biota [*Galloway et al.*, 2008]. In such areas, denitrification, which reduces nitrate (via nitrite and nitrous oxide) to dinitrogen gas, occurring in anaerobic hyporheic zones can be an important sink for anthropogenic nitrogen [*Hill et al.*, 1998; *LeFebvre et al.*, 2004; *Kasahara and Hill*, 2006a, 2007b]. While less is known about phosphorus dynamics in the hyporheic zone, both abiotic sorption to oxides and heterotrophic bacterial metabolism are possible sinks for phosphorus in hyporheic zones [*Mulholland et al.*, 1997; *Hendricks and White*, 2000]. Such effects on phosphorus are of particular note because phosphorus is often the limiting nutrient in streams and rivers [*Hendricks and White*, 2000]. The capacity for biogeochemical reactions in the hyporheic zone can be reduced by humans via any actions that reduce hyporheic hydraulics, but also by removing critical reaction substrates. For example, labile carbon sources are required to develop the necessary redox conditions for denitrification [*Duff and Triska*, 2000]. Activities that reduce input of particulate organic carbon (POC) to streams (e.g., riparian deforestation), reduce retention of POC in streams (e.g., channel simplification) or reduce entrainment of POC into hyporheic sediments (e.g., altering stream or river flow regime) therefore have the potential to reduce denitrification in the hyporheic zone.

2.4. Toxins

The unique reactivity of the hyporheic zone can similarly benefit stream or river organisms and also humans by reducing the concentration of dissolved toxins. For example, aerobic conditions are present in some hyporheic zones, which allow mineralization of toluene, pentachlorophenol, and fuel oxygenates [*Pignatello et al.*, 1983; *Kim et al.*, 1995; *Bradley et al.*, 1999; *Landmeyer et al.*, 2001]. On the other hand, where carbon sources allow sufficient hyporheic metabolism, anaerobic conditions that allow denitrification to occur also allow reductive dehalogenation of chlorinated solvents [*Conant et al.*, 2004] as well as mineralization of fuel oxygenates under certain conditions [*Bradley et al.*, 2001a, 2002]. In addition to degradation reactions, sorption processes can also attenuate pollutant migration. For example, organic carbon present in hyporheic zones can retard migration of organic pollutants in streams [*Smith and Lerner*, 2008]. Similarly, redox gradients and a variety of sorption sites can induce precipitation or sorption of a wide variety of metals under a wide range of conditions [*Bencala et al.*, 1984; *Cerling et al.*, 1990; *Kimball et al.*, 1994; *Benner et al.*, 1995; *Moser et al.*, 2003; *Brown et al.*, 2007; *Gandy et al.*, 2007]. Hyporheic processes that impact water quality (either nutrients or toxins) can affect water moving through the hyporheic zone from upstream surface water or from deeper groundwater headed for the stream and can have a significant cumulative impact at the basin scale [*Harvey and Fuller*, 1998].

2.5. Habitat

The hyporheic zone, including the shallowest (i.e., benthic) layer of sediments, is habitat for a wide variety of organisms [*Gibert et al.*, 1994; *Brunke and Gonser*, 1997]. Some invertebrate organisms are permanent hyporheos, spending their entire life cycles in the subsurface [*Williams and Hynes*, 1974]. Other organisms are occasional hyporheos, including water column and even terrestrial species that utilize the hyporheic or benthic zones for particular life stages [*Brunke and Gonser*, 1997; *Wood and Armitage*, 1997] or during adverse conditions (i.e., as a refuge) [*Dole-Olivier et al.*, 1997]. Such species, and by extension hyporheic conditions, can therefore have considerable importance for water column and terrestrial ecosystems. For example, many species of salmonids are important components of river, estuary, and even ocean ecosystems [*Quinn*, 2005] that can, in turn, impact adjacent terrestrial ecosystems via importation of marine nitrogen [*Gende et al.*, 2007; *Quinn et al.*, 2009]. Yet many salmonids incubate their eggs in stream sediments, where viability depends greatly on interstitial conditions (e.g., oxygen, nutrients, and temperature), which are influenced by surface water-groundwater interactions [*Webster and Eiriksdottir*, 1976; *Baxter and Hauer*, 2000; *Yamada and Nakamura*, 2009]. Additionally, certain winged terrestrial insects in the orders Ephemeroptera, Plecoptera, and Trichoptera spend their larval stages on or in the benthic layer where their species composition has been shown to be influenced by hyporheic exchange [*Pepin and Hauer*, 2002]. Finally, some organisms do not reside in the hyporheic zone but take direct advantage of its benefits, including both aquatic and terrestrial plants [*Coleman and Dahm*, 1990; *Mouw et al.*, 2009].

Humans degrade hyporheic habitat primarily by reducing hyporheic exchange. In particular, clogging of sediment pores by fine sediments can decrease hyporheic flows, dissolved oxygen supply, removal of metabolic wastes, and substrate suitability for hyporheic organisms [*Wood and Armitage*, 1997], although clogging with fine sediment is not always the issue [*Sowden and Power*, 1985; *Peterson and Quinn*, 1996]. Impairment of adjacent surface water or groundwater quality (e.g., toxins, excess nutrients, and elevated temperatures) can also impact the suitability of hyporheic habitat. Reversing any of these human impacts represent opportunities for restoration of hyporheic habitat [*Boulton*, 2007].

3. STREAM RESTORATION TECHNIQUES THAT ENHANCE HYPORHEIC EXCHANGE OR FUNCTION

To be effective, hyporheic restoration goals must be translated into design plans or channel management plans. Such plans generally entail adding various geomorphic features [*Boulton*, 2007] included solely for hyporheic enhancement or included for other reasons but modified to maximize hyporheic impact. Here we refer to these geomorphic features as hyporheic features. Larger-scale actions beyond the channel (e.g., techniques to reduce influx of fine sediments such as riparian corridor restoration or sediment management at construction sites) can also contribute to achieving hyporheic restoration goals, and these are discussed in section 4.3.

Hyporheic features aim to enhance factors that contribute to hyporheic exchange or hyporheic function, but are also subject to engineering limitations. Such features typically enhance exchange by creating localized drops in water level (head drops), increasing the length or area of stream engaged in hyporheic exchange or increasing hydraulic conductivity (K) of sediments (Table 1). Other factors that can affect hyporheic exchange such as stream discharge, substrate heterogeneity, turbulence of channel flow, and bed load movement are less amenable to engineering control. Head drops are caused by in-channel features, such as local steepening of the streambed (e.g., steps and riffles) or obstacles that create backwater (e.g., debris dams and large wood) [*Gooseff et al.*, 2006; *Lautz and Siegel*, 2006; *Hester and Doyle*, 2008]. Features that increase planform complexity (e.g., meanders, islands and side channels) can increase the length of stream engaged in hyporheic exchange [*Boano et al.*, 2006; *Cardenas*, 2008; *Poole et al.*, 2008]. Increased K

Table 1. Stream Features/Techniques That Enhance Hyporheic Exchange

Hydraulic Function	Feature/Technique	Subtype	Examples
Head drop	in-channel features	streambed steepening	steps riffles
		backwater	debris dams log dams boulder weirs large woody debris
Area/length of exchange	channel planform complexity	sinuosity branching	meanders islands alcoves side channels braids bars
Hydraulic conductivity	substrate coarsening		substrate augmentation substrate cleaning

Table 2. Types and Sources of Data Useful for Hyporheic Engineering

Hyporheic Function	Feature/Technique	Engineering Parameter	Example References
Hyporheic hydraulics[a]	in-channel features	feature size	*Kasahara and Wondzell* [2003], *Storey et al.* [2003], *Gooseff et al.* [2006], *Lautz and Siegel* [2006], *Tonina and Buffington* [2007], *Hester and Doyle* [2008]
		feature type	*Kasahara and Wondzell* [2003], *Kasahara and Hill* [2006b], *Lautz and Siegel* [2006], *Hester and Doyle* [2008], *Crispell and Endreny* [2009], *Wondzell et al.* [2009]
		conceptual	*Harvey and Bencala* [1993], *Mutz et al.* [2007], *Knust and Warwick* [2009]
	channel planform complexity	feature size	*Boano et al.* [2006], *Cardenas* [2008, 2009]
		feature type	*Kasahara and Hill* [2007a]
	sediment texture	sediment texture	*Packman and MacKay* [2003], *Saenger et al.* [2005], *Velickovic* [2005], *Kasahara and Hill* [2007b]
Temperature moderation	in-channel features	feature size	*Hester et al.* [2009]
		example	*White et al.* [1987], *Hendricks and White* [1991], *Evans and Petts* [1997], *Moore et al.* [2005], *Loheide and Gorelick* [2006]
	channel planform complexity	example	*Fernald et al.* [2006], *Arrigoni et al.* [2008], *Burkholder et al.* [2008]
	general temperature	conceptual	*Story et al.* [2003], *Johnson* [2004]
Nutrient processing	in-channel features/nitrogen	example	*Hill et al.* [1998], *LeFebvre et al.* [2004], *Groffman et al.* [2005], *Kasahara and Hill* [2006a]
	in-channel features/phosphorus	example	*Hendricks and White* [1991, 2000]
	channel planform complexity/nitrogen	feature type	*Kasahara and Hill* [2007b]
	general nutrient processing	conceptual	*Coleman and Dahm* [1990], *Jones et al.* [1995], *Mulholland et al.* [1997], *Fischer et al.* [2005], *Craig et al.* [2008], *Smith et al.* [2008]
Toxin attenuation	sediment texture	sediment texture	*Bradley et al.* [2001b]
	general organic pollutant mineralization	conceptual	*Pignatello et al.* [1983], *Kim et al.* [1995], *Bradley et al.* [1999, 2001a], *Conant et al.* [2004], *Chapman et al.* [2007], *Landmeyer et al.* [2010]
	general precipitation/sorption of metals	conceptual	*Bencala and Walters* [1983], *Cerling et al.* [1990], *Kimball et al.* [1994], *Benner et al.* [1995], *Harvey and Fuller* [1998], *Fuller and Harvey* [2000], *Moser et al.* [2003], *Brown et al.* [2007], *Gandy et al.* [2007]
Hyporheic habitat[b]	in-channel features	example	*Geist and Dauble* [1998], *Sliva and Williams* [2005], *Davy-Bowker et al.* [2006]
	channel planform complexity	example	*Dole-Olivier and Marmonier* [1992], *Geist and Dauble* [1998]
	sediment texture	sediment texture	*Olsen and Townsend* [2003], *Boulton* [2007]
	general habitat	conceptual	*Coleman and Hynes* [1970], *Stanford and Ward* [1988], *Boulton et al.* [1992], *Dole-Olivier et al.* [1997], *Baxter and Hauer* [2000], *Pepin and Hauer* [2002], *Olsen and Townsend* [2003], *Boulton* [2007], *Mouw et al.* [2009]

[a]Hyporheic exchange rate, residence time, or hyporheic zone size.

[b]Including hyporheic organisms and surface water organisms that use hyporheic zone temporarily.

can be accomplished by adding coarser sediment to [*Grant et al.*, 2006] or removing fine sediment from (cleaning) existing substrate [*Meyer et al.*, 2008].

Engineering data are necessary to translate hyporheic goals into design or management plans. A large number of studies have been published that quantify various aspects of hyporheic exchange, function, or habitat (Table 2). We discuss a representative cross section of those studies here, in particular, demonstrating two types of papers that have particular value for hyporheic engineering. First, we discuss studies that provide insight into the likely feasibility or efficacy of restoration activities on hyporheic exchange or function. Next, we discuss studies that relate exchange and its benefits to stream morphology. Last, we finish this section with a discussion of the importance of hydrologic context and its variation in time and space.

3.1. Efficacy and Feasibility

The ability of hyporheic restoration to enhance hyporheic exchange will be site specific and depend on both local (e.g., existing channel or floodplain sediment texture) and upstream conditions (e.g., fine sediment load arriving from upstream). Predictions of efficacy are thus not possible a priori, and instead, we discuss potential ranges of efficacy that are supported by the literature and refer the reader to the original sources for more information. Because of its overriding importance for all hyporheic functions, we discuss hyporheic hydraulics in detail first, followed by briefer coverage of other hyporheic benefits/functions.

The importance of hyporheic hydraulics to stream hydrology can be quantified as the percent of surface flow in the channel that moves through the hyporheic zone in a given reach of stream. Estimates of this percentage, and therefore the potential efficacy of hyporheic restoration, vary over many orders of magnitude and seem to be controlled primarily by K, channel morphology, and length of channel considered. At the river segment scale, 30% of the flow of a sizeable cobble/boulder bed river can move through a floodplain along a ~6 km segment when K is in the range of 10^{-2} to 10^{-1} m s^{-1} [*Poole et al.*, 2006]. On the other hand, when both K (10^{-5} to 10^{-3} m s^{-1} typical of sand/silt mixtures) and length of channel considered (~100 m) are less, percent exchange can range from a few tenths to a few percent of surface flow for riffles, step pools, sinuosity, or in-channel large wood [*Saenger et al.*, 2005; *Wondzell et al.*, 2009]. Nevertheless, *Kasahara and Wondzell* [2003] found that within reaches of similar length (~100 m) and K (10^{-5} to 10^{-3} m s^{-1}), 75–100% of channel water was exchanged in headwater areas and 1–10% of channel water was exchanged in fifth-order reaches.

At the subreach scale, exchange flux rates can be as high as 50% of surface flow for a single weir or step type structure if K is high enough (e.g., 10^{-2} m s^{-1}, typical of fine gravel or coarse sand) [*Hester and Doyle*, 2008]. Hyporheic flux can similarly be greater than 10% of surface flow for a single cross vane or rock vane structure (1.4 L exchange s^{-1} m^{-1} of streambed in 50 L s^{-1} surface discharge for structures at least 5 m long) [*Crispell and Endreny*, 2009]. On the other hand, hyporheic flow can be well less than 0.1% of surface flow for individual in-channel features or even large meander bends when K is in the range of 10^{-6} to 10^{-4} m s^{-1} (typical of silts with even some clay) or even less [*Boano et al.*, 2006; *Kasahara and Hill*, 2006b; *Lautz and Siegel*, 2006]. The most important factor controlling exchange per length of stream appears to be K, followed by geomorphic feature type [*Hester and Doyle*, 2008]. K dominates control of hyporheic exchange largely due to its extreme variability, which ranges from 10^{-10} to 10^{-1} m s^{-1} in fluvial sediments [*Calver*, 2001]. Even within the most frequently reported range of K values (10^{-7} to 10^{-3} m s^{-1}), the percent of water moving through the hyporheic in a given reach can vary from highly significant to insignificant, underscoring how site specific the potential for hyporheic restoration necessarily is.

The significance of the percent of surface flow cycling through the hyporheic zone for beneficial hyporheic functions depends fundamentally on the nature of the function. For example, while thermal impacts of hyporheic exchange can be locally significant even to bulk main stem flow (e.g., >1°C given by *Story et al.* [2003] and *Loheide and Gorelick* [2006]), surface stream temperatures re-equilibrate with the atmosphere given sufficient downstream movement [*Mohseni and Stefan*, 1999] (Figure 3). Therefore, while locally insignificant thermal impacts of structure-induced hyporheic exchange on main stem flow (e.g., ≤~0.01°C given by *Burkholder et al.* [2008] and *Hester et al.* [2009]) may sum over distance if many structures are present, re-equilibration with the atmosphere means that such impacts can sum only over finite distances. Nevertheless, even where hyporheic impacts on bulk main stem flow temperatures are minimal, impacts on thermal heterogeneity in both surface water and hyporheic water can be much larger and therefore significant to organisms present (e.g., >1°C given by *Arrigoni et al.* [2008], *Burkholder et al.* [2008], and *Hester et al.* [2009]) (Figure 4).

The impact of hyporheic biogeochemical reactions on surface stream conditions is controlled by both the rate of biogeochemical processes in the hyporheic zone and the percent of surface flow cycling through [*Findlay*, 1995] (Figure 5). The cumulative impact of these reactions over long distances, in turn, depends on whether the reactions are reversible. For example, mineralization of xenobiotic organic toxins like toluene and pentachlorophenol has been shown

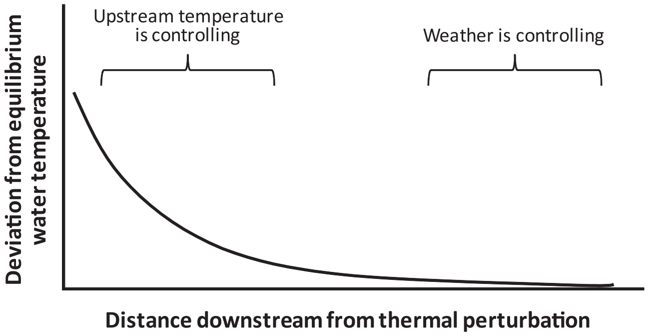

Figure 3. General representation of the influence of upstream thermal perturbations on downstream water temperature. Stream temperature at some distance comes into equilibrium with atmospheric conditions in terms of both cooling from a warm source and warming from a cool source. Modified from *Mohseni and Stefan* [1999, Figure 8].

to reduce contaminant concentrations in surface flow by up to ~90% in both laboratory and field experiments under certain conditions (e.g., warmer temperatures in summer) [*Pignatello et al.*, 1983; *Kim et al.*, 1995]. Other studies have shown fuel oxygenates in groundwater discharging to streams to be reduced by up to ~90% in both the field and laboratory [*Bradley et al.*, 2001b; *Landmeyer et al.*, 2010]. Mineralization of such organic toxins is irreversible so that any reductions are fully cumulative over distance.

On the other hand, transformations of inorganic compounds like metals or nutrients [*Triska et al.*, 1993; *Jones et al.*, 1995; *Gandy et al.*, 2007] are cumulative only to the extent that those reactions are not reversed, which varies with process and setting. For example, hyporheic processes have been shown to remove approximately 20% of the dissolved manganese (from former mining operations) flowing out of a drainage basin [*Harvey and Fuller*, 1998]. In terms of nutrients, nitrogen species such as ammonium and nitrate may be assimilated by biota or they may be converted among inorganic forms, but are still generally available to hyporheic communities (Figure 6). For example, hyporheic nitrification has been found to range from 1.7 to 38.5 ugN L^{-1} sediment h^{-1} and satisfy 89% of surficial algal demand in a desert stream minimally impacted by humans [*Jones et al.*, 1995]. On the other hand, denitrification can convert nitrate to less available forms and has been reported on the order of 168 ugN m^{-2} h^{-1} (accounting for 16% of nitrate removal) in small streams [*Mulholland et al.*, 2004]. Similarly, flow through gravel bars and meander bends has been shown to remove 68–98% of nitrate entering the hyoprheic zone, although in this case, hyporheic exchange was small enough relative to surface stream discharge that hyporheic denitrification overall comprised less than 0.1% of the stream load at base flow [*Kasahara and Hill*, 2007b]. Attenuation of both nutrients and toxins arriving in upstream surface flow (unlike those upwelling from deeper groundwater) will be sensitive to surface discharge, with the percent of surface flow exchanging through the hyporheic zone greatly decreasing during storm flow. Regardless, techniques such as riparian reforestation, large woody debris placement, and in-channel features can increase carbon sources and carbon retention, providing necessary substrates for certain hyporheic biogeochemical functions [*Webster et al.*, 1994; *Wallace et al.*, 1997; *Crenshaw et al.*, 2002].

The feasibility and efficacy of hyporheic restoration features are context-dependent in a variety of ways. For example, increasing channel planform complexity may be more feasible in lowland agricultural settings than in urban or steep mountainous areas, where the channel corridor may be too confined, either naturally or by human infrastructure. On the other hand, adding in-channel features is often straightforward in small streams of moderate to steep gradient [*Craig et al.*, 2008], but in larger rivers would be both more difficult (due to high flows and wide channels) and less effective (due to low channel gradients minimizing the ability of in-channel features to increase head gradients). Different types of features will also

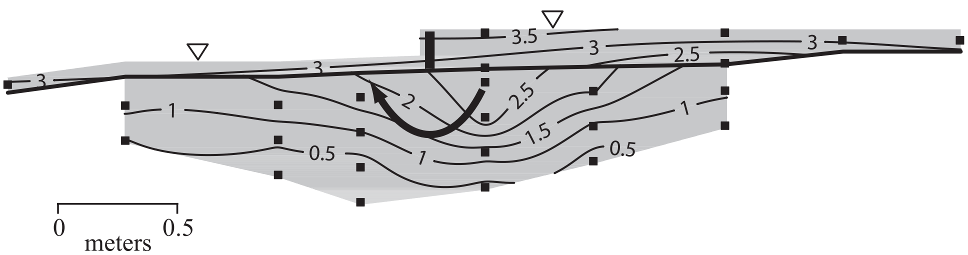

Figure 4. Longitudinal (side) view of stream and hyporheic zone temperatures near an in-stream structure. Contours denote variation in magnitude of diel temperature fluctuations (in degrees Celsius) in presence of in-stream structure (weir representing log or debris dams) that drives hyporheic exchange. Stream flow is from right to left. Modified from *Hester et al.* [2009, Figure 4d].

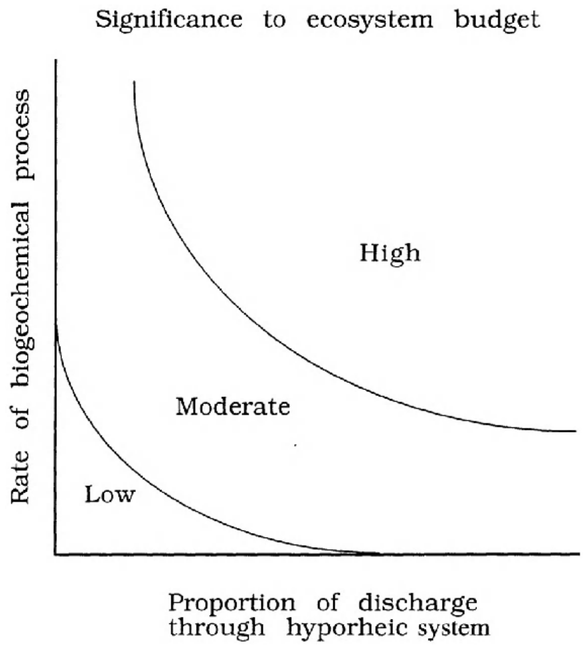

Figure 5. Hypothetical influence of hyporheic biogeochemical processes on overall stream metabolism and biogeochemical cycling. From *Findlay* [1995, Figure 2, p. 161]. Copyright 1995 by the American Society of Limnology and Oceanography, Inc.

have compounding effects in certain situations. For example, addition of in-channel features and coarser sediment may be necessary to fully engage new meanders; reconfiguring the channel with new meanders may not, by itself, enhance hyporheic exchange [*Kasahara and Hill*, 2007a, 2008]. Furthermore, certain common stream restoration strategies such as floodplain reconnection could enhance hyporheic exchange by incorporating multiple feature types (e.g., both in-channel features and planform complexity). Finally, substrate augmentation or substrate cleaning is generally easier in smaller watercourses, but can also be accomplished for portions of larger rivers [*Grant et al.*, 2006; *Meyer et al.*, 2008]. However, the long-term persistence of such changes is likely to be constrained by watershed conditions (i.e., sources of fine sediments) upstream.

3.2. Control by Stream Morphology

In order to estimate the impact of a specific set of proposed hyporheic restoration features, we must relate exchange and its benefits to stream morphology. A growing number of papers in the scientific literature are starting to provide this type of information. The most useful papers are those that provide quantitative relationships showing how hyporheic exchange or function varies with design or planning parameters (i.e., parameters that are necessary for creating stream restoration design drawings or channel management plans) such as hyporheic feature size or type. Examples of such papers are included in Table 2 in rows where "Feature size" or "Feature type" is specified in the "Engineering parameter" column. For example, the amount of hyporheic exchange induced by in-channel features like steps and debris dams is shown to increase with structure size and be affected by structure type [*Hester and Doyle*, 2008] (Figure 7a). Similarly, hyporheic flux is shown to increase with the sinuosity of meander bends [*Cardenas*, 2009] (Figure 7b). Other studies provide relationships between hyporheic function and other relevant parameters, such as the relationship between toxin mineralization and sediment texture [*Bradley et al.*, 2001b] (Figure 8). The size of Table 2 indicates there is much available knowledge that is applicable to hyporheic restoration (although this listing is not exhaustive). Nevertheless, quantitative data relating stream morphology to hyporheic function are not available in certain categories (e.g., toxin attenuation). For these cases, Table 2 contains references that quantify the impact of an example hyporheic feature (designated as "Example" in Table 2) or, failing that, references that quantify the impact of the hyporheic zone that may not be associated with a particular hyporheic feature or design parameter (designated as "Conceptual" in Table 2). Although not listed in Table 2, many of these references are also valuable sources of information concerning how hyporheic exchange induced by hyporheic features varies with hydrologic or geologic context (e.g., channel slope).

Data are limited for the effects of specific design parameters on hyporheic habitat and most of the hyporheic functions. In

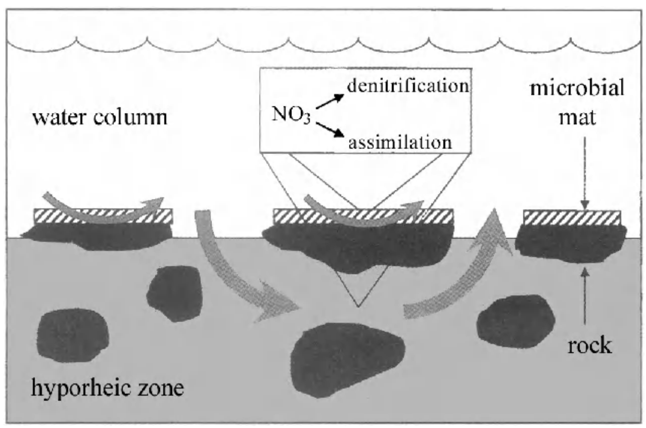

Figure 6. Conceptual model of influence of stream-water exchange through hyporheic zone and microbial mats on the channel bed on the fate of nitrate in a small desert stream. From *Gooseff et al.* [2004, Figure 2, p. 1886]. Copyright 2004 by the American Society of Limnology and Oceanography, Inc.

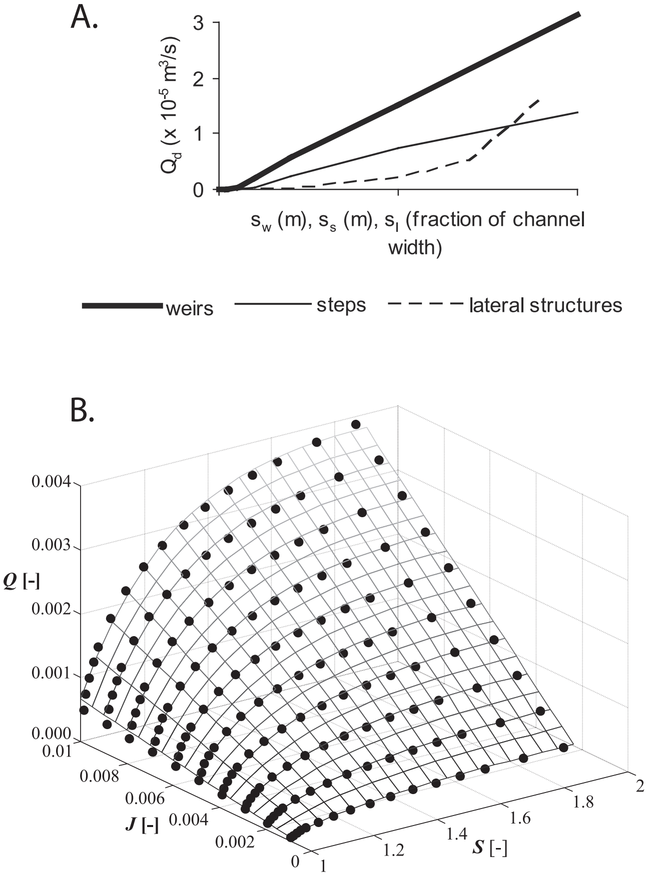

Figure 7. Relationships between restoration feature dimensions and induced hyporheic exchange. (a) Effect of in-channel feature size (s_w, weir height; s_s, step height; s_l, lateral structure width; normalized to maximum structure size) on induced downwelling flux (Q_d). From *Hester and Doyle* [2008, Figure 3]. (b) Effect of meander bend size (sinuosity, S) and channel slope (J) on lateral hyporheic exchange (Q). From *Cardenas* [2009, Figure 4].

particular, data appear to be most limited for the impact of stream morphology on degradation and transformation rates of toxins in the hyporheic zone. These studies can be more challenging than nutrient studies because sites must be found that have sufficient concentrations of the toxin of interest. Furthermore, the consortia of biofilms in hyporheic zones are not well known, and there has not yet been significant effort to modify these ecosystems by introducing or augmenting microbial or other populations. Metals, for example, have often been studied in acid mine drainage conditions (versus industrial locations), and the majority of organic pollutants (e.g., pesticides) have not been evaluated (although there are some data for riparian attenuation). Furthermore, most studies of toxin attenuation in the hyporheic zone have focused on sources in deeper groundwater rather than sources in upstream surface water. However, because hyporheic attenuation of toxins is the hyporheic function with one of the highest potential benefits to human health, it warrants additional study. In general then, the conceptual underpinnings of hyporheic function are well established, but practical data useable in the hyporheic design or planning process are limited, particularly for chemical reaction rates.

3.3. Control by Hydrologic Context

The ability of channel modifications to enhance hyporheic exchange will depend not only on head gradients in the channel and K, but also on the bounding hydrologic conditions in adjacent aquifers. In particular, strong background head gradients toward the stream can reduce or eliminate hyporheic exchange induced by stream morphology [*Cardenas and Wilson*, 2007b; *Hester and Doyle*, 2008]. These boundary conditions are not particularly amenable to engineering control, but do vary considerably in time and space [*Winter et al.*, 1998]. For example, streams are more strongly gaining in areas of steeper terrain or higher precipitation, all other factors being equal. Such variations have also been documented over the course of storm events and among seasons [*Wondzell and Swanson*, 1996; *Wroblicky et al.*, 1998; *Wondzell and Swanson*, 1999]. Even on a daily time scale, it is possible that the boundary condition forcing would influence the extent of the hyporheic zone or the magnitude of exchange [*Loheide and Lundquist*, 2009; *Wondzell et al.*, 2010]. Hence, the daily, seasonal and annual hydrologic conditions, under which one does preliminary assessment and for which one designs, may significantly influence the conclusions regarding hyporheic restoration targets and efficacy.

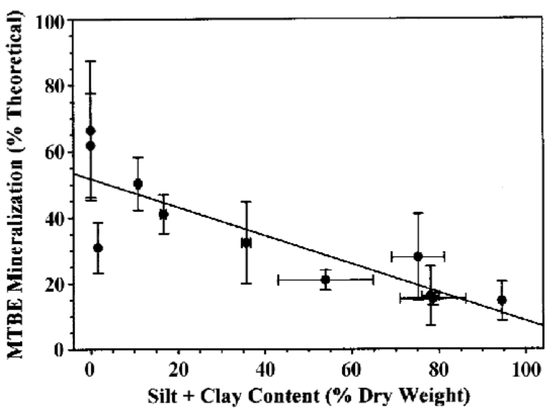

Figure 8. Relationship between shallow sediment texture and mineralization rate of the fuel oxygenate methyl tert-butyl ether (MTBE). From *Bradley et al.* [2001a, Figure 3].

4. RESTORATION PROCESS

Because stream restoration design guidance is already well established (elsewhere in this volume, also *Federal Interagency Stream Restoration WorkingGroup* (*FISRWG*) [1998], *Doll et al.* [2003], *Shields et al.* [2003], *Natural Resources Conservation Service* (*NRCS*) [2007]), we focus here on the process specific to restoring hyporheic exchange and function. We present a preliminary outline of the process and propose that detailed guidance be added to existing general restoration guidance in the future. In this section, we also consider the question of whether hyporheic restoration is appropriate for a particular restoration project in the first place. This question of whether to attempt hyporheic restoration is presented in parallel with the restoration process because many of the analyses required for design or planning also contribute to feasibility assessment.

4.1. Preparation and Data Gathering

The first step of any stream or river restoration effort is to determine project goals (Figure 9) [*Wohl et al.*, 2005]. Goals will vary with context, from complete restoration to some preindustrial or reference condition, to restoring a subset of reference conditions, or creation of functions that may not have originally existed [*Roni*, 2005; *NRCS*, 2007]. For hyporheic restoration, goals will generally entail enhancing hyporheic exchange generally or a more specific subset of possible hyporheic functions (section 2). Hyporheic restoration goals should also be set within the context of the full set of restoration goals for the entire stream restoration project. In other words, hyporheic restoration should be assigned a priority among the stated goals of the project, so that tradeoffs that inevitably crop up among goals as the project proceeds can be appropriately addressed.

Once project goals are determined, the next step is selecting sites for restoration (Figure 9) [*Roni et al.*, 2002]. Many projects will skip this step because a particular property or landowner agreement dictates the location of restoration work. However, where possible, this step can be extremely important in determining the ultimate success of the project. Site selection should consider as large a portion of the watershed as feasible and start with aspects of the site itself that vary with watershed position. For projects where a location is predetermined, these conditions will instead help determine the feasibility of hyporheic restoration at the given location. For example, sites with coarser bed sediment, deeper bedrock, and minimal hydrologic gaining would have higher potential to enhance hyporheic exchange. For larger rivers, availability of floodplain space for re-creation of channel planform complexity (e.g., meander bends, see Table 1) would also be beneficial. On the other hand, longer hyporheic residence times, along with sources of labile carbon, may be important for certain beneficial hyporheic chemical reactions. Such conditions are more often found in lowland watercourses [*Vannote et al.*, 1980], although considerable variability exists. Some of these preferred conditions may not coexist, resulting in a tradeoff. For example, coarser substrates tend to be in headwater regions where steep valley walls increase hydrologic gaining.

Site selection or evaluation should next consider the relation of the site to the watershed, particularly physical, chemical, and biological fluxes from upstream. For example, projects intending to augment nutrient processing or toxin

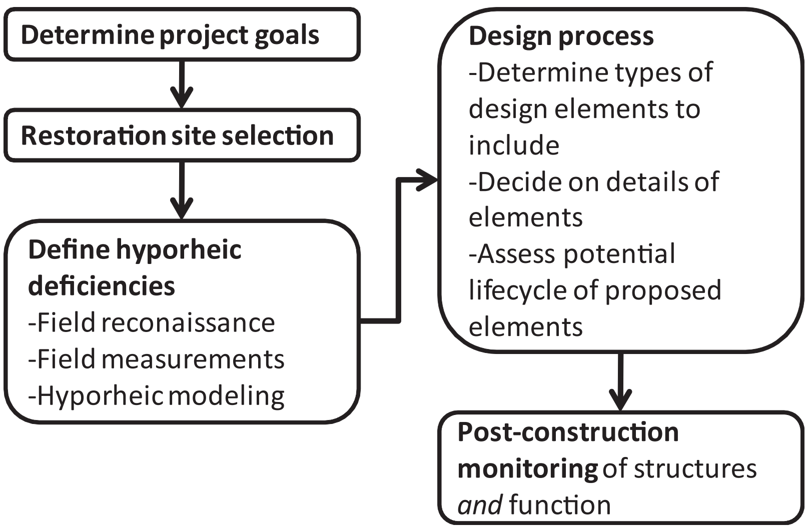

Figure 9. Recommended work flow for hyporheic restoration.

attenuation need to be located at the most effective location downstream of the source. On the other hand, projects intending to restore hyporheic habitat should consider the location of pollution sources that might render hyporheic restoration efforts ineffective. In addition, the cumulative impact of multiple projects needs to be considered in a spatial context. Finally, site selection should consider the long view, particularly the question of maintaining site functions. The most critical component of maintenance is sediment transport through the site. Excess fine sediment from urbanizing or agricultural areas upstream can quickly clog the surface layer of sediments, erasing any increase in hyporheic exchange created by a project. On the other hand, fully built-out areas upstream can decrease sediment loads and increase peak flows, leading to erosion of the streambed and any geomorphic features placed by the project.

Once the project site is determined, hyporheic deficiencies can be quantified (Figure 9). For restoration based on reference conditions, goals can be stated in terms of deficiencies relative to a reference condition [*Shields*, 1983; *Smith et al.*, 2008]. This reference condition may be some knowledge of historical conditions at the project site or current conditions at a more pristine but otherwise similar site elsewhere [*NRCS*, 2007]. Hyporheic deficiencies at a particular site can also be viewed as the hyporheic restoration potential of that site. Determining hyporheic deficiencies/potential can be challenging because hyporheic exchange is not readily observed and entails some combination of field observations, field measurements, and modeling to determine the prevalence of hyporheic exchange and associated hyporheic functions and/or fauna. Reconnaissance level observations include visual inspection for existing geomorphic features that enhance exchange (e.g., debris dams and meander bends) and sediment bed texture. Reconnaissance observations for hyporheic function and/or fauna will be more challenging but may still be possible. For example, carbon entrained in the hyporheic zone would be an indicator of potentially favorable conditions for certain hyporheic reactions and might be ascertained by looking for organic matter sources (e.g., leaves and woody debris from riparian trees), presence of organic debris among surface and shallow subsurface sediment, and evidence of bed load transport. Hyporheic fauna can also be sampled [*Hauer and Lamberti*, 2007], though it should be considered that some are subject to seasonal or life stage patterns of hyporheic occupation.

While reconnaissance is generally qualitative, field measurements are by definition quantitative. Measurements are therefore recommended for determining hyporheic deficiencies/potential if possible because they can assess a wider variety of hyporheic parameters, and they make assessment of deficiencies more concrete. Nevertheless, measurements are more involved than simple reconnaissance. Local hyporheic exchange can be measured in many ways, including directly via seepage meters or estimated using Darcy's law informed with measurements of vertical hydraulic head gradient between surface water and groundwater (measured using water levels in piezometers) and estimates of sediment hydraulic conductivity (K, measured by falling head tests in piezometers) [*Kalbus et al.*, 2006; *Rosenberry*, 2008]. Hydraulic conductivity as well as hyporheic exchange itself can also be estimated using heat and other tracers [*Lapham*, 1989; *Harvey and Wagner*, 2000; *Su et al.*, 2004]. Conservative tracers released in the stream, together with solute transport modeling, can be used to quantify transient storage, which is caused by water temporarily residing in surface transient storage zones (pools, eddies, etc.) and the hyporheic zone [*Bencala and Walters*, 1983]. New transient storage models that distinguish between surface and hyporheic transient storage zones provide enhanced ability to quantify hyporheic exchange from stream solute study data [*Marion et al.*, 2008; *Briggs et al.*, 2009], and natural fluctuating tracers can be used even in larger rivers [*Knust and Warwick*, 2009]. Isotope tracers can be used to determine subsurface flow paths [*Domenico and Schwartz*, 1998] and, together with transient storage modeling, may be used to quantify hyporheic exchange [*Gooseff et al.*, 2003] and the reactive capability of the hyporheic zone [*Mulholland et al.*, 1997; *Thomas et al.*, 2003].

The presence of microbial activity and the potential for chemical reactions in the subsurface can be assessed by sampling for biogeochemical parameters such as oxygen, nutrients, carbon, and metals [*Duff et al.*, 1998; *Hauer and Lamberti*, 2007]. Microbial populations can also be sampled directly using freeze core techniques [*Moser et al.*, 2003]. Field measurements should generally be guided by the results of the reconnaissance. For example, downwelling would be expected just upstream of geomorphic features that reconnaissance indicated create a head drop in the stream.

Finally, the most involved method for determining hyporheic deficiencies/potential is hydrodynamic (e.g., groundwater flow) modeling. This process would start with calibrating a coupled surface water-groundwater hydraulic model using field measurements such as hydraulic head, stream or river discharge, and K. Model components to simulate solute transport and transformation can then be added if desired and potentially require additional field measurements to determine parameters that quantify dispersion, sorption, and reaction processes. Modeling would be used to estimate the extent and degree of hyporheic exchange and possibly hyporheic functions currently occurring in an existing reach. When information is available concerning historical channel configuration, these models can also estimate the historical extent of hyporheic exchange and functions.

Hydrodynamic modeling is particularly useful because the results can be used to compare current and historical hyporheic function to different restoration alternatives. Nevertheless, proper model execution requires considerable field data, training, and time. Hydrodynamic modeling may therefore be prohibitively expensive for typical restoration projects unless a model already exists from a previous project at that location.

A final consideration before beginning design or planning is determining constraints at the chosen site. Working around existing infrastructure is often necessary in urban areas [*Bernhardt and Palmer*, 2007]. In addition, for projects without latitude for site selection, many site selection parameters become constraints at the given site. Many restoration sites will be chosen based on land availability, financial constraints, or for potential for nonhyporheic components of stream restoration, which may not always align with hyporheic restoration potential. If site options are restricted to those where hyporheic restoration potential is poor, hyporheic restoration may not be a realistic project goal. In areas with good hyporheic restoration potential, the cost of adding hyporheic restoration as a component of a stream restoration project depends on the extent to which the desired hyporheic restoration features can also fulfill other project goals. For example, many hyporheic features (e.g., log dams and steps) may also provide habitat [*Hilderbrand et al.*, 1997; *Roni et al.*, 2006] and channel stabilization [*NRCS*, 2007] benefits.

4.2. Design

Design and planning entail taking project goals (section 2), translating them into hyporheic features using available data (section 3), and subjecting them to site constraints (section 4.1). Because these processes are creative and individual, and thoroughly addressed in many textbooks [e.g. *Dym and Little*, 2003], we do not discuss them in detail here. Further, many of the hyporheic features listed in Table 1 (e.g., meander bends and in-channel features) are frequently installed for other purposes, and the considerable existing guidance [e.g., *FISRWG*, 1998; *NRCS*, 2007] and theory [e.g., *Boano et al.*, 2006; *Tregnaghi et al.*, 2007; *Chin et al.*, 2008] for design of such features readily applies to design for hyporheic restoration. We therefore focus here on guidance specific to hyporheic restoration.

To begin, two basic levels of decisions are necessary. The first is to decide what types of hyporheic features to include that will enhance exchange. Examples include in-channel features, meander bends, and gravel augmentation. Other techniques, such as riparian vegetation or large woody debris can increase carbon sources and other desirable hyporheic functions. The second level of decision concerns the details of each type of hyporheic feature or technique. For example, for in-channel features, size, type, number/density, material, and placement/layout need to be determined. In many cases, decisions will be appropriately made through consulting suitable engineering data (e.g., Table 2) and applying traditional design practice and best professional judgment. However, many projects intending to augment hyporheic restoration will benefit from numerical modeling of hyporheic processes. The most important process to model is hyporheic water flow through channel sediment [e.g., *Wroblicky et al.*, 1998; *Poole et al.*, 2004; *Gooseff et al.*, 2006], to estimate the extent of exchange zones, hyporheic flux rates, and hyporheic residence times in the project reach before and after the project is built. Solute transport and transformation models can be added to the flow models to simulate biogeochemical processes [e.g., *Lautz and Siegel*, 2006], but much can also be inferred about these processes from the flow model alone. A two-dimensional (2-D) vertical model will often be sufficient for in-channel features. On the other hand, features such as meander bends, side channels, or islands will require horizontal 2-D models or 3-D models. In-channel stages and floodplain groundwater levels will be necessary for a flow model, but may be already available as part of a broader stream restoration effort.

Because streams and rivers are dynamic, a final consideration is expected design life of constructed features. There are three primary ways that enhanced hyporheic function can diminish over time. First, installed geomorphic features can erode, either slowly over time, or spectacularly during spates, leaving a simplified channel. Second, features can be buried where significant coarse sediment is supplied from upstream, again simplifying the channel [*Elmore and Kaushal*, 2008]. Finally, hyporheic flow can be reduced or eliminated where fine sediment from upstream clogs sediment pores [*Wood and Armitage*, 1997; *Velickovic*, 2005]. All three processes can occur as part of natural cycles. Floods have been documented to substantially modify hyporheic zones by completely re-organizing the morphology of a reach [*Wondzell and Swanson*, 1999]. However, channel form just prior to any given flood is, in turn, due partly to earlier floods. It is therefore expected that channels that have characteristics appropriate for the channel setting (e.g., sediment load and flood hydrograph characteristics) will be realigned and reconfigured in flood events so that hyporheic exchange, though rearranged, will be maintained overall. Such dynamic equilibria can occur with both in-channel features and channel planform features. Human activities can greatly impact such equilibria. For example, construction can greatly increase fine sediment loading and hence sediment clogging [*Wood and Armitage*, 1997]. On the other hand, fully urbanized areas can increase peak discharge and decrease sediment supply to magnify erosive power [*Leopold*, 1968].

Funding organizations, regulatory agencies, and design engineers need to be clear about the expected lifetime of any engineered features and associated hyporheic functions that are part of a restoration project. Design life can be specified in terms of expected years of service and can, in theory, be broken down by function. In the case of in-channel features, lifetime can alternatively be specified as the return interval of storm that is expected to compromise the structures. Design life of geomorphic features can generally be extended against erosive failure by increasing the size of construction materials (log or rock size), though the consequences related to hyporheic exchange potential (i.e., K) should be considered. Conversely, design life can be reduced if natural regeneration of features is planned (e.g., riparian forest restoration to restore self-forming in-channel features such as log dams). On the other hand, burial of features by large sediment or clogging of interstices by fine sediment can only be addressed by reducing upstream sediment loading. For hyporheic restoration, the most common threat to restored function will likely be fine sediment, such that rigorous sediment control practices at all current or future upstream agricultural or construction sites will be critical to project success.

4.3. Implementation and Monitoring

Implementation of hyporheic restoration largely follows the established trajectory for stream and river restoration, in general [*FISRWG*, 1998; *NRCS*, 2007]. Project features (hyporheic features) constructed for hyporheic enhancement purposes will have similar installation to similar features constructed for other restoration purposes (e.g., in-stream structures [see *Shields*, 1983]). Any procedural modifications required for hyporheic restoration will probably differ mainly in terms of degree. For example, implementation of hyporheic restoration projects may require greater excavation than those without a hyporheic component if replacement of significant amounts of bed material is desired to augment hyporheic exchange. Such excavation will cost more time and money, may require additional planning to avoid underground utilities and other infrastructure, and will be more disruptive geomorphically and ecologically.

Postconstruction monitoring is critical to the success of any restoration project [*FISRWG*, 1998; *Roni and Quimby*, 2005]. Monitoring hyporheic restoration aims to determine whether the project has enhanced hyporheic exchange to the desired degree and whether those improvements last over time. Monitoring is particularly critical, while hyporheic restoration practice is still in its infancy. Adopting an adaptive management approach where past lessons learned can drive future project adjustments can therefore improve the chances of meeting project goals. The basic strategy for monitoring hyporheic restoration components of stream restoration projects is similar to that for other restoration goals. The main difference concerns how hyporheic enhancement is measured because hyporheic processes generally cannot be viewed from the surface, unlike more traditional restoration monitoring metrics such as bed scour, growth of riparian vegetation, or the presence of fish. Such measurement techniques are the same as those introduced in detail above to determine hyporheic deficiencies during the preparation and data gathering phase of a project.

The discussion in this chapter focuses on a traditional view of stream or river restoration practice, which generally entails direct intervention in stream channels, often with heavy construction equipment [*Bethel and Neal*, 2003; *Downs and Gregory*, 2004]. Such traditional approaches will no doubt continue in the short term out of inertia and can be helpful to jump start recovery or where rehabilitation cannot occur any other way (e.g., stream daylighting [*Elmore and Kaushal*, 2008]). Nevertheless, direct intervention in stream and river channels and floodplains creates a significant geomorphic and ecological disturbance, particularly to systems that have some remnant ecological functionality [*Tikkanen et al.*, 1994]. Even if great care is taken, the direct intervention process is inherently disruptive. As such, a number of less invasive approaches that conceptualize the restoration process on a larger scale are preferred where feasible. These processes are more passive in nature and are both ecologically and economically more sustainable in the long term.

The first larger-scale approach to consider is riparian corridor restoration [*Sterba et al.*, 1997; *Berg et al.*, 2003], which can have many benefits for hyporheic exchange. For example, a healthy riparian forest can supply sufficient tree fall to a stream to create a naturally sustainable population of in-channel features (e.g., log dams or steps) that can significantly enhance hyporheic exchange. Riparian forest may also supply a range of particulate organic carbon (e.g., leaves and twigs), which can become entrained in the hyporheic zone during bed load turnover during storms. Such entrained carbon sources can then encourage conditions necessary for certain hyporheic functions (e.g., denitrification [*Schindler and Krabbenhoft*, 1998; *Duff and Triska*, 2000]). A riparian corridor provides room for channel migration, which also increases recruitment of in-channel wood [*Berg et al.*, 2003]. Further, sufficient corridor width is important to accommodate meander bends, which can also promote hyporheic exchange [*Boano et al.*, 2006]. In areas of incised channels and fine sediment banks (common along the eastern seaboard of the United States), a riparian corridor allows room for sufficient channel bank setbacks which reduces the rate of sediment sloughing into the channel, reducing corresponding streambed clogging and associated reductions of hyporheic exchange. Finally, riparian restoration can also help prevent upland sources of fine sediments from urbanizing or agricultural areas from reaching receiving streams, decreasing a source of clogging sediments. Riparian corridor restoration can be accomplished actively, through grading and planting of riparian and floodplain areas or, passively, by purchasing land and allowing nature to take its course.

Ultimately, the most sustainable solution to aquatic impairment is watershed-scale restoration and planning [*Wohl et al.*, 2005]. Watershed restoration entails altering upland practices and conditions to improve hyporheic exchange or function in downstream waterways [*Palmer et al.*, 2010]. Many of the watershed practices that benefit hyporheic exchange are storm water control best management practices that also have a variety of other stream and watershed benefits. For example, reducing or disconnecting impervious surfaces in urban areas decreases storm flow and increases base flow, which may increase the overall percentage of flow that exchanges through the hyporheic zone. Similarly, implementing tighter sediment controls at construction sites would reduce streambed clogging and prevent reductions of hyporheic exchange. At another level, a watershed view of restoration involves coordinating restoration projects throughout a watershed in order to maximize overall watershed benefit from individual restoration actions. A watershed view of hyporheic restoration is tightly coupled with site selection for individual projects, but differs conceptually in that site selection focuses on maximizing the hyporheic potential of a single project with the watershed context as a given, while watershed planning considers multiple actions coordinated throughout a watershed to achieve increased hyporheic exchange. A watershed approach to restoration can be accomplished on several levels. At a minimum, the cumulative impact of multiple stream, river, or riparian restoration projects can be taken into account at the watershed scale, particularly for hyporheic functions that impact surface water. A more ambitious approach could site projects where they will have the most impact (see previous section) and may involve rezoning or moving buildings out of floodplains. The most ambitious approach would set aside significant swaths of land within watersheds in order to restore large contiguous regions of natural habitat. Many of these activities would also benefit many aspects of streams and rivers beyond hyporheic exchange.

5. VISION FOR THE FUTURE

While awareness of the importance of the hyporheic zone is increasing among practitioners and regulatory officials involved in stream and river restoration, hyporheic restoration is still rarely incorporated as a project goal, and appropriate guidance does not exist. We argue that hyporheic function should be formally incorporated as a restoration goal as more governments adopt mitigation requirements for stream and river impacts, and the goal of restoring stream ecosystem function becomes more common. Formal requirements for hyporheic restoration should then lead to content on hyporheic restoration being added to existing and future stream and river restoration guidance documents [e.g., *FISRWG*, 1998; *NRCS*, 2007]. Hyporheic exchange and function then needs to be monitored after project construction at restoration sites along with other restoration success criteria. Guidance can then be refined and updated to create meaningful yet flexible definitions of hyporheic restoration success [in the sense of *Palmer et al.*, 2005].

Research should occur in parallel with hyporheic restoration activity, continuing to expand our understanding of how the hyporheic zone functions in pristine and human-dominated settings. To begin, we need to know more about how species composition and abundance of hyporheic fauna vary with the degree and type of impairment in order to assess hyporheic health before and after restoration. Additional research needs to explicitly evaluate how hyporheic functions vary with design parameters of hyporheic features. For example, we need to know more about how temperature moderation, nutrient processing, and toxin attenuation vary with size, type, and sediment texture of geomorphic features such as riffles, steps, and meander bends. Engineering data are most limited for toxin attenuation in particular, a hyporheic function with one of the highest potential benefits to human health. For this function, studies associating function with particular hyporheic features such as riffles are needed, as are studies that focus on contaminant sources from upstream surface water (more studies already exist for contaminants coming from deeper groundwater). Hyporheic attenuation of additional toxic pollutants (e.g., pesticides) and in additional environments (e.g., metals in nonacidified systems) needs to be studied. Finally, we need to know much more about how hyporheic functions measured and understood on the reach scale produce cumulative effects on the watershed scale [*Harvey and Fuller*, 1998; *Boulton*, 2007; *Smith et al.*, 2008]. In addition to research experiments, many of these questions would benefit from examining and synthesizing data being collected at numerous restoration projects by government and the private sector. Such data could potentially be incorporated into existing synthesis programs such as the National River Restoration Science Synthesis (NRRSS Database, National Biological Information Infrastructure, 2008, available at http://www.nrrss.nbii.gov/) [*Bernhardt et al.*, 2005].

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Diversity of Macroinvertebrate Communities as a Reflection of Habitat Heterogeneity in a Mountain River Subjected to Variable Human Impacts

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Most sections of the Czarny Dunajec River, Polish Carpathians, have been considerably modified by channelization and gravel mining-induced channel incision. As a result, the river morphology now varies from a single-thread, incised, or regulated channel to an unmanaged, multithread channel. For 18 cross sections with one to five flow threads, diversity of benthic invertebrate communities was determined and compared with low-flow channel width and the variation in flow depth, velocity, and bed material size. The increased number of flow threads in a cross section was associated with a larger aggregated width of low-flow channels and greater complexity of physical habitat conditions. Single-thread cross sections hosted four to seven invertebrate taxa, mostly eurytopic, which represented two or three functional feeding groups. In multithread cross sections, 7 to 19 taxa were recorded, with the assemblages representing all five functional groups of invertebrates and comprising taxa typical of both lentic and lotic habitats, sometimes within the same braids. The number of invertebrate taxa increased linearly with increasing number of low-flow channels in a cross section and variation in flow depth, velocity, and bed material grain size, while it was unrelated to flow width. Thus, it is the increase in habitat heterogeneity rather than simple habitat enlargement that supported the increased diversity of macroinvertebrate fauna in the multithread cross sections. This study shows that the simplification of flow pattern and the resultant homogenization of physical habitat conditions, caused by human impacts, is reflected in notable impoverishment of invertebrate communities and that restoration of morphological complexity of the river will be necessary for future recovery of these communities.

1. INTRODUCTION

Under natural conditions, mountain rivers are dynamic systems exhibiting three-dimensional (3-D) connectivity [*Kondolf et al.*, 2006] and supporting a vast array of aquatic and terrestrial habitats for fauna and flora [*Tockner et al.*, 2003]. Over the last century, mountain rivers in densely populated areas experienced a variety of pressures related to human activity. Common modifications to the rivers worldwide included channelization works, in-channel gravel mining, as well as diversions and regulation of flow by dams, and were frequently accompanied by basin-scale changes in land use [*Bravard and Petts*, 1996; *Wohl*, 2006, and references

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therein]. These modifications have led to detrimental changes in the physical structure of the hydrosystems such as a reduction in morphological complexity of the channels and changes in sediment flux, isolation of rivers from the floodplains, channel instability and incision, in-channel concentration of flood flows, and alteration to channel boundary material, including bed armoring and bedrock exposure [e.g., *Bravard et al.*, 1997; *Kondolf*, 1997; *Liébault and Piégay*, 2001; *Surian and Rinaldi*, 2003; *Rinaldi et al.*, 2005]. As the aforementioned changes frequently resulted in a degradation of habitat integrity of the rivers and their riparian zones [*Muhar and Jungwirth*, 1998], they were usually followed by a decrease in the biodiversity of aquatic and riparian ecosystems [e.g., *Roux et al.*, 1989; *Muhar et al.*, 2008].

Similar impacts, accompanied by a tendency toward channel incision, are also typical of Polish Carpathian rivers [*Wyżga*, 2008], which were significantly altered because of direct and indirect human disturbances during the last century. Most significant impacts comprised channelization works, leading to slope-channel decoupling, bank stabilization, substantial channel narrowing, and an almost complete replacement of former multithread channels by straight, single-thread channels [*Wyżga*, 1993, 2001; *Korpak*, 2007], extensive gravel mining [*Radecki-Pawlik*, 2002; *Rinaldi et al.*, 2005], construction of check dams, and partitioning the rivers with dam reservoirs. The increase in transport capacity of Carpathian rivers caused by their channelization and the reduced availability of sediment for fluvial transport have led to rapid bed degradation that resulted in channel incision of the rivers by 0.5 to 3.8 m during the twentieth century [*Wyżga*, 2008]. In many river reaches, the rate of incision increased during the second half of the century because of further reduction in catchment sediment supply resulting from land use changes, mainly an increase in forest cover, and the associated increase in slope stability [*Lach and Wyżga*, 2002]. Although construction of concrete weirs in the channels locally arrested bed degradation, it further altered bed material transport and disrupted the continuity of the rivers for fish [*Bojarski et al.*, 2005; *Wiśniewolski*, 2005]. Finally, with increased flow capacity of deepened channels, the rivers became largely disconnected from their floodplains, which reduced delivery of organic matter and wood debris from the riparian zone and limited the availability of remnant channels and ponds in the floodplains for river biota.

Increasing recognition of the adverse effects of human pressures on river channels has recently led to a great number of restoration projects undertaken worldwide to improve the geomorphological and ecological conditions of modified watercourses [*Lüderitz et al.*, 2004; *Hillman and Brierley*, 2005; *Habersack and Piégay*, 2008]. In the European Union, these restoration efforts have been strengthened by the requirement of the Water Framework Directive [*European Parliament, Council*, 2000] to reestablish good ecological status of rivers by 2015. The presence and structure of aquatic biota communities are commonly used to monitor the ecological integrity of watercourses [e.g., *Jungwirth et al.*, 2000; *Hering et al.*, 2006], but to date, the monitoring has focused on changes in riverine biocoenoses resulting from a degradation of water quality. If restoration measures are to be successful, they must be based on understanding to what extent the diversity and structure of physical habitat in watercourses influence the condition of riverine ecosystems. Hence, identification of the relationships between the structure of river biocoenoses and the hydromorphological

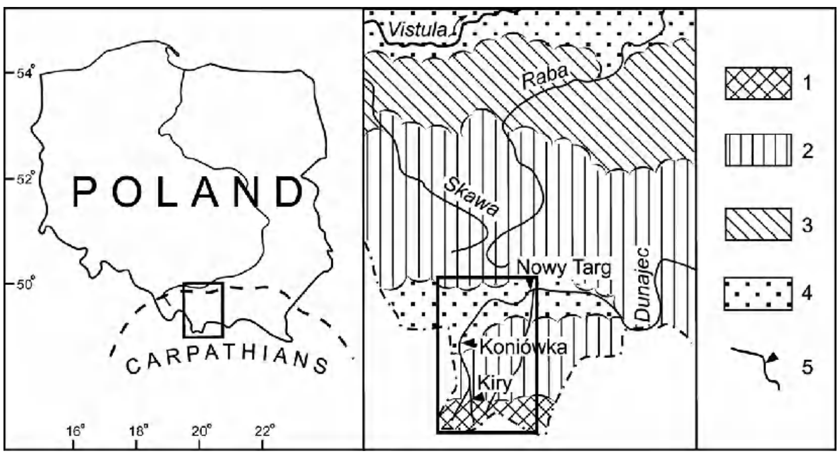

Figure 1. Location of the Czarny Dunajec River in relation to physiogeographic regions of southern Poland. Numbers indicate the following: 1, high mountains; 2, mountains of intermediate and low height; 3, foothills; 4, intramontane and submontane depressions; 5, flow-gauging stations. Box indicates the area shown in detail in Figure 2.

characteristics of the habitats is now becoming a focus of the efforts to integrate ecological, geomorphological, and hydrological expertise on rivers [*Vaughan et al.*, 2009]. This chapter adopts such an ecohydromorphological approach in an attempt to determine relationships between the diversity and composition of benthic invertebrate communities and physical habitat conditions in a number of cross sections of the Czarny Dunajec River, Polish Carpathians, which during the past few decades was subjected to considerable though spatially varied modifications caused by human impacts.

2. STUDY AREA

The Czarny Dunajec constitutes the upper part of the Dunajec (Figure 1), the second largest river of the Polish Carpathians. Its headwaters are located in the high-mountain Tatra massif, which determines the hydrological regime of the river, with low winter flows and floods occurring between May and August. Fed by coarse material in the Tatras, the river formed a noncohesive alluvial plain in the Tatra Mountains foreland, flowing in a braided channel for a length of 38 km to its confluence with the Biały Dunajec River (Figure 2).

The relatively complex multithread channel pattern along the river course changed during the second half of the twentieth century, when the channel was subjected to considerable, spatially variable human impacts [*Krzemień*, 2003; *Zawiejska and Krzemień*, 2004; *Zawiejska and Wyżga*, 2010]. In the 1950s and 1960s, gravel was intensely mined from the riverbed at several locations [*Dudziak*, 1965], and in subsequent decades, larger cobbles were widely extracted from the channel [*Krzemień*, 2003]. Up to 3.5 m of channel incision followed these activities, and in many sections within the Gubałówka Hills, the alluvial channel has transformed to a bedrock system [*Zawiejska and Wyżga*, 2010]. A 7 km long stretch in the middle river course was progressively channelized in the 1960s–1990s, resulting in (1) replacement of the former multithread channel by a nearly straight, single-thread channel, (2) narrowing of the channel by up to five times, and (3) a reduction in channel gradient due to the construction of a number of concrete drop structures ranging from 0.7 to 2.1 m in height. Downstream of the channelized stretch, the river flows in an unmanaged channel over a length of 4 km, and its channel pattern ranges from braided, through island braided, to heavily island braided. Farther downstream, channelization works, which were completed by the 1980s, resulted in straightening and considerable narrowing of the river throughout its lower course, but the channel gradient was not reduced by construction of drop structures.

This chapter reports on the study performed in a 17 km long reach in the middle course of the river, which is located on a large, fluvioglacial-alluvial fan formed during the Quaternary within the intramontane Orawa-Nowy Targ Basin [*Baumgart-Kotarba*, 1992]. Owing to this location, the catchment area increases relatively little along the reach, and the river receives no significant inflow from tributaries (Figure 2). The reach comprises a deeply incised channel in its upper part, a regulated channel with drop structures in its middle part, an unmanaged channel in the lower part, returning to a regulated channel at its downstream end. The study reach exhibits considerable variation in river morphology [*Wyżga and Zawiejska*, 2005], with single-thread and multithread sections along with incised and vertically stable sections. These contrasting channel conditions, in conjunction with differences in channel management, result in highly variable physical habitat conditions within the reach, which are likely to be reflected in differences between local biocoenoses.

3. STUDY METHODS

Eighteen river cross sections, representing the range of hydromorphological conditions present in the reach, were selected for the study. Initial observations indicated that a scale of the reduction in habitat complexity caused by river channelization and channel incision was greater in pools than riffles. Therefore, cross sections were located across pools in order to examine differences in macroinvertebrate communities and physical parameters of habitats between particular types of river morphology and channel management rather than those related to pool-riffle sequences in the river. During base flow conditions in late March and the first 2 weeks of April 2008, elevation profiles were surveyed at the cross sections, followed by measurement of water depth, flow velocity, and mean grain size of surface bed material at 1 m intervals across the low-flow channel(s). Flow velocity was measured at 0.6 of depth (depth-averaged velocity) and 1 cm above bed surface (near-bed velocity) using Ott Nautilus C 2000 electromagnetic current meter. The construction of this current meter, lacking a propeller, enables measurements at a short distance above the bed surface. Transect sampling was used to establish mean grain size of gravelly sediments. This sampling method yields the same results as the "grid by number" procedure [*Diplas and Sutherland*, 1988], and both are equivalent to bulk sieve analysis [*Diplas and Sutherland*, 1988; *Shirazi et al.*, 2009]. At each measurement site within a cross section, 15 particles were collected from the bed; this sample size was selected to keep the sampling within the area characterized by hydraulic measurements. The distribution of the "b" axis diameters of collected particles was then determined, and their mean size was calculated as an average of the 3rd, 8th, and 13th grain in the sequence. The calculated

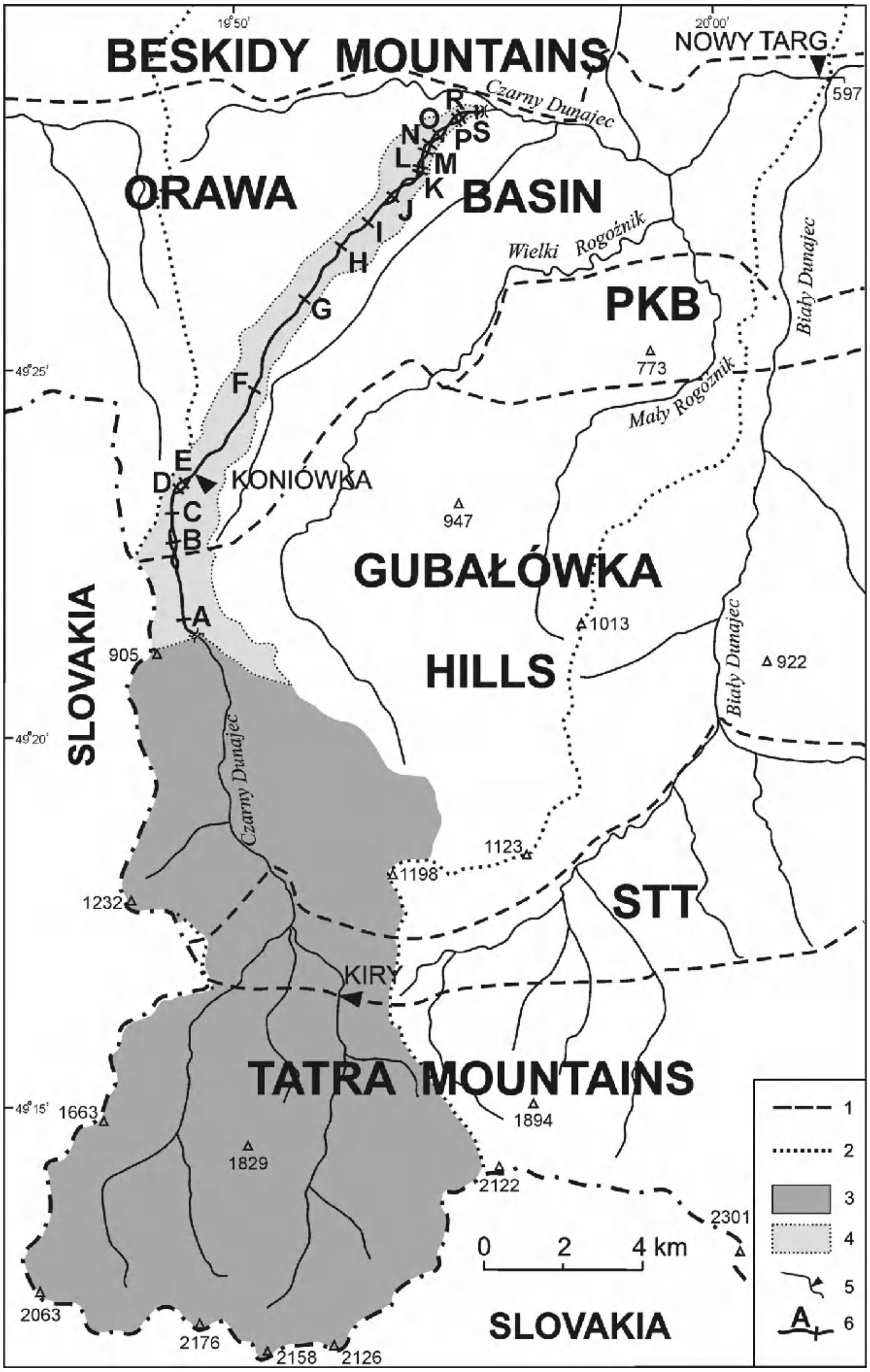

Figure 2. Drainage network of the Czarny Dunajec catchment and detailed setting of the investigated river cross sections. Numbers indicate the following: 1, boundaries of physiogeographic units; 2, boundary of the Czarny Dunajec catchment; 3, the Czarny Dunajec catchment to the beginning of the study reach; 4, catchment area increment along the study reach; 5, flow-gauging stations; 6, river cross sections investigated. PKB is Pieniny Klippen Belt; STT is Sub-Tatran Trough.

mean grain size reflected the 20th, 50th, and 80th percentiles of the grain-size distribution and is the closest available approximation of the formula of *Folk and Ward* [1957]. Samples of sandy and silty sediments were taken to a laboratory where their grain-size distribution was determined using sieving or hydrometer analyses. Mean grain size of the sediments was subsequently calculated from the same percentiles of the distribution. Finally, on the basis of data for the sites spaced at 1 m intervals, means and coefficients of variation of the analyzed parameters were calculated for each cross section.

The cross sections were sampled for benthic invertebrates on 21 and 22 March 2008, at similar low-flow conditions. Sampling was conducted in a single period of 1 year to maintain invertebrate communities representative of the habitat conditions established during the measurement campaign in this highly dynamic river. Since the abundance of particular groups of invertebrates varies within a year [e.g., *Hynes*, 1970; *McCafferty*, 1998], the sampling did not focus on this aspect but rather aimed to identify the taxonomic and functional diversity, as well as the taxonomic richness, of benthic invertebrate communities in particular cross sections or low-flow channels within a cross section. As the maximum number of flow threads in the study reach varies with time, we collected the same number of invertebrate samples from each low-flow channel rather than from every cross section. This provides a repeatable sampling strategy should the study be replicated in subsequent years when the number of low-flow channels in the cross sections may have changed. In each low-flow channel, samples were collected at three sites representing principal, visually identified habitat conditions (combinations of water depth, flow velocity, and substrate type). At each of the sampling sites, invertebrates were collected from approximately 0.25 m^2 of the channel bed using triangular dip net, Ekman grab, mosquito dipper, and tweezers (for gathering the sprawlers from cobbles). Special attention was paid to keep the sampled area and the sampling time similar for all samples [cf. *Fiałkowski et al.*, 2005], whereas the number of collected individuals varied considerably among the samples depending on a substrate type and, especially, the presence/absence of caddisflies exhibiting a strongly aggregated distribution (*Philopotamus* sp.). The invertebrates were identified in a laboratory, partly from non-preserved material during 2 to 3 days after the sampling and partly from the samples preserved with 70% ethanol. The rule of unquestionable identification was assumed, with all the specimens identified to the lowest practical taxonomic level.

In the analysis of riverine habitats, the general pattern of river morphology in the surveyed cross sections was considered in the context of various human impacts that had modified particular river stretches in the past decades. Linear regression models were estimated to test the statistical significance of hypothesized relations between measured physical characteristics of the habitats and the number of low-flow channels in a cross section. Differences in the variation of habitat conditions between single-thread and multithread river cross sections were then examined by analyzing scatter diagrams for the pairs of measured habitat characteristics. Differences in physical habitat conditions between individual braids of multithread cross sections were demonstrated with cross section L, which contains five low-flow channels; their statistical significance was analyzed with the Kruskal-Wallis test.

In the analysis of invertebrate communities, taxa indicative of good and high water quality [*Hawkes*, 1998; *Dumnicka et al.*, 2006] were identified, and their distribution in the study reach was examined to determine whether differences in the taxonomic richness of the communities observed between surveyed cross sections could be attributed to a decrease in water quality. Differences in the number of invertebrate taxa between single-thread and multithread cross sections, as well as between single-thread cross sections and individual braids of the multithread channels, were analyzed with the Mann-Whitney test. Differences in the taxonomic richness and composition of invertebrate communities between particular cross sections and between individual low-flow channels of multithread cross sections were then measured by Jaccard's similarity coefficients which range from 0% (no common taxa) to 100% (identical composition). Taxa from various functional feeding groups [*Cummins and Klug*, 1979; *Lampert and Sommer*, 2007] and taxa preferring either lentic or lotic habitats, as well as eurytopic ones [*Kołodziejczyk and Koperski*, 2000; *Kownacki*, 2003], were also identified and their distribution in the study reach determined to highlight differences in the composition of invertebrate communities between single-thread and multithread channel sections. Finally, simple and multiple regression models were estimated to test the statistical significance of relations between the number of invertebrate taxa and physical characteristics of riverine habitats in the surveyed cross sections.

4. RESULTS

4.1. General Pattern of Variation in Channel Morphology in the Study Reach

Single-thread, channelized, and incised channel sections represent about 60% of the total length of the study reach. However, of the 18 investigated cross sections (Figure 2), only 6 were located in single-channel sections in order to avoid overrepresentation of this type of channel morphology within the sample, whereas the remaining 12 were multi-thread cross sections with between two and five low-flow channels.

Cross sections A through E were located in the incised, upper part of the study reach and made up two single-thread cross sections (A and C) and three bar-braided cross sections. Of the bar-braided cross sections, E had two low-flow channels, while B and D had three low-flow channels. Cross sections F through H were located in the channelized river stretch that included drop structures and was typified by a single-thread morphology, narrow channel, and artificially reinforced banks (Figures 3 and 4, cross section F). The same

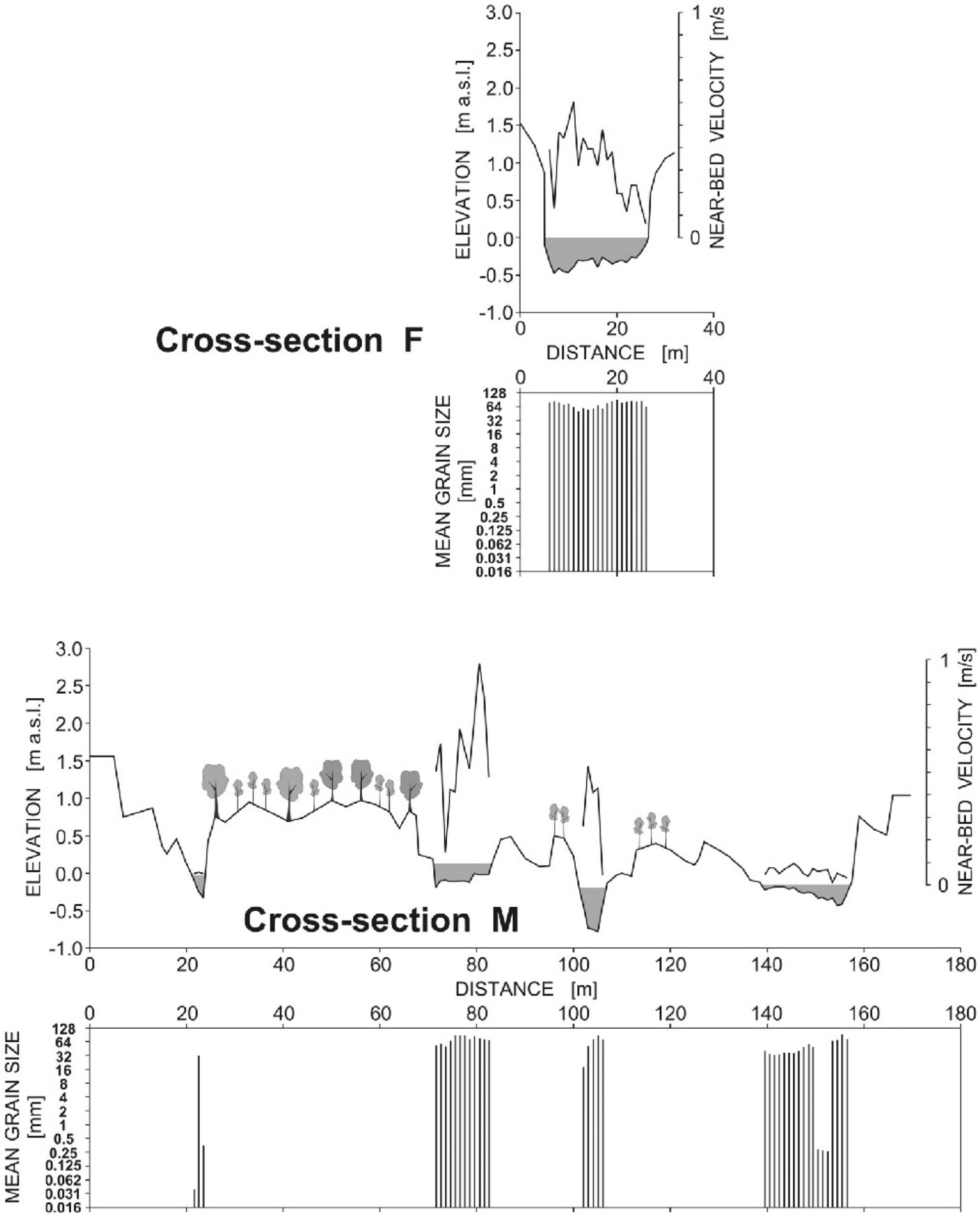

Figure 3. Examples of cross-sectional morphology of the Czarny Dunajec in (top) channelized river section and (bottom) an unmanaged section. For low-flow channels, mean grain size of surface bed material and near-bed flow velocity are indicated at 1 m intervals. The scale for velocity commences at the water surface for each low-flow channel.

Figure 4. View of the Czarny Dunajec River in single-thread cross section F and cross section M with four braids. Horizontal arrows indicate a location of the investigated cross sections, and vertical arrows point to particular low-flow channels in cross section M. Cross section F is viewed downstream, and cross section M in the upstream direction.

channel morphology is also found at cross section S located in the channelized river section at the downstream end of the study reach. Cross sections J to M, which were located in the central part of the unmanaged river stretch, had four or five flow threads and supported an abundant or moderate occurrence of wooded islands (Figures 3 and 4, cross section M). Finally, cross section I and cross sections N to R, located in the transition zones between the fully unmanaged channel in the center and the upstream and downstream channelized river stretches, respectively, had one riprap reinforced bank and were typified by a bar-braided morphology with two or three low-flow channels.

4.2. *Physical Parameters of Riverine Habitats*

Hydraulic parameters were measured at discharges varying between 3.42 and 3.86 $m^3 s^{-1}$ among the surveyed cross sections. No significant systematic change in the measured

Table 1. Results of the Linear Regression Analysis for Relationships Between Physical Characteristics of the Czarny Dunajec River and the Number of Flow Threads in the Investigated Cross Sections[a]

Dependent Variable	Beta Value	Goodness of Fit	Significance
Low-flow channel width	$\boldsymbol{B = 0.54}$	$\boldsymbol{R^2 = 0.29}$	$\boldsymbol{p = 0.02}$
Flow depth: mean	$B = -0.41$	$R^2 = 0.17$	$p = 0.09$
Flow depth: coefficient of variation	$\boldsymbol{B = 0.74}$	$\boldsymbol{R^2 = 0.55}$	$\boldsymbol{p = 0.0005}$
Depth-averaged velocity: mean	$B = -0.16$	$R^2 = 0.03$	$p = 0.53$
Depth-averaged velocity: coefficient of variation	$\boldsymbol{B = 0.49}$	$\boldsymbol{R^2 = 0.24}$	$\boldsymbol{p = 0.04}$
Near-bed velocity: mean	$B = 0.02$	$R^2 = 0.00$	$p = 0.94$
Near-bed velocity: coefficient of variation	$\boldsymbol{B = 0.49}$	$\boldsymbol{R^2 = 0.24}$	$\boldsymbol{p = 0.04}$
Grain size: mean	$\boldsymbol{B = -0.75}$	$\boldsymbol{R^2 = 0.56}$	$\boldsymbol{p = 0.0003}$
Grain size: coefficient of variation	$\boldsymbol{B = 0.80}$	$\boldsymbol{R^2 = 0.64}$	$\boldsymbol{p = 0.00008}$

[a]Relationships with p values <0.05 are indicated in bold.

discharge was found either with the distance of a cross section from the beginning of the study reach (linear regression, p = 0.30) or with the catchment area to a given cross section (p = 0.33). Owing to the specific hydrographic setting of the study reach (Figure 2), no downstream increase in discharge was thus registered, and the variation reflected runoff variability during the measurement campaign. This enabled us to consider the data from all cross sections as representing similar flow conditions and to analyze whether the physical parameters of riverine habitats were dependent on the complexity of the flow pattern.

No significant relations were identified between mean flow depth, as well as mean cross-sectional values of depth-averaged and near-bed velocity, and the number of low-flow channels in a cross section (Table 1). Increasing flow pattern complexity was, however, reflected in an increase in the aggregated width of low-flow channels (Figure 3 and Table 1); the latter increased by 4 m, on average, with every additional low-flow channel in a cross section (Figure 5). Moreover, mean grain size of surface bed material was found to decrease with increasing number of low-flow channels in a cross section, and the relationship was very highly significant (Table 1).

An increase in the flow pattern complexity was associated with increasing variation in flow depth (p = 0.0004), depth-averaged (p = 0.04) and near-bed flow velocity (p = 0.04), and mean grain size of surface bed material (p = 0.00008) in a cross section (Figure 5 and Table 1). As a consequence, single-thread cross sections and cross sections with a few braids differed markedly in the degree of variation of the physical characteristics of riverine habitats (Figure 3). In the former, the pattern of flow depth and velocity was relatively regular, and a gravel bed occurred across the whole channel width (Figure 3, cross section F). In the latter, braids with strong water current were accompanied by those with slow-flowing water, and thus, the prevalent gravelly parts of the bed were accompanied by areas covered with sand or mud (Figure 3, cross section M). Mean values of depth-averaged velocity and near-bed velocity were relatively strongly correlated among the surveyed cross sections (r = 0.81; p = 0.00004), and the coefficients of variation of both parameters showed a very strong correlation (r = 0.96; p = 0.000001), indicating that the parameter variability is highly similar within the cross sections.

Differences in the variation of physical characteristics of habitats in the Czarny Dunajec, which exist between single-

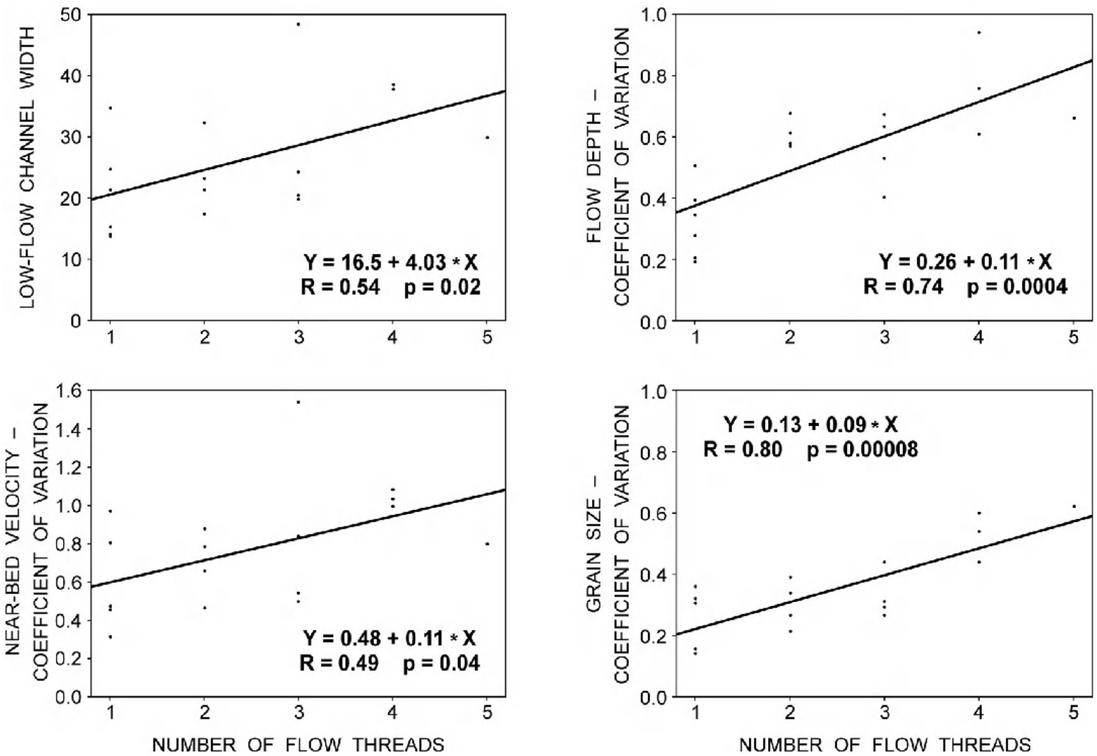

Figure 5. Scatterplots and estimated regression relationships between the aggregated width of low-flow channels and the coefficients of variation of flow depth, near-bed flow velocity, and mean grain size of surface bed material and the number of flow threads in the investigated cross sections of the Czarny Dunajec River.

thread cross sections and multithread cross sections with a larger number of low-flow channels, are well illustrated by scatter diagrams for the pairs of the parameters measured at 1 m intervals. First, in single-thread cross sections, flow velocity (both near-bed and depth-averaged) generally increased with increasing water depth (Figure 6). This reflected a progressive shift from slow-flowing water in shallow channel areas to faster water in deeper parts of the cross sections

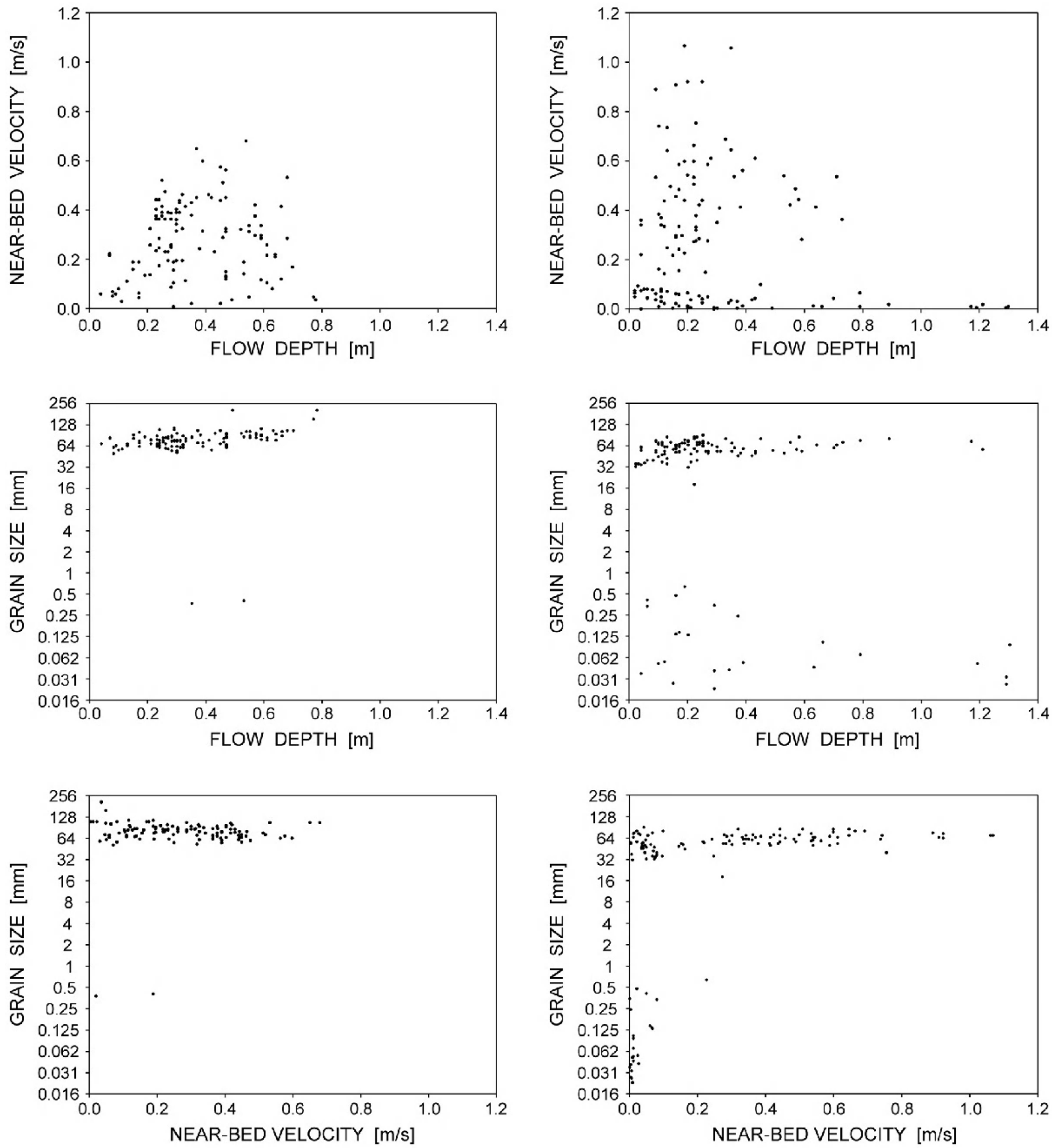

Figure 6. Scatterplots for the pairs of physical characteristics of the Czarny Dunajec measured at all sampling sites in the investigated cross sections of the Czarny Dunajec River: (left) six single-thread cross sections and (right) four multithread ones with the largest number of low-flow channels.

(Figure 3, cross section F). No relation between water depth and flow velocity was, however, observed in multithread cross sections (Figure 6). Here, very different near-bed velocities were associated with relatively shallow flows, depending on whether the measurements were made at the main braids conveying most flow or in lateral channels partly disconnected from the main flow (Figure 3, cross section M). At the same time, in some braids, low velocities were recorded in shallow channel areas where flow retardation due to bed roughness was greatest, while in others, low velocities occurred in deep pools because of backwater effects.

Second, in single-thread cross sections, grain size of the cobble bed was unrelated to measured base flow velocities (Figure 6), as the bed forms at substantially higher velocities during floods. At the same time, the shallow-water, low-velocity areas within the cross sections lacked fine bed sediments (Figure 6). This indicates that shallow-depth, slow-flow conditions are transient in these cross sections, and fine sediments, even if deposited on the bed at low to moderate flows, are readily and regularly flushed out from such sites. In multithread cross sections, two populations of bed material were identified: (1) pebble to cobble sediments, which predominated in the main braids conveying most flow, and (2) muddy-sandy sediments, which usually occurred in the lateral braids exhibiting slower flow but also occurred in the low-velocity areas within main braids (Figure 6). The grain size of the former apparently reflects depositional conditions during flood flows, while that of the latter is adjusted to the hydraulic conditions at low to moderate flows. With relatively long-lasting disconnection of the upstream end of some lateral braids from the main water current, fine sediment overlying gravelly material can persist and accumulate, at times attaining quite a considerable thickness.

The high heterogeneity of habitat conditions in multithread channel sections is illustrated by Figure 7, showing both the range and the average values of flow depth, near-bed velocity, and mean grain size of surface bed material for five low-flow channels of cross section L. The mean values of the parameters differed significantly among the low-flow channels, and the individual braids exhibited a distinct range of variation in particular parameters. As a result, each low-flow channel exhibited specific combinations of hydraulic and bed substrate conditions, suitable for different macroinvertebrate taxa.

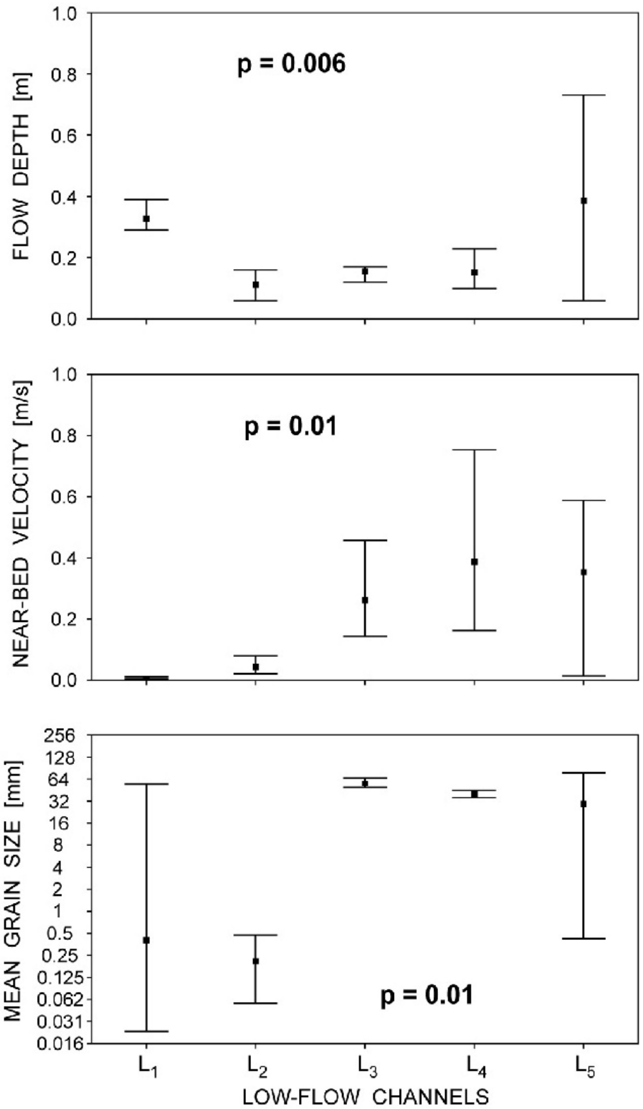

Figure 7. Range and mean of the flow depth, near-bed flow velocity, and mean grain size of surface bed material in particular low-flow channels of cross section L with five flow threads and the results of a Kruskal-Wallis test for the significance of difference of the parameter means among the low-flow channels.

4.3. Invertebrate Communities

Twenty-one invertebrate taxa were found in the investigated reach of the Czarny Dunajec, with four identified at the species level, fourteen at the genus level, and three at the family level (Table 2). No taxa were present in all surveyed cross sections; however, perlodid stoneflies (*Perlodes* sp.) were found in 17 cross sections, and two taxa, stoneflies (*Perla* sp.) and one species of spiny crawler mayflies (*Ephemerella ignita*), were found in 16 cross sections. No distinct shift in the taxonomic composition of invertebrate communities along the reach was observed, with the exception of the larvae of mites (family *Acari*), which were found only within one low-flow channel in cross section J (Table 2). Notably, taxa indicating good or high water quality occurred in all surveyed cross sections. In single-thread cross sections, two to six good/

Table 2. Invertebrate Taxa Recorded in Particular Low-Flow Channels of the Investigated Cross Sections of the Czarny Dunajec River as Well as the Total Number of Taxa Recorded in Particular Low-Flow Channels and the Investigated Cross Sections

Taxon	A	B_1	B_2	B_3	C	D_1	D_2	D_3	E_1	E_2	F	G	H	I_1	I_2	J_1	J_2	J_3	J_4	K_1	K_2	K_3	K_4
Crenobia alpina			x	x	x			x			x						x				x		
Dendrocoleum carpathicum				x				x							x								
Nematoda		x					x									x			x				
Lumbricidae		x					x							x		x			x				
***Perla* sp.**	x	x	x	x	x	x	x	x	x		x				x	x	x		x	x		x	
***Perlodes* sp.**	x	x	x	x		x	x	x	x	x	x	x	x	x	x	x	x	x		x	x	x	x
***Leuctra* sp.**		x		x				x			x				x		x	x		x			x
***Caenis* sp.**		x		x		x		x		x	x				x	x				x	x		
Ephemerella ignita		x	x	x	x	x	x	x		x	x	x	x		x		x	x		x		x	x
***Heptagenia* sp.**	x			x				x							x		x	x				x	x
***Agapetus* sp.**		x						x							x		x				x		
***Goera* sp.**		x		x							x				x					x		x	
Hydropsyche sp.	x		x	x	x	x		x	x								x						x
***Philopotamus* sp.**			x			x			x	x					x		x	x			x		
***Ryacophila* sp.**	x			x		x		x				x	x		x		x	x		x		x	x
Simulium sp.								x				x					x						
Tabanus sp.							x							x		x							
Tipula sp.							x							x		x			x				
Chironomus sp.							x							x		x			x				
Acari																		x					
Ancylus fluviatilis			x	x					x				x		x		x				x		x
Number of taxa in low-flow channel	5	9	7	12	4	7	8	12	5	4	7	4	4	5	12	8	12	7	5	7	6	6	7
Number of taxa in river cross section	5		16		4		18		7		7	4	4	16			19				13		

Taxon	L_1	L_2	L_3	L_4	L_5	M_1	M_2	M_3	M_4	N_1	N_2	O_1	O_2	O_3	P_1	P_2	P_3	R_1	R_2	S
Crenobia alpina			x			x							x				x		x	
Dendrocoleum carpathicum				x										x			x			
Nematoda	x				x				x	x					x					
Lumbricidae	x									x		x			x	x				
***Perla* sp.**			x			x	x	x			x	x	x	x	x	x	x		x	x
***Perlodes* sp.**			x	x		x	x	x			x	x	x		x		x		x	x
***Leuctra* sp.**			x				x	x		x		x	x			x		x		
***Caenis* sp.**			x					x								x	x		x	
Ephemerella ignita			x	x	x			x			x			x			x	x		
***Heptagenia* sp.**			x	x	x	x	x					x				x	x	x		
***Agapetus* sp.**					x			x						x			x		x	
***Goera* sp.**		x										x					x		x	
Hydropsyche sp.		x	x					x			x			x			x		x	
***Philopotamus* sp.**				x		x		x					x				x		x	x
***Ryacophila* sp.**		x		x				x					x	x			x		x	x
Simulium sp.						x													x	
Tabanus sp.					x				x	x		x						x		
Tipula sp.					x				x						x			x		
Chironomus sp.	x						x		x	x	x	x		x	x	x		x		
Acari																				
Ancylus fluviatilis				x				x					x						x	
Number of taxa in low-flow channel	3	3	8	7	6	6	5	10	4	5	5	8	7	7	6	6	12	6	11	4
Number of taxa in river cross section			19				17			9			16			17		17		4

[a]Bold indicates taxa of good and high water quality [cf. *Hawkes*, 1997; *Kownacki et al.*, 2002].

high water quality taxa were found, which represented 50% to 86% of all the taxa recorded in these cross sections, whereas multithread cross sections hosted between 4 and 10 indicator taxa with 44% to 77% of the assemblage composition (Table 2). Remarkably different numbers of indicator taxa were recorded in neighboring cross sections that contained a different number of flow threads; for instance, 10 such taxa occurred in cross section B with three low-flow channels, 2 taxa in single-thread cross section C, and 9 in cross section D with three braids (Table 2).

Taxonomic richness of invertebrate communities varied markedly within the study reach, with particular cross sections hosting anywhere from 4 to 19 taxa. In general, multithread cross sections exhibited greater diversity of invertebrates than single-thread cross sections. In single-thread cross sections, invertebrate communities comprised 4 to 7 taxa (4.7 on average), whereas 7 to 19 taxa (mean 15.3) were found in multithread cross sections, with the difference being statistically significant (Mann-Whitney test, p = 0.001). As the amount of sampling effort is known to exert an influence on recorded taxonomic richness of invertebrate assemblages [*Larsen and Herlihy*, 1998], two analyses were made to verify a possibility that the aforementioned difference reflected greater numbers of samples collected in the multithread cross sections rather than greater variability in habitat conditions typifying these cross sections in comparison with the single-thread cross sections. First, the numbers of taxa found in single-thread cross sections were compared with those recorded in individual braids of multithread cross sections. With equal sampling effort in both types of low-flow channels, 3 to12 taxa (7.1 on average) were recorded in the braids in comparison with between 4 and 7 taxa (mean 4.7) found in the single-thread cross sections, a statistically significant difference (Mann-Whitney test, p = 0.01). Of the 37 braids surveyed in the multithread cross sections, 33 hosted a greater number of taxa than the mean for the single-thread cross sections (Table 2). Second, 11 taxa were found in all surveyed single-thread cross sections, and this number was compared with those recorded in individual multithread cross sections. Although the aggregated number of samples collected in the single-thread cross sections (18) exceeded those taken in particular multithread cross sections (6 to 15), 10 of 12 multithread cross sections exhibited greater taxonomic richness of the invertebrate assemblages than in all single-thread cross sections (Table 2).

Because of the small number of invertebrate taxa hosted by individual single-thread cross sections, assemblages found in single-thread and multithread cross sections exhibited a low degree of similarity, as shown by the values of Jaccard's coefficient, which amounted to 28% on average (Table 3). Considerable differences in the taxonomic composition of invertebrate communities also existed between single-thread cross sections, with the average value of the coefficient for the pairs of such cross sections amounting to 27% (Table 3).

Table 3. Jaccard's Coefficients (in Percentage) of the Taxonomic Similarity Between Invertebrate Communities Recorded in Particular Cross Sections of the Czarny Dunajec River[a]

	A(1)	B(3)	C(1)	D(3)	E(2)	F(1)	G(1)	H(1)	I(2)	J(4)	K(4)	L(5)	M(4)	N(2)	O(3)	P(3)	R(2)	S(1)
A(1)																		
B(3)	31																	
C(1)	29	25																
D(3)	29	74	24															
E(2)	30	26	33	32														
F(1)	20	30	38	33	36													
G(1)	33	18	14	17	33	22												
H(1)	22	25	14	17	33	22	60											
I(2)	21	68	11	74	33	35	18	25										
J(4)	25	71	20	76	40	29	20	20	64									
K(4)	39	71	31	58	40	54	21	31	61	50								
L(5)	28	79	22	84	37	39	16	22	70	81	72							
M(4)	29	65	24	79	47	33	24	24	65	85	67	84						
N(2)	27	39	30	60	31	33	30	18	47	45	29	50	44					
O(3)	31	68	25	74	33	35	18	25	78	64	71	89	65	56				
P(3)	29	74	24	89	32	41	17	17	74	61	67	94	70	44	74			
R(2)	29	65	24	70	47	41	24	24	74	76	77	84	89	37	74	70		
S(1)	43	25	14	24	33	22	33	33	25	20	31	22	24	18	25	24	24	

[a]Figures in parentheses indicate the number of low-flow channels in a given cross section.

Table 4. Jaccard's Coefficients (in Percentage) of the Taxonomic Similarity Between Invertebrate Communities Recorded in Particular Low-Flow Channels of Cross Section L of the Czarny Dunajec River

	L_1	L_2	L_3	L_4	L_5
L_1					
L_2	0				
L_3	0	10			
L_4	0	11	25		
L_5	13	0	17	18	

Widely tolerant perlodid stoneflies (*Perlodes* sp.) and stoneflies (*Perla* sp.) were most commonly found in single-channel sections, the former being recorded in five and the latter in four investigated cross sections. In turn, completely absent in these sections were limnophilic taxa (*Nematoda*, *Lumbricidae*, *Chironomus* sp., *Tabanus* sp.), while surprisingly rare were rheophilic taxa such as larvae of *Crenobia alpina*, *Heptagenia* sp., and *Goera* sp., and river limpet *Ancylus fluviatilis*, with the first taxon recorded in two single-thread cross sections and the remaining ones in single such cross sections. Finally, of the five functional feeding groups of benthic invertebrates, only predators and shredders were found in each of the surveyed single-thread cross sections, and one additional group (either filter feeders or gatherers) was also represented in five of these cross sections.

As a markedly greater proportion of the reach-wide pool of invertebrate taxa were represented in multithread channel sections (Table 2), assemblages found here exhibited a relatively high degree of similarity, with the values of Jaccard's coefficient calculated for the pairs of multithread cross sections amounting to 63% on average (Table 3). Taxa from all five functional feeding groups of invertebrates (i.e., predators, gatherers, scrapers, shredders, and filter feeders) were represented in 11 of the 12 investigated multithread cross sections. In contrast to the high degree of taxonomic similarity among multithread cross sections, remarkable differences existed between invertebrate assemblages found within individual low-flow channels in a given cross section. They are illustrated by Jaccard's coefficients calculated for the pairs of low-flow channels of cross section L with five braids. The values of the coefficient ranged between 0% and 25% (Table 4), indicating that particular braids supported different invertebrate communities, with zero or only a small number of common taxa (Table 2). In multithread sections, most low-flow channels hosted rheophilic taxa associated with varying numbers of eurytopic taxa (B_3, D_3, I_2, J_2, K_2, L_3, L_4, M_1, O_2, P_3, and R_2), while some supported an occurrence of limnophilic taxa, such as horsehair worms (*Nematoda*), aquatic earthworms (*Lumbricidae*), midges (*Chironomus* sp.), or horse and deer flies (*Tabanus* sp.), in association with ecologically tolerant taxa (braids D_2, I_1, J_1, J_4, and P_1). Interestingly, there also occurred low-flow channels, which supported an occurrence of both limnophilic and rheophilic taxa (braids B_1, L_5, M_2, O_1, P_2, and R_1).

4.4. Relationships between the Diversity of Invertebrates and Explanatory Variables

The detailed set of physical parameters for the surveyed cross sections made it possible to determine whether they explain the observed variability in taxonomic diversity of benthic invertebrates among the cross sections. Results from simple regression analysis indicated that the number of invertebrate taxa increased linearly with decreasing cross-sectional averages of flow depth and grain size of surface bed material and with increasing variation of flow depth, velocity (both depth-averaged and near-bed), and bed material grain

Table 5. Results of the Linear Regression Analysis for Relationships Between the Number of Invertebrate Taxa Recorded in the Investigated Cross Sections of the Czarny Dunajec River and Physical Characteristics of the Cross Sections[a]

Independent variable	Beta Value	Goodness of Fit	Significance
Low-flow channel width	$B = 0.41$	$R^2 = 0.17$	$p = 0.09$
Flow depth: mean	$\boldsymbol{B = -0.56}$	$\boldsymbol{R^2 = 0.31}$	$\boldsymbol{p = 0.016}$
Flow depth: coefficient of variation	$\boldsymbol{B = 0.66}$	$\boldsymbol{R^2 = 0.44}$	$\boldsymbol{p = 0.003}$
Depth-averaged velocity: mean	$B = -0.08$	$R^2 = 0.01$	$p = 0.74$
Depth-averaged velocity: coefficient of variation	$\boldsymbol{B = 0.51}$	$\boldsymbol{R^2 = 0.26}$	$\boldsymbol{p = 0.03}$
Near-bed velocity: mean	$B = 0.13$	$R^2 = 0.02$	$p = 0.60$
Near-bed velocity: coefficient of variation	$\boldsymbol{B = 0.49}$	$\boldsymbol{R^2 = 0.24}$	$\boldsymbol{p = 0.04}$
Grain size: mean	$\boldsymbol{B = -0.71}$	$\boldsymbol{R^2 = 0.50}$	$\boldsymbol{p = 0.001}$
Grain size: coefficient of variation	$\boldsymbol{B = 0.58}$	$\boldsymbol{R^2 = 0.34}$	$\boldsymbol{p = 0.01}$
Number of flow threads	$\boldsymbol{B = 0.84}$	$\boldsymbol{R^2 = 0.71}$	$\boldsymbol{p = 0.00001}$

[a]Relationships with p values <0.05 are indicated in bold.

size. Other physical parameters under consideration exerted no significant effect on taxonomic richness of invertebrate communities (Table 5). However, the strongest and most significant relationship was with the number of flow threads in a cross section. The taxonomic richness of invertebrate communities increased as the number of low-flow channels increased (Table 5 and Figure 8), and this relationship explained 71% of the total variation in the number of invertebrate taxa among the surveyed cross sections.

Relationships between the number of invertebrate taxa and physical habitat parameters were further investigated by means of multiple regression analysis. A stepwise regression procedure was used, and particular variables were allowed to enter the model if they were significant at $p < 0.05$ in the final equation. The following equation was obtained:

$$\mathrm{NIT} = 16.2 + 3.4 \times \mathrm{NFT} - 29 \times \mathrm{DEPTH} - 0.4 \times \mathrm{WIDTH} + 10 \times \mathrm{VNBV} \quad (R^2 = 0.90; \quad p = 0.000003),$$

where NIT is the number of invertebrate taxa in a cross section, NFT is the number of flow threads, DEPTH is the cross-sectional average of flow depth (m), WIDTH is the aggregated width of low-flow channels (m), and VNBV is the coefficient of variation of near-bed velocity. The equation indicates that with other parameters being held constant, the number of invertebrate taxa in a cross section increased by more than 3 on average with each addition of a low-flow channel and by 1 with each 0.1 increase in the coefficient of variation of near-bed velocity; the number of invertebrate taxa decreased by 1 with each 2.5 m increase in the aggregated width of low-flow channels and by nearly 3 with each 0.1 m increase in mean flow depth. These four variables together explained 90% of the observed variation in the number of invertebrate taxa among the investigated cross sections. Mean grain size of surface bed material appeared redundant in the model, as it was significantly correlated with the number of flow threads in a cross section, as did coefficients of variation of flow depth and bed material grain size, which were correlated with the coefficient of variation of near-bed velocity.

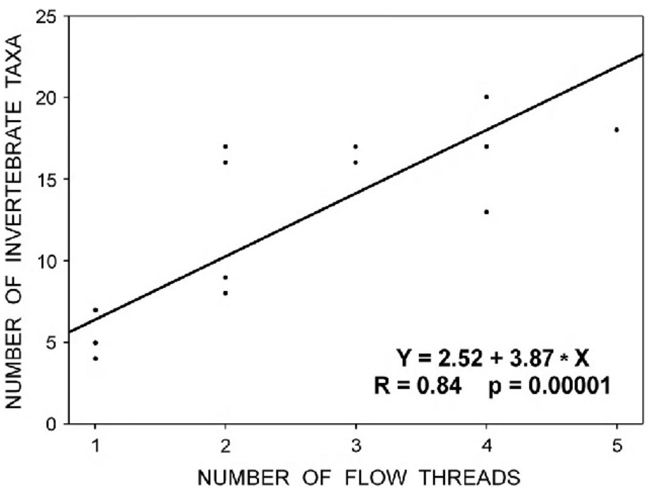

Figure 8. Scatterplot and estimated regression relationship between the number of invertebrate taxa recorded in the investigated cross sections of the Czarny Dunajec River and the number of flow threads in the cross sections.

5. DISCUSSION

The study performed in the mountainous Czarny Dunajec River indicated (1) gradients in physical conditions of aquatic habitats that accompany an increase in flow pattern complexity and (2) differences in hydromorphological conditions between channel sections heavily modified by channelization and gravel mining-induced channel incision (single-thread channels) and those which remained in a relatively undisturbed state (multithread channels). An increasing number of flow threads is associated with an increase in low-flow channel width and a decrease in bed material grain size. Multithread cross sections located in the lower part of the study reach exhibit markedly finer bed material than in single-thread cross sections in either the upper or lower reaches. However, this disparity reflects not only a downstream fining of bed material but also reflects differences in active channel width [*Wyżga et al.*, 2009b] that induce distinct differences in unit stream power during flood flows [*Wyżga and Zawiejska*, 2005; *Wyżga*, 2007]. During low to moderate flows, the difference in bed material grain size between the single-thread and multithread channel sections is enhanced by deposition of fine sediments on the bed of less active braids.

An increase in the flow pattern complexity is also associated with increasing cross-sectional variability in flow depth, depth-averaged and near-bed velocity, and mean grain size of surface bed material [cf. *Jähnig et al.*, 2008]. Single-thread cross sections exhibit a simple gradient from shallow, low-energy to deep, high-energy habitats. The multithread cross sections are typified by a markedly greater range of variation in the measured physical parameters than the single-thread cross sections and exhibit significant differences of physical habitat conditions not only between individual low-flow channels but also within some braids. In the multithread cross sections, various combinations of habitat conditions occur, such as slow velocities and fine bed substrate associated with both shallow and deep channel areas. Moreover, slow-velocity conditions persist relatively longer in some lateral braids, as evidenced by considerable thickness of fine-grained sediment covering the bed. This contrasts with a short-lived occurrence of such conditions in single-thread channels [cf. *Negishi et al.*, 2002], indicated by the lack of fine material on the bed, even in the pool cross sections investigated.

The study indicated that benthic invertebrate communities occurring in the surveyed cross sections derive from the same, reach-wide pool of invertebrate taxa, with its particular components recorded at various locations along the study reach. It is not surprising given the similar geomorphic, geological, and climatic characteristics [*Kukulak*, 1997] and the lack of major tributaries precluding downstream changes in hydrologic regime within the reach. The lack of a systematic downstream shift in the taxonomic composition of invertebrate communities is important because it enables the structure of particular assemblages to be interpreted as resulting from a sorting process through local environmental filters [cf. *Malmqvist*, 2002]. Although benthic invertebrates in mountain watercourses are quite sensitive to water pollution [*Kownacki et al.*, 2002; *Hering et al.*, 2006], the occurrence of taxa indicative of good/high water quality in all the surveyed cross sections, their considerable proportion in the structure of each local assemblage, and short distances between the neighboring cross sections supporting the occurrence of remarkably different numbers of indicator taxa provide evidence that a decrease in water quality was not a major limiting factor for benthic invertebrates in the Czarny Dunajec. Instead, differences in the taxonomic richness of invertebrate assemblages recorded within the studied reach of the Czarny Dunajec predominantly reflect variability in physical habitat complexity among the surveyed cross sections.

In single-thread cross sections, a low degree of physical habitat variability was associated with an occurrence of invertebrate assemblages with a small number of taxa, mostly eurytopic ones. The narrow range of the cross-sectional variation in substrate types and hydraulic conditions probably led to their colonization by only a portion of the reach-wide pool of invertebrate taxa. This primary selection of taxa seems to be indicated by the occurrence of only some functional groups of invertebrates in the single-thread cross sections, as functional diversity is easier to reconstruct with limited sampling effort than taxonomic richness [*Bady et al.*, 2005]. Narrow, channelized river sections are typified by a relatively high rate of velocity increase with increasing discharge [*Negishi et al.*, 2002], and this, together with a general absence/scarcity of flow refugia allowing invertebrates to escape the shear forces, might have resulted in further decreases in taxonomic richness of the communities in such sections during flow pulses [cf. *Negishi et al.*, 2002]. Although the existence of a hyporheic zone may provide refugia for benthic invertebrates during flow pulses, allowing subsequent recolonization of the streambed [*Brunke and Gosler*, 1997], this recovery mechanism is impossible in the single-thread incised sections of the Czarny Dunajec, where the bedrock is covered with only a thin veneer of coarse-grained alluvium [*Zawiejska and Wyżga*, 2010]. Field observations indicated a small snowmelt flood wave occurred during the winter months preceding the invertebrate sampling when repeated colonization of the streambed by insects was impossible. The impoverishment of primary communities during this or earlier flow pulses can explain the relatively small number of invertebrate taxa found in the surveyed cross sections and, most of all, little similarity in the taxonomic structure of invertebrate samples between individual single-thread cross sections, shown by low values of the Jaccard's coefficient.

Multithread cross sections, which exhibited greater variability in physical habitat conditions, supported all functional groups of invertebrates and a significantly greater number of taxa than single-thread cross sections. With their greater range of cross-sectional habitat conditions, and the high diversity of hydraulic units (i.e., patches of uniform flow and substrate [see *Thomson et al.*, 2001]) among individual low-flow channels and within some braids, the multithread channel sections could be colonized not only by eurytopic taxa but also by invertebrates that prefer either lentic or lotic environments. These diverse habitat conditions increase the chance for the taxa with unique ecological niches to avoid strong and opportunistic competitors [e.g., *Thorup*, 1966; *Protasow*, 1994]. Moreover, with a lower rate of velocity increase with increasing discharge in these wider channels [cf. *Leopold and Maddock*, 1953] and the relatively high persistence of slow-velocity conditions in lateral braids, it is easier for the invertebrates to find flow refugia and escape the shear forces, hence reducing potential decreases in the taxonomic richness during flow pulses. In the incised part of the study reach, where the low-flow channel beds are underlain by only a thin veneer of coarse-grained alluvium, the occurrence of midchannel bars may enable the invertebrates to hide in gravel bar interstices during flow increases.

The observed relations between the number of benthic invertebrate taxa in the Czarny Dunajec and physical conditions of the channel most likely manifest genuine links between channel morphology, habitat conditions, and riverine communities [*Smiley and Dibble*, 2005]. Data presented in this study evidenced the dependence of the taxonomic richness of benthic invertebrate communities in the river on the variability in water depth, flow velocity, and bed material grain size but did not confirm its positive relation with an aggregated width of low-flow channels. This shows that greater diversity of invertebrates recorded in the multithread channel sections results from greater habitat heterogeneity, not from the enlargement of habitat area in these sections. The crucial role of habitat heterogeneity in determining taxonomic richness of invertebrate assemblages was demonstrated by a significantly greater average number of taxa recorded in individual braids of multithread cross sections than in single-thread cross sections. However, it is no coincidence that the relationship with the number of low-flow channels in a river cross section explained a considerably greater portion of the total variation in the number of invertebrate taxa than did regression models developed for the variation in single physical parameters. Increased complexity of the flow pattern is associated with greater cross-sectional variability in many environmental parameters, such as water quality including the amount of dissolved oxygen [*Fernald et al.*, 2006], water temperature [*Arscott et al.*, 2001], retention of wood debris [*Gurnell et al.*, 2000; *Wyżga and Zawiejska*, 2005], fallen leaves, and fine organic matter, which together with the parameters measured in this study determine the heterogeneity of riverine habitats. This increased habitat heterogeneity is beneficial for a variety of invertebrate taxa and together with greater persistence of slow-water conditions and greater availability of flow refugia in the multithread channel sections explain the strong relationship that exists between the diversity of invertebrates and the number of flow threads in the river.

6. CONCLUDING REMARKS

This study has shown distinct differences in the degree of heterogeneity of riverine habitats and in the diversity of benthic invertebrate communities associated with differences in the complexity of the flow pattern in the mountain Czarny Dunajec River resulting from spatially varied human impacts. Single-thread cross sections modified by channelization and channel incision were typified by relatively low variability of habitat conditions and hosted invertebrate communities composed of a few, mostly eurytopic, taxa representing only some functional feeding groups. In contrast, where multithread channel pattern was preserved, the examined river cross sections exhibited high variability of habitat conditions, reflected in the occurrence of various combinations of flow depth, velocity, and bed material grain size. These cross sections supported the occurrence of more diverse invertebrate communities, with taxa typical of both lentic and lotic habitats and representing all functional groups of macroinvertebrates.

The identified relationships between the number of invertebrate taxa and the degree of riverine habitat complexity [cf. *Taniguchi and Tokeshi*, 2004] indicate that, apart from a decrease in water quality, simplification of the flow pattern and morphology of mountain rivers due to human impacts must be considered a very important reason for a degradation of their biocoenoses. While generally the outcomes from this study confirm those obtained from a previous work on the relations between the condition of fish communities in the Czarny Dunajec and hydromorphological river quality [*Wyżga et al.*, 2009a], an important difference in the response of the two groups of organisms to the human disturbances exists, emphasizing the utility of monitoring the condition of various groups of aquatic biota in the assessment of ecological integrity of watercourses [*Jungwirth et al.*, 2000; *Hering et al.*, 2006]. Incised channel cross sections, either with single-thread or multithread morphology, supported only two fish species in comparison with four species recorded in the unmanaged part of the study reach, with some fish species lost from the incised sections over the last three decades. Considerable bed material coarsening associated with channel incision most likely made fish spawning more difficult, whereas the construction of weirs in the channelized, middle part of the study reach prevented fish migration from the downstream sections, which have maintained high-quality habitat conditions [*Wyżga et al.*, 2009a]. In turn, the invertebrate communities from the incised, upper part of the study reach exhibited low diversity only in the single-thread cross sections, whereas both the taxonomic and functional diversity of the invertebrates found in the multithread cross sections was comparable to that recorded in the examined cross sections in the unmanaged river stretch. Apparently, habitat heterogeneity in the multithread, incised channel sections is sufficient to support diverse invertebrate communities, and even if some taxa may be temporarily removed from these sections by flood flows, aerial dispersal of emergent invertebrates in their adult stage of the life cycle [*Malmqvist*, 2002] enables repeated colonization of suitable habitats as long as the invertebrates are present in the reach pool of taxa.

Considerable impoverishment of fish [*Wyżga et al.*, 2009a] and invertebrate communities in the Czarny Dunajec arising from the simplification of flow pattern and the resultant homogenization of physical habitat conditions indicates that future recovery of these communities in this and other modified mountain rivers will require restoration of the morphological channel complexity. Importantly, results from the studies on the condition of both groups of biota enable formulation of complementary recommendations about necessary restoration procedures. Restoration of the lost lateral connectivity of riverine ecosystems in both channelized and incised river sections will be required to increase the heterogeneity of habitats and the availability of refugia for invertebrates and fish. In the incised channel sections, the reduction in flow velocity will be necessary to stimulate sediment accumulation on the channel bed and to reduce bed material grain size. This would restore a hyporheic zone and vertical connectivity in the sections, with invertebrate community benefits [*Boulton*, 2007], and would also reestablish suitable spawning conditions for fish. Finally, removing unnecessary weirs and the construction of fish passes in the channelized section will be essential for restoration of the longitudinal connectivity in the river. Thus, it is evident that substantial improvement of the ecological integrity of a river modified by channelization and channel incision cannot be achieved without restoration of 3-D connectivity of the riverine ecosystem [*Kondolf et al.*, 2006; *Jansson et al.*, 2007].

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Combining Field, Laboratory, and Three-Dimensional Numerical Modeling Approaches to Improve Our Understanding of Fish Habitat Restoration Schemes

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Despite a growing consensus that in-stream structures used in fish habitat restoration schemes should be nested within the larger catchment context, they may, nevertheless, provide rapid results, which are often required when fish habitat is urgently needed. However, the design, location, and placement of many of these in-stream structures are often based on very little scientific assessment. Of the various existing structures, deflectors (also called groins, abutments, vanes, or spur dikes) are reported to be the most common and the most successful in fish rehabilitation projects. Several field, laboratory, and numerical modeling studies have been conducted to improve our understanding of the complex flow and sediment dynamics around in-stream structures. The objectives of this chapter are first to summarize the current scientific knowledge on flow deflectors for stream restoration of fish habitat based on laboratory experiments, fieldwork, and numerical modeling and, second, to present findings from a research program based on the Nicolet River (Quebec) case study where several paired deflectors were installed in the 1990s to enhance fish habitat. Combining field, laboratory, and three-dimensional (3-D) numerical modeling approaches, this case study highlights the important feedback between the excavated pool morphology, complex 3-D flow field during high flow when structures are overtopped, and sediment transport. The larger particles falling in the excavated pool do not appear capable of exiting the pool even during floods. The design and position of the excavated pool do not appear appropriate in this case, which will likely hamper the long-term success of this enhancement project.

1. INTRODUCTION

The use of fish habitat restoration schemes, although often criticized [*Frissell and Nawa*, 1992; *Roper et al.*, 1997; *Thompson*, 2002a, 2006], is widespread in both North America and Europe [*Roni et al.*, 2008]. (Note that the term "stream restoration" is used loosely to designate almost any type of stream corridor manipulation [*Sear*, 1994; *Shields et al.*, 2003; *Wohl et al.*, 2005] even if it should, strictly speaking, be defined as the complete return of an ecosystem to a predisturbance state [*Cairns*, 1991; *National Research Council*, 1992]. True restoration of ecosystems is nearly impossible [*Wheaton et al.*, 2006], but it is a term that is now widely used and that should be

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understood in this chapter as encompassing rehabilitation and enhancement.) Reliance on this management strategy has been a response, in recent times, to the observation that fish are declining at alarming rates within freshwater systems [*Allan and Flecker*, 1993; *Richter et al.*, 1997; *Ricciardi and Rasmussen*, 1999; *Fausch et al.*, 2002; *Lake et al.*, 2007]. The need for restoration also reflects the overall poor state of rivers. For example, in the United States, only 2% of river kilometers are nonimpacted by human activities, and over 70% of riparian forests have been lost [*Palmer et al.*, 2007]. In many cases, the restoration of a river involves the placement of structures within the channel, such as large woody debris, boulder complexes, or deflector-type structures, the latter of which receive various names depending on the context (e.g., groins, abutments, vanes, and spur dikes). Although there is a growing consensus that these structures should be nested within the larger catchment context [*Wohl et al.*, 2005; *Wheaton et al.*, 2006; *Lake et al.*, 2007], they may nevertheless provide rapid results which are often required when fish habitat is urgently needed [*House*, 1996; *Wheaton et al.*, 2004; *Whiteway et al.*, 2010]. However, the designs of many in-stream structures have not been modified since the 1930s [*Thompson and Stull*, 2002], and their location and placement is often based on judgment with little or no analysis [*Downs and Kondolf*, 2002; *Lacey and Millar*, 2004]. Despite annual investments of over US $1 billion in aquatic habitat rehabilitation activities [*Bernhardt et al.*, 2005; *Wohl et al.*, 2005], there is still very little spent on monitoring or on evaluating these projects [*Kondolf and Micheli*, 1995; *Bernhardt et al.*, 2005; *Thompson*, 2006; *Brooks and Lake*, 2007; *Palmer et al.*, 2007]. Consequently, little feedback exists to improve the state of our knowledge, and no clear guidelines exist to inform structure design [*Biron et al.*, 2004, 2005]. This situation makes it very difficult for practitioners to install in-stream structures without relying on a trial and error approach [*Wheaton et al.*, 2004].

Of the various existing structures, deflectors are reported to be the most common and the most successful in fish rehabilitation projects [*Hunter*, 1991; *Brookes et al.*, 1996; *Mitchell et al.*, 1998; *Thompson*, 2002a; *Thompson and Stull*, 2002], although earlier evaluations of their success did not always present a strict statistical assessment and may have been overly enthusiastic [*Bond and Lake*, 2003; *Thompson*, 2006]. Used in isolation or in cross-channel pairs, deflectors can also have various angles, lengths, and heights [*Hey*, 1996]. The main purpose of these structures is to limit channel width and accelerate flow through the constricted section, thus causing local scouring [*Hey*, 1996]. However, in many cases, a pool needs to be excavated downstream from the structure in order to ensure that suitable habitat is immediately available (e.g., for sport fishing or to favor the survival and recruitment of threatened species). Again, very little guidance is provided to river managers to determine the best location and dimensions of these pools based on deflector design, river dynamics, sediment supply, and other factors. In general, our knowledge of the river morphodynamics caused by various restoration measures is still very poor [*Zhang et al.*, 2007].

The lack of instructions to assist deflector design is evidenced from a case study on the Nicolet River (Quebec, Canada). There, a fish rehabilitation project was initiated in 1993 by implementing four artificial riffles and three paired deflectors [*Genivar*, 1996]. The cost of each pair of wooden deflectors varied between CDN $35,000 and $40,000. The original design of these deflectors was modeled on projects at several stream restoration sites in New England and in Ontario, in which downstream oriented V deflectors were typical. However, notable differences between the Nicolet River and these other streams exist. First, the Nicolet River is much wider than most of the compared streams (bankfull width of about 35 m compared to less than 10 m in most observed cases). It also has a larger ratio between high and low discharge. Seasonal ice cover further adds to the complexity of determining a suitable design, particularly in terms of structure height. The main objective of the Nicolet project was to quickly create deep pools suitable for salmonid habitat (brook (*Salvelinus fontenalis*), brown (*Salmo trutta*) and rainbow (*Oncorhynchus mykiss*) trout). Thus, a 1.5 m deep pool was excavated downstream from each pair of deflectors. As this was not deemed sufficient to create high-quality habitat for trout, pool depths were increased to 2.5 m in 1996. However, after only a few more years, the wooden deflectors suffered serious damage (Figure 1), and the pools became partly filled. This prompted a complete change in the structures' design for reaches subsequently restored: boulder deflectors, oriented upstream (at 135°) instead of downstream, were used (Figure 2). These resilient structures were not only more aesthetically pleasing, but also less expensive (between CDN $15,000 and $25,000 per pair).

The trial and error approach that characterized the Nicolet River case prompted the development of a research program that would look at the impact of in-stream structures on fish habitat in a more scientific way. The hope was that future projects could benefit from a sound knowledge of how in-stream structures and, in particular, paired deflectors, affect flow dynamics, sediment transport, and scouring processes. This research program was formulated so as to combine laboratory experiments on various deflector designs [*Biron et al.*, 2004, 2005], fieldwork on flow dynamics, and sediment transport around existing structures [*Carré et al.*,

Figure 1. An example of the downstream-oriented (45°) paired wooden deflectors installed in the Nicolet River in 1993. (a) View looking upstream at high flow showing that the deflectors were placed in a sinuous part of the river (1999). (b) Side view at high flow illustrating the importance of the constriction ratio (1999). (c) Side view at low flow in 2001 showing damaged structure. (d) Side view in 2006 of the same structure that is no longer maintained.

2007], as well as three-dimensional (3-D) numerical modeling of the complex flow surrounding both field and laboratory deflectors [*Carré et al.*, 2006; *Haltigin et al.*, 2007a, 2007b]. The purpose of assembling such a comprehensive data set was to improve our understanding of river processes around structures and provide useful guidelines for stream managers.

The objectives of this chapter are to (1) summarize the current scientific knowledge on flow deflectors for stream restoration of fish habitat based on laboratory experiments,

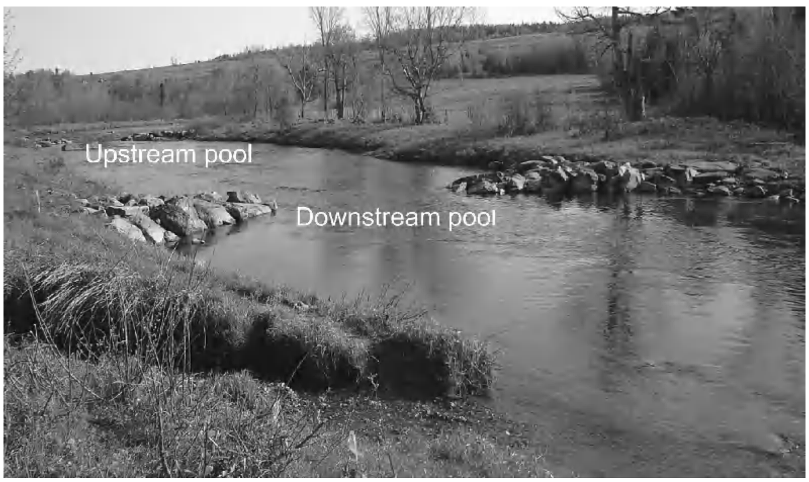

Figure 2. Upstream-oriented (135°) paired boulder deflectors used in the Nicolet River installed in 1997 after the unsuccessful installation of the 45° wooden deflectors located farther upstream. The view is looking upstream, where the upstream paired deflectors are visible.

fieldwork, and numerical modeling and (2) present the most recent findings from the research program based on the Nicolet River paired deflector case study.

2. THE SCIENCE OF STREAM RESTORATION: GEOMORPHOLOGICAL IMPACTS OF IN-STREAM STRUCTURES

2.1. Field Studies

Some geomorphological field studies have been carried out to assess the long-term impact of restoration projects involving in-stream structures, but very few have focused on the hydrodynamic processes taking place around these structures. For example, *Thompson* [2002b] conducted research on the Blackledge and Salmon rivers (Connecticut), restored in the 1930s using a series of grade-control structures, riprap revetments and deflectors, to determine the current state of the structures. Bed topography measurements were taken to assess pool depth upstream and downstream from the structures, in addition to channel width. This research highlighted the long-term problems that can occur when installing structures that have a 20 year projected lifespan [*Frissell and Nawa*, 1992; *Thompson*, 2002b]. After 40 to 60 years, several of these structures had become ineffective and, in the worst cases, detrimental to channel morphology and fish habitat quality. Similar findings were obtained by *Champoux et al.* [2003] in Wisconsin, where most of the structures in high-energy reaches were no longer efficient after 33 years, although many in the lower-gradient zone were still in good condition. There is clearly some uncertainty in the use of in-stream structures in the long term, which is a problem for restoration designers when they need to plan for a project life beyond 20 years [*Niezgoda and Johnson*, 2006]. At a much shorter time scale (1–2 years after restoration measures were initiated), *Shields et al.* [1995] observed a fivefold increase in pool-habitat availability due to enlargement of scour holes near deflectors (groins), though deflector extensions had failed partly due to sand erosion under the structures.

It is increasingly believed that spatially detailed descriptions of the flow field are required to correctly assess fish habitat [*Shields et al.*, 1995; *Shields and Rigby*, 2005]. For example, the Physical Habitat Simulation System (PHABSIM) is based not only on field measurements of channel shape, water depth, and substrate but also on flow velocities [*Bovee*, 1982; *Maddock*, 1999]. Several studies have collected these measurements at the scale of transects, reaches, or segments in order to delineate suitable habitats. From an ecological perspective, these scales are appropriate. However, even the cross-sectional scale is not sufficient to provide an understanding of the impact of in-stream structures on flow dynamics. Most studies focusing on these impacts come from the engineering literature, where field studies are the exception rather than the rule. As seen in the numerical modeling section below, detailed flow information can now be obtained from numerical simulation models, but field data are still needed to calibrate and validate these models. For example, *Lacey and Millar* [2004] took 1-D velocity measurements at 70 random points throughout their study reach to validate a 2-D model. However, most numerical models have been validated using laboratory rather than field data, and few field studies provide the detailed bathymetric and 3-D velocity data that are required for this purpose, in particular around in-stream structures.

Velocity data are laborious to collect when using point measurements such as those obtained with acoustic Doppler velocimeters (ADV) [e.g., *Daniels and Rhoads*, 2003, 2004; *Rhoads et al.*, 2003]. Yet *Engelhardt et al.* [2004] used an ADV to obtain 3-D velocity data in a groin field on the Elbe River (Germany). The contribution of their work was significant, as it showed the importance of large-scale vortices originating from groin heads in shaping phytoplankton dynamics. Meanwhile, *Carré et al.* [2007] quantified, with an ADV, the marked increases in bed shear stress that occur between paired deflectors. Methods such as the acoustic Doppler current profiler (ADCP) also show promise, as measurements can be obtained simultaneously along a vertical profile [*Shields and Rigby*, 2005]. However, ADCP is limited to relatively deep water environments (>1 m). It also has a fairly large blanking distance (0.2 to 0.5 m) that prevents measurements from being taken near the water surface. Moreover, the sampling volume increases with distance from the probe so it can be quite large near the bed [*Shields and Rigby*, 2005]. This is problematic, given the great importance of near-bed flow field in characterizing fish habitat [*Crowder and Diplas*, 2006; *Shen and Diplas*, 2008]. The ADCP also produces a spatial averaging problem, making it difficult to adequately estimate bed shear stress, a variable that is needed to determine bed load transport. By comparison, pulse-coherent acoustic Doppler profilers (PC-ADP) can be used in shallower water environments, but they also suffer from the spatial averaging problem near the bed [*Tilston and Biron*, 2006]. Another promising field technique is large-scale particle image velocimetry (LSPIV), which provides simultaneous surface planform velocity data [*Creutin et al.*, 2003; *Jodeau et al.*, 2008]. Unlike laboratory particle image velocimetry, it is limited to 2-D measurements and can only be used at the water surface. LSPIV at the Nicolet River was successful in revealing the very complex pattern in the recirculation zone during low-flow conditions [*Carré et al.*, 2006]. It has also been used by

Muto et al. [2002] in a river to measure the free surface flow around a groin field. Measurements of high flow are needed, however, to improve our understanding of the flow dynamics when deflectors are submerged. High-flow LSPIV values were obtained in a river reach by *Jodeau et al.* [2008], but, as will be described below, we were less successful in using this approach next to deflector structures in the Nicolet River.

2.2. Laboratory Experiments

Deflector-type structures have been studied extensively in laboratory environments, mainly in the field of engineering, since scouring poses a risk to the stability of these structures [*Ahmed and Rajaratnam*, 1998; *Ali and Karim*, 2002]. Many of these studies have focused on developing equations to predict maximum scour depth [*Melville*, 1997; *Kuhnle et al.*, 1999, 2002; *Fael et al.*, 2006]. However, not many have examined flow dynamics and scouring processes around these structures within a fish habitat restoration framework, where the objective is to maximize, and not minimize, scour zones [*Kuhnle et al.*, 1999, 2002].

In a test on deflector orientation, *Kuhnle et al.* [2002] observed that deflectors angled at 135° (i.e., oriented upstream) generate the largest scour volume and yet minimize bank erosion (and thus channel instability). Thus, the authors recommended this model as optimal for restoration schemes. By comparison, in a study conducted by *Biron et al.* [2004], the deflectors oriented at 90° were found to produce a larger volume of scour than those oriented at 45° or 135°. The discrepancies between these two studies as to which model results in the largest pools may be due to differences in the shape of the deflectors tested, a vertical plate [*Biron et al.*, 2004] versus a concrete block with a trapezoidal crest [*Kuhnle et al.*, 2002], though they could also be related to differences in structure length. A numerical simulation of flow around paired deflectors revealed that the maximum dynamic pressure is greatest for 90° deflectors when the structures are short (contraction ratio: length of structures/width of channel <0.2), but that pressure values are greatest and nearly equal for both 90° and 135° deflectors for larger contraction ratios [*Haltigin et al.*, 2007a]. In general, laboratory studies have indicated that an increase in structure length (or contraction ratio) results in greater values of velocity amplification at the obstruction nose, leading to greater scour volumes and depth [*Molinas et al.*, 1998; *Biron et al.*, 2004, 2005]. However, for longer structures (the length of which is greater than the flow depth), scour depth becomes independent from the length parameter [*Melville*, 1992, 1997]. Clearly, more research is required before specific guidelines can be provided to practitioners on the appropriate length and angle of structures for a given restoration project.

In order to understand the development of scour near instream structures, it is essential to gain insight into the complexity of the flow-scour interactions [*Kondolf*, 1998; *Ahmed and Rajaratnam*, 2000; *Shamloo et al.*, 2001; *Ali and Karim*, 2002; *Chrisohoides et al.*, 2003; *Biron et al.*, 2005; *Haltigin et al.*, 2007a; *Koken and Constantinescu*, 2008a]. It is generally acknowledged that in-stream structures such as deflectors cause a very complex 3-D, highly turbulent flow field near the structure due to the separation of the incoming flow and the complex vortex systems that result [*Chrisohoides et al.*, 2003; *McCoy et al.*, 2007; *Koken and Constantinescu*, 2008a]. These vortices in turn affect bed shear stress values, which need to be properly quantified for accurate scour predictions. Laboratory studies measuring bed shear stress around deflectors have found that the maximum value occurred at the structure's tip [*Rajaratnam and Nwachukwu*, 1983; *Molinas et al.*, 1998]. For various contraction ratios, this maximum was observed to be three to five times larger than the shear stress in the approach flow [*Rajaratnam and Nwachukwu*, 1983; *Ahmed and Rajaratnam*, 2000]. In contrast, *Molinas et al.* [1998] reported a maximum amplification of bed shear stress around the structure to be around 10 times the bed shear stress value of the approach flow, while *Biron et al.* [2005] observed it to be 15 times.

In nature, overtopping of the structures by the flow occurs very frequently. It is thus important to examine the role of deflector height on scouring, as overtopping conditions can significantly alter the characteristics of the vortices [*Kuhnle et al.*, 1999, 2002; *Shamloo et al.*, 2001; *Thompson*, 2002a; *Biron et al.*, 2004, 2005; *McCoy et al.*, 2007]. Low deflectors, even if they constrict the flow markedly, may not be able to create adequate pool habitat [*Thompson*, 2002a]. In general, structures built too high are believed to lead to an unnecessary high-flow resistance during floods, whereas those that are too low poorly confine the flow at normal water levels [*Uijttewaal*, 2005; *McCoy et al.*, 2007]. The flow field around submerged structures is far more complicated, with intensified three-dimensionality of the flow, than that around emergent structures [*Uijttewaal*, 2005; *McCoy et al.*, 2007]. As will be seen below, 3-D numerical models provide particularly useful information for these cases, since detailed experimental measurements of the whole flow field, and particularly of bed shear stress (which is needed to understand scouring processes), are difficult to obtain even with modern equipment such as particle image velocimetry (PIV) [*McCoy et al.*, 2007]. Indeed, the relationship between the overtopping ratio (flow depth/height of structure) and scour depth or volume is rather complicated when examined for various approach flow intensities [*Kuhnle et al.*, 2002] and requires more attention. For example, the idea that deflectors become drowned out and ineffective as they become overtopped [e.g., *Thompson*, 2002a] may be too simplistic [*McCoy et al.*, 2007]. The impact of deflector roughness also needs to be investigated, as the overwhelming majority of studies so far have used smooth obstructions, while natural deflectors, which are frequently overtopped, are always rough (independent of the construction material).

In addition to the lack of knowledge previously mentioned, not many studies have investigated the flow field past the initial stages of the scouring process [*Koken and Constantinescu*, 2008b]. The flow dynamics in mobile bed experiments with a developed scour is very different from that over a flat, smooth bed [*Biron et al.*, 2005]. The presence of a primary necklace vortex inside the scour hole seems to play a key role in scour development [*Kwan and Melville*, 1994; *Dey and Barbhuiya*, 2005, 2006a, 2006b]. However, our understanding of the complex vortex dynamics that take place near in-stream structures, which is necessary if we want to improve scour predictions and restoration designs [*Koken and Constantinescu*, 2008a], is limited when measurements are obtained using an intrusive device such as the ADV. Only PIV methods seem capable of providing the required detailed flow field measurements, but even PIV studies are restricted if, as is mostly the case in laboratory experiments, they remain in 2-D; 3-D PIV would be the ideal tool to investigate these flows, but they have not yet been used around deflector-like structures [*Koken and Constantinescu*, 2008a]. In fact, laboratory measurements around deflector-like structures are often limited to surface measurements using LSPIV because of the difficulties in obtaining 2-D PIV from the side for high width-to-depth ratio situations, which are required for deflector tests in a laboratory setting [*Koken and Constantinescu*, 2008a]. However, LSPIV offers the advantage of covering a considerably larger field of view than the

Table 1. Factors Affecting the Performance of Flow Deflectors

Factor	Impact/Importance	Known Issues
Orientation	Deflector angle affects maximum scour depth and risk of bank erosion; depth of scour generally increases as angle increases from 15° to 90°.	Angle generating maximum scour not consistent between studies (e.g., 135° [*Kuhnle et al.*, 2002], 90° [*Biron et al.*, 2004].
Length (contraction ratio)	Larger contraction ratios generate larger velocity amplification and larger scour.	Length of structures may explain discrepancies in the role of deflector angles [*Haltigin et al.*, 2007a, 2007b]. Scour depth does not appear to be affected when length exceeds depth of flow [*Melville*, 1992, 1997].
Height (overtopping ratio)[a]	Overtopping conditions generate more complex three-dimensional flow field; less scour when flow overtops structures.	No clear relationship between overtopping ratio and scour depth [*Kuhnle et al.*, 2002]. Confounding effect of increasing turbulent shear stress during high flow (overtopping) conditions [*Uijttewaal*, 2005; *McCoy et al.*, 2007].
Incoming flow[a]	Ratio of shear stress over critical shear stress in the incoming flow affects amplification of bed shear stress near structures and maximum scour depth.	Shear stress amplification ratios vary markedly between studies (from 3–5 [*Rajaratnam and Nwachukwu*, 1983; *Ahmed and Rajaratnam*, 2000] to 15 [*Biron et al.*, 2005]. Most studies on overtopping impact use constant incoming flow, which does not replicate the natural situation of higher flow stage corresponding to larger incoming shear stress values.
Mobility of the bed[a]	Interactions between bed morphology and flow dynamics through primary vortex in scour; reduced shear stress amplification over a mobile bed.	Different flow velocity and bed shear stress patterns over flat bed and mobile bed [e.g., *Biron et al.*, 2005]. Feedback between deflector dynamics and scour morphology need to be further assessed.
Roughness of structure[a]	Increased turbulence around rough structures.	Very few laboratory studies with rough structures and impact on vortex dynamics and scouring not clear.
Heterogeneity of the bed	Increased turbulence near the bed; impacts on scour of possible armoring effects for coarser particles.	Smooth beds or uniform-sand beds typically used in laboratory studies. Thus, this impact is poorly understood.

[a]Addressed in this study.

conventional PIV method, which is important when investigating the flow field in the wake region of structures [*Koken and Constantinescu*, 2008a].

Overall, there are few studies measuring sediment transport around in-stream structures, despite the importance of this variable in the design of stream restoration schemes in mobile bed channels [*Shields et al.*, 2003, 2004; *Bhuiyan et al.*, 2007]. Furthermore, one very serious limitation of laboratory studies is that they exclusively deal with sand-bed channels, whereas many, if not most, restoration schemes for fish habitat are in coarser-particle rivers, where armoring of the surface layer plays a fundamental role in determining whether or not particles will move for a given shear stress [*Laronne and Carson*, 1976; *Brayshaw et al.*, 1983; *Parker and Sutherland*, 1990; *Vericat et al.*, 2006]. Assessing sediment transport around in-stream structures in nature is further complicated by the high variability of flow discharge. The impact of increasing both water stage and velocities around deflectors is not clear because many laboratory experiments have used constant flow intensity conditions for various overtopping ratios [e.g., *Kuhnle et al.*, 1999; *Biron et al.*, 2004, 2005]. Laboratory experiments on the role of deflector length, height, and orientation have yielded useful information, although our understanding of the effect of these variables on flow dynamics is still incomplete [*Thompson*, 2002a; *Biron et al.*, 2004]. Insight into the interaction between flow, scour, and deflector shape and geometry has also improved via lab-based investigations using ADV, ADCP, PIV, and LSPIV [*Biron et al.*, 2005; *Uijttewaal*, 2005; *Koken and Constantinescu*, 2008a]. However, in order to provide useful results for practitioners in stream restoration projects and to improve further our scientific knowledge of the impact of in-stream structures, future laboratory studies need to replicate field conditions more closely. Table 1 summarizes the key factors affecting deflector performance and known issues that need to be addressed.

2.3. Numerical Modeling

Numerical models in one dimension, two dimensions, and, more recently, three dimensions have been used heavily in the field of stream restoration [*Niezgoda and Johnson*, 2006]. The simplest 1-D models (e.g., PHABSIM [*Bovee*, 1982]), still in widespread use, provide width- and depth-averaged velocity values at several cross-sections. At the reach scale, these models can generate estimates of physical habitat quality for a wide range of flows [*Federal Interagency Stream Restoration Working Group*, 1998; *Spence and Hickley*, 2000; *Booker and Dunbar*, 2004; *Moir et al.*, 2005; *Niezgoda and Johnson*, 2006]. Nevertheless, many consider them to be inadequate descriptors of physical habitat; they cannot provide information on spatial velocity gradients which are important for fish [*Crowder and Diplas*, 2000a; *Shields and Rigby*, 2005]. Thus, 2-D numerical models, which though also depth-averaged, do take the lateral velocity variability into account, have been advocated as being superior for assessing sediment transport, hydrogeomorphic processes, and, consequently, local fish habitat [*Crowder and Diplas*, 2000a, 2000b, 2002; *Gibbins et al.*, 2002; *Lacey and Millar*, 2004; *Clifford et al.*, 2008].

Lacey and Millar [2004] successfully used a 2-D model to simulate the mean flow field around various in-stream structures. Yet, it is becoming more and more apparent that 3-D models, which incorporate longitudinal, lateral, and vertical variability, are necessary for characterizing the quality of habitat at the small scale and mesoscale, whether around simple boulders [*Crowder and Diplas*, 2002, 2006; *Shen and Diplas*, 2008] or around deflector-like structures [*McCoy et al.*, 2007, 2008]. For example, vortices shed from a cylinder can be exploited by trout to reduce their swimming energy cost [*Liao et al.*, 2003], but this can only be quantified by a 3-D model [*Shen and Diplas*, 2008]. Much of the literature indicates that higher levels of spatial habitat heterogeneity, as generated by spatial gradients associated with eddies, wakes, and transverse flows, support higher levels of biodiversity [e.g., *Gorman and Karr*, 1978; *Shields and Rigby*, 2005; *Schwartz and Herricks*, 2008]. Thus, a scientific assessment of in-stream structures, installed for the purpose of restoring biological productivity in streams, should be based on a 3-D understanding of the flow field. Caution also warrants the use of 3-D models, as environments previously believed to be 2-D, such as shallow embayments in groin fields, were later observed to have significant 3-D flow components [*McCoy et al.*, 2008].

Even if 3-D models are known to provide a more detailed and accurate representation of stream physics [*Lane et al.*, 1999; *Clifford et al.*, 2005; *McCoy et al.*, 2007], several researchers have been reluctant to use them. These models are much more complex to operate and, despite evidence from fundamental research, they are not necessarily perceived as relevant for many important geomorphic and ecological functions critical to environmental management [*Pasternack et al.*, 2008]. Furthermore, calibration and validation of 3D models for large reaches remain unrealistic, and their computational demands are still prohibitive for most problems in management of stream ecosystems [*Pasternack et al.*, 2006; *Papanicolaou et al.*, 2008; *Brown and Pasternack*, 2009]. In fact, to work around the needs for 3-D models, several studies have coupled 2-D hydrodynamic models with sediment transport equations. Hence, they were able to predict scour potential, create local habitat suitability curves, and determine physical habitat quality for various organisms [*Leclerc et al.*, 1995; *Hardy*, 1998; *Pasternack et al.*, 2004; *Wheaton et al.*, 2004; *Gard*, 2006]. Meanwhile, one of the rare successful attempts to use a 3-D model in this context is that of *Nagata et al.* [2005] who used a time-accurate Reynolds-averaged Navier Stokes (RANS) solver with a nonlinear k-ε closure and wall functions to predict the scour evolution around an isolated 90° deflector. The relatively poor performance of currently available sediment transport formulae remains a problem when simulating scouring processes around structures, but adopting a 3-D rather than 2-D modeling approach should at least improve the accuracy of bed shear stress estimates [*Pasternack et al.*, 2006; *McCoy et al.*, 2007; *Shen and Diplas*, 2008]. It is thus important that researchers in this field demonstrate to practitioners the usefulness of adopting a 3-D numerical modeling approach to help improve the success rate of restoration projects by predicting scour evolution following the installation of in-stream structures.

Not only is it apparent that 3-D models can contribute to our understanding of the complex flow field around structures, but it is also becoming increasingly clear that these models should examine unsteady rather than time-averaged simulations. Indeed, flow separation, strong interactions between eddies in the mixing layers around structures and the main flow field, vortex shedding, and strong adverse pressure gradients require full 3-D nonhydrostatic simulations with eddy-resolving techniques such as large eddy simulation (LES) to accurately describe the complex flow and the dynamics of the large-scale turbulence around deflector-like structures, especially when they are submerged [*McCoy et al.*, 2007; *Koken and Constantinescu*, 2008a]. *Uijttewaal and van Schijndel* [2004] have compared the standard (time-averaged) k-ε model with a horizontal (unsteady) LES (HLES) simulation running in 2-D (depth-averaged Delft-3D) and found that the predictions for the HLES model were improved compared to the k-ε model. However, the agreement for the submerged cases was not as good as for the emergent case, highlighting the need to use 3-D LES instead of 2-D LES [*McCoy et al.*, 2007]. Furthermore, RANS models cannot take into account the role played by large coherent structures in the scouring mechanisms, since they require velocity fluctuations to be simulated [*Koken and Constantinescu*, 2008a, 2008b]. One interesting finding of the 3-D LES studies is that the coherence, structure, position and shape of the horseshoe vortex system are highly variable in time. This system is very important as it induces high turbulent kinetic energy, and thus high shear stress, ultimately responsible for scouring [*Koken and Constantinescu*, 2008a].

It is more and more common to see 3-D numerical modeling used in river restoration projects [*Carré et al.*, 2006; *Clifford et al.*, 2008; *Rhoads et al.*, 2008]. However, field data for model calibration and validation remain essential [*Clifford et al.*, 2008], and this needs to be addressed in future research.

3. NICOLET CASE STUDY

The Nicolet River is located approximately 200 km east of Montreal (Quebec) and drains into the St. Lawrence River (Figure 3a). Due to human activity, primarily land use changes, the section chosen for restoration had become degraded, and the trout population had dwindled. The primary problems identified by the Corporation de Gestion des

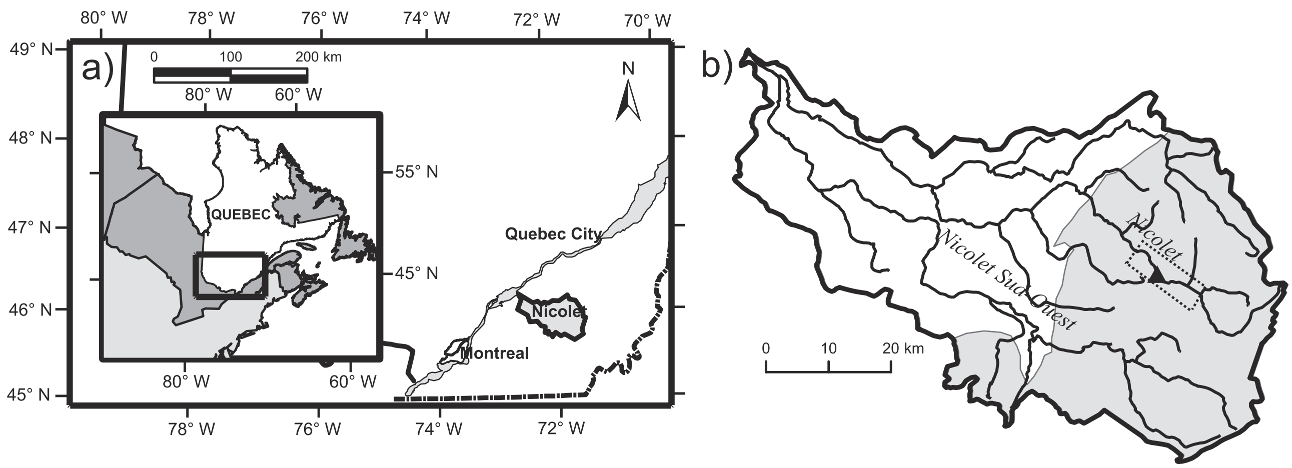

Figure 3. (a) Location map of the Nicolet watershed. (b) Location of the 14 km restored reach (dotted rectangle) and of the study reach with paired deflectors (black triangle). The gray area represents the Appalachian part of the watershed, whereas the white zone is located in the St. Lawrence Lowlands.

Rivières des Bois-Francs (CGRBF), a watershed agency, before restoration activities were commenced were the lack of pools and cover for trout as well as elevated summer water temperatures [*Corporation de Gestion des Rivières des Bois-Francs*, 1993]. The restoration plan included bank stabilization, fish shelter construction, tree planting, in-stream structure construction, and fish stocking. The in-stream structures consisted of paired deflectors, single deflectors, and weirs, all of which were designed to increase pool depth and volume [*Whiteway*, 2009].

Restoration structures were implemented gradually from 1993 to 1999 in a 14 km reach in the Appalachian headwater section of the watershed (Figure 3b). In total, 17 deflector structures were installed, including the two failed pairs of wooden deflectors (Figure 1), seven pairs of boulder deflectors, seven single boulder deflectors and one triple boulder deflector. The drainage area of the restored reach is 332 km^2, with a bankfull width of about 35 m, a bankfull discharge of 95 $m^3\ s^{-1}$ and an average bed slope of 0.0015. While one of the recommendations for the design of current deflectors is to avoid rivers with vast flow fluctuations [*Wesche*, 1985; *Brookes et al.*, 1996], the Nicolet River exhibits a 30-fold fluctuation in discharge, which decreases the project's likelihood of success. Restored reaches consist mainly of gravel beds, with some sand areas (average D_{50} and D_{84} of 39 and 70 mm, respectively). Since one of the objectives of this restoration project was to attract anglers to this site, fish stocking (brook, brown, and rainbow trout) is performed weekly between May and September each year. On average, 200 adult trout are stocked each week: brook and brown trout are stocked early in the season, while rainbow trout are stocked later as they have a higher maximum temperature threshold.

The first research project undertaken at this site focused on field measurements of bed topography and grain size. However, it soon became apparent that laboratory experiments and numerical modeling were needed to help understand the impact of these structures on the flow field and scouring processes. The advantage of both laboratory and modeling studies is the possibility to test different structure designs.

There is agreement that knowledge of the supply, transport, and storage of sediments in rivers and their watersheds is required for their sustainable management, and that this is highly relevant to channel design and maintenance [*Newson*, 1993; *Sear*, 1996]. Ideally, river restoration design includes information on the sediment load entering the reach and its variability over time, as well as on the routing of this load through the reach [*Sear*, 1996]. The latter is seldom analyzed, so it was decided to include sediment transport measurements in the research program on the Nicolet River. However, monitoring sediment transport rates directly is known to be a complex task for bed load [*Sear*, 1996]. The morphological method, which consists in executing repeated topographic surveys to calculate volumes of erosion and deposition within a reach, can be used [*Ashmore and Church*, 1998]. The volumetric changes, if combined with an average transport length of particles, can be converted into a sediment transport rate or load [*Sear*, 1996]. However, a minimum level of detection for particle movement must be used to distinguish geomorphic changes from survey noise [*Wheaton et al.*, 2010], with displacements of at least 0.1 m recommended in coarse-bedded rivers [*Brasington et al.*, 2000]. One method for measuring particle transport length is by means of tracer particles. These particles are marked so that they can be located and followed through time after they have been introduced to a stream. Marking techniques vary from simple bright paint to more advanced technologies such as passive integrated transponder (PIT) tags and active radio transmitters. *Hassan and Ergenzinger* [2003] provide a comprehensive review of marking techniques and their respective advantages and disadvantages for different applications.

Recent findings from the ongoing research program on the Nicolet River restoration project are presented by examining field, laboratory, and numerical modeling results related specifically to flow and sediment transport around paired deflectors. This is followed by an ecological assessment of the Nicolet River project.

3.1. Field

A 300 m reach of the Nicolet River that contains two sets of paired deflectors (Figure 2) has been surveyed in detail since 1999, providing information regarding the morphology of pools that were excavated downstream from each deflector pair. The initial area and volume of pools and their initial position with respect to deflectors throughout the 14 km restored reach varies; in the study section, the downstream pool is the largest (Figure 4a). Permanent benchmarks were established at the onset of this research program to obtain detailed digital elevation models (DEMs) from a total station at a spatial resolution which was maintained constant through the years (~0.15 points m^{-2}). This approach allows temporal patterns of erosion and accumulation to be examined at a yearly scale as each survey was collected in May, before vegetation would hinder data collection near the banks. Since one of the primary objectives of this restoration project was to (re-)create deep pools, a key component in assessing its long-term viability is to examine in more detail the variation in pool area and pool volume over the years to determine if the paired deflectors are capable of maintaining

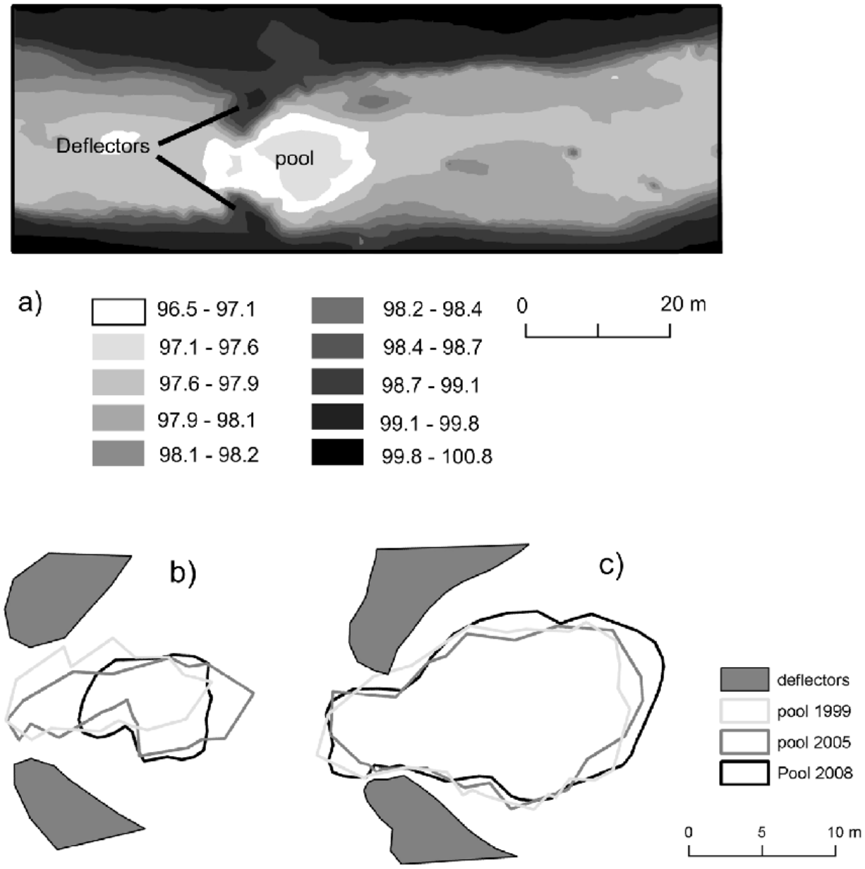

Figure 4. (a) Bed topography near the downstream paired deflectors, pool limit changes for (b) the upstream and (c) the downstream deflectors between 1999, 2005, and 2008. Flow is from left to right.

this habitat or if frequent maintenance will be required. As illustrated in Figures 4b and 4c), the upstream pool has evolved markedly since 1999, whereas the downstream pool appears much more stable. This is also reflected in pool volume which has decreased considerably in the upstream pool (Table 2). The downstream pool exhibits variations with time, but no systematic trend in volume (Table 2), although it seems to have migrated slightly downstream with time (Figure 4c).

In order to understand the adjustments that have occurred in the Nicolet River following the installation of in-stream structures, data on flow dynamics, bed shear stress and

Table 2. Pool Volume (m^3) of the Upstream and Downstream Pool at the Nicolet River

Year	Upstream Pool	Downstream Pool
1999	27.4	99.9
2005	7.0	58.9
2008	5.0	79.7

sediment transport were required. However, obtaining detailed velocity measurements in an intermediate-size river such as the Nicolet River is more complex than in smaller streams, where it is possible to wade or to use a portable bridge structure, or than in larger rivers, where measurements can be taken from a boat. In addition, as this project was designed for sport fishing, field devices such as cables installed to control a boat could not be used; they would have been in the way of the anglers. Under most flow conditions, flow measurement devices such as ADCP or PC-ADP could only have been used in the pool areas because the flow is too shallow in nonpool zones (0.30–0.50 m). Furthermore, as mentioned previously, bed shear stress estimates obtained from these devices in complex zones such as the recirculation zone downstream from the deflectors are not necessarily reliable [*Tilston and Biron*, 2006].

For these reasons, 3-D velocity was measured using an ADV. However, because of time constraints, it was not possible to cover the entire study reach for constant flow conditions using this approach, so data were limited to the zone near the downstream deflectors during low-flow conditions. As expected, these data revealed a marked increase in mean velocity and bed shear stress between the pair of deflectors [*Carré et al.*, 2007]. However, the limited spatial resolution of these point data restricts their usefulness in calibrating and validating a numerical model. Thus, LSPIV was also used to obtain simultaneous planform velocity data at the water surface. There are obvious drawbacks to this method: it cannot produce the vertical component of velocity and it is limited to water surface data, which are of little use to estimate bed shear stress. However, flow field information can be obtained for the entire water column from a 3-D model once it is validated with LSPIV data. For low-flow conditions, LSPIV compared well with ADV measurements [*Carré et al.*, 2006]. The agreement with the simulated flow field in the 3-D model Phoenics was qualitatively good, in particular, in the recirculation zone downstream of the deflectors, but the quantitative comparison between simulated and measured velocities resulted in correlation coefficients that were not very high (r = 0.67 for the velocity magnitude) [*Carré et al.*, 2006]. More research is currently underway to calibrate the 3-D model and improve the correlation with water surface LSPIV data.

Based on the success of LSPIV at low flow and on the difficulties in obtaining ADV measurements at high flow, the LSPIV approach was adapted to higher discharge conditions. The hope in collecting these data was to validate a high-flow 3-D model which would then have provided information about near-bed velocities and bed shear stress, variables that are needed to assess bed load transport. The same seeding material (confetti) and camera (720 × 480 pixel resolution) as *Carré et al.* [2006] were used in our high-flow experiments. However, since it was not possible to stand in the river at high flow to spread confetti, a dispenser system that could be controlled from the banks was developed. Eleven confetti dispensers were attached to a clothing line running across the river upstream from the pair of deflectors (Figure 5a). A tubing system allowed each dispenser to be agitated using a single handle. Another difficulty in collecting high-flow LSPIV measurements is ensuring that four benchmarks are in the camera view. This was overcome by placing a reflective prism on a float controlled by two operators, so that coordinates could be obtained from a total station (Figure 5b). The camera was also placed as high as possible on the bank to minimize the obliqueness of the camera view and the resultant distortion. Despite this, surface flares confounded the recognition of

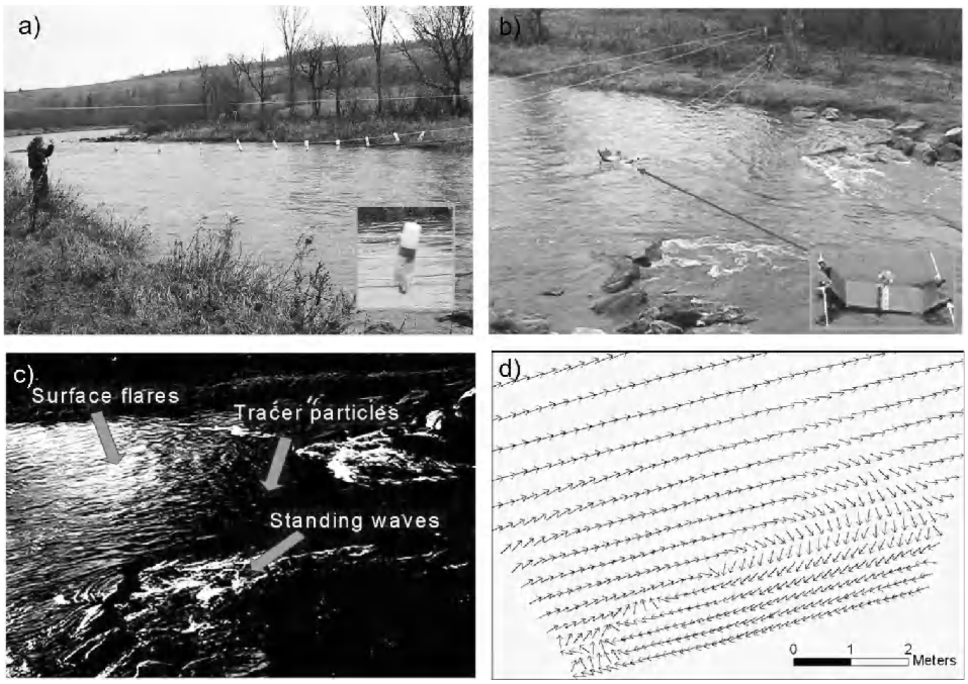

Figure 5. (a) Seeding of tracer particles (confetti) for PIV analysis at high flow in the Nicolet River. (b) Prism float used to record benchmark coordinates at the water surface. (c) Comparison of the appearance of surface flares, standing waves, and tracer particles in a PIV image. (d) Recirculation zone in the wake of the right-bank deflector as measured from LSPIV (arrows only indicate flow direction).

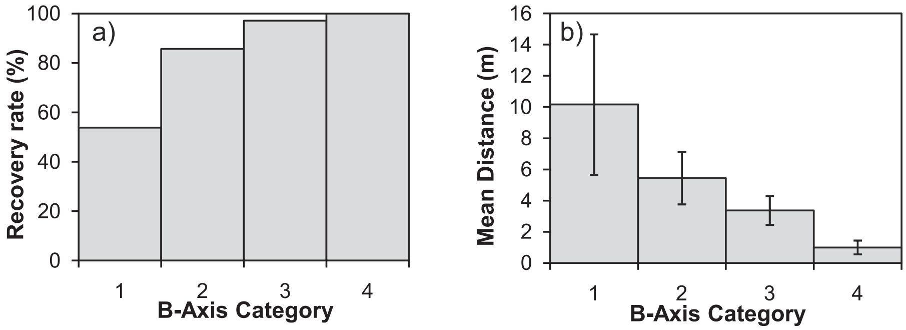

Figure 6. (a) Recovery rates of rocks for different sizes from May 2007 to September 2008. (b) Mean travel distance per size during the same period. Particle sizes are as follows: 1, 5.0 cm; 2, 7.4 cm; 3, 10.6 cm; and 4, 15.9 cm.

tracer particles in the PIV processing software (Figure 5c). Furthermore, standing waves either trapped tracer particles or generated white foam which resulted in the appearance of constant unmoving patches (Figure 5c). This resulted in an underrepresentation of the true water surface velocities. Nevertheless, LSPIV data provided useful information on streamline orientation at the water surface, in particular, the zone of flow reversal in the wake of the deflectors (Figure 5d). This method should be further investigated, perhaps using other types of tracers such as Ecofoam chips [*Jodeau et al.*, 2008]. Different types of camera filters could also be tested to limit the impact of surface flares, but keeping the tilt angle as small as possible (by raising the camera as high as possible) seems a key parameter to reduce the error with this method [*Hauet et al.*, 2008].

Bed load sediment transport was investigated using PIT tags [*Lamarre et al.*, 2005; *Carré et al.*, 2007]. This method allows individual particles to be monitored as each PIT tag has a unique code that is detected by an antenna, even when particles are buried under up to 0.60 m of sediment. Four sizes of particles were used, with an average *b* axis of 5.0 cm (size 1), 7.4 cm (size 2), 10.6 cm (size 3), and 15.9 cm (size 4). One hundred and ten PIT-tagged rocks were first introduced in 2005 with the objective of understanding

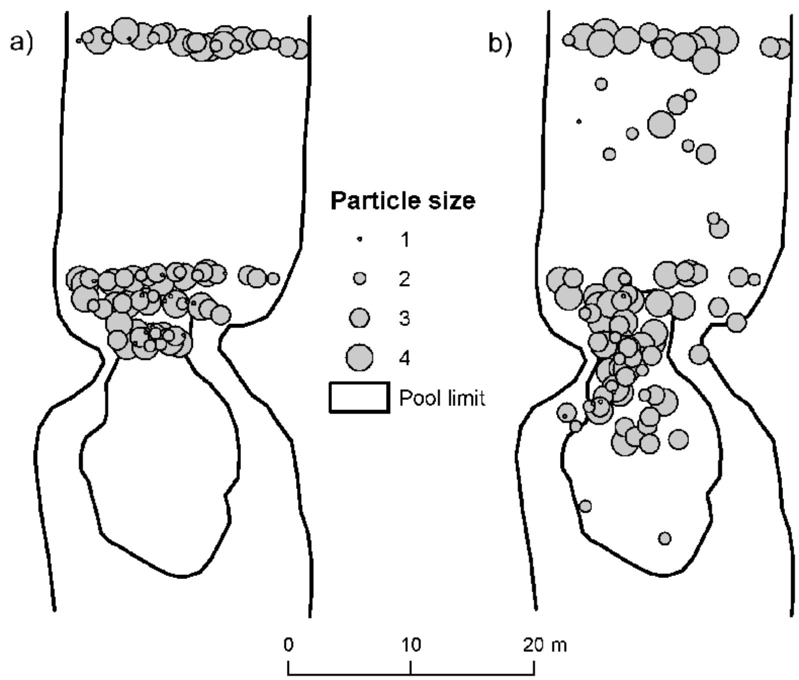

Figure 7. PIT tag movement at the Nicolet River. (a) Initial position in May 2008. (b) Positions at the end of August 2008. Particle sizes are as follows: 1, 5.0 cm; 2, 7.4 cm; 3, 10.6 cm; and 4, 15.9 cm. Flow is from top to bottom.

particle motion in the downstream deflector zone. Therefore, rocks that moved too far downstream were moved back upstream from the deflectors. From the original rocks placed in the spring of 2005, three were moved back upstream in July 2005. These rocks remained in the river until May 2007, when 53 were located. Another 59 rocks were tagged and added in 2007, and others from the 2005 cohort were rediscovered throughout 2007 and 2008. In 2008, two rocks were found to be broken so were removed, re-measured, and replaced. The method has been very successful in terms of recovery rate (Figure 6a), particularly for the coarser particles. As expected, mean travel distance decreases with increasing particle size (Figure 6b).

Tracer rocks were placed along several lines upstream from the deflectors in May 2008 (Figure 7a). By September 2008, several of these rocks had moved into the pool area (Figure 7b). A key question is whether these particles are capable of leaving the pool afterward, since it determines whether the pool will be maintained in the long term. Results indicate that from 2005 to 2008, of the 117 PIT-tagged particles that fell in the pool, only 27 are known to have exited. Many of the others moved erratically within the confines of the pool zone (Figure 8a), and some were lost. Smaller particles were more likely to move through the pool (23 out of the 27 that left the pool were in the two smallest categories), and none of the 30 largest rocks entering the pool escaped. Some particles were buried fairly deeply in the pool zone. Figure 8b illustrates the locations through time of the four particles in the size 3 category, which were discovered downstream of the pool in 2008.

A large flood in August 2008 with discharge exceeding bankfull discharge by approximately 15% provided an opportunity to quantify the role of a large flood in particle transport. Ten particles were found downstream of the pool area after that event: none from the largest size (although one had moved to the downstream slope of the pool very close to the pool limit), two from the size 3 category (Figure 8b), and the remaining eight from the two smallest sizes. Eight of these rocks were from among the 64 particles known to be in the pool area before the flood. The other two had not been seen since late 2007, so were probably extricated from under more than 30–40 cm of bed material by the flood, using a conservative estimate of the maximum range of the detection equipment which had been used to try to locate them. They may or may not have also been transported during the event. An even larger flood occurred in January 2008, but it is difficult to distinguish that flood's effects from the spring freshet. What is clear is that floods that meet or exceed the spring flood are required to move large particles and that the largest size particles will likely remain in the pool for many years before a sufficiently large event can move them downstream.

3.2. Laboratory

The initial laboratory experiments on flow deflectors were designed to provide a better insight on flow dynamics and scouring processes near various deflector designs [*Biron et al.*, 2004, 2005]. These results revealed that 90° deflectors generated larger scour than did 45° and 135° angles [*Biron*

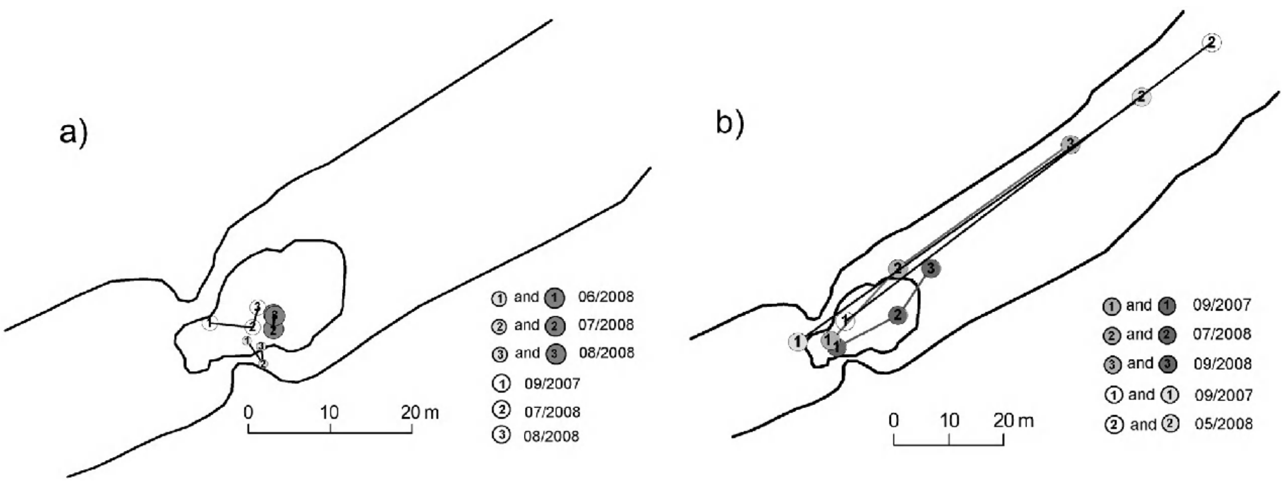

Figure 8. Examples of movement of particles in the pool area during the summer 2008: (a) Very little movement within the pool (size 2 to 4). (b) Patterns of the four size 3 particles that have traveled through the pool and were found downstream. Flow is from bottom left to top right.

Table 3. Experimental Conditions in Laboratory Runs Based on Nicolet River Deflector Design[a]

Run	Mean Deflector Height h (m)	Water Depth z (m)	Overtopping Ratio (z/h)	Shear Velocity u_* ($\mathrm{m\ s^{-1}}$)	Shear Velocity Ratio u_*/u_{*c}	Froude Number $U/(gz)^{1/2}$	Discharge Q ($\mathrm{m^3\ s^{-1}}$)	Reynolds Number (Uz/v)	Slope
1	0.056	0.110	1.96	0.028	0.82	0.242	0.011	24280	0.0007
2	0.056	0.062	1.11	0.026	0.77	0.302	0.010	22800	0.0005

[a]Here u_* is shear velocity; u_{*c} is critical shear velocity. U is mean approach flow velocity, g is acceleration due to gravity, and v is kinematic viscosity.

et al., 2004]. The laboratory experiments tested various deflector angles, lengths, and heights for constant flow conditions. However, they were not representative of natural situations where deflector roughness is important and where overtopping conditions are frequent and associated with varying flow conditions. In order to examine these questions, a new series of experiments was initiated to reflect more closely the Nicolet River situation. The experimental conditions of these new experiments are described in Table 3.

Ideally, a scaled model would have been used. However, several constraints are associated with scale models [*Peakall et al.*, 1996; *Ettema and Muste*, 2004; *Maynord*, 2006]. For example, the width scaling factor between the flume (0.4 m) and the field is 63.2. In theory, the vertical dimension should be scaled using the same factor. This would have resulted in flow depth of only 2.3 cm at bankfull level, which can lead to surface tension effects [*Henderson*, 1966]. Instead, the vertical factor was assigned a value of 13.2, which created a distortion factor deemed acceptable from earlier practice [*Henderson*, 1966; *Maynord*, 2006], although ideally, a smaller distortion factor should be used when examining scouring processes [*Lee and Sturm*, 2009].

Furthermore, it was not possible in the flume to represent the variation in grain size that is observed at the field site

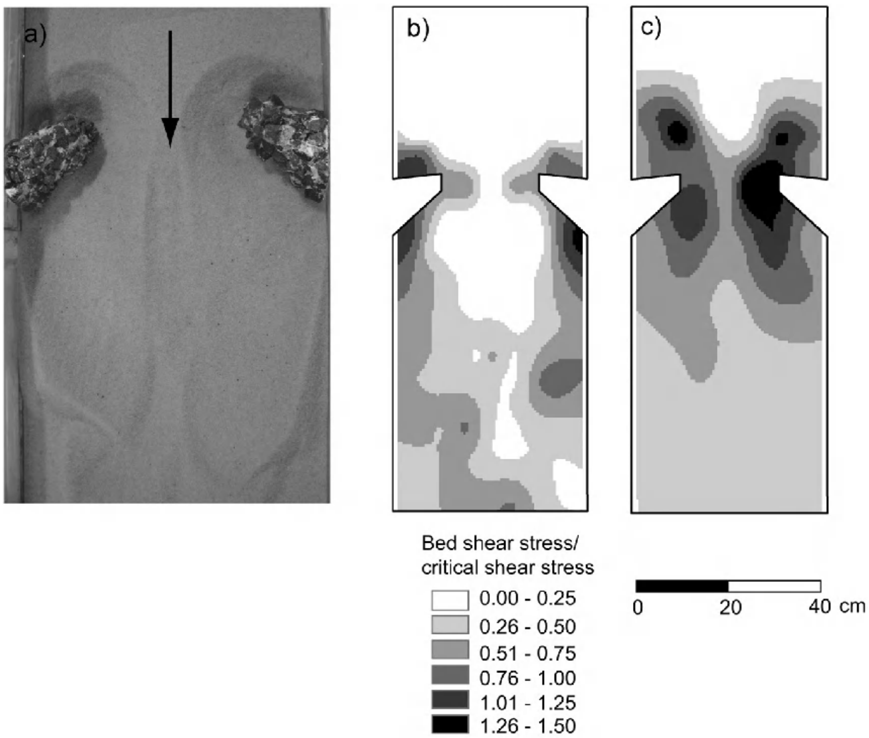

Figure 9. (a) Laboratory setup with boulder deflectors representing the Nicolet deflectors at the end of a run corresponding to medium-flow stage (Figure 9b). (b) Bed shear stress (turbulent kinetic energy method) divided by critical shear stress around the boulder deflectors at medium level with $u_*/u_{*c} = 0.77$. (c) Bed shear stress divided by critical shear stress around deflectors at high flow with an overtopping ratio of 1.96 and $u_*/u_{*c} = 0.82$. Flow is from top to bottom.

with the median diameter (D_{50}) ranging between 5 and 225 mm (mean: 39 mm). Instead, uniform sand particles (D_{50} = 1.1 mm) were used. Clearly, this is an important constraint as (1) the scaling of sediment size to depth and structure size can have important effects on prediction of scour (e.g., around bridge piers [*Lee and Sturm*, 2009]); (2) it is well known that the dynamics of uniform sand and heterogeneous mixture of sediments are very different [*Andrews*, 1983; *Buffington and Montgomery*, 1997; *Wilcock and Crowe*, 2003]; and (3) armoring, which can greatly modify the entrainment conditions in coarse-bedded rivers, will not occur in uniform sand bed [*Klingeman and Emmett*, 1982; *Gomez*, 1983]. However, the main objective of these laboratory experiments was not to replicate precisely the sediment dynamics of the Nicolet River, but rather to investigate the impact of overtopping flow conditions on the scour for deflectors that were closer to designs typically used in natural rivers (i.e., log- or boulder-based in-stream structures) than those typically used in laboratory experiments.

Deflector skeletons were built from Plexiglass and then covered in gravel with marine epoxy to create a component of roughness. Furthermore, the shape of the deflectors was designed to resemble the deflectors in the field (Figure 9a). Figures 9b and 9c present the pattern of bed shear stress for a medium flow situation, where deflectors are barely submerged, and for higher flow stage with an overtopping ratio of 1.96. Unlike previous studies, where the ratio of shear velocity (u_*) to critical shear velocity (u_{*c}) was maintained constant between experiments [*Kuhnle et al.*, 1999, 2002], the higher flow stage had a higher shear velocity ratio (u_*/u_{*c}) compared to the medium flow stage (0.82 compared to 0.77, Table 3). Results show that bed shear stress around the deflectors is markedly larger for the high-flow condition (Figure 9c) than the medium-flow one (Figure 9b), despite the overtopping ratio being larger and close to 2 in this high-flow case. This corresponds well to bed topography data in the laboratory experiments, which reveals that the high-flow experiment produced scour volumes 3.6 times those observed for the medium flow run. However, in other experiments where 90° deflectors of various heights were used with a constant approach shear velocity ratio of 0.97, higher overtopping ratios were associated with smaller scour volumes. These results illustrate the importance of taking into account field characteristics such as varying shear velocity ratio before coming to conclusions on the effect of the overtopping ratio in the scouring process. Furthermore, it is interesting to note that pools that developed around the Nicolet-like deflectors in the laboratory experiments were always located near the deflector tips (Figure 9a) [*Rodrigue-Gervais et al.*, 2011]. Thus, the excavated pool, located in the center of the channel in the Nicolet River, does not correspond to the zone that would be scoured by these structures due to shear forces acting on the bed.

3.3. Three-Dimensional Numerical Model

Our first attempts to investigate the 3-D flow field around deflectors numerically with the Phoenics model were based on laboratory experiments [*Biron et al.*, 2005], where it is easier to obtain detailed 3-D velocity measurements to calibrate and validate the model. One of the difficulties in using 3-D models with angled deflectors when the bed geometry is complex lies in the design of the numerical mesh [*Biron et al.*, 2007]. Here, a method involving an "object-bed," which is a 3-D representation of the DEM, was used with a Cartesian grid to avoid major distortions of the numerical grid. The Phoenics model with an object bed proved successful in simulating the complex flow field around deflectors in the laboratory [*Haltigin et al.*, 2007a, 2007b]. Thus, the same approach was used to simulate the flow field in the Nicolet River, using both ADV and LSPIV to calibrate and validate the model [*Carré et al.*, 2006]. As is shown in Figure 10a, the flow pattern near the deflectors is highly 3-D and could not be represented adequately in a 2-D model. A qualitative comparison between streamlines and PIT-tagged particle movement in the pool area reveals similar trends, i.e., both streamline and bed load movement are oriented mainly toward the left bank when looking downstream (on the right in Figure 10a) at the exit of the pool. However, the shear stress pattern for both low flow (Figure 10b) and high flow (overtopping ratio of 1.38, Figure 10c) is markedly different from patterns observed in the laboratory experiments (Figures 9b and 9c) where the pool zones were located on each side of the deflectors. This suggests that feedback between bed morphology (here, an excavated pool), flow dynamics, and sediment transport are important. It also raises the question of how the dimensions and location of excavated pools are determined for different deflector designs in restoration projects. In the case of the Nicolet project, no tests were made to verify if the excavated pool was optimally positioned or dimensioned based on the upstream deflectors. A better understanding of scouring processes near structures, as revealed from laboratory analyses, could have been used to modify the design, for example, excavating a pool on each side instead of a single pool in the center.

Work is currently under way to examine higher overtopping flow conditions, with shear stress values sufficiently large to generate significant bed load transport, using the 3-D model. This is essential to further investigate bed load dynamics in the pool zone, which is shown from the field experiments (PIT-tag data) to occur only at flows near

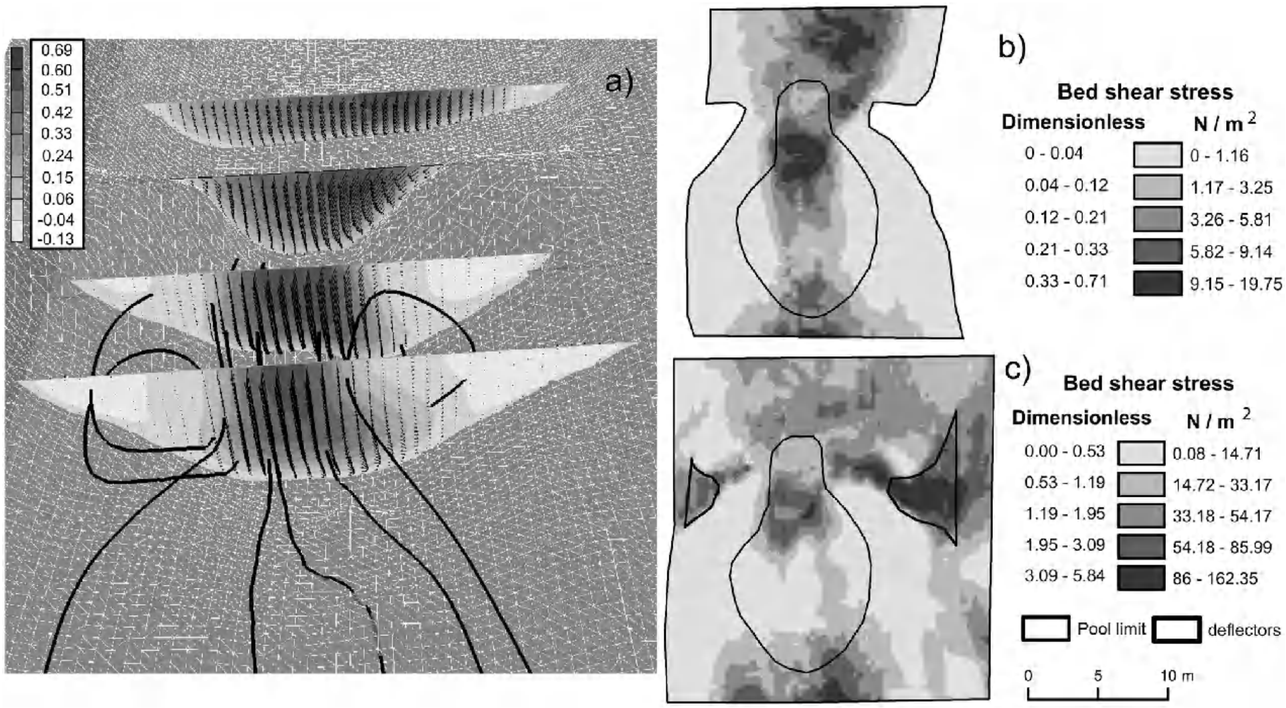

Figure 10. (a) Simulated three-dimensional downstream velocity (m s^{-1}) at low flow in the Nicolet River around paired deflectors and in the pool area. The view is looking upstream. Streamlines close to the bed are also shown as black curves. Simulated bed shear stress (obtained from the law of the wall) at (b) low-flow condition and (c) high-flow condition (overtopping ratio of 1.38). Dimensionless values are obtained by dividing by (Shields) critical shear stress. Flow is from top to bottom.

bankfull discharge. Eventually, various deflector and excavated pool designs will be tested using a 3-D model to assess which one is more likely to maintain pools in the long term. Considering the cost of installing in-stream structures, this would be a valuable tool in river management projects.

3.4. Habitat Utilization

Most of the scientific effort in assessing the Nicolet River restoration project, so far, has been limited to a relatively small study reach and has focused on physical parameters. A recent ecological assessment was therefore conducted at the scale of the entire restored reach (14 km, Figure 3b) to determine whether restored areas result in improved trout habitat. The initial objective of this project was to determine which habitat was used by the three species of trout in the Nicolet by snorkeling through the reach (note that electrical fishing was not an option because of the presence of anglers and depth of water). However, it proved impossible to link physical habitat with actual trout habitat with this approach. Snorkeling observations require that the fish are undisturbed long enough for identification to be made. This is usually accomplished by moving upstream and observing the fish before they notice the snorkeler [e.g., *Shuler et al.*, 1994; *Thurow and Schill*, 1996; *Mullner et al.*, 1998; *Thurow et al.*, 2006]. In this case, very few fish were observed, and of those, the majority were disturbed and fled upstream. Low visibility was a problem. There were few days when visibility exceeded 1.5 m, the minimum suggested for making underwater observations [*Goldstein*, 1978], and visibility never exceeded 2 m. Low visibility made it impossible to see the entire width of the river, and in some of the deeper pools, the bottom was not visible. This is problematic as *Peterson et al.* [2005] found that salmonids responded to the presence of a snorkeler at 10 to 20 m; thus, fish may react to the presence of a snorkeler before the snorkeler could actually see them in the Nicolet River. While many studies have used underwater observation to determine fish abundance and habitat [e.g., *Goldstein*, 1978; *Shuler et al.*, 1994; *Thurow and Schill*, 1996; *Mullner et al.*, 1998], the majority was conducted on streams much smaller than the Nicolet and with better visibility.

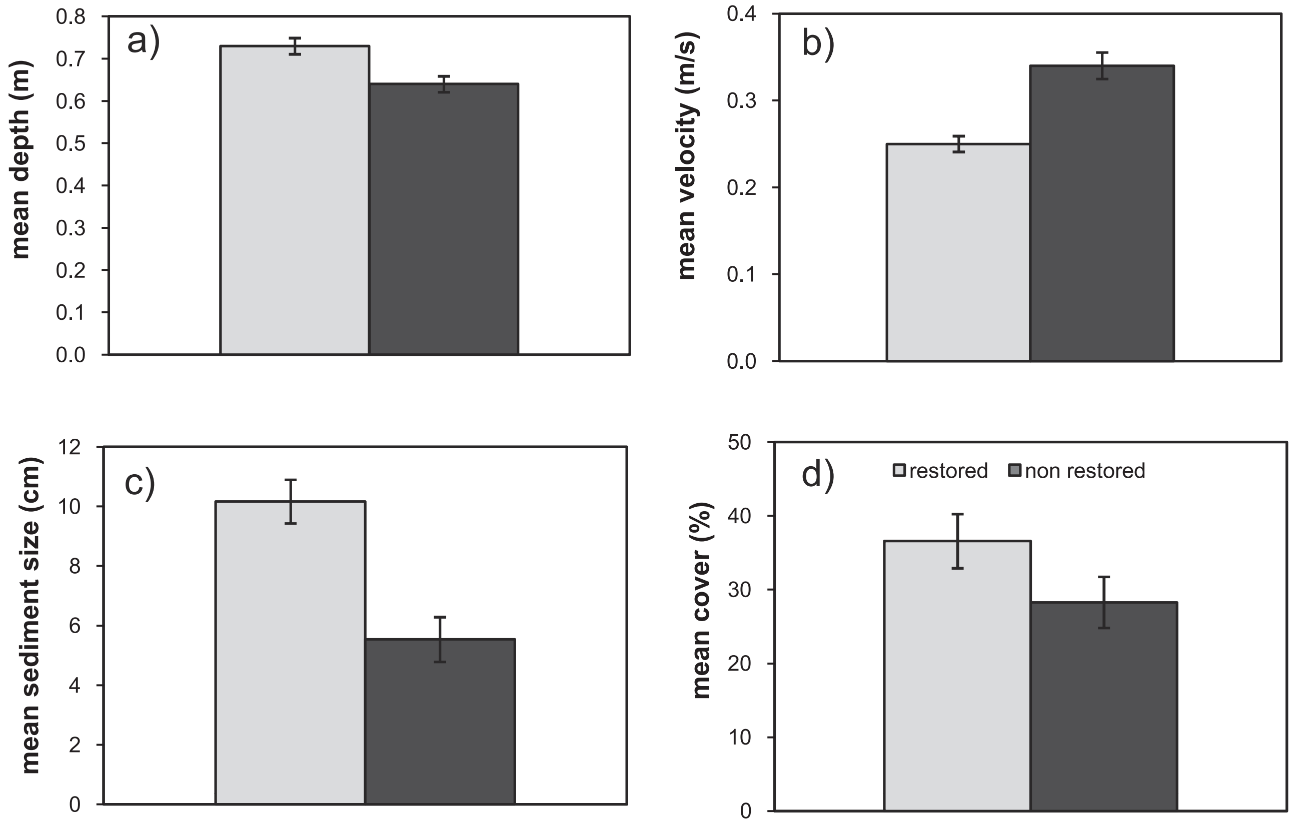

Figure 11. Habitat differences in restored and nonrestored reaches of the Nicolet River: (a) mean depth, (b) mean velocity, (c) mean sediment size, and (d) mean cover. Here $n = 502$ for restored and 246 for nonrestored areas (except for percentage cover where $n = 16$ for restored and 10 for nonrestored habitat).

The physical habitat of pools downstream from a restoration structure (restored pools) was compared with that of naturally occurring, nonrestored pools. The 14 km project reach consists of 70 pools, of which 30 were restored (i.e., excavated) and 40 were nonrestored pools. The average pool depth of nonrestored pools was 0.63 m, whereas restored pool average depth was 0.74 m (for low-flow conditions). The physical habitat assessment reveals that restored

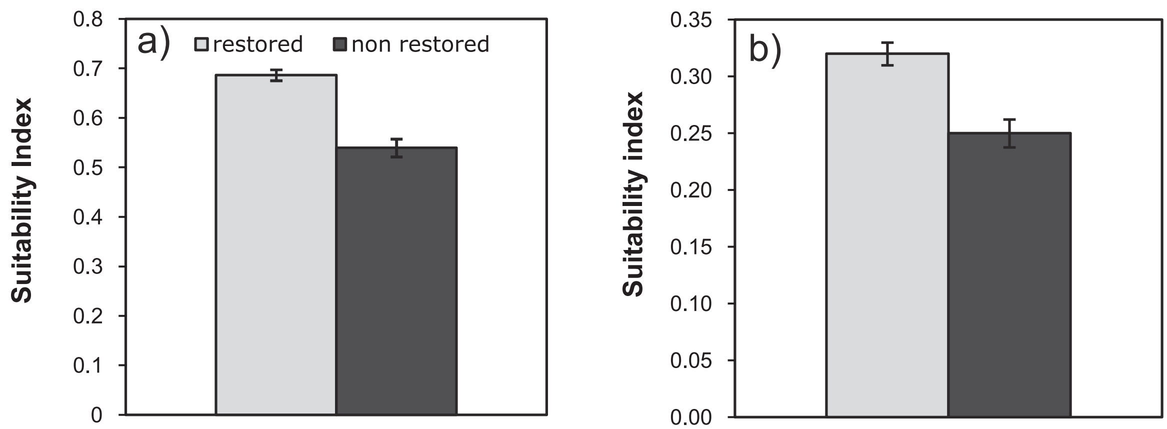

Figure 12. Habitat suitability index for (a) rainbow trout [*Raleigh et al.*, 1984] and (b) brown trout [*Raleigh et al.*, 1986] for restored and nonrestored reaches at the Nicolet River.

pools are deeper, slower, have larger sediment size, and different forms of cover than the nonrestored pools (Figure 11). Here, velocity represents average water column velocity, measured at 0.4 of the flow depth when depth was less than 0.5 m, and at an average from 0.2 and 0.8 of the flow depth when depth was over 0.5 m. Cover was considered present if there was any object large enough to provide a velocity refuge for a 15 cm fish within 1 m of the sampling location. Boulders provided almost all of the cover in the restored pools, whereas the nonrestored pool cover was fairly evenly divided between cover provided by boulders, woody debris, and undercut banks.

This corresponds to an overall improved habitat for both rainbow and brown trout when using a suitability index developed for similar rivers (Figure 12). This indicates that the physical habitat changes created by the in-stream structures have the potential to improve trout summer habitat. However, there are major limitations to using habitat preference curves developed in other rivers [*Moyle and Baltz*, 1985; *Rosenfeld*, 2003]. Differences in intraspecific competition, habitat availability and food abundance are all listed as potential difficulties for transferability of suitability curves [*Strakosh et al.*, 2003]. For that reason, the recommended protocol is to develop, or modify, suitability curves on site [*Raleigh et al.*, 1986; *Glozier et al.*, 1997; *Strakosh et al.*, 2003]. Thus, the restoration project appears to have successfully altered the physical habitat of the Nicolet River in a manner that may increase trout habitat suitability even if it was not possible to determine whether or not trout are actually using this newly created habitat.

One of the complications in determining the habitat preference of trout in the Nicolet River is the patchiness of stocking and fishing pressure. A survey of anglers during the summer of 2008 showed that the majority of trout catches occur in pools that either are stocked or are adjacent to stocked pools. It has been found that stocked trout are unlikely to move more than a few kilometers from their stocking site [*Cresswell*, 1981; *Helfrich and Kendall*, 1982; *Hesthagen et al.*, 1989; *Aarestrup*, 2005]. This distance is further reduced when they are stocked in warm water, in the same season in which they are caught, into larger streams or rivers [*Cresswell*, 1981], or into pools rather than riffles [*Helfrich and Kendall*, 1982]. All of these factors make it unlikely that the stocked fish in the Nicolet will fully use the 14 km of restored habitat. Since the stocking sites are known to local anglers, fishing pressure is also highest around the stocked pools. After analyzing the location of 434 trout catches, there was no evidence that fish were more likely to be caught in restored pools. However, the uneven stocking and fishing effort make drawing any conclusions about habitat preference impossible.

4. CONCLUSION

In the last decade, remarkable improvements were made in 3-D modeling (e.g., LES) and laboratory measuring devices (e.g., PIV), which have greatly improved our knowledge of flow dynamics around in-stream structures used for fish habitat rehabilitation schemes. However, more work is clearly needed to improve the success rates of these projects. In particular, more field data are required during high-flow conditions in which bed particles are in motion. Sediment transport in gravel bed environments needs to be assessed to determine the long-term effectiveness of in-stream structures. This is still a challenge since laboratory studies cannot easily deal with a scaled representation of gravel bed rivers, and roughness is difficult to simulate adequately in 3-D models (although it is much more accurately addressed than in 2-D or 1-D models).

A combination of field, laboratory and numerical modeling approaches appears to be the most efficient way of tackling this problem. Hopefully, more long-term monitoring research programs such as the one on the Nicolet River will be initiated in the future, as a lack of monitoring is a major problem in the science of stream restoration.

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Connectivity and Variability: Metrics for Riverine Floodplain Backwater Rehabilitation

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The importance of floodplain aquatic habitats that are seasonally or periodically connected to the main channel (backwaters) within lowland riverine ecosystems is well established. However, backwaters are becoming rare as development is transforming floodplain landscapes. Therefore, rehabilitation, protection, and management of riverine backwaters are becoming increasingly common, with annual expenditures in the millions of dollars. Even with the increasing number of projects, general criteria for selecting restoration goals and evaluating project outcomes are lacking. To address this need, Kondolf et al. (2006) proposed an approach for evaluating river restorations that is based on assigning a position to the system in a four-dimensional space that represents hydrologic temporal variability on one axis and connectivity in the three spatial dimensions on the remaining three axes. Use of the Kondolf approach for evaluating restoration of a backwater adjacent to a medium-sized river in northern Mississippi is presented as a case study, in which nearby degraded and less impacted backwaters were used as references. The restoration project resulted in a reduction in main-channel connectivity and lower levels of variability for the treated backwater. Additional responses to treatment included increased summer water depth, moderation of severe diurnal water quality fluctuations, and reductions in concentrations of solids, nutrients, and chlorophyll *a*. Fish species richness, numbers, and biomass were unchanged following rehabilitation, but trophic structure shifted away from omnivorous species and toward predators. Ecological services provided by floodplain riverine backwaters may be enhanced by modest management measures, but regaining and maintaining connectivity with adjacent ecological functional patches remains difficult.

1. INTRODUCTION

Freshwater ecosystems in the United States are exceptionally diverse, even compared with the tropics [*Master et al.*, 1998]. In particular, streams in the southeastern United States ("Southeast") are important ecological resources, but resident aquatic fauna are experiencing accelerated extinction rates [*Ricciardi and Rasmussen*, 1999; *Warren et al.*, 2000; *Karr et al.*, 2000]. Apparently, faunal declines reflect disruption of important connections between main channel and slack water habitats such as wetlands, abandoned channels, sloughs, severed meander bendways, and borrow pits [*Buijse et al.*, 2002; *Ward et al.*, 2001; *Wiens*, 2002; *Jackson*, 2003; *Kondolf et al.*, 2006], or in more current terminology, disruptions of connections between hydrogeomorphic patches [*Thorp et al.*, 2006]. The timing, frequency, and duration of hydrologic connections between rivers and their backwaters have important ecological implications. Many plant and animal species native

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to river corridors have life cycles that require access to backwater habitats during certain seasons or while certain climatic conditions exist. For example, reproduction may jointly depend on certain ranges of water temperature, photoperiod, and flooding. Furthermore, disrupted connections usually lead to drying and gradual terrestrialization of backwaters [*Gore and Shields*, 1995; *Schramm and Spencer*, 2006].

The Lower Mississippi River alluvial plain ("the Delta") is a case in point. This region contains numerous floodplain lakes that experience varying levels of hydrologic connectivity during periods of high stage in adjacent streams and rivers. Many of these water bodies receive significant inflows of water and associated pollutants from cultivated lands and have experienced precipitous declines in water quality and fisheries in recent decades. Recent studies of Delta lake fisheries indicate that lake area, lake elongation, and lake water clarity are key abiotic variables that control fish community structure, with small, shallow, elongated lakes most seriously degraded [*Miranda and Lucas*, 2004]. Backwater ecosystems often suffer from problems associated with hydrologic perturbation due to levees, dams, main channel incision, and backwater sedimentation [*Bellrose et al.*, 1983; *Bhowmik and Demissie*, 1989; *Hesse and Sheets*, 1993; *Jackson*, 2003]. Additional issues include water quality degradation, aquatic plant infestation and die-off, and extreme variation in water temperature and habitat volume [*Claflin and Fischer*, 1995; *Light et al.*, 2006; *Justus*, 2009]. One of the most pernicious problems may be described as vertical disruption of lateral connectivity. Hydrologic connections between the river and backwaters become shorter and less frequent when river stages are lowered through channel incision or when controlling elevations for floodplain water bodies are raised by sediment deposition [*Light et al.*, 2006].

Ecological restoration may be thought of as an attempt to return an ecosystem to its historic (predegradation) trajectory [*Society for Ecological Restoration International Science and Policy Working Group*, 2004] (accessed 23 November 2009). Restoration workers attempt to establish this "trajectory" through a combination of information about the system's previous state, studies on comparable intact ecosystems, information about regional environmental conditions, and analysis of other ecological, cultural, and historical reference information [*Society for Ecological Restoration International Science and Policy Working Group*, 2004]. In lightly altered natural systems, backwaters tend to follow a trajectory similar to classical lake eutrophication: due to sedimentation and perhaps migration of the river main stem, these areas become shallower, and connections to the river become briefer and less frequent. However, the formation of new backwaters due to main channel avulsion and more gradual processes continues as old backwaters become wetlands and eventually terrestrial systems. In altered floodplains, however, backwater formation processes are hindered or absent. Flood control and channel stabilization prevent formation of new backwaters as existing backwaters age, becoming shallower, more turbid and often experiencing lower dissolved oxygen (DO) concentrations [*Miranda*, 2005]. Extremely shallow backwaters tend to experience lower DO than deeper ones due to respiration occurring throughout the water column [*Miranda et al.*, 2001]. These systems also tend to have lower water transparency due to benthivorous fish and phytoplankton [*Roozen et al.*, 2003; *Miranda and Lucas*, 2004; *Lin and Caramaschi*, 2005]. Fish communities in such systems exhibit strong linkages to abiotic factors and are dominated by tolerant omnivores with few predators [*Miranda and Lucas*, 2004].

Since a hallmark of river corridor development is reduction of lateral linkages, many river restoration projects have focused on managing floodplain water bodies and their connectivity with the main channel (e.g., *Holubova et al.* [2005], but see *Pegg et al.* [2006]). Based on a study of 29 floodplain lakes in the region containing our sites, *Miranda and Lucas* [2004] recommended rehabilitation efforts focus on watershed management, dredging or water level control, and fishery management. Existing backwater rehabilitation projects feature practices such as pumping in water, breaching levees, reopening relatively small connecting channels, or by constructing water control structures to increase water depth during dry periods [*Shields and Abt*, 1989; *Theiling*, 1995; *Galat et al.*, 1998; *Amoros*, 2001; *Buijse et al.*, 2002; *Valdez and Wick*, 1981; *Grift et al.*, 2001; *Shields et al.*, 2005; *Schultz et al.*, 2007; *Julien et al.*, 2008]. Substantial sums have been spent in these efforts, but little information is available regarding the performance of existing projects to guide future design efforts [*O'Donnell and Galat*, 2007; *Palmer et al.*, 2007]. At least three approaches (or combinations of these) for generating criteria are possible. First, backwater treatments may be designed, maintained, and operated to meet habitat requirements for a selected species or group of species [*Galat et al.*, 1998]. Second, criteria may be set to produce selected characteristics of a reference site. Third, using an approach described by *Kondolf et al.* [2006], backwater physical conditions may be assessed in terms of hydrologic variation and main channel connectivity, as described below. The objective of this paper is to show how the ecological performance of a backwater rehabilitation project may be assessed using the Kondolf approach.

2. KONDOLF DIAGRAM

Kondolf et al. [2006] proposed use of hydrologic connectivity and variability (also referred to as flow dynamics) as key descriptors of riverine ecosystem status. Hydrologic connectivity was defined as water-mediated fluxes of material, energy, and organisms among the major ecosystem components: main channel, floodplain, aquifer, etc. [*Amoros and Bornette*, 2002]. Connectivity occurs in all three spatial dimensions: longitudinal (upstream and downstream), lateral (main channel and floodplain), and vertical (surface water and the hyporheic or deeper subsurface regions). Variability was primarily defined as temporal variation in discharge, but it also encompasses parameters such as temperature, sediment, and trophic levels [*Hughes et al.*, 2005]. Connectivity and variability tend to be related. For example, construction of a dam to regulate flow often reduces the frequency and duration of floods downstream, reducing lateral connectivity and flow variation. Furthermore, the dam may reduce longitudinal connectivity by presenting a barrier to movements of sediment and organisms. Connectivity tends to be reduced by human activities (e.g., construction of dams, levees, channelization, flood reduction, and blockage of side channels) or by geomorphic change produced by human activities (e.g., channel incision). The status of a given riverine ecosystem may be mapped by plotting a point representing the system within a Cartesian plane with the horizontal axis representing variability and the vertical axis representing connectivity in a selected dimension (Figure 1). Multidimensional plots may be used if connectivity is mapped in more than one dimension. If information is available, points may be plotted representing predegradation and current conditions, giving a degradation trajectory. Ideally, restoration would simply follow the reverse path of the degradation vector, returning the system to its predegradation connectivity and flow variability. If predegradation data are not available, reference conditions may be inferred from lightly degraded sites.

To illustrate this concept, *Kondolf et al.* [2006] plotted degradation trajectories for 23 rivers using at least one dimension to measure connectivity. Month-to-month flow variation, with special emphasis on the probability of intermittent flow, was used to indicate streamflow variability, with sites arrayed along a continuum ranging from spring fed to snowmelt to rain fed to intermittent or ephemeral regimes. Restoration trajectories were plotted for systems that were sites for restoration projects. In general, the bivariate plots showed that systems tended to follow paths that resulted in reduced connectivity and variability as they degraded, although some sites (e.g., base flow diversions and channelization) became more variable as they were degraded. Rehabilitation or restoration often increased connectivity but rarely increased variability. Preparing a "Kondolf diagram" for a system selected for restoration requires completion of four key tasks: assessment of historical conditions, definition of degradation in process-based terms, identification of factors triggering degradation, and setting goal trajectories for

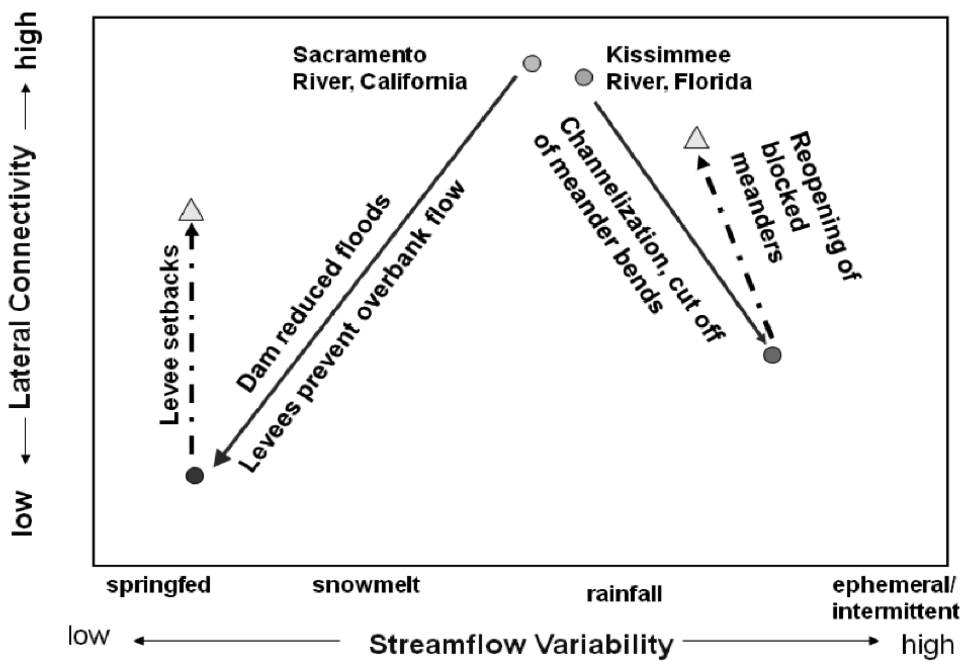

Figure 1. Diagram for assessment of aquatic ecosystem status. Solid arrows represent ecological degradation, and dashed arrows represent restoration trajectories plotted on axes of lateral connectivity and flow dynamics. After the work of *Kondolf et al.* [2006].

selected processes. Herein, we adopt this approach, not for river reaches as originally proposed, but for individual floodplain water bodies or backwaters. Clearly, the overarching goal of backwater rehabilitation is to contribute positively to the entire river ecosystem, but the open nature of the river system and the mobility of its fauna make measurement of the effects of restoring one or a few backwaters on the entire river ecosystem impossible. This project seeks to gauge impacts of rehabilitation of a single backwater body on its connectivity and variability and to relate these outcomes to the backwater's ecological functions as manifest in water quality and fish populations.

3. STUDY SITES

A reach of the Coldwater River about 20 km downstream from Arkabutla Dam in northwestern Mississippi was selected for study due to the presence of more than 20 severed meander

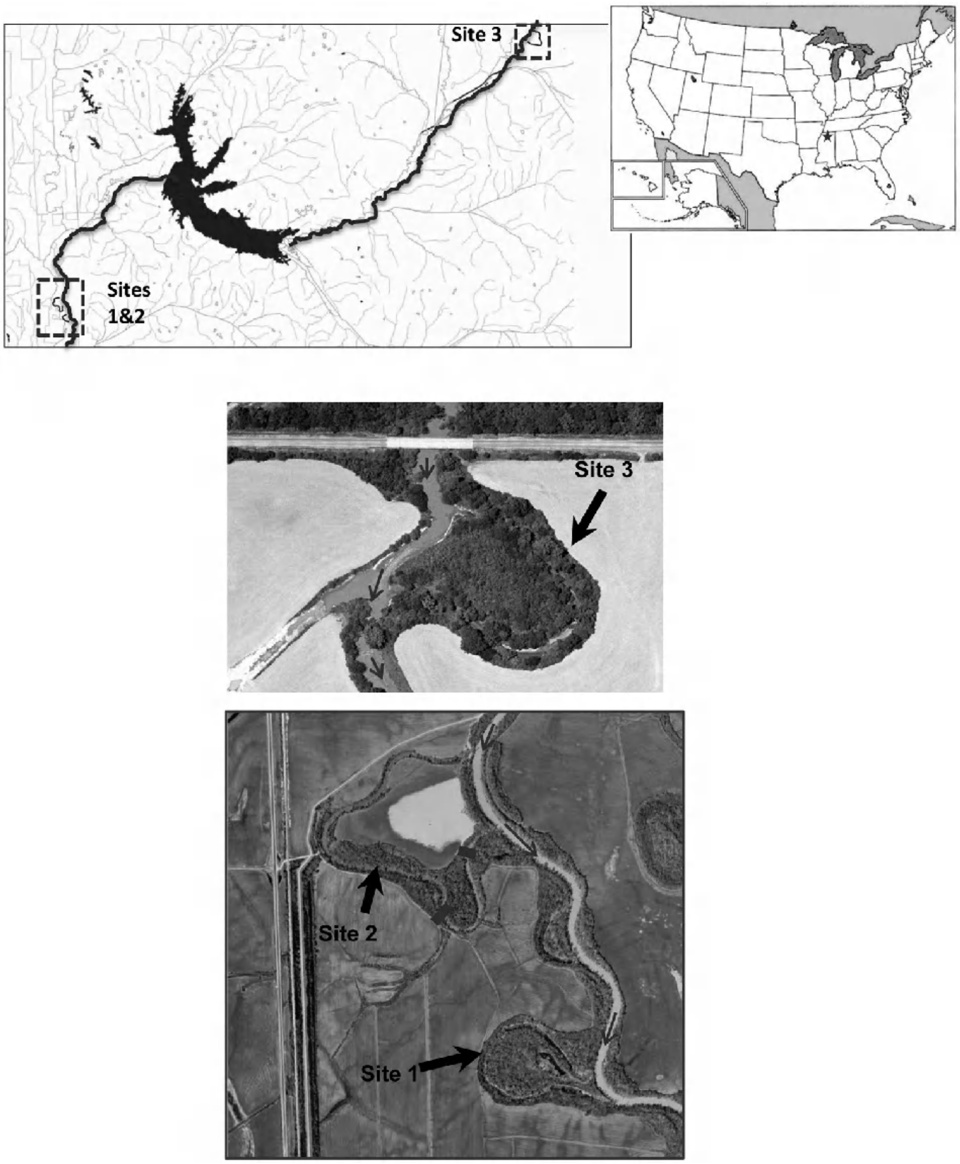

Figure 2. Study locations upstream and downstream from Arkabutla Reservoir. Air photos show site 3 in 2000 and study sites 1 and 2 in 1957. Sites 1 and 2 (34°40.024′N, 90°13.373′W) were cut off from Coldwater River in 1941–1942, and site 3 (34°51.572′N, 89°48.375′W) was cut off prior to 1991. Site 2 was treated by addition of weirs (gray rectangles) in 2006.

bends and other floodplain water bodies along the river. Elevated suspended sediment concentrations, habitat reduction associated with sedimentation and water pollution associated with agriculture are primary resource problems in this locale [*Mississippi Department of Environmental Quality*, 2003, U.S. Corps of Engineers, Coldwater River Basin below Arkabutla Lake, Mississippi, Section 905(b) reconnaissance report, undated, Vicksburg, Mississippi]. In addition, flows are highly regulated by the upstream impoundment, which is operated for flood control and recreation. Despite these problems, 13 to 22 species of fish were captured annually between 1990 and 1994 from this stretch of the Coldwater. Catch per unit effort (hoop nets) for the Coldwater River (all species and seasons) exceeded the four other Yazoo basin rivers sampled during the same time period [*Jackson et al.*, 1995].

Three Coldwater River floodplain backwaters were selected for study (Figure 2). Two were severed meander bends along the aforementioned reach below Arkabutla Dam, while the third was upstream from the reservoir. Sites were designated 1, 2, and 3 from downstream to upstream and were used as degraded reference, rehabilitation site, and least-impacted reference, respectively. Sites 1 and 2 were severed meander bends created by man-made cutoffs constructed in 1941–1942 [*Whitten and Patrick*, 1981]. Both were 1.5 to 2 km long and 40 m wide and were inside the main stem flood control levee. Lands outside the old bends were in row-crop cultivation, while lands inside the bends were in forest (site 1) or fallow (site 2). Buffers of natural vegetation 5–100 m wide were on both banks of the old channels. Both backwaters received runoff from cultivated fields. Backwater levels were tightly coupled with Coldwater River stage when the river stage exceeded the controlling elevation in the downstream connecting channel, but during the warmer months, the river was 1 to 3 m lower than the backwaters, and the backwaters became quite shallow. Site 1 was almost completely choked with aquatic plants during warmer months. Probing bed sediments at both sites with metal rods and sampling site 2 with a Vibracore apparatus revealed 2–2.5 m of fine-sediment deposition, with mean annual rates of about 3.1 cm yr^{-1} based on vertical profiles of sediment density and Cs-137 activity [*Shields et al.*, 2010]. Previously reported sediment sample chemical analyses and invertebrate bioassays indicated sediment metal concentrations were likely not high enough to create toxic impacts, but several insecticides were detected and impacted bioassays [*Knight et al.*, 2009a, 2009b].

A third severed bendway (site 3) located on the same river, but upstream from Arkabutla Lake and outside of the zone of reservoir influence, was used as a less impacted reference. There were no significant local inflows, and runoff from adjacent fields was diverted away from the bend by a low levee. The backwater channel was about 0.35 km long and 20 m wide and was subjected to more frequent connection with the river, with fully developed lotic conditions (velocities ~0.3 m s^{-1}) occurring during high river stage. This type of long-duration, pulsed connectivity is typical of the regime that persisted at the degraded site downstream of the reservoir prior to reservoir and levee construction, and fish species in this system are adapted to such conditions [*Jackson*, 2003]. However, although the stage hydrograph at site 3 was less perturbed relative to sites 1 and 2, investigations after this study began revealed that site 3 habitat was impacted by deposition of sandy sediments contaminated with the organochlorine insecticide heptachlor [*Knight et al.*, 2009b]. Probing bed sediments with metal rods revealed 1–2 m of deposition. The backwater was simply a series of small, isolated pools during periods of low river stage. Therefore, site 3 provided a hydrologic reference, but not a suitable reference for less impacted backwater water quality and ecology.

4. REHABILITATION

For rehabilitation, site 2 was modified by constructing two low weirs across the old channel. Weirs divided the backwater into two compartments: a lake cell and a wetland cell. The southern (upstream) weir was located so as to divert runoff from agricultural fields away from the lake cell. The remainder of this paper focuses on our effort to restore the lake cell as a riverine backwater and gauge progress toward that goal using the Kondolf axes representing hydrologic variability and connectivity. The wetland cell was managed using the downstream weir in order to reduce loadings of sediment, nutrients, and pesticides to the river, and results of that work have been reported elsewhere [*Lizotte et al.*, 2009; *Shields and Pearce*, 2010]. Weirs consisted of low (<2 m high) earthen embankments placed at right angles to the old river channel and covered with stone riprap. Each weir included a

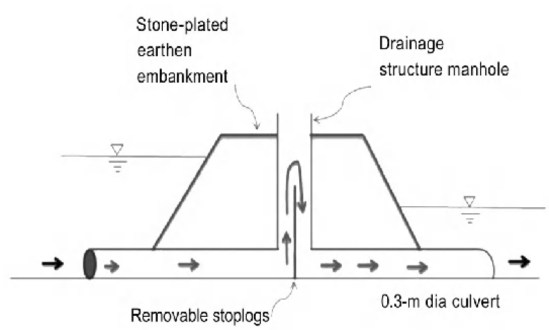

Figure 3. Schematic of weir structure.

water control structure that consisted of a 0.3 m diameter pipe that penetrated the embankment bisected by a flashboard riser "manhole" (Figure 3). Weir water control structures were operated to retain water during March–November and were opened to allow more frequent connection to the Coldwater River during December, January and February.

5. METHODS

Once daily, Coldwater River stage data for a 44 year period of record (1960–2004) were transferred from nearby gauges to the backwater sites using regression formulas between the gauge data and measurements made at the backwater mouths during this study. These data were analyzed to determine the average annual duration (or probability) of connection between the backwaters and the main channel given the site geometry found at the outset of this study, assuming stationary hydrologic conditions. In addition, backwater stage and temperature were logged at all three sites at 30 min intervals for about 18 months before and 36 months after rehabilitation of site 2 (2004–2009). Additional water quality constituents were determined for site 1 (degraded reference) during the first and final years of this period, while water quality in site 2 (rehabilitated) was sampled throughout the study period. Specifically, pH, dissolved oxygen, turbidity, and specific conductance were logged at 4 h intervals and were measured weekly using handheld meters. Grab samples were collected weekly and analyzed for solids, nutrients, and chlorophyll. Water quality loggers were placed near the apex of each of the old bends, and grab samples were collected at the same sites. Loggers were deployed so that their sensors were 0.2 to 0.6 m below the water surface; warm season vertical stratification in these waters was weak due to the shallow depths. At site 1, sensors were sometimes more deeply submerged during floods, but these occurred only during colder periods when there was no vertical stratification. At sites 1 and 2, fish were collected using a boat-mounted electroshocker at least semiannually in spring and fall. Fish were collected from each site during four, 20 min sampling periods using pulsed DC current. Because conductivities varied at the collection sites both temporally and spatially, voltages were adjusted to provide the maximum catch possible for the given conditions. All major habitats were sampled including shorelines, debris piles, and open water. At site 3, fish were sampled on two dates, 1 year apart using a backpack electroshocker due to the extremely shallow depths. Each collection consisted of one or two 20 min sampling runs depending upon the amount of surface water present such that all major habitat patches were sampled. All fish collections were processed in the same fashion. Fish were identified to species, enumerated, and measured for length, which was used to calculate weight. Weights and numbers of fish were used to calculate catch by numbers, catch by weight, catch per unit of effort, and numbers per unit of effort for each sample.

The backwater stages measured during 2004–2009 were used to compute mean depth at each measurement interval using digital elevation models based on lidar coverage of terrestrial zones and bathymetric data collected using boat-mounted echo sounders coupled with differentially corrected GPS. Mean daily values of backwater stage and mean depth were further examined using the suite of indices of hydrologic alteration proposed by *Richter et al.* [1998]. Hydrologic data from all three study sites were used to construct a two-dimensional (2-D) Kondolf diagram featuring lateral connectivity. Since water quality data were not normally distributed, nonparametric analysis of variance (ANOVA) (Kruskal-Wallis one-way ANOVA on ranks with Dunn's method for multiple comparisons) was used to compare distributions before and after rehabilitation [*Glantz*, 1992; *Systat Software, Inc.*, 2009]. Effects of rehabilitation on fish community structure similarity was examined by computing Bray-Curtis coefficients using lists of the numerical abundances of the 11 most abundant fish species from each of the backwaters [*Pegg et al.*, 2006]. For this analysis, collections from site 2 before and after rehabilitation were listed separately. Bray-Curtis values are lower for higher levels of similarity, with identical collections having values equal to 0 and collections with no species in common having coefficients equal to 1 [*Bray and Curtis*, 1957]. All abundances were fourth-root transformed prior to computation of Bray-Curtis coefficients to meet assumptions of multivariate normality and to moderate the influence of species abundance extremes. Pearson product-moment correlation coefficients were computed between key descriptors of fish collections and physical (hydrologic) variables [*Systat Software, Inc.*, 2009]. For correlation analyses, all fish samples from a given site on a given date were pooled to compute collection characteristics (number of fish, mean size of fish, percent of catch biomass composed of piscivores, etc.). Correlation coefficients were computed between these values and the mean water depth computed for that date.

6. RESULTS

Comparison of 1960–2004 once daily river stages with prerehabilitation (circa 2005) geometry indicated that sites 1 and 2 experienced backwater connection with the river channel 15% (site 1) and 12% (site 2) of the time, while the less impacted reference (site 3) was connected an average of 22% of the time (assuming static geometry circa 2005). Sites

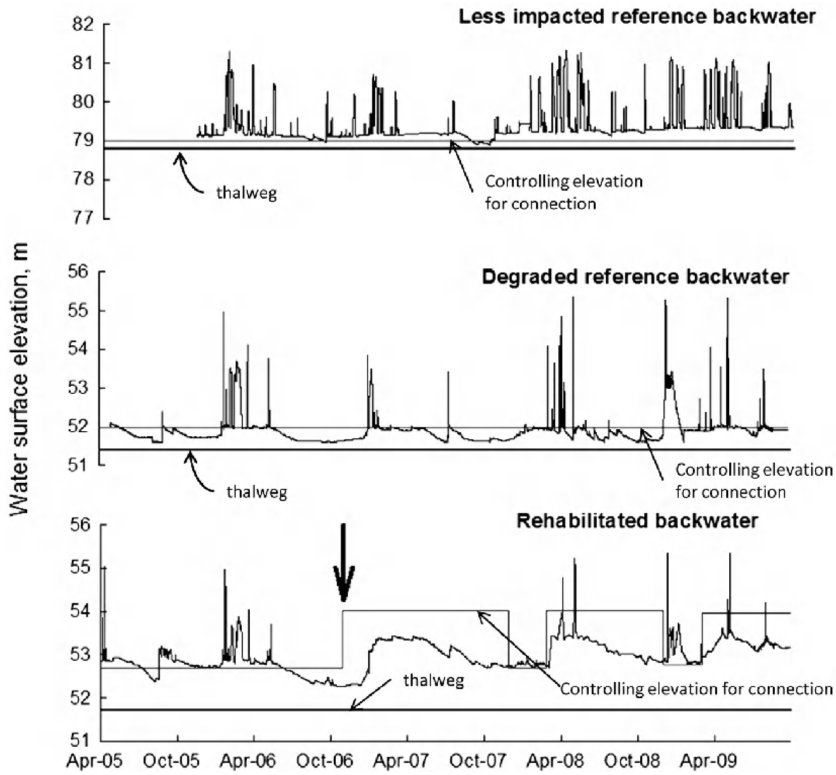

Figure 4. Stage hydrographs for less impacted reference backwaters, degraded reference backwater, and rehabilitated backwater. Vertical black arrow indicates date for completion of weir construction.

1 and 2 were connected with the river at both their upstream and downstream ends, allowing lotic conditions to develop, 2% and 3% of the time, respectively, while site 3 enjoyed such connection 10% of the time.

The actual effects of rehabilitation on backwater hydrology were measured using data we collected before and after construction of the weir in site 2 (Figure 4). Prior to rehabilitation, water depths in both degraded backwaters were extremely shallow, with monthly mean water depths generally <0.65 m. Periods with deeper water, which were driven by high river stages, were brief and limited to winter and spring. The rehabilitation weir increased dry season (summer-fall)

Table 1. Hydrologic Conditions in Study Backwaters Before and After Rehabilitation[a]

	Mean (STD) Water Depth (m)	River Connection (% of time)	Median Rise Time (m d^{-1})	Median Fall Time (m d^{-1})
Degraded reference (site 1)				
Before rehabilitation	0.53 (0.22)	5.7	0.046	0.073
After rehabilitation	0.54 (0.20)	5.5	0.193	0.087
Rehabilitated backwater (site 2)				
Before rehabilitation	0.59 (0.15)	4.4	0.028	0.073
After rehabilitation	0.69 (0.16)	2.2	0.013	0.008
Less impacted reference (site 3)				
Before rehabilitation[b]	NA	24.5	0.172	0.028
After rehabilitation	NA	55.6	0.190	0.018

[a]Median rise and fall times were computed using software package Indices of Hydrologic Alteration [*Richter et al.*, 1998]. NA indicates not applicable.

[b]These values are based on shorter period of record (only one water year, 2006) than the site 1 and site 2 “before rehabilitation” entries, which were based on two water years (2005–2006).

water depths there by 0.15 to 0.30 m, while conditions in site 1 remained unchanged (Table 1). Dry season extreme lows were greatly moderated by the presence of the weirs in site 2 (Table 2).

In general, the weir moderated stage fluctuations and made hydrologic conditions less variable (Figure 4). The high stage rise rate (median of all positive differences between consecutive mean water depths that exceeded base stage elevation) decreased 50%, while the fall rate decreased by an order of magnitude (Table 1). Rise and fall rates for the treated site 2 were similar to those for the degraded reference backwater (site 1) before rehabilitation, but an order of magnitude smaller afterward. Weir placement made stage variability at site 2 more similar to the less impacted reference site 3 during the postrehabilitation period.

Weir placement reduced connectivity between the backwater and river channel (Table 1). The degraded backwater, site 1, was hydraulically connected to the river about 6% of the time during the period of observation. The rehabilitated backwater, site 2, was connected about 4% of the time prior to weir construction, but only 2% following weir placement. These values are far lower than those observed at the less impacted site 3, which was connected to the river about one fourth of the time during the water year immediately prior to rehabilitation of site 2 and more than half the time during the three water years following rehabilitation. The ecological importance of the connection of lateral habitats to main channels is a function of timing as well as the duration of such connections. Native organisms are adapted to a hydrograph dominated by the Lower Mississippi River, which features regular high stages during the December–May timeframe, with highest stages in April [*Baker et al.*, 1991]. Prior to construction of flood control levees and dams, this regime likely produced flooding of low-lying areas across the alluvial plain containing our sites. Seasonality of connection frequency for our sites followed these trends before and after rehabilitation, although the fraction of time connection occurred tended to be low for the degraded and rehabilitated sites relative to site 3 (Figure 5).

Table 2. Medians of Annual Extreme Mean Depths (m)[a]

	30 Day Minimum	90 Day Minimum
Degraded reference backwater (site 1)	0.37	0.39
Site 2 before rehabilitation	0.39	0.45
Site 2 after rehabilitation	0.54	0.55

[a]Values computed using software package Indices of Hydrologic Alteration [*Richter et al.*, 1998].

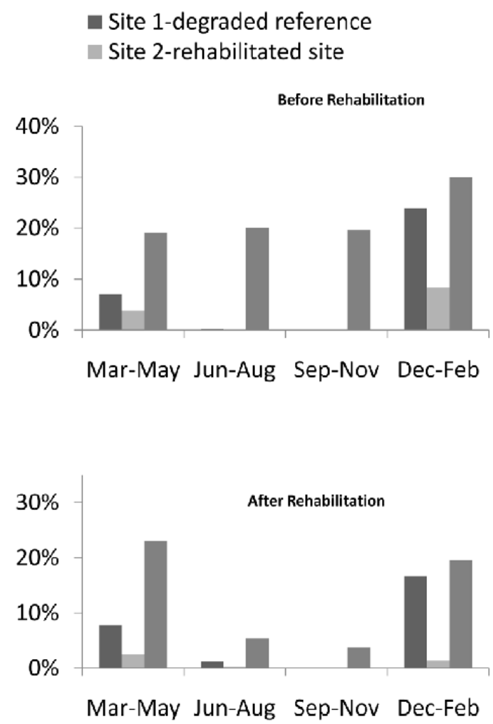

Figure 5. Percent of time backwaters were connected to main channel by quarter before (2005–2006) and after (2007–2009) rehabilitation.

In order to gauge the effects of rehabilitation, a 2-D Kondolf diagram was constructed using the less impacted reference site 3 as a predegradation condition, site 1 as an indicator of degraded status, and the postrehabilitation conditions at site 2 (Figure 6). The fraction of time that the backwaters were hydraulically connected to the river channel was used as a measure of connectivity [*Heiler et al.*, 1995], while the median rate of stage change during the falling limbs of high stage events was adopted as a measure of variability. Sites 1 and 2, both degraded backwaters, plotted very close to each other prior to rehabilitation. Since the weirs reduced connectivity and moderated the flashy stage hydrographs, rehabilitation made the treated site 2 plot closer to the less impacted reference (site 3) on the variability (x) axis, but translated it farther away from the target condition on the connectivity (y) axis.

Prior to weir placement, water quality conditions in the two degraded backwaters were similar, except dissolved oxygen and chlorophyll *a* were lower, and total N was greater, in the degraded reference, site 1 (Table 3). These differences were likely due to the heavy mat of floating duckweed (*Lemna* sp.) that covered the water surface in the degraded reference site during all but the coldest months.

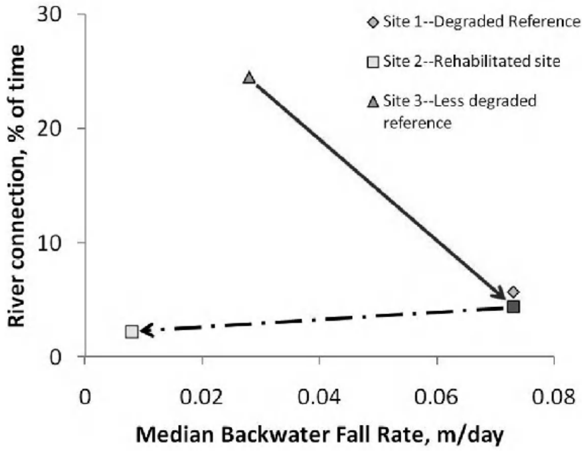

Figure 6. Kondolf diagram for study backwater. Since there were no predegradation data for the rehabilitation site 2, the upstream site 3 was used as a less impacted reference. Rehabilitation translated the status of the treated site away from the degraded reference and toward the less impacted reference on the variability axis (median fall rate for high stage events) but had the opposite effect on connectivity (y axis).

Weir placement transformed site 2 water quality, making it less similar to the degraded site 1. Diversion of agricultural runoff away from the lake cell in site 2 (Figure 2) resulted in reductions in turbidity and suspended solids of about 70%, while nutrient levels were 30% to 60% lower. Accordingly, chlorophyll *a* values were about half as great after weir placement. Summer diurnal fluctuations in temperature and dissolved oxygen were moderated by greater depths produced by the weir, but maximum temperatures (Figure 7) and minimum dissolved oxygen levels were not (Figure 8). The less impacted site continued to experience maximum summer temperatures that were about 5°C cooler than the other two sites following rehabilitation.

The rehabilitated backwater was sampled for fish on 11 occasions over the course of the study, with a total effort of 818 min of electrofishing, producing 2523 fish representing 32 species with a total mass of 259 kg. The degraded reference site yielded 402 fish representing 19 species with a total mass of 20 kg when sampled on two dates with a total effort of 65 min. The less impacted reference site was sampled twice, but yielded only eight individuals of two species total, likely due to the aforementioned insecticide contamination [*Knight et al.*, 2009b]. Two species, *Ictiobus bubalus* and *Lepisosteus oculatus* composed 53% of the biomass from the rehabilitated site 2 and 66% of the biomass from the degraded reference site 1. Fish populations in both backwaters appeared relatively insensitive to antecedent connection to the river but were influenced by mean water depth (Table 4). Greater depths in the treated backwater were associated with larger fish, more fish species, and a shift in species composition from planktivores to piscivores (Figure 9). When all fish collections from sites 1 and 2 were considered, dominance (as percent of sample biomass) of the top predator, *Micropterus salmoides*, was positively correlated with mean water depth, while the tolerant insectivore, *Lepomis humilis*, and the planktivore, *Dorosoma cepedianum*, were negatively correlated with mean water depth (Table 4). Thus, as site 2 water depth increased following rehabilitation, its fish assemblage became less similar to site 1. The Bray-Curtis dissimilarity coefficient between

Table 3. Medians for Mean Water Depth and Selected Water Quality Variables From Degraded Reference Backwater and Rehabilitated Backwater Before and After Addition of Weirs[a]

Variable	Degraded Reference Backwater (Site 1)	Rehabilitated Backwater (Site 2)	
		Before Weir	After Weir
Mean depth on days when samples were collected (m)	0.56*	0.60*	0.71**
pH	6.8*	6.7*	6.0**
Dissolved oxygen (mg L^{-1})	4.2*	5.4**	6.2**
Secchi disk depth (cm)		21*	40**
Turbidity (NTU)	38*	51*	16**
Suspended solids (mg L^{-1})	40*	60*	17**
Total P (mg L^{-1})	1.18*	0.77*	0.33**
Filterable P (mg L^{-1})	0.061*	0.052*	0.041**
NH_3– (mg L^{-1})	0.001*	0.012*	0.009*
Total N (mg L^{-1})	1.056*	0.132**	0.092***
Chlorophyll *a* (μg L^{-1})	23*	78**	39***

[a]Medians with different superscripts are significantly different ($p < 0.05$, Dunn's method for multiple comparisons, Kruskal-Wallis analysis of variance (ANOVA) on ranks). Boldface variable names indicate significant differences in the rehabilitated backwater before and after rehabilitation.

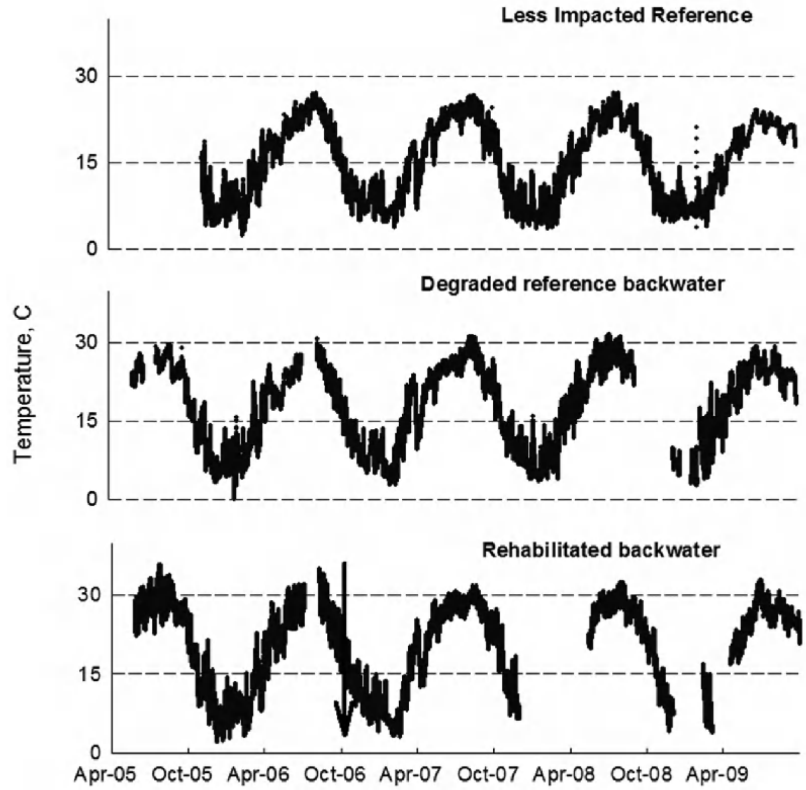

Figure 7. Water temperature for less impacted reference backwater, degraded reference backwater, and rehabilitated backwater. Vertical black arrow indicates date for completion of weir construction.

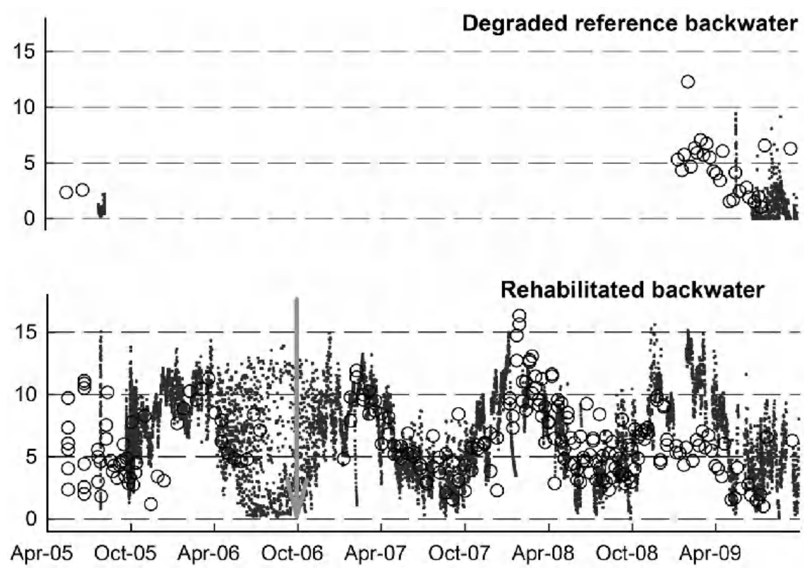

Figure 8. Dissolved oxygen concentrations for degraded reference backwater and rehabilitated backwater. Open circles are weekly measurements using handheld meter, while solid symbols are values from loggers collected at 4 h intervals. Vertical arrow indicates date for completion of weir construction.

Table 4. Pearson Correlation Coefficients *r* Between Descriptors of Electrofishing Samples From the Rehabilitated Backwater and Key Physical Variables[a]

	Days With Hydraulic Connection to River		
	During Previous 6 Months	During Previous 3 Months	Mean Water Depth
Number of fish species	0.139 (0.684)	0.329 (0.323)	**0.579 (0.062)**
Mean fish size (g)	−0.287 (0.391)	−0.209 (0.537)	0.501 (0.116)
Catch biomass as piscivores (%)	−0.132 (0.698)	0.117 (0.733)	0.512 (0.107)
Catch biomass as planktivores (%)	−0.161 (0.599)	−0.170 (0.616)	**−0.589 (0.057)**
Catch biomass as *Micropterus salmoides* (%)	0.233 (0.491)	0.417 (0.202)	0.515 (0.105)
Catch biomass as *Lepomis humilis* (%)	0.311 (0.351)	0.156 (0.647)	**−0.817 (0.002)**
Catch biomass as *Dorosoma cepadium* (%)	−0.081 (0.813)	−0.170 (0.618)	**−0.592 (0.055)**

[a]Numbers in parentheses are p values. Values in boldface indicate $p < 0.10$.

sites 1 and 2 was 0.28 prior to rehabilitation but 0.13 afterward (Table 5).

7. DISCUSSION AND CONCLUSIONS

Future development of stream corridors should adopt an ecological engineering paradigm [*Mitsch and Jørgensen*, 2004] that manages ecosystems for the totality of services they can provide. Since cutoff bends and other types of floodplain backwaters are common along large, lowland rivers, these areas merit special attention [*Zalewski*, 2006]. Cutoff bends may be managed using a combination of water control/flow diversion techniques [*Shields et al.*, 2005]. Key questions regarding the design of these measures have to do with the timing and duration of flow connection with the main channel [*Shields et al.*, 2009]. Alternative designs may be evaluated by comparing the level of main channel connectivity and hydrologic variability they produce relative to degraded and least impacted sites [*Kondolf et al.*, 2006].

Installation and operation of a low weir in the degraded cutoff bend described here reduced main channel connectivity and stage variation relative to the preconstruction and degraded reference site conditions. Observed chemical and biological changes were evidently related to moderating temporal hydrologic variations by increasing dry season water depths by about 0.15 m and by diverting agricultural runoff from about 350 ha of cultivated fields. In general, water quality improved as solids and nutrient concentrations declined. Others have reported floodplain lake quality improvements following diversion of polluted runoff [*Cooper*, 1993; *Filipek et al.*, 1993; *Cooper et al.*, 1995]. Water quality impairment has been directly linked to shallow depths in floodplain lakes within this region due to coincident problems associated with nutrient enrichment and biochemical oxygen

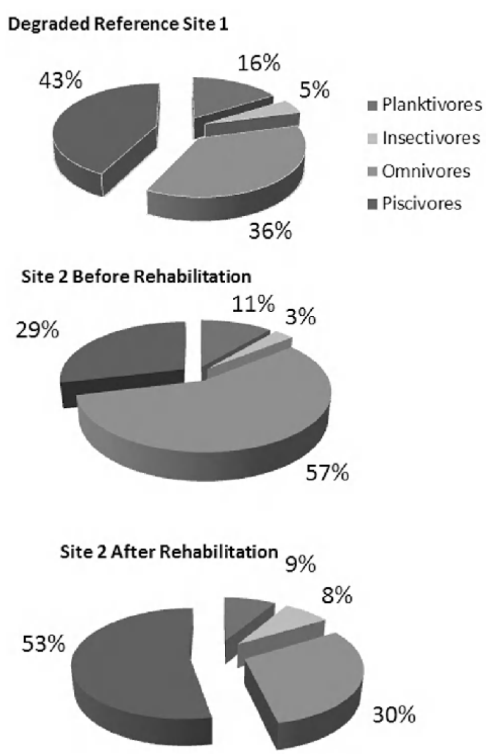

Figure 9. Trophic structure of fish assemblages in study backwater sites 1 and 2.

Table 5. Bray-Curtis Dissimilarity Coefficients for Sites 1 and 2 Based on Abundances of the 11 Most Abundant Fish Species

	Degraded Reference Backwater (Site 1)	Site 2 Before Rehabilitation	Site 2 After Rehabilitation
Degraded reference backwater (site 1)	0.00		
Site 2 before rehabilitation	0.13	0.00	
Site 2 after rehabilitation	0.28	0.16	0.00

demand from allochthonous organic matter [*Miranda and Lucas*, 2004]. In addition, shallow depths are more susceptible to increased turbidity from wind action and bottom-feeding fishes and to DO depletion by benthic respiration. Reduced depth means there is less oxygen in the water column to support such respiration [*Miranda et al.*, 2001] and less water to absorb incident solar energy. Shallow lakes in this region often experience wide diurnal swings in temperature, DO, and pH during warmer months [*Justus*, 2006].

Despite the continued problems with low DO in summer at the rehabilitated site 2, fish responded to greater depth and reduced variability in a fashion similar to that reported by *Miranda and Lucas* [2004] based on a study of 11 oxbow lakes along the Mississippi River. Degraded backwaters supported assemblages dominated by, "species that thrive in turbid, shallow systems with few predators and low oxygen content." Fish assemblages in the treated site 2 trended away from those typical of shallow, small systems studied by *Miranda and Lucas* [2004], but did not shift toward assemblages Miranda found in highly connected backwaters. Water level stabilization using a levee controlling an Illinois River backwater produced a shift in fish species community structure similar to the one reported here [*Pegg et al.*, 2006].

The backwater rehabilitation project described here had three main shortcomings. First, it failed to fully address the problem of hypoxia during warmer months. Evidently, the cyclic hypoxia reflects the high level of nutrient enrichment and attendant algal activity common to shallow backwater systems in cultivated floodplains in this region [*Miranda et al.*, 2001; *Justus*, 2006]. Others have reported anoxia in riverine backwaters and have suggested that these conditions may be ameliorated by introducing flow from the river through the backwater [e.g., *Theiling*, 1995]. In fact, we were able to produce dramatic water quality improvements in site 2 by pumping a modest amount of water from the adjacent river into the backwater for about 4 weeks early in the prerehabilitation period [*Cooper et al.*, 2006]. The second main failing of the rehabilitation project was its adverse effect on lateral connectivity between the backwater and the river main stem. The less impacted reference site was connected to the river more than half of the time in the postrehabilitation period, while the rehabilitated site enjoyed connection only about 2% of the time. Although many attest to the importance of connectivity as a determinant of backwater fish community structure [e.g., *Valdez and Wick*, 1981; *Grift et al.*, 2001; *Lusk et al.*, 2003; *Penczak et al.*, 2004; *Miranda*, 2005] and perhaps the value of the backwater as nursery habitat for river species [*Csoboth and Garvey*, 2008], the level of connectivity needed to produce a given level of ecological benefits is unknown. Others have reported fish migrating over and through water control structures to access floodplain backwaters [*Schultz et al.*, 2007; *Csoboth and Garvey*, 2008]. Third and finally, questions arise regarding the sustainability of restoration efforts like ours. The weirs we constructed create more favorable hydrologic variation based on observations of the reference sites, and diversion of polluted agricultural runoff will slow degradation of the rehabilitated site. Nevertheless, over the long term, this area will continue to experience sedimentation and eutrophication even if at a reduced rate. Full recovery of floodplain ecosystem services will require manipulation of main channel flows [*Theiling*, 1995] and floodplain vegetation and topography [*Baptist et al.*, 2004].

Rehabilitation research is challenging due to the complexity of natural ecosystems and our inability to replicate these systems or isolate the influences of key variables. This study attempted to use a before-after-control-impact approach to assess rehabilitation effects with the "control" role filled by a degraded and a less degraded site. However, our efforts were hampered by lack of resources required to study the degraded site in as great a detail as the rehabilitated site. Furthermore, the site selected as "less degraded" proved to be a poor biological reference, perhaps due to toxic residues [*Knight et al.*, 2009a, 2009b], demonstrating yet again how hard it is to find suitable reference sites for studies such as this. The Kondolf method was a useful tool in assessing physical performance of our project, but selection of the most ecologically appropriate measures of variability and connectivity is a key step. More work is needed to refine the use of the Kondolf approach for aquatic ecosystem evaluation.

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Quantitatively Evaluating Restoration Scenarios for Rivers With Recreational Flow Releases

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Rivers worldwide have been drastically altered in terms of hydrology and morphology. To date, restoration has focused heavily on channel design (i.e., channel morphology), despite the well-known importance of flow regime on controlling ecological processes. In addition, there are few quantitative tools with which to directly and quantitatively compare alternative restoration scenarios. Here we leverage the ecological dominant discharge approach to develop a simple method for quantitatively comparing the effect of different restoration strategies on an ecological metric, in-stream hydraulic habitat (i.e., limiting depths and velocities of flow for fish). We apply this approach to the case of a river ecosystem impacted by recreational flow releases. The analysis shows that enhancing channel morphology will result in limited ecological gains and that greater habitat could be gained through changes in the hydrologic regime. Greater quantitative tools are needed for directly comparing the potential ecological gains of different restoration approaches, and these tools should be usable during the initial evaluation stages of a restoration project; our approach provides a starting point in this direction.

1. INTRODUCTION

1.1. Quantitative Analysis of Design Alternatives

Rivers worldwide have been modified both geomorphically and hydrologically [*Costa et al.*, 1995]. Geomorphic modifications can be considered either direct, such as channelization [*Brookes*, 1988], or indirect, such as those changes that often follow urbanization [*Chin and Gregory*, 2005]. Hydrologic changes range from moderate alterations of hydrographs via increases in watershed imperviousness to fundamental alterations in the quantity and timing of flows through dams, reservoirs, and associated with irrigation, hydropower, and flood control [*Poff et al.*, 1997]. There is a vast literature on the ecological and societal ramifications of alterations of channel morphology and the hydrologic regime, and well-documented impacts include the loss of species, altered biogeochemical cycles, loss of riparian vegetation, increased flood frequency, and degraded water quality [*Downs and Gregory*, 2004].

Because of these impacts, there is growing interest in the potential to restore the natural ecological processes in streams, and there is a rapidly growing literature on stream restoration as a science [*Roni et al.*, 2008] and as an industry [*Bernhardt et al.*, 2005]. Considerable research has focused on designing channels that are geomorphically stable, with several alternative approaches to such stability-centric design

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[*Shields et al.*, 2003]. The bulk of stream restoration effort to date has been focused on restoring or recreating channel forms (i.e., geomorphology) that existed prior to anthropogenic impacts. That is, stream restoration science to date has focused heavily on geomorphology.

There are two limitations to previous studies of stream restoration designs and alternatives. First, many previous studies evaluate the ecological response to a change in geomorphology, yet hydrologic regime is as often perturbed as the geomorphic template. Indeed, if hydrologic change is the driver of the geomorphic change [*Booth*, 1990], then it is unclear whether geomorphic restoration alone could be considered a viable approach to restoration. As well, much ecological research has shown that ecosystems can be as driven by hydrologic change alone as by physical channel changes [*Poff et al.*, 1997; *Doyle et al.*, 2005].

A second limitation is more pragmatic: in evaluating restoration designs, there are few methods with which design alternatives can be quantitatively evaluated and compared. During most restoration projects, early stages of the design process include development of alternatives, which are evaluated by some oversight group (e.g., property owner, funding or regulatory agency) before final designs and construction are begun. Alternatives generally include options such as no action, bank stabilization only, and complete re-meandering and restoration. Comparing these alternatives currently relies on estimates of their varying degree of geomorphic stability, some estimates of effects on ecology (usually very qualitative), and changes to flood stages. Cumulative geomorphic or ecological metrics from which different design alternatives can be directly and quantitatively compared are mostly lacking. That is, it is difficult to specifically answer the question, "If we use channel design option A instead of B, what is our percentage gain in ecological restoration?" In the absence of such directly comparable metrics, options tend to be evaluated on the most easily developed and compared metric: cost, even though the goal of the project is ecological restoration.

This puts a river restoration design team in the awkward position of having insufficient tools with which to gauge the potential success of alternative proposed actions. As importantly, for agencies or funders of restoration programs, it precludes the potential to develop quantitative approaches through which alternative design scenarios can be developed.

1.2. Recreational Flow Releases

River morphology and ecology are clearly impacted by land development, power generation, and other land use change or industrial activities. Yet over the past few decades, an emerging concern is over the use of rivers for recreation, specifically recreational flow releases [*Whittaker et al.*, 2005]. Recreational flow releases are used to facilitate activities such as whitewater rafting or fishing and are a relatively new consideration for river managers. Recreational flow releases are problematic because, like flows released during hydropower operations, the release schedule can be vastly different from the natural flow regime to which downstream ecosystems have adapted.

Based on our experiences in the northeastern United States, management decisions about recreational flow releases are being made in a vacuum of data or understanding of local system properties and processes. River and environmental management agencies are aware of the lack of data and understanding, and several have initiated studies specific to recreational flow releases [e.g., *Baldigo et al.*, 2010]. In lieu of available studies, managers currently rely on the vast literature on "natural flow regime" which has provided the foundation for understanding the effect of flow alteration on river ecosystems [*Poff et al.*, 1997], as well as relatively simple tools to assess the statistical magnitude of flow changes [*Richter et al.*, 1996].

Previous approaches, however, provide limited quantitative or predictive abilities. Specifically, such approaches do not allow quantitatively evaluating (1) whether altered flow regime is the ecologically limiting factor or (2) how much restoration of the flow regime would be needed to restore specific functions of the ecosystem. That is, in a recreational flow release situation, the critical question for managers is "How much restoration of the natural flow regime is required to restore specific ecological processes?" In addition to questions of hydrologic restoration, similar questions arise about whether geomorphic restoration could accomplish comparable levels of restoration. That is, "Can the same ecological restoration be accomplished via geomorphic restoration rather than hydrologic restoration?"

For rivers targeted for restoration, environmental managers are left in a conundrum of whether to target restoring hydrologic regime, which affects the local recreational industry, or restoring channel morphology, which can be very expensive with unknown benefits. In cases where flow quantity and timing are critical to local industry (e.g., hydropower, navigation, irrigation), then geomorphic-focused restoration could be a preferable approach to gaining ecological recovery, or at least mitigating some of the effects of flow releases. Alternatively, in tightly constrained landscapes, such as urban regions or levee-constrained rivers, geomorphic modification may be impossible, and so flow manipulation may be a preferable avenue to ecological recovery.

There are currently few quantitative tools available with which to evaluate these alternative approaches to restoration. One approach that was recently used was de-coupling hydrology from geomorphology in a staged analysis of hydraulic habitat availability for endangered species on the Lower Missouri River [*Jacobson and Galat*, 2005]. This approach provided a critical step in that it enabled river managers to see that altering channel morphology could not make up for the loss of function provided by the predisturbance flow regime. That is, to recover the habitat for a specific species of interest, no amount of channel restoration could make up for what was needed in flow restoration.

One limitation of the Jacobson-Galat approach was that it only provided a look at a snapshot in time in terms of discharge. Many management decisions must be made over many years, and so an additional step should be to develop a time-integrative approach to analyzing the contributing effects of hydrology and geomorphology on an ecological variable like hydraulic habitat. Here we do this by leveraging the concept and analytical approach of ecological dominant discharge.

1.3. Ecological Dominant Discharge: Application to Habitat Availability and Restoration

The concept and analytical approach of dominant discharge is a cornerstone of geomorphology and, with it, natural channel design in river restoration [*Shields et al.*, 2003]. Briefly, the dominant or "channel forming discharge" concept suggests that alluvial channels adjust over long periods of time to accommodate a discharge of certain magnitude and frequency [*Wolman and Miller*, 1960], and there are a series of approaches to quantify measures of channel forming discharge (see review by *Doyle et al.* [2007]).

More recently, *Doyle et al.* [2005] proposed extending the dominant discharge concept to ecological processes and applied the approach to metrics ranging from organic matter loads to nutrient spiraling. In this approach, the ecological effective discharge is that discharge which, over time, maximizes a specific ecological process. For example, if a particular river has a range of flows from 1 to 1000 $m^3\ s^{-1}$, then consider the question, "At what discharge is primary productivity maximized?" To answer this question, we must consider both the rate of primary productivity at each discharge between 1 and 1000 $m^3\ s^{-1}$, but we must also evaluate how frequently and for what duration each discharge occurs over time. It is this balance of frequency and magnitude that determines the relative effectiveness of a particular discharge on primary productivity.

For calculation, the long-term ecological effectiveness of a discharge of particular magnitude is the product of the effect of that flow multiplied by its duration of occurrence (Figure 1). A flow duration curve is created using discharge records ($f(Q)$). Also needed is an ecological rating curve, or the value of the ecological process as a function of discharge, across the range of discharges at the river. The product of the

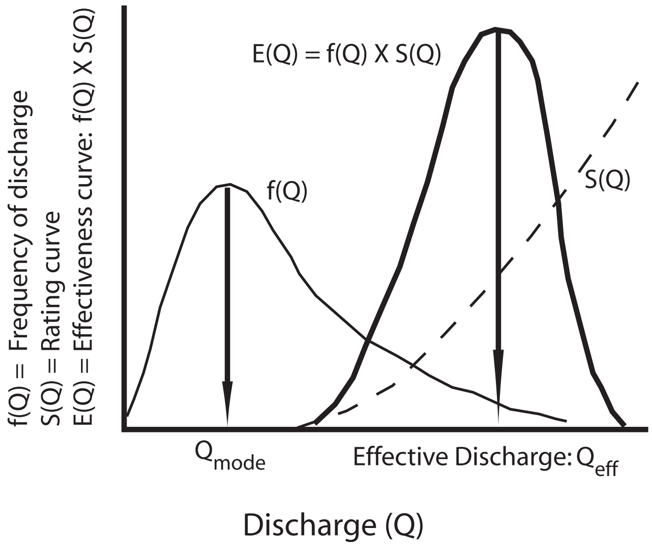

Figure 1. Components of effective discharge (Q_{eff}). Adapted from *Wolman and Miller* [1960]. Note that for the present analysis of habitat, the shape of $S(Q)$ will be quite different from that shown above (see Figure 7 below).

hydrologic frequency curve and the ecological rating curve produces the effectiveness curve, the peak of which is the ecological effective discharge, Q_{eff}. The integral of the Q_{eff} curve (curve $E(Q)$ in Figure 1) is the expected value of the function, or the total hydraulic habitat, H_{tot}. That is, for a given flow regime and a given ecological rating curve, the H_{tot} gives the cumulative expected average condition for the entire time period of interest. While Q_{eff} provides a relative metric of ecological effectiveness, H_{tot} provides an integrative metric that accounts for the relative contributions of hydrology and ecological response to flow. For evaluating the relative effects of hydrology versus ecological response on total stream ecosystem condition, the H_{tot} provides a quantitatively analytical method with which to synthesize the available information into a single metric.

The primary utility of the H_{tot} analytical approach is that it lends itself to sensitivity analysis: the contributing parts of the analysis can be de-coupled and then manipulated individually in order to analyze their relative effects on cumulative ecological conditions. For analyzing flow releases, we are interested in de-coupling hydrology from geomorphology with respect to their individual contributions to some ecological response variable. To consider the potential importance of restoration, we can then individually analyze the relative contribution of restoring hydrology versus the contribution of restoring geomorphology and determine which is more efficient.

There are numerous ecological variables which could be analyzed using the Q_{eff} and H_{tot} approach. We chose to

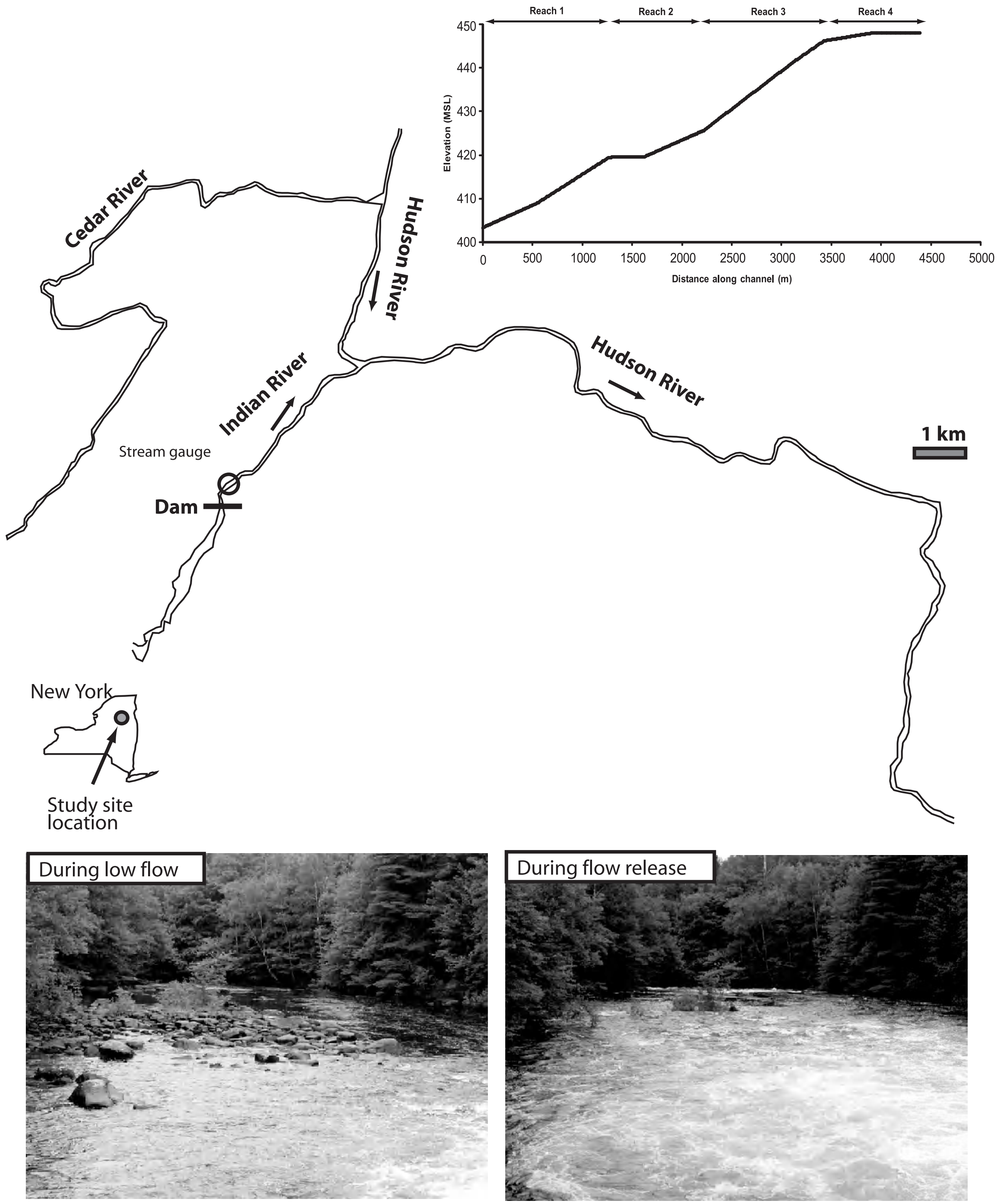
Reach 1
Reach 2
Reach 3
Reach 4
Elevation (MSL)
450
440
430
420
410
400
0
500
1000
1500
2000
2500
3000
3500
4000
4500
5000
Distance along channel (m)
Cedar River
Hudson River
Indian River
Hudson River
1 km
Stream gauge
Dam
New York
Study site location
During low flow
During flow release

Figure 2

analyze the relative effects of hydrology and geomorphology on hydraulic habitat availability. The effect of discharge on hydraulically defined habitat availability is a widespread approach used in environmental flow analysis [*Acreman and Dunbar*, 2004]. Typically, such studies employ hydraulic models to assess the distribution of depth and velocity as a function of discharge. Ecosystem responses in terms of fish population or other ecological community metrics are then related empirically to indices calculated from physical habitat variables. Through this approach, quantitative inventories of physical habitat known to be important for specific biota are used as a surrogate for direct quantification of population dynamics. We used a very simplified version of this approach to develop the Q_{eff} and H_{tot} approach, and to illustrate its potential utility in analyzing alternative restoration approaches.

Here we analyze the relative contributions of hydrology and geomorphology to cumulative, time-integrated hydraulic habitat availability using the Q_{eff} and H_{tot} quantitative analysis approach. We do this using a river subjected to recreational flow releases that facilitates a whitewater rafting industry, an increasingly common management issue. We use a sensitivity analysis to simulate the potential effects of restoring hydrology or enhancing geomorphic complexity on the river as an example for how the approach can be used to guide channel design alternatives evaluation and river management decisions.

2. METHODS

We sought to use the analysis tools that are most readily available and widely used among river restoration designers to facilitate the potential application of this method. While other analysis models can provide important additional information, we instead used those tools that can provide the most efficient entry-level analysis; we note throughout the methods below, and discussion, the limitations imposed by our approach, and the additional analysis that could be conducted in the future. The primary limitation to our approach was that we used standard stream surveying techniques and standard 1-D (one-dimensional) hydraulic modeling approaches rather than multidimensional modeling. This influenced and limited some aspects of our analysis and resolution of the ecological variable, habitat. Yet the gains in broader applicability and relative rapidity of analysis merited this approach in this introduction to the concept.

2.1. Study Site

The study was conducted in the Indian River in the Adirondack Mountains, New York. The Indian River is a mixed bedrock and alluvial river. It is geomorphically moribund, largely unable to geomorphically adjust to contemporary changes in hydrology or sediment regime. The 4.5 km river segment in which we worked is made up of three distinct geomorphic reaches (Figure 2): two high-gradient reaches (slope ~0.02 – 0.03), and two lower-gradient reaches (slope ~0.003). The width of the channel ranges from 20 to 50 m, with substrata consisting of boulders, cobbles, and gravel. Regional hydrology is driven by snowmelt peak flows in April or May, low flow in July and August interrupted by local thunderstorms, but also large flows via frontal systems often associated with tropical depressions in the early autumn.

Along the 4.5 km segment on the Indian River between the Abanakee Dam and its confluence with the Hudson River, recreational flow releases elevate discharge regularly throughout the summer months, creating extremely high frequency, short duration disturbances. These releases increase discharge from a base flow of 2.5–4.0 to 35–43 $\mathrm{m^3\ s^{-1}}$ (Figure 3); flow during the releases are close to the geomorphic bankfull discharge in the segment, but less than observed peak discharges at the site (e.g., 97 $\mathrm{m^3\ s^{-1}}$ on 29 June 2006). Flow releases last from 90 to 120 min and occur at 10:00 AM at least four days a week (Tuesday, Thursday, Saturday, and Sunday) from April to October during the whitewater rafting season. When upstream reservoirs are full during the autumn, additional flow releases can occur through November and into December as well. Recreational flow releases 5 days a week are common in the late spring and early summer when there is sufficient water in the upstream reservoir. Depth in the river rises from 30–50 cm at base flow to >1 m during a release. Boats cannot be floated downstream during the summer low-flow, nor can a person wade the channel (i.e., fishermen) during high flow releases. A U.S. Geological Survey stream gauge was installed immediately downstream of the dam, providing 15 min data discharge data.

Within this region, non-native brown trout (*Salmo trutta*) are a fish species of interest for management, primarily for recreational purposes. They are stocked in the Indian and other rivers of the region, and there is an active trout fishing tourism industry in the area. As the whitewater rafting

Figure 2. (opposite) Site location map and longitudinal profile (inset) for study reach. Study reach is on Indian River between the dam and the confluence with the Hudson River. For longitudinal profile, note four different slopes along reach for reach 1 (downstream 1.3 km, i.e., RK 0–1.3), low slope of reach 2 (RK 1.3–2.2), high slope of reach 3 (RK 2.2–3.6), and low slope of reach 4 (RK 3.6 to the dam). Stream gauges from which hydrology was developed were on upper Hudson, lower Hudson, and Cedar Rivers, all adjacent to the study site.

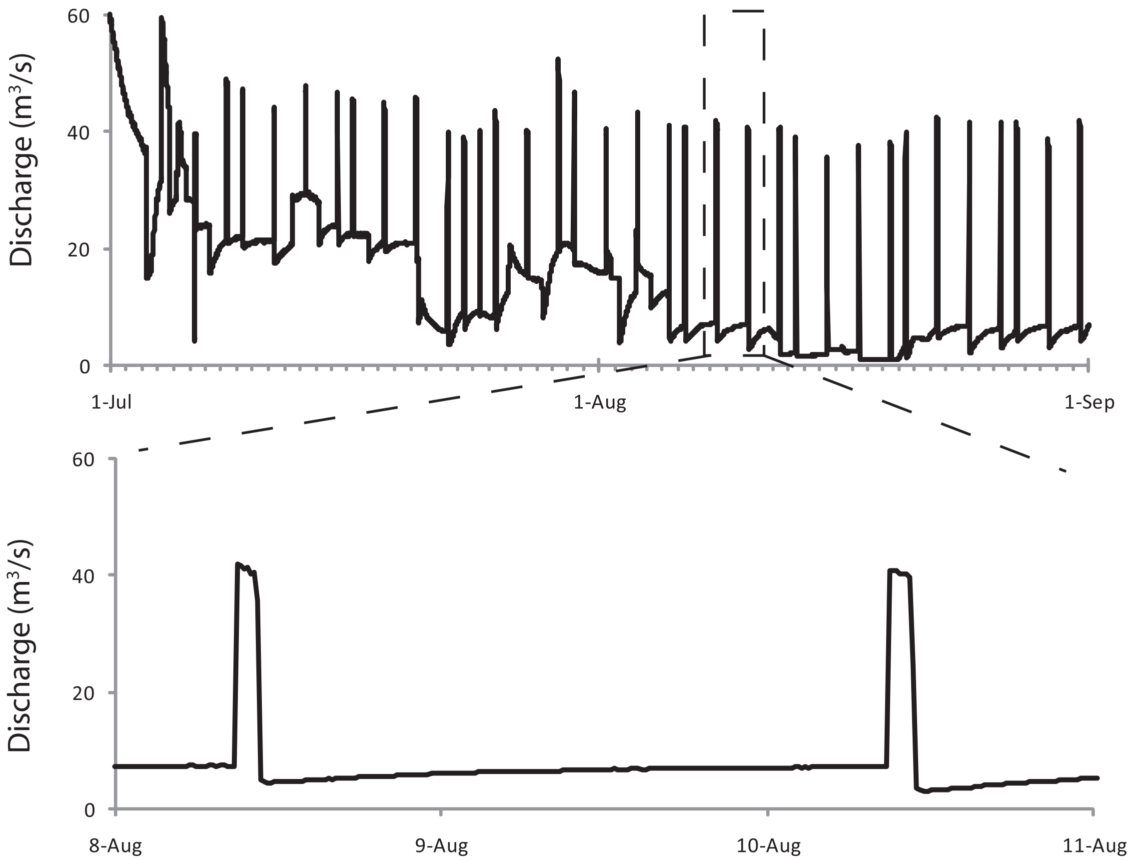

Figure 3. Hydrograph at Indian River subject to recreational flow releases. (top) River discharge through recreational flow release months. (bottom) A typical several day sequence of flow on the Indian River flood hydrograph.

industry has grown over the past decade, the New York Department of Environmental Conservation has become interested in the potential environmental impacts of the flow releases within the Indian River and the downstream reaches of Upper Hudson River. Because of the economic benefits of the whitewater rafting, there is keen interest in mitigating any effects of the recreational flow releases, i.e., the potential to eliminate the flow releases altogether is limited. The effects of the flow releases on the downstream ecosystems, including trout communities, are mostly unknown, but multiple studies are now ongoing [*Fuller et al.*, 2011; *Boisvert*, 2008].

2.2. Field Surveys and Hydraulic Modeling of Existing and Restoration Scenarios

We surveyed a series of channel cross-sections and a longitudinal profile using standard surveying techniques and instruments (total station instrument) [*Harrelson et al.*, 1994]. We also quantified bed material grain size distributions using Wolman pebble counts with $n > 100$ [*Wolman*, 1954]. We used the surveys to construct a 1-D step-backwater hydraulic model using the U.S. Army Corps of Engineers Hydrologic Engineering Center River Analysis System (HEC-RAS) modeling software [*U.S. Army Corps of Engineers*, 2008]. The U.S. Geological Survey (USGS) stream gauge, in addition to frequent and multiple water surface elevation observations along the river segment during different flows, allowed us to calibrate the roughness coefficients.

The current Indian River channel is essentially devoid of instream woody debris jams along the 4.5 km study reach; most jams that form are removed by rafting guides. Other rivers in the region in natural conservation areas are replete with woody debris jams [*Kraft and Warren*, 2003]. Woody debris jams on these rivers provide substantial hydraulic habitat [*Manners et al.*, 2007], and thus were likely important hydraulic habitat features for the stream ecosystems in the past.

In conceptualizing restoring the river, our hypothetical goal was enhancing habitat availability for a specific fish species. Thus, we were not so much analyzing the ability to restore to some historic condition channel and ecosystem condition as much as we were analyzing the ability to ecologically enhance specific habitat conditions. Thus, we considered proposed changes in morphology to be "enhancements" of morphology rather than restoration. For hydrology, however, we did consider an actual restoration scenario, i.e., converting the flow regime back to a historic condition, in addition to hydrologic regime changes specifically targeted at increasing habitat.

We considered several alternative morphology enhancement scenarios. The more common approach of re-meandering was not feasible because the river lies within a geologically constrained valley, restricting any potential lateral manipulations of the channel geometry. Also, because of the lack of hydraulic habitat along the study segment, the approach we thought most likely would be enhancement via construction of woody debris jams [*Abbe and Montgomery*, 1996]. Using debris jams to create or increase habitat is a potentially low-cost method of enhancement, particularly when large wood is locally available, as it is in this region of the Adirondacks.

We sought to simulate the effect of introducing woody debris jams in the reach by including flow-blocking structures in the HEC-RAS hydraulic model. We assumed that the debris structures would span one-third of the channel width, and extend vertically to the top of the channel bank elevation. These structures would thus create low velocity and high depth habitat areas upstream and downstream of the jams, but high velocity areas immediately adjacent to the jams [*Abbe and Montgomery*, 1996]. A jam of these approximate dimensions along this study segment, the only one of such size, was studied by *Manners et al.* [2007]. It created usable habitat for brown trout and was also negotiable by the rafters. These assumptions and the corresponding hydraulic model gave us simulated enhanced channel conditions.

The simulated habitat enhancement structures were placed in the model only in the high-gradient reaches at the upper and lower ends of the segment, as these were the reaches where velocity and depth were most limiting for habitat. We simulated enhanced conditions for ~1 km in each of these reaches. For sensitivity analysis, we simulated two scenarios: (1) partial channel enhancement in which only the upper reach (reach 3) was manipulated and (2) full channel enhancement in which both the upper and lower reaches were manipulated (reaches 1 and 3).

2.3. Hydrologic Analysis and Modeling

USGS stream gauge 15 min data were used to develop a frequency distribution of flows (i.e., flow duration curve) for the Indian River. This frequency distribution of current conditions represents the existing regulated hydrology of the site. Only data for the months of April–September were used, as these are the months in which flow is manipulated for the rafting industry and when habitat availability is of greatest concern for brown trout. This is referred to as the "existing-regulated" hydrology scenario (Figure 4).

The additional scenarios we sought to analyze were to develop an estimate of what the hydrology of the site would be like without the upstream dam and associated flow regulation, i.e., restored hydrology scenarios. We analyzed the flow frequency distribution from four nearby, unregulated rivers (Figure 4) and developed simple statistical representations of these data [*Stedinger et al.*, 1992] and using the scaling approximation approach of *Mueller and Pitlick* [2005]. This approach provided an estimate of flow frequency distribution for the Indian River if the hydrology was not regulated. We refer to this as the "restored hydrology" scenario (Figure 4) because it is an estimate of what restoration to historic or previous conditions would be.

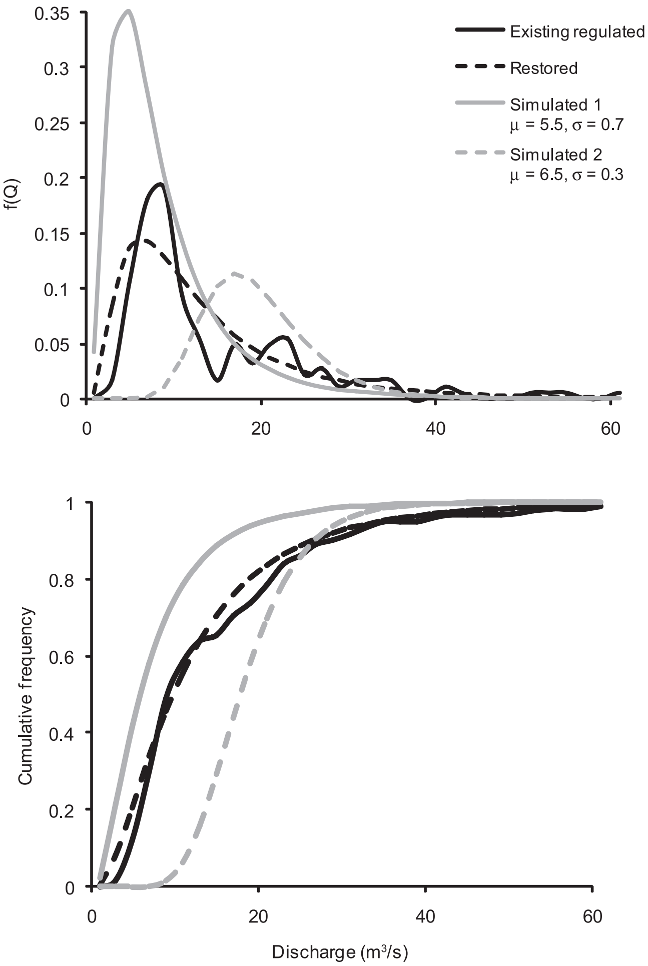

Figure 4. Flow distribution functions for existing regulated hydrology, restored hydrology, and the two simulated flow conditions based on stream gauge analysis of adjacent rivers.

Similar to how we analyzed alternative scenarios for the channel morphology enhancements, we also sought to develop alternative scenarios for hydrology. To do this, we simulated simple flow frequency distributions using a log-normal distribution of flows, which captures the primary features of near-modal flow frequency distribution, which are primary concern for habitat.

$$f(Q) = \frac{1}{Q\sigma\sqrt{2\pi}} e^{-\frac{(\ln Q-\mu)^2}{2\sigma^2}}, \qquad (1)$$

where μ and σ are the mean and the standard deviation of the logarithm of the discharge, respectively. We used this to generate a simple flow frequency distribution with which to estimate the potential impacts of alternative flow scenarios on increasing habitat, and these are the "simulated" flow regimes (Figure 4). It is important to note that the log-normal distribution alone would be inappropriate if large, rare flows were an issue of concern [*Stedinger et al.*, 1992].

2.4. Effective Discharge Analysis of Habitat Suitability

We used well-established curves to calculate habitat suitability and weighted useable area over a range of flows at each station along the river [*Bovee*, 1982; *Parasiewicz and Dunbar*, 2001]. We also analyzed two different life history stages (juveniles and adult). Habitat suitability curves are generally based on frequency of use of different microhabitats and are standardized to a maximum of 1 (Figure 5). While this is an extremely simplified ecological metric, the advantage of this approach is that it can be readily adapted to other fish species, as well as other ecological variables, thus developing quantitative relationships with flow. This also allows for rapid analysis of broad trends across watersheds [e.g., *Rosenfeld et al.*, 2007] and rapid assessments of management implications of decisions [e.g., *Brown and Pasternack*, 2009].

The simulated hydraulic habitat scenarios were analyzed to calculate a total habitat availability, H_{tot}, for the different channel morphology and hydrology scenarios. This approach allowed us to calculate a single quantitative metric, which represented the combination of geomorphic and hydrologic contributions to hydraulic habitat. It is in this single, collapsed, cumulative metric that the benefits of our approach lie, as the single metric allows more direct, and quantitative comparisons of hydrologic-geomorphic alternatives. This single metric then allows additional analysis, such as cost comparisons, that are not as directly feasible with other, existing analytical methods of habitat restoration.

For each combination of geomorphic and hydrologic conditions, the hydraulic model was run for discharges ranging

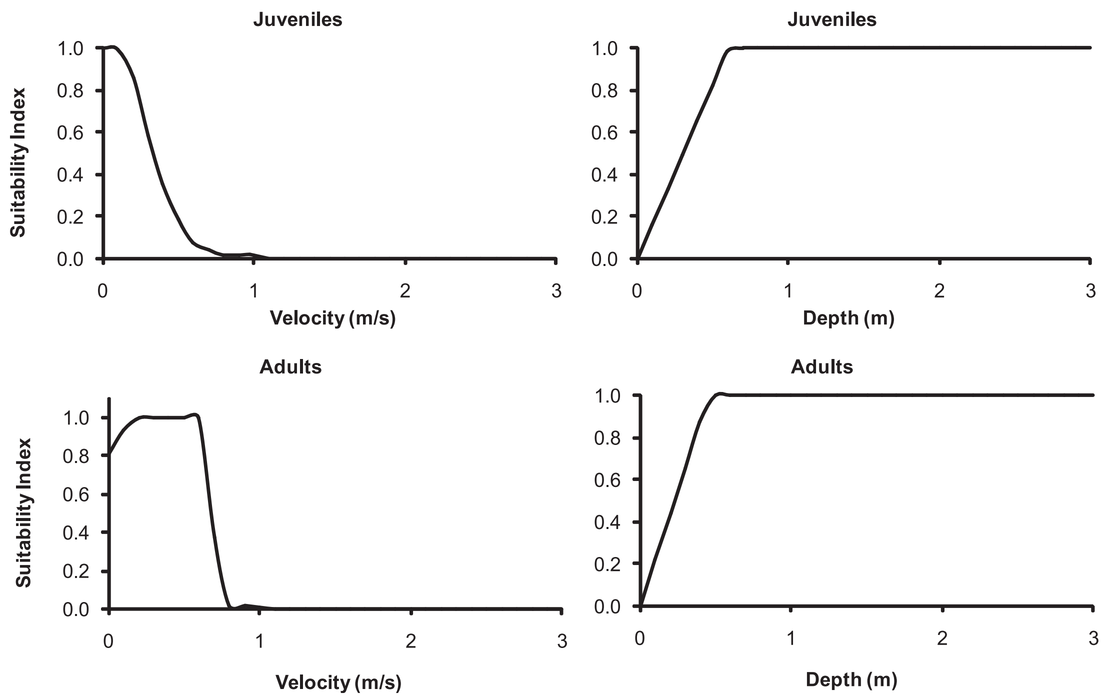

Figure 5. Depth and velocity habitat suitability curves for different life history stages of brown trout (adapted from *Bovee*, 1982).

from 1 to 99 $m^3\ s^{-1}$, which represented the range of discharges at the site. The hydraulic model was used to quantify the average depth and velocity of flow at each of the cross-sections at each of the simulated discharges. For a particular discharge, flow and depth results from the model were combined with the habitat suitability index to calculate a combined habitat suitability index, which was then weighted by the channel area. This product gave the total channel area along the study segment of the Indian River suitable for brown trout for a particular discharge.

We ran this simulation for each discharge to produce a curve of total hydraulic habitat availability as a function of discharge, thus producing a habitat-discharge rating curve (curve $S(Q)$ in Figure 1). This curve is what is needed to conduct the ecological effective discharge analysis. The habitat-discharge rating curve was then multiplied by the flow frequency curve (curve $f(Q)$ in Figure 1), the result of which quantified the ecological effectiveness curve (curve $E(Q)$ in Figure 1). Integrating the effectiveness curve gave H_{tot} or the expected value of the function. This value can be interpreted as the expected habitat availability for the particular combination of geomorphology and hydrology. That is, over time, for the particular channel morphology and the particular expected distribution of flow frequencies, the H_{tot} is the statistically expected total value of available habitat. We conducted this analysis to calculate H_{tot} for each of the simulated geomorphic or hydrologic scenarios and compared the results of the different conditions by directly comparing the expected value function H_{tot}.

3. RESULTS

Relationships of hydraulic modeling were evaluated in terms of the relations between discharge and habitat area in the existing and hypothetically enhanced morphology and by examining the availability of habitat under regulated, restored, and other discharge flow regimes. These results are summarized in the metric of H_{tot}, the expected value of total habitat area, calculated using the ecological effective discharge analysis approach.

3.1. Hydraulic Conditions

Valley-scale geomorphic conditions along the study segment of the Indian River created four reaches with distinct hydraulic conditions. Reaches 2 (from RK 1.3 to 2.2; RK is river kilometers upstream from confluence) and 4 (RK 3.6 to the dam) were relatively lower-gradient than reaches 1 and 3 (Figure 2), and the lower-gradient created greater depths and lower velocities (Figure 6, left). These depths and velocities were mostly outside the limits of brown trout usability, particularly for juveniles (Figure 5). Under existing channel conditions, flow velocities in these steep reaches, even at very low discharges (~1 $m^3\ s^{-1}$), exceeded 0.5 $m\ s^{-1}$, the approximate threshold for suitability for juvenile brown trout. Because of these hydraulic conditions, there was essentially no suitable habitat in reaches 1 or 3 for juveniles, even at low flow (Figure 6, bottom). For adults, whose threshold flow velocity was closer to 0.8 $m\ s^{-1}$, flows near 9 $m^3\ s^{-1}$ created velocities unsuitable in the high-gradient reaches.

In the lower-gradient reaches 2 and 4, flow velocities were reduced and depths increased, and conditions were suitable for juveniles. However, this was the case only for low discharges, <11 $m^3\ s^{-1}$; (Figure 6). The only habitat available for most discharges for the existing channel morphology was in reach 2 and in the upstream portions of reach 4, and this was only the case for discharges <11 $m^3\ s^{-1}$ for juveniles and 15 $m^3\ s^{-1}$ for adults.

For the enhanced channel morphology conditions, flow depths and velocities oscillated among the restored debris jams, as the woody debris jams reduced flows immediately upstream of the jams but increased velocities and decreased depths immediately adjacent to and downstream of the jams (Figure 6, right). The increase in available habitat area was limited to the area immediately upstream of the debris jam where flows could be slowed down by the backwater effects of the debris jams. This backwater region was, however, relatively limited, but did create small patches of suitable habitat for adults and juveniles.

This change in morphology is more clearly represented by the change in relationship between habitat availability and discharge, the ecological rating curve (see curve $S(Q)$ in Figure 1, Figure 7): more habitat was available across a wider range of discharges in the area of channel immediately upstream of the debris jams than was available with the existing channel morphology.

3.2. Effective Discharge Analysis

The modal discharge of the existing (regulated) hydrology regime was 9 $m^3\ s^{-1}$, with a secondary mode at 22 $m^3\ s^{-1}$, and several flows >20 m^3/s that occurred somewhat frequently because of the recreational flow releases (Figure 4). The restored hydrology had a modal discharge of 6 $m^3\ s^{-1}$, which occurred 15% of the time, and there were few flows <20 $m^3\ s^{-1}$. The two simulated flow regimes contrasted greatly in that the Simulated-1 regime was dominated by lower flows, while the Simulated-2 regime had more frequent large flows, with modal discharges of 5 and 17 $m^3\ s^{-1}$, respectively; over time, the same quantity of water was passed through the river with each distribution.

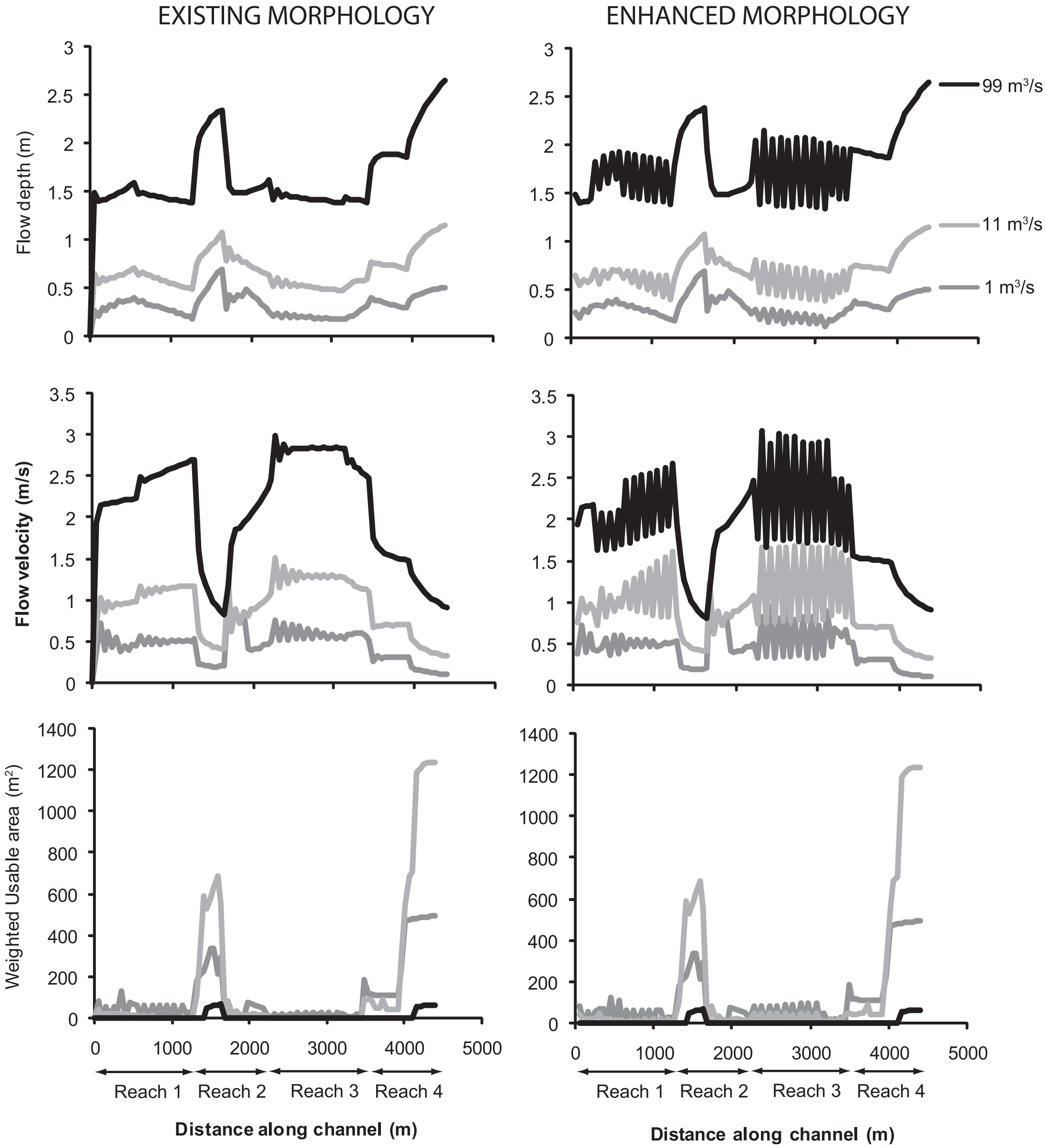

Figure 6. Distribution of flow velocities, depths, and habitat availability along Indian River under (left) existing morphology and (right) enhanced morphology. All results presented here are for juveniles only.

The Q_{eff} analysis combined these hydrologic frequency distributions with the ecological rating curves. For instance, under existing conditions, there was some usable habitat available at cross-sections in the steep reaches for flows up to 5 $m^3\ s^{-1}$ (Figure 7, existing morphology). However, under existing flow conditions, the majority of time was at flows >5 $m^3\ s^{-1}$ (Figure 4); as such, over time for existing flow and existing channel conditions, essentially no habitat was available through time along the steep reaches (Figure 8a). However, when considering restored flow conditions or Simulated 1 flow conditions, there were greater periods of time at flows <5 $m^3\ s^{-1}$; there was thus some usable area through time under existing channel conditions, but only under these alternative flow conditions at this cross-section. When channel morphology is enhanced, greater portions of the study site have lower velocities, which when combined with the existing or restored flow conditions, produces greater areas of usable habitat (Figure 8b).

When all cross-sectional hydraulic habitat availability was quantified and then weighted by the flow frequency

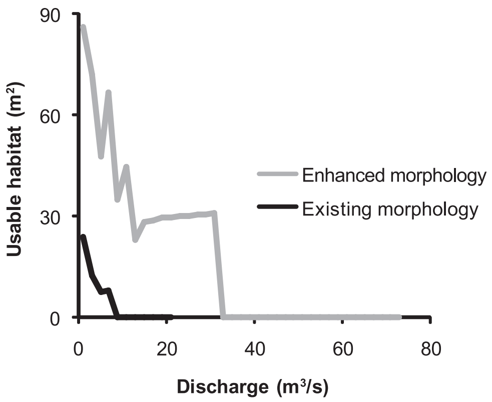

Figure 7. Habitat-rating curve. Effect of flow on usable habitat area for existing and enhanced morphology in a high-gradient reach of the Indian River (RK 2520). Note the increased habitat availability for enhanced morphology conditions for discharges between 10 and 30 m^3 s^{-1}.

distribution to calculate the Q_{eff}, overall results showed that the small additions in hydraulic habitat for the enhanced morphology up to 10 m^3 s^{-1} (e.g., Figure 4) became substantial because these habitat additions came during low flows, and it was these low flows that were very frequent under existing flow conditions. For instance, under existing hydrologic conditions, introducing debris jams in reaches 1 and 3 increased the available habitat from ~1 to >110 m^2 in the area immediately upstream of the location of each debris jams (for adults only; juveniles had very modest increases in habitat availability).

Collapsing all of the Q_{eff} analysis into the single metric of H_{tot} showed the relative contributions of each geomorphic and hydrologic change to expected total habitat availability over the entire study segment and integrated for an annual expected condition. For existing channel conditions (i.e., no restoration), restoring natural flow regime would increase time-averaged habitat by 17% for adults and 22% for juveniles (Figure 9). Much greater changes were possible through the simulated flow regime 1, in which case H_{tot} could be increased by a factor of 2.4 for adults and 2.8 for juveniles. The amount of habitat available under the flow regime 2 was substantially reduced for existing channel conditions.

Surprisingly, changes in channel morphology provided only marginal increases to total expected habitat, H_{tot}. Enhancing the morphology of both reaches 1 and 3, but retaining current hydrology, increased expected habitat availability by 20% for adults, but did not increase habitat at all for juveniles. Enhancing channel morphology and restoring hydrology to preregulated conditions increased habitat by 30%. However, it is important to note that restoring hydrology alone resulted in comparable cumulative habitat increases. Enhancing morphology and manipulating hydrology to those conditions in Simulated-1 would almost triple available habitat, but again, most of this gain was through hydrology effects rather than geomorphic effects.

4. DISCUSSION

4.1. Uses and Limitations of Approach

River ecosystems are sensitive to many abiotic drivers, of which hydrology and geomorphology are critical. Here, using a newly developed analytical technique, we show that time-integrated hydraulic habitat availability was more sensitive to changes to flow regime, and less sensitive to changes in morphology. However, the changes driven by flow regime

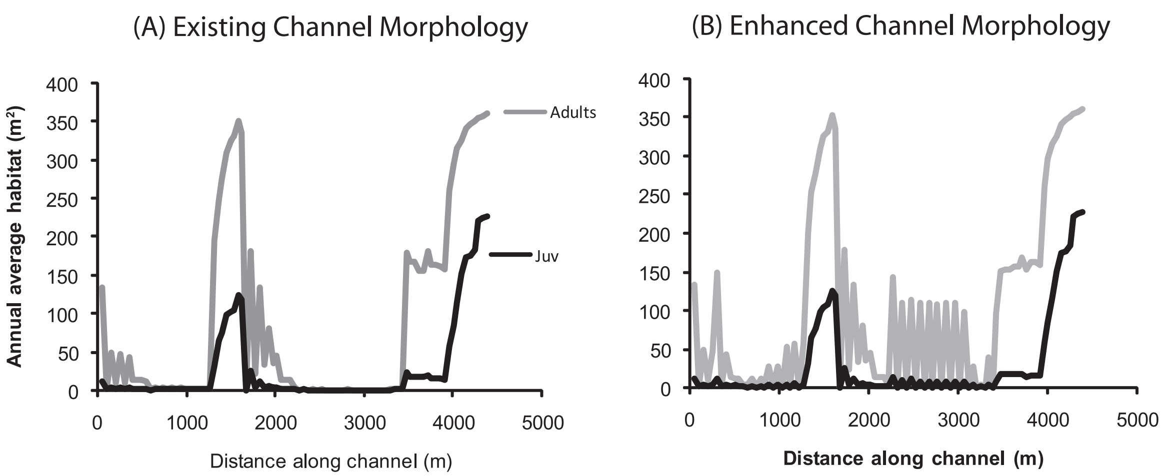

Figure 8. Time-integrated, expected value of annual average habitat availability for (a) existing and (b) enhanced channel morphologies. These curves are spatially distributed but time-integrated values of habitat availability.

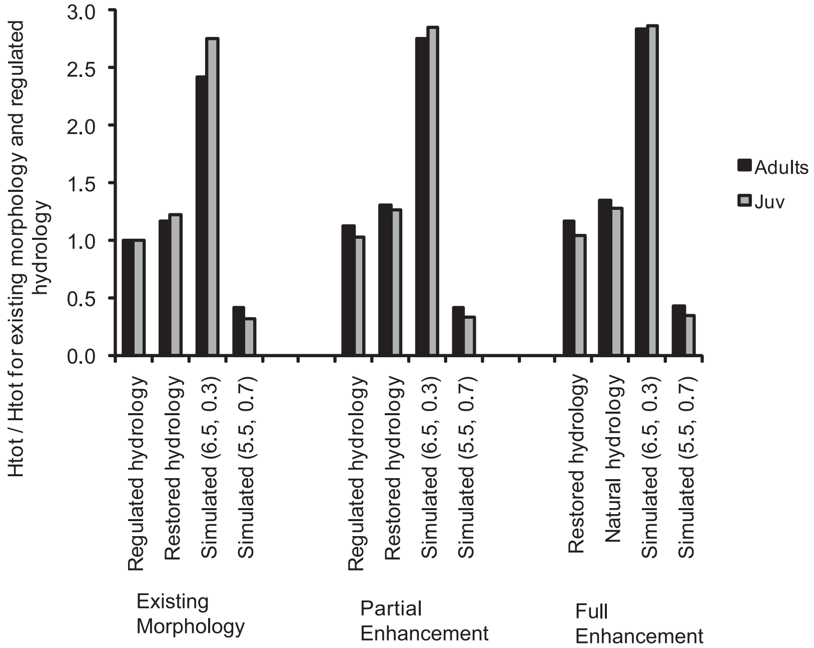

Figure 9. Values of H_{tot} for existing conditions of hydrology and morphology compared to H_{tot} values for all other restoration scenarios.

were for large flow regime changes; when constrained to restoration of natural flow regime, only slight gains in habitat could be made.

The purpose of our modeling approach was to have an easily understood and a readily adaptable structure. In this sense, it can be considered a "fast-and-frugal" modeling approach (in the sense of *Carpenter* [2003]). Our results demonstrate the utility of analytical techniques that allow decoupling abiotic drivers of hydraulic habitat. Previous studies of hydraulic habitat availability have primarily focused on existing channel conditions and potential changes in flow manipulation [e.g., *Richter et al.*, 1996]. Relatively few, but recent studies have examined alternative channel scenarios in addition to alternative flow scenarios, most notably the work of *Jacobson and Galat* [2005].

The advantages of the additional analysis of Q_{eff} as done here is that the final metric, H_{tot}, allows a single quantitative metric by which the different scenarios can be directly compared. For instance, rather than relying on the assumption that restoring hydrology will improve conditions for brown trout, our analysis showed that restoring hydrology to predam conditions will increase habitat availability for adult brown trout by <10%; a small gain for a decision with potentially large economic costs.

Yet we also fully acknowledge the many limitations of our current approach. For studying habitat, reliance on a 1-D model was problematic as has been indicated by more thorough analysis of other rivers [*Brown and Pasternack*, 2009]; clearly, moving toward more detailed, multidimensional modeling would increase the accuracy of the approach. Our application of a 1-D model to represent flow associated with debris jams is a key limitation, as our previous studies of this type of jam on this river shows that small patches of habitat are created that are not well captured by the 1-D assumption [*Manners et al.*, 2007].

More generally, users of this approach should be cautious, lest it be overused or used inappropriately; the Q_{eff} and H_{tot} captures a single ecological metric among a range of critically important factors limiting stream ecosystems. Other studies at the field site have demonstrated the importance of thermal habitat for brown trout at the site, and the complexity of ecological response to the flow releases [*Boisvert*, 2008; *Fuller et al.*, 2011]. Also, many other ecological metrics could have been used [e.g., see *Doyle et al.*, 2005, Table 1 and Figure 13], and just within flow controls on organisms like fish, there are numerous other aspects of the flow regime that control organism behavior [*Poff et al.*, 1997], and sequences of flows as disturbing events are particularly important [*Fisher et al.*, 1982].

The manipulation of hydrologic regime, as modeled here, is realistic in some rivers, but not in others. The ability to release large floods will be limited by infrastructure such as size of gates on the dam, as well as the presence/absence of upstream flow-regulating dams. Manipulating hydrology will not be feasible on unregulated rivers or on those more heavily constrained by flow restrictions. This is, in part, why we began our analysis on a river affected by recreational flow releases. Also, we have assumed that hydrology and geomorphology are dynamically decoupled, i.e., changes in hydrologic regime do not directly impact geomorphic drivers of habitat. In our case of a glaciated, geomorphically moribund channel [see *Fuller et al.*, 2011], this assumption is valid. But in other geomorphically dynamic systems, a more coupled analytical approach would be required.

Despite these limitations, our simple analysis of habitat is used as a method of quantifying the specific tradeoffs in river restoration and management decisions; including the numerous other influences on stream ecosystems would make such quantification impossible, or at least prohibitively difficult in most situations. Our analysis represents a simple, yet potentially important first step in quantitatively evaluating alternative scenarios for restoration of rivers where both channel morphology and flow hydrology are known to be disturbed.

4.2. Use of Approach for Cost-Benefit Analysis and Recreational Habitat Estimates

Beyond the issues of habitat restoration, we also sought to explore the uses of this approach for the specific issue of recreational flow releases, and there were two other aspects of this issue that we sought to understand. First, we sought to provide some sense of cost approximations for context to the restoration or enhancement actions. Second, we considered other potential applications of this approach to recreational flow release analysis in general.

For our cost approximation, we attempted to estimate costs of different decisions to the nearest order of magnitude, noting that true costs of such decisions contain significant amounts of information than that contained here [e.g., *Sanders et al.*, 1990]. We started by assuming that a very rough cost of a single woody debris jam would be $10,000 (M. Brunfelt, Inter-Fluve, Inc., personal communication, September 2009). For rough estimate of cost of design, permitting, and installation of the jams for partial restoration, we estimated $150,000 for partial enhancement and $250,000 for full enhancement (a simple assumption of economy of scale for the restoration of the second reach).

We also made gross approximations of the effects of flow manipulation scenarios on revenues. While now somewhat out of date, the NY DEC in their Unit Management Plan in 2002 estimated that 10,000–12,000 people raft this segment of river per year, and that costs were, on average, $80 per rafter, or on the order of $1,000,000 per year in revenues due to the flow releases for generating whitewater rafting industry. Under current flow scenario, there are 4 days per week of rafting during the peak flow release summer season. For simplicity, we assumed that, on average over the course of a summer, restoring the flow to predam conditions would result in the loss of 2 days of rafting per week, flow scenario 1 in the loss of 3 days per week, and scenario 2 in the gain of 3 days per week (i.e., rafting on all days of the week during the summer months).

Under these simple assumptions, the H_{tot} approach allowed us to directly compare rough cost estimates of each restoration scenario with the gains in hydraulic habitat. For instance, keeping the existing channel conditions but restoring hydrology would cut rafting revenues by almost 30%, but increase habitat by <10%. Our habitat analysis showed that the greatest ecological gains came under flow scenario 1, which would more than double available habitat. Yet this scenario could reduce revenues by almost 50% under our cost assumptions. Insightfully is that including channel

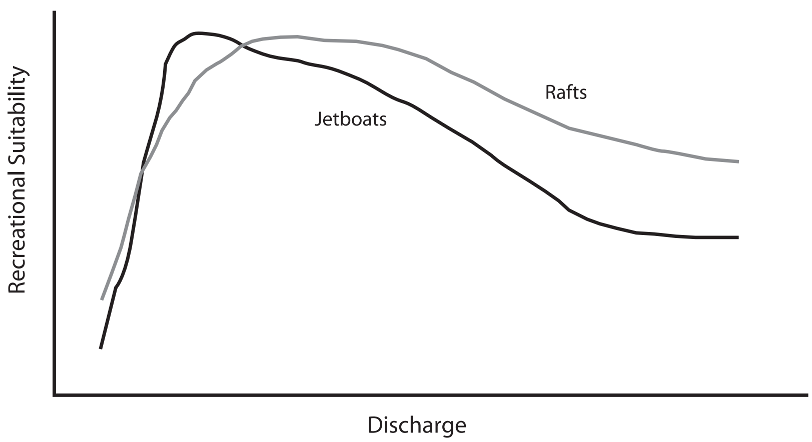

Figure 10. Recreation suitability curves for rafting and jet boating. Adapted from results of *Whittaker and Shelby* [2002] for rafting industry surveys downstream of Hells Canyon Dam, Snake River.

restoration rather than just flow restoration greatly increases the costs, but only marginally increases habitat availability.

These results indicate how the Q_{eff} and H_{tot} analysis can increase the ability to quantitatively compare the gains and losses of restoration as would be done for cost estimates in scenario building as part of project planning. Yet it is clear that hydraulic habitat and rafting are not the only considerations for this and other rivers subjected to recreational flow releases. Most notably, fishability is a key concern on many rivers like the Indian River, and the Q_{eff} and H_{tot} approach could also be useful in analyzing for this problem or for other types of recreation on a range of rivers. For instance, *Whittaker and Shelby* [2002] developed an approach to evaluate the effects of flow releases on different types of recreational boating, using rafts or jet boats. They used surveys and observations at various discharge to develop curves from which to evaluate the suitability of different flows for different recreational activities (e.g., Figure 10). These curves are, essentially, "recreational rating curves," analogous to curve $S(Q)$ in Figure 1. It would be straight forward to continue the Q_{eff} and H_{tot} analysis to include total, time-averaged "recreational habitat" in addition to actual fish habitat as part of restoration scenario building. Depending on the recreational activity of interest, specific curves could be developed based on surveys or could be approximated based on previous studies. These additional analyses would provide a greater quantitative basis for specific river management decisions in balancing ecological needs, recreational demands, and costs.

5. CONCLUSIONS

The effect of flow on geomorphic and ecological processes is well known. We developed a method to quantitatively evaluate the potential effects of restoration scenarios, particularly comparing the contribution of geomorphic enhancement versus hydrologic restoration and manipulation. Previously developed approaches for analyzing flow manipulation on ecological communities or processes are largely statistical or descriptive and do not allow for direct comparison to the end-result of restoration quantitatively. We suspect that in lieu of direct quantitative metrics, costs will drive decisions. While cost may drive decisions even when quantitative, cumulative metrics are available, at least some sense of the magnitude of tradeoffs among scenarios will be better understood. The proposed method and metric here, while limited, is a step in advancing quantitative comparisons and will hopefully serve as a starting point for further refinement.

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Sediment Source Fingerprinting (Tracing) and Sediment Budgets as Tools in Targeting River and Watershed Restoration Programs

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Management strategies that focus on stream and watershed restoration with the goal of reducing erosion and sediment flux should be founded on a sound understanding of the sediment sources contributing to the sediment flux and the overall sediment budget of the watershed in question. This understanding can be provided by undertaking complementary sediment source fingerprinting and sediment-budgeting investigations. The sediment fingerprinting approach quantifies the relative importance of the potential sediment sources in a watershed. The sediment budget approach provides information on the magnitude of the fluxes and the links between sources, sinks, and sediment output. Combining the two approaches can provide resource managers with information on where to target measures to reduce erosion, sediment delivery, and the net transport of sediment.

1. INTRODUCTION

Worldwide, sediment is increasingly recognized as a major pollutant in aquatic systems [*Walling*, 1995, 1999]. In many cases, the sediment of concern is fine-grained silts and clays [*Walling and Moorehead*, 1989]. The transport of fine sediment by a stream will increase its turbidity and reduce light penetration, and the deposition of this sediment can reduce reservoir capacity and impair water supply intakes as well as degrade aquatic habitats [*Mahmood*, 1987; *Phillips*, 2002; *Ruether et al.*, 2005]. Furthermore, because of the importance of fine sediment as a vector for the transport of nutrients and organic contaminants, such as pesticides, reduction of sediment mobilization and stream sediment loads is often critical to reducing contaminant and nutrient transport [*Walling et al.*, 2003a; *Schoellhamer et al.*, 2007; *Lick*, 2008]. Watershed restoration, which involves rehabilitating the stream corridor and/or upslope areas outside the stream corridor (hereinafter referred to as "upslope areas"), is often used as a management strategy to reduce sediment loadings and improve aquatic habitats [*Shields and Knight*, 2004; *Rosgen*, 2006]. Restoration of upslope areas may include, but not be limited to, reduction of storm runoff, grazing management, gully control, and vegetation plantings [*Federal Interagency Stream Restoration Working Group*, 1998; *Poesen et al.*, 2003; *Evans et al.*, 2006; *Randle et al.*, 2006; *Natural Resources Conservation Service* (*NRCS*), 2007]. Stream corridor restoration may involve engineering of the channel form, vegetation planting, flow manipulation, and bank stabilization [*Federal Interagency Stream Restoration Working Group*, 1998; *Shields et al.*, 2003; *Moerke and Lamberti*, 2004; *Randle et al.*, 2006; *NRCS*, 2007].

In watersheds where excess sediment has been identified as a problem, design of an effective restoration project for reducing sediment flux requires the consideration of a

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number of key questions, such as (1) What are the main sources of the sediment of concern? (2) Where are these sources located? (3) What are the pathways and sinks associated with the movement of sediment through the watershed? (4) Is the sediment problem an upslope area problem or a stream corridor problem?

Answers to the first two questions provide important information for targeting restoration measures to the key sediment sources in a watershed, but it is also important to consider the third question in order to be able to understand the links between sediment mobilization and sediment output from a watershed and, thus, the likely effectiveness of controlling sediment mobilization in reducing the downstream sediment flux. If extensive erosion occurs in the upslope areas of a watershed, successful control of that erosion might have only a limited effect on the downstream sediment flux, if much of the mobilized sediment was previously deposited within the watershed and therefore did not reach the watershed outlet. Results of this kind also highlight the need to manage these sediment storage sites so that future remobilization of sediment does not occur. The fourth question is important because strategies to reduce sediment flux may differ markedly based on whether the concern is sediment derived from the channel or upslope areas. This point was highlighted by *Pess et al.* [2003, p.186] who stated, "... repairing excessive stream bank erosion along a downstream reach may be a waste of valuable resources if altered upstream or upslope conditions remain conducive to accelerated surface runoff, streamflows, and sediment production rates." Conversely, if only a small proportion of the sediment mobilized from upslope areas reaches the channel system, sediment mobilized from the channel is likely to represent the main source of the sediment reaching the watershed outlet, and management strategies should target stream bank erosion.

Assessment of upslope areas and the stream corridor is important in determining baseline conditions, and identifying areas of high erosion, areas of sediment deposition, and areas of net transfer [*Kondolf*, 1995; *Montgomery and MacDonald*, 2002; *Pess et al.*, 2003; *Shields et al.*, 2003; *Evans et al.*, 2006]. Failure to undertake watershed and channel assessments prior to stream restoration in Pennsylvania was recognized by *Moses and Longenecker* [2003] as a key constraint on the success of the projects. An evaluation of the important watershed sediment sources and the overall sediment budget of the watershed was important in determining the success and cost effectiveness of restoration activities [*Moses and Longenecker*, 2003]. A study undertaken in the Murray-Darling watershed, Australia, demonstrated that by targeting the significant sources of sediment in a watershed, the implementation of best management practices (BMPs) would be more cost effective in reducing loads [*Lu et al.*, 2003]. Similarly, *Walling and Collins* [2008] emphasized the benefits of adopting a sediment budget approach when designing sediment control strategies. In the United States, the State of Virginia recognized the importance of targeting and prioritizing stream corridor erosion in their guidelines for the design of stream restoration programs and outlined three key steps, namely, (1) an initial assessment to identify degraded stream reaches, (2) ranking stream reaches based on the initial assessment, and (3) prioritizing stream reaches for restoration [*Virginia Department of Conservation and Recreation*, 2004].

It was estimated that 1 billion dollars was spent every year on stream restoration in the United States [*Shields*, 2009]. Monitoring sediment loads and sediment sources before and after implementation of sediment reduction measures, rather than relying on estimated or modeled impacts, will help to improve understanding and prediction of the efficacy of management practices and other control measures in reducing sediment and nutrient fluxes; however, such monitoring is rarely undertaken [*Kondolf*, 1995; *Moerke and Lamberti*, 2004; *Shields and Knight*, 2004; *Florsheim et al.*, 2006; *Minella et al.*, 2008; *Shields*, 2009]. By identifying the significant sources of sediment before restoration commences, assessments of whether the restoration efforts were successful in reducing erosion and sediment loads will have more significance.

In the United States, the U.S. Environmental Protection Agency (U.S. EPA) has developed a protocol for establishing total maximum daily loads (TMDLs) for sediment, to assist states and Indian Tribes in meeting the requirements of the Clean Water Act. The protocol, which can be applied to both riverine and estuarine systems, supports the development of rational, science-based assessments, and decisions that will lead to the establishment of understandable and justifiable sediment TMDLs. Sediment source assessment and the establishment of sediment budgets are again recognized as important steps in defining sediment TMDLs (Figure 1) [*U.S. EPA*, 1999] because without this information, it is not possible to specify appropriate management actions that are needed to reduce sediment loads.

There are many approaches for determining sediment sources and establishing sediment budgets. In this chapter, we focus on the use of sediment fingerprinting techniques for identifying the primary sources of sediment within a watershed and on the sediment budget approach, as a framework which provides an understanding of the linkages among source areas, depositional sites, and sediment export (Figures 2a and 2b). The identification of sediment sources represents a key product of both approaches. These approaches are most applicable where fine sediment (<0.063 mm) is a problem and where management practices are being considered to remediate the

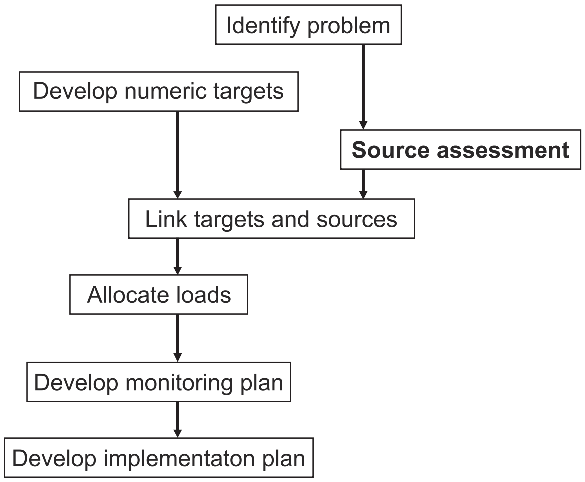


Figure 1. Components in the sediment total maximum daily load (TMDL) procedure [*U.S. Environmental Protection Agency*, 1999].

erosion and sediment problems. Although the sediment fingerprinting and sediment budget approaches can be used at most watershed scales, our experience is that their utility for informing the stream-restoration process is most appropriate for the watershed scales used by many land-management agencies (i.e., extending to less than 300 km^2). This is also the scale at which monitoring the effects of sediment control is likely to be most successful.

2. SEDIMENT SOURCE FINGERPRINTING (SEDIMENT SOURCE TRACING)

The sediment fingerprinting approach provides a direct method for identifying the primary sources of fine-grained suspended sediment within a watershed and estimating their relative contribution to the measured sediment flux [*Walling and Woodward*, 1995; *Collins et al.*, 1997; *Motha et al.*, 2003; *Walling*, 2005]. This approach is based on characterizing each of the potential sediment sources within a watershed by a composite fingerprint, defined by a number of physical or geochemical properties of the source materials, and comparing the fingerprint of suspended sediment sampled at the watershed outlet with the fingerprints of the potential sources (Figure 3). By using an unmixing model, it is possible to estimate the relative contribution of the potential sources to the sampled sediment.

Potential sediment sources within a watershed can be defined in terms of their spatial distribution (e.g., parts of the watershed underlain by different rock types or soil types), but in most situations, emphasis is placed on distinguishing what are commonly referred to as "source types." Classification of source types could involve a simple distinction between upslope areas (sediment mobilized by sheet and rill erosion) and sediment mobilized from the channel system by channel erosion (stream bank erosion). In many cases, however, this classification is extended to include surface soils under different land uses (e.g., cultivation, pasture, and forest), whereas channel erosion could be subdivided to include gully erosion, ditches, channel beds, tributaries, different stream orders, and the main channel system. In addition, specific sources such as unpaved roads, construction areas, and mass movements could also be included [*Nelson and Booth*, 2002; *Gruszowski et al.*, 2003; *Motha et al.*, 2003]. The suite of potential sources selected for a watershed will depend on the local conditions and the nature of the sediment problems in the watershed.

Field reconnaissance and aerial photographs can provide valuable information on both the nature and spatial distribution of upslope area sediment sources, including landslides and unpaved roads, in addition to the location of the main eroding reaches of the channel network [*Mosley*, 1980; *Murray et al.*, 1993; *Kondolf and Downs*, 1996; *Ries and Marzolff*, 1997; *Reid and Trustrum*, 2002; *Yetman*, 2002; *Vermont Agency of Natural Resources*, 2007]. A geographic information system may also prove useful in determining slopes and land cover (including impervious surface extent) for planning the sampling program required to characterize the potential sediment sources [*Kothyari and Jain*, 1997; *Nelson and Booth*, 2002; *Flügel et al.*, 2003].

The procedures commonly used for sampling the potential sediment sources within a watershed are shown in Figure 4. Upslope area sediment sources are commonly sampled by collecting representative samples of the source material from the upper 1–2 cm of the soil [*Carter et al.*, 2003; *Collins and Walling*, 2007; *Gellis et al.*, 2009]. This is seen to be representative of material that could be mobilized by erosion, and sampling points are frequently selected to target areas that are likely to experience active erosion. Because of the likely spatial variability of source material properties, a stratified sampling plan, involving the collection of bulked composite samples from a representative selection of sampling locations, is likely to be the most effective approach [*Carter et al.*, 2003; *Collins and Walling*, 2007]. Locations for collecting samples of stream bank material can be selected randomly or based on a reconnaissance, which develops a stratified sampling plan, such as for different stream orders or land use types. At each location, the bank surface is sampled along several vertical profiles, typically within a channel width of each other and composited into a single sample [*Gellis et al.*, 2009; *Banks et al.*, 2010; *Collins et al.*, 2010]. In stream bank sampling, both channel bends and straight reaches are sampled as well as stream banks on either side of the channel. Again, emphasis is frequently placed on

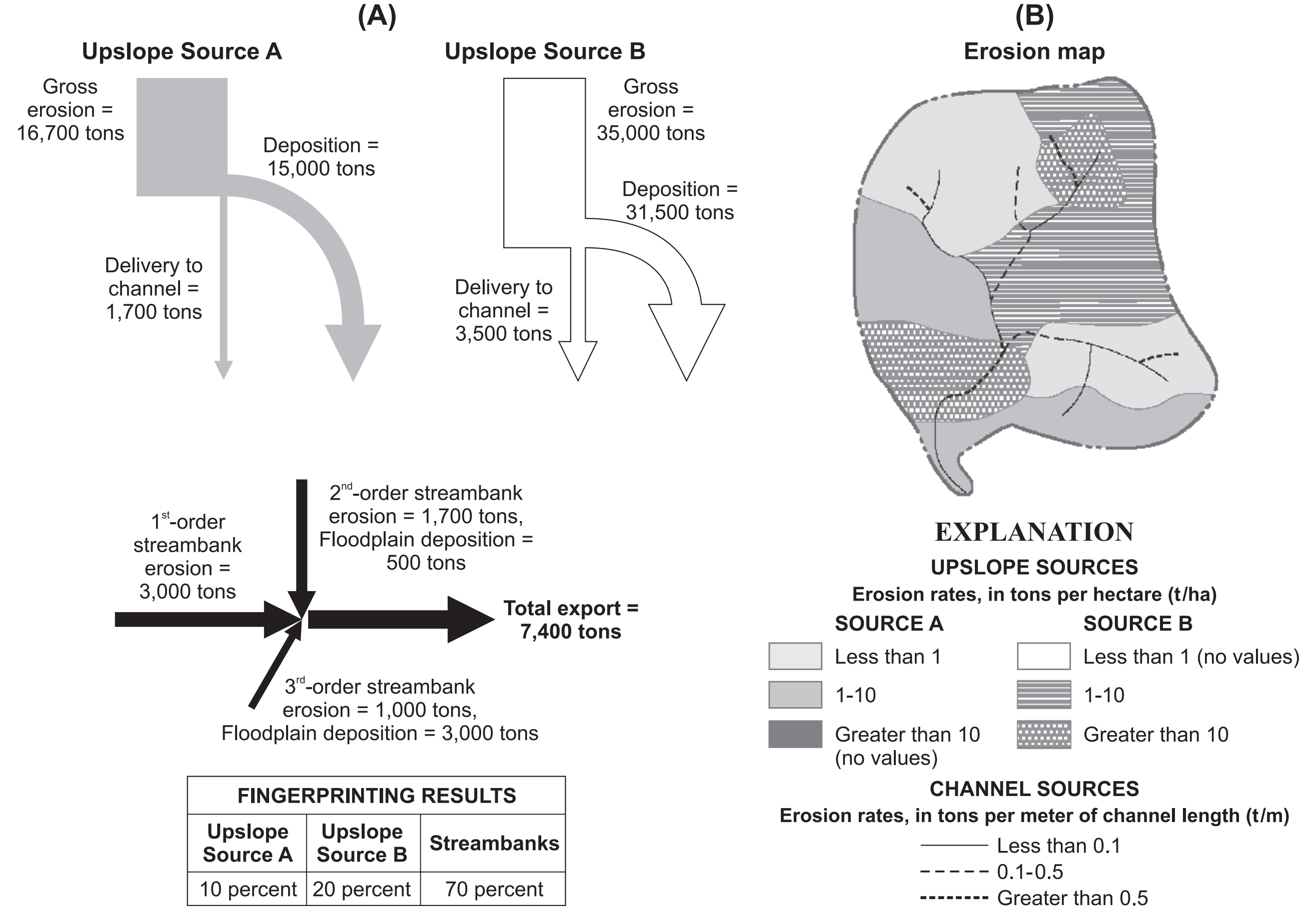


Figure 2. (a) Hypothetical diagram from a third-order watershed illustrating the benefit of combining the sediment budget and sediment fingerprinting approaches. In the sediment budget example (labeled A), 5200 tons of upslope sediment reaches the channel network from sources A and B. In the channel network, 5700 tons are eroded from stream banks (first, second, and third-order streams) and 3500 tons are deposited in the floodplain; the net export out of the watershed is 7400 tons. Although 5200 tons are from upslope sources, based on the sediment fingerprinting results, 30% of the upslope-derived sediment is exported out of the watershed. The difference in percentages of eroded and delivered upslope sediment is due to the mass of sediment in storage (3500 tons), which in this example would be inferred to be dominantly upslope-derived sediment. The sediment budget quantifies the net erosion and deposition, and sediment fingerprinting quantifies the delivery of this sediment. (b) Example of how a sediment budget can be used to assist in management decisions related to watershed restoration. In this example an "Erosion" map is produced from the sediment budget measurements made in Figure 2a. Similarly, a sediment storage map could also be constructed.

sampling actively eroding banks, since the aim is to characterize the sediment originating from channel sources. At construction sites, individual construction areas can be sampled and sediment composited into a single sample. If a sediment detention pond is present within a construction site, this may provide a valuable sampling location because it is likely to contain sediment mobilized from a wider area [*Gellis et al.*, 2009]. The number of samples collected for a given source type or source area will vary according to the size of the watershed, the area of the upslope sediment sources, and the length of the channel system. However, 20 to 30 samples will typically be collected from each source type or area. Where previous fingerprinting studies have been undertaken within the local region, information on the variability of source material properties can be used to inform decisions as to the number of samples required to characterize the variability. For example, such decisions could be based on the number of samples required to estimate the mean concentrations of a set of source material properties to a given precision, at a given level of confidence.

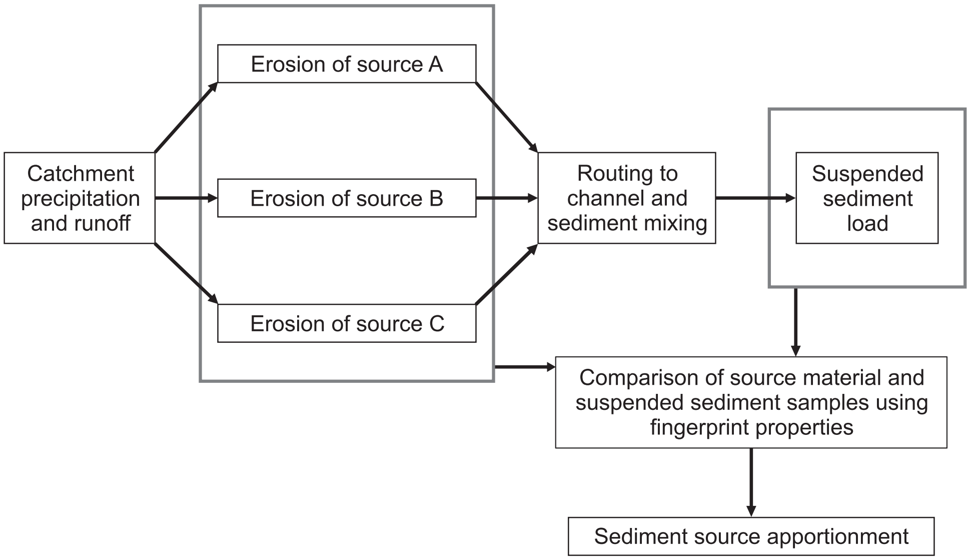


Figure 3. Outline of the sediment fingerprinting approach [from *Walling and Collins*, 2000].

Suspended-sediment samples are traditionally collected at the outlet of the watershed. The temporal basis of the sampling program should be considered carefully and linked to the purpose of the study. If, for example, the aim is to provide information on the relative contributions of the potential sources to the annual sediment yield from the watershed, it is important that the samples collected are representative of the annual sediment flux, and therefore, the samples should be collected during events of different magnitude, at different times during flood events, and at different times of the year. It is important to determine the source of sediment over the course of an entire storm event to avoid bias introduced by the varying travel times of sediment originating from different parts of the study watershed. Sediment samples can be collected by pumping water into containers or using appropriate suspended-sediment sampling equipment (i.e., isokinetic samplers) [*Edwards and Glysson*, 1999]. Large sample volumes may be required where substantial quantities of sediment are required for subsequent analysis of the fingerprint properties, such as for radionuclides. Use of a continuous flow centrifuge or settling and possibly subsequent centrifugation may be required in some situations. Passive trap-type samplers that are installed in the channel and collect a time-integrated sample of suspended sediment have been shown to be effective in

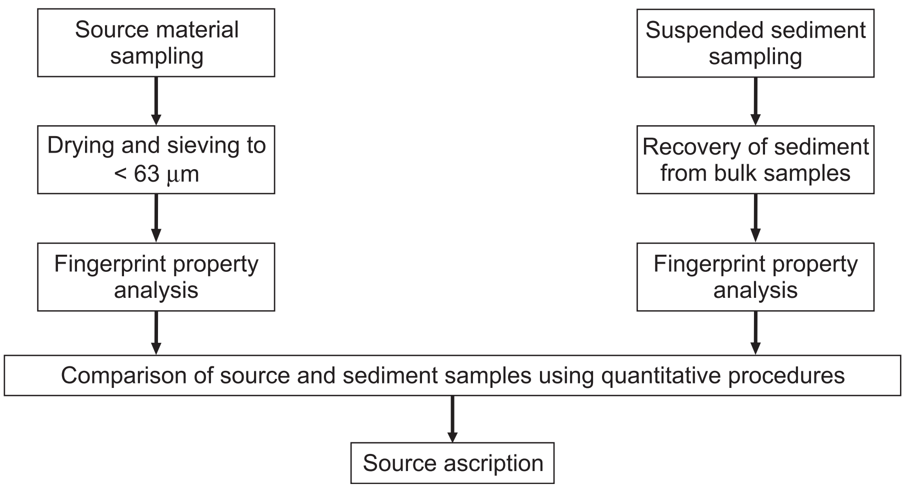


Figure 4. Outline of the sediment fingerprinting sampling procedure [from *Walling and Collins*, 2000].

sampling entire storm events and longer periods and reduce the need for a large number of individual samples [*Phillips et al.*, 2000; *Russell et al.*, 2000; *Brodie*, 2005]. Where it proves impossible to collect suspended sediment samples, due, for example, to lack of time, lack of events, or cost, some workers have successfully used fine surface overbank deposits from river floodplains as a surrogate for suspended sediment [*Collins et al.*, 2010]. If this approach is taken, it is important to recognize that such deposits will reflect sediment transported during overbank flood events, rather than lower magnitude events. Furthermore, the deposited sediment may not be totally representative of the suspended sediment load of the river or stream, if, for example, there is selective deposition of the coarser fractions.

Source material and sediment samples are dried prior to disaggregation and subsequently sieved through a 0.063 mm sieve, in order to recover the <0.063 mm fraction, which is commonly used for sediment source analysis. Sieving of both source material and sediment samples to the same size removes major contrasts in grain-size composition between source material and sediment samples, which could exert an important influence on subsequent geochemical analysis. Drying of both source material and sediment samples commonly involves air drying or oven drying at a low temperature. However, freeze drying may prove an effective means of drying fine-grained sediment because this largely avoids the need to grind and disaggregate the sample after drying.

Identifying the best fingerprint properties for discriminating potential sources in a particular watershed is an important component of the sediment fingerprinting process. *Walling* [2005] emphasized that the search for a single fingerprint property, capable of discriminating several potential sources, is likely to prove elusive, and most studies employ composite fingerprints comprising several fingerprint properties that provide a robust means of discriminating several potential sources. The set of fingerprint properties to be used is commonly selected empirically by analyzing a range of potential fingerprint properties and using statistical tests to identify the properties which provide the best discrimination between the sources. In many studies, however, there may be a need to limit the initial list of potential fingerprint properties to conserve available funds or the mass of sediment available for analysis. In view of the difficulty of identifying effective fingerprint properties on an a priori basis, a compromise approach is adopted in most studies, whereby analysis is initially constrained to a limited list of potential properties, and statistical tests are used to select those that will provide the best discrimination and to identify the best composite fingerprint [*Walling*, 2005].

The tracers used are likely to vary according to whether the focus of the fingerprinting study is on establishing the relative contribution of different source types or different source areas. In the former case, emphasis may be placed on tracers that can discriminate between surface and subsurface material and between cultivated land and land under pasture or rangeland. In the latter case, tracers must be capable of discriminating sediment mobilized from different areas of a catchment and may thus be linked to contrasts between individual soil types and rock types. Tracers that have successfully been used as fingerprints include mineralogy [*Motha et al.*, 2003], color or spectral reflectance properties [*Martínez-Carreras et al.*, 2010], radionuclides [*Walling and Woodward*, 1992; *Collins et al.*, 1997; *Nagle et al.*, 2007; *Walling et al.*, 2008], trace elements [*Walling et al.*, 2008; *Devereux et al.*, 2010; *Mukundan et al.*, 2010], magnetic properties [*Caitcheon*, 1993; *Slattery et al.*, 2000; *Gruszowski et al.*, 2003; *Evans et al.*, 2006], organic matter (C, N) [*Walling et al.*, 2001; *Fox and Papanicolaou*, 2007; *Minella et al.*, 2008; *Gellis et al.*, 2009], stable isotopes (^{15}N and ^{13}C) [*Fox and Papanicolaou*, 2007; *Gellis et al.*, 2009], and compound-specific stable isotopes [*Gibbs*, 2008].

Fallout radionuclides (^{137}Cs and ^{210}Pb$_{ex}$) have often been used in sediment source fingerprinting studies aimed at distinguishing sediment derived from surface and subsurface sources and from cultivated and noncultivated areas [*Walling and Woodward*, 1992]. The depth distributions of these radionuclides in the upper ~30 cm of the soil are influenced by cultivation, which reduces the surface activity by mixing the radionuclide throughout the plough layer. Surface soil from cultivated and uncultivated areas is therefore characterized by different ^{137}Cs and ^{210}Pb$_{ex}$ activities. Furthermore, the activity of these radionuclides declines rapidly with depth, and soil below a depth of about 30 cm is unlikely to contain significant ^{137}Cs or ^{210}Pb$_{ex}$. However, analytical costs for ^{137}Cs and ^{210}Pb$_{ex}$ are high, and their use is precluded in many studies. Recent sediment fingerprinting studies have highlighted the potential for using elemental analysis (metals and base cations) to provide effective source fingerprints (Table 1) [*Walling*, 2005; *Collins and Walling*, 2007; *Banks et al.*, 2010; *Collins et al.*, 2010; *Devereux et al.*, 2010]. The advantage of using such elements as tracers is that, currently (2010), elemental analysis is relatively inexpensive, and more than 30 different elements can be analyzed simultaneously within a small mass of sediment (~0.2 g) using inductively coupled plasma–mass spectroscopy (ICP-MS). Because of the large number of elements derived from elemental analysis, emphasis is placed on the use of statistical techniques to identify a final subset of elements that can reliably discriminate between the potential sources. Depending on budget constraints, other tracers that are relatively inexpensive and may be used in conjunction with elemental analysis are C, N, P, and stable isotopes (Table 1).

Table 1. Example of Fingerprint Properties Used in Sediment Fingerprinting Studies

Fingerprint Property	Sediment Mass Requirements (g)
Inductively coupled plasma–mass spectroscopy (ICP-MS) elemental analysis[a,b]: Al, As, Ba, Bi, Ca, Cd, Ce, Co, Cr, Cs, Cu, Dy, Er, Eu, Fe, Ga, Gd, Ge, Hf, Ho, La, Li, Mg, Mn, Mo, Na, Nd, Ni, Pb, Pd, Pr, Rb, Sb, Sc, Sm, Sn, Sr, Tb, Ti, Tl, V, Y, Yb, Zn, and Zr	~0.2
ICP-MS elemental analysis[c]: Ag, Al, As, Ba, Be, Bi, Cd, Ce, Co, Cr, Cs, Cu, Ga, K, La, Li, Mg, Mn, Mo, Na, Nb, P, Pb, Rb, Sb, Se, Sr, Ti, Th, Tl, U, V, Y, and Zn,	~0.2
Stable isotopes[c]: delta carbon-13 ($\delta^{13}C$), delta nitrogen-15 ($\delta^{15}N$)	~0.1
Radionuclides: cesium-137 ($^{137}Cs^{a,d}$), excess lead-210 ($^{210}Pb_{ex}{}^{a,d}$), and radium-226 ($^{226}Ra^{a}$)	10 or more
Organic: total carbon ($C^{a,d}$) and total nitrogen ($N^{a,d}$)	~0.1
Inorganic: total $P^{a,d}$	~0.1

[a] *Collins and Walling* [2007].
[b] *Collins et al.* [2010].
[c] *Banks et al.* [2010].
[d] *Gellis et al.* [2009].

Radionuclide analysis may cost two to three times more than elemental analysis. The tracers highlighted in Table 1 are not limited to a particular environment, but they are useful for a wide range of geologic, climatic, and spatial settings. In all cases, however, there is a need to test the ability of the tracers selected to discriminate between potential sources before incorporating those tracers in the composite fingerprint used to establish the relative contribution of the potential sources to the downstream sediment load.

Several statistical procedures are used to identify the optimum set of fingerprint properties that will be used in the final composite fingerprint to distinguish the potential sources and establish their relative contribution to the sediment flux at the watershed outlet. The aim is to identify those properties which clearly discriminate the potential sources and to select a small subset of these properties that optimizes the discrimination provided by the composite fingerprint. These statistical procedures incorporate the following steps:

1. In step 1, a bracketing test is performed to confirm that the property values obtained for the fluvial sediment are within the range of the equivalent values obtained for the potential sources. This is an essential prerequisite for the use of an unmixing model to determine source contributions.

2. Step 2 is to identify the fingerprint properties that can distinguish between the potential sediment sources using the nonparametric Kruskal-Wallis H test.

3. In step 3, where a large number of potential properties are used, stepwise multivariate discriminant function analysis can be used to identify an optimum composite fingerprint from the properties selected after step 2.

Once the properties to be included in the composite fingerprint have been identified, the composite fingerprint is used in conjunction with an unmixing model [*Walling*, 2005] to quantify the contributions from the individual potential sources.

$$\mathrm{Res} = \sum_{i=1}^{n} \left[\frac{Cssi - \left(\sum_{s=1}^{m} CsiPsZs \right)}{Cssi} \right]^2, \tag{1}$$

where Res is the residual sum of squares, n is number of fingerprint properties comprising the optimum composite fingerprint, m is number of sediment source categories, $Cssi$ is the concentration of tracer property i in the fluvial sediment, Csi is the mean concentration of the tracer property in the source group s, Ps is the relative contribution from source group s, and Zs is a particle-size correction factor.

An iterative optimization algorithm is commonly used to establish the combination of Ps values, which minimizes the residual sum of squares.

The unmixing model equation assumes that $0 \leq Ps \leq 1$ and

$$\sum_{s=1}^{m} Ps = 1.$$

The particle-size correction factor (Zs) is introduced into equation (1) to take into account contrasts in grain-size composition between the fluvial sediment and source material samples. Even though all samples may be sieved to <0.063 mm to minimize contrasts in grain-size composition between sediment and source material samples, differences in the grain-size composition of the <0.063 mm fraction can still exert an influence on the values obtained for individual properties and thus preclude direct comparison of property values for sediment and source materials. Frequently, the grain-size correction factor is based on the ratio of the specific surface area of the sediment to that of the source material [*Walling*, 2005]. Grain size can also be used in the correction factor.

The unmixing model provides estimates of the relative (%) contribution of the individual sources to a given suspended sediment sample. Different results can be expected for different storms, in response to the different hydrologic and seasonal conditions associated with the sampling of the fluvial sediment. Where instantaneous samples are involved, the timing of the sample within a storm runoff event and in relation to the routing of sediment from different portions of the watershed is also likely to influence the result [*Webb and Walling*, 1982]. Bearing in mind the critical importance of storm events in accounting for a major proportion of the sediment flux from a watershed [*Wolman and Miller*, 1960; *Guy*, 1964; *Webb and Walling*, 1982; *Francke et al.*, 2008], it is important that such events are included in the sampling strategy. Taking the average of the source contributions for all sampled events may not be the most meaningful method for establishing the sources of the longer-term sediment yield from a watershed. Because sediment loads vary with storm magnitude, a load-weighted mean contribution is likely to provide a more reliable estimate of the relative contribution of a set of sources to the sediment yield [*Walling et al.*, 1999].

In several recent studies, attention has been directed to the uncertainty associated with the estimates of source contribution provided by the unmixing model. These uncertainties relate to both the potential equifinality of the optimized solution of the model [*Rowan et al.*, 2000] and the use of mean values derived from a relatively small number of samples to represent the properties of the individual potential sources. In the latter case, Monte Carlo procedures can be used to take account of the spatial variability of source properties and to establish the likely confidence limits of the source contribution estimates [*Collins and Walling*, 2007; *Martínez-Carreras et al.*, 2008]. *Gellis and Landwehr* [2006] developed an unmixing model specifically for small sample data sets that is based on normalizing the standard deviation of a mixture.

3. PLACING SEDIMENT SOURCE FINGERPRINTING STUDIES WITHIN A WIDER SEDIMENT BUDGET CONTEXT

Establishing the sources of sediment output from a watershed, through sediment fingerprinting, for example, will frequently represent a key step in assembling a sediment budget for that watershed [*Walling and Collins*, 2008]. Although the sediment fingerprinting approach is able to provide information on the dominant sources of the sediment flux at the outlet of a watershed, it may not provide detailed information on the precise location of the main sediment sources. For example, sediment fingerprinting may indicate that the stream banks are a major source of sediment, but sediment fingerprinting analysis does not determine which stream reaches have the highest rates of bank erosion and therefore need to be targeted by an effective sediment control program. The sediment budget approach provides a valuable basis for placing the information on sediment sources within the broader context of the processes of sediment mobilization, transfer, storage, and export operating within a watershed (Figure 2a).

Large amounts of sediment could be mobilized by erosion from agricultural land within the upslope areas, but if much of the mobilized sediment is stored within intermediate areas of the watershed, the contribution of this source to the sediment yield at the basin outlet might be very small and secondary to channel erosion in the lower reaches of the watershed. In this situation, reduction of channel erosion is likely to be the most effective means of reducing the sediment yield from the watershed, even though the amounts of sediment mobilized by erosion in the upslope areas are much higher. Information assembled for the sediment budget approach, such as an assessment of the spatial distribution and magnitude of bank erosion rates, can often be used to identify "hot spots" or areas of high erosion in the watershed where management actions may be directed (Figure 2b).

A sediment budget can be represented as an equation:

$$I \pm \Delta S = O, \tag{2}$$

where I is the sediment input or sediment mobilization, ΔS is the change in sediment storage, and O is the sediment output.

Sediment budget studies have used a variety of techniques to quantify the mobilization, transport, and storage of sediment (Table 2). For the purposes of this chapter, we highlight techniques that we believe are important in developing sediment budgets to support watershed (channel and upslope) restoration.

3.1. Upslope Area Erosion

Upslope area erosion is considered to include sheetwash, rill erosion, and mass movements. Surface erosion rates can be measured using a variety of techniques (Table 2) including sediment traps, erosion pins, small dams, and surveys of ponds [*Leopold et al.*, 1966; *Loughran*, 1989; *Gellis et al.*, 2001a, 2001b]. Many of the techniques listed in Table 2 can be separated into field- and office-based procedures. Field techniques such as sediment traps and erosion pins can provide useful data on an event basis, but are labor intensive, require maintenance shortly after storm events, and measure erosion from small areas. Office-based techniques include photogrammetry and models [*Evans et al.*, 2006] (Table 2). These techniques, although less intensive in terms of labor and time, may produce a wide range of results and need to be validated with data collected from the watershed of interest.

Table 2. Approaches and Tools Used to Construct Sediment Budgets

Technique	Used to Measure Erosion or Deposition	Watershed Elements Quantified	Time Scales	Applicable Spatial Scales	References
Monitoring suspended sediment loads	erosion and deposition	watersheds and tributaries, channel reaches	years, decades, individual storm events	km	*Milliman and Meade* [1983], *Parker* [1988], *Fitzpatrick et al.* [1999], *Syvitski et al.* [2003], *Gellis et al.* [2006]
Aerial photography and LIDAR[a]	erosion and deposition	stream banks, roads, land use/land cover	years, decades, individual storm events	m, km	*Bartley et al.* [2008], *Keen et al.* [2008], *Pizzuto et al.* [2007], *Piégay et al.* [2005], *Thoma et al.* [2005], *Gellis* [2002], *Evans et al.* [2006], *Lawler* [1993]
Aerial photography[a]	erosion	mass movements	years, decades, individual storm events	km	*Wallbrink et al.* [2002], *Larsen and Santiago-Roman* [2001], *Dietrich et al.* [1982]
Historical maps	erosion and deposition	channels, channel bars	years, decades	m, km	*Kesel et al.* [1992]
Erosion pins	erosion and deposition	sheetwash	years, days, individual storm events	cm, m	*Leopold et al.* [1966], *Visser et al.* [2007]
Field surveying, historical surveys, and inventorying	erosion and deposition	channels, gullies, channel bars	years, decades, individual storm events	m, km	*Leopold et al.* [1966], *Munro et al.* [2008], *Marzolff and Ries* [2007], *Gellis et al.* [2001b], *Wijdenes and Bryan* [2001], *Kesel et al.* [1992], *Trimble* [1983]
Radionuclide inventories (^{137}Cs, ^{210}Pb)	erosion and deposition	upslope areas of varying land use and land cover	decades	km	*Campbell et al.* [1988], *Ritchie and McHenry* [1990], *Walling and Bradley* [1990], *Walling and Quine* [1991], *Walling and He* [1999b], *Nagle et al.* [2000], *Wallbrink et al.* [2002], *Walling et al.* [2003b], *Zapata* [2002]
Radionuclides as time horizon markers (^{137}Cs, ^{210}Pb)	deposition	floodplains, lake and pond sediment	decades	km	*Amos et al.* [2009], *Ritchie et al.* [2004], *Terry et al.* [2002], *Walling and He* [1997b], *Walling et al.* [2001]
Dendrochronology	erosion and deposition	channels, floodplains	decades, years	m, km	*Allmindinger et al.* [2007], *Hupp and Bazemore* [1993], *Dietrich et al.* [1982]
Clay pads	deposition	floodplains	years, individual storm events	m	*Gellis et al.* [2009], *Hupp and Bazemore* [1993]
Bank pins	erosion and deposition	stream banks	years, days, individual storm events	cm, m	*Thorne* [1981], *Bartley et al.* [2006], *Lawler* [1993]
Sediment traps	erosion	upslope areas of varying land use and land cover	years, days, individual storm events	m, km	*Gellis et al.* [2006], *Larsen et al.* [1999]
Resuspension cylinder	deposition	channel storage	days, individual storm events	M	*Lambert and Walling* [1988]
Sediment fingerprinting	erosion	watershed areas under varying land cover or geologic type	years, individual storm events	km	*Walling and Woodward* [1995], *Collins et al.* [1997], *Motha et al.* [2003], *Walling* [2005]
Ponds and lakes	deposition	lake bottoms and deltas	decades, years	km	*Leopold et al.* [1966]
Sediment models[b]	erosion and deposition	hillslopes, channels	decades, years, days, individual storm events	m, km	*Fu et al.* [2010], *Evans et al.* [2006], *Croke and Nethery* [2006], *Aksoy and Kavvas* [2005], *Chen and Lai* [2005], *Merritt et al.* [2003], *Bhuyan et al.* [2002]

[a]Aerial photography can be used to establish the location of potential sediment sources but can also be used over successive time periods to quantify change.

[b]The types and versions of models available to budget sediment are far too numerous to summarize here. For more information, the reader is directed to the references provided.

Another useful technique that has been increasingly applied to estimate upslope area erosion is the use of fallout radionuclides, particularly cesium-137 (^{137}Cs), which have been successfully used to document soil erosion and sediment redistribution in a variety of locations worldwide (Table 2). ^{137}Cs is an artificial radionuclide with a half-life of 30.7 years, which was introduced into the stratosphere by the testing of aboveground thermonuclear weapons in the late 1950s and early 1960s and subsequently deposited as fallout. The ^{137}Cs fallout reaching the land surface is rapidly and strongly fixed by soil and sediment particles, and the post-fallout redistribution of the radiocesium directly reflects the mobility and redistribution of soil particles [*Ritchie and McHenry*, 1990; *Zapata*, 2003].

The loss or gain of ^{137}Cs from a particular measuring point is determined by comparing the ^{137}Cs inventory measured at a sampling point with the equivalent inventory for a neighboring reference site or sites, which is undisturbed, stable over time, and not eroding. The selection of a stable reference site may prove problematic in some areas under intensive land use, but areas of permanent pasture, orchard, low-density forest, and cemeteries may prove to be appropriate. The relationship between the loss and gain of ^{137}Cs and the soil redistribution rate averaged over the period since the main period of radiocesium fallout in the late 1950s and early 1960s is usually derived theoretically, and a number of conversion models have been developed for this purpose [*Walling and He*, 1997a, 1999a; *Zapata*, 2002].

A useful review of the field and laboratory procedures associated with the use of the ^{137}Cs technique is provided in the manual for its use produced by the International Atomic Energy Authority [*Zapata*, 2002]. The design of the field sampling program is particularly important because this must account for both the systematic variation of the ^{137}Cs inventory in response to soil redistribution, as well as small-scale variability associated with sampling variability and soil heterogeneity [*Bachhuber et al.*, 1987]. Soil sampling for ^{137}Cs is commonly undertaken using a cylindrical coring device or a box coring device [*Campbell et al.*, 1988]. Information on the ^{137}Cs depth distribution is required for determining sampling depths and estimating rates of soil redistribution on undisturbed soils and for confirming that reference sites are undisturbed. The depth of sampling must extend below the maximum depth of ^{137}Cs penetration. In cultivated areas, this will reflect the tillage or plough depth. In undisturbed soils, ^{137}Cs activity commonly declines exponentially with depth below the surface peak, and little activity is commonly found below 20 cm [*Walling and Quine*, 1992]. This depth will, however, vary according to soil type and the local conditions. Trial sampling may be required to establish this depth and thereby ensure that soil cores provide a representative assessment of the total inventory of the soil. In this context, it is important to recognize that ^{137}Cs will be found to greater depths in depositional areas in both cultivated and undisturbed areas [*Fitzpatrick et al.*, 2008]. Analysis of samples for ^{137}Cs activity commonly focuses on the <2 mm fraction and involves gamma spectrometry.

3.2. Stream Corridor Erosion and Deposition

The stream corridor component of the sediment budget should provide information on three aspects of sediment mobilization and sediment storage: (1) the magnitude of sediment mobilization by channel erosion, (2) sediment storage and remobilization, and (3) the spatial distribution of eroding and aggrading reaches. Field methods for determining the rates and location of stream bank erosion and streambed changes include benchmarking and resurveying of channel cross sections over time to determine rates of erosion and deposition [*Harrelson et al.*, 1994]. Bank pins consisting of metal pins inserted into the bank, which are remeasured at intervals to document the rate of retreat of the bank face, can be used to measure the retreat of the banks over time (Table 2). The number and spacing of such pins depends upon cost, time, and labor constraints. *Thorne* [1981] recommended that pins should be spaced longitudinally every 1 to 5 m, with two or more pins inserted at each vertical section. Analysis of aerial photographs over time can also provide valuable information on rates of bank retreat and elongation of gully networks (Table 2). More recently, ground-based and airborne lidar have been shown to be useful in quantifying bank retreat and gully erosion [*Thoma et al.*, 2005; *Pizzuto et al.*, 2007; *Perroy et al.*, 2010].

Quantification of rates of floodplain accretion and associated sediment storage can involve dendrogeomorphic (tree ring) analyses and clay pad analyses, as well as the use of the environmental radionuclides ^{137}Cs and excess lead-210 ($^{210}Pb_{ex}$) to establish rates of accretion (Table 2). Dendrogeomorphic techniques use tree-ring information to date and interpret the age of various geomorphic surfaces and can be used to determine the net rate of floodplain sediment deposition, and in some cases, rates of erosion and subsidence [*Sigafoos*, 1964; *Hupp and Bornette*, 2003]. Typically, six or more trees are sampled at a monitoring point. Replication is necessary to account for local variation in deposition rates and to ensure the determination of a mean rate with an acceptable SE less than the mean. This technique has been used with considerable success along streams within the Atlantic Coastal Plain [*Hupp et al.*, 1993; *Jolley and Lockaby*, 2006] and the Great Plains [*Friedman et al.*, 1996] regions of the United States, and in

Europe [*Piégay et al.*, 2008]. Although dendrogeomorphic analyses are not as precise as other techniques for measuring erosion and deposition, they are relatively inexpensive and may provide long-term erosion and deposition rates, often with considerable spatial and temporal detail wherever ring-producing trees grow.

Both ^{137}Cs and ^{210}Pb$_{ex}$ provide a means of establishing rates of floodplain accretion (Table 2). For both radionuclides, it is necessary to collect a sediment core from the floodplain, to section this core into 1 or 2 cm depth increments and to analyze the resulting sections for the appropriate radionuclide. In the case of ^{137}Cs, the sediment deposited at the time of peak fallout in 1963 (or 1964 in the Southern Hemisphere) can be identified by the occurrence of maximum activity of ^{137}Cs in one of the increments. In some cases, the depth at which ^{137}Cs first appears is dated to the onset of radiocesium fallout in 1954. However, this second approach is less reliable because there can be some postdepositional downward migration of the ^{137}Cs fallout, which results in overestimation of accretion rates.

In the case of ^{210}Pb$_{ex}$, evidence of the age-depth relationship is provided by the rate of downward decline of the ^{210}Pb$_{ex}$ activity, which reflects both the half-life of the radionuclide and the rate of sediment accretion. Interpretation of the pattern of downward decline of the radionuclide in a sediment core is, however, complicated by the existence of two possible sources of ^{210}Pb$_{ex}$ in the sediment profile, namely, direct fallout to the floodplain surface and deposition of ^{210}Pb$_{ex}$-rich sediment mobilized by erosion from upslope areas. Although ^{137}Cs can only provide a time horizon linked to 1963, ^{210}Pb$_{ex}$ can provide information on the age-depth relationship, covering a period of more than 100 years.

It should also be emphasized that infrequent events, such as floods, are important in the erosion and transport of sediment, initiation of landsliding, and modification of the channel banks, bed, and floodplain. Because large events may cause substantial upland erosion and channel change, quantifying the effects of floods by inventorying landslides, resurveying channels, floodplains, reservoirs, or measuring bank pins after large runoff events may improve the sediment budget [*Schmidt*, 1994; *Johnson and Warburton*, 2002; *Fuller*, 2008; *Schwab et al.*, 2008].

3.3. Suspended-Sediment Export

Quantification of the suspended-sediment export from a watershed is a key element of any sediment budget study because this represents the output term O in equation (2). Equally, during sediment source fingerprinting investigations, the availability of information on the sediment yield from a watershed can greatly extend the utility of the results obtained, transforming estimates of the proportion supplied by a particular source to an estimate of the magnitude of the amount of sediment involved. Depending on the grain size fraction of interest, measurements of sediment export could include both the suspended-sediment and the bed load fluxes.

When a reservoir has been constructed at the outlet of a watershed and traps all or most of the sediment input, reservoir surveys and sediment cores may provide a basis for quantifying the amount of accumulating sediment and, thus, the total sediment exported from the watershed over the period between surveys [*Holliday et al.*, 2003; *Slaymaker*, 2003; *Walling and Collins*, 2003]. The period between surveys is likely to comprise several years, and reservoir surveys provide estimates of longer-term sediment flux. Although reservoirs will commonly trap all of the bed load, some of the suspended load may not be trapped, and it is possible to account for this by assessing the trap efficiency of the reservoir. Trap efficiency will depend primarily on the residence time of the inflowing water within the impoundment and thus the capacity/inflow ratio, although the grain-size composition of the suspended load will also exert an influence [*Vanoni*, 1975].

In most situations, a reservoir will not be available at the outlet of the study watershed, and in the absence of an existing measuring station, a program for measuring the sediment load will need to be established. Such programs are likely to focus on the suspended load, but bed load measurements can also be included. The majority of the sediment output from most watersheds is transported during a few high magnitude events, and information on the sediment load transported during shorter periods or by individual events will be required. To obtain accurate information on the suspended-sediment load, it is necessary to define the continuous trace of sediment concentration [*Porterfield*, 1972] and to combine this with an accurate record of water discharge. Sediment surrogates, such as turbidity, are often continuously recorded to facilitate the production of a continuous record of sediment concentration, but the turbidity measurements need to be calibrated to site-specific conditions [*Lewis*, 1996; *Ziegler*, 2003]. In the absence of these detailed records, estimates of daily, monthly, or annual sediment loads are frequently produced using rating curves or regression relationships between suspended-sediment concentrations or loads and water discharge. However, it should be noted that this approach may introduce errors in excess of 100% [*Walling*, 1983] because the suspended-sediment load of a river is commonly supply-dependent rather than transport capacity-dependent, and any relationship between sediment concentration or load and water discharge is likely to be poorly defined and may involve considerable scatter and well-developed hysteresis.

3.4. Sediment Budget Uncertainties

Notwithstanding its utility, any attempt to establish a sediment budget for a watershed frequently faces many problems and uncertainties when quantifying the various components of the budget. In theory, all the inputs and storage terms of the budget, when combined, should balance the output (equation (1)). However, the budget is rarely balanced on the basis of the available measurements [*Kondolf and Matthews*, 1991]. If the sediment input is higher than the sediment output, the difference is usually assigned to sediment storage [*Kondolf and Matthews*, 1991]. *Kondolf and Matthews* [1991] reported imbalances in sediment budgets as high as 104% of the total measured sediment output. Sources of error in balancing the budget include those associated with the techniques used to quantify the individual components of the budget, the temporal variability of the processes involved, and spatial extrapolation of point measurements to the entire watershed. The timescales involved in quantifying erosion may vary among different approaches, thereby introducing additional uncertainty. Although the different timescales and spatial scales used in a sediment budget may increase the variability of the results, it is important to have measurements at different temporal and spatial scales, in order to provide multiple lines of evidence on the sediment processes involved. Another common problem in constructing sediment budgets is that erosion and deposition are measured, but the delivery of sediment to the channel is not. The sediment delivery ratio is defined as the amount of sediment that reaches the watershed outlet divided by the amount of sediment mobilized by erosion within the watershed. Sediment delivery ratios estimated from sediment budgets range from close to zero to over 100% [*Smith and Dragovich*, 2008; *Walling and Collins*, 2008]. Of the many benefits associated with the sediment budget approach, one of the most important is the information obtained on rates and locations of erosion and the interrelationship between sediment mobilization, sediment storage, and sediment output, which are important for informing decision making when designing restoration programs. Sediment budgets can provide information on changes in sediment sources and depositional areas over time, whereas sediment fingerprinting provides the relative contribution of different sources to the sediment output over the sampling period.

4. TARGETING SEDIMENT SOURCES: CASE STUDIES

In this chapter, we emphasize the importance of using both the sediment source fingerprinting and sediment budget approaches before implementation of a sediment control or watershed-restoration program (Figure 5). This preliminary assessment will assist land managers in developing an improved understanding of the functioning of a watershed. Performing a sediment fingerprinting/sediment budget survey will require an initial capital investment. Depending on scale, site location, resources, and other factors, the cost of this investment will vary. In most cases, the investment should be a small percentage of the overall cost of the restoration. In our opinion, without this initial investment to determine the significant sources of sediment, the restoration project may not achieve its desired goal(s) or attain the full potential for reducing sediment loadings.

The sediment source fingerprinting and sediment budget approaches have now been used both separately and in combination to support the development of management strategies to restore watersheds and to reduce erosion and sediment transport worldwide [*Gellis et al.*, 2009; *Minella et al.*, 2008; *Walling et al.*, 2008; *Walling and Collins*, 2008; *Evans et al.*, 2006; *Jordan*, 2006; *Merz et al.*, 2006; *Walling et al.*, 2006; *Slaymaker*, 2003; *Reid and Trustrum*, 2002; *Nelson and Booth*, 2002; *Wallbrink et al.*, 2002; *Walling et al.*, 2001]. The problems faced, the approaches used, and the management implications for some of these studies are summarized in Table 3. A brief description of some of the projects that the authors have been involved with that have used either the sediment fingerprinting and/or sediment budget approaches are provided in the case studies that follow [*Gellis et al.*, 2009; *Walling et al.*, 2006; *Walling et al.*, 2001; *Gellis et al.*, 2001b].

4.1. Southern Zambia

Walling et al. [2001] used the sediment budget approach, combining suspended-sediment source fingerprinting with measurements of upslope area erosion (^{137}Cs), floodplain and reservoir sedimentation (^{137}Cs), and sediment yield from the watershed to examine the sediment dynamics in the 63 km^2 Upper Kaleya River in Southern Zambia between 1997 and 1999. Erosion and sedimentation problems in the Upper Kaleya River watershed are representative of similar problems throughout much of Southern Africa [*Walling et al.*, 2001]. The objectives of the study were to target areas of high sediment production within the watershed that contributed substantially to the downstream sediment yield, to inform decision makers, and improve sediment control policies. Sources of sediment included (1) channel banks and gullies, and (2) surface erosion in areas under commercial cultivation, (3) communal cultivation, and (4) bush grazing.

A variety of sediment fingerprints, including metals, organic matter (C, N), cations, and radionuclides were used to establish the relative contribution of the four potential sediment sources identified above to the sediment yield at the

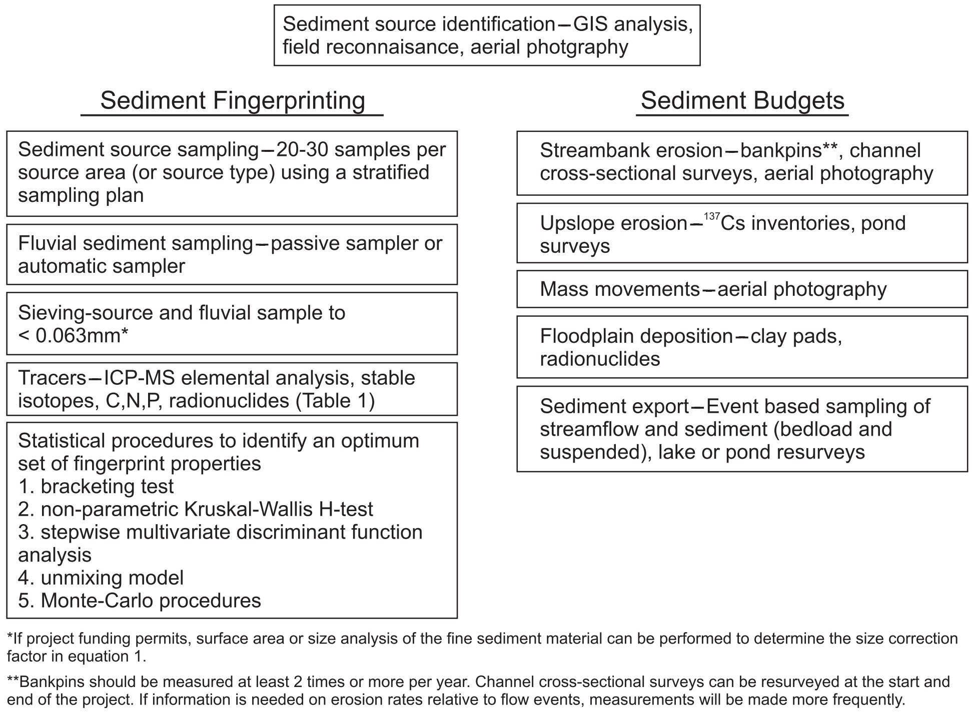


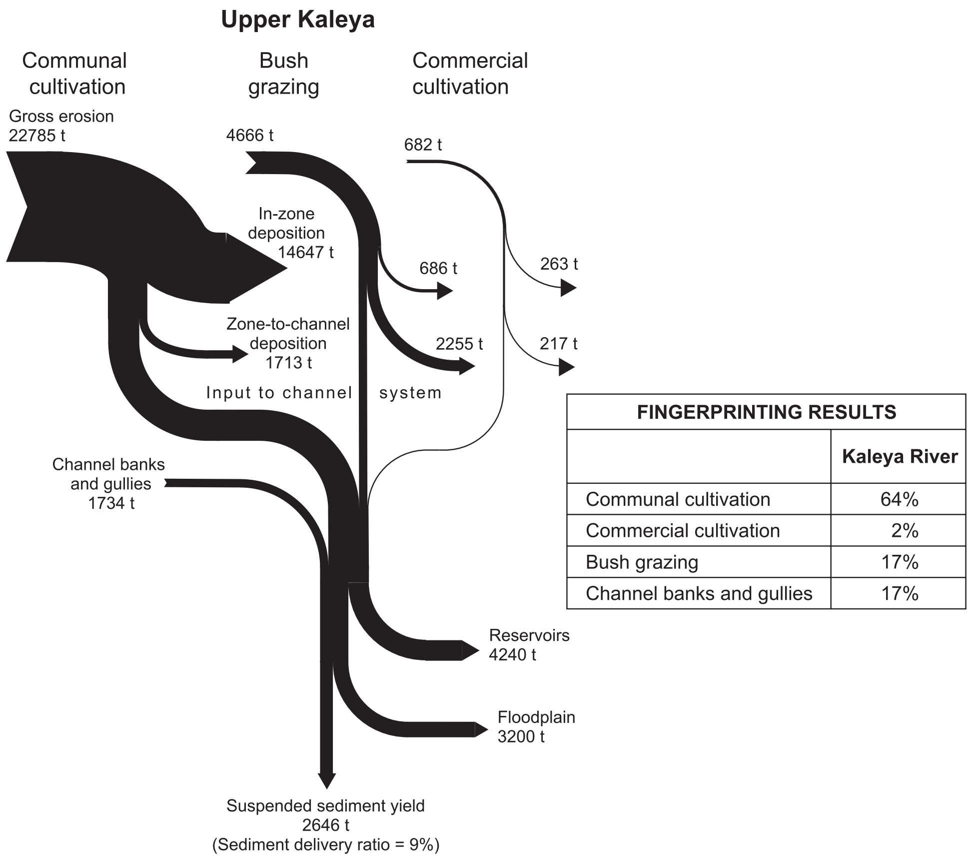
Figure 5. Techniques highlighted in this paper to determine sediment sources in watersheds where excess sediment has been identified and where management practices are being considered.

watershed outlet. The results of the sediment source fingerprinting study showed that the load-weighted mean relative contributions for a representative set of 13 storms were channel banks and gullies, 17%; commercial cultivation, 2%; communal cultivation, 64%; and bush and bush grazing, 17%. Estimates of rates of sediment mobilization from the watershed slopes were obtained using ^{137}Cs measurements. These provided estimates of both gross and net rates of soil loss, with rates of net soil loss from areas of commercial cultivation of 0.43 kg m^{-2} yr^{-1}; from communal cultivation of 0.25 kg m^{-2} yr^{-1}; and from bush grazing of 0.29 kg m^{-2} yr^{-1}. Based on the available measurements, the mean annual suspended-sediment output from the Kaleya watershed was estimated to be 2646 t yr^{-1}, which is equivalent to a specific sediment yield of 42 t km^{-2} yr^{-1}. Significant conveyance losses were associated with overbank sedimentation on the floodplain bordering the lower reaches of the main channel and in off-channel water storage reservoirs supplied from the main channel, which reduced the sediment output from the watershed by 3200 and 4240 t yr^{-1}, respectively. By combining the estimate of sediment yield from the watershed with the estimates of conveyance loss associated with floodplain and reservoir sedimentation, the total mass of sediment delivered to the main channel system from 1997 to 1999 was estimated to be 10,086 t yr^{-1}. The results obtained from the source fingerprinting study were used to apportion this to the four main sources, providing estimates of the annual contributions from channel and gully erosion (1734 t), commercial cultivation (202 t), communal cultivation (6425 t), and bush grazing (1725 t). Although commercial cultivation had the highest rates of erosion (0.43 kg m^{-2} yr^{-1}), it only comprised 1.3 km^2 of the watershed.

The estimates obtained for the individual components were used to construct the overall sediment budget for the Kaleya River watershed (Figure 6). Using the ^{137}Cs measurements, upslope deposition was subtracted from gross upslope erosion to estimate the loss of sediment from areas of commercial and communal cultivation, and bush grazing. Thus, for example, the gross soil loss estimated for communal cultivation was 22,785 t yr^{-1}, and the deposition was 14,647 t yr^{-1}, which resulted in a net sediment loss of 8138 t yr^{-1} (Figure 6). Sediment fingerprinting results indicated that 6425 t yr^{-1} was contributed from communal cultivation. The difference between the soil erosion value and the sediment fingerprinting results (1713 t yr^{-1}) was attributed to conveyance losses associated with the transport of sediment from the slopes to the channel network (zone-to-channel deposition in Figure 6). In the case of channel and gulley erosion, efficient delivery to the main channel system with no depositional storage was assumed.

Table 3. Examples of Sediment Budget and Sediment Fingerprinting Studies That Have Been Used to Guide Management Actions to Reduce Erosion and Sediment Transport

Location of Project and Scale	Sediment Associated Problem(s)	Objective(s)	Approaches Used	Results and Land Management Implications	References
Guapore River, southern Brazil; 1.19 km^2	accelerated soil loss from agricultural land	assess the impact of improved land management on sediment mobilization, transport, and delivery	sediment fingerprinting, water and sediment flux measurements for individual storm events	results of the study clearly demonstrated the potential for using sediment fingerprinting to assess the impact of land management on reducing erosion and sediment yields	*Minella et al.* [2008]
Issaquah Creek, western Washington; 144 km^2	flooding, loss of fish habitat, and water quality	develop a sediment budget for an urbanizing watershed and evaluate how development has altered predevelopment processes	sediment budget based on established estimates of regional erosion, USLE, geographic information system (GIS), landslide measurements, sediment transport equations	results indicated the contribution of fine sediment (roads), versus mixed-grained (landslides, stream banks) and management implications for reducing sediment loads.	*Nelson and Booth* [2002]
Bush Catchment, Northern Ireland, 340 km^2	salmon habitat	guide management actions to reduce fine sediment loadings through identification of the sources and contribution of fine sediment transport	sediment fingerprinting, GIS, bank erosion pins, helicopter survey	management actions to reduce erosion and sediment delivery were justified by the sediment source assessment, which showed drainage maintenance, bank erosion, agriculture, and livestock poaching to be the main sediment sources	*Evans et al.* [2006]
Redfish Creek (26.2 km^2) Laird Creek (15 km^2), Gold Creek (95 km^2), southeastern British Columbia, Canada	effects of forestry operations on water quality, fish habitat, channel stability	illustrate the practical application of the sediment budget approach in assessing the impacts of forestry operations	sediment budget based on surveys of natural and development-related sediment sources and yields	forest roads are a major source of sediment	*Jordan* [2006]
Unnamed watershed, southeast New South Wales, Australia; 12 ha	effects of timber harvesting on sediment delivery	quantify the transfer and storage of sediment related to forestry operations and assess the effectiveness of erosion mitigation strategies	sediment budget using ^{137}Cs and ^{210}Pb inventories, aerial photography, GIS	tracks formed by dragging logs and log storing areas have the highest rates of erosion. Management efforts to reduce erosion in these areas have been largely successful	*Wallbrink et al.* [2002]



FINGERPRINTING RESULTS	
	Kaleya River
Communal cultivation	64%
Commercial cultivation	2%
Bush grazing	17%
Channel banks and gullies	17%

Figure 6. Final sediment budget for Kaleya River watershed, 1997 through 1999 (t indicates metric tons). Modified from *Walling et al.* [2001].

The sediment budget for the Kaleya watershed (Figure 6) indicates that the upslope areas, rather than channel and gully erosion, were the main sediment source and that these areas were well connected to the main channel system. Any attempt to develop a management strategy aimed at reducing the sediment yield from the watershed should therefore focus on reducing soil loss from the areas of commercial and communal cultivation and bush grazing [*Walling et al.*, 2001]. Equally, sediment storage during sediment transfer between the cultivated fields and grazing areas and the main channel system was seen as a major component of the sediment budget. The storage areas should therefore be carefully managed in order to continue their important role in reducing sediment delivery to the main channel network and to ensure that the stored sediment is not remobilized [*Walling et al.*, 2001]. The latter could clearly result in increased sediment inputs to the channel system. A similar conclusion was reached in the Murray-Darling River watershed, Australia, where results indicated that both erosion rates at the source areas and sediment delivery efficiency need to be considered to achieve effective targets for reducing the downstream delivery of fine sediment [*Lu et al.*, 2003].

The Upper Kaleya watershed sediment budget investigation provided a valuable demonstration of the potential for combining ^{137}Cs measurements and sediment source fingerprinting techniques with traditional sediment monitoring, to establish a watershed sediment budget, and variants of the approach have been applied in several other watersheds in different areas of the world. Furthermore, it afforded a timely demonstration, for a watershed in a developing country, of the importance of developing an understanding of sediment sources, transfer pathways, sediment storage, and watershed connectivity, before initiating a sediment control or management program.

4.2. Chesapeake Bay

The Chesapeake Bay watershed, which contains the largest estuary in the United States and has a drainage area of 165,800 km^2, was listed as an “impaired water body” in 2000 under the Clean Water Act because of excess sediment and nutrients [*Phillips*, 2002]. Excess fine-grained sediment has reduced water clarity, which in turn has affected submerged aquatic vegetation [*Phillips*, 2002]. In order to reduce sediment inputs to the bay, it is necessary to identify the main sources of fine-grained sediment.

The large scale of the Chesapeake Bay watershed (165,800 km^2) necessitated determining erosion, sediment transport, and deposition at several scales and in a variety of environments using several approaches. At the larger spatial scales, suspended-sediment data, and the Spatially Referenced Regressions on Watershed Attributes (SPARROW) model showed that modern sediment yields (20th century) were highest in the Piedmont Physiographic Province and lowest in the Coastal Plain Physiographic Province (Plate 1) [*Gellis et al.*, 2009; *Brakebill et al.*, 2010]. Erosion indices based on meteoric beryllium-10 showed the highest rates of soil erosion from the piedmont part of the Susquehanna River watershed (Plate 1), specifically in the Conestoga River watershed. In contrast to the recent 20th century land-use disturbance and high rates of erosion in the Piedmont Province, geologic rates of erosion (between 10,000 years and 100,000 years) measured with in situ beryllium-10 were lowest in the Piedmont Province.

Sediment source analysis using geochemical fingerprints was performed for three watersheds draining the Chesapeake Bay watershed, two in the Coastal Plain (Pocomoke River, 157 km^2 and Mattawoman Creek, 134 km^2) and one in the Piedmont (Little Conestoga Creek, 110 km^2), between 2001 and 2004 to determine whether upslope areas or the channel corridor were the most important source (Plate 1) [*Gellis et al.*, 2009]. Sediment properties used to distinguish sediment sources in the three watersheds included radionuclides (^{137}Cs, ^{210}Pb), stable isotopes ($\delta^{13}C$, $\delta^{15}N$), and C, N, and P.

Six storm events were sampled in the agricultural and forested Pocomoke River between July 2001 and November 2002. The six storm events had peak flows ranging from 0.31 to 20.0 $m^3\ s^{-1}$ and had a recurrence interval of <1 to 2 years [*Gellis et al.*, 2009]. The resulting sediment fingerprinting results were weighted according to the sediment transported by each event and indicated the following source contributions: cropland, 46%; ditch beds, 34%; stream banks, 7%; and forest, 13% (Figure 7). Many parts of the Pocomoke River watershed were ditched to lower groundwater levels, thereby providing more farmable land, and the main stem channel of the Pocomoke River was actively channelized from the late 19th century until the late 20th century. Ditching, channel straightening, and continual dredging have created conditions favorable for channel-corridor erosion along the Pocomoke River.

Six storm events were sampled in the agricultural and mixed land use (forest, agricultural, and urbanizing) Mattawoman Creek between December 2003 and September 2004. The six storm events had peak discharges ranging from 5.58 to 9.49 $m^3\ s^{-1}$, and all had recurrence intervals that were close to 1 year [*Gellis et al.*, 2009]. The sediment fingerprinting results, which were also load-weighted, indicated the following source contributions: stream banks 30%; forest, 29%; construction areas, 25%; and cropland, 17% (Figure 7). The Mattawoman Creek watershed is within commuting distance of Washington, D.C. and drains a rapidly developing area with 182 ha (approximately 1.26 % of the watershed) under construction at the time of sampling. This may explain the importance of construction areas as a source of sediment. The significance of forest areas as a sediment source in the Mattawoman Creek watershed may indicate that the forests are being disturbed and degraded and are experiencing substantial erosion.

Twelve storm events were sampled in the urban and agricultural Little Conestoga Creek watershed between March 2003 and June 2004. The 12 storm events had peak discharges ranging from 5.4 to 28.9 $m^3\ s^{-1}$ [*Gellis et al.*, 2009]. Owing to the short period of discharge record for this station, recurrence intervals for these storms could not be determined. The load-weighted source contributions obtained for this watershed were stream banks, 23% and cropland, 77 % (Figure 7). ^{137}Cs inventories, measured in nine cropland sites in the Little Conestoga Creek watershed, provided an estimate of the average cropland erosion rate of 19.4 t $ha^{-1}\ yr^{-1}$. If this erosion rate is extrapolated to the 13% of the watershed that is under cropland, then cropland could contribute almost four times the measured suspended-sediment load transported out of the watershed (27,600 t yr^{-1}), indicating that much of the eroded sediment was deposited and stored in channel and upstream sinks. The results of the sediment fingerprinting analysis indicated that cropland contributed 77% of the sediment output from the watershed, and therefore, the sediment delivery ratio for sediment mobilized from cropland sources was 20%.

The sediment source fingerprinting results for the three watersheds draining to the Chesapeake Bay show that both the stream corridor and upslope areas have been important sediment sources and need to be considered when planning remediation measures. Little Conestoga Creek drains the Piedmont Province, which is the region of the Chesapeake Bay watershed with the highest specific sediment yields [*Gellis et al.*, 2009], and in the Little Conestoga Creek watershed, cropland was the dominant sediment source [*Gellis et al.*, 2009]. The ^{137}Cs measurements indicated that on-site cropland erosion rates were high, but that only a small proportion of the mobilized sediment reached the watershed outlet. The importance of storage areas in reducing the transfer of sediment eroded from the cropland areas to the

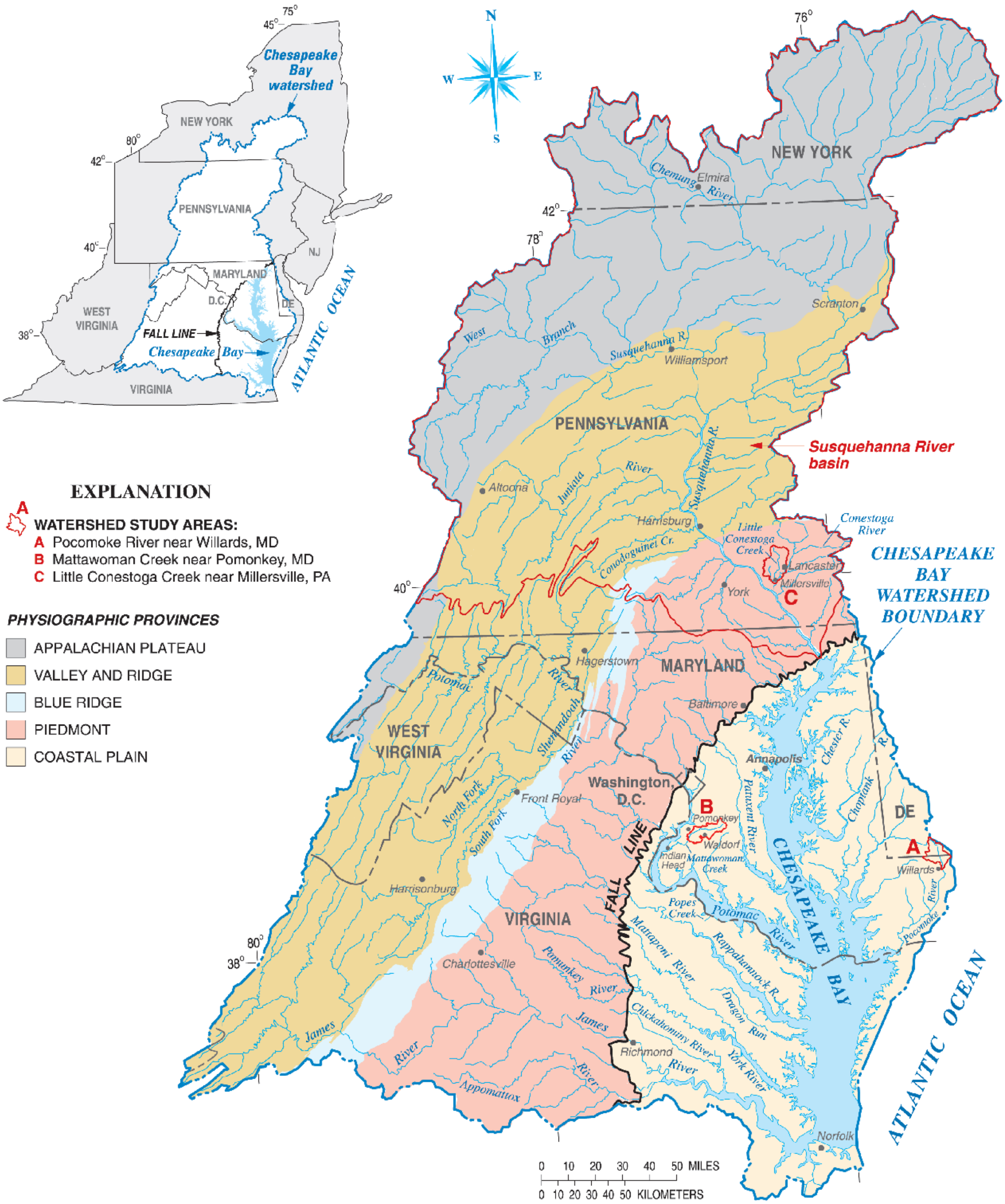


Plate 1. Location of three study watersheds for sediment source analysis in the Chesapeake Bay watershed: Pocomoke River near Willards, Maryland (area A), Mattawoman Creek near Pomonkey, Maryland (area B), and Little Conestoga Creek near Millersville, Pennsylvania (area C).

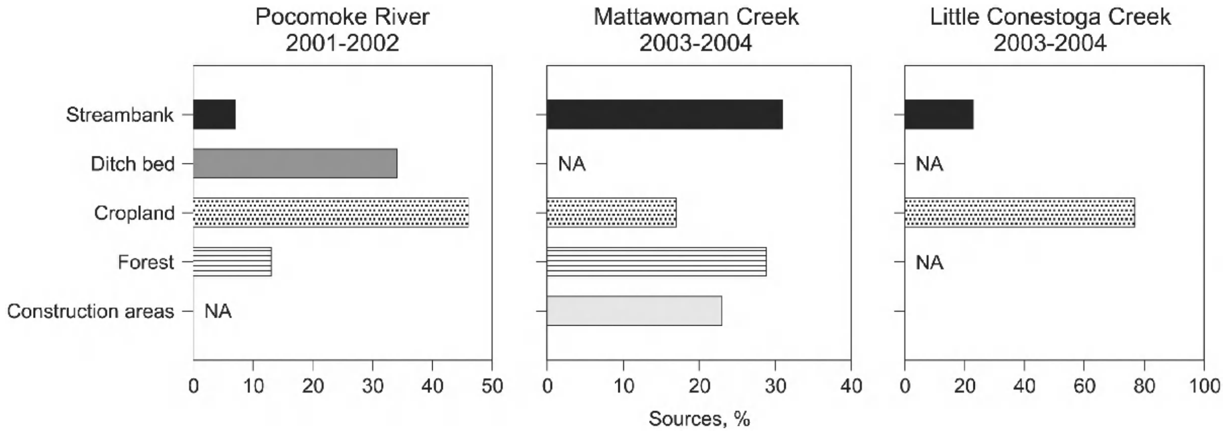


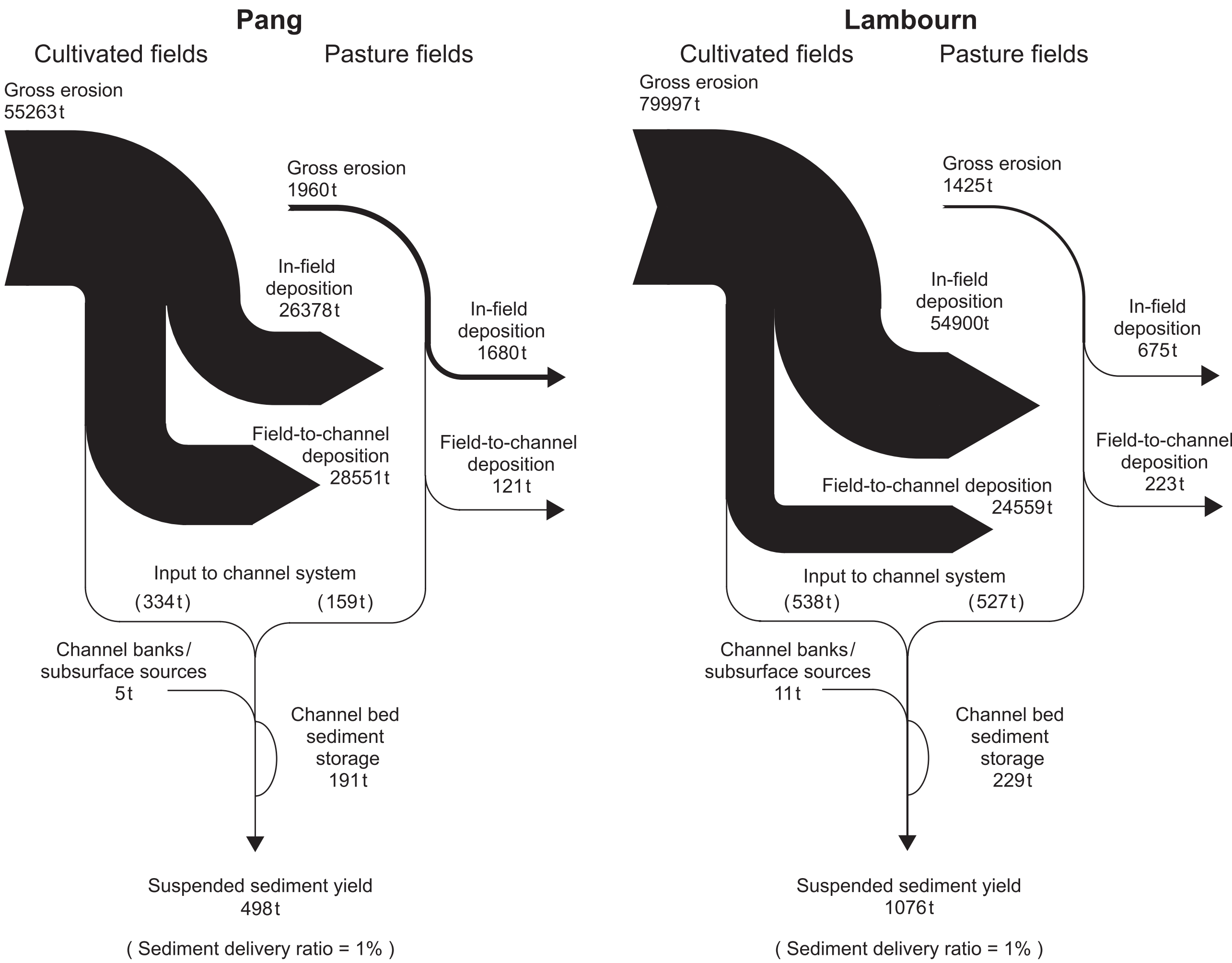
Figure 7. Results of sediment fingerprinting for the Pocomoke River, Mattawoman Creek, and Little Conestoga Creek. NA indicates sediment source was not investigated in the selected watershed.

watershed outlet would need to be recognized in future watershed management strategies, to ensure that these sink areas retain their role as sediment stores and that the stored sediment is not remobilized. The SPARROW model, sediment fingerprinting, and sediment budget approaches are currently being incorporated into approaches to target sediment sources by the Chesapeake Bay Program (a partnership of Federal, State, and citizen groups, State Environmental Agencies, and Counties).

4.3. A Chalk Area, Southern England, United Kingdom

A sediment source fingerprinting and sediment budget approach similar to that used in the study of the Kaleya watershed in Zambia described above was also used by *Walling et al.* [2006] to establish the key sources of sediment in the streams of a Chalk area in southern England, United Kingdom. In their undisturbed state, Chalk streams are groundwater-fed, clear-running streams that support a diverse habitat of plants and benthic invertebrates and important salmonid fisheries. With the intensification of land use, and particularly the conversion of permanent grassland to cropland, as well as the general development of these areas, increased suspended-sediment concentrations and loads have resulted in the degradation of the aquatic ecosystems and habitats and are frequently cited as the prime cause of reduced fish stocks. The study undertaken by *Walling et al.* [2006] was designed to obtain an improved understanding of the sediment sources and the functioning of the sediment budgets of such Chalk watersheds. Two watersheds in the Chalk area were chosen as study areas, the Pang (166 km^2) and Lambourn (234 km^2). A variety of tools were employed in the study, including ^{137}Cs measurements, sediment source fingerprinting, monitoring of sediment yields, and periodic measurements of the storage of fine sediment on the channel bed. ^{137}Cs measurements were used to provide information on sediment mobilization and redistribution rates on the watershed slopes. Sediment fingerprinting was used to establish the relative importance of major watershed sediment sources (cultivated areas, pasture, and stream banks). Fingerprints used included trace elements; inorganic, organic, and total P; and radionuclides. Measurements of the storage of fine sediment on the channel bed were performed by inserting a 1 m cylinder (with an area of 0.16 m^2) into the surface of the riverbed at several locations and resuspending the fines stored on the surface and in the top 5 cm of bed material. The estimates of fine sediment storage (mass per unit area) were scaled up to determine the total bed storage of fines within the two watersheds at the time of sampling [*Walling et al.*, 2006].

The sediment budgets and sediment fingerprinting results for these two watersheds (shown in Figure 8) emphasize the dominance of cultivated land and the limited contribution of stream banks as sediment sources. However, the sediment delivery ratios estimated for both watersheds were very low (about 1%), indicating that most of the sediment mobilized from the surface of the watershed was deposited and stored either in the fields or during transfer from the edge of fields to the stream network. From a management perspective, the interpretation that most of the sediment mobilized from the slopes of the watershed was deposited before reaching the stream channel is important because it indicates that a reduction in rates of soil erosion in the cultivated areas may have limited effect on the downstream sediment load. It also indicates that the watersheds are likely to be highly sensitive to any disturbance that could lead to increased connectivity between the slopes and the stream network and thereby increase the efficiency of sediment delivery. In these



FINGERPRINTING RESULTS

	Pang	Lambourn
Cultivated areas	67%	50%
Pasture	32%	49%
Streambanks	1%	1%

Figure 8. The annual suspended sediment budgets for the Pang and Lambourn study catchments, Chalk area, southern England, United Kingdom. Modified from *Walling et al.* [2006].

watersheds, maintaining, and perhaps even decreasing further, the limited connectivity between the slopes and channel network should be seen as a priority for future management [*Walling et al.*, 2006]. In addition, the sediment budgets shown in Figure 8 emphasize the importance of channel storage in retaining sediment within the channel system because they indicate that 20–40% of the annual fine sediment load was temporarily stored on the channel bed. Such storage is relatively short term in nature, demonstrating a seasonal cycle with accumulation during the summer and remobilization and removal during the winter. The results of this study indicated that most of the annual suspended-sediment load moving through the channel systems of the two watersheds entered temporary storage and that such storage was a key feature of the sediment response of the watersheds. Although only relatively small amounts of sediment were transported by the streams, it appears that these have increased in recent years, causing degradation of the aquatic ecosystems and associated habitats. To achieve an improvement, future management strategies should aim to reduce both the sediment input to the channel system and the storage of that sediment on the channel bed, possibly by increasing flows through reduction of groundwater abstraction [*Walling et al.*, 2006].

The results obtained from this study of the Pang and Lambourn watersheds have proved valuable in developing sediment management strategies for these and other groundwater-dominated chalk streams in the United Kingdom, by emphasizing the importance of maintaining and further reducing the limited connectivity between the watershed slopes and the stream corridor and reducing in-channel sediment storage. More generally, the findings of this and similar studies of other watersheds have contributed to the development of the Catchment Sensitive Farming Initiative promoted by the Department for Environment, Food and Rural Affairs and its Safeguarding Soils and Future Water program, by demonstrating the importance of slope-channel connectivity, emphasizing the need to consider sediment sources other than agricultural fields (i.e., channel erosion) as potentially contributing to downstream sediment fluxes and highlighting the potential importance of in-channel storage in attenuating sediment transfer through the main channel system.

4.4. *Zuni Reservation, New Mexico*

As a result of erosion problems on the Zuni Indian Reservation (1653 km^2) in western New Mexico, United States (Figure 9), some of which may date back to the 19th century, the Zuni Tribe began a program of watershed rehabilitation in the 1990s. A prioritization approach was

Table 4. Factors Considered in the Selection of a Subbasin for Rehabilitation in the Rio Nutria Watershed, Zuni Reservation, New Mexico[a]

Features	Y-Unit Draw	Lower Nutria	Coal Mine Canyon Draw	Burnt Timber Canyon	Crow Canyon	Wetlands	Garcia Draw	Benny Draw
Basin area (km^2) (draining within the Zuni Reservation)	24.7	103	26.0	28.7	4.5	39.8	30.1	3.0
Physical features								
Headcut density (headcuts km^{-2})	1.5	0.8	6.0	6.7	6.8	1.0	8.8	17.5
Change in arroyo density, 1934–1988 (m km^{-2})	3	−39	151	−18	239	120	170	70
1988 arroyo density (m km^{-2})	145	17	436	398	573	250	385	441
Width-to-depth ratios (m m^{-1})	7.3	5.6	11.7	5.5	ND	3.6	5.0	3.5
Average sheetwash erosion rates (ppm)	2662	10,609	5033	6395	838	4928	5425	914
Qualitative physical features (rankings are from 10, the potential for the most erosion, to 1, the potential for the least erosion)								
Average value of main channel erosion	5.3	3.3	7.3	7.7	5.3	2.7	5.3	5.7
Average value of tributary erosion	4.8	3.4	7.4	7.4	1.4	7.0	7.6	7.0
Visual estimate of watershed erosion	6	3	9	7	4	5	9	10
Roads								
1988 density of dirt roads (m km^{-2})	834	833	1161	1283	1299	1541	1067	1807
Change in dirt road density, 1934–1988 (m km^{-2})	559	384	1063	1092	935	972	723	1807
Erosion control structures								
Number of failed dams	0	0	9	6	0	7	3	1
Number of structures more than 50% silted	0	0	10	5	1	5	5	3
Number of headcuts below structure	0	0	4	4	0	3	1	0
Socioeconomic factor								
Agricultural area (%)	0	1.0	0	0	3.0	11.9	3.3	0

[a]Locations of subbasins are shown in Figure 9b. ND indicates no data available. From *Gellis et al.* [2001b].

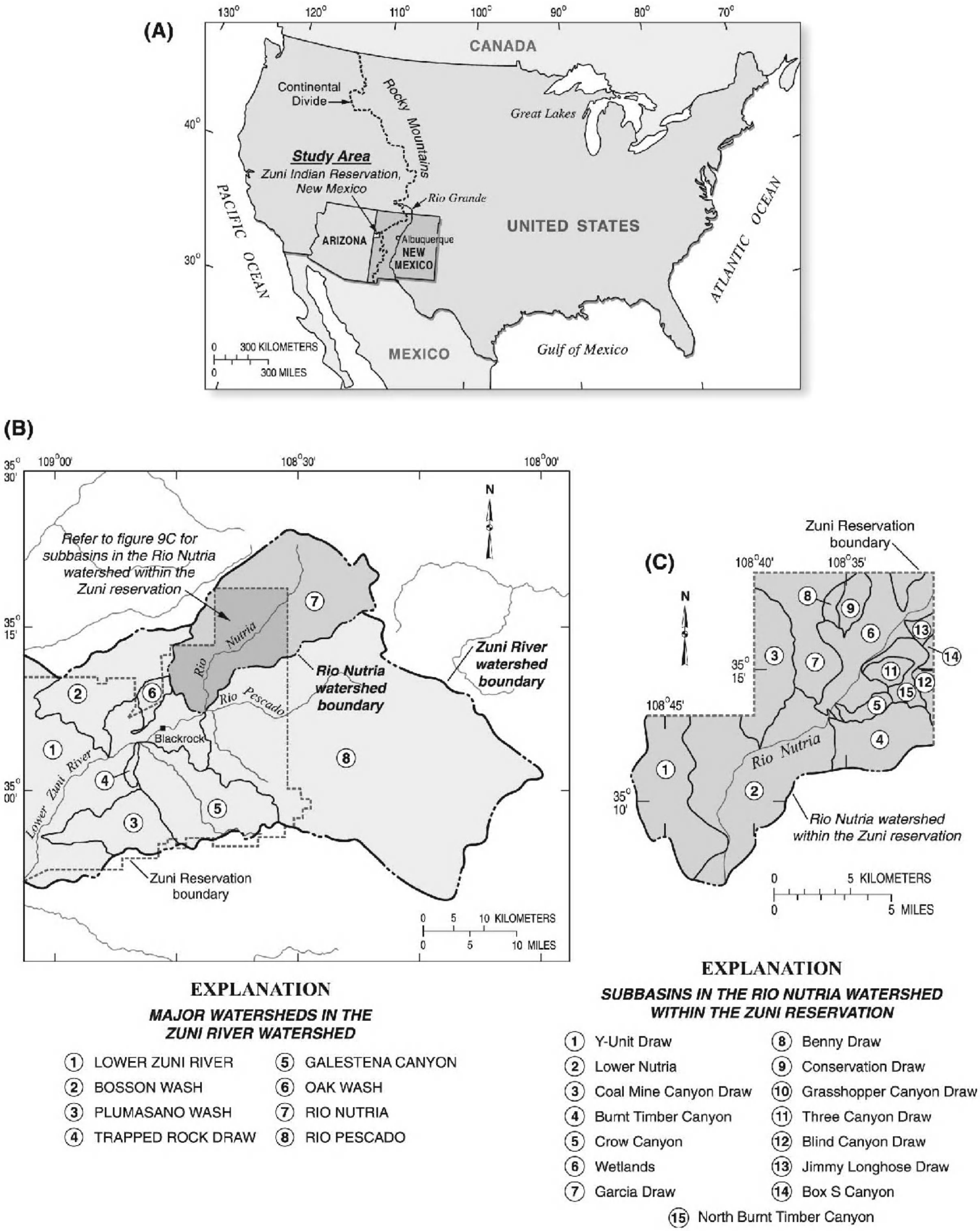


Figure 9. (a) Map of the United States highlighting the study area. (b) Location of major watersheds in the Zuni River watershed, Zuni Reservation, New Mexico. (c) Location of subbasins in the Rio Nutria watershed. Modified from *Gellis et al.* [2001b].

developed to rank the most important watersheds for rehabilitation [*Gellis et al.*, 2001b]. The approach was based on information on geomorphic and anthropogenic characteristics collected during a 3 year study (1992–1995) in a pilot watershed draining the Zuni Reservation, the Rio Nutria watershed (602 km^2) (Figure 9). Information collected in this watershed included (1) changes in the cross-sectional area, width, and depth of incised channels (arroyos), (2) information on the condition of historical erosion-control structures, and (3) an assessment of sheetwash erosion. The results of the initial survey undertaken during the 3 year study indicated that 61 out of 85 arroyo cross sections aggraded, whereas channels with lower width-to-depth ratios appeared to be eroding. The results of the resurveys indicated that narrow, deep channels were more erosive and that measurements of changes in cross-sectional area over time provided a better indicator of channel erosion than measurements of channel-bed lowering. The assessment of historical erosion-control structures, some dating back to the 1930s, indicated that 60% of the earth dams and 22% of the rock-and-brush structures in the Rio Nutria watershed had been breached or isolated [*Gellis et al.*, 1995]. The failure of these structures led to the headward migration of headcuts through the channel deposits and a renewed cycle of erosion. Sheetwash erosion measured at five sites representative of different land covers (sagebrush, pasture, chained piñon and juniper, unchained piñon and juniper, and ponderosa pine) indicated that chained piñon and juniper sites and pasture sites generated the highest volume-weighted sediment concentrations of 13,000 and 9970 ppm, respectively. Chaining is a procedure used to clear vegetation, such as piñon and juniper, to encourage growth of grass for livestock. Chaining is accomplished by dragging a heavy chain through trees between two vehicles to uproot the vegetation. Owing to the nature of chaining, where trees are uprooted, vegetation may be slow to recolonize these areas effectively, and large areas of bare ground were present.

Once the pilot study was completed, interpretations derived from the study were used in a two-stage process to rank watersheds in terms of the need and scope for rehabilitation. In the first stage, the Zuni Reservation was divided into eight major watersheds (9.4 to 480 km^2) (Figure 9a). For each watershed, data on physical condition factors (headcut density, percentage of bare ground, percentage of chained areas, and a qualitative ranking of watershed erosion) and anthropogenic and socioeconomic factors (density of dirt roads, number of failed earthen dams, and area of agricultural land) were collected. The local community was also surveyed to determine its perception of the major watershed most in need of rehabilitation. A system for scoring the various factors based on weighted scores was devised, and the eight watersheds were ranked, based on the summed scores. The Rio Nutria watershed was selected as the priority watershed for rehabilitation (Table 4). In the second stage, the Rio Nutria watershed was subdivided into 15 subwatersheds (3 to 103 km^2) (Figure 9b). For each subwatershed, data on physical factors (including headcut density, arroyo density in 1988, channel width-to-depth ratios, sheetwash erosion, change in arroyo density per unit basin area from 1934 to 1988, and a qualitative assessment of watershed erosion) and anthropogenic factors (changes in dirt road density per basin area, dirt road density in 1988, characteristics of earthen dams built for erosion control, and percentage of agricultural area) were assembled (Table 4). The scores for each subwatershed were summed, and the subwatersheds were ranked according to their priority for rehabilitation. The three subwatersheds selected as priorities for rehabilitation were Benny Draw, Coal Mine Canyon Draw, and Garcia Draw (Table 5). This study highlighted the use of geomorphic measurements and cultural assessments to target specific watersheds for rehabilitation by use of a ranking system. The results of this study and the rankings of watersheds for rehabilitation were subsequently used by the Zuni Tribe to prioritize the location of erosion-control measures.

Table 5. Results for Selection of the Most Appropriate Subbasin for Rehabilitation in the Rio Nutria Watershed[a]

Subbasin[b]	Sum of Rankings[c]
Benny Draw	165
Coal Mine Canyon Draw	164
Garcia Draw	162
Burnt Timber Canyon	157
Wetlands	146
Conservation Draw	136
Three Canyon Draw	136
Crow Canyon	135
Blind Canyon Draw	130
North Burnt Timber Canyon	116
Box S Canyon	115
Y-Unit Draw	114
Jimmy Longhose Draw	109
Lower Nutria	104
Grasshopper Canyon Draw	93

[a]From *Gellis et al.* [2001b].

[b]Location shown in Figure 9b.

[c]The highest value for ranking indicates the subbasin with the most erosion.

5. SUMMARY AND CONCLUSIONS

In restoration projects where the goal is to reduce sediment flux or erosion, it is important to target the most important sediment sources. The two most important watershed sources of sediment are typically stream bank erosion and upslope areas (areas comprising various land uses). Erosion-control strategies vary widely between these two critical sediment sources, and thus, it is important to assess their relative contribution. By targeting the important sediment sources, BMPs will be more cost effective in reducing loads. As BMPs are implemented, it may prove beneficial that "before and after" monitoring of sediment loads can be undertaken in order to provide directly measured assessments of the success of those strategies, as distinct from "modeled" or "predicted" assessments. This information will likely prove valuable in developing improved estimates of the efficiency of management practices in reducing nutrient and sediment loads and can be further extended by the use of source fingerprinting techniques to include information on posttreatment changes in sediment sources.

Prior to the implementation of stream or watershed rehabilitation practices aimed at reducing erosion and the associated sediment flux, an assessment phase could be conducted whereby sediment sources are investigated using two approaches: (1) the sediment fingerprinting approach and (2) the sediment budget approach (Figure 5). The sediment fingerprinting approach quantifies the main sources of the sediment that is exported from a given watershed. The sediment budget approach provides information on the location within the watershed of the sediment sources and on the importance of sinks and stores in influencing the connectivity of these sources to the watershed outlet. For example, if sediment fingerprinting indicates that cropland erosion is an important source of the sediment yield from a watershed, but if only a small proportion of the sediment mobilized from cropland areas reaches the watershed outlet, reduction of cropland erosion rates may not result in a substantial reduction in sediment yield. In contrast, stream bank erosion results in a direct input of sediment to the river channel, and there is often less opportunity for sinks to reduce this input. As a result, the control of stream bank erosion may have a more direct and immediate impact on the downstream sediment load. If stream bank erosion is measured as part of the sediment budget, this data can be used to identify "hot spots" in the watershed and may be useful for resource managers to target resources to these stream reaches.

The sediment fingerprinting and sediment budget approaches will be most successful in what are defined as management-scale basins (<300 km^2). The case studies presented here illustrate the use of both of these approaches in the development of management strategies to reduce erosion and sediment transport. Information provided by the combined sediment source fingerprinting and sediment budget approach and its interpretation can, in turn, provide valuable input to the design of BMPs for reducing sediment problems in rivers and their watersheds.

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Closing the Gap Between Watershed Modeling, Sediment Budgeting, and Stream Restoration

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The connection between stream restoration and sediment budgeting runs both ways: stream restoration is proposed as a means to reduce sediment yields, but an accurate understanding of sediment supply is necessary to design an effective project. Recent advances in monitoring technology, geochemical techniques, high-resolution topography data, and numerical modeling provide new opportunities to estimate sediment erosion, transport, and deposition rates; upscale them in a geomorphically relevant fashion; and synthesize sediment dynamics at watershed scales. For practical application at large scale, watershed models used to predict yield often do not resolve lower-order channels, leaving an essential "blind spot" regarding sediment processes. We illustrate the challenges and emerging approaches for estimating sediment budgets using examples from two very different physiographic settings: the Mid-Atlantic Piedmont and the agricultural plains of southern Minnesota. We highlight common challenges and themes in defining an effective watershed sediment model. In both cases, reliable estimates of sediment yield depend essentially on the accurate identification of sediment sources and sinks and, hence, require careful delineation of landscape units and identification of dominant sediment sources and sinks. The primary elements needed to bridge the gap between sediment budgeting, watershed modeling, and stream restoration are (1) specificity regarding location, mechanism, and rates of erosion, (2) accurate accounting of sediment storage, (3) appropriate methods for upscaling local observations, (4) efficient means for incorporating multiple lines of evidence to constrain budget estimates, and (5) stream restoration methods that incorporate sediment supply in assessment and design procedures.

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1. INTRODUCTION

The need for estimates of sediment sources and yields is not new; this is a field with a long and rich history of research and application. Sediment fluxes and their mass balance, a sediment budget, have been developed for both research and management purposes. The need for such work has intensified with the increasing recognition of the effect of sediment and turbidity on the health of receiving waters and with the advent of total maximum daily loads (TMDLs) specified for sediment or turbidity reduction. Stream restoration is increasingly viewed as a viable option for reducing sediment loads.

The connection between stream restoration and sediment budgets runs both ways: stream restoration is not only proposed as a means to reduce sediment yields, but an accurate understanding of sediment supply is often needed to design an effective stream restoration project. The two directions are closely linked in practice, addressing questions such as the following: Where is the best place in the watershed to reduce sediment yield? Over what time period will sediment reductions occur at the watershed outlet? What is the sediment supply to a designated restoration site over different time scales? How can information on location and rates of erosion and deposition guide the selection of best management practices? In either direction, practical application of sediment budget information requires that sources and sinks be specifically identified as to location, mechanism, controls, and rates.

Watershed hydrologic models are increasingly used to predict sediment yields. By predicting water flux and applying a sediment mass balance, such models provide a potentially powerful tool for estimating sediment supply and yield. They have also increased in their resolution and the number of physical processes that are simulated [e.g., *Flanagan and Nearing*, 1995; *Langendoen*, 2002; *Neitsch et al.*, 2005]. However, such models face difficult challenges when applied to a range of watershed scales.

1. Sediment erosion and deposition are extremely variable in place and in time, with the bulk of sediment movement often happening in highly localized, short-term events, which makes prediction of sediment yield as a function of temporal and spatial mean quantities prone to large error.

2. Entrainment, transport, and deposition mechanisms are nonlinear with respect to the driving water flux and the sediment available for transport. This leads to potentially large errors from even relatively small errors in flow and sediment input.

3. The fraction of eroded sediment that is stored between source and sink can vary from zero to unity and the duration of storage can range from intraevent to geological. Some watershed models now include overland and channel components and can compute storage changes at a fine spatial and temporal scale, but none have been demonstrated to adequately represent sediment storage and release across all scales.

For practical application at large scale, watershed models often do not resolve lower-order channels, leaving an essential "blind spot" regarding sediment processes. Low-order channels can act as net sources or sinks of sediment. Their dynamics can include a suite of mechanisms that differ strongly from those acting within upland hillslopes or larger valley bottoms. These distinctions are essential in developing a reliable estimate of sediment supply and for focusing restoration efforts. If, for example, a watershed model includes only third- and higher-order channels, the sediment dynamics of first- and second-order streams are necessarily grouped into a simple, often scalar parameter that specifies the fraction of the upland sediment production delivered to the stream network. Because a large fraction of most watersheds is drained via first- and second-order channels, representing these features by a simple filter or delivery factor can result in substantial error. Values reported for sediment delivery ratios (the ratio of sediment yield to sediment production) vary from >1 to <0.1 [*Walling*, 1983; *De Vente et al.*, 2007], indicating that reliable, independent estimates of sediment sources and sinks are essential if watershed sediment budgets are to be successfully connected to stream restoration projects.

The emerging availability of high-resolution topography and GIS offers the opportunity for more realistic representation of sediment processes in low-order subwatersheds, but reliable and efficient methods have not yet been assembled into a widely used package. One approach to addressing the resolution problem uses watershed models that implement physical relations governing sediment production, flux, storage, and delivery at high spatial resolution [e.g., *Flanagan and Nearing*, 1995]. Although defined explicitly, the mechanisms incorporated may not represent the actual suite of mechanisms and their rates at the process scale. Indeed, the physical basis for these models can become a limitation when insufficient information is available to specify the many detailed boundary conditions required. Further, the specific physical relations used in these models must be applied to a wide range of topographic and hydraulic conditions over which they are unlikely to apply consistently.

In response to these challenges, watershed sediment models can be modified to incorporate independent information on sediment sources and sinks. For example, a Hydrological Simulation Program—Fortran (HSPF) model of the Minnesota River Basin uses sediment fingerprinting results to constrain the proportion of sediment derived from different sources [*Tetra Tech, Inc.*, 2008]. In the examples presented here, net sediment contributions from colluvial deposits, floodplains, and stream banks are determined from direct observation and upscaled using topographic analysis to estimate the area and location of sites serving as net sources and sinks. Sediment fingerprinting techniques are used to estimate the proportion of the yield derived from agricultural fields. When such independent information is used to constrain the results of a watershed model, the model provides a useful role as an accounting system for the sediment mass balance, but its ability to predict future sediment yield is no better than the independent information used.

The need for direct observation of sediment sources and sinks and for using multiple lines of evidence to constrain a sediment mass balance differs little from sediment budgets assembled in the predigital era. The challenge at present is to develop a system within which the power of watershed numerical models can fully integrate available information and for which the predictive capability of supplemental information is demonstrated. The nature of the information will necessarily vary with circumstance and conditions in different watersheds, and an effective combination of approaches is needed to close the gap in predicting watershed sediment yield.

The primary elements needed to bridge the gap between sediment budgeting, watershed modeling, and stream restoration are (1) specificity regarding location, mechanism, and rates of sediment erosion, (2) accurate treatment of changes in sediment storage, (3) appropriate methods for upscaling local observations, (4) efficient means for incorporating multiple lines of evidence to constrain budget estimates, and (5) stream restoration methods that effectively incorporate sediment supply in assessment and design procedures. A combination of existing and new technology provides an excellent opportunity to estimate sediment sources and sinks in a manner that discretizes over space and integrates over time, including (1) field observations and spatial analysis of topography, soil distribution, and land cover to locate, quantify, and upscale erosion estimates in a way that accounts for the effects of geomorphic setting and watershed location on sediment supply and (2) measurements of sediment accumulation in ponds, reservoirs, and lakes combined with radiogenic and isotopic chemistry methods for sediment fingerprinting and dating to develop a reliable estimate of sediment yield over decade to century time scales in order to provide a strong constraint on estimated sediment budgets.

We illustrate the challenges and emerging approaches for estimating sediment budgets using examples from two very different physiographic settings: the Mid-Atlantic Piedmont and the agricultural plains of southern Minnesota. Relief, watershed age, climate, and land use histories differ substantially between the two. However, reliable estimates of sediment yield and specification of restoration alternatives depend essentially on accurate identification of sediment sources and sinks in both cases, phenomena that have not been well captured in existing modeling approaches. The cases we describe in this chapter do not represent the balance of sediment processes in all regions. For example, sediment budgets in mountainous and arid watersheds can be dominated by episodic delivery of coarse sediment to the channel network, processes that are present but less significant in the cases presented here. Although the imperative to accurately identify mechanisms, locations, and rates of sediment delivery is the same in mountainous watersheds, the spectrum of processes and the methods needed to quantify them (e.g., landslide and road inventories) are different and have been well summarized by the work of *Reid and Dunne* [1996, 2003].

2. APPROACHES FOR WATERSHED SEDIMENT MODELING

2.1. Universal Soil Loss Equation

The universal soil loss equation (USLE) has been a primary tool for estimating long-term average erosion rates for decades [*Wischmeier and Smith*, 1978; *Soil and Water Conservation Society*, 2003]. This approach applies estimated rainfall and runoff conditions to erosion, soil erodability, slope conditions, and land management techniques. Extensive data from plot studies have been assembled throughout the past century in support of the model. The USLE has often been used to predict upland sediment supply. However, a shortcoming of the model and its revised forms, RUSLE and RUSLE2, is the ability to relate erosion at the plot scale to sediment delivery to the river channel network and outlets of large watersheds [*Renard et al.*, 1997; *Trimble and Crosson*, 2004]. This is the original “gap” between sediment production and yield that recent work has tried to address. Accounting for sediment delivery motivated the development of a subsequent version of the model called the modified USLE or MUSLE through direct consideration of runoff rates and hillslope curvature [*Williams*, 1975]. The MUSLE approach was designed to estimate sediment delivered from small watersheds for individual storms.

Wischmeier and Smith [1978] identified limitations of the USLE for predicting sediment supply. Later modifications improved adaptability, time resolution, and prediction of small watershed sediment delivery, but the model is fundamentally limited by the lack of terms to estimate erosion, deposition, and transport in both colluvial and alluvial settings. Accurately scaling up USLE estimates to large watersheds has been criticized for being impractical for these reasons [*Boomer et al.*, 2008].

Despite the limitations, USLE and its descendents remain highly useful for the appropriate purpose: estimating sediment yield at the field scale, particularly because of the rich legacy of plot observations and broad availability of Natural Resources Conservation Service (NRCS) county soil surveys, which include site-specific values needed to use the model. The USLE remains the most thoroughly tested approach for field-scale erosion estimation. At the same time, there is abundant evidence that USLE cannot do what it was not intended to do: estimate the transport and fate of sediment once it leaves the field.

2.2. Hydrologic Models With Sediment Flux Components

Demands for large basin sediment yield estimates have led to widespread use of watershed hydrology models as loading and transport simulation tools. Multiple models have been developed that used lumped parameter approaches to watershed simulation. Among those commonly used in the United States are the Soil and Water Assessment Tool (SWAT) [*Arnold et al.*, 1998] and HSPF [*Bicknell et al.*, 2001]. HSPF is a component of the Better Assessment Science Integrating Point and Non-point Sources environmental analysis system and a primary watershed modeling tool of the U.S. Environmental Protection Agency (U.S. EPA). HSPF provides a platform for continuous simulation of surface and subsurface hydrology and suspended sediment transport [*Donigian and Huber*, 1991; *Bicknell et al.*, 2001; *U.S. Environmental Protection Agency (U.S. EPA)*, 2008]. The model allows for the integrated simulation of land and soil runoff processes coupled with terms to represent simplified river hydraulic conditions related to sediment deposition and transport.

HSPF is framed with some physical basis for detaching and routing sediment downstream. Nonetheless, the modeling "gap" remains in its application, as illustrated by the application of the model to the Chesapeake Bay watershed. Erosion from the land surface is simulated using a continuous time series of precipitation combined with specified land uses to calculate edge-of-field (EOF) loads that are calibrated to estimates of soil erosion from the RUSLE and adjusted relative to the efficiency of implemented best management practices (NRCS, National Resources Inventory, 2003—Soil Erosion, U.S. Department of Agriculture, 2007, accessed 23 June 2011, http://www.nrcs.usda.gov/technical/NRI/2007/nri07erosion.html, hereinafter referred to as NRCS, data, 2007). EOF loads are delivered to the stream network after reduction by a scalar sediment delivery factor that is a function of drainage area [*Roehl*, 1962; *Natural Resources Conservation Service (NRCS)*, 1983; *U.S. EPA*, 2008]. In the latest version of the Chesapeake model, the minimum stream size is prescribed by an annual average flow rate of 2.83 m^3 s^{-1}, which typically corresponds to streams of third or fourth order. Smaller river segments can be included in simulations, but headwater streams are not modeled when HSPF is applied for TMDL purposes in most watersheds in Maryland where we have focused attention here. The relatively large size of the rivers considered by the model and exclusion of smaller tributaries establishes a substantial gap in the watershed simulations (Figure 1). The range of erosion and deposition processes in lower-order streams are complex and vary among physiographic settings, making the universal application of a single delivery operator in the model a large source of uncertainty.

2.3. Hydrologic/Hydraulic/Geomorphic Erosion Models

More recent tools used by the U.S. Department of Agriculture (USDA) to estimate soil erosion have been compiled within the Water Erosion Prediction Project (WEPP). WEPP

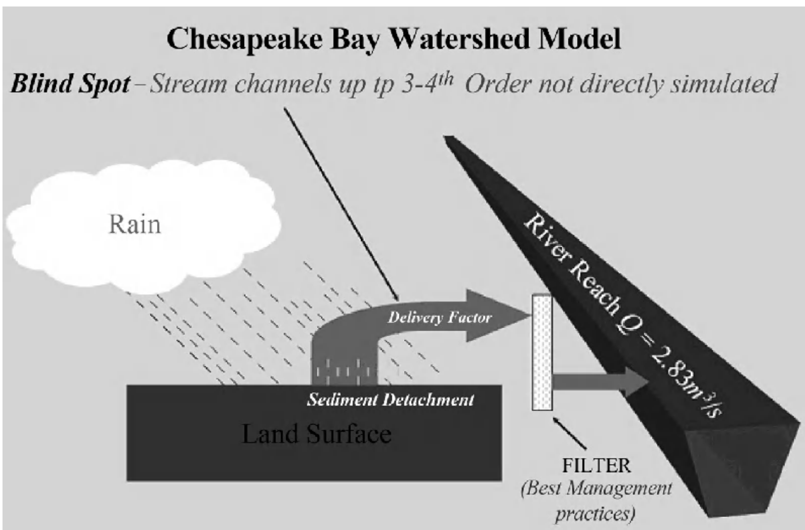

Figure 1. Chesapeake Bay watershed model framework. Source is G. Shenk, U.S. EPA Chesapeake Bay Program.

uses a process-based simulation approach for hillsides and small watersheds [*Foster and Lane*, 1987; *Flanagan and Nearing*, 1995]. WEPP components include databases and subroutines for climate, hydrology, hydraulics, plant growth, and soil conditions. Sediment erosion and deposition are simulated using a steady state continuity equation and process-based transport rates for fields and small waterways. Gully erosion processes are not included despite their relevance to watershed sediment yield [*Howard*, 1999]. Accumulating local calculations of flow and sediment transport to larger scales requires specification of spatially and temporally varied soil, hydraulic, topographic, and vegetation conditions. Parameterization and input specification is a daunting task for practical application of process-based models like WEPP to large watersheds [*Scatena*, 1987].

Channel processes are explicitly incorporated in the USDA Conservational Channel Evolution and Pollutant Transport System [*Langendoen*, 2000, 2002]. The model simulates sediment transport and channel morphology using an unsteady one-dimensional hydraulic model that relates calculated transport capacity to upstream sediment supply in order to determine sediment erosion and deposition. Channel width adjustment can be estimated based on simulated bank material entrainment and bank gravity failure using input streambed and bank information. Floodplain processes are not simulated, so the effects of overbank flooding on hydraulic conditions and sediment storage are not quantified. Although the model includes a larger suite of physical mechanisms than spatially lumped models, model accuracy still faces the challenges of unresolved local heterogeneities and error amplification when using averaged quantities to estimate flux with nonlinear relations.

Watershed models bring obvious benefits to the problem of estimating sediment supply and yield. Physical mechanisms can be explicitly incorporated, sediment can be routed over long distances, and the models can provide a useful basis for developing a sediment mass balance. Each model has strong points and weaknesses, but none provides a complete framework that reliably identifies and predicts all production, transport, and storage terms at the appropriate time and space scales within a system that is practicable for typical watershed management and stream restoration applications. We argue that successful sediment supply and yield estimates must combine watershed modeling with the classical sediment budget imperative to apply multiple lines of independent evidence. This evidence can be developed using a mix of existing and new field, remote sensing, fingerprinting, and analysis techniques. A key challenge is to develop a watershed modeling system that can accommodate a diverse range of local and integral measures of sediment flux and storage.

3. SEDIMENT YIELD IN THE MID-ATLANTIC PIEDMONT PROVINCE

3.1. Site Description

The Piedmont Plateau physiographic province comprises nearly 23% of the 165,759 km^2 Chesapeake Bay watershed [*Langland et al.*, 1995]. The province is an old, dissected landscape dominantly composed of metamorphic crystalline bedrock such as schist, quartzite, and gneiss, with some areas underlain by carbonate bedrock [*Smith et al.*, 2009]. The Blue Ridge physiographic province abuts the western side of the Piedmont, and the eastern side has a boundary coinciding with a relatively abrupt drop in the bedrock surface below an overburden of Coastal Plain sediment. This “fall zone” transition of the bedrock defines the head of navigable waters and a location attractive for hydropower in the colonial period, focusing development of urban centers that continue to grow today.

Once dominated by temperate humid forests, large-scale European colonization began about 350 years ago, resulting in extensive deforestation of the landscape and conversion of the land to agriculture [*Grumet*, 2000]. Forest cover of the region was smallest around the turn of the nineteenth century, with some recovery occurring in the twentieth century as a result of the decline in agriculture [*Brush*, 2008]. A second significant landscape conversion is still underway with suburban development increasing in the region over the past century.

Historic changes in land and river use have created a complex system of watershed sediment supply and delivery in the contemporary landscape. The conversion to agriculture and transition to suburban development substantially altered watershed hydrology and sedimentation patterns [*Gottschalk*, 1945; *Wolman and Schick*, 1967; *Jacobsen and Coleman*, 1986]. Extensive soil erosion during the peak agricultural period produced intense sediment delivery to valley bottoms and eventually to the Chesapeake Bay. A large fraction of the eroded sediment was stored as colluvium in upland areas or alluvium within valleys [*Costa*, 1975]. Evacuation of the deposits may take hundreds to thousands of years at the current rates of removal and replacement [*Scatena*, 1987]. Widespread construction of mill ponds augmented the storage of fine sediment along river channels [*Happ*, 1945; *Walter and Merritts*, 2008]. Breaching or intentional removal of these dams represents a potentially important perturbation and modern source of fine sediment to the channel network [*Schenk and Hupp*, 2009].

Concern about sediment supply has increased over the recent decades, as efforts to improve water quality in the Chesapeake Bay have proved largely ineffective [*U.S. EPA*, 2008]. Sediment and turbidity, along with nitrogen and phosphorous, are identified as critical pollutants requiring reduction [*U.S. EPA*, 2010; Chesapeake Bay Program, Chesapeake 2000, Chesapeake Bay agreement, 2000, available at http://www.chesapeakebay.net/content/publications/cbp_12081.pdf]. Stream bank stabilization and floodplain storage are increasingly seen as alternatives for reducing sediment loading to the Bay [*Langland and Cronin*, 2003; *Hassett et al.*, 2005]. Management efforts require specificity regarding sediment source location and amount, which provided the motivation to develop a sediment budget for a Piedmont watershed in Maryland.

We focus here on the upper Patuxent River watershed (UPRW), a 203 km^2 watershed draining relatively homogenous Mid-Atlantic Piedmont physiography (Figure 2) [*Reger and Cleaves*, 2003]. Land cover in the watershed is a mix of forest, field, and suburban development whose proportions have remained relatively stable over the past half-century. The Patuxent River is a fifth-order tributary at the downstream extent of the study area, where it is impounded by the Triadelphia Reservoir. The reservoir was constructed in 1943, and approximately decadal bathymetric surveys provide a record of sediment yield over more than 50 years. A U.S. Geological Survey (USGS) gauge at the town of Unity is located on the Patuxent River main stem immediately above the reservoir and has provided a continuous flow record from 1944 with periodic measurements of suspended sediment [*Lizarraga*, 1999]. Sediment load has also been estimated for the UPRW for the purpose of TMDL requirements using flow records, sediment grab samples, USGS

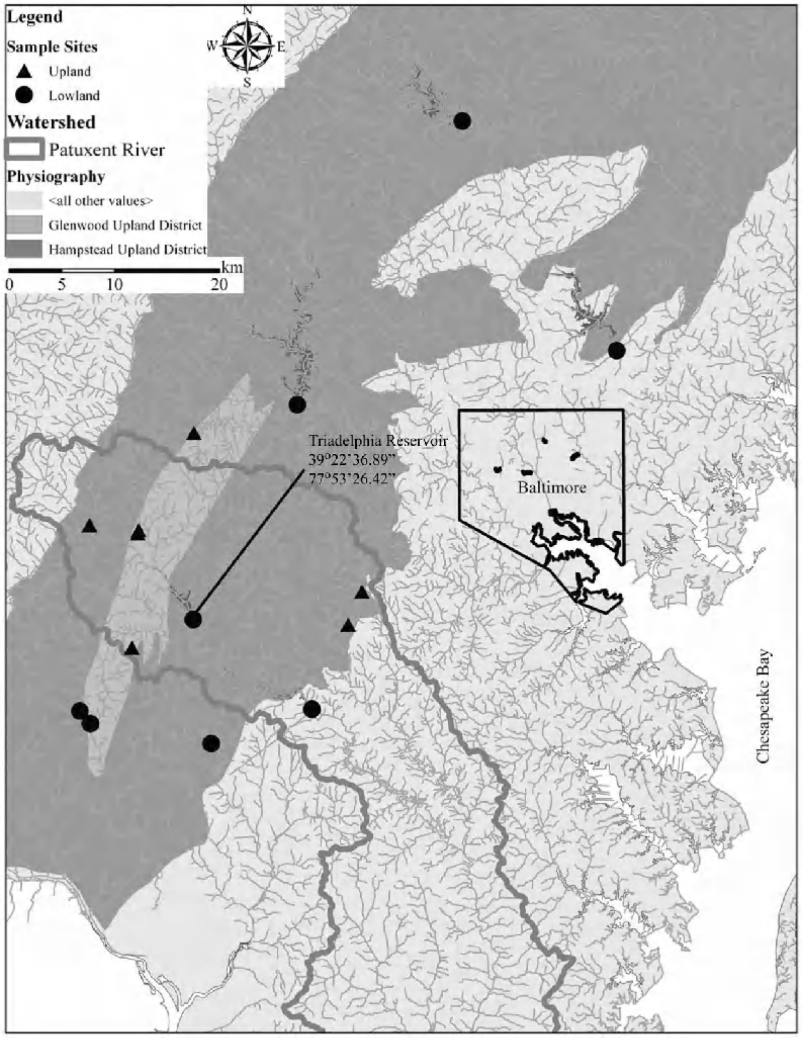

Figure 2. Physiographic districts and landform analysis study areas. Study areas were located in either upland (triangles) or lowland (circles) settings.

ESTIMATOR software, and an HSPF watershed model [*Cohn et al.*, 1989; *Interstate Commission the Potomac River Basin (ICPRB)*, 2006].

The starting point for understanding sediment processes in the UPRW involved delineation of relevant landscape units and the channel network. Field observations and measurements were used to estimate sediment production rates for upland landscape units. Sediment yield for first-order watersheds was determined from accumulation in ponds. The subwatersheds selected for study in the UPRW and adjacent areas were dominated by one of the three land cover conditions typical of the region: suburban, agricultural, and forest. The measurements in the basins thereby provided an indication of sediment yield as a function of land cover. Sediment yield was upscaled based on the area of relevant landscape and land cover units, then evaluated against sediment accumulation in the Triadelphia Reservoir, as well as other large impoundments in similar physiographic settings.

3.2. Landscape Delineation

The study area and the entire UPRW lie within the delineated boundaries of two similar Piedmont subunits, the Hampstead Uplands District and the Glenwood Uplands District (Figure 2) [*Reger and Cleaves*, 2003]. Both districts have predominantly crystalline bedrock and modest relief of less than 100 m, with exception of areas within the major fall zone gorges at the eastern boundary. We examined the portion of watersheds entirely upstream of the fall zone region.

Accounting for contributions to contemporary sediment yield requires accurate delineation of landforms relevant to the quantification of net erosion and storage in the landscape. The landscape was broadly divided into upland and lowland complexes [*Cleaves*, 1974]. Key objectives of the delineation were to define the location of (1) upland landform subunits with consistent controls, mechanisms, and rates of sediment production, (2) channel heads and, therefore, the extent of the channel network, and (3) the channel network transition from dominantly erosional (with little to no sediment storage) to alluvial with floodplain storage. Based on the typical observation that first-order channels generally do not have active floodplains, we broadly divided the landscape into upland and lowland landform units at the confluence of first-order and higher-order channels. The general landform partition conformed to the classification considered by previous investigations and has relevance to the dominance of erosion- or transport-limited conditions [*Cleaves*, 1974; *Costa and Cleaves*, 1984; *Howard*, 1999].

3.2.1. Uplands. Nearly balanced chemical weathering and mechanical erosion over the Quaternary Period produced a dissected, dendritic tributary drainage network in upland portions of the Piedmont [*Cleaves*, 1974; *Costa and Cleaves*, 1984; *Pavich*, 1989]. However, increased rates of erosion from runoff over the past three centuries have substantially increased the rates of mechanical erosion relative to chemical weathering [*Langland and Cronin*, 2003]. Factors governing production and conveyance of runoff in the modern landscape play a key role in determining the magnitude and extent of continued dissection, the resulting sediment supply, and the transport efficiency within and from upland areas.

Upland landform subunits include hillslopes, hollows, and channels [*Hack*, 1960]. Hollows are vaguely defined but can be described as nonchanneled or zero-order upland valleys that form shallow concentrated surface runoff patterns in response to precipitation events. Sediment yield from these units was determined using field observations and event-based flow and sediment monitoring. The yield from upland units is strongly influenced by both present and past land cover conditions. For example, reforested agricultural land can produce relatively large rates of overland flow and sediment transport that may be explained by the removal of surficial soil horizons, leaving less permeable soil at the ground surface [*Costa*, 1975]. First-order channels in forested and agricultural areas are often incised into in situ and colluvial material and show little evidence of alluvial deposition. Sediment yield from first-order basins can be estimated from both event-based flow and sediment monitoring and by measuring sediment accumulation in small ponds that are commonly constructed for agricultural uses, sediment control or storm water management.

3.2.2. Lowlands. The Piedmont above the fall zone contains large, low-gradient alluvial valleys. The common concavity of longitudinal river profiles and down-valley increase in cumulative valley flat area create conditions conducive to floodplain development [*Hack*, 1957; *Bloom*, 1998]. The net exchange of sediment between stream channels and floodplains depends, in part, on the space accommodation within alluvial valleys, as well as local base level controls provided by structures such as dams and culverts [*Schenk and Hupp*, 2009]. Flood magnitude and the elapsed time between major runoff events also influence sediment deposition and storage within valley networks [*Wolman and Gerson*, 1978].

3.2.3. Network geomorphology. Identification of the channel heads defining the upper limit of the stream network is necessary for reliable delineation of upland and lowland landforms and estimation of sediment storage associated with valley deposition. The 72 km^2 fourth-order Cattail Creek subwatershed in the UPRW was selected as a focus for the evaluation. Channel heads were defined as the upstream limit of a persistent eroded channel. Their locations were identified using field and air photo reconnaissance. The mean drainage source area to channel initiation derived from the channel head data set was 0.15 km^2.

Figure 3 shows portions of the derived Cattail Creek drainage network on a topographic map with 1.52 m (5 ft) contours. The channel heads are shown, as well as the channel network created using the mean channel initiation source area. Also shown is the extended tributary network derived using an initiation source area of 0.04 km^2 that corresponded to the minimum source area measured in the channel head data set. The tributary network delineated by the dashed lines was much larger than the total length of the channel network, indicating the extent of shallow confined flow pathways in the landscape. The resulting map identifies the external tributary links within nonchanneled, zero-order upland valleys of the Piedmont, most of which are poorly documented in the spatial data layers commonly used by government agencies.

Results from the tributary network delineation provided a basis for estimating the relative extent of upland landform units. The minimum measured source area to channel initiation (0.04 km^2) indicated that 35% of the Cattail Creek watershed was occupied by zero-order basins draining through nonchanneled upland valleys. The mean first-order basin area derived for the watershed was 0.3 km^2, occupying 62% of the total drainage area (Figure 4). The lowlands that

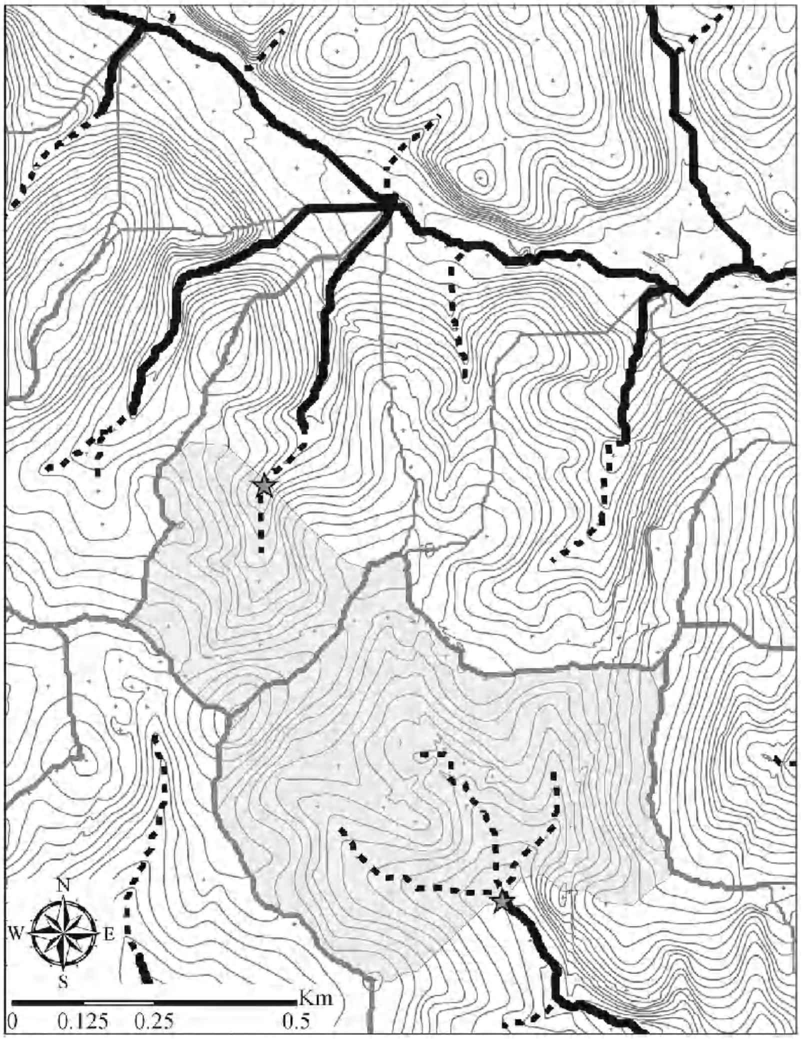

Figure 3. Channel head identification (stars), source area delineations (shaded), channels delineated using a 0.15 km^2 source area (solid lines), and zero-order tributaries delineated using 0.04 km^2 source areas (dashed lines). First-order basins are delineated by thick gray lines. Contours representing 1.52 m (5 ft) elevation intervals are shown by thin gray lines.

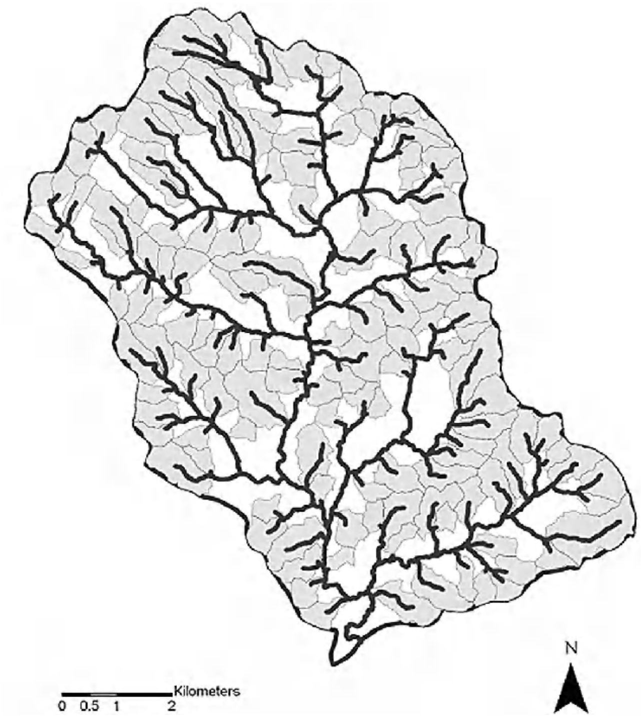

Figure 4. First-order channels and their respective basins (shaded) in the Cattail Creek watershed of the UPRW delineated using a source area to channel initiation of 0.15 km^2.

receive input from the upland landforms and serve as locations for substantial sediment storage were estimated from available floodplain delineations (Howard County, Maryland floodplains—Vector digital data, Federal Emergency Management Agency, Washington, D. C., 1986, http://msc.fema.gov/webapp/wcs/stores/servlet/FemaWelcomeView?storeId=10001&catalogId=10001&langId=-1). The mapped floodplains comprised approximately 3% of the Cattail Creek basin, which provided an indication of the relative extent of alluvial valley bottomland area in the Piedmont.

The morphometry of headwater drainage networks in the Piedmont exhibits a clear imprint from long-term weathering, landscape development, and mechanical erosion [*Costa and Cleaves*, 1984]. Zero-order tributaries can be difficult to delineate and morphologically altered by accumulations of agricultural sediment in topographic convergence zones [*Costa*, 1975]. The source area to channel initiation within the deposits varies with the land use history and direct alterations to upland drainage patterns. Colluvial deposits in upland valleys can create the appearance of alluvial floodplains in some upland basins. Channels can incise within the deposits, but lateral flows outward from the channel are usually minimal or do not occur because of the combined effects from enlarged channel capacity created from erosion and small contributing drainage areas. In some locations, remobilization of stored legacy sediment may be occurring via upstream propagation of channel incision through head cutting mechanisms at the upper termini of the first-order channel links.

The transition from dominantly erosional to storage-exchange valley bottoms has not been clearly identified in most settings, including the Mid-Atlantic Piedmont. Alluvial floodplains typically become recognizable features along second- or third-order channels [*Allmendinger et al.*, 2007]. Variations in the actual limits are strongly influenced by the history of upland sediment supply, watershed hydrology, valley profile, bedrock control, and artificial structures, including dams [*Jain et al.*, 2008]. The reality that consistent metrics for floodplain delineation are unavailable requires that surrogates be employed for identifying the boundary between upland and lowland landforms. The computation of sediment yield over progressively larger spatial scales can serve as one such approach to determine where substantial alluvial storage and therefore floodplain development occurs in the contemporary landscape.

3.3. Sediment Yield, Land Use, and Spatial Scales

Sediment yield from different land uses and spatial scales in the Piedmont are shown in Figure 5, providing a basis for comparisons among the conditions characterizing the contemporary landscape. The higher stream orders on the *x* axis correspond to larger watershed sizes [*Dunne and Leopold*, 1978]. Estimates for first-order basins were derived from sedimentation measurements in farm and storm water ponds [*Verzstraeten and Poesen*, 2001]. Each sampled basin was dominated by one of the three land cover types under consideration. Yield from third- and fifth-order watersheds was obtained from surveys of larger artificial lakes and water supply reservoirs, all of which received drainage from a mix of land uses [*Gottschalk*, 1948].

3.3.1. Land use comparisons. Several trends were apparent from the comparison of the geomorphic settings, land cover types, and spatial scales. Sediment yields from zero-order basins were often much smaller than typical land cover specific EOF values from NRCS (data, 2007), indicating colluvial storage was occurring in zero-order basins. The yield from zero-order basins was smaller than from first-order basins under similar forest and agricultural land cover conditions. This suggested that enlargement and extension of first-order channels played an important role in increasing upland sediment yield. These observations were supported by morphological evidence and precipitation event sampling in basins dominated by one of the three land cover types.

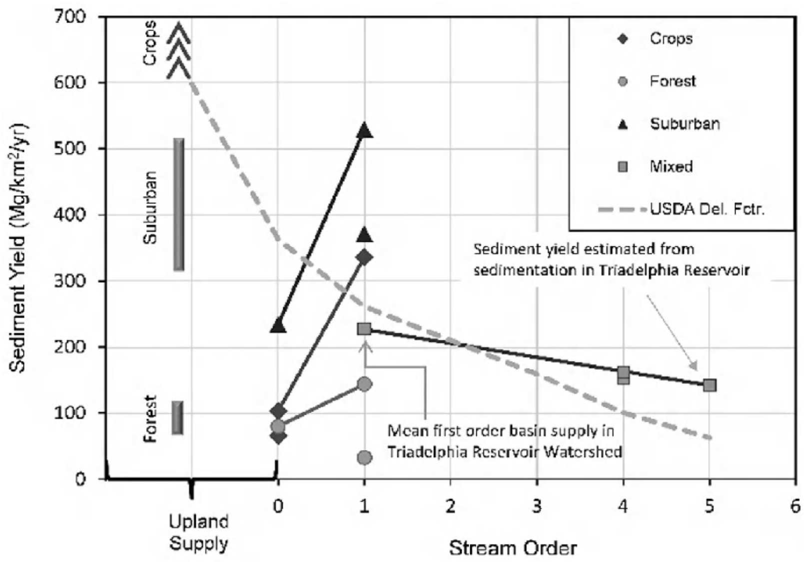

Figure 5. Sediment yield from flume, pond, and reservoir data plotted as a function of land cover and stream order. Lines connect observations within the same subbasin. Typical NRCS (data, 2007) upland erosion values are shown at left. Note that crop values are off the top of the chart. Sediment yield at the scale of zero-order basins was larger than upland supply when a substantial extent of the range of the NRCS values were considered, indicating upland storage. Yield from first-order basins tended to be larger than the NRCS values, indicating net supply from first-order channel enlargement. Sediment yield decreased from first order to fifth order, indicating storage along the valley bottom of second- and higher-order channels. The USDA delivery factor, anchored at land use-weighted annual EOF value of 598 Mg km^{-2} [*ICPRB*, 2006], produced an inverse relation between sediment yield and drainage area at every spatial scale.

The upland basin comparisons clearly showed that sediment yield was influenced by land cover conditions. However, relative rankings of land use yield changed over the spatial scale range of the three Piedmont upland landform subunits considered. Hillslope sediment yield predicted by the NRCS (data, 2007) NRI database is smallest for forest and largest for cropped land, with suburban land use having intermediate values. The relative order of the land use trend was different at the scale of first-order basins, where yield values were smallest for forested conditions and largest for suburban land use, with agricultural watersheds having intermediate values.

The sediment yield from a first-order forested basin evaluated during the study was considerably larger than previously documented for small forested basins [*Cleaves et al.*, 1970; *Yorke and Herb*, 1978; *Patric et al.*, 1984]. This was attributed to several factors, the most apparent being active upland channel extension and enlargement. Another relevant process was observed further upslope within a nonchanneled upland valley. Field observations and storm runoff sampling within a measured forested basin revealed that commonly occurring overland flows were competent in their ability to move and imbricate gravel clasts within zero-order tributaries. Like much of Maryland's Piedmont, the basin had been cleared of trees and farmed over the past two centuries. The mobility of small gravel was unexpected, but conformed to suggestions by others that the erosion of permeable upper soil horizons and removal of organic matter has increased runoff and amplified erosion in the Piedmont uplands [*Costa*, 1975; *Pavich*, 1989].

The relatively large sediment yield measured in suburban first-order basins of the UPRW was intriguing because the development was completed decades ago, and it has been a commonly held view that urban areas become sediment starved following the period of initial construction [*Smith et al.*, 2008]. Sediment yield derived from storm event sampling in a similarly mature suburban basin in the UPRW was also high relative to values reported in literature even though there were minimal opportunities for channel erosion. The cycle of sedimentation in urbanizing watersheds described by *Wolman* [1967] included reductions in sediment yield following urban development. However, it does not appear reasonable to assume that the sediment yield from mature urbanized areas can be exclusively attributed to channel enlargement based on the observations. Localized disturbances capable of generating elevated supplies of sediment offer an explanation for departures from the prediction. *Wolman and Schick* [1967] showed that construction sites can produce a very large yield of sediment compared to undisturbed landscapes in temperate humid environments. Modern sediment control technology has partly addressed this problem. However, sediment trapping efficiency has often been reported in the range of 50% to 75%, allowing relatively large loads to occur during periods of construction [*Schueler and Lugbill*, 1990]. The areas under active construction at any one time period may be relatively small in an aging suburb. The combination of large unit sediment yield and small size of the areas under construction make the cumulative contributions especially dependent on the number of locations being disturbed and the effectiveness of mitigation measures.

3.3.2. Spatial scale comparisons. A comparison of upland sediment supply to the yield measured at Triadelphia Reservoir was made by weighting observed first- and zero-order basin yield values by the UPRW land cover composition (Table 1). The area of first-order basins was estimated using results from the Cattail Creek drainage network delineation. Zero-order basins draining directly to second-order or higher tributaries were estimated by subtraction of the first-order basin and mapped floodplain areas from the total area of the UPRW. The approach thereby accounted for the contributions from nonchanneled and channeled uplands. Sediment contributions from construction were calculated based on average annual estimates of development activity over the lifespan of the reservoir. Results from the analysis were consistent with the commonly described trend of decreasing yield with increasing drainage area. The yield comparison indicated that second- to fifth-order valleys have stored more than one third of the upland sediment supply over the recent half century time period considered by the evaluation.

Although the comparison of first- and fifth-order basins in Figure 5 and Table 1 predicted a net reduction in sediment yield with increasing drainage area, examination at a finer resolution produced a more complex pattern that has important implications for targeting of locations to address watershed sediment problems. Most notably, event sampling within upland landform units in the UPRW indicated that sediment yield can increase with drainage area through upland portions of the watershed, reaching a maximum at the outlet of first-order basins. A "local" sediment yield ratio (SYR_n) defined as

$$SYR_n = \frac{SY_n}{SY_{n-1}}, \qquad (1)$$

where SY is sediment yield and n is stream order can highlight where net additions from erosion or subtractions from sediment storage occur. A value exceeding unity is produced where sediment yield at the lower boundary of a landform subunit, expressed in the numerator, is higher than at the upper boundary expressed in the denominator. Tributary erosion is the common cause of such a result. A SYR_n value less than unity is produced where sediment yield is larger at the upper boundary and internal sediment storage has occurred within the landform subunit under consideration.

Multiple factors influence spatial and temporal SYR trends within a watershed. Previous investigations showing varied rates of regolith development in the Piedmont suggest that background upland SYR values are strongly influenced by lithology [*Cleaves et al.*, 1974; *Costa and Cleaves*, 1984; *Pavich*, 1989]. Limiting the range of lithology conditions compared in this study was an important consideration for that reason. The trends in Figure 5 indicated that land cover can also influence the ratio and that wide variations in the relation are likely within the generalized upland land use categories.

Land cover and management conditions can vary considerably within both suburban and rural headwater areas. EOF sediment yield predicted by the NRCS (data, 2007) NRI database can be relatively large in rural areas, particularly for agricultural land uses. Locations characterized by high rates of hillslope sediment supply and inadequate transport capacity within downslope upland valleys produce sediment storage and a yield ratio less than the unity condition, $SYR_0 < 1$. Ratios can also be below unity where sediment best management practices have been successfully deployed. For example, the sediment yield from agricultural fields can be substantially reduced where grassed buffer strips are in place and where zero-order tributaries are maintained as "grassed waterways." Conversely, augmentation of upland sediment supply can occur where shallow concentrated surface flows frequently form on exposed soil during periods of rainfall runoff in upland valleys, producing $SYR_0 > 1$ as a result of upland valley erosion. Such conditions occur in suburban

Table 1. Sediment Yield Estimates for the Upper Patuxent River Watershed

Land Use	Drainage Area (%)	Sediment Yield (Mg km^{-2} yr^{-1})
Agricultural	52	336
Forest	33	125
Urban	15	450
Construction[a]	0.2	2102
Weighted upland area average[b]		227
Fifth-order reservoir		142

[a]Construction yield assumed 75% efficiency for sediment control measures.

[b]Weighted by land use and proportion of the UPRW composed of zero- and first-order basins draining to valleys with second- or higher-order tributaries.

areas experiencing storm water infrastructure problems, poorly managed agricultural drainage conveyances, and in forested zero-order tributaries affected by the legacy effects of intense farming activities.

The link between upland sediment production and yield is clearly not a simple one, but worthy of close attention to identify the sources influencing the sediment loads from large watersheds. Once in the channel network, SYR values change considerably with increasing scale from first- to higher-order channels. Figure 5 provides evidence that channel erosion augments upland hillslope sediment supply, as shown by the yield increases over the scale change from zero- to first-order basins in the plot. There was a consistent pattern of SYR_1 >1 in the UPRW, which indicated that additional sediment was being produced in upland channels regardless of the current land use. The downstream SYR trend was reversed only within larger valleys with sufficient space to store the upland sediment in alluvial floodplain deposits.

At watershed scales larger than the area of first-order basins, land cover is most often mixed, and the sediment yield reported is usually a weighted average for all of the contributing land cover conditions. SYR_{3+} values in settings sampled in Maryland's Piedmont were generally less than unity, indicating net sediment storage over the decadal time scale evaluated. It is important to consider that localized reaches of alluvial valleys have the capacity to augment the sediment supply, particularly where historic accumulations of sediment are in the process of being reworked under the influence of altered hydrologic regimes [*Jacobsen and Coleman*, 1986; *Schenk and Hupp*, 2009; *Walter and Merritts*, 2008; *Smith et al.*, 2008]. Even with the known existence of those processes, the net contribution calculated for the UPRW alluvial valley network was one of sediment storage over decadal time scales. The timing and rate of valley sediment evacuation is governed by the occurrence of relatively large runoff events, complicating predictions over time scales of less than a decade [*Wolman and Gerson*, 1978].

It is readily apparent from the comparison in Figure 5 that changes in sediment yield with spatial scale can differ from the simple inverse trend given by the USDA delivery factor. Most notably, the delivery factor predicts that sediment storage exceeds supply within the upland portions of the landscape. Although sediment storage can dominate between EOF and the outlet of zero-order basins ($SYR_0 < 1$), conditions causing hillslope sediment supply to be augmented by upland valley erosion can occur in all contemporary Piedmont land cover conditions, including forests. The consistently observed $SYR_1 > 1$ trend suggested that channel erosion was a substantial contributor to the total upland sediment yield to alluvial valleys. A likely culprit associated with upland tributary erosion in the contemporary landscape was increased runoff resulting from past and present land alterations.

The intention of the landform SYR calculations from the UPRW data was to account for net sediment supply and storage in defined upland and lowland settings. Framing the application of SYR values relative to geomorphic setting, lithology, and land use provided a useful basis for interpreting sediment yield calculations. Adjustments to upland valley geomorphic conditions are partly dependent on the water and sediment supply from upstream hillslopes. SYR values provided an index of the ability of a tributary reach to pass the supplied load. Despite the utility of the ratio, caution is necessary when applying SYR values to a range of EOF yield conditions that are estimated rather than predicted.

The SYR trends in the UPRW imply that the net effect of sediment management investments such as stream stabilization on the watershed sediment yield depends on the condition and location of the settings selected for the interventions. Sediment processes at different spatial scales are unlikely to be properly represented by the USDA delivery factor, particularly in upland areas where zero- and first-order tributaries influence the net sediment supply to alluvial valleys. Drainage network simulations that include only third and higher tributaries present substantial limitations because of the potential for substantial sediment contributions from headwater tributary erosion. The complex relations between EOF values and the upstream limit of the modeled watershed cannot be reliably estimated using a simple delivery factor. Landform-specific observations and multiple lines of evidence are needed to locate and estimate sediment sources at the scale of low-order basins. This must involve consideration of the cumulative hydrologic and hydraulic effects from lithology, land use, and watershed history.

4. SEDIMENT YIELD IN THE MINNESOTA RIVER BASIN

4.1. Site Description

The Le Sueur River drains a 2880 km^2 watershed in south central Minnesota, joining the Blue Earth River just before draining into the Minnesota River (Figure 6). Although relief in most of the watershed is very small, the surficial geology and river longitudinal profiles clearly indicate that this has been an active and dynamic landscape over the past few millennia.

This part of south central Minnesota was deglaciated approximately 14,000 radiocarbon years before present (rcybp) when the Des Moines lobe of the Laurentide Ice Sheet retreated, leaving behind a relatively flat terrain

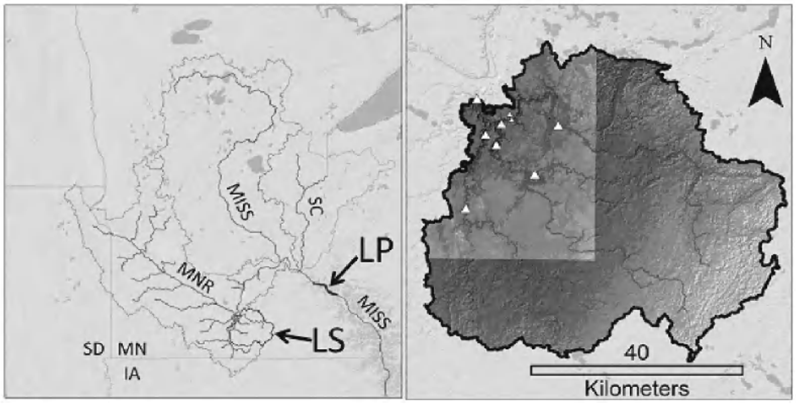

Figure 6. Location of the Le Sueur watershed (LS), south central Minnesota (MN), and Lake Pepin (LP) on the Mississippi River (MISS) in southeastern Minnesota. Also shown are the Minnesota (MNR) and St. Croix (SC) rivers. Triangles in the right panel indicate the locations of gauging stations on the main stems (large) and ravines (small). The lighter grayscale portion in the northwest corner of the Le Sueur watershed DEM shows the extent of lidar data. Mouth of Le Sueur watershed is located at 44°07′36″N, 94°02′52″W.

underlain by a 50–60 m thick package of interbedded fine-grained (65% silt and clay) till and glaciofluvial sand strata. The southern and western half of the watershed comprised Glacial Lake Minnesota for several millennia, leaving behind a thin mantle of lacustrine deposits in that part of the basin [*Thorleifson*, 1996]. Throughout the watershed, remnants of the active late Pleistocene history can be found, including large subglacial and proglacial channels, large meltwater lakes, small kettle lakes, and stagnant ice moraines [*Jennings*, 2010].

Approximately 13,400 years before present (11,500 rcybp), Glacial Lake Agassiz drained through the Minnesota River, causing as much as 70 m of incision near the confluence with the Blue Earth River [*Clayton and Moran*, 1982; *Matsch*, 1983; *Gran et al.*, 2009]. In response to the base level fall, the Blue Earth and Le Sueur systems began incising rapidly causing a knickpoint that has propagated 40 km up through the Le Sueur network [*Belmont*, 2011]. Throughout much of the Holocene, the watershed contained many internally drained wetlands and lakes and a fragmented stream network, which presumably developed better connectivity over time, particularly with the passage of the knickpoint in the lower reaches.

European-style agriculture began circa 1830, initially draining wetlands and clearing forest and prairie to plant a diversity of crops. In the past few decades, nearly all arable land is in row crop production (primarily corn and soybean), with narrow grass and forest buffers lining streams. For agricultural purposes, the fine-grained soils require tillage, and the thermal regime (with freezing temperatures occurring as late as May) deters use of cover crops that would otherwise reduce erosion in the spring.

In addition to clearing vegetation and tilling the soil, agriculture has profoundly changed the watershed hydrology in several significant ways. The vegetation change and bare spring soils have reduced evapotranspiration. Ditches throughout the watershed have greatly increased hydrologic connectivity and effectively increased the drainage area. In addition, subsurface tile drainage has been introduced, initially as ceramic pipes and more recently as plastic corrugated tubing buried various depths below the plow line, to increase runoff efficiency. The extent and density of drain tiles is not well documented, but artificial drainage appears to be nearly ubiquitous, with spacing between tiles as close as 15–20 m. The hydrologic effects of these drain tiles are generally understood, but quantitative models have struggled to accurately predict drain tile effects under the wide range of environmental conditions that exist [*Blann et al.*, 2009]. In terms of sediment dynamics, it is expected that drain tiles have both positive and negative impacts.

Detrimental impacts of excessive sedimentation throughout the Minnesota River Basin, and specifically in the Le Sueur River, are well documented [*Minnesota Pollution Control Agency*, 2008]. The problem is pervasive, with many reaches of the Minnesota and Le Sueur Rivers listed as impaired under the Clean Water Act (1972). Similar scenarios have been described in agricultural landscapes throughout the Midwestern United States and elsewhere [*Hooke*, 2000; *Montgomery*, 2007], but the south central Minnesota landscape appears to be particularly sensitive. Sedimentary records from Lake Pepin, a naturally dammed lake on the Mississippi River downstream from the confluence with the Minnesota River, indicate that the Minnesota River has been the dominant sediment source throughout the Holocene and that sediment delivery from the Minnesota River basin has increased tenfold since the mid-1800s [*Kelley and Nater*, 2000; *Engstrom et al.*, 2009].

A broad effort is underway to improve water quality in Minnesota. In 2008, state taxpayers approved an amendment to the state constitution to increase sales tax for the exclusive purpose of protecting and restoring water, wildlife, and cultural resources. The amendment is expected to generate over $150 million in tax revenue per year, providing an extraordinary opportunity and a compelling obligation to effectively implement watershed rehabilitation and restoration. Reducing sediment loading to the Minnesota River and Lake Pepin are primary objectives for restoring clean water and improving the ecosystem. The Le Sueur accounts for a significant part of the problem, contributing as much as one third of the Minnesota River suspended sediment load, while comprising only 7% of the watershed area [*Wilcock*, 2009].

Developing an effective sediment reduction strategy for the Le Sueur drainage basin requires explicit consideration for the location, mechanisms, and rates of sediment sources and sinks throughout the watershed. Implementing such a strategy requires additional economic and social considerations that will not be considered here.

4.2. Landscape Delineation and Constraints on Rates and Mechanisms

As is the case for the Maryland Piedmont, an estimate of sediment supply and yield must begin with the delineation of landscape units and their rates of sediment production and storage. The morphological conditions and processes in the Le Sueur watershed require a different mix of techniques to delineate landscape elements and constrain rates. Consistent with the above, we delineate sediment sources and sinks and identify a critical transition between alluvial and erosional portions of the landscape. However, the alluvial portion of the channel network upstream from the knickpoint is also upstream of the primarily erosional portion of the channel network associated with the knick zone in the case of the Le Sueur.

Three primary sediment sources exist in the Le Sueur watershed: uplands, bluffs, and ravines (Figure 7). Floodplains and stream banks are inherently exchange landforms, serving as both sediment sources and sinks. They represent an important challenge for developing sediment budgets and are considered separately in the next section.

High-resolution topographic data and spatial data analysis software currently allow the location and morphology of sediment sources to be defined with a precision not previously available. Constraining erosion rates from these landform units and determining the fate of the eroded sediment remains a considerable challenge. Generating accurate estimates of erosion requires a combination of targeted measurements and reasonable assumptions. This section discusses the techniques used and challenges encountered in constraining the locations, mechanisms, and rates associated with sediment sources throughout the watershed.

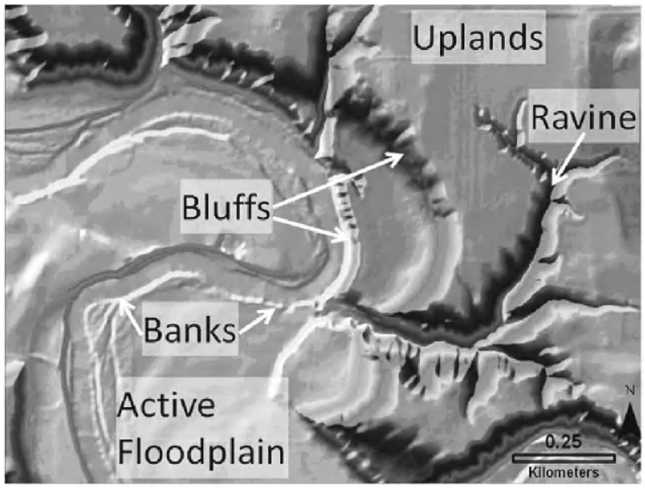

Figure 7. Sediment sources and sinks in the Le Sueur watershed including flat agricultural uplands, ravines, bluffs (shown is a 27 m tall bluff connected to river and 40 m tall paleobluff separated from the river by a fluvial terrace), banks, and active floodplain.

More than 90% of the vast, flat uplands in the Le Sueur watershed are used for row crop production. Processes of erosion in the uplands include sheet and rill erosion, gully development, and enlargement of drainage ditches. But the rates at which these processes actually convey sediment to the channel are difficult to constrain due to extraordinary spatial heterogeneity and temporal variability. Upland erosion estimates computed from the USLE [*Wischmeier and Smith*, 1978], or its derivatives, modified USLE [*Williams*, 1975] and revised USLE [*Renard et al.*, 1997], must be viewed skeptically in this landscape for two reasons. First, surface erosion is highly sensitive to the threshold at which surface runoff occurs, which cannot currently be predicted with accuracy in this artificially drained landscape. Second, a relatively wide range of sediment delivery ratios likely exists, driven by relatively subtle topographic features. This causes large uncertainty in sediment delivery, which is especially problematic because the source area is so large.

Bluffs are tall, near-vertical features that exist almost exclusively within the knick zone of the Le Sueur (see Figure 8). They are primarily composed of glacial sediments and can be very large, (>50 m high and hundreds of meters long) or relatively small (3 m high and <10 m long). Some bluffs are directly connected to the river. Others were previously

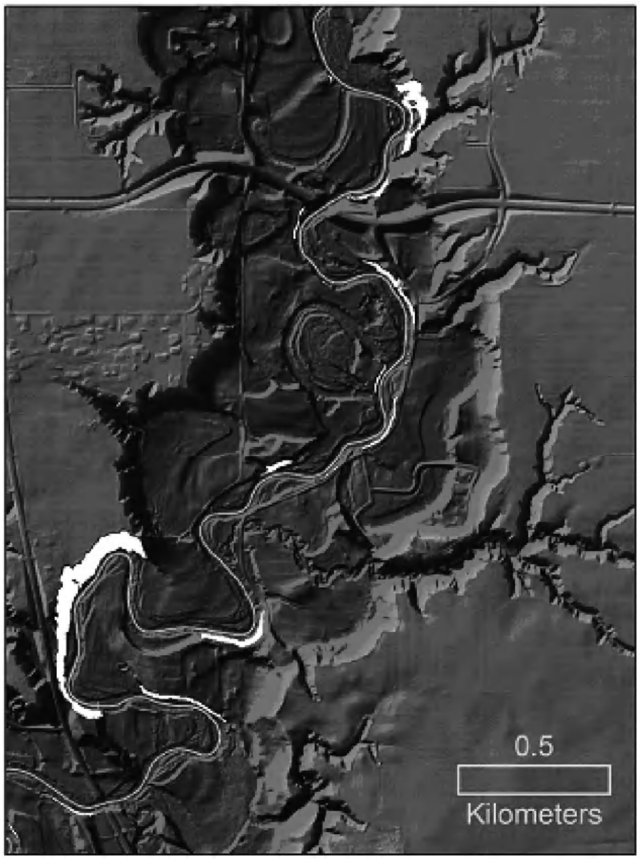

Figure 8. Bluffs (in white) along the main stem of the Le Sueur River automatically delineated using neighborhood analysis (focal range) as described in text.

connected to the river, but the river has since migrated away and incised, leaving them stranded behind as strath terraces. Identification of bluffs is relatively straightforward using a simple algorithm that extracts cells based on a local relief threshold (e.g., 3 m of relief within a 9 m by 9 m neighborhood, Figure 8), but measuring meaningful erosion rates is challenging.

Bluff erosion is driven by fluvial undercutting at the toe, which triggers slope failure. The rate of erosion is influenced by physical properties of the layered glacial material of which they are composed, including cohesive strength, hydraulic conductivity, vegetation, and moisture content. In theory, bluff erosion rates could be modeled from hydrology, geotechnical properties [*Simon et al.*, 2000], and vegetation effects [*Simon and Collison*, 2002; *Bankhead and Simon*, 2010], but upscaling from a few bluffs on which measurements can reasonably be made to all (300+) bluffs throughout the watershed is confounded by the spatial heterogeneity of glacial deposits.

Bluff erosion rates can be directly measured from historic air photos, comparing bluff crests over multiple decades. Such erosion rates are defined for the time scale over which they are measured and may or may not be applicable to shorter or longer time scales. In addition, careful consideration of bluff retreat processes and geometry are needed to determine sediment supply. Over century times scales, the crest and toe of the bluffs can be assumed to retreat in parallel, as long as they remain connected to the primary driver of erosion, the river. However, over shorter time scales, the rate of sediment supply can be smaller if the bluff crest retreats more rapidly than its toe, with a minimum obtained if the toe erosion rate is zero. A bluff erosion rate for the Le Sueur was developed by combining estimates of bluff crest erosion rate with bluff toe erosion rates determined from channel migration measured separately.

Bluff erosion rates can also be measured utilizing ground-based surface elevation scanning technology. The precision of these instruments (less than 1 cm) provides an extraordinary opportunity to directly measure erosion on annual or subannual time scales, but a number of logistical complications must be overcome, and ultimately, multiyear erosion rates measured on a few bluffs must be extrapolated to all other bluffs throughout the system. Selecting sites that cover the full range of bluff types in terms of size, composition, aspect, and proximity to roads is critical for upscaling in a process-sensitive manner (see S. S. Day et al. (Change detection on bluffs using terrestrial laser mapping technology, submitted to *Earth Surface Processes and Landforms*, 2010) for detailed discussion).

Ravines are small, steep channel networks primarily found in the incised portion of the basin, connecting the broad, flat uplands to the incised Le Sueur river channel. These features can easily be identified and delineated from high-resolution topography data [*Wing*, 2009]. Constraining accurate sediment contributions from ravines is challenging because they erode by a combination of hillslope and fluvial processes. In addition, ravines can serve as sediment sinks, storing significant amounts of sediment behind landslides and woody debris jams. Such fine-scale sediment storage and release processes are not easily predicted from topography data alone, but field observations indicate that fill terraces can dominate the sediment contributions from some ravines. Release of stored sediment from ravines can be exacerbated under conditions where precipitation is increasing or flow is being concentrated in the ravines by artificial drainage of the uplands.

Erosion in ravines is ultimately driven by fluvial incision and subsequent undercutting, hillslope creep, or mass wasting. Measuring sediment yield from ravines is challenging because of the flashy nature of these systems. Samples collected from a dozen events in a few (two to four) ravines between 2008 and 2010 indicate that most sediment is transported through these systems in a matter of hours, and sediment is only mobilized during relatively large precipitation events (K. Gran, personal communication, 2010). Extrapolating sediment yields from a few ravines to the more than 100 ravines found throughout the Le Sueur watershed is problematic considering the great diversity in ravine size, shape, relief, etc. In addition, sediment export from these systems is likely to be highly nonlinear as a function of runoff, so it is essential that ravines are monitored over a wide range of environmental conditions before proper constraints can be made.

4.3. Sediment Storage in the Uplands and Fluvial Network

An enduring problem in geomorphology is the understanding and prediction of the mechanisms, rates, and timing of sediment storage in the landscape [*Trimble*, 1977; *Wolman*, 1977; *Walling*, 1983]. The Le Sueur watershed provides a relatively unique opportunity to study sediment storage. Above the knick zone, sediment storage is widely distributed and spatially complex, representative of flat, agricultural landscapes that dominate the Midwestern United States. Within the knick zone, sediment transport and storage processes are dominated by adjustments within a steep, rapidly incising valley.

There is abundant evidence that a significant portion of sediment eroded from fields is deposited before reaching the river network. In the Le Sueur, field evidence of eroded sediment that remains stored within the landscape includes deposits of windblown sediment less than a few centimeters thick on snow patches every spring. In addition, some agricultural fields that are apparently subjected to strong winds have been observed to produce “mud dunes” as high as a meter at the edge of fields where vegetation provides the necessary roughness to trap windblown sediment.

Evidence for sediment storage within the landscape has been observed in many watersheds covering a wide range of tectonic and climatic environments [*Costa*, 1975; *Meade*, 1982; *Phillips*, 1991; *Trimble*, 1999; *Bierman et al.*, 2005]. However, actually quantifying the location, mechanism, volume, and duration of storage within the landscape is difficult within heavily modified agricultural settings. *De Alba* [2001] developed and applied a numerical model to quantify the amount of soil redistribution that can be attributed to tillage. Such models make predictions that can be field tested, but they are difficult to apply at the watershed scale. The relevance of such models depends on whether or not human dynamics can be adequately captured. This modeling problem is common to heavily engineered landscapes, as discussed above in the context of construction sites in the UPRW.

Sediment storage in valley bottoms is more spatially focused than upland storage. Nevertheless, the Le Sueur channel network exemplifies some of the challenges for making meaningful estimates of sediment storage in the channel and floodplain. One complication arises from the nonuniform structure of the stream network. The “natural” channel network of the Le Sueur, not including human-engineered ditches, includes four *Strahler* [1957] stream orders. However, the notion of stream order loses some meaning in the relatively flat, human-modified landscape. For example, first-order streams exhibit a wide range of contributing drainage areas (<1 to 217 km^2), due in part to the once-internally drained areas that have been connected to the channel network either naturally or by humans using surface ditches or subsurface drain tiles. The total length of agricultural drainage ditches is over 450 km, comprising nearly a quarter of the total surface drainage network.

A more meaningful way to categorize the network is in terms of sediment storage and transport dynamics, as discussed for the UPRW. According to this categorization, the Le Sueur drainage network can be separated into four distinct types, low-gradient agricultural ditches, low-gradient natural channels above the knickpoint (average slope is 0.0004), high-gradient main stem channels within the knick zone (slope is 0.002), and high-gradient, mostly ephemeral ravines that connect the uplands to the incised river, primarily within the knick zone. Each of these distinct channel types plays a potentially important role in establishing sediment sources and sinks and exhibits different challenges in determining rates of sediment storage over annual to decadal time scales.

The ditches are generally straight channels with 45° grassed side slopes. Despite the apparent uniformity in planform, these human-designed features exhibit remarkable diversity in sediment transport rates. Many serve as sediment sinks for silt and clay, while others actively transport fine gravel. Ditches are “cleaned” as needed, typically once every 10 to 50 years, but the criteria used to determine when ditches need cleaning are rather arbitrary. Sediment excavated from the ditch is typically placed back on the levee of the ditch, or back on the adjacent agricultural field, and the amount of sediment removed is not documented (C. Austinson, Blue Earth County Ditch Manager, personal communication, 2009). For these reasons, ditches are challenging systems to incorporate into a sediment budget or routing model.

Most of the agricultural ditches drain to low-gradient natural channels, which define the Le Sueur network above the knick zone. These channels migrate laterally, but at a relatively slow pace (<10 cm yr^{-1} on average), and maintain a floodplain by lateral and vertical accretion. Floodplains represent a large potential source of sediment directly accessible to the channel, but accounting for net exchange of sediment between the channel and floodplain requires consideration of both erosion and deposition. Erosional processes include bank retreat, channel widening, and vertical incision. Depositional processes include point bar deposition and overbank deposition.

The effort required to measure the actual fluxes of sediment in and out of the floodplain can be considerable, but is necessary in systems for which substantial valley bottom storage is indicated [*Walling*, 1999]. Basic geomorphic observations may be sufficient to indicate whether a floodplain system is aggrading or degrading in some cases. Generally speaking, in a net aggradational environment, the floodplain should be accessed frequently by the river, and deposition from large events should be measurable. Topographic lows in the floodplain, such as cutoff channels, should not persist for long periods of time. In contrast, key morphological indicators of net floodplain degradation include entrenchment of the channel or systematic differences in floodplain elevation on either side of the channel, such that cut banks are significantly taller than depositing banks [*Lauer and Parker*, 2008a, 2008b]. In addition, changes in channel width can be used to estimate net storage or evacuation of sediment from the floodplains [*Dean and Schmidt*, 2011].

The floodplains associated with the low-gradient, natural channels of the Le Sueur river network appear to be near a state of mass flux equilibrium with no signs of systematic floodplain aggradation or degradation in the recent past. To quantitatively test this observation, we used the Planform Statistics Tool (available from the National Center for Earth-surface Dynamics Stream Restoration Toolbox, http://www.nced.umn.edu/content/tools-and-data) to estimate net erosion that has resulted from channel migration between 1938 and 2005. This tool computes migration distance at user-specified intervals along the river (every 10 m in this study) between two points in time. The tool also extracts bank elevations at each node from high-resolution ground elevation data and combines the migration rate with the difference in bank elevation to compute local, net sediment contributions from stream banks [*Lauer and Parker*, 2008a].

The approach described above primarily accounts for floodplain deposition by lateral accretion, but vertical accretion from overbank deposition must also be considered. In the simplest form, overbank deposition can be modeled as the product of floodplain discharge and suspended sediment concentration. A trapping efficiency can be empirically calibrated and is expected to change as a function of vegetation and suspended sediment grain size. Concentration can vary by several orders of magnitude over the course of individual storm hydrographs and varies significantly from event to event, which becomes a significant problem when direct observations are few in number.

Floodplain vegetation poses two additional problems. For one, dense vegetation in floodplains is often not adequately filtered out in the process of generating a bare-earth digital elevation model (DEM), resulting in an inaccurate surface. The vegetation also influences the hydraulic conditions and sediment dynamics. For some floodplain environments, vegetation can be treated as relatively static, with a single trapping efficiency over time. In other floodplain environments, an understanding of seasonal growth patterns must be coupled with flow data. The field of ecohydraulics is currently making important gains in modeling the hydraulic implications of vegetation, but much work remains before reliable network-scale models are available [*Perona et al.*, 2009; *Corenblit et al.*, 2009].

Even when this suite of information is available, the challenge of predicting overbank deposition is formidable. In addition to knowing the concentration of sediment in transport, deposition is mediated by the grain size distribution of suspended sediment, which may change considerably over the course of a flow event. Instruments that measure sediment concentration and grain size distribution are helpful for constraining this problem, but cost and logistical complications preclude their use for constraining network-scale grain size dynamics.

Hydrologic analysis indicates that high flows in the Le Sueur are increasing in frequency and magnitude [*Novotny and Stefan*, 2007]. Field observations suggest that the increases are causing channel widening throughout much of the channel network. To account for the amount of sediment contributed from banks and floodplains via channel widening, we measured channel width from historic air photos at multiple times between 1938 and 2005. Accurate estimates of channel width were obtained by manually delineating polygons (each 500 to 1000 m long) outlining the active channel and dividing by length. The net contribution of sediment from channel widening was then computed as the product of the change in channel width, average channel depth, and the length of channel that has experienced channel widening.

In the Le Sueur channel network, the calculations above provide reasonable constraints on the amount of sediment derived from widening and meander migration. However, neither of these approaches account for sediment storage and erosion related to large woody debris jams, which occur in the low-gradient natural channels at a frequency of approximately once every 2 km. Our field observations indicate that erosion and deposition are approximately balanced in the vicinity of debris jams, although more detailed surveying would be needed to confirm this. Given the sparse number of debris jams, the apparent balance between erosion and deposition, and absence of detailed information, no source or sink of sediment from debris jams was included in the budget.

The relatively high-gradient main stem channels within the knick zone of the Le Sueur exhibit transport and storage dynamics that are substantially different from the low-gradient channels discussed above, similar to but in reverse order to the channel network of the UPRW. Meander migration rates within the knick zone are relatively high (20–30 cm yr^{-1}) according to historic air photo analyses. This is due in part to the dramatic increases in sediment loading within the knick zone (see Table 2) and significant increase in the caliber of sediment contributed as gravel and boulders are eroded from bluffs, terraces, and ravines. Relatively rapid vertical incision of the river, currently and throughout the Holocene, causes floodplains to be abandoned. Strath terraces that are preserved throughout the incised river valley are exceptionally uniform in thickness, between 2 and 3 m with a thin base of gravel, a relatively thick package of laterally accreted sand and mud capped by a variable, but typically thin mantle of fine-grained overbank deposits. These terrace deposits represent net long-term storage within the knick zone, but the volume stored is relatively small compared with the volume that has been removed over the course of the Holocene.

The morphology of the modern floodplain through the knick zone is strongly controlled by Holocene base level fall [*Belmont*, 2011]. The floodplains become progressively narrower with distance downstream through the knick zone. Confined flows with a relatively steep gradient are less inclined to deposit sediment, so decadal scale net sediment storage is minimal. One important implication of the steep knick zone in the lower reaches of the network is that sediment delivery ratios increase with downstream distance, contrary to many systems where sediment delivery ratios have been demonstrated, or assumed, to decrease downstream [*NRCS*, 1983].

Ravines play a complicated role in sediment storage and release. In general, ravines are net degradational, as discussed above. However, landslides and woody debris jams can cause backwater conditions in the otherwise steep channels. As a result, a significant amount of sediment can be temporarily stored in fill terraces, similar to the alluvial storage behind small dams discussed in the UPRW above. Sediment stored in a fill terrace can be excavated over a relatively short period of time when the physical barrier causing the backwater conditions is breached. Fill terraces of various sizes have been observed in ravines throughout the Le Sueur watershed. Because of the morphology of the ravines and poor filtering of dense ravine-bottom vegetation in the bare-earth lidar DEM, fill terraces can often be identified from the lidar DEM, but the volume of sediment trapped in fill terraces cannot readily be measured other than in the field.

4.4. Assembling the Pieces

Sediment budgets have been established for the Le Sueur watershed using HSPF [*Tetra Tech, Inc.*, 2008], WEPP [*Maalim*, 2009], and SWAT [*Folle*, 2010]. Calibration and validation of these models have produced contrasting results [*Wilcock*, 2009]. The primary data used to calibrate the models is total suspended sediment (TSS) loading measured from a gauge network in the watershed. Although the network is relatively extensive with a gauge above and below the knick zone in each of the three primary subwatersheds and a long-running gauge at the watershed mouth, a fundamental problem arises in that the available sediment measurements used for load computation do not distinguish between different sources. Table 2 shows sediment loads measured at gauging stations throughout the watershed, both above and below the knickpoint.

Loads measured at the upper gauges in each watershed are primarily derived from uplands and stream banks, but the proportion of sediment derived from each source cannot be determined and might be expected to differ in dry versus wet years. Sediment yield increases dramatically between the upper and lower gauges on each tributary. This corresponds

Table 2. Sediment Loads for All Years of Record for Each Gauge in the Le Sueur Watershed

		TSS Load (Mg yr^{-1})								
Basin	Contributing Drainage Area (km^2)	2001	2002	2003	2004	2005	2006	2007	2008	2009
Upper Maple	800	–	–	–	–	–	7,900	13,300	6,100	3,500
Lower Maple	880	–	–	18,600	101,200	85,100	22,300	37,900	22,300	4,900
Upper Cobb	335	–	–	–	7,500	8,200	4,000	4,400	3,100	1,600
Lower Cobb	735	–	–	–	–	–	33,400	21,800	14,600	6,300
Upper Le Sueur	870	–	–	–	–	–	–	42,200	22,400	4,300
Mid Le Sueur	1210	–	–	–	–	–	86,600	74,600	42,800	13,400
Mouth Le Sueur	2880	346,500	90,200	71,100	338,000	219,300	135,400	136,400	86,300	29,100

to increasing prevalence of nonupland sediment sources such as bluffs and ravines but may also be due to increased connectivity of uplands to the channel and therefore higher sediment delivery ratios. Although the gauge data provide a good indication of the magnitude of sediment flux, it cannot inform about the location and mechanism of sediment supply.

Geochemical fingerprinting provides an alternative approach to constraining upland sediment yields. In the Le Sueur, meteoric lead-210 (^{210}Pb) and beryllium-10 (^{10}Be) have been used in combination to quantify the proportion of sediment derived from uplands. Both tracers exhibit high concentrations in upland soils and low concentrations in bluffs and ravines. However, sediment temporarily stored in floodplains is diminished in ^{210}Pb and enriched in ^{10}Be concentration. Therefore, if used independently, either of the tracers would be systematically biased depending on the amount of channel-floodplain sediment exchange. When the two tracers are used in combination, this bias can be corrected. Understanding the geochemical systematics of the tracers as well as mix of geomorphic processes conveying the sediment are both essential in implementing an effective fingerprinting study.

When used together, sediment gauging and sediment fingerprinting can be used to constrain both the proportion and rate of sediment supply from different landform units. By using multiple lines of evidence, one can begin to address the problem of equifinality inherent in watershed modeling in which multiple parameter combinations can be tuned to get the “right” upland erosion rates. Without such information, a watershed erosion modeler often has little more than intuition on which to base decisions about parameter tuning. By further incorporating upscaled sediment yield estimates for different landform units, as discussed above, a reliable estimate of sediment sources and sinks can be developed.

5. DISCUSSION

Stream restoration, rehabilitation, and stabilization are increasingly proposed as an approach to resolve watershed sediment problems. To date, many projects have been opportunistic, based on the availability of land, space on a development site, or local stakeholder interest. This approach is not likely to efficiently achieve desirable water quality changes. Instead, a broader strategy is needed that can target the best opportunities for sediment load reduction. The need to place best management practices in locations promising the greatest efficiency requires a thorough understanding of the geomorphic processes associated with mechanical erosion and landform adjustment in the contemporary landscape. Careful identification and sampling of upland and lowland landforms can guide the stream management approach proposed to address water quality problems, particularly if mechanisms of sediment supply can be identified. Approaches used to reduce sediment supply from uplands include runoff control, channel stabilization to reduce tributary incision, and the use of vegetation and buffers to trap surface erosion before it is delivered to the channel network. In contrast, practices proposed to achieve sediment reductions in alluvial valleys attempt to reduce the evacuation of stored sediment and enhance the trapping of newly delivered sediment.

Although the effort involved in developing a reliable watershed sediment model can seem large, the costs will generally be small compared to those involved in implementing restoration and other actions to address watershed sediment issues. The potential savings and benefits of implementing an effective program can be substantial. The requirements for an accurate watershed sediment model are similar to those needed for informed targeting of sediment source reductions. Both require specificity regarding location, mechanisms, and rates of erosion and sediment deposition. Planning and design require development of an understanding of landscape organization, documentation of the effects of management practices on sediment production, and tracking of the locations of practice implementation and effectiveness thereof. Implementation without these tasks will make it difficult to satisfy watershed sediment yield objectives over the long term.

In the UPRW, delineation of upland and lowland landscape units, channel head locations, and the transition from erosional to depositional valley bottoms was based on analyses of air photos, elevation data, and other catalogued spatial information. Rates of erosion and storage in upland units were characterized using field observations and event-based sampling. Integrative records used to constrain uncertain upland erosion and deposition estimates were based on sediment accumulation in ponds. Land cover data were necessary for upscaling local erosion rates to the watershed scale. Comparison of sediment yield values indicated that sediment yield increased from the edge of field to the outlet of first-order watersheds and that net storage occurred within the higher-order watersheds. This pattern cannot be captured in a simple delivery factor intended to link edge-of-field soil erosion rates to sediment supply to higher-order rivers.

Delineation of landscape elements in the Le Sueur watershed, including agricultural fields, bluffs, ravines, and the channel-floodplain system, used a combination of analyses exploiting high-resolution topography and air photos as well as field surveys and mapping. Adequately constraining rates of sediment inputs from each source required an understanding of erosion mechanisms. Upscaling estimates of erosion from a few features where detailed measurements can be made (e.g., a dozen bluffs) to similar features throughout the watershed required constraints on spatial variability and delineation of essential geomorphic features, such as distinguishing the proportion of bluffs that are actively undercut. Design of sampling and monitoring programs require critical evaluation of these factors as well as lithology, relief, landform subunit, and land use. Future work needs to focus on automating the processes by which landscape elements can be identified, enhancing techniques for geomorphic change detection on spatially extensive landforms, and accounting for uncertainty in identification, change detection, and upscaling.

Sediment production, transport, and storage for individual landscape units must be upscaled in a geomorphically informed fashion. The advent of widespread coverage of high-resolution elevation data, the availability of long-term air photo records, and the power of spatial data software offer excellent resources for upscaling in a superior, topographically sensitive fashion. Landform-specific sediment flux observations provide a basis for transferring data to appropriate locations within a catchment and linking the components together in a defensible manner.

Accurate treatment of sediment storage remains a difficult problem that can be addressed by constraining a sediment budget using sedimentation records of a decadal time scale or longer in order to integrate over a range of runoff and climatic conditions. Such observations of channel enlargement and sediment yield in first-order Mid-Atlantic Piedmont watersheds indicate that sediment storage is currently minimal, and sediment production is substantial in contemporary upland valleys. This contrasts with the conclusion of *Costa* [1975] that over half the sediment eroded during peak nineteenth century Piedmont agriculture remains stored in colluvial sheet wash deposits.

Variability in higher-order tributaries set within alluvial valleys requires consideration of base level controls and the role of large storms in setting annual to decadal sediment delivery patterns. Recent measurements indicate that contemporary floodplains in the Mid-Atlantic region are actively storing sediment, but the temporal and spatial limits to storage are not well documented [*Schenk and Hupp*, 2009; *Noe and Hupp*, 2009]. Comparison of third- and fifth-order alluvial valleys in the UPRW indicated that storage opportunities increase with drainage area. Local geologic conditions that govern valley geometry can strongly influence the availability of storage opportunities. The narrow gorges in the Piedmont fall zone are an example of a geologic feature that limits the capacity for floodplain development. However, the constriction also provides a hydraulic control that can affect sediment accumulation trends upstream.

Similarly, the Le Sueur channel-floodplain network exhibits distinct zones (agricultural ditches, ravines, low-gradient natural channels, and high-gradient natural channels) that must be delineated and treated separately for the purpose of estimating watershed sediment patterns. As discussed above, the sediment transport and storage dynamics differ significantly in each of these zones, so identifying if or where a problem exists and considering various stream restoration solutions to the problem must be done in a context-sensitive manner.

Given the large inherent uncertainty in any estimate of sediment erosion, transport, and storage, a credible watershed sediment model requires the use of multiple lines of evidence to constrain the estimated values. Regardless of the methods used, a sediment supply prediction that relies on a single estimate, or calculates budget terms as a residual, cannot produce reliable results. Approaches that rely on sediment concentration measurements and sediment rating curves are not only subject to considerable error, but do not provide a basis for prediction under altered conditions, do not identify actionable sources for locations between gauges, and can involve considerable, often prohibitive logistics and expense in order to build a data set across multiple spatial scales. Approaches based on local erosion measurements provide the observations and interpretation needed to specify the mechanism and location relevant for restoration efforts, but face considerable uncertainty in upscaling episodic and nonlinear rates. Sediment fingerprinting offers important advantages for source identification, but generally provides only percentages from different sources. An effective fingerprinting campaign can be defined using a combination of deposited sediment and sediment in transport, although this raises logistical issues similar to direct load measurements [*Rowan et al.*, 2000]. All of these methods can be used in combination to improve the accuracy of sediment supply and yield estimates, although the strongest constraint, and therefore the most useful for developing a credible sediment budget, is a record of erosion or sedimentation that spans both spatial and temporal scales. It is very difficult to develop a credible sediment budget without some estimate of integrated erosion or deposition for the entire watershed over decadal or longer periods.

Observations of sediment accumulation in impoundments can be used to constrain the sediment yield estimate and related error. Impoundment measurements can be obtained through direct measurement of smaller structures and via the monitoring and maintenance that government agencies pursue for safety, water supply, and storm water quantity management purposes. The record of reservoir sedimentation is growing, and a concerted effort is underway to organize and distribute this information (http://ida.water.usgs.gov/ressed/), which can provide an invaluable constraint on future sediment yield estimates. An important opportunity can be realized

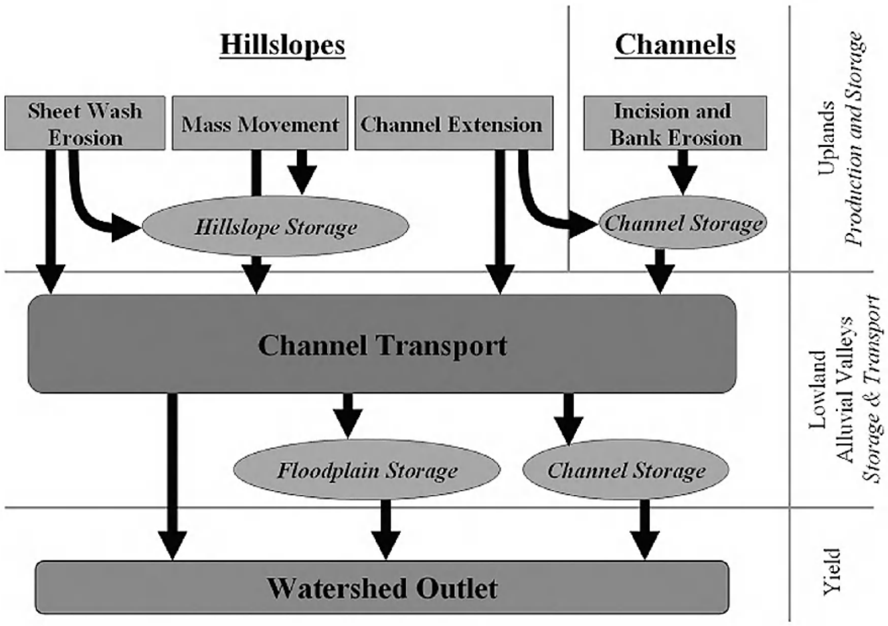

Figure 9. Watershed sediment budgeting framework.

from better coordination with watershed maintenance efforts. For example, monitoring of sediment accumulation in reservoirs, ponds, and storm water facilities may be achieved as part of water quality and storage maintenance purposes.

Sediment accumulation in UPRW was determined for watersheds from first- to fifth-order and for time periods of one to many decades. In the case of the Le Sueur River, we took advantage of a well-defined incision history where the initial surface elevation and the timing of base level drop are

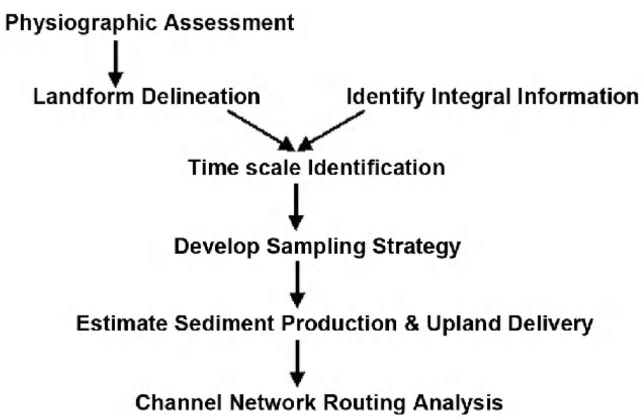

Figure 10. Watershed sediment process and yield analysis framework.

precisely known to compute long-term sediment evacuation rates. Ongoing work will constrain the unsteady rates of knick migration and valley excavation over time throughout the Holocene. These data and hydrologic reconstructions combine to constrain natural background turbidity levels.

A broad conceptual framework can be proposed to aid in organization of a watershed sediment model. Landscape delineation, estimates of sediment yield in individual landscape units, and an approach for upscaling, coupling, and routing local sediment yield are the starting elements of any approach. Figure 9 provides a simple schematic of common elements of a watershed sediment model. *Reid and Dunne* [1996] provide an excellent handbook for evaluating the different parts of the budget. The role of zero- and first-order valleys, upland channels in Figure 9, in producing and storing sediment is poorly represented in any modeling approach, and improvements in that regard are a priority.

Figure 10 outlines a conceptual sequence of activities that can be used to organize efforts to develop an estimate of sediment supply or yield and address the scaling challenges of matching watershed models to sediment budgets. Before identifying modeling time scales, it is necessary to delineate the potential dominant sources of sediment and identify the types of integral data (reservoirs, long-running gauges, sediment fingerprinting, historical channel analysis) that might be available and reliable. Once the units and time scale are identified, an appropriate sampling strategy can be developed. This may combine field observations of erosion and deposition, historic analysis of slope and channel shift, and supporting information regarding land use and hydrologic alteration. By combining this information, it should be possible to compare upland sediment production to integral measures of sediment yield in a way that identifies the location and rates of sediment erosion and storage. When scaling up to larger watersheds, an approach for routing sediment along channels, including an estimate of net storage, is needed.

6. SUMMARY

A variety of factors, relief, landscape history, climate, and land use history cause the locations, mechanisms, and rates of sediment production and storage to vary in space and time. Recent advances in monitoring technology, geochemical techniques, high-resolution topography data acquisition and analysis, geographic information system software, and numerical modeling approaches provide new opportunities to constrain geomorphic rates, upscale them in a geomorphically relevant fashion, and synthesize sediment dynamics at the watershed scale. In the two examples examined here, the upper Patuxent River in Maryland and the Le Sueur River in Minnesota, the mechanisms and many of the sediment budget components are substantially different. Remarkably, the upstream-to-downstream position of dominantly erosional and depositional landscapes is different between the two watersheds. The UPRW has a more typical erosional-to-depositional sequence, whereas because of low gradient and the Holocene base level drop, erosional reaches in the Le Sueur occur down valley of low-gradient reaches with substantial storage. Nonetheless, there are common challenges and themes in defining an effective watershed sediment model. In both cases, reliable estimates of sediment yield depend essentially on the accurate identification of sediment sources and sinks and, hence, require careful delineation of landscape units. Upscaling local contributions to watershed sediment yield requires reliable estimates of sediment transport across multiple time scales and the use of multiple lines of evidence to constrain uncertain estimates.

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Mitigating Channel Incision via Sediment Input and Self-Initiated Riverbank Erosion at the Mur River, Austria

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At the border reach of the Mur River between Austria and Slovenia, systematic river training and hydroelectric power plants have led to a channel incision with considerable ecological and technical consequences. A sediment transport model predicted further incision if no countermeasures are implemented. The thin gravel layer (≈0.5 m) poses the threat of a riverbed breakthrough, calling for urgent action. In a Basic Water Management Concept, several types of ecologically oriented countermeasures have been proposed. Recently, one measure that combines self-initiated riverbank erosion with sediment input from a newly constructed sidearm has been implemented. To determine the success of the measure, we conducted a detailed survey along with particle tracking by telemetry. The results show the anticipated response. At least for the present, short-term development, the measure effectively mitigated the incision.

1. INTRODUCTION

Narrowing of former braided rivers and a lack of sediment input from upstream because of weirs, torrent control, or gravel mining has degraded the riverbed of many rivers in the Alpine region [*Habersack and Nachtnebel*, 1998; *Habersack and Piégay*, 2008], and similar examples exist worldwide [*Darby and Simon*, 1999]. Channel incision has generated a variety of ecological problems such as the disconnection of the floodplain forest and a decrease in habitat and biodiversity [*Jungwirth et al.*, 2003]. It has also created numerous technical problems, including scouring and subsequent destabilization of bank-protection structures and bridge piers, or problems in the water supply because the adjacent groundwater table has been lowered [*Habersack and Nachtnebel*, 1998].

To stop channel incision, to ensure flood protection, and to improve the ecological integrity of the river ecosystem, many restoration projects have been undertaken in Austria over the

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past 10 years. River engineering measures such as spur dykes and longitudinal bank protection, which imposed fixed lateral boundaries on rivers, have been removed [*Habersack et al.*, 2000]. The channel incision at the border reach of the Mur River between Austria and Slovenia, first observed in 1970, calls for urgent action: The ongoing erosion of the thin gravel layer could cause a riverbed breakthrough into finer-grained, Tertiary material, potentially leading to a total loss of the gravel layer as in the Salzach River [*Hengl*, 2004]. Once a breakthrough occurs, ecologically oriented methods like riverbed widenings are no longer effective [*Habersack and Piégay*, 2008]. To date, only few pioneering restoration projects have dealt with sediment transport and related river morphology in the Alps [*Habersack and Piégay*, 2008]. At the Mur River, a long-term reestablishment of a balanced sediment budget is the main goal; restoration projects have to consider sediment transport processes because they are the key parameters for success. The direct use of natural morphological processes like bed load input from bank erosion has been discussed in a Basic Water Management Concept for the Mur River (H. Habersack et al., River engineering, final report for the Austrian-Slovenian Standing Committee for the Mur River, Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished, 2001a) and applied at a recently implemented measure.

The implemented measure is part of a series of measures proposed for the Mur River. Using the Mur River as a case study, this chapter discusses an ecologically oriented countermeasure against channel incision based on an innovative monitoring concept and evaluates the success of that measure.

2. SITE DESCRIPTION, CHANNEL INCISION, AND HISTORICAL BACKGROUND

The Mur River originates in the Austrian Alps at an altitude of 1900 m above sea level and flows into the Drava River in Croatia after having covered a distance of approximately 450 km. It drains a catchment basin of 13,824 km^2. The studied reach represents the border between Austria and Slovenia and is about 34 km long (Figure 1). At the end of this border reach, the Mur has a length of about 355 km and drains 10,340 km^2. This reach is characterized by a mean

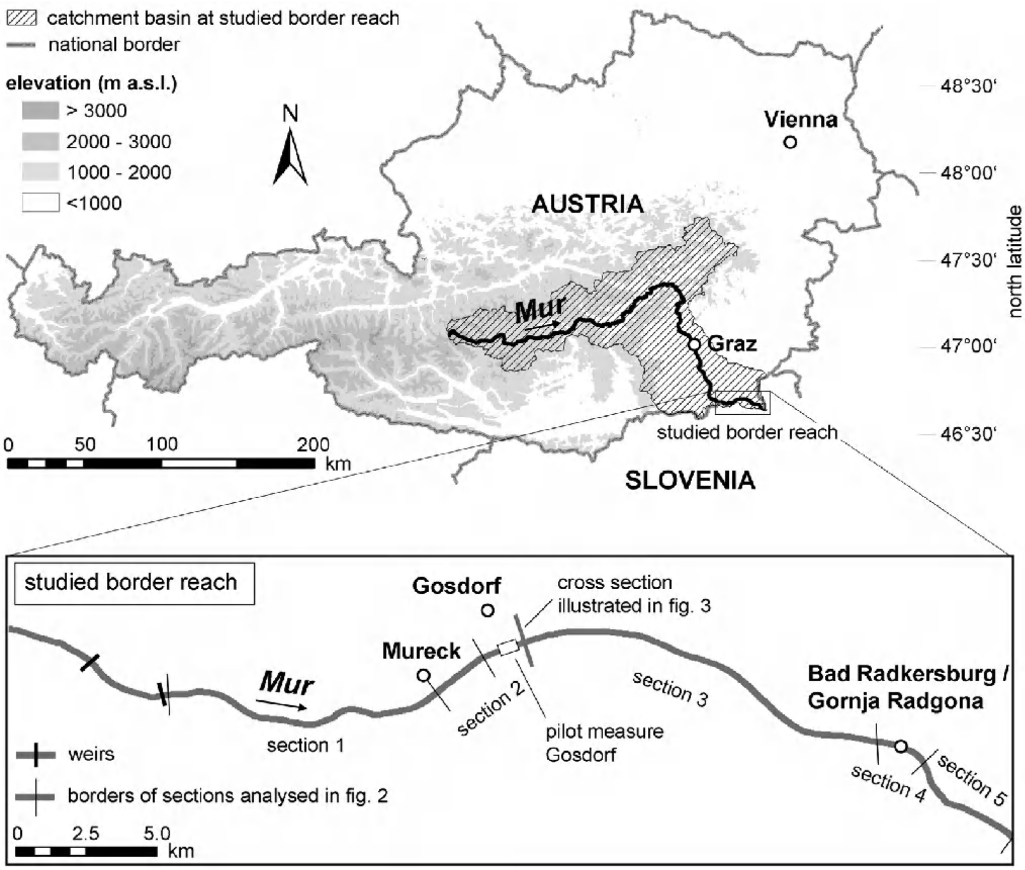

Figure 1. Mur River catchment and location of the studied border reach.

slope of 1.4 ‰ and a mean discharge of 150 m^3 s^{-1}. A 1 year flood reaches 700 m^3 s^{-1}, and a 100 year flood, 1800 m^3 s^{-1}. Given the alpine catchment, the highest discharges occur during snowmelt in May; second flow peaks occur in July/August (Hydrographic Service Styria). The Mur is a gravel bed river; at the border reach, the bed material has a mean diameter of approximately 35 mm.

At the border reach, the Mur River was once a braided river system. A landslide in the 15th century led to the beginning of a lateral shifting of the river. The already dynamic river system imposed great problems on the densely populated area. Several settlements and important infrastructures were destroyed (H. Habersack et al., unpublished report, 2001a). The human impacts on the river morphology started with local regulations in the Middle Ages, culminating in a systematic channelization between 1875 and 1891. At that time, channel incision was set as a goal. Owing to increasing water transport capacity within the channel boundaries, the incision was appraised as a positive effect of the channelization. Earlier, this stretch of the Mur River was up to 1200 m wide. Since systematic channelization, this stretch is constrained into a straight, 76 m wide channel. This resulted in larger river radii and hence shorter length, which ultimately increased the channel slope [*Habersack and Schneider*, 2000] (Figure 2). As a result, the bed load transport capacity increased due to higher shear stresses. During channelization works, the riverbanks were protected with riprap, which inhibit bank erosion and any lateral sediment input. Additionally, at the beginning of the 20th century in the upstream reach, the construction of a chain of hydroelectric power plants started and bed load input into the investigated reach diminished. Before the channelization, the Mur River had a balanced sediment budget or was in a slightly aggrading state (H. Habersack et al., River morphology, final report for the Austrian-Slovenian Standing Committee for the Mur River, Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished, 2001b). Based on these new boundary conditions, the riverbed began to incise. Between 1970 and 2000, the mean degradation in the 34 km long border reach was about 0.5 m, the maximum about 1.2 m (J. Plattner, Bed level change, final report for the Austrian-Slovenian Standing Committee for the Mur River, Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished, 2001). This yielded a deficit of sediment volume of approximately 900,000 m^3 in this time period in the studied reach; the degradation between 1977 and 1995 is depicted in Figure 4.

The channelization and the incision increasingly led to technical/economic and ecological problems. The incision induced scouring and destabilization of bank protection structures in built-up areas, separated old river branches and alluvial forests from the main channel, and lowered the adjacent groundwater table. Bank protection structures had to be repaired or repeatedly reconstructed to avoid uncontrolled erosion and flood damage. The lowering of the

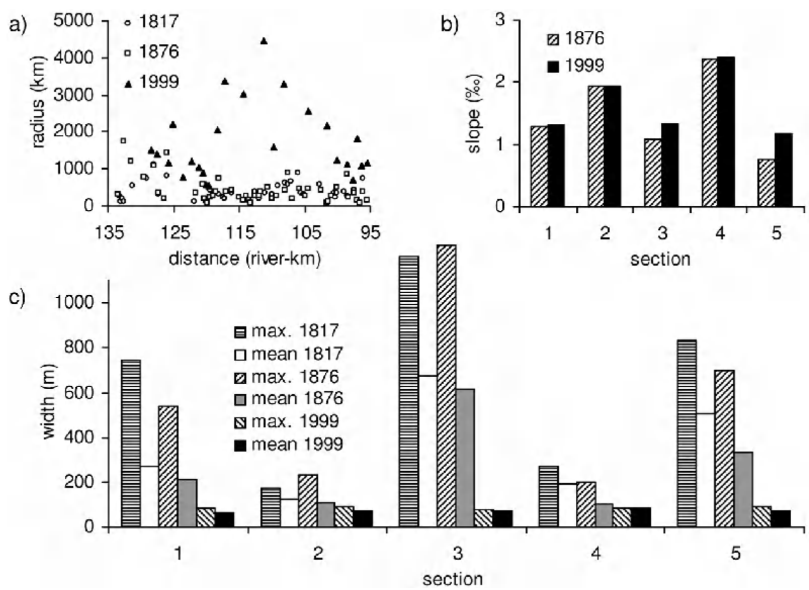

Figure 2. Changes of morphological parameters over time at the Mur River: (a) radius, (b) slope, and (c) width. Sections 1–5 according to geomorphic characteristics are as follows: 1, straight; 2, gorges; 3, historically braided; 4, constrained in urban areas; and 5, former braided-meandering. For location of the sections, see Figure 1. Modified after *Habersack and Piégay* [2008, p. 706]. Copyright Elsevier 2008. Reprinted with permission.

groundwater table compromised the water supply. The morphological change of the Mur River severely impacted the fish and macrozoobenthos population. Compared with historically documented biodiversity data, the number of fish and macrozoobenthos species remained stable, but shrinking habitat size and growing habitat uniformity considerably reduced the population size of individual species. From an estimated several hundred kilograms per hectare in the reference state, the biomass diminished to approximately 60 kg ha^{-1} (M. Jungwirth et al., Fish ecology, final report for the Austrian-Slovenian Standing Committee for the Mur River, Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished, 2001). The habitat loss is clearly evident in the disappearance of gravel bars: Once reaching sizes of up to 80,000 m^2 per bar occurred; most bars today measure <1000 m^2 (H. Habersack et al., unpublished report, 2001b). The lack of backwaters and reduced connectivity to the main channel removed self-maintaining populations of fish groups requiring strong lateral connectivity (M. Jungwirth et al., unpublished report, 2001). Biodiversity remains high due to the open river continuum (1000 km long) downstream between the Mur, Drava, and Danube rivers and due to the higher habitat diversity immediately downstream of the border reach in Slovenia. Given these conditions, a high recolonization potential can be expected for restored sections.

Compared to other rivers suffering from far greater incision (e.g., Magra River [*Rinaldi et al.*, 2009]), the incision at the Mur River might seem to be negligible. Nonetheless, investigations on the thickness of the remaining gravel layer proved to be of critical importance in the Mur River: Drillings into the sediment showed that, in the year 2000, a cross section of the Mur, which is representative of the area, had an only 0.5 m thick gravel layer (Figure 3). The coarse Quaternary sediment is underlain by finer Tertiary sediment, which, once reached by the incision, would quickly erode. In Austria, a riverbed breakthrough has already occurred at the Salzach River [*Hengl*, 2004]. Without countermeasures, the process is irreversible. Once the gravel layer is lost, the incision cannot be stopped without major intervention into the river system (weir or sill construction). Without a gravel riverbed, widening as a restoration measure would be futile [*Habersack and Piégay*, 2008]. Accordingly, with respect to the goal of achieving good ecological status by 2015, as mandated by the European Water Framework Directive, ecologically oriented methods are preferable. For the Mur, such measures are urgent.

3. BASIC WATER MANAGEMENT CONCEPT

Between 1998 and 2001, an Austrian-Slovenian expert team completed a Basic Water Management Concept for the border reach of the Mur River [*Austrian-Slovenian Standing Committee for the Mur River*, 2001]. Recognizing the problems outlined above, objectives were to (1) prevent further incision of the riverbed, (2) guarantee protection of inhabited areas and infrastructure against the 100 year flood, and (3) allow the water system to develop according to its natural dynamics in the long term.

A sediment transport model was applied (M. Hengl et al., Bedload transport model, final report for the Austrian-Slovenian

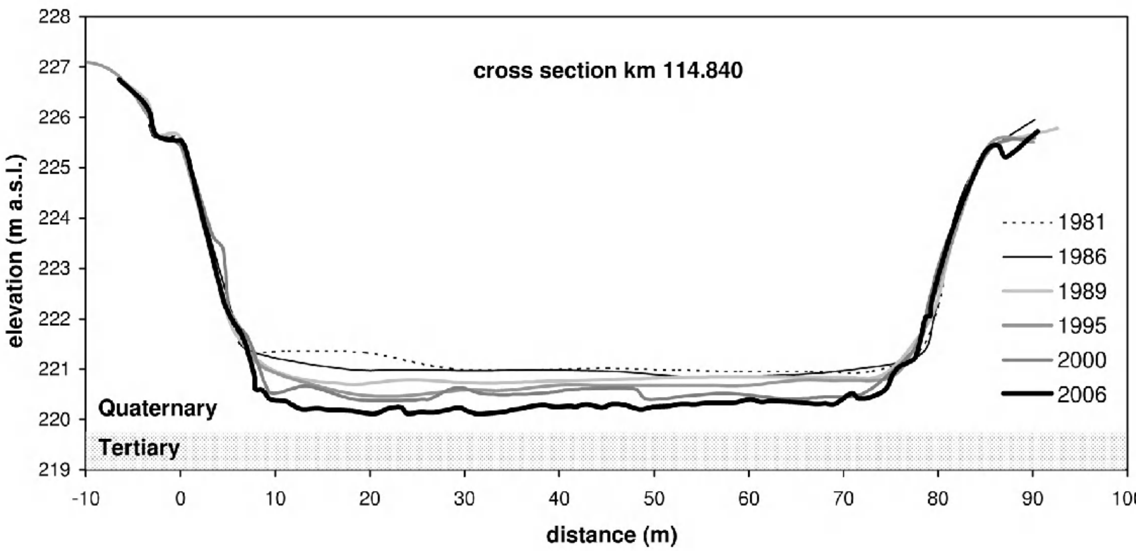

Figure 3. Degradation in the cross section with the highest incision rate and declining distance to the finer Tertiary sediment, view downstream.

Standing Committee for the Mur River, Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished, 2001) to predict the further development of the riverbed. Simulations were performed for the case without countermeasures and to test the effect of different restoration scenarios. The employed model MORMO was developed at ETH Zürich and is a one-dimensional (1-D), quasi-steady state model used to calculate sediment-transport rates and bed changes of rivers. In fact, a 1-D sediment transport model is limited to calculating the mean cross-sectional bed level change. However, given the length of the investigated reach (>30 km) and the length of the time period to be modeled (60 years), a 2-D sediment transport model would not have been applicable. The bed load transport equation is based on the *Meyer-Peter and Mueller* [1948] formula, modified to calculate fractional sediment transport by including a hiding function into the formula, following suggestions from *Hunziker* [1995]. Small shear stresses, which lie below the critical value that initiates motion, may exceed the critical shear stress if the values fluctuate. To account for this effect, the approach of *Pazis and Graf* [1977] has been integrated into the calculation. Variations of depth and flow velocity within a cross section have been considered by replacing the irregular, natural profile by a rectangular one with empirically derived dimensions after the method of *Zarn* [1997]. The armoring layer of the riverbed has been incorporated by using two sediment layers of different erodibility.

Cross sections surveyed every 100 m in 1998 served for reconstruction of the channel geometry in 1977, the beginning of the time period selected for model calibration. Sediment input into the reach from upstream was estimated to be zero because of the interruptions of the sediment continuum

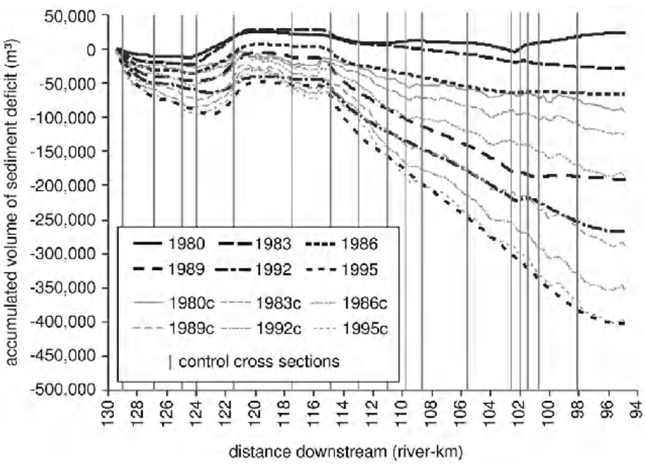

Figure 4. Volumetric sediment deficit in the studied river reach compared to the state in 1977 and calibration results (labeled with 'c') from the sediment transport model. Modified after the work of M. Hengl et al. (unpublished report, 2001).

by the weirs of hydroelectric power plants. Model calibration was conducted by comparing measured and modeled change of sediment balance in the investigated reach (Figure 4). Sediment balance was calculated based on the bed level changes in repeatedly surveyed cross sections distributed over the studied border reach (14 cross sections from 1977 to 1995, three additional cross sections in 1992 and 1995). Additionally, the modeled bed level changes were validated separately in these cross sections.

The flow used for predicting future bed level change was generated based on the hydrograph from 1977 to 1995 measured at the gauging station Mureck (situated within the investigated reach). Without changing the existing conditions, the model predicted further incision exceeding the critical distance to the Tertiary sediment layer. A bed breakthrough within 60 years was a distinct possibility.

Applying the 1-D hydrodynamic flow model Hydrologic Engineering Center River Analysis System (HEC-RAS) to a schematic channel representing the Mur River geometry enabled calculating the shear stresses at different bed widths and channel slopes. The results showed that the riverbed width strongly influences the shear stresses exerted by the flow. The conclusion was that channel incision can best be stopped by implementing riverbed widenings, which effectively reduce shear stresses and hence the sediment transport capacity (H. Habersack et al., unpublished report, 2001a).

Riverbed widening has become a very common restoration measure in Austria and Switzerland [*Jaeggi and Zarn*, 1999; *Rohde*, 2004]. According to *Hunzinger* [1998], the self-dynamic morphological changes along the longitudinal profile associated with river widening are the following: (1) The mean bed level in the widening is stepped vertically relative to the bed level in the upstream and downstream channel to ensure continuity and energy conservation. (2) A new equilibrium slope becomes established. After widening, the initial reduction of transport capacity is compensated by the formation of a slope steeper than the slope of the original narrow streamway. In the case of river widenings over longer stretches, this effect increases the upstream bed level. (3) Bars are formed, creating a more diverse flow pattern. At the same time, cross flows and scouring increase the hydraulic load on the riverbanks. (4) The flow is concentrated in the downstream part, causing intense scouring at the constriction. (5) Sediment is retained within the widening, causing temporary downstream erosion (based on *Habersack and Piégay* [2008]). Widenings temporarily cause a sediment deficit in the downstream reach. If the widening is accomplished by self-initiated bank erosion that continuously inserts sediment into the channel, this effect can be temporarily avoided or even inverted, perhaps even until a balanced transport through the widened section is achieved.

Moreover, the construction of sills and ramps at regular intervals across the river length was discussed during the development of the Water Management Concept. While sills and ramps would not improve the ecological situation, widened river sections best match the historical reference state and generate a comparable habitat diversity and ecological status (H. Habersack et al., unpublished report, 2001a). In addition, costs for construction and maintenance of riverbed widenings are comparatively low. Considering these arguments, the Basic Water Management Concept concentrated on riverbed widening as a solution to channel incision.

Natural or infrastructural constraints within the studied reach make only parts of the river suitable for riverbed widenings. Accordingly, additional sediment input is needed to supply nonwidened sections with sediment. As mentioned above, short widenings do not automatically lead to a sediment deficit downstream, if they are combined with self-dynamic bank erosion. As long as bank erosion mobilizes sediment in the widened section, sediment is supplied to the downstream, channelized section. Finally, self-dynamic widening by lateral erosion to the doubled width was suggested. Solely removing the bank protection structures introduced bank-derived sediment into the channel over a long time period, with the erosion rate depending on hydrology, bank erodibility, and morphodynamics within the widened bed.

The floodplain across the whole river length of the border reach was investigated for stretches suitable for riverbed widenings. The migration and high bed load transport of the historical river system stored high amounts of Quaternary deposits in the floodplain. The whole river length was evaluated to determine the amount of gravel in the adjacent floodplains and the relative elevation of the gravel layer compared to the elevation of the riverbed. The suitability for riverbed widenings across the river length was assessed by also considering natural and infrastructural constraints to the channel (e.g., only 86 m river width in town Bad Radkersburg). By coupling the suitability with the historical and the predicted degradation, sections suitable for riverbed

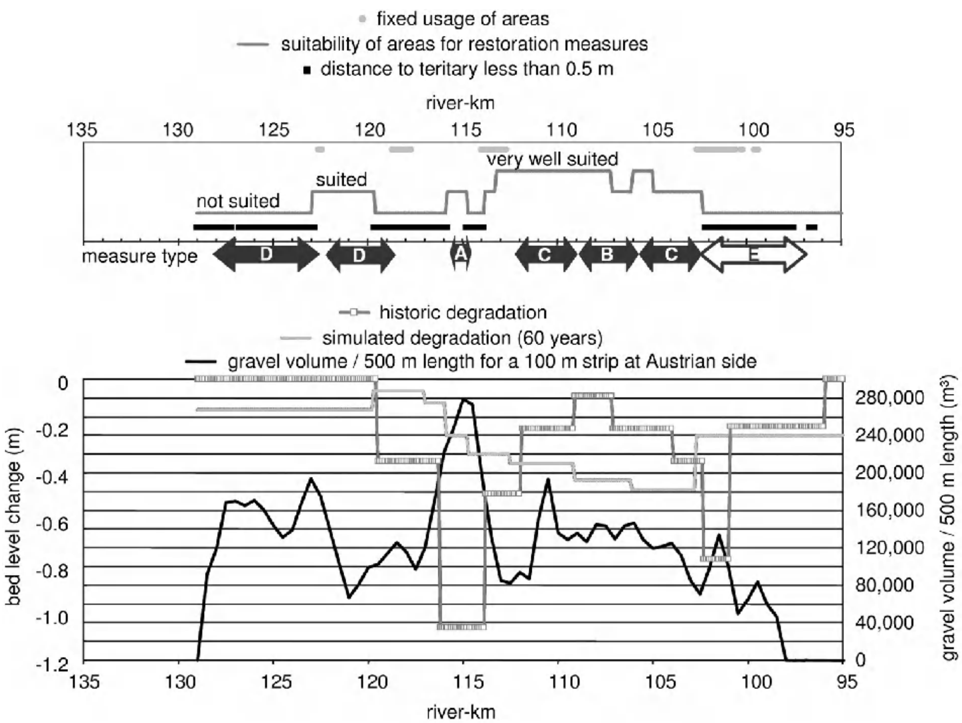

Figure 5. Recommended positioning and types of restoration measures according to the section suitability (determined according to gravel availability in the floodplain, thickness of remaining gravel layer, and constraints) and to the historical and predicted bed degradation: A, riverbed widening to about 200 m and artificial bed load supply; B, self-forming side erosion and increase of riverbed width, C, initial riverbed widening, followed by side erosion, D, activation of gravel deposits without changing the effective bed width in combination with sidearm reconnections, and E, alternative bed stabilization by ramps or local grain size increases. Arrows show the length of the various measure types. Modified after the work of *Habersack and Piégay* [2008, p. 725]. Copyright Elsevier 2008. Reprinted with permission.

widenings were localized (Figure 5). High priority was given to reaches with extreme degradation. Sections with a gravel bed thickness <0.5 m above the Tertiary were excluded because bed breakthrough due to scouring processes remains a threat even if widening takes place (e.g., during bed form migration) [*Habersack and Piégay*, 2008]. Finally, five types of measures were identified: A, riverbed widening to about 200 m and artificial bed load supply; B, self-forming side erosion and increase of riverbed width; C, initial riverbed widening, followed by side erosion; D, activation of gravel deposits without changing the effective bed width in combination with sidearm reconnections; and E, alternative bed stabilization such as by ramps or local grain size increases. A stepwise realization of the measures was therefore suggested to optimize gravel availability and extend the lifetime of the measures, recognizing that long-term gravel input from outside the reach is essential [*Austrian-Slovenian Standing Committee for the Mur River*, 2001; *Habersack and Piégay*, 2008].

The effects of these five types of measures were simulated with the calibrated sediment transport model. Bed widening was integrated into the 1-D sediment transport model for simulating its effects on sediment transport. Given the lack of observations on self-dynamic widening at the Mur River at the time the Basic Water Management Concept was completed, only the simulation of a widening scenario could be conducted. Bank erosion was estimated to take place as soon as sediment transport was fully active (at 300 $m^3\ s^{-1}$). At discharges above this threshold, the bank erosion rate was estimated to increase linearly with discharge. The channel width was adjusted accordingly in the 1-D model. It is now the aim of the monitoring to verify this simplified widening employed in the model, leading to the adaption of planned measures and their succession of implementation.

The numerical simulations showed overall positive effects of the measures. A stepwise instead of a simultaneous realization of the measures extended the lifetime of the measure combination. Several successions of measure implementation were tested. The succession that yielded the best result is shown in Figure 6. Considering that bed load supply from upstream is essential for the functioning of riverbed widenings, the bed load input from upstream is increased from an estimated 150 $m^3\ yr^{-1}$ at present to 2600 $m^3\ yr^{-1}$.

Based on the calculated 900,000 m^3 of sediment deficit between 1970 and 2000, the mean annual deficit amounts to 29,000 $m^3\ yr^{-1}$. Shifting bed load mobilization from bed degradation to lateral erosion still yields a deficit but does not affect the riverbed (M. Hengl et al., unpublished report, 2001). Following the implementation schedule depicted in Figure 6, the model showed that enough sediment is available for 60 years to stop the existing degradation tendency. These 60 years would provide enough time to develop and implement measures to improve the sediment input into this reach (e.g., by optimizing the weirs and reservoir conditions to support gravel throughput during floods [*Habersack and Piégay*, 2008]). Figure 7 shows the simulated longitudinal

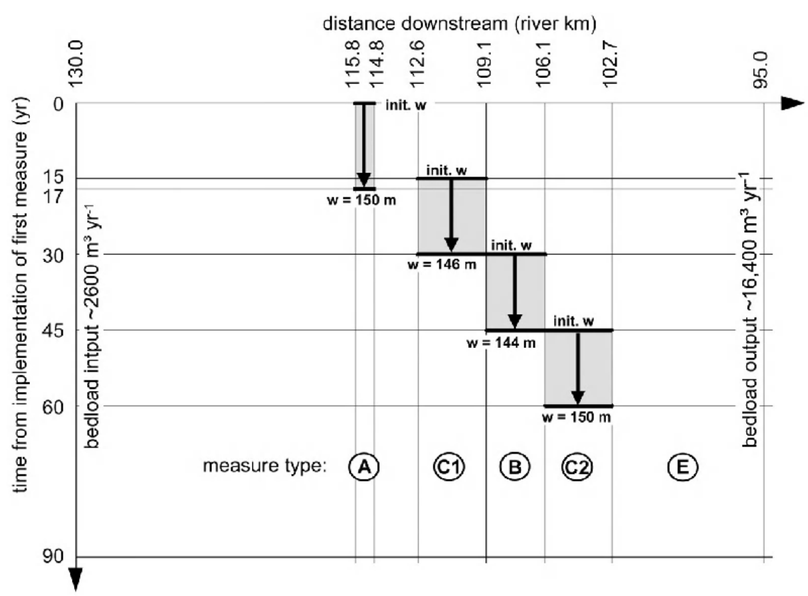

Figure 6. Succession of measure implementation that best mitigated channel incision in the sediment transport model (w, riverbed width; init. w, initial riverbed width). Modified after the work of M. Hengl et al. (unpublished report, 2001).

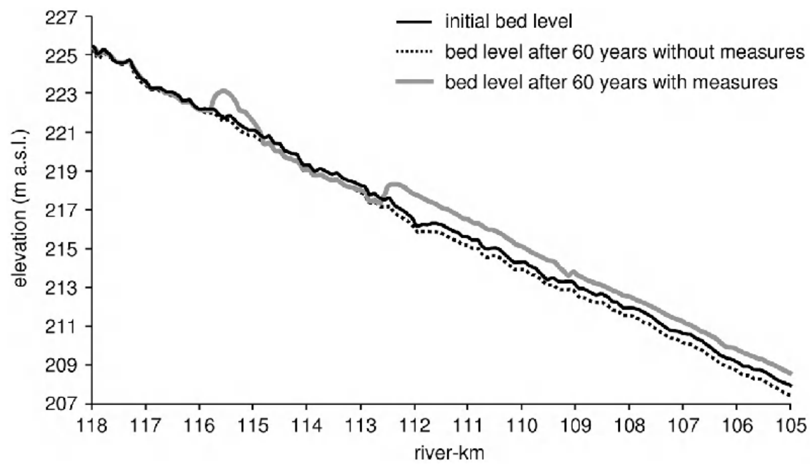

Figure 7. Simulated development of the longitudinal section after 60 years. Modified after the work of M. Hengl et al. (unpublished report, 2001).

section after a 60 year development of two variants, one in which no measure is implemented and one corresponding to the measure combination in the succession outlined in Figure 6.

4. PILOT MEASURE AT GOSDORF

4.1. Measure Implementation

Since 2001, four small measures have been implemented, each of them contributing 12,000 to 25,000 m^3 gravel by self-initiated side erosion or as an immediate artificial input. As recommended in the Basic Water Management Concept, the first major measure was implemented between river km 115 and 116 at Gosdorf, directly at the beginning of a reach suffering from severe bed degradation (see measure type A in Figures 5 and 6). There, the Concept proposed a riverbed widening to about 200 m and artificial bed load supply. Widening due to bank erosion was enabled by removing the bank protection structures along the left bank over a length of 1 km. A new sidearm was excavated, and the dredged material was introduced into the main channel as an immediate artificial bed load supply (approximately 150,000 m^3, Figure 8). According to the assumed lateral development, another 300,000 m^3 bank-derived gravel will enter the channel. In the middle of this section, a bedrock sill emerged from the bed because of the past incision. A brook with a mean discharge of 1.6 $m^3\ s^{-1}$, which previously flowed directly into the Mur River in the middle of this section, now flows into the sidearm (J. Plattner, Einreichprojekt Mur Gemeinde Gosdorf Flussaufweitung, technical report for the Styrian Water Management Authority and Austrian Ministry of Agriculture, Forestry, Environment and Water Management, unpublished report, 2005) (Figure 9). The measure was completed in 2006/2007. The border reach of the Mur River, with its alluvial forests on Austrian territory, is designated as a European Conservation Area (Natura 2000). The construction of the sidearm required removing large vegetated areas, considerably reducing the originally mapped area for habitats according to the European fauna, flora and habitats directive.

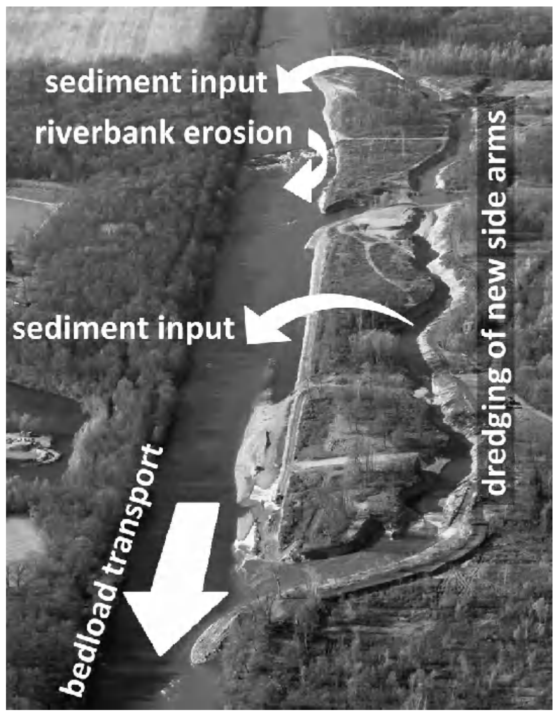

Figure 8. Schematic drawing of the functionality of the measures at Gosdorf.

Nonetheless, the expected ecological benefits in the near future led to acceptance of the initial work [*Hornich and Baumann*, 2008].

4.2. Monitoring Activities

An intensive monitoring program was conducted to observe and evaluate the development of the measure and its effectiveness in mitigating further channel incision. The measure is considered effective if the following can be achieved: (1) bank erosion occurs and leads to bed widening; (2) widening reduces the transport capacity in the restored section and stabilizes the bed level; (3) the artificially introduced gravel and the gravel originating from bank erosion compensate the bed load deficits in the downstream, regulated section; (4) the lifetime of the measure, depending on the velocity by which the introduced gravel is transported downstream and on the rate banks are eroded in the restored section, is according to plan (Figure 6). To provide a good database for model calibration, the monitoring was conducted at short time scales. This enabled processes to be linked to individual flow events or even to individual conditions of flow dynamics [*Klösch et al.*, 2008].

This approach included (1) echo sounder measurements of riverbed geometry, (2) tachymetric survey of riverbanks and hinterland, (3) survey of "control cross sections" in the downstream section, (4) terrestrial photogrammetry at selected riverbanks, (5) particle tracking using telemetry, (6) basket sampler bed load measurements, (7) grain size analysis of bed material, (8) time lapse camera surveys, and (9) monitoring of riverbank stability. The ecological status is monitored to evaluate the implications of the measures on ecology.

4.2.1. Monitoring of morphology. The geometry monitoring combined methods yielding high spatial resolution with methods yielding high temporal resolution. High spatial resolution of the riverbed geometry was obtained from dense echo sounder measurements (distance of measured cross sections <3 m). The hinterland was surveyed tachymetrically, whereas the geometries of the riverbanks were surveyed using reflectorless measurements. The resulting digital elevation models (Figure 10) yielded maps displaying elevation changes and allowed mass balances to be calculated [*Formann et al.*, 2007]. To observe changes of bed elevation, the periodic survey of those profiles surveyed since 1970 was continued. Profiles were added in the close-up range of the restored section.

The previously outlined survey of the restored section was repeated once or twice per year. Several methods are added to obtain a higher temporal resolution in the monitoring of the geometry: (1) four webcams, originally installed mainly for public information, were used to gain bank geometry data during flood events, (2) selected riverbanks were surveyed after every flood event using terrestrial photogrammetry, (3) in the sidearm, cross sections were surveyed after every flood event. In the same cross sections, erosion pins were installed in the riverbanks; if the pins are not submerged and

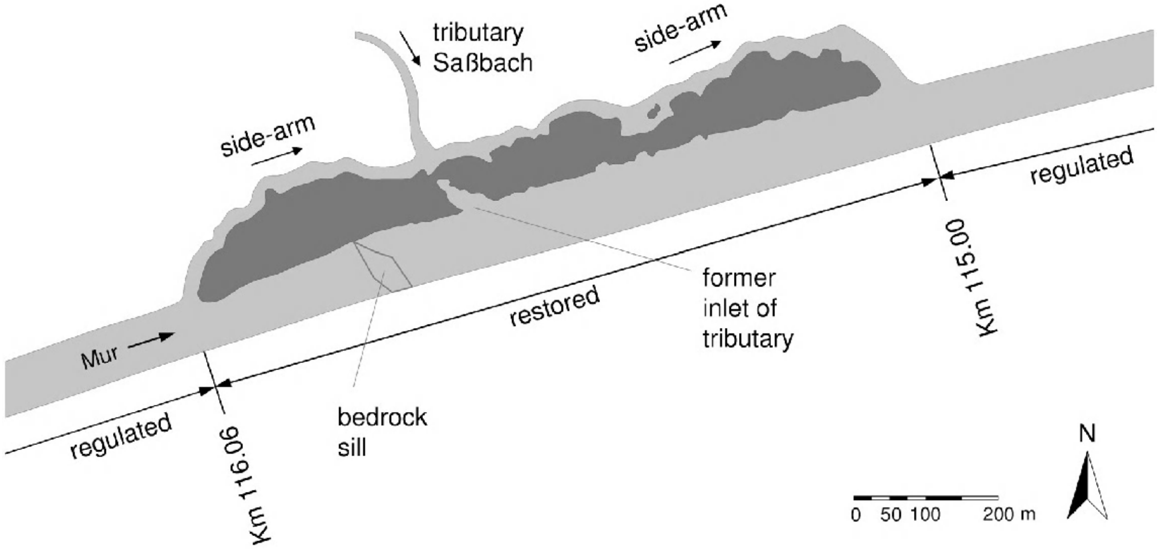

Figure 9. Plan of the measure.

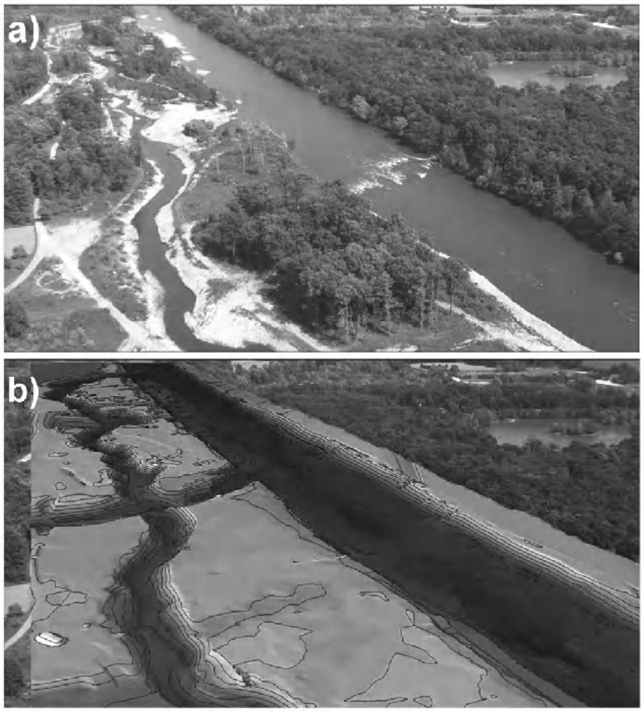

Figure 10. (a) Restored section at Gosdorf (aerial photograph from Regional Government of Styria) and (b) its replication as a digital elevation model.

if personnel are available on-site, then the pins can be observed during flood events to monitor riverbank geometry.

4.2.2. Monitoring of bed load transport. The use of particle tracking via radio telemetry shows the transport paths of artificial stones representing coarse gravel [*Habersack*, 2001; *Habersack and Klösch*, 2008; *Klösch et al.*, 2008]. Fifteen stones of mean bed material diameter were deployed. Their positions were surveyed periodically by boat using a receiver and GPS. Five stones were observed continuously to detect the initiation of motion using antennas installed on the riverbank, a data logger, and transmitters with a motion detector [e.g., *Habersack*, 2001]. Additional measurements with a basket sampler were conducted to measure cross-sectional bed load transport.

4.2.3. Monitoring of riverbank stability. Samples from the riverbank sediment were taken in different depths in several cross sections after removal of the riprap and grain size distributions were analyzed. The riverbanks of the restored section consist of gravel overlain by silty sand of varying thickness. Sandy soils are weakly cohesive, but at unsaturated conditions, shear strength increases due to apparent cohesion [*Fredlund et al.*, 1978]. This stabilizes even vertical banks. Riverbank stability varies according to variations in pore water pressures as a consequence of rainfall, evapotranspiration, and water stage variations [*Rinaldi and Casagli*, 1999]. Tensiometers and gypsum blocks were installed to monitor matric suction to analyze the variations of riverbank stability. Positive pore water pressures were measured with two piezometers. Precipitation and water stage were measured directly at the investigation site [*Klösch et al.*, 2008].

4.2.4. Monitoring of ecological integrity. To monitor the ecological status, the biotopes, fish fauna, and macrozoobenthos were monitored before and after implementation of the measures (e.g.,. Jungwirth et al., unpublished report, 2001; A. Wilfling et al., Aufweitung Mur bei Gosdorf-Post-monitoring 1, report for the Styrian water management and nature authorities, unpublished report, 2010). To observe the development of aquatic habitat diversity, we computed flow velocities, water depths, and shear stresses for the regulated channel and for the restored state using a 2-D hydrodynamic numerical simulation.

4.3. Monitoring Results

4.3.1. Transport of deployed gravel. The measured transport paths of the tracers show that the gravel was transported during comparatively small flow events (Figure 11). The data were separated into movements within the restored section and within the channelized downstream section. Note that once the gravel enters the channelized section, motion is accelerated. There, transport paths of up to 2.5 km were observed during small flow events. The results demonstrate the reduced transport capacity within the restored section but also the supply of the downstream channelized section with gravel.

The continuous observation of five tracers allowed the initiation of transport to be detected during several flow events. The 2-D flow model (CCHE2D, NCCHE) allowed the shear stresses at the tracer positions to be calculated during initiation of transport. The average value of the critical shear stress for gravel of mean diameter was 26.7 N m^{-2} [*Puchner*, 2009]. The flow model was also used to compare the shear stress distribution on the riverbed in the restored versus former, channelized state (Figure 12). Drawing a threshold at the evaluated shear stress for gravel of mean diameter revealed that, at a discharge of 600 m^3 s^{-1}, the riverbed area in which gravel of mean diameter is transported is reduced by 11%. High shear stresses were reduced significantly, while the gravel bars in the restored state induced large areas of low shear stresses. Altogether, this proves the efficacy of the widening in reducing the transport capacity.

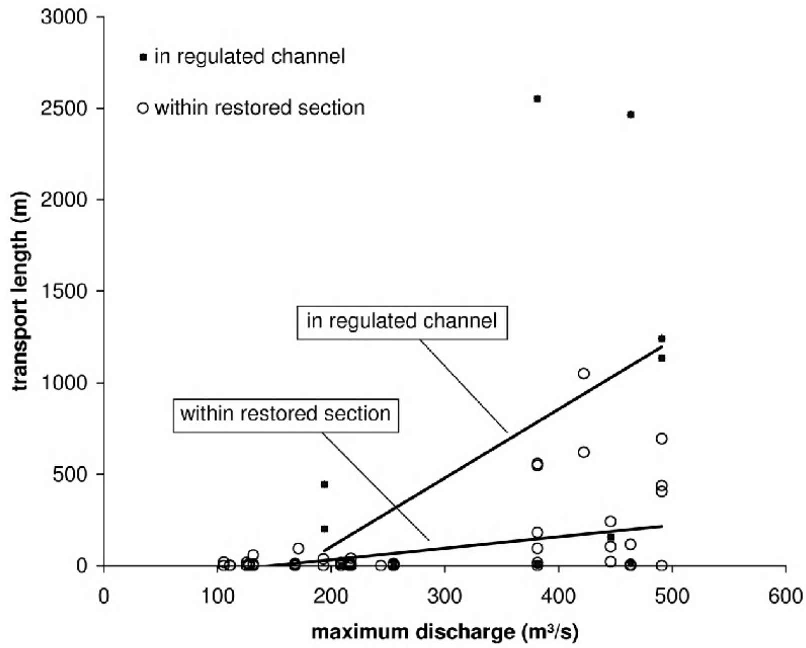

Figure 11. Transport length of tracers related to the maximum discharges between tracer positioning.

4.3.2. Riverbank erosion. Using terrestrial photogrammetry, digital elevation models of a 50 m long riverbank in the main channel were generated at four different dates to determine the eroded sediment volumes corresponding to three flow events. Because part of the bank consists of silty sand, not all bank-derived sediment is available as bed load. The data were therefore separated into eroded gravel volume and total sediment (Figure 13). The first flow event in 2008 (maximum discharge 464 $m^3\ s^{-1}$) removed relatively large amounts of fine sediment, which mainly accumulated on the bank toe in winter due to freeze/thaw processes. The comparatively high amount of eroded sediment in July might be explained by the earlier smaller flow events not considered in the monitoring. Owing to bank erosion, the three flow events introduced 1.8 m^3 gravel per meter riverbank length as bed load into the channel. The overlying fine sediment layer of the investigated riverbank is relatively thick (up to 1.5 m). Other reaches of the restored banks

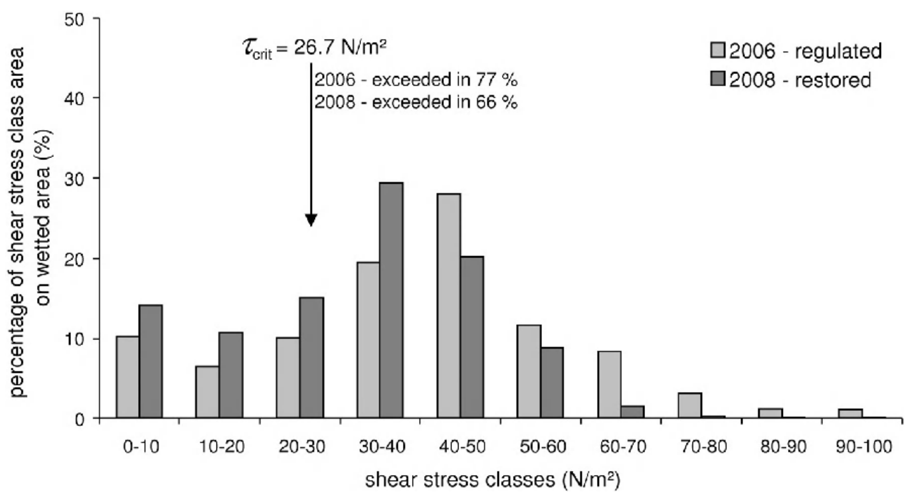

Figure 12. Shear stress distribution in the former regulated channel (2006) and in the restored status (2008), related to the measured critical shear stress for gravel of mean diameter.

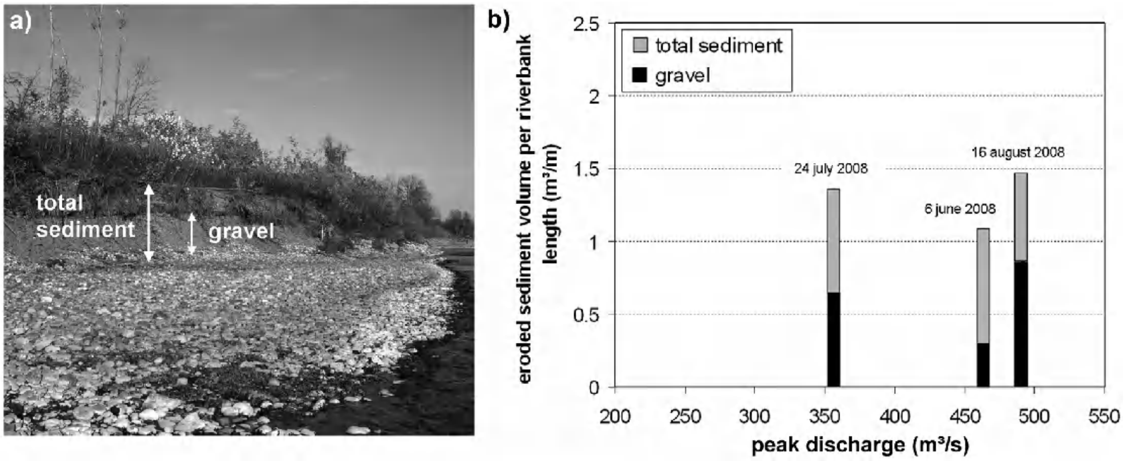

Figure 13. (a) Riverbank consisting of gravel and finer-grained sediment. (b) Eroded sediment volume per riverbank length due to three flow events along a 50 m long riverbank section.

show a higher percentage of gravel, which lead to a larger gravel input when they are eroded. Additionally, at banks mainly consisting of noncohesive gravel because of lower shear strength, higher retreat rates can be expected. Next to geotechnical properties, the retreat differs along the river from bank to bank according to the exposure to shear stresses of the flow. At the Drava River, the development of cross flows due to braiding has been identified as a key process for active bank retreat. Gravel bars laterally deflect the flow and hence increase shear stresses on the surrounding riverbanks (H. Habersack et al., Flussmorphologisches Monitoring an der Oberen Drau, final report, Carinthian Water Management Authority, Klagenfurt, Austria, unpublished report, 2010).

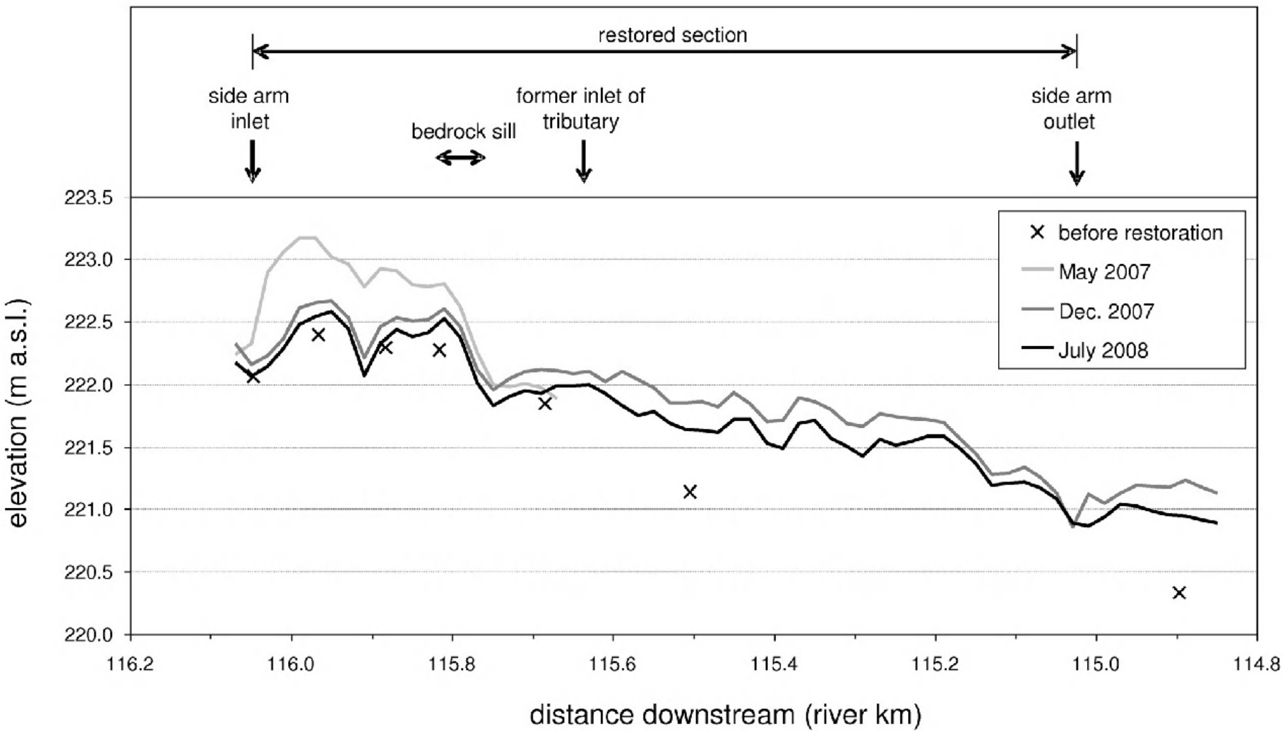

Figure 14. Comparison of mean bed elevation in the regulated status and after measure implementation.

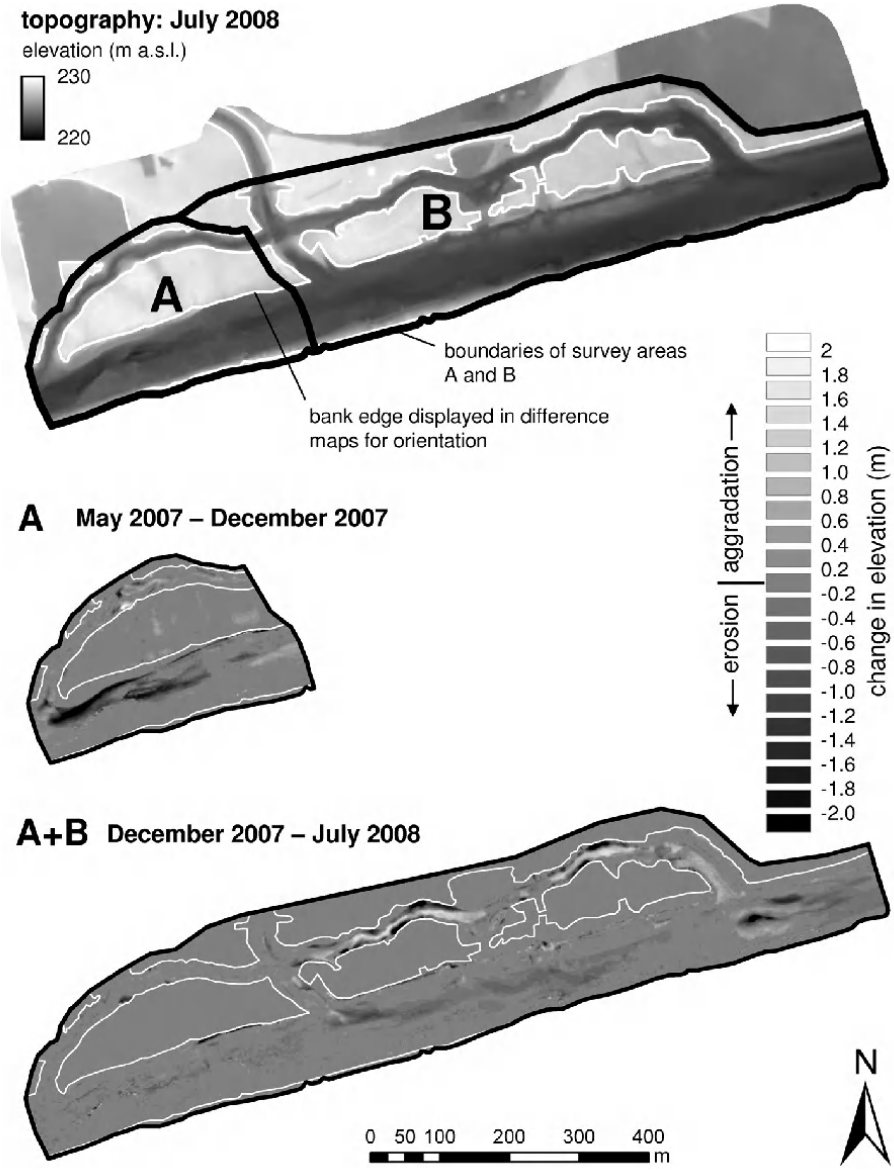

Figure 15. Changes in elevation within the restored section in two time intervals.

4.3.3. Bed level adjustment and sediment balance within the restored section. Three digital elevation models are available for the restored status (May 2007, December 2007, and July 2008), whereby the first survey was limited to the upstream section (the downstream section was still under construction). The input of 150,000 m^3 of gravel resulted in an aggradation of the riverbed, with a mean value per cross section of up to 0.75 m (Figure 14). Comparing the mean bed elevation of the former regulated channel with the present situation, the mean elevation still lies above the former incised bed. Because of further widening, the degradation is expected to diminish and turn into aggradation. The differences in elevation between the elevation models have been calculated to illustrate aggradation and erosion on a map (Figure 15). The hydrograph corresponding to the investigated time period is illustrated in Figure 16. At the initial geometries, even the small flow events induced strong morphodynamics (area A from May 2007 to December 2007 and area B from December 2007 to July 2008 in Figure 15). In the upstream section, the artificial sediment input generated a gravel bar that was approximately 150 m long and up to 25 m wide; lateral erosion started at relatively low discharges and

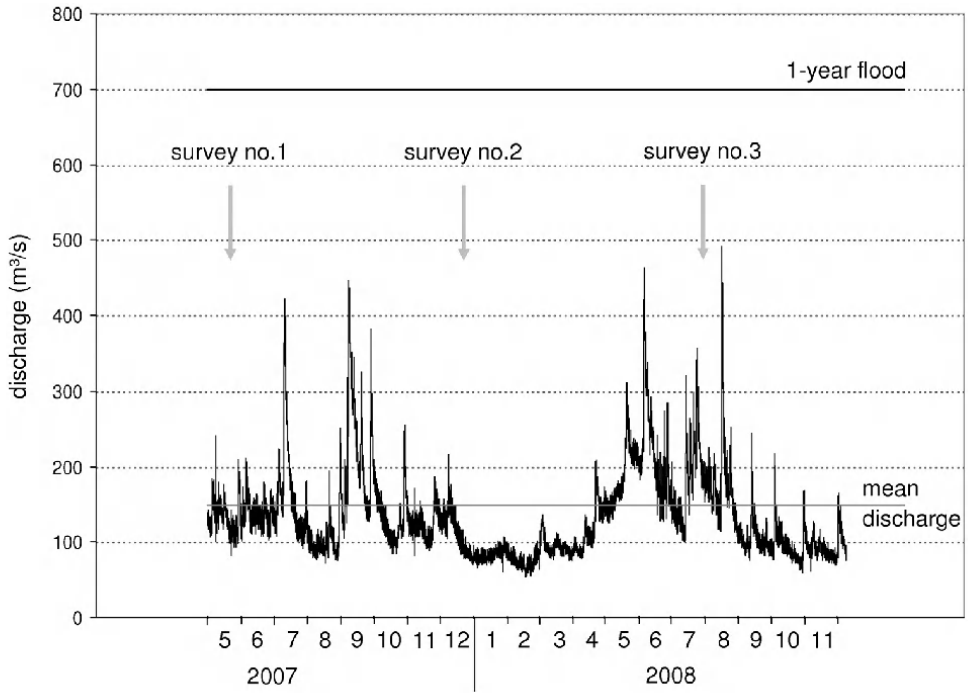

Figure 16. Hydrograph at gauging station Mureck (2 km upstream of restored section) between measure implementation and third survey.

caused high bed level changes. In the main channel of the downstream section, the artificial sediment was introduced more uniformly, resulting in wide-ranging, but less pronounced, erosion across nearly the entire river length. The upstream part of the sidearm remained stable, except for some local bank failures, in both time intervals. After the downstream part of the sidearm was connected, it experienced high morphological changes during a small flood of the tributary in June 2008. The flood mainly caused riverbank failures at the outer banks, accompanied by bed aggradation of locally up to 1.5 m.

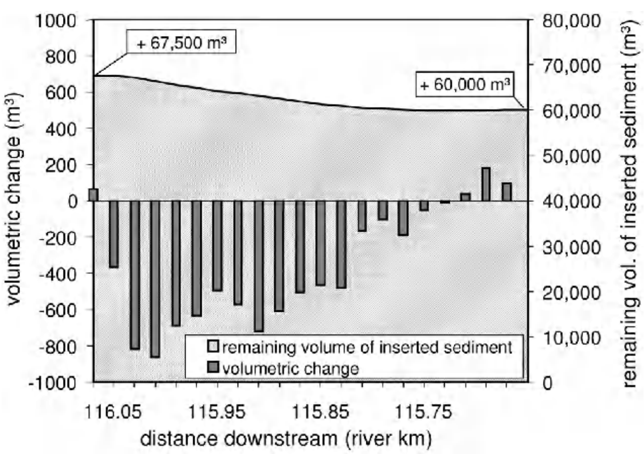

Figure 17. Mass balance and volumetric changes in section A between May 2007 and December 2007.

Calculating the volumetric changes across the whole river length enabled calculating the change of sediment volume within the restored reach and the amount of sediment supply into the downstream section. The volumetric change reflects bed load transport and bank erosion. The large amount of transported sediment volume upstream of the bedrock sill between May 2007 and December 2007 (Figure 17) mainly derives from the lateral erosion of the artificial gravel bar. Only 67,500 m^3 of the inserted 150,000 m^3 sediment was still in the restored section after the construction works were completed because part of the inserted sediment is fine-grained and hence transported as suspended load, and part of the bed load was already transported out during construction. Neglecting the minimal bed load input from upstream, 7500 m^3 of inserted and bank-derived sediment have been transported out of this section between May 2007 and December 2007, leaving behind at least 60,000 m^3 of the inserted material. Between December 2007 and July 2008, the volumetric change was negative nearly across the entire length, resulting in 13,500 m^3 of transported sediment. However, still not less than 46,500 m^3 remain in the restored section (Figure 18).

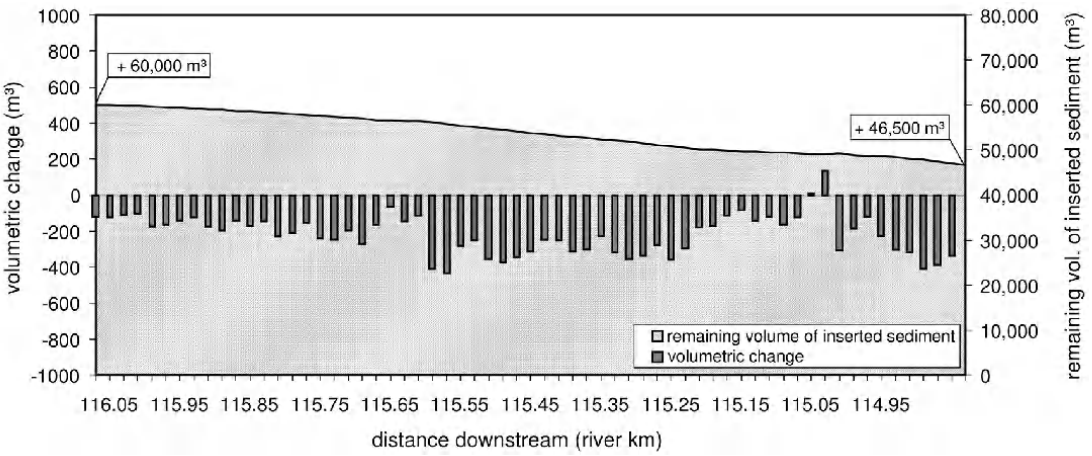

Figure 18. Mass balance and volumetric changes in the entire restored section between December 2007 and July 2008.

The upstream sidearm initially (from May 2007 to December 2007) experienced little morphological change and, mainly due to aggradation, a volumetric change of plus 800 m^3. The volumetric deficit of the second period, from December 2007 to July 2008 (after the downstream part of the sidearm was connected), increased continuously, in spite of some sediment input peaks due to bank erosion, with the distance downstream to more than 2000 m^3. At the same time, aggradation in the end of the sidearm reduced the deficit, but still more than 1000 m^3 of sediment ultimately entered the main channel (also neglecting bed load input from the tributary, Figure 19). Neglecting bed load supply from upstream (an estimated 150 m^3 yr^{-1}) and from the tributary, 14,500 m^3 of gravel were transported out of the restored section and introduced into the downstream, regulated channel as bed load supply between December 2007 and July 2008.

4.3.4. Bed level adjustment in the downstream, regulated section. Examining the development of bed elevation in the cross sections in the downstream, regulated reach helps evaluate the effectiveness of the measure in supplying the incised reach with gravel. It also reveals the distribution of the gravel. According to *Lisle* [2008], sediment waves with material equivalent to the ambient bed material tend to be more dispersing than translating. Accordingly, the supplied gravel would be well distributed over a long reach. In most downstream cross sections surveyed after measure implementation, strong bed level change occurred, which showed that the gravel deriving from the restored section was well distributed. The downstream cross sections also indicated that the incision there could already be stopped, significantly reduced, or even turned into an aggrading state (Figure 20). The aggradation in the cross sections 108.55 to 106.50 km

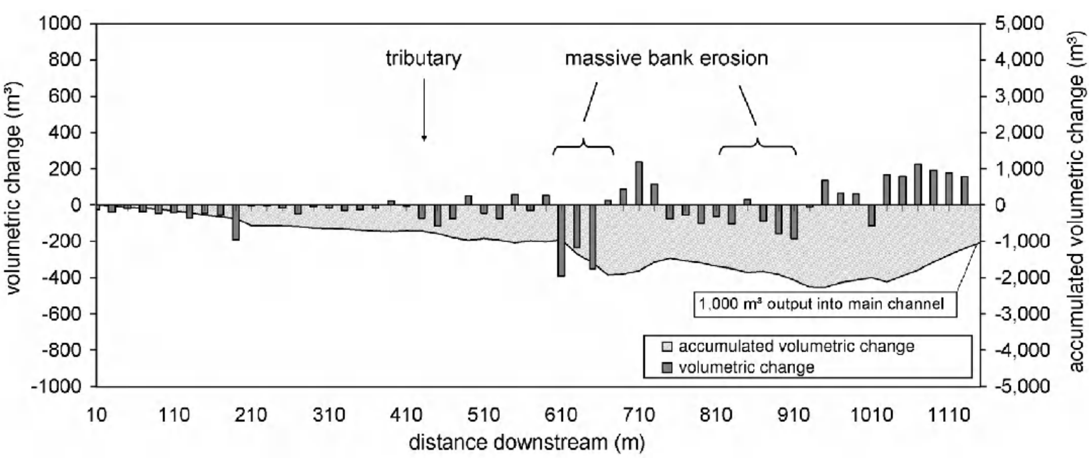

Figure 19. Mass balance and volumetric changes in the sidearm between December 2007 and July 2008.

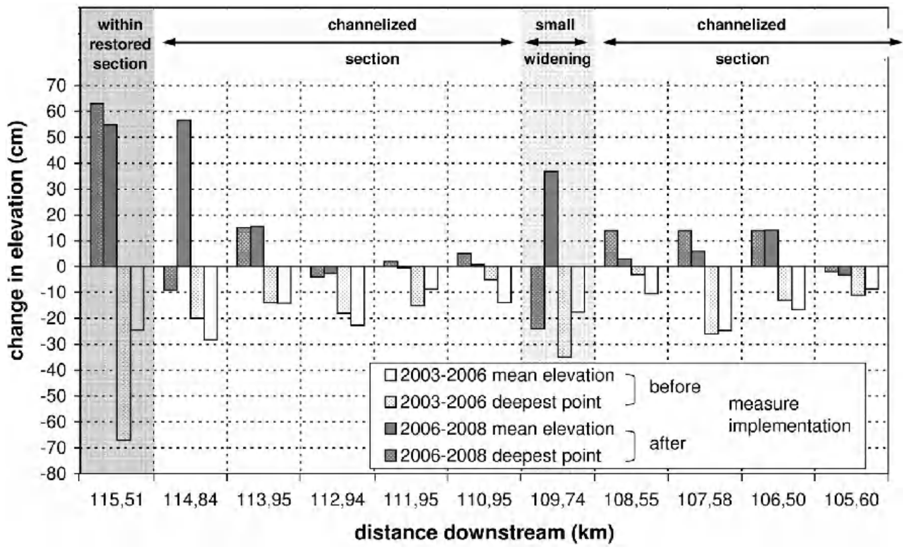

Figure 20. Bed level adjustment in the downstream regulated section, comparison of developments before and after measure implementation.

may also have been generated or increased by the small widening at river km 109.74 (completed in 2006) with artificial bed load supply (22,000 m^3).

4.3.5. Implications for ecology. Immediately after measure implementation, the length of the bank line was tripled compared to the initial state. This indicates an improved habitat diversity. The steep natural banks, gravel bars, and fine sediments were recolonized by endangered bird and insect species (H. Brunner et al., Zoologisches Post-Monitoring in Aufweitungen der streirischen Grenzmur, report for the Styrian water management and nature protection authorities, unpublished, 2010). The change in aquatic habitat diversity was illustrated by evaluating the results from a 2-D flow simulation at 600 m^3 s^{-1}. The percentage of water depth classes and flow velocity classes in the regulated and restored state are shown in Figures 21 and 22. Note that for comparison, also for the restored state, only the main channel has been evaluated. The development of the water depths shows that the distribution became flattened and more heterogenic, which results in higher habitat diversity. The high flow velocities (>3 m s^{-1}) diminished significantly, while the percentage of low flow velocities

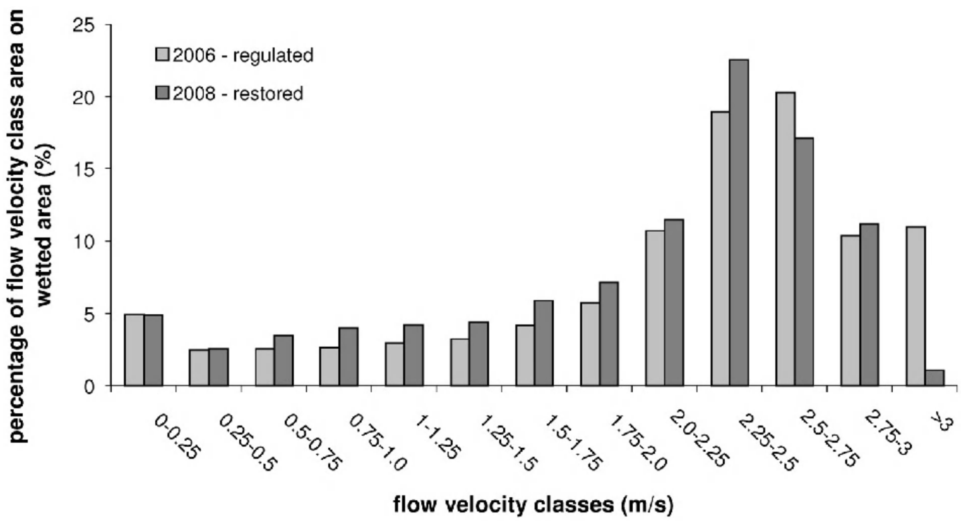

Figure 21. Comparison of flow velocities in the main channel in the regulated and restored state.

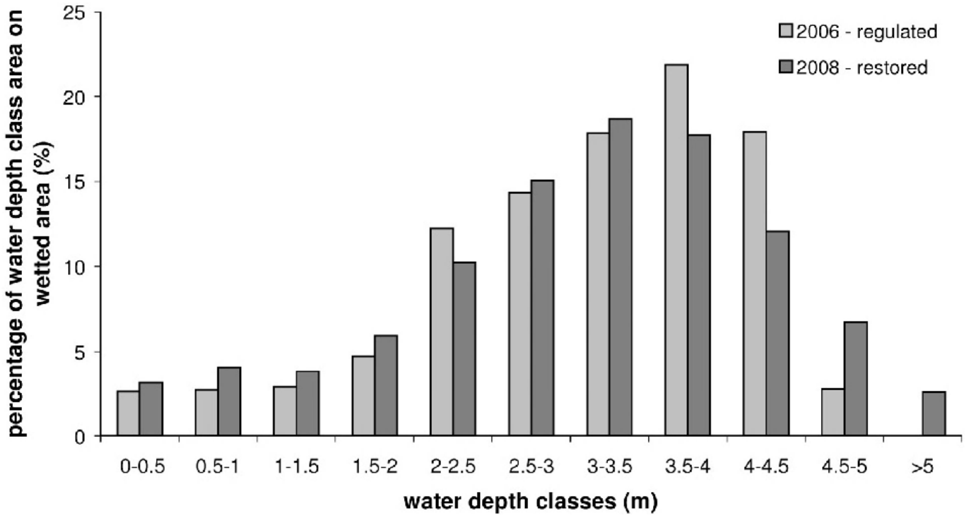

Figure 22. Comparison of water depths in the main channel in the regulated and restored state.

increased. At discharges as the simulated (600 m^3 s^{-1}), the habitats with little flow velocities are of great value for aquatic fauna as refuges.

5. CONCLUSIONS

This chapter describes measures that mitigate channel incision with ecologically oriented methods. It also evaluates the short-term success of a recently implemented measure. The riverbed gravel is recognized as a limited resource, with a potential threat of riverbed breakthrough and total loss of the gravel layer. Given the thin gravel layers in the Mur River, timely action is needed, especially if ecologically oriented methods are planned. The applicability of a sediment transport model to predict further developments with different scenarios is demonstrated. River widening has been identified as effectively reducing transport capacity. When implemented by self-initiated bank erosion, combined with additional artificial gravel input, the widenings also positively affected nonwidened sections downstream. At the Mur River, a stepwise realization of several measures is suggested to extend the lifetime of the measures. At the pilot measure in Gosdorf, the artificial sediment supply was derived from a newly dredged sidearm, which in the same time improved the ecological status.

An innovative monitoring concept to evaluate the effectiveness of the measure is described, and the short-term developments are presented. The results show the predicted response to the measure with respect to both the distribution of the inserted gravel and the self-initiated bank erosion. The gravel in the restored section was mobilized and minimized the incision in the downstream, regulated channel. The bed level within the restored reach has been raised, and the observed riverbank erosion indicates further widening. Nevertheless, sediment supply from widenings is time-constrained. This calls for solutions that reestablish sediment continuity from upstream.

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Salmon as Biogeomorphic Agents in Gravel Bed Rivers: The Effect of Fish on Sediment Mobility and Spawning Habitat

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Spawning salmon have been known to affect streambed texture, influence sediment transport, and play an important geomorphological role in streams and rivers. We examined the impact of salmon and floods on channel morphology, bed material dispersion and yield, bed surface texture and stability, fine-sediment dynamics, and nutrient retention of small gravel bed streams. Repeated channel surveys indicate that salmonids change postflood channel morphology, creating a hummocky bed surface through several cycles of redd creation. In streams with dense populations of sockeye salmon, the whole surface of spawning reaches may be modified, bars are excavated, and pools are filled. Analyses of coarse and fine sediment show that salmon increase sediment mobility by disturbing fine materials and preventing the development of an armored bed surface. Carbon-nitrogen ratios of the suspended sediment at the field sites and the gravel stored sediment in the flume indicate that salmon carcasses are the primary source of nitrogen to these systems. Further, the data suggest that the fine-grain bed sediments, in particular, are good retainers of salmon organic matter and thus function as nutrient stores for subsequent salmon generations. The study shows a sharp increase in the biochemical oxygen demand when salmon decay products combine with fine sediment and settle to the gravel bed; thus, the influx of nutrients and reworked gravel may aid in sustaining the salmon stocks and other biotic activity. Consequently, bed excavation by salmon plays a major geomorphic role in streams and improves the overall health of the ecosystem.

1. INTRODUCTION

Anthropogenic impacts that degrade fish spawning and rearing habitats have contributed to severe declines and extirpations of native salmonids around the world [e.g., *Gresh et al.*, 2000; *Montgomery*, 2003; *Finstand et al.*, 2007]. Riverine habitat degradation caused by human activities may partly explain the negative trends in fish populations because salmonids require suitable habitats for egg incubation as well as in-stream rearing before they migrate to sea [*Finstand et al.*, 2007; *Suttle et al.*, 2004]. When considering the restoration of salmonid habitat, one must consider the flow regime, channel morphology, and sediment quality. Sediment is a key factor for spawning, incubation of eggs, and rearing habitat for young fry. Many in-stream structures alter flow and

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sediment regimes and, in turn, degrade habitat [*Kondolf*, 2000a]. In particular, the area immediately downstream from dams is severely affected due to alteration of sediment and flow regimes. This frequently leads to downstream gravel starvation, bed surface armoring, and channel degradation (lowering of bed elevation), which decrease gravel bar deposits and result in reduced quantity and quality of salmonid habitat [*Kondolf*, 2000b]. Alternatively, road construction, urbanization, excessive logging, poor land use practices, and agricultural activities can result in an influx of fine sediment which can cement and infill the bed surface, infiltrate the stream bed, or be deposited on top of critical spawning gravel habitat [*Suttle et al.*, 2004; *Kondolf*, 2000b]. The cumulative impacts of human activities have reduced the amount and quality of salmonid habitat in many North American river systems.

Salmon also transport sediment through their spawning activity. The geomorphic role of spawning salmon involves nest (redd) excavation that modifies both the streambed topography and bed sediment texture and, hence, sediment mobility [*Gottesfeld et al.*, 2008; *Hassan et al.*, 2008]. Redds are excavated by the flexing action of the female, which creates lift forces that mobilize sediment to form a depression. Depending on the species and the size of the fish, the depth of this depression ranges from ~0.05 to 0.50 m [*DeVries*, 1997]. Fine silts and sands are lifted into the water column and carried downstream. Coarser pebbles and gravels accumulate in a pile, called the tail spill, at the downstream edge of the egg pocket [*Chapman*, 1988]. After spawning, the female proceeds to cover the eggs with gravel from upstream. In this process, the fine sediments are again carried downstream, so that eggs are covered with relatively coarse-grained sediment [*Rennie and Millar*, 2000]. The female will dig multiple egg pockets within the redd, a process that may take one or several days [*Gottesfeld et al.*, 2004]. Salmon tend to spawn on the upstream and downstream ends of riffles, and the edges of bars, but in streams with high spawning densities, their redd generation may disturb the entire channel bed [*Montgomery et al.*, 1996; *Gottesfeld et al.*, 2004, 2008; *Hassan et al.*, 2008].

The amount of sand in the gravel is reduced during spawning, resulting in the redd substrate becoming coarser than the surrounding undisturbed substrate [*Kondolf et al.*, 1993]. Redd sediments are therefore very permeable, which promotes flow of oxygen-rich water to the eggs and thereby increases fry survival [*Chapman*, 1988]. Indeed, eggs may not survive without the cleaning of gravel during spawning [*Zimmermann and Lapointe*, 2005]. Permeability decreases as interstitial voids fill with fine sediment as well as due to the development of sand caps that block flow paths on the bed surface. Reduced permeability decreases survival, both because of decreased dissolved oxygen availability and because the capping by fine sediment can make it more difficult for fry to successfully emerge.

Research has also focused on bed material scour and the probability of egg pocket erosion. *Montgomery et al.* [1996] compared depths of channel scour with egg burial depth for streams in Alaska and Washington and found that, in both regions, eggs were generally buried just below the average depth of channel scour. Similar findings were reported by *Rennie and Millar* [2000] and *Gottesfeld et al.* [2004]. Based on their results, *Montgomery et al.* [1996] suggest this to be an evolutionary adaptation of the fish to the long-term hydrologic regime. However, they warn that the close correspondence between scour depth and burial depth makes salmonid populations vulnerable to variations in scour. Scour depths may vary due to changes in the hydrologic regime and/or the supply of sediment associated with land use disturbances such as dam construction and timber harvesting.

While sediment composition is an integral component of fish habitat and therefore a regulating factor in habitat quality, most restoration plans pay little (if any) attention to fish-driven bed texture modification, sediment mobility, and changes in channel morphology. While abiotic forces shape fluvial habitat structures to a large extent, organisms like salmon that act as ecosystem engineers are pervasive and may exert strong feedback by mediating physical processes [*Jones et al.*, 1994; *Hassan et al.*, 2008]. Here we illustrate the effects of sediment transport and channel modification by fish and floods on salmonid habitat and suggest their consideration in restoration approaches.

2. STUDY STREAMS

The Stuart-Takla experimental watersheds are tributaries of the upper Fraser River in northern British Columbia, Canada (Figure 1). The study was conducted in the lower 2 km of three tributary creeks to the Middle River drainage, Forfar, Gluskie, and O'Ne-ell (Figure 1). At the river mouth, the drainage areas of the watersheds range between 38 and 76 km^2, with stream widths ranging from 5 to 20 m. These streams represent highly productive habitat for early and late run sockeye salmon, as well as a number of resident salmonid species [e.g., *Macdonald and Herunter*, 1998; *Gottesfeld et al.*, 2004]. The watersheds accumulate a substantial snowpack over the winter with a mean annual precipitation of 487 mm. In most years, maximum flows are associated with an extended spring freshet prompted by snowmelt in late May or early June, but occasional large flows occur following rain on snow events in late spring or following mid-to-late summer frontal storms. A 17 year record of flows (1992–2007) is available at the river mouth of the watersheds (Figure 1). The

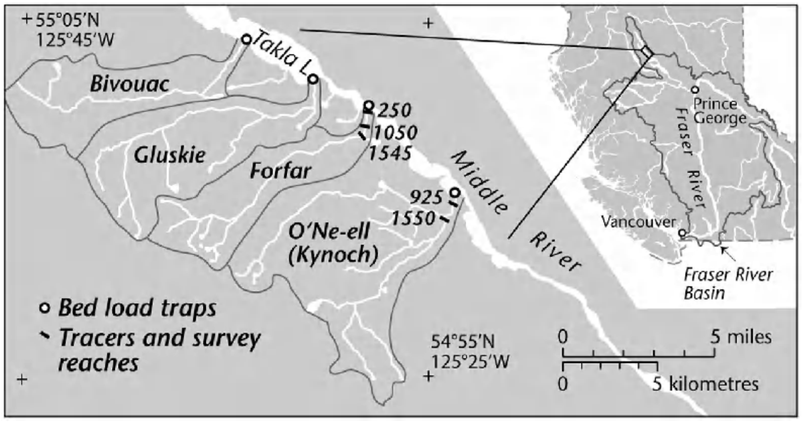

Figure 1. Location map of the study creeks. The values attached to each reach represent the distance upstream of the creek mouth in meters.

mean annual flood, 5 year, and 10 year return period floods are reported in Table 1. During the cold months (November–April), ice and snow cover the streams.

Bedrock geology consists of metamorphosed fine clastic sedimentary rocks of the Cache Creek complex of Pennsylvanian to Triassic age. All but the steepest slopes are drift covered, with till being widespread on the valley sides [*Plouffe*, 2000]. The texture of the till varies according to the underlying bedrock lithology. The headwaters of the creeks descend steeply through outwash fan material that has been incised since the close of the Fraser glaciation [*Ryder*, 1995]. The most important determinant of the surface geology and the sediment supply to the creeks, however, is late Pleistocene glaciation.

Table 1. Watershed, Flood, and Morphological Characteristics of the Study Creeks[a]

Watershed	Forfar	Gluskie	O'Ne-ell
Basin area (km^2)	38.4	51.2	76.4
Mean channel gradient ($m\ m^{-1}$)	0.064	0.038	0.066
Valley flat (%)	7.0	38.0	4.2
Steep land area (%)[b]	0.1	8.2	0.8
Bankfull width (m)	14.2	14.8	14.4
Bankfull depth (m)	1.03	1.30	1.51
Mean annual flood ($m^3 s^{-1}$)[c]	12	8	23
Five year flood ($m^3 s^{-1}$)[c]	14	11	28
Ten year flood ($m^3 s^{-1}$)[c]	18	14	32
Large wood debris ($N\ m^{-2}$)[d]	0.0641	0.0676	0.0485
Median size of bed material (mm)	31	22	29
Pool/riffle ratio (bankfull width)	1.7	2.3	4.4

[a]Partially based on the work of *Hogan et al.* [1998, Tables 2, 5, and 6].
[b]Percent of drainage basin area.
[c]Based on 17 year record.
[d]Pieces per unit area.

Six reaches in three tributaries (three in Forfar, two in O'Ne-ell, and one in Gluskie) of the Middle River were selected for detailed study (Figure 1). The reaches are characterized by diverse channel morphologies including highly variable depth, width, sediment texture and woody debris loading [*Hogan et al.*, 1998]. The morphology of the channels is significantly influenced by abundant large woody debris [*Hogan et al.*, 1998]. The study reaches have a number of important distinctions between them. The higher reaches generally have a greater slope, somewhat higher stream power, and a coarser bed. For more details about the watersheds and the study reaches see the works of *Gottesfeld et al.* [2004, 2008] and *Hassan et al.* [2008].

Sediments arriving in the main stem of the study streams are recruited almost entirely from discrete sources, including episodic bank erosion, tributary input, and slumps and slides within Quaternary sediments along the tributaries and lower valley side. Sediments delivered to upland channels may be moved relatively quickly downstream or may become stored in the alluvial fans, channel bars, or floodplain along the main stems of the creeks, where they may remain for decades. Although there are spatial differences along the streams, repeated channel surveys indicate that the channel position and morphology are relatively stable [*Hogan et al.*, 1998]. Between 1992 and 1997, *Hogan et al.* [1998] reported minor net changes in bed elevation, bank, and woody debris characteristics.

3. FISH RETURN AND REDD EXCAVATION

The excavation of redds results in bed disturbance and the mobilization of sediment. All salmon spawn in gravels, although the spawning habitat, rearing areas, adult feeding areas, and age of return to their natal streams vary between species [*Groot and Margolis*, 1991; *Quinn*, 2005; *Gottesfeld*

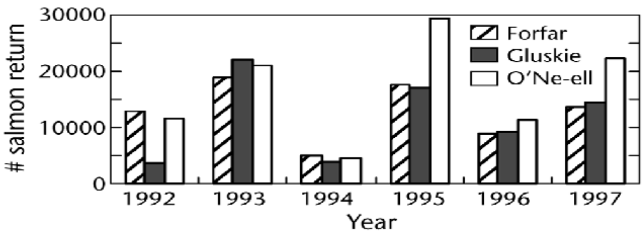

Figure 2. Adult female stock assessments in the three study watersheds.

et al., 2008]. The optimum bed sediment texture and resulting redd size is species-specific. For example, Chinook salmon, the largest species, may spawn in cobble-sized material (64–256 mm), while rainbow trout usually spawn in smaller gravels (6–64 mm) [*Gottesfeld et al.*, 2008]. In the case of sockeye salmon, the average redd dimensions and depth are 0.9 m^2 and 0.09 m, respectively [*McCart*, 1969]. *Scrivener and Macdonald* [1998] measured the depths of sockeye redds in the study streams and found depths of excavation on the order of 0.20 m. Working in the same streams, *Gottesfeld et al.* [2004, 2008] found the boundaries of a given redd difficult to delineate, but estimated the average volume of excavated material as about 0.3 m^3. Based on the depth of egg burial and scour depths, *Montgomery et al.* [1996] suggested that salmon tend to lay their eggs at a depth just below the scour level associated with bankfull discharge.

The average sockeye escapement estimates near the river's mouth between 1992 and 1997 are presented in Figure 2 (Fisheries and Oceans Canada, nuSEDS V1.0 database, available at http://www.fishwizard.com/2001, 2001). Sockeye escapement varies significantly between years; relatively low returns were recorded for 3 years out of 6 (Figure 2). Among the study streams, however, Forfar experiences the highest intensity of fish activity during the spawning period. Spawning densities are greatest within 2 to 3 km of the river mouth and tend to decline upstream because the early and stronger fish take possession of lower gradient reaches, leaving the remaining fish to vie for increasingly steep and coarse sections upstream [*Gottesfeld et al.*, 2004]. Salmon arrive in late July, and the peak of spawning is usually in the first 2 weeks of August. Spawning occurs when discharge is low, well below the threshold discharge for initiating sediment transport, making it possible to unambiguously distinguish between flood and fish-induced transport.

4. CHANNEL MORPHOLOGY

We studied changes in bed surface elevation, channel morphology, scour, and fill by floods and spawning activities in three reaches in Forfar and two reaches in O'Ne-ell (Figure 1). Bed surface changes by floods and spawning were documented over 2 years (1996 and 1997) by repeated, detailed topographic mapping. Using a total station, total reach lengths of between four and nine channel widths were surveyed at a survey data density ranging from 4 to 9 points m^{-2}. The measurements allow the calculation of net changes in the channel elevation between transporting events and therefore the comparison between the erosional effectiveness of floods and fish.

A digital elevation model example from Forfar 1050 (the values attached to each reach represent the distance upstream of the creek mouth in meters) prior to the nival flood of May 1996 is presented in Plate 1. The nival flood of May 1997 produces a typical channel morphology that one would find in these streams consisting of deep pools, riffles, and bars (Plate 1a). Plate 1b shows the net change in bed elevation following the 1997 nival flood. About 50% of the channel study area underwent net sedimentation, 40% was scoured, and roughly 10% saw no net change. Sockeye excavation/bioturbation in August resulted in major changes to the channel morphology. About 60% of the channel study area underwent net scour, 30% was aggraded, and roughly 10% saw no net change. The general trend is a net excavation of riffles and bars that are the areas most suitable for fish spawning and the filling of deep pools with sediments from the excavated bars. The several cycles of redd excavation and fill resulted in a hummocky surface with small-scale mound and hollow topography that persisted through the fall, winter, and early spring. The bed surface structure created through bioturbation remains until late spring, acting as the initial condition on which the nival flood operates. As stream flow increases in the spring snowmelt and summer floods, sediments previously deposited in pools from bioturbation are remobilized and deposited downstream, creating new bars and riffles that characterize flood morphology. In turn, these freshly deposited bars and riffles become suitable areas for the next seasons' salmon spawning. In general, the resulting bed morphology from fish excavation is antiphasic to the morphology associated with flood events. This pattern recurs annually reflecting flood magnitude and fish return values.

Using the channel surveys, we calculated the net volumes of fill, scour, and net change for all the five reaches. To compare the different reaches, we report the calculations in volume per area (m^3 m^{-2}) (Figure 3). Three values are presented in Figure 3, net fill, net scour, and net change (the difference between fill and scour). The magnitudes of fill/scour and net change for floods and fish are comparable: the median net scour and fill values for both fish and flood events are approximately 0.10 m (assuming survey errors are equivalent to the median particle size and the range of median would fall between 0.07 and 0.13 m). Thus, the depth of change (or change in elevation) is usually small, and few

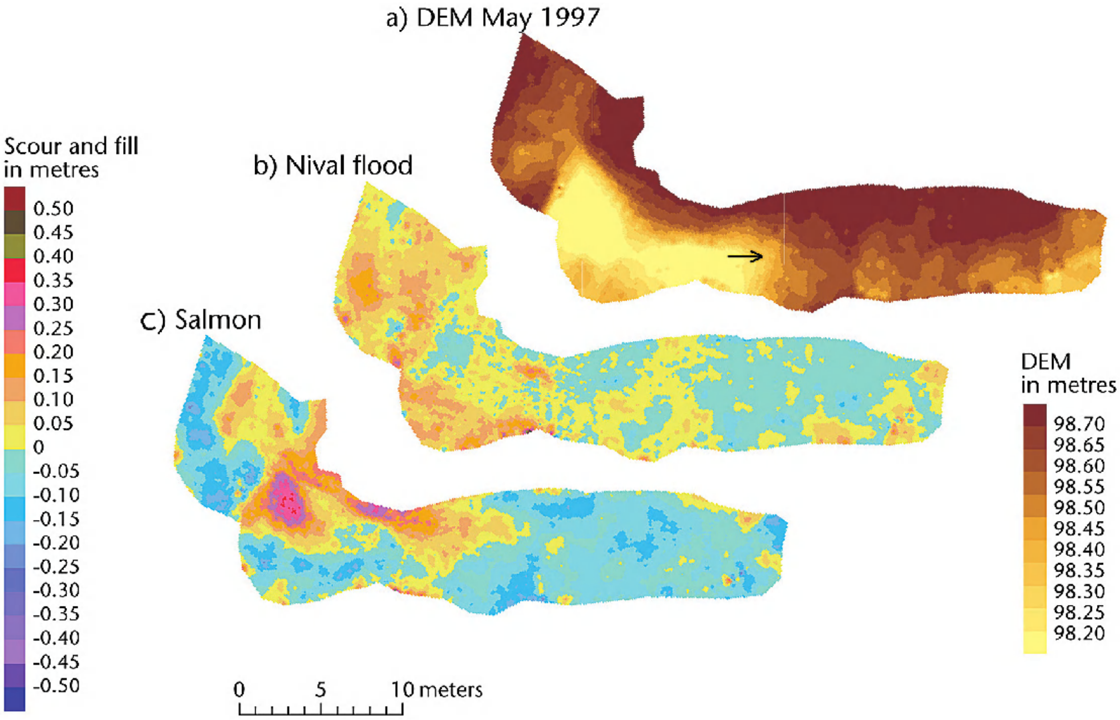

Plate 1. (a) Topographic map of Forfar 1050 subreach prior to the 1997 flood (surveyed on 19 August 1996). (b) Isopach map of net scour and fill during nival floods (surveyed on 24 May 1997). (c) Isopach map of net scour and fill during salmon transport (surveyed on 13 September 1997). Cold colors indicate decreases in elevation; warm colors indicate increases in elevation. The values attached to each reach represent the distance upstream of the creek mouth in meters. Modified from *Hassan et al.* [2008].

areas experience more than 0.30 m of net erosion or net deposition. Overall, the volume of sediment mobilized by salmon was lower in 1996 than in 1997. The net changes in sediment volume following floods ranged between 0.04 and -0.07 m^3 m^{-2}, equivalent to the median size of the bed material. Although our observations are based on only 2 years of data, all study reaches fluctuated between net scour and net fill, except for O'Ne-ell 925, which degraded in both years (Figure 3d).

Bed excavation by fish resulted in a hummocky bed morphology and comparable net scour and fill to the nival and summer floods. As mentioned before, fish activity declined upstream and, hence, so did the relative net change in bed elevation. For example, the net change due to fish activity was higher in Forfar 250 (near the river mouth) than Forfar 1050 or Forfar 1550 (Figure 3).

5. COARSE-SEDIMENT DISPERSION: TRAVEL DISTANCES AND BURIAL DEPTHS

We studied sediment dispersion, travel distances, and burial depth using magnetically tagged particles in three reaches in Forfar Creek and two reaches in O'Ne-ell Creek. Particles 40–200 mm in diameter collected from the surface of riffles were magnetically tagged, marked for identification, and replaced in lines across the channels (of the same reaches in Forfar and O'Ne-ell creeks). After a transport event, whether due to spring snowmelt, summer storm, or salmon spawning, tracer stones were recovered, and the distances they had moved and the depths to which they were buried were recorded. The recovery rate was high, ranging between 60% and 90% after snowmelt events and 90% to 100% following spawning [*Gottesfeld et al.*, 2004].

A plot of the travel distances of tagged particles mobilized by nival floods and fish between 1992 and 1996 shows that for any given year, both flood and fish events produce comparable median travel distances and ranges (Figure 4a). As distance from the stream mouth increases, spawning fish numbers decrease, and therefore, the significance of fish spawning on sediment travel distance decreases accordingly. In terms of the burial depth of the tagged particles, in some years, floods and fish have comparable results, while in other years, the fish bury particles deeper than floods: years in which the fish show a much larger effect on vertical mixing are associated with relatively small floods (Table 1 and Figure 4b). Burial depths were typically shallow; most transported clasts were recovered from within the surface layer of the gravel, 58% and 43% for floods and fish, respectively. Recovery of loose clasts on the surface was rare. A smaller portion was buried two or three clasts deep within the upper 0.15 m (measured from the particle top). Few clasts were

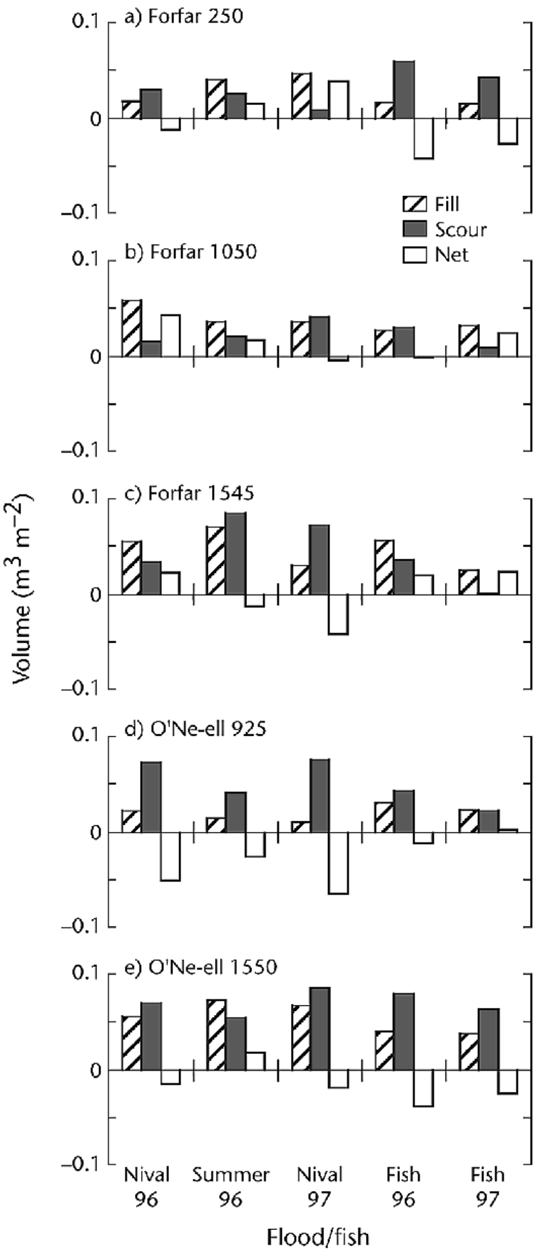

Figure 3. (a–e) Net scour/fill and net change in bed elevation by floods and fish in (a, b, and c) Forfar and (d and e) O'Ne-ell creeks study reaches.

deposited at deeper depths where they form part of the filling of scoured holes. The average burial depth ranged from 2 to $10D_{50}$ (D_{50} is the median size of the bed material). The average burial depth was approximately the same after flood transport and after sockeye salmon bioturbation. Years in which the fish show a much larger effect on vertical mixing are associated with relatively small floods (Table 1 and Figure 4b).

Scrivener and Macdonald [1998] determined the depth of the egg pockets by collection of freeze core samples. Using the same data, *Gottesfeld et al.* [2004] compared the average burial depth of the magnetic tracers with the average egg pocket depth. They reported that the depths of the egg pockets were usually greater than the mean burial depth of the flood- or sockeye-mobilized clasts. At low gradient sites where spawning salmon density was high, the mean burial depth achieved by the fish was greater than that of the floods for all seasons except 1993. This finding supports the suggestion that salmon tend to lay their eggs at a depth just below the scour level associated with bankfull discharges [see *Montgomery et al.*, 1996].

6. BED MATERIAL SEDIMENT YIELD

We studied sediment yield caused by floods and fishes using pit traps and magnetically tagged tracers. The traps were deployed in the lower reaches of Forfar, O'Ne-ell, and Gluskie creeks. The traps sampled bed load, which we assume to be equivalent to bed material transport in the gravel channels. Details of the trap construction and operation are given in the works of *Scrivener and Macdonald* [1998] and *Hassan et al.* [2008]. Measurements were conducted throughout the spring freshet, summer floods, and fish spawning periods between 1992 and 1997. The pit traps and the tagged particle data were used to estimate sediment yield. Since these are two completely different estimation methods, they are likely to yield different values [*Hassan et al.*, 2008].

For the tagged particles, the sediment yield for floods and fish was estimated using the mean travel distance, mean channel width at the study reach, and the mean burial depth. Given the little net change in bed elevation at the study reaches over the study period [*Hogan et al.*, 1998], we assumed that the mean burial depth approximated the mean scour depth. For the pit traps, we developed sediment rating curves ($Qs = aQ^b$, where Qs is the transport rate, Q is discharge, and a and b are constants) for the floods. In the case of the fish, the total accumulation over the spawning period was used. The observed transport rates are extremely low: in fact, more than half of our values fall below the reference transport rate adopted by *Parker et al.* [1982] to signify the practical threshold of motion. Depending on the

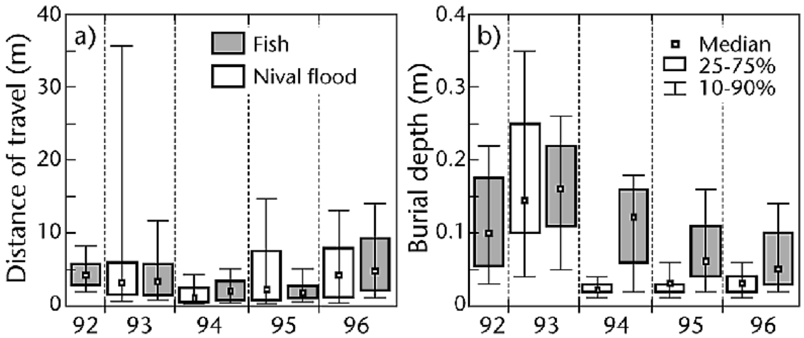

Figure 4. Forfar Creek reach 250: (a) Median travel distance (m) of tracer particles by floods and fish. (b) Median burial depth (cm) of tracer particles by floods and fish.

reach gradient, the threshold for sediment movement is about 0.1–0.3 $m^3\ s^{-1}$ (Figure 5). The pit trap sediment transport data are scattered with a clear trend of increased sediment transport with associated increases in discharge. The exponents (*b*) of the rating relations were relatively low in comparison to the typical value reported in the literature for gravel bed streams [e.g., *Whiting et al.*, 1999; *Hassan and Church*, 2001]. This could be due to trap filling during high flow, implying underrepresentation of these flows and, thus, low exponent values.

Annual sediment yield associated with floods largely depends on nival flood magnitude and duration, and the presence or absence of summer storm floods (Figure 6a). Pit trap values ranged between relatively low values during low nival floods to high values during wet and/or summer floods. Sediment yield by fish ranged widely between years, partially due to the position of redds relative to the location of traps and tracers. Years of high spawning return may produce low yield values if the fish spawn in areas distant from the traps and tracers. Conversely, years of low return may yield high values if the returning fish target areas close to the traps and tracers. Overall, pit trap results indicate fish mobilized between 10% and 60% of the annual yield. For Forfar, O'Ne-ell, and Gluskie creeks, the ratio between fish/flood mobilized sediment ranged between 0.09 and 0.6, 0.01 and 0.4, and 0.20 and 0.5, respectively. Similar trends, although of different values, of sediment mobilization by floods and fishes were estimated using the tracer data. Fish mobilize between 40% and 300% of the values attributed to floods (Figure 6b). The extremely high value of sediment yield from tracers in 1994 is likely attributed to fish spawning in close proximity to the tracer locations. On average, fish mobilize about 88% of the sediment moved by floods during the study period [*Hassan et al.*, 2008].

Plotting the fish/flood mobilization ratio versus flood return period yield similar trends (Figure 7). For return periods of <2 years, the fish/flood sediment yield ratio ranged between 0.01 and 0.6 (Figure 8). For events with return period between 1 and 3 years, fish/floods ranged from about 0.01 up to 3 indicating that fish contribution is significant, but variable, for years with low magnitude events. For floods with return period >4 years, the fish/flood ratio is close ranged between 0.01 and 0.06 indicating little contribution by the fish during years with relatively large floods. The differences in the fish/flood ratio values between the pit traps and the tracers are largely related to differences between the two bed load measuring techniques. In addition, traps estimate sediment transport moving through a cross section, while the tracers represent the areal sediment mobilization.

The amount of sediment mobilized by fish is likely to depend on the number of returning adult female salmon and the location of fish activity relative to the pit trap/tracers location. To explore the amount of sediment mobilized by fish, we plotted the sediment yield during the spawning period versus the number of returning female adult salmon (Figure 8). Relatively poor correlation was found between annual female escapement and spawning-derived transport (Figure 8). This could be due to the small sample size in addition to the location of the fish activity relative to the

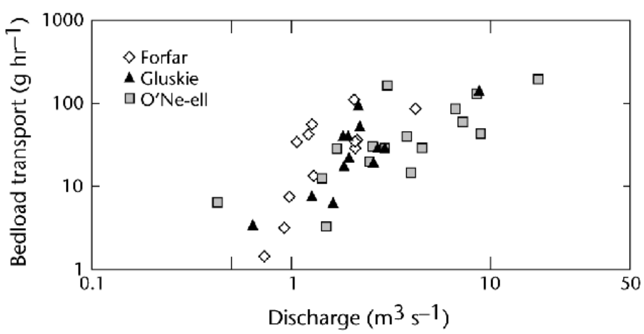

Figure 5. Total bed material rating curves for the three study creeks. Modified from the work of *Hassan et al.* [2008].

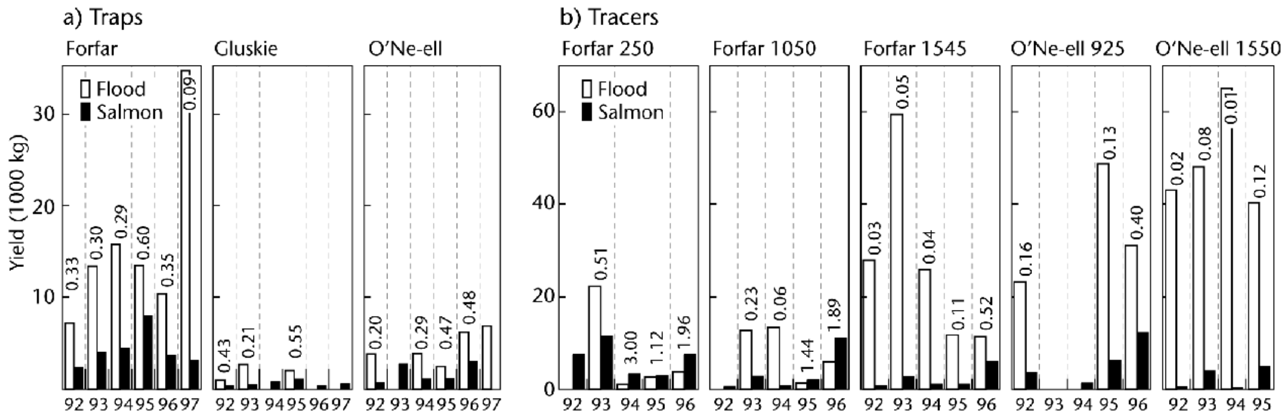

Figure 6. Sediment yield as estimated for floods and fish spawning for the years 1992–1997 using (a) pit traps and (b) tracer data. Numbers presented for each year indicate the fish/flood bed load yield ratio. Modified from the work of *Hassan et al.* [2008].

measuring devices. Furthermore, in years of high escapement, fish utilize reaches farther upstream [*Tschaplinski*, 1998], and the resulting mobilized sediment is not recorded in the traps because they are located in the lower reach of the stream.

7. BED MATERIAL TEXTURE

To assess the impact of floods and fish on changes of bed material texture, *Scrivener and Macdonald* [1998] collected freeze-core samples between 1993 and 1995. Using a 50 mm in diameter core which penetrated 0.30 m into the gravels, a 0.20–0.30 m long sample of frozen gravels was extracted from the bed. The samples were taken before and after the spawning period from areas excavated by fish and nearby areas with no spawning activity. In order to avoid problems arising from small samples (for discussion, see *Zimmermann et al.*, [2005]), individual samples were taken from the same morphology from excavated and undisturbed areas. Before combining the samples, each core was divided into surface (0–0.15 cm) and subsurface (0.15–0.30 m) material for analyses. As the core diameters were usually more than two times the diameter of the largest particles in these reaches of the creeks, the method was considered to be representative of the full gravel size range [*Scrivener and Macdonald*, 1998]. Owing to their size, freeze core samples tend to overestimate the large stones or even not sample them. Any systematic error introduced by the sampling methodology will likely affect data [*Hassan et al.*, 2008]. To avoid sampling problems, however, we limit our reporting to general trends in the data. In humid and snowmelt hydrological regimes, the surface is often armored (the bed surface is usually coarser than the subsurface), which increases bed stability and reduces bed load transport. Given the hydrological regime in the study streams, we expected to find armored bed surfaces. However, there is little difference in the size composition of the surface and subsurface after flood and fish spawning

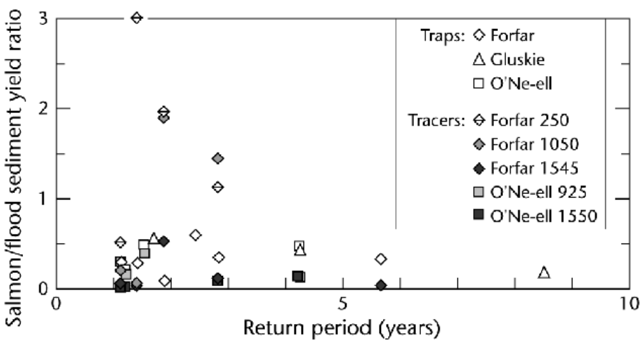

Figure 7. Relationship between the fish/flood annual yield ratio and flood return period (calculated using data from 1991–2007) for Forfar, Gluskie, and O'Ne-ell creeks. Modified from the work of *Hassan et al.* [2008].

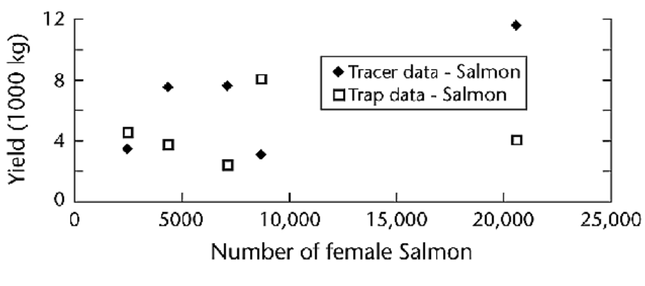

Figure 8. Relation between sediment yield and the number of returning female sockeye salmon in Forfar Creek for the years 1992–1997.

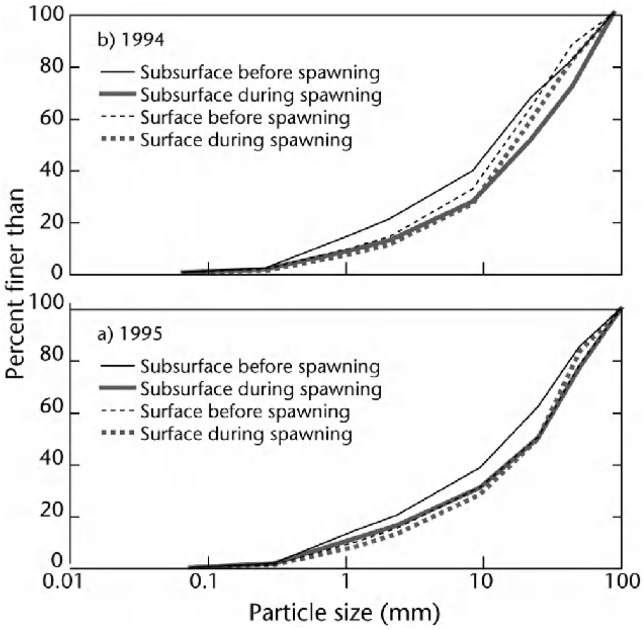

Figure 9. Examples of particle size distribution of the surface and subsurface before and during fish spawning in Forfar Creek: (a) 1995 and (b) 1994.

events (Figure 9). Hence, in addition to directly inducing bed load transport, salmon also conditions the streambed by mixing the sediment and, hence, limit or prevent the development of an armored surface layer.

8. FINE-SEDIMENT DYNAMICS

The definition of fine sediment varies among fisheries habitat studies with sands and small gravels from 0.1 up to 4 mm being identified as problematic for egg survival [*Chapman*, 1988]. These particle sizes exhibit fast settling rates and in the moderate flows of spawning season settle out of the water column quickly. While they act to cap the surface gravels or fill up the intergravel pore spaces, they are inorganic and therefore have a limited oxygen demand. Although the upper end of the grain sizes of concern identified in the fisheries literature varies, in all habitat degradation studies, the grain sizes smaller than 63 μm, silts, and clays are considered problematic due to their clogging potential. These finer inorganic grain sizes are found in gravel beds, but once resuspended during redd preparation generally are thought to be advected out of the system as they should remain in suspension due to their slow settling rates. However, silts and clays have the ability to aggregate (flocculate), which modifies their transport and settling behavior [*Droppo*, 2001]. A number of conditions that aid this process have been documented and include high conductivity and suspended sediment concentrations and/or the presence of biological breakdown products (extracellular polymeric substances (EPS)), which act like a glue binding particles together [*Liss et al.*, 1996; *Wotton*, 2007]. In freshwater systems, the influence of bacteria, and their EPS, appear to be the more dominant factor [*Rex and Petticrew*, 2008], especially in environments with low suspended sediment concentrations, as might exist in natural spawning streams.

8.1. Fine-Sediment Mobilization and Storage

Studies of fine-sediment mobilization and storage were also undertaken in O'Ne-ell Creek in the Stuart-Takla experimental watersheds. In these streams, the proportion of sediment <74 μm accounts for less than 1% by weight [*Scrivener and Macdonald*, 1998], indicating their good quality as spawning systems. Given that the amount of stored fines is low, investigations of suspended sediment concentrations during spawning periods did not consistently show an increased response to the disturbance of the bed by the digging of redds [*Cheong et al.*, 1995; *Beaudry*, 1998]. However, fish resuspension of fine sediments was documented clearly with the use of a Benthos underwater camera [*Petticrew*, 2006] and indicated that the resuspended fines were comprised of aggregated particles that settled quickly following resuspension (Figure 10). Silhouette images of particles suspended in the water column were analyzed for particle size and shape [*Petticrew*, 2006] to provide a sediment size distribution for in situ populations of aggregates, termed the effective particle size distribution (EPSD). The absolute particle size distribution (APSD) was determined on the same samples using a Coulter counter, following organic removal and mechanical disaggregation [*Petticrew*, 2006] (Figure 10). The mode and maximum size of the APSD in the immediate vicinity of fish digging are 294 and 512 μm, representing medium sands, while the EPSD of the same suspended sediment has a mode and maximum of 588 and 1024 μm. Given that the EPSD comprises a large number of particles greater than the maximum size of the constituent particles (APSD), it is clear that the physical action of digging fish resuspends aggregated fines as well as sands. Farther downstream, the particle size distribution of the fish-suspended sediment has modes of 169 μm for the EPSD and 16 μm for the APSD, reflecting a significant reduction in both effective and absolute grain size but also lower water column concentrations of sediment. All of the aggregates suspended in the water column at this distance downstream of the disturbance are smaller than 400 μm and are comprised of inorganic sediment less than 85 μm, indicating the loss, by settling, of the individual sand grains and larger (>400 μm) aggregates. This settling of sands and aggregates

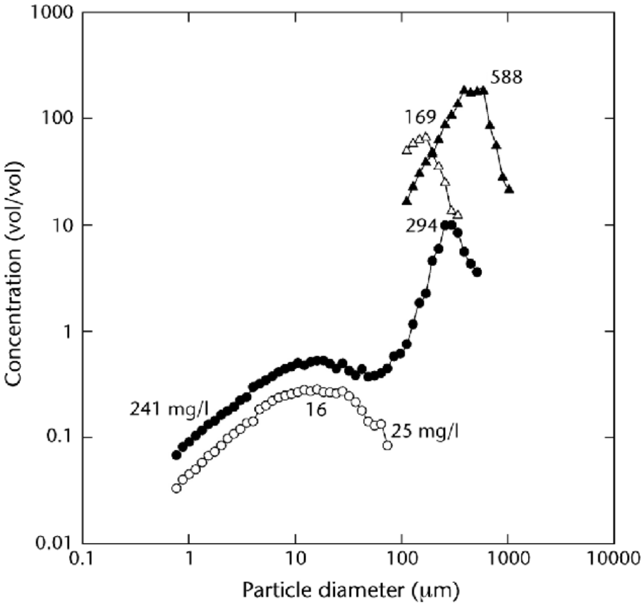

Figure 10. Particle size spectra for sediment resuspended by redd-digging fish immediately downstream (solid symbols) and 4 m downstream (open symbols) of redds. The circles represent the disaggregated inorganic component of the suspended sediment (APSD), while the triangles represent the effective particle size distribution (EPSD). The suspended sediment concentrations of the APSD samples are noted (mg l^{-1}) along with the modal size of each sediment spectra (μm). From the work of *Petticrew* [2006]. Reprinted with permission of CAB International.

over a 4 m distance indicates that fish are able to clean their immediate environment but near-field downstream impacts are generated by the resettling of sands and aggregates. Implications of this settling on incubation conditions include the physical impacts of clogging and capping but also the impacts of biochemical oxygen demand as the aggregates incorporate organic matter.

The suspended sediment in a reach approximately 1500 m upstream of the mouth of O'Ne-ell Creek was characterized for concentration, particle size, and organic matter composition during 2001 [*McConnachie and Petticrew*, 2006]. Stream discharge, precipitation, and fish returns in the reach were also monitored between early May and late September. Averaged values of middepth grab samples of suspended sediment concentrations were compared during periods of spring snowmelt and before, during, and after spawning (Figure 11a). While spring melt concentrations were the largest observed over the sample period, significant increases ($\rho < 0.05$) in suspended sediment were observed during spawning compared to the prespawn period. These ambient average values ($n = 3$) were taken over 5 days following the midpoint of the fish returns (Figure 11c) reflecting large numbers (>2000) of active spawners in or above this reach. In the following period of fish die-off, suspended sediment concentrations decreased, although a large summer storm generated a suspended sediment response in the stream.

Approximately 2 weeks prior to the 2001 salmon return, 12 infiltration bags [*Lisle and Eads*, 1991] were buried in the streambed of this same reach of O'Ne-ell Creek to collect fine sediment moving horizontally and vertically into and within the gravel bed. Infiltration bags consist of a waterproof fabric bag approximately 0.20 m in diameter and 0.35 m long attached to a steel ring (0.20 m diameter). The fabric bag was collapsed into the ring and was placed in a 0.25–0.30 m deep hole excavated in the bed that was filled with excavated gravel that had been washed and sieved to remove particles less than 2 mm in size. Duplicate bags were retrieved over a 71 day period following installation. The six retrieval dates represent: (1) the period before the fish return to the river to spawn (17 July); (2) the early spawn (28 July); (3) midspawn (3 August); (4) two dates during the major fish die-off (12 and 16 August); and (5) a sample when there was no visual evidence of live or dead carcasses in the stream, termed postfish (22 September). The material captured in the infiltration bags was wet-sieved through a 2 mm screen to obtain a mass of fine sediment collected by the gravel over the infiltration bag (estimated as mass >2 mm extracted from the bag). Figure 11b shows the cumulative amounts of fine sediment (normalized mass <63 μm) and infiltrated sediment (% <2 mm) captured by the bags and allows a comparison of these values to the hydrologic conditions, the salmon numbers, and the suspended sediment regime which reflects both the fish activity and precipitation events. Over the 71 day period, the proportion of sediment <2 mm did not increase consistently at the infiltration bag sites but rather showed maximum values during active spawning (August 3). On the subsequent two dates, the portion of <2 mm sediment decreased even though no large changes in discharge were noted. As this value is a ratio, it implies that either more gravel was added to the area above the bags or the gravels were cleaned of fines. The mechanism for either of these changes during this period would have to be the fish given that the flow regime at the time was not able to modify these components.

The normalized mass of silts and clays collected by the infiltration bags shows a similar temporal trend: low values prespawn, increases during the two active spawn samples, and decreases in the late spawn period. For these size classes, the sample collected on 22 September, 71 days after burial, had the highest and most variable mass of fines. The gravel bags retrieved on 3 August had, over a period of 21 days, accumulated ~0.02 kg of silt and clay. Removal of the bags after 71 days, which integrated the effects of spawning as

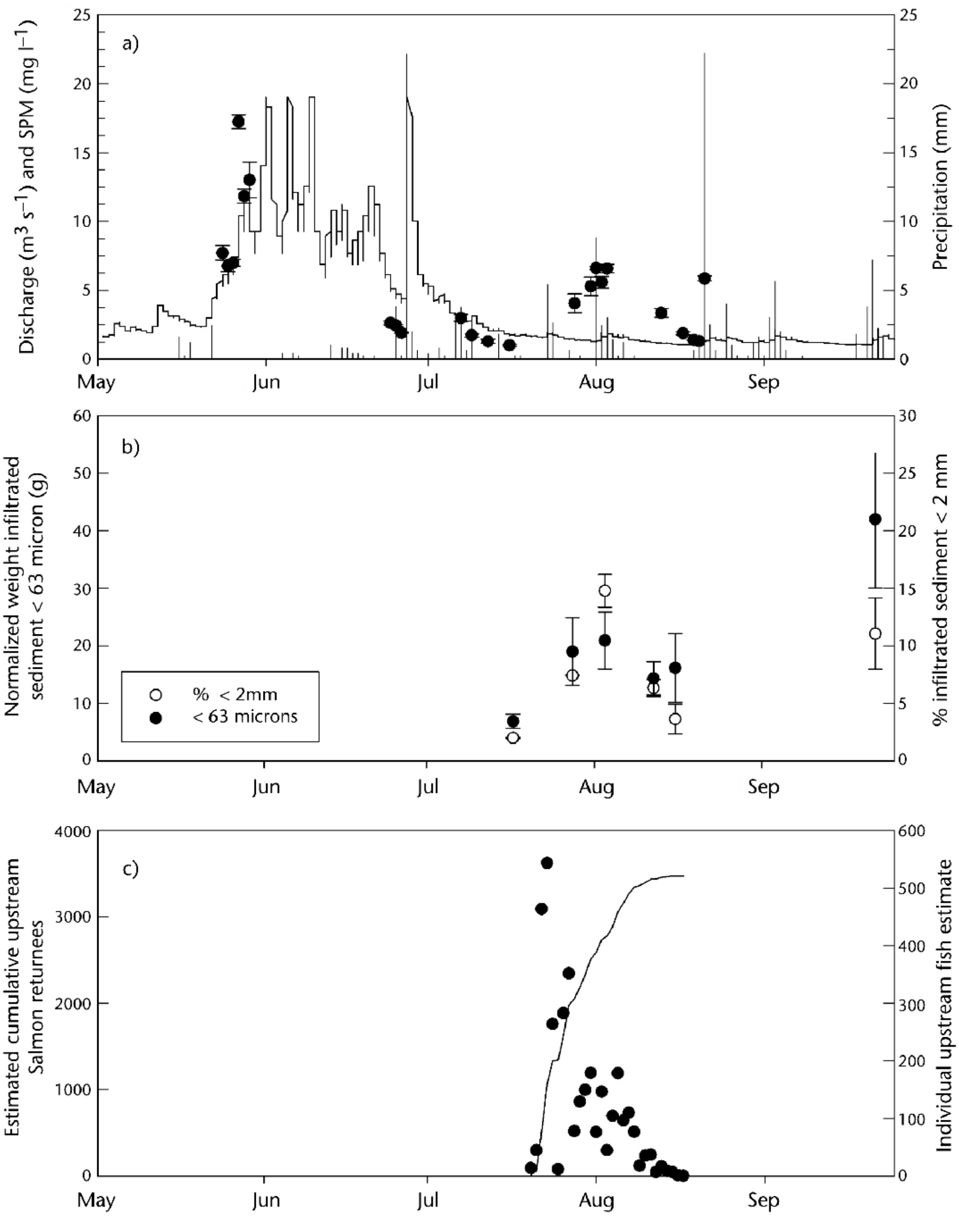

Figure 11. (a) Discharge, precipitation, and suspended sediment in O'Ne-ell Creek for periods of spring melt, prespawning, active spawn, and salmon die-off. (b) Weight of fine-sediment infiltration into gravels in an active spawning reach and the proportion of material <2 mm accumulated in the gravels before and following fish presence in the stream. (c) Estimated number of sockeye salmon entering the upstream spawning reach on a daily basis (circles) and the cumulative numbers (line) for the full 2001 spawning season. All error bars represent ±1 standard error. Modified from the work of *Petticrew* [2006]. Reprinted with permission of CAB International.

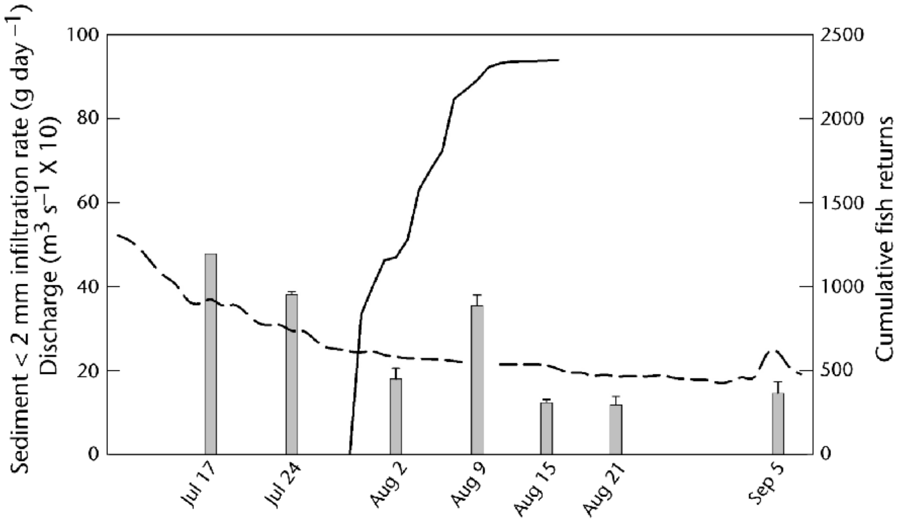

Figure 12. Infiltration rates of <2 mm sediment for the sample period in 2002. Error bars represent ±1 standard error. Cumulative counts of fish returning to O'Ne-ell Creek are shown by the solid line, while stream discharge (×10) is represented by the dashed line. From the work of *Petticrew and Rex* [2006], reproduced with permission of IAHS Press.

well as a large summer storm, indicated a doubling of this amount with on average 0.04 kg of silt- and clay-sized particles (<63 μm). Clearly, this environment is dynamic with both stream discharge and fish playing an active role in modifying the mobilization and storage of intergravel fine sediments.

On 9 July 2002, 14 infiltration bags were installed in two riffles between 1000 and 1500 m upstream of the mouth of O'Ne-ell Creek. Again, the discharge and fish returns above the reach were monitored. The bags were installed in early July with the expectation that over a 2 month period, summer low flows preceding the salmon return would allow a comparison of pre- and postspawning sediment infiltration. Figure 12 presents the accumulation of <2 mm sediment collected in the bags as a daily infiltration rate. In 2002, the discharges in early July were higher than 2001 (>4 $m^3\ s^{-1}$) due to an extended spring snowmelt, and the two prespawn infiltration rates are the highest of the data set. The infiltration rates for this size class of sediment over the sampling period tracks the discharge values exceptionally well except for the one value on 9 August, the peak of active spawning. The regression relationship between the cumulative discharge for the period of 5 days preceding bag removal (CQ_5) versus infiltration rate (IR) for the six dates (excluding 9 August) has an $r^2 = 0.97$, ($n = 6$, $p < 0.05$) indicating how strongly discharge affects the rate of sediment moving into the gravels. Using this discharge-infiltration relationship ($IR = 4.11 \times 10^{-5}\ (CQ_5) - 22.76$), the infiltration rate for 9 August is predicted to be 0.017 kg d^{-1}, whereas the observed value was 0.035 kg d^{-1}. This doubling effect indicates the influence of the active spawners on fine-sediment redistribution during the peak of spawn in 2002.

8.2. Fine-Sediment Deposition

Results of work undertaken in O'Ne-ell Creek in 2001 and 2002 indicate that fish and flows regulate the mobilization and storage periods of intergravel fine sediments in salmon-spawning streams. However, terrestrial riparian vegetation and salmon carcass decay materials represent the two largest sources of organic matter (OM) to O'Ne-ell Creek. Stable isotope analysis of suspended [*McConnachie and Petticrew*, 2006] and gravel-stored sediment [*Petticrew*, 2006] indicate that salmon-derived organic matter can make up as much as 40% of the OM in suspended sediments following the die-off, while gravel-stored sediments show an increasing salmon OM signature with increasing upstream fish counts. The increase in suspended sediment sizes and settling rates following the fish die-off, observed over a number of seasons in O'Ne-ell Creek, [*McConnachie and Petticrew*, 2006; *Petticrew*, 2006], suggested that these changes were linked to the increased retention of salmon organic matter in the gravel via settling and/or interception of these flocculated sediments by the gravel bed.

As a means of evaluating the occurrence and significance of flocculated sediments on the storage and retention of fine sediment and salmon organic matter in salmon streams, a series of experiments were conducted in outdoor flumes at the Quesnel River Research Centre (QRRC), British Columbia [*Rex and Petticrew*, 2006]. In 2004, a 30 m long, 2 m wide and 2 m deep recirculating flume was seeded with 0.4 m of cleaned gravels and set up to simulate depth and flow velocity conditions similar to that of O'Ne-ell Creek during spawning periods (0.10–0.20 m and 0.05–0.10 m s^{-1}). Infiltration bags were installed in the bed, and three bags were collected following each of the treatments to allow an assessment of the size and composition of infiltrated sediment. The experiment took place over a period of 6 days, with an initial baseline evaluation of suspended and gravel-stored sediment. This was followed by the addition of sediment from the Takla region (modal size 8 μm), which was then followed by three daily treatments of decayed salmon organic matter (SOM), mixed with Takla sediment. The concentrations of inorganic suspended sediment (<5 mg L^{-1}) and SOM (30–100 g m^{-2}) were similar to those measured in O'Ne-ell Creek in the previously mentioned studies.

Figure 13 presents Coulter counter results of inorganic, APSD for sediments collected in infiltration bags from the flumes following the three treatments, baseline, Takla sediment, and salmon-plus-Takla sediment. Coulter results for gravel-stored fine sediments indicate a coarsening of the inorganic portion of flume-bed samples as the treatments progress. Increased proportions in the silt fraction between ~5 and 40 μm were collected in the infiltration bag samples following the addition of salmon-plus-Takla sediment. The proportion of sediment greater than 10 μm captured in the gravel bed increased over the treatment periods as only 2.4% of the baseline streambed sediment exceeded 10 μm, followed by 37.3% of the sediment-only samples, and 57.5% in the salmon-plus-Takla sediment samples.

In the presence of the Takla sediments, the APSD of the receiving gravel bed shifted as silts collected on and in the bed (Figure 13). Following the Takla sediment plus salmon treatment, the gravel-stored inorganic sediment distribution coarsened even further. The coarsening of the bed between ~5 and 40 μm shows that these silt-sized particles are being sequestered from the water column by SOM and delivered to the gravels. These data indicate that flocs or aggregated particles formed in the water column increased the delivery

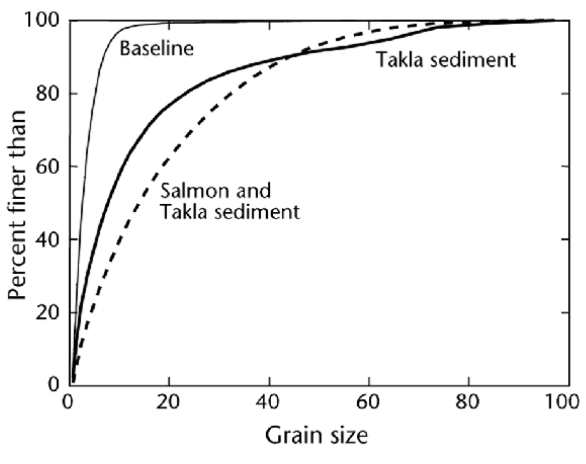

Figure 13. Mean percent composition (n = 3) for inorganic fine-grained sediment (APSD) captured in infiltration bags in the recirculating flume.

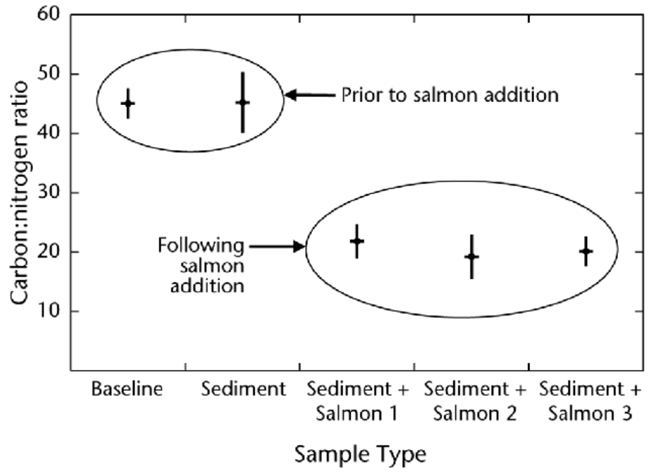

Figure 14. Variability in the carbon:nitrogen ratio for background, sediment only, and the combined salmon-plus-sediment treatments (n = 3). From *Petticrew and Rex* [2006], reproduced by permission of IAHS Press.

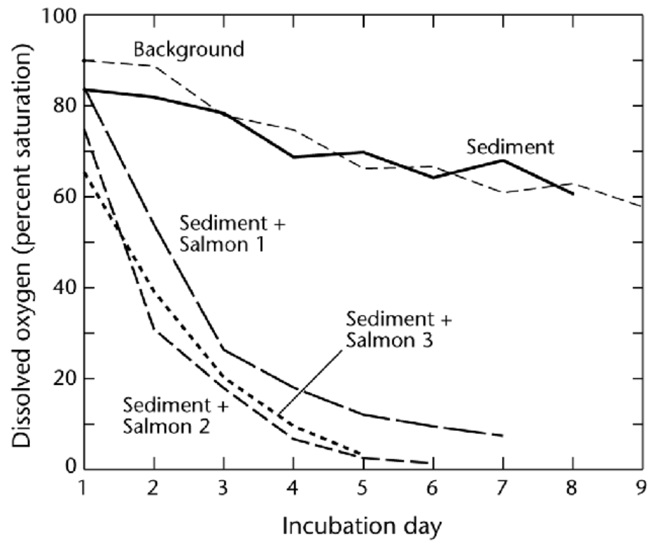

Figure 15. Temperature-corrected biochemical oxygen demand curves for background, sediment only, and the three salmon-plus-sediment treatments (n = 3). From the work of *Petticrew and Rex* [2006], reproduced with permission of IAHS Press.

of larger proportions of silt fractions to the gravel bed of the flume.

9. NUTRIENT RETENTION

As a means of characterizing the effect of inorganic and organic fine-grained additions to the gravel bed, two methods were used in the flume study at the QRRC in 2004. First, carbon and nitrogen ratios of the gravel-stored fine sediment collected from the infiltration bags were determined using the Dumas method of total combustion. Second, the biochemical oxygen demand (BOD) of the gravel-stored sediments was determined by measuring oxygen concentrations on a daily basis for a period of 7 to 9 days or until dissolved oxygen levels had reached near zero [*Clesceri et al.*, 1998]. BOD samples were incubated in the flume and then a laboratory water bath of the same temperature as the flume (~10°C) [*Rex and Petticrew*, 2006].

The gravel-stored sediments collected after salmon-plus-sediment treatments had a significantly lower carbon to nitrogen ratio than the background and sediment-only infiltration bag samples ($\rho = <0.05$, Figure 14). Lower values of C:N in the gravel-stored sediment in the flume following the addition of SOM indicates retention and enrichment of the sediment with SOM as fish decay products are the only source of N added to the system. Similarly, the BOD samples exhibited a considerably different trend between the background, sediment, and salmon-plus-sediment samples. The initial streambed samples (background) and sediment-only BOD samples decreased to 60% oxygen saturation within 9 days (Figure 15). The three salmon-plus-sediment samples had lower starting oxygen concentrations than the background and sediment samples and decreased to near zero within 7 days. The increase in BOD is a result of the mineralization by benthic bacteria of the additional SOM delivered to the gravel bed.

Both the C:N ratio and BOD indicate that SOM is being delivered to the gravel bed and modifying the quality of the habitat. Follow-up work in the QRRC flumes verified a model for a flocculation feedback loop [*Rex and Petticrew*, 2008] in which the abundance of in-stream SOM during carcass decay enhanced flocculation with available inorganic sediments such that delivery of SOM to the gravels was increased. This results in the in-stream retention of fish-derived nutrients, which helps to maintain the productivity of natal streams. This reflects another means by which salmon behavior (spawning and die-off in natal streams) acts to regulate the future success of their offspring. While it is obvious that some retention of SOM will be of value to the longer-term productivity of the stream, the level that the stream can handle while still allowing suitable conditions for incubation has not yet been determined.

10. IMPLICATIONS

Our analysis shows that fish modify channel morphology, change bed surface structure, and mobilize both gravels and fine sediment in our study streams. While abiotic forces shape habitat structure to a large degree, organisms that act as "ecosystem engineers" are pervasive and may exert strong feedback by mediating physical processes [*Jones et al.*, 1994]. It has previously been argued that salmon significantly modify their own habitat [e.g., *Kondolf et al.*, 1993; *Kondolf and Wolman*, 1993; *Montgomery et al.*, 1996]. In this way, salmon can be considered as both geomorphic agents and ecosystem engineers [*Hassan et al.*, 2008]. Salmon redd excavation creates a distinct hummocky channel morphology superimposed on the longer-wavelength pool-riffle bed forms. *Montgomery et al.* [1996] argue that redd construction increases particle size and sorting and changes local morphology, increasing form drag. From theoretical calculations, they showed that these changes act to decrease the mobility of the bed, thereby decreasing the potential for redd scour. In areas of high spawner density, the combined effect may improve gravel quality for spawning over time. However, in this study, we observed that fish activity, through redd construction, prevents the formation of an armor layer, loosens the gravel framework, breaks up surface structures and particle imbrication, and hence decreases bed stability and increases sediment mobility. The outcome of fish activity, therefore, largely depends on the balance between the increase in surface roughness, which is likely to increase bed stability, versus the decrease in bed surface armoring, which likely increases sediment mobility. In other words, the balance between these opposing factors is likely to determine the relative impact of fish on channel morphology and sediment transport.

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Restoring Habitat Hydraulics With Constructed Riffles

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Riffles and rapids may be added to channels for a variety of purposes, including increasing hydraulic complexity, stabilizing mobile bed streams, increasing aquatic habitat, or restoring fish passage. To increase hydraulic complexity, there are several options for introducing locally varied hydraulic conditions through the creation of riffles, rapids, runs, and pools. This involves increasing the frequency of transitions between several conditions of uniform, gradually varied and rapidly varied flow. To improve fish passage, riffles and rapids are normally designed as fish-passable hydraulic structures, often replacing traditional fixed drop structures or low dams in channelized streams. The provision of more diverse hydraulics and fish access may be a project objective, but the intricacies of specific aquatic habitat types are beyond commonly used one-dimensional open channel hydraulic equations. Consequently, reliance is placed on mimicking the hydraulics of preferred habitats surveyed in natural reference streams. The hydraulics observed in several preferred aquatic habitat types are broadly summarized, and a design method for riffles, runs, and pools with six project examples is presented.

1. HYDRAULICS AND HABITATS

Rapidly varying flows occur in streams at man-made structures, variations in bed topography, large roughness elements, or changes in channel shape. The local velocity and trajectory of flow is shaped by the flow boundaries and often by discrete cells of water moving within one another. Contrary to the uniform flow assumptions, a surprising amount of water may flow both in a cross-stream and upstream direction, such as when fast water enters a slowly moving pool (Figure 1). Energy losses result primarily from turbulent contraction, expansion, and deceleration. These conditions occur over such a short reach that frictional losses on the boundaries are generally ignored for design purposes.

Rapidly varied flow conditions are the focus of this chapter. For a full treatment of open channel hydraulics and hydraulic structures, readers may refer to several historic and recent sources: introductory concepts [*Brater and King*, 1976; *Kay*, 1998], open channel hydraulics [*Chow*, 1959;

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Figure 1. In this short reach at the head of a pool, two thirds of the flow directions are upstream or across the channel in two back eddies on either side of the channel and in the "horseshoe vortex" formed behind the boulder. Aeration occurs as the rapid flow penetrates the slower moving pool water, carrying air under the surface that emerges as noisy bubble-popping white water. This is the only source of flowing water sound and a sure indicator of rapidly varying flow.

Chanson, 1999], fluvial processes [*Leliavsky*, 1966; *Chang*, 1988], and habitat hydraulics [*Vogel*, 1981; *Allan*, 1995].

The state of the flow is defined by how the potential and kinetic energy components are partitioned [see *Chow*, 1959, chapter 1]. The potential and kinetic energy components are expressed in an equation named for Daniel Bernoulli (1700–1782) in recognition of his founding hydraulic concepts by Euler (1707–1783) [*Rouse and Ince*, 1957] where

$$E = Z + D + \alpha V^2/2g \quad \text{(Bernoulli's equation)}. \qquad (1)$$

Applied to stream channels, E is the sum of all of the energy of the flow at that point in the channel, Z is the elevation of the channel bed above a determined datum (m), D is depth of flow (m), V is the mean velocity (m s^{-1}), g is the acceleration due to gravity (9.8 m s^{-2}), and α is an adjustment factor for cross-section changes in the short reach (assumed to be 1.0 in this discussion). The term $V^2/2g$ describes the kinetic energy of the flow (m). The specific energy of the flow H (m) is the sum of the water depth and kinetic energy at a point in the channel where

$$H = D + V^2/2g. \qquad (2)$$

There are only three states of flow based on velocity and depth combinations: (1) subcritical, the depth is two times or more greater than the kinetic energy, typically in deeper pools and mildly sloping channels; (2) critical, the depth is two times the kinetic energy as the water flows over riffle crests, protruding boulders, or in steeply sloping streams; and (3) supercritical, the depth of flow is less than two times the kinetic energy in waterfalls and short vertical drops below boulders and ledges for example.

The Froude number Fr, where $Fr = V/(gD)^{1/2}$ is equal to 1 for critical flow, less than 1 for subcritical flow, and greater than 1 for supercritical flow (Figure 2).

When $Fr = 1$, the critical velocity is

$$V_c = (gD_c)^{1/2}. \qquad (3)$$

The critical depth D_c and critical specific energy H_c may be expressed in terms of the discharge Q ($m^3\ s^{-1}$) and width of the flow W (m):

$$D_c = (Q^2/gW^2)^{1/3} \qquad (4)$$

$$H_c = 1.5(Q^2/gW^2)^{1/3}. \qquad (5)$$

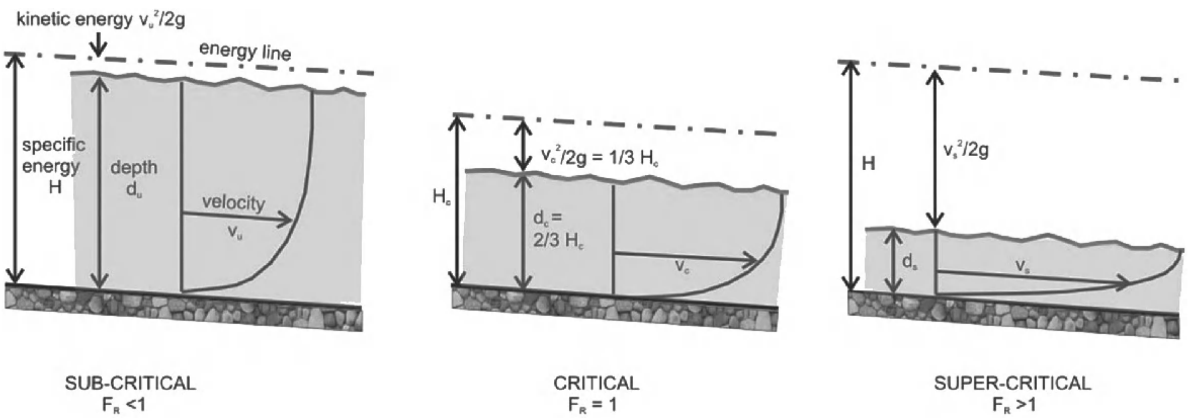

Figure 2. Three possible states of flow based on velocity and depth combinations for rectangular channels and associated Froude numbers.

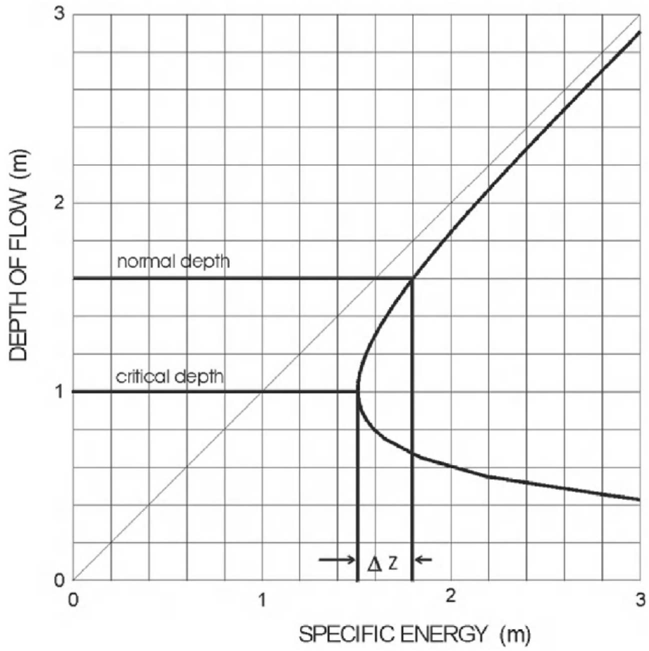

Figure 3. Combinations of depth and velocity that can occur for a given discharge and width are shown in this specific energy diagram. The minimum specific energy occurs at the critical depth. If the resistance-governed normal depth, in this case for a uniform channel with no backwater effects, is greater than the critical depth, there is room for the manipulation of the local channel cross section without changing the flood stage [*Chanson*, 1994]. In this case, the bed may be raised by the height ΔZ. In some cases, for example, in incised streams, riffle crests may be set much higher to allow normal flood discharges to enter the floodplain.

All combinations of depth and velocity for a given discharge in a channel cross section can be expressed as a specific energy curve where the depth of flow D is plotted against the specific energy H (Figure 3). The point of minimum specific energy required for a given width and discharge occurs at the critical depth (D_c), where the curve reverses slope between the subcritical zone and supercritical zone, in this case at a depth of 1.0 m.

2. LOCAL HYDRAULICS AND HABITAT PREFERENCES

Flows described by the Froude number have been mapped and linked to three traditional stream habitat classifications: riffles, runs, and pools [*Jowett*, 1993]. Species-specific habitat preferences based on observed combinations of velocity, depth, and substrate (termed habitat suitability indices) are compiled for a wide variety of fish [*Bovee*, 1982; *Keeley and Slaney*, 1996]. Four examples of observed habitats and subsequent projects designed to mimic the site-specific channel geometry and hydraulics follow.

2.1. Fish Habitat Preferences: Jumping Pound Creek, Alberta, Canada

The abundance of fish species was observed to increase with the drainage area and stream width in the upper reaches of Jumping Pound Creek, Alberta, Canada (Figure 4) [*Glozier*, 1989].

The fish are not distributed uniformly in the sample reaches. The velocity and depth preferences for species in

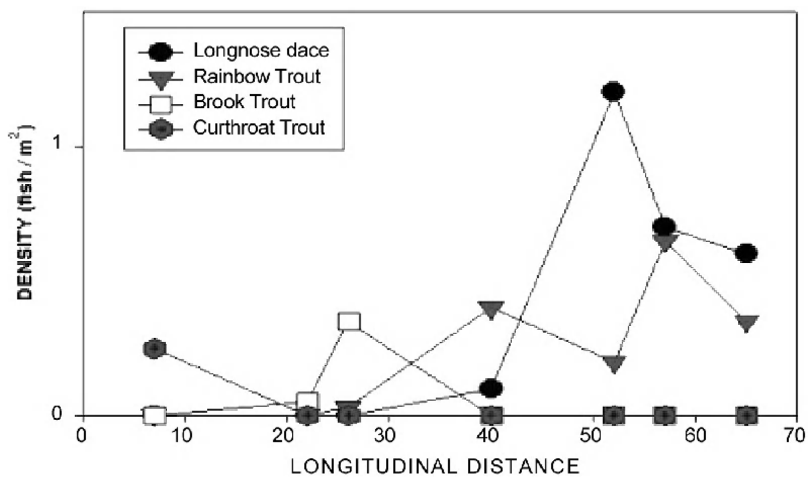

Figure 4. Increasing number of fish species from the headwaters to near the mouth of Jumping Pound Creek, Alberta, measured in kilometers [*Glozier*, 1989].

Figure 5. Lower sample reach of Jumping Pound Creek, Alberta.

Table 1. Froude Number at Observed Fish Locations in the Lower Reach of Jumping Pound Creek, Alberta, Canada

Fish Species	Life Stage	Froude Number
Longnose Dace	fry	0.026
	adult	0.055
River Shiner	fry	0.027
	adult	0.038
Spottail Shiner	fry	0.026
	adult	0.037
Brook Stickleback	juvenile	0.074
White Sucker	juvenile	0.036
Rainbow Trout	juvenile	0.276

the lower reach (Figure 5) are shown in Figure 6 and Table 1 [*Swanson and Ray*, 2003].

The Froude number of habitats increases as the smaller species mature from fry to adult. Larger juvenile rainbow trout (47 times fry lengths) were found in higher velocity water with Froude numbers that were 10 times greater than those in the smaller fish locations. Similar observations have been made for cutthroat trout in Chapman Creek, British Columbia, Canada (see example 4 of *Bates* [2000]).

Stream restoration projects for a range of species and life stages can be designed to create a similar diversity of hydraulic conditions by altering the reach hydraulics with pool, riffle, and run profiles. For example, the dimensions of natural meanders with proven adult trout habitats were mimicked in recreating meanders in the North Pine River (Figure 7) [*Newbury and Gaboury*, 1993]. The project plan and profile are shown in Figure 8. Similar profiles showing equal drops in runs and riffles have been observed in meandering streams [*Leopold et al.*, 1964].

2.2. Benthic Habitats in Critical Flow

The life cycle for many benthic insects requires a period of attachment or occupation on the streambed in an optimum feeding location for grazing surface algae or capturing organic drift [*Allan*, 1995]. Algae growth occurs where there is maximum light penetration in zones of minimum depth. This occurs in shallow runs and on the surface of large cobbles

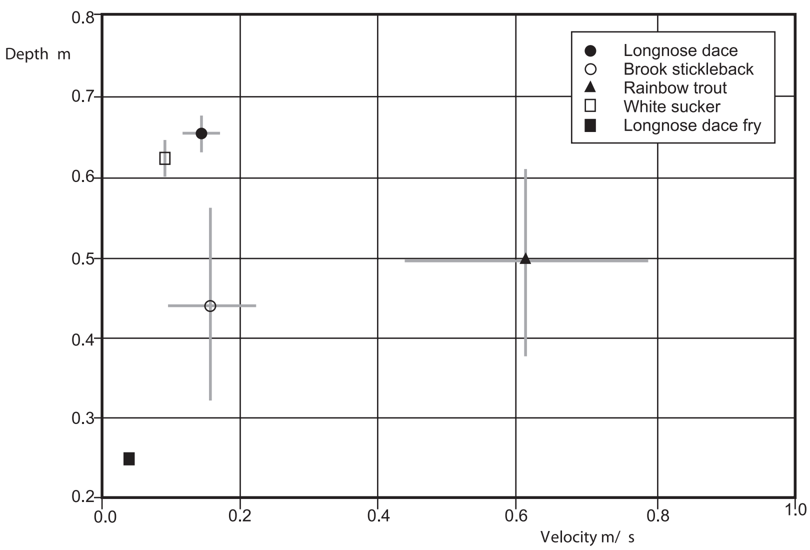

Figure 6. Depth and velocity measurements at fish locations observed in the lower reach of Jumping Pound Creek, Alberta.

Figure 7. Riffle, runs, and meanders constructed though a straightened highway crossing on the North Pine River, Manitoba, in 1990. The reconstructed meanders mimic natural adult trout habitats found elsewhere in the river. Photo by K. Kansas, Manitoba Fisheries.

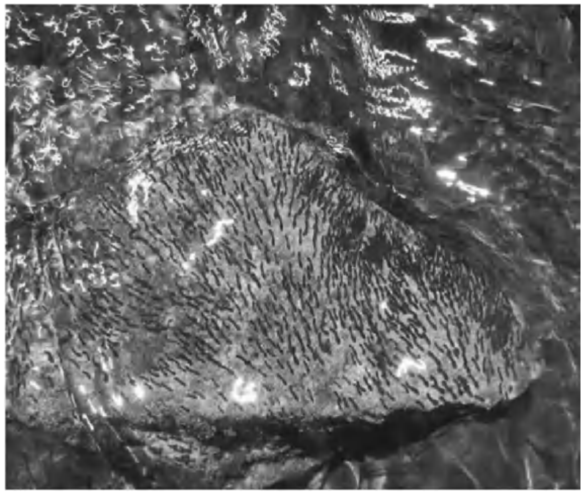

Figure 9. Blackfly larvae in a critical flow zone on the surface of a large cobble, Battle Creek, Saskatchewan.

and boulders in riffles and rapids. Insects in these high-velocity habitats have developed streamlined bodies with low drag coefficients and employ a variety of anchoring strategies, such as hooks, claws, glues, and the attachment of heavier materials to their bodies or external cases. Blackfly larvae holding on to the rock surface with a salivary glue and hooks and Caddisfly larvae in cases made of fine bed materials glued to the rock surface are shown in Figures 9 and 10. Riffles are an important source of food production for fish because of the relatively high macroinvertebrate production relative to other parts of the channel [*Allan*, 1995].

Stream restoration projects designed to sustain resident fish populations must create local hydraulic conditions in riffles and rapids that are elastic in the sense that the target conditions will persist over a range of discharges as the stream stage changes. This can be accomplished by arranging cobbles and boulders of different sizes and elevations of the surface of riffles and rapids that maintain similar near-emergent conditions over a wide range of depths. Resilient designs that

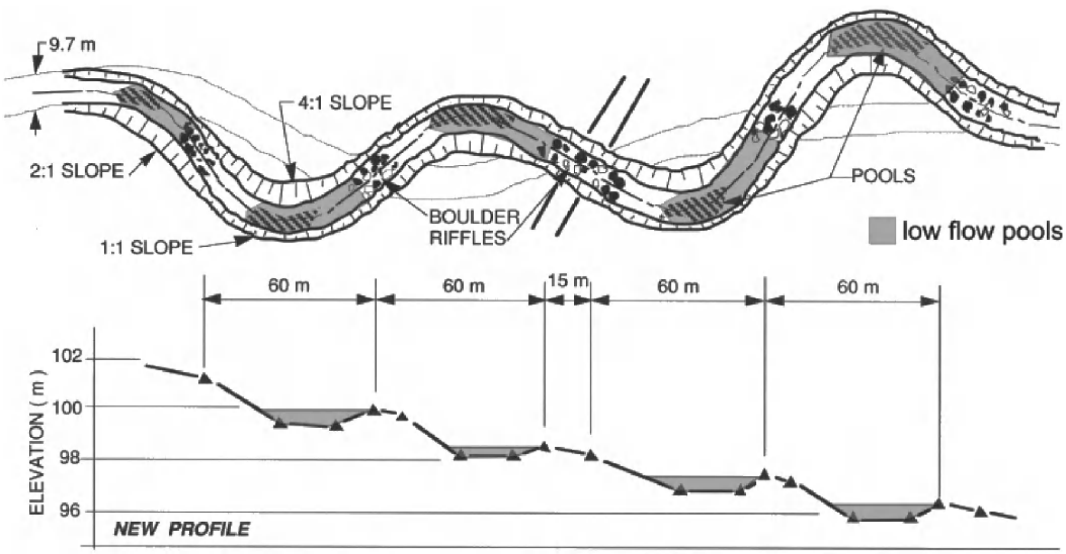

Figure 8. Plan and profile of the North Pine River, Manitoba, trout pool project. The still water levels extend halfway around the meander bends observed in other meandering reaches. The spacing and radius of curvature surveyed in nearby unaltered trout pools are the average observed in meandering alluvial rivers [*Chang*, 1988].

Figure 10. Caddisfly larvae cases in a back eddy zone below critical flow over a large cobble, Jumping Pound Creek, Alberta.

address a range of flow conditions while still meeting project objectives are also more likely to persist into the future under changing hydrology due to climate and land use change. For example, in 1991, emergent boulders and cobbles were added to a uniform bedrock reach of the Whiteshell River MB to create pools and insect habitats in a reach regularly stocked with rainbow trout from an adjacent hatchery (Figure 11). The reach was developed for the Manitoba Fly Fishers Association as a training site for student fishers [*Newbury*, 2010]. Other than a loss of cover as woody debris decayed, the created habitats have persisted for 17 years (Figure 12) [*AAE Tech Services*, 2008].

Figure 12. Re-surveys of the Whiteshell River project reach in 2008 found an abundant trout population and no change in the project reach configuration [*AAE Tech Services*, 2008]. Woody debris cover was lost through decay. Photo by M. Lowdon.

2.3. Spawning Habitats on Gravel Bars and Riffles

Salmonids and other fish species spawn by excavating shallow covered egg beds or redds [*Long*, 2007]. To be successful, the redd locations must meet several conditions:

1. Substrate must be clean and coarse enough to allow interstitial flow to carry dissolved oxygen to the eggs during the incubation period. This occurs typically on gravel bars, runs, and approaches to riffles.

2. Bed materials must be easily disturbed at spawning period discharges to allow the fish to excavate the bed and

Figure 11. Rapids and boulders added to a uniform bedrock reach of the Whiteshell River, Manitoba, to create benthic and trout habitats at a fly fishers training site in 1991 [*Newbury and Gaboury*, 1994].

Figure 13. Typical spawning sites on the main stem and a side channel entrance in the natural reach located upstream from the restoration project the Okanagan River near Oliver, British Columbia.

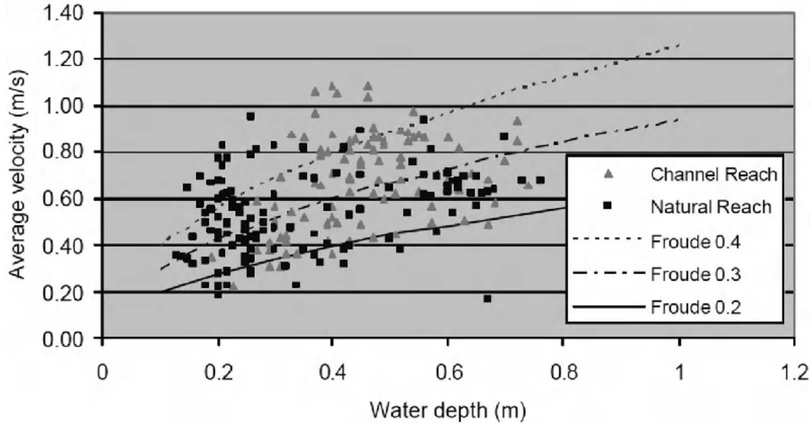

Figure 14. Range of Froude numbers based on observed pairs of velocities and depths at Sockeye spawning redds in a natural and channelized reach of the Okanagan River, British Columbia [*Long*, 2007].

then cover the eggs by displacing materials from upstream with a minimum of effort. This implies that the velocity and depth at the site exert a shear stress that approaches that required for bed load transport.

3. The velocity at the site must be low enough to allow sustained swimming while the redds are being built and fertilized.

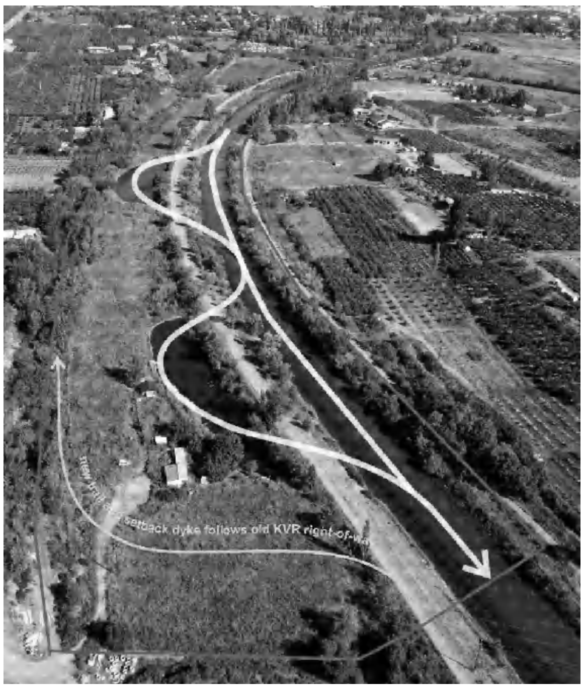

Figure 15. Okanagan River re-meandering and spawning bar project plan near Oliver, British Columbia. Photo by Wild Earth Photography, Kelowna, British Columbia.

In a study of Sockeye salmon in the Okanagan River, British Columbia (Figure 13), paired values of velocity and depth at typical redd sites expressed as a Froude number characterized the preferred sites more strongly than velocity or depth alone [*Long*, 2007]. Spawning occurred in both natural and channelized reaches where the Froude numbers range between 0 (back eddies) and 1.0 (riffle crests) (Figure 14). Egg survival was more successful in the higher velocity, shallower depth sites in the natural reach where midwinter sedimentation was low. A restoration project in a 1 km long channelized reach based on the observed hydraulics in the natural reach was undertaken in 1990.

To restore natural spawning conditions in the channelized reach, abandoned meanders were rejoined with entrance and exit riffles and gravel bed bars added to the main stem that were designed with the Froude number range observed at

Figure 16. Entrance to upper meander bend showing spawning ramp in main stem (Fr = 0.35 at the regulated spawning discharge of 8 m^3 s^{-1}) and riffle habitat in the Lougheed meander entrance completed in August 2009.

Figure 17. Sockeye spawning redds (lighter gravel circles) on the Lougheed spawning bar, September 2009. Photo by Kevin Dunn, Kelowna, British Columbia.

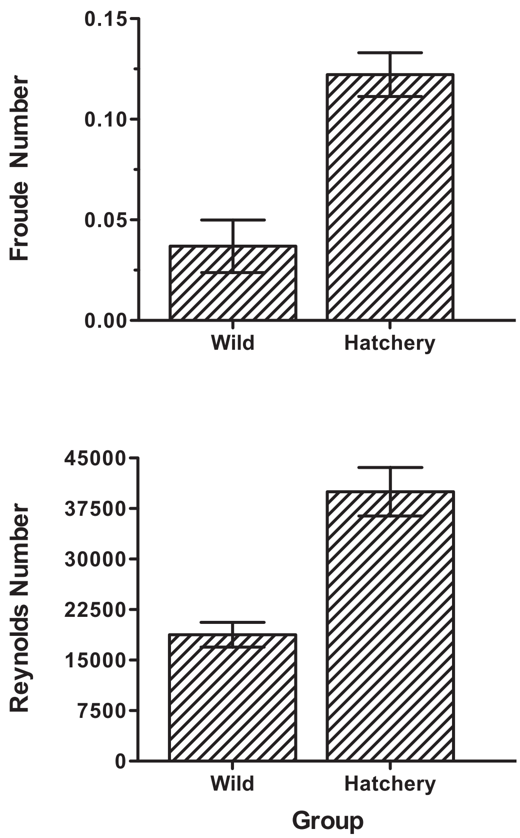

Figure 18. Froude and Reynolds number preferences in habitats selected by wild and hatchery-raised 1 year old cutthroat trout [*Bates*, 2000]. The higher Froude number areas occur in a deep central channel that is scoured by floods. The lower Froude number areas occur in shallow back eddies on the margins of the stream.

Figure 19. An elevated shallow pool in a hatchery release reach of Chapman Creek, British Columbia, formed by back flooding from a downstream rock riffle.

high-velocity redd sites in the natural reach. A dyke was set back to access the original floodplain and meanders immediately downstream of the natural channel in 2008 (Figure 15).

The reconnected channel, entrance riffle, and main stem spawning bar constructed in August 2009 are shown in Figure 16. Sockeye spawning redds on the main stem bar in October 2009 are shown in Figure 17. The geometry and hydraulic conditions are analogous to the natural reach shown in Figure 13.

2.4. Hatchery-Raised Fish Release Habitats

Bates [2000] mapped the hydraulic conditions of selected locations used by juvenile coastal cutthroat trout released into Chapman Creek, a small coastal British Columbia

Figure 20. A rock riffle was added to Chapman Creek to retain spawning gravels and maintain water depths. The bankfull depth at the site is 1.6 m. The height of the riffle crest is 0.28 m.

Table 2. Summary of Chapman Creek Riffle Calculations[a]

	Definition	Function	Value
Riffle Height			
Q	design discharge (m^3 s^{-1})		68
D	depth of flow approaching riffle (m)		1.6
W	average width of flow (m)		21
V	approaching velocity (m s^{-1})	Q/WD	2.02
$V^2/2g$	velocity head (kinetic energy of approaching flow) (m)		0.21
H	specific energy (m)	$D + V^2/2g$	1.81
D_c	critical depth of flow (m)	$D_c = (Q^2/gW^2)^{1/3}$	1.02
$V_c^2/2g$	critical velocity head (m)	$D_c/2$	0.51
H_c	critical specific energy (m)	$H_c = D_c + v_c^2/2g$	1.53
R_H	riffle height above channel bed (m)	$H - H_c$	0.28
Dimensions			
S_B	channel slope		0.03
S_{RU}	slope of upstream riffle face		0.5 (2:1)
S_{RD}	slope of downstream riffle face		0.1 (10:1)
R_U	distance of heel to crest m	$R_U = R_H/(S_{RU} + S_B)$	0.53
R_D	distance of crest to toe m	$R_D = R_H/(S_{RD} - S_B)$	4.0
Y_D	height of bed at the crest above toe m	$Y_D = R_D\ (S_B)$	0.12
	total drop in chute m	$Y_D + R_H$	0.4

[a]These may be easily summarized in a riffle design spreadsheet. The dimensions are shown in Figure 22.

stream (Figure 1). The juveniles sampled included fish of "wild" populations and hatchery-reared trout. The hatchery fish, raised in tanks with set flows and circulation patterns, dispersed into central areas of the stream cross section seldom used by wild trout. The areas overlapped with those used by larger aquatic predators and, more importantly, were subject to the main torrent of flood flows with velocities greater than sustained juvenile swimming speeds. Consequently, the initial density of released fish in the central parts of the channel decreased dramatically when the first post-release freshet surged down the channel into the estuary. Densities observed in back eddy and shallows on the edge of the channel that were preferred by wild fish remained the same.

The preferred flow regions were characterized using two hydraulic descriptors, the Froude number Fr and the Reynolds number Re (Figure 18). The Reynolds number $Re = V(D)/\eta$, where η is the kinematic viscosity.

The hydraulic preferences suggested that stream reaches used for hatchery release may be restored by providing shallow side channels and varied cross sections that will allow a range of lower Froude number locations to persist as refugia during high flow events. Riffles were added to the hatchery release reach in 1990 to allow access to the floodplain and shallow side channels with low Froude number conditions that could be accessed over a range of flood flows (Figure 19). A typical Chapman Creek riffle design is presented in the following section.

3. RIFFLE DESIGN

Lack of hydraulic complexity typifies many channelized and uniformly graded streams. As such, restoration of these channelized streams may be accomplished through the creation of locally varied flow conditions by adding riffles or constricting the channel. For example, a rock riffle was constructed in Chapman Creek, British Columbia, to create a shallow pool for hatchery releases, retain spawning gravels, and reduce bank erosion in the channelized reach.

Under the channelized uniform flow conditions, the median annual flood discharge 68 m^3 s^{-1} just filled the channel to

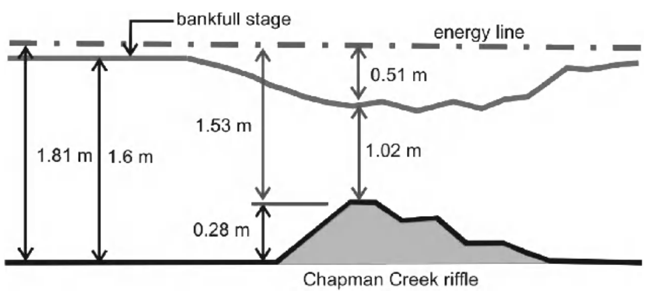

Figure 21. Schematic profile of the change in state of subcritical to critical flow in an open channel as it passes over a riffle crest in Chapman Creek without increasing the flood stage.

the floodplain/bankfull stage. The average cross section in the reach was 1.6 m deep and 21 m wide (Figure 20).

The initial design problem was to determine desired habitat features, which in this case were a shallow pool and bed materials that would support the life history of spawning and rearing salmon. The second design challenge was to determine maximum pool depth upstream of the riffle that could provide access to shallow side channel refugia while maintaining the same flood stages. To meet this condition, the height of the riffle had to be less than or equal to the difference between the specific energy of the approaching flow in the pool at the same preproject flood stage and the minimum specific energy required to pass the same discharge at the critical depth. With these criteria, preflood stages and frequencies would not change for a nearby hatchery and trailer park located on the floodplain.

At the bankfull stage in the pool above the riffle and at a discharge of 68 $m^3\ s^{-1}$, the average velocity from continuity is 2.02 $m\ s^{-1}$ with a specific energy of 1.81 m (equation (2)). The critical depth at 68 $m^3\ s^{-1}$ is 1.02 m (equation (4)), the critical velocity is 3.16 $m\ s^{-1}$ (equation (3)), and the critical specific energy is 1.53 m (equation (2)). Based on these values, the maximum height of the riffle is limited to $\Delta Z = H - H_c = 0.28$ m in order to pass the same flood flow at the

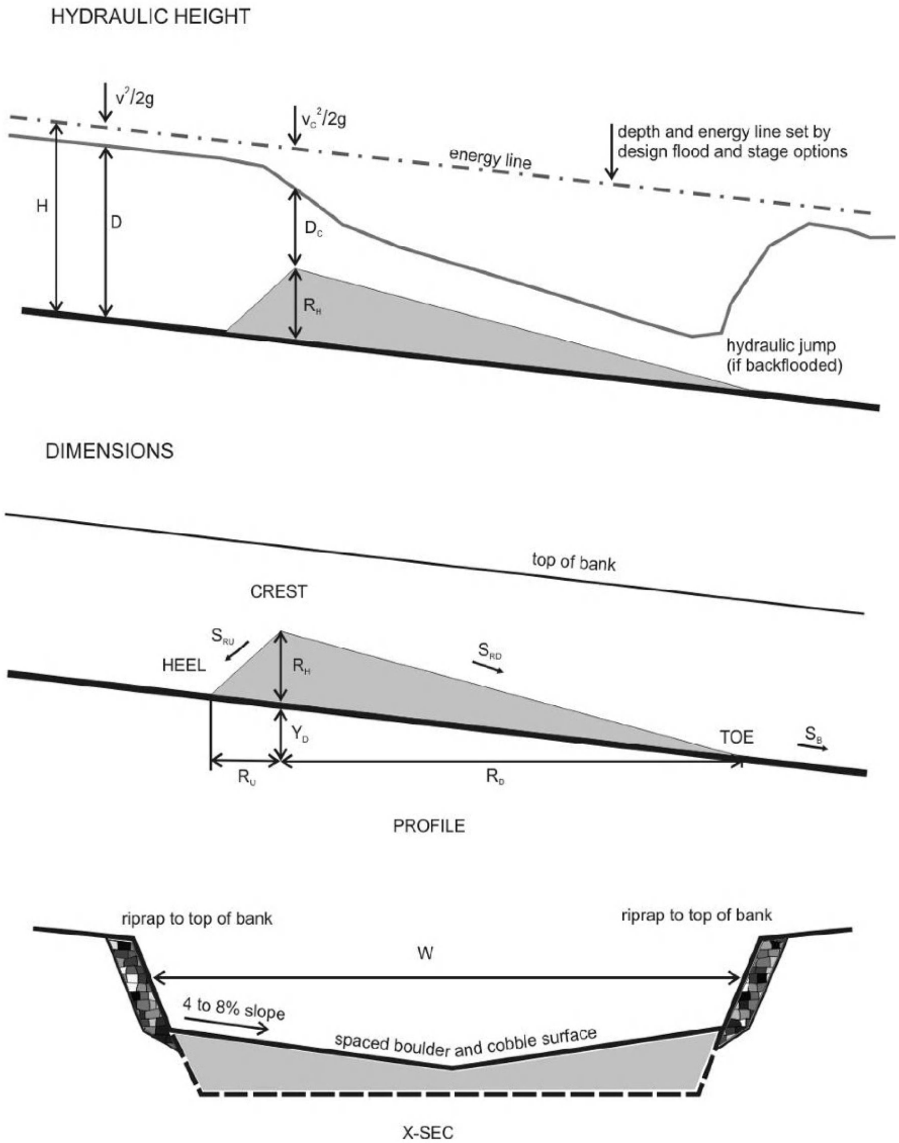

Figure 22. Riffle dimensions.

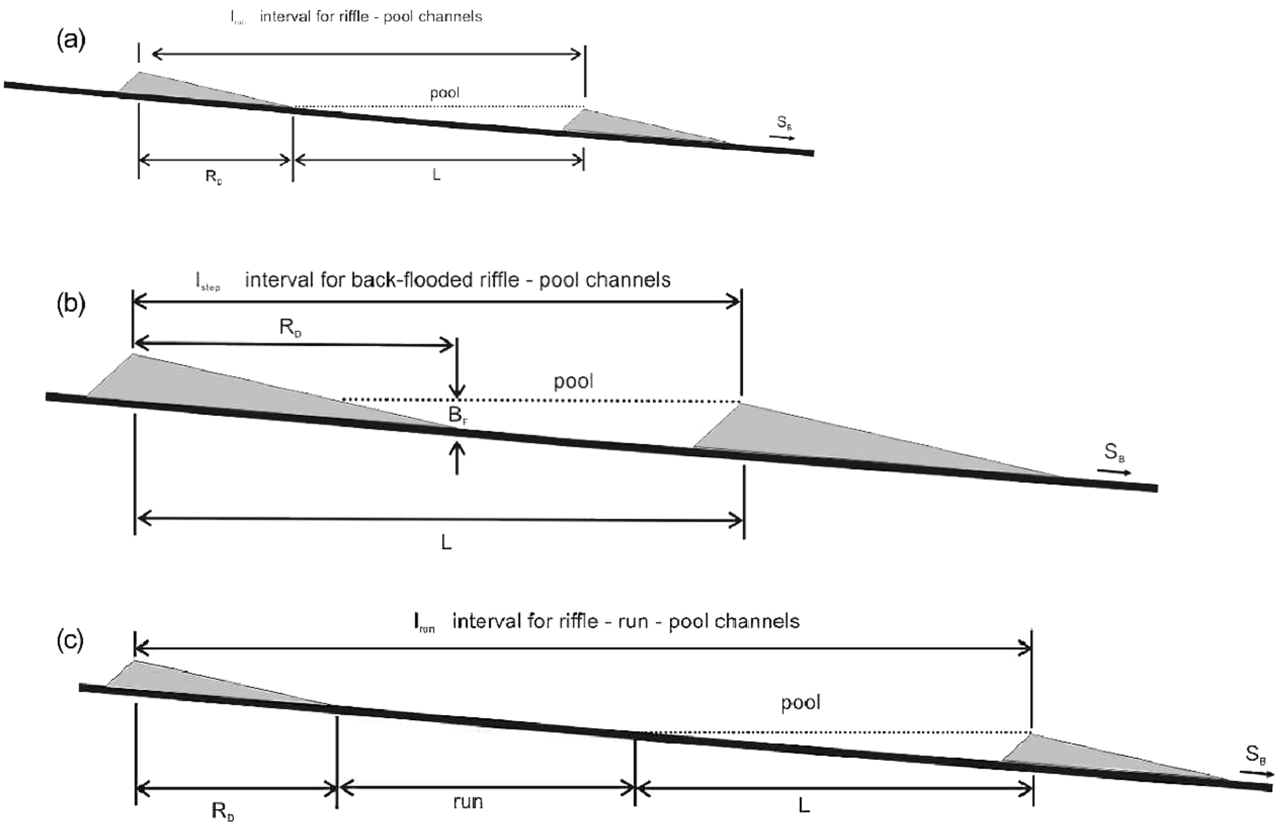

Figure 23. Pool, run, and riffle spacing (see Table 3). (a) Stepped channel with no back flooding of riffle. (b) Stepped channel back flooding the upstream riffle. (c) Pools and riffles separated by normally flowing runs.

floodplain elevation as shown in the specific energy diagram Figure 3. The riffle calculations and dimensions are summarized in Table 2 and Figures 21 and 22.

Programmed solutions for the riffle and reach hydraulics (Hydrologic Engineering Center River Analysis System program, 2008, available at hec.usace.army.mil/software) and stable construction materials (CHUTE program, 2003, available at ewatercrc.com.au) are readily available. Adjustments in the suggested roughness coefficients and bed configurations are generally required to mimic natural conditions, particularly at lower discharges.

In the Chapman Creek example, the selected design flood stage just matched the existing flood stage. Other flood stages may be selected to create pools and runs that increase the inundation frequency of an incised stream or reactivate an abandoned floodplain.

4. POOL, RIFFLE, AND RUN REACHES

A pool, riffle, and run morphology provides productive habitats for both lotic fish communities and macroinvertebrates. Several stream profiles are possible depending upon the steepness of the channel, riffle heights, and pool lengths. However, choosing the optimal design option will depend on a host of factors beyond the physical parameters of the stream including, but not limited to, project goals and objectives, adjacent land use, and flood risk tolerance. For example, pools and riffles separated by runs (Figure 23c) will allow for more natural lateral migration. If channel migration is not

Table 3. Riffle and Pool Spacing Dimensions[a]

	Definition	Function
L	pool length with no back flooding (m)	$L = R_H/S_B$
B_F	height of back-flooding on upstream riffle (m)	
I_{step}	interval between crests with back-flooding (m)	$I_{step} = L - (B_F/S_B) + R_D$
I_{run}	interval between crests with run and pool (m)	$I_{run} = L + R_D + \text{run}$

[a]See Figure 23.

Figure 24. A typical gabion basket and lined channel failure in Dickson Brook 10 years after installation.

Figure 25. Pool and riffle profile constructed between 2004 and 2009 by Parks Canada staff in the formerly channelized reach of Dickson Brook as it flows through Fundy National Park golf course, New Brunswick. Riparian vegetation has been planted in nonplay areas. Cover boulders and tree wads are added to the reaches in fairways. Photo by J. Watts.

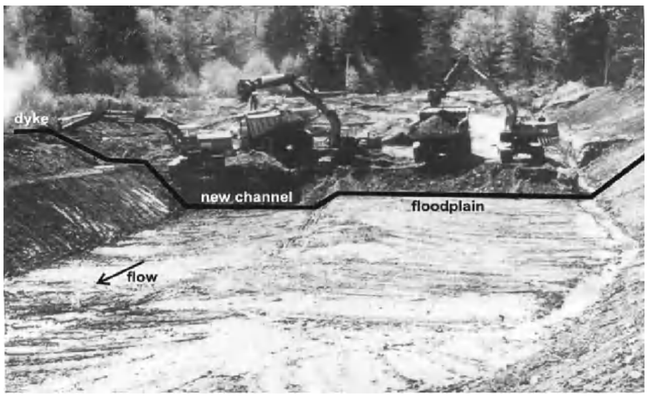

Figure 26. Channelized Oulette Creek diversion under construction in 1978.

Figure 27. Oulette Creek, British Columbia, in 2004, 10 years after pools and riffles were added to the channelized reach.

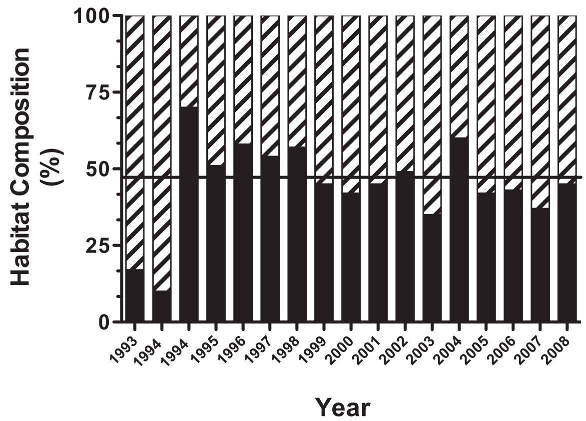

Figure 28. The distribution of riffles and pools in the Oulette Creek diversion channel before and after the addition of riffles in 1994 (average postproject configuration shown by solid histograms) (D. J. Bates, Review of rehabilitation success on stream rearing salmonids in Oulette Creek, unpublished data, FSCI Biological Consultants, Halfmoon Bay, British Columbia, Canada, 2009).

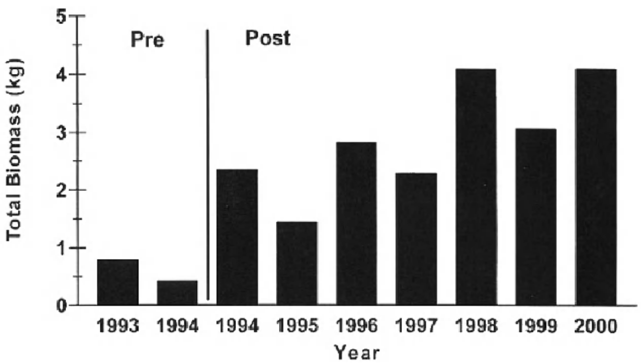

Figure 29. Response of fish biomass to the creation of a pool and riffle profile in Oulette Creek in the 7 year biological monitoring period (D. J. Bates, unpublished data, 2009).

acceptable because of land use constraints, then another design option, such as a stepped channel with pools that just reach or back flood the upstream riffle, may be more appropriate (Figures 23a and 23b). The dimensions are summarized in Table 3.

Ideally, riffle spacing would mimic natural conditions. However, in many urban and developed landscape settings, stream migration is constrained, often with repeated attempts to stabilize banks. This was the condition in Dickson Brook, New Brunswick, as it flows through an active golf course following playability criteria developed by a named designer (Figure 24).

Beginning in 2004, a closely stepped pool and riffle channel was constructed to stop bank and channel erosion without riprap, boulder, or gabion-lined banks [Figure 25, *Newbury*, 2010]. In addition, water stored in the back-flooded pools provided an allowance for evaporation during drought periods and an extended period of fish passage for the intermittent tributaries.

The shape of the riffles and the entrance and exit pools may be adjusted to create different habitat requirements. The riffle cross sections in the Dickson Brook project are broad and flat to maintain pool depths and allow golfers to easily cross the stream at numerous locations. The exit velocities are below the threshold of bed erosion at the design flood flow. In contrast, the riffle cross sections built in the channelized reach of Oulette Creek, British Columbia (Figure 26), are narrow V shapes designed to form a central torrent that erodes a deep segment of the downstream pool for overwintering coho salmon (Figure 27). The number of pools and riffles and the fish biomass has been maintained in the reach in spite of repeated overbank flood flows every year (Figures 28 and 29).

5. CONCLUSIONS

Constructed riffles have been used in a wide variety of stream systems for several decades [*Harper et al.*, 1998]. While goals and objectives of stream management and restoration projects have changed over time, the basic design approach has remained rather constant, employing essentially the same suite of one-dimensional flow equations. By using additional stream habitat information in project design, in particular natural reference reach surveys, it is possible to design in-stream structures that improve, rather than degrade, aquatic habitat.

Acknowledgments. The project examples were researched, designed, and built with Marc Gaboury, LGL Ltd. Nananaimo, British Columbia, formerly Manitoba Fisheries (North Pine River, Whiteshell River); Rick Wowchuk, Swan Valley Sport Fishing Association, Manitoba (North Pine River); Grant McBain, DFO Canada (Chapman Creek); Rob Lidden, Terminal Forest Products, Gibsons British Columbia (Oulette Creek); Martin Erickson, Manitoba Fisheries Branch (North Pine River); Jane Watts, Donald Porter, and Jeff Rossiter, Parks Canada, Fundy National Park, New Brunswick (Dickson Brook); and Stuart Mould and Jody Good, Mould Engineering Ltd. Kelowna, British Columbia (Okanagan River). The habitat data were gathered by talented graduate students and in project-specific natural channel reference sites. The lead author is grateful for the guidance and coaching when a graduate student of *Ed Kuiper* [1965] and M. Gordon Wolman that led to a career of Play: the Handmaiden of Work [*Wolman*, 1995].

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Pool-Riffle Design Based on Geomorphological Principles for Naturalizing Straight Channels

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Many channelized streams have straight trapezoidal channels that lack geomorphological structure and physical habitat, such as pools and riffles. This chapter focuses on design and implementation of pool-riffle structures in straight channels through examination of three case studies: one in which pool-riffles have been implemented successfully, one in which implementation failed, and one in which implementation has yet to occur. Basic geomorphological principles indicate that a key element in the design of pool-riffle units is the need for strong flow convergence from riffles into the pools to maintain transport of sediment through the pools as discharge increases, thereby promoting removal of accumulated sediment. This idea is evaluated in the first case study through one-dimensional (1-D) and 3-D numerical modeling of flow through a basic pool-riffle design, along with field and laboratory measurements of flow through the designed pool-riffle structures to confirm the validity of the numerical modeling. The design was implemented successfully in a straight urban channel with limited sediment load. The second case study, where implementation failed, involved crude construction techniques that did not conform to design criteria and thus did not provide appropriate hydraulic conditions for pool scour under conditions of high sediment load, resulting in pool infilling. Lessons learned are being applied in a third case study, where the basic design has been modified to accommodate straight channels with high sediment loads. Modification involves enhancement of forced constriction of flow into the pools to produce appropriate hydraulic conditions for removal of accumulated sediment from pools at high flows.

1. INTRODUCTION

Many streams around the world have been transformed from natural features into straight, trapezoidal channels to increase flow conveyance, contain floodwaters, and promote land drainage. This practice, known as channelization [*Brookes*, 1988], has been especially prevalent in urban and rural areas of the midwestern United States. In urban settings, the process of urbanization often produces dramatic increases in the amount of impervious surface cover, resulting in enhanced rates and amounts of storm water runoff [*Rhoads*, 1995]. Historically, this change in watershed conditions has exacerbated local flooding, leading to channelization of urban streams in an attempt to maximize flow conveyance and alleviate flooding. Much of the land throughout rural portions of the Midwest is relatively flat and poorly drained, a product

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of widespread continental glaciation during the late Pleistocene Epoch. The concern here has been the need for adequate land drainage to support industrial agriculture [*Rhoads and Herricks*, 1996]. Drainage tile systems have been installed beneath many farm fields to improve subsurface drainage of rich prairie soils. In turn, local streams have been deepened, straightened, and widened or extended headward through mechanical ditching to provide adequate outlets for field drainage systems. The result is that as much as 100% of headwater streams in some areas of the Midwest have been channelized [*Mattingly et al.*, 1993].

Over the past two decades, public opinion toward stream management has undergone a gradual transformation. Concern about the environmental consequences of channelization, including deleterious effects on ecological, water quality, and geomorphological conditions of streams, has been growing, as have efforts to try to undo or mitigate these effects. Many community-based environmental groups are now advocating for multiobjective approaches to management that include consideration of environmental aspects of stream systems. Such approaches are seeking to change highly modified channels into naturalized fluvial environments. These efforts are still design-based, but seek to develop environmentally friendly designs that incorporate ecological and geomorphological components.

The purpose of this chapter is to demonstrate how geomorphological principles can contribute to the naturalization of straight channelized streams by providing guidance for the design and implementation of pool-riffle sequences in such streams. The focus of analysis is on situations where cost constraints, extant infrastructure surrounding a stream, or strong opposition from particular stakeholders preclude reconfiguration of the channel pattern through practices like remeandering. The design approach developed here is aimed at the establishment of sustainable pool-riffle sequences within straight, flat-bottomed channels with low gradients. The chapter examines (1) the development of a basic design for pool-riffle sequences in straight channels, (2) implementation of the basic design in a straight urban channel with limited bed material transport, (3) the success of this implemented design as determined from postimplementation monitoring, (4) failure of a pool implementation project that did not conform with design principles in a straight rural channel with a high sediment load, and (5) modification of the basic design to accommodate straight channels with abundant amounts of mobile bed material.

2. WHAT IS STREAM NATURALIZATION?

The attempt to design and implement pool-riffle sequences in straight channels is part of a research program aimed at developing a scientific and technological basis for "naturalizing" streams in agricultural and urban landscapes of the midwestern United States [*Frothingham et al.*, 2002; *Rhoads and Herricks*, 1996; *Rhoads et al.*, 1999; *Wade et al.*, 2002]. The concept of stream naturalization represents a specific perspective on environmental management of aquatic resources in human-dominated environments that was developed as an alternative to the concept of restoration as outlined by the National Research Council [*National Research Council*, 1992]. An important emphasis of this perspective is that all stream-management initiatives, especially community-based efforts, are fundamentally social in nature. In virtually all cases, environmental management of streams is guided by scientific input but is not determined by this input, an important detail that scientists and technical experts often fail to fully recognize. Attempts to "naturalize" channelized streams in human-dominated settings typically are driven by an environmental vision that has as much to do with aesthetics and the perceived benefits that an attractive stream environment has for economic growth, community pride, and utilitarian concerns (e.g., flood control and land drainage) as it does with geomorphological or ecological considerations. In other words, the very notion of "natural" in human-dominated contexts is socially determined and cannot be reduced to an objective standard defined by technical experts (such as the predisturbance condition: the reference state of restoration as defined by the *National Research Council* [1992]). Naturalization in this sense is highly place-based: what one community views as natural, another may see as unnatural or artificial. Thus, the second point of emphasis is that desired target states for naturalization are highly variable. In particular, the classical notion of restoration, the return of a system to its undisturbed state (i.e., its pristine condition), often becomes meaningless in agricultural and urban contexts given the extensiveness of human modification of the landscape. Because the goal commonly is to undo deleterious environmental effects of channelization, the reference state, if one can be defined at all, becomes the extant, channelized condition of the stream, which has been homogenized in form and function. The goal is not to move toward the reference state, as it is in restoration, but to move the system away from the current highly simplified state toward alternative conditions that establish sustainable, morphologically and hydraulically varied, yet dynamically stable fluvial systems that are capable of supporting healthy, biologically diverse aquatic ecosystems [*Rhoads et al.*, 1999; *Wade et al.*, 2002]. The concept of stability is somewhat elusive, and tension often exists between the dynamic character of natural streams and socially acceptable levels of stream dynamics [*Rhoads et al.*, 2008; *Shields et al.*, 2008]. From a scientific perspective, stability is often defined as a balance between erosion and deposition over a period of years or as the lack of accelerating rates of change in fluvial processes [*Biedenharn et al.*, 2008; *Rhoads*, 1995]. Thus, a stream can be highly dynamic and still be considered stable. An important aspect of naturalization is that stream-management alternatives are defined on the basis of social negotiations within the community of relevant stakeholders. Scientists and technical experts should engage in these negotiations to help define objectives that are consistent with scientific understanding of natural processes and to determine how objectives derived from a community-based environmental vision can be met within practical limits of technical feasibility and cost.

3. POOL, RIFFLES, AND STREAM NATURALIZATION IN NORTHBROOK, ILLINOIS, A SUBURB OF CHICAGO

A naturalization project along the West Fork of the North Branch of the Chicago River (WFNBCR) in downtown Northbrook, Illinois, served as the initial impetus for development of a pool-riffle design for straight channelized streams. The process of naturalization involved balancing local stakeholders' environmental vision of enhanced aesthetics, erosion control, fish habitat, and water quality with scientific and technical input about what type of enhancements are feasible and sustainable. The community of Northbrook initially sought to beautify the WFNBCR within the downtown area, which over the years had been transformed into a straight trapezoidal drainage channel intended mainly to convey increased storm water runoff from growth of the city and surrounding suburban communities. Initial plans focused on clearing of weedy trees along the channel, reshaping and stabilizing channel banks, protection of the bank toes from erosion, and development of a riverside walkway within a public park along the lower part of the project reach. The Friends of the Chicago River, a volunteer nonprofit organization committed to improving the environmental quality of the river, took an interest in the project, and through their involvement, the city decided to naturalize the aquatic environment by enhancing in-stream habitat for fish.

Preliminary assessment of geomorphological conditions and fish community characteristics in the reach provided baseline information to guide the design [*Rhoads et al.*, 2008]. This assessment revealed bed topography in the reach, which has a gradient of about 0.003 m m^{-1}, was relatively uniform and lacked the development of bar-scale geomorphological features, such as pools and riffles. Bed material consisted mainly of coarse sand and fine gravel, but the availability of this material for sediment transport was limited. The veneer of alluvium generally was thin (<20 cm), and in many places, underlying glacial till was exposed on the channel bottom. Several small concrete riffles emplaced in the channel in the 1930s were highly stable and showed no signs of sediment accumulation upstream of these structures, which resembled low (height <20 cm) check dams. The lack of deep-pool habitat (>0.5 to 1.0 m) during typical base flow conditions (discharge [Q] <1 m^3 s^{-1}) was identified as a major factor limiting fish diversity and abundance in the 900 m long project reach [*Wade et al.*, 2002]. As a result, habitat improvement focused on design and installation of pool-riffle structures in the reach to enhance geomorphological, hydraulic, and ecological diversity. Details of the design and implementation of these structures are provided by *Rhoads et al.* [2008] and *Wade et al.* [2002]. The following discussion focuses on field evaluation of the morphology, morphological stability, and hydraulic performance of the installed structures.

The naturalization design drew upon geomorphological principles concerning the form and function of pools and riffles in natural rivers. The occurrence of pools and riffles in straight channels has been associated with the development of alternating bar units [*Frothingham et al.*, 2002] (Figure 1). The structure of bar units leads to flow convergence in pools, promoting scour, and flow divergence over riffles, promoting deposition. Bar units also typically deflect flow laterally triggering bank erosion and the possible initiation of channel meandering [*Rhoads and Welford*, 1991]. However, a key consideration in the design of the pool-riffle structures in the WFNBCR was to promote self-maintenance of these structures, yet not produce hydraulic conditions that would lead to erosion of channel banks within the reach, which was bordered closely by parking lots and roadways (Figure 2). The resulting design promoted convergence of the flow into a pool in the center of the channel and mild divergence of flow over a riffle

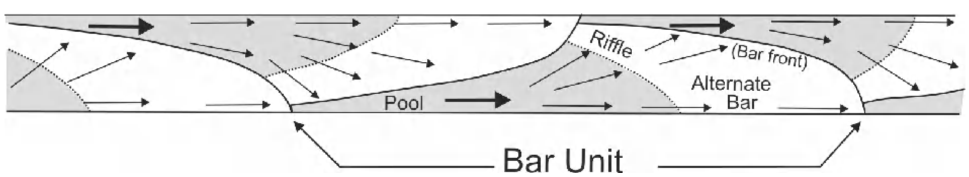

Figure 1. Plan view of typical bar units and pool-riffle structure in straight channels. Arrows show flow paths, and shading corresponds to portions of the bed below the mean bed elevation.

that extends across the width of the channel bed (Figure 3). Flow convergence into pools and divergence over riffles is a key attribute of pool-riffle maintenance [*Harrison and Keller*, 2007; *MacWilliams et al.*, 2006; *Thompson et al.*, 1999]. A goal of the design was to have the rate of increase in bed shear stress within the pools exceed the corresponding rate of increase for riffles as flow stage increases. This characteristic causes shear stress magnitudes in pools to converge on or even exceed the magnitudes in riffles with increasing stage: a phenomenon that has been documented for natural pools and riffles [*Booker et al.*, 2001; *Cao et al.*, 2003; *Lisle*, 1979; *Milan et al.*, 2001]. At low flow, riffles have the highest bed shear stress, but at flows near bankfull, the bed shear stress in pools approximates or exceeds the bed shear stress in riffles. The differential rates of increase in bed shear stress with increasing stage are viewed as a critical factor in the self-maintenance of pools and riffles. Such conditions should promote higher rates of sediment transport in the pools than at the riffles at high discharges, resulting in flushing of accumulated sediment out of the pools.

Experimental studies using a physical model indicated that the pools and riffles, as designed, promote flow convergence and divergence and exhibit differential rates of increase in bed shear stress with the bed shear stress along the centerline of pools exceeding the bed shear stress along the centerline of the riffles at high stage [*Rhoads et al.*, 2008]. These results provided experimental confirmation that the hydraulic performance of the pool-riffle structures should support self-maintenance. After further testing using a three-dimensional (3-D) computational fluid dynamics model reinforced these findings, the city of Northbrook decided to install the structures in the WFNBCR.

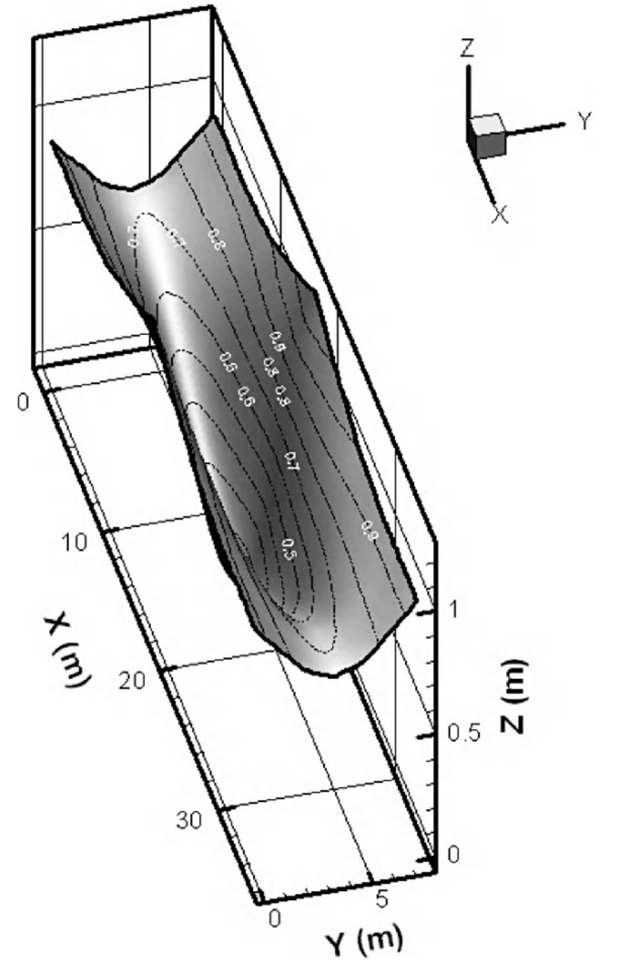

Figure 3. Three-dimensional (3-D) image of pool-riffle design in downtown Northbrook (contour interval = 0.1 m).

Figure 2. Parking lots and alleys flanking the WFNBCR in downtown Northbrook, November 2001.

Installation involved on-site supervision of construction of the pool-riffle structures by members of the University of Illinois (UI) research and design team. Construction began in November 2001 and continued through May 2002. In all, 11 pool-riffle sequences were installed along a 900 m reach of the WFNBCR. Reshaping of channel banks, which in the design was intended to contribute to flow constriction in the pools (Figure 3), occurred prior to modifications to the channel bed and was not supervised by the research and design team. Thus, this element of the design was not effectively incorporated into construction. Instead, an alternative component, arcuate rock ribs, positioned halfway between the riffle crests and pool centers, was included in construction to enhance deflection of flow toward the center of the pool (Figure 4).

The morphology of the pool-riffle structures as constructed approximated the design shape (Figure 5), but the degree of constriction within the pools in the design was not reproduced

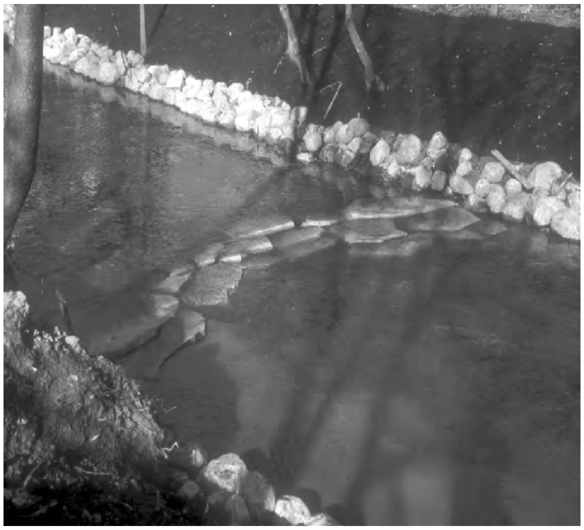

Figure 4. Arcuate rock rib between riffle crest and pool center.

in construction. The proposed amount of constriction would have required modification of the lower banks, which were reshaped without the involvement of the UI team prior to modification of the channel bed. Pools as constructed generally were about 0.65–0.75 m lower than the riffles, whereas the design aimed to provide pools about 1.0 m lower than adjacent riffles. Two factors affected pool depths in the field: (1) the approximate method of determining depths of excavation at the time of construction, which involved probing with a surveying rod to determine the shape of the pool and (2) limits on the depth of penetration of the backhoe into the dense glacial till underlying the shallow alluvium on the channel bed.

After completion of construction in May 2002, a moderate flow event occurred in the project reach in early June 2002 (Figure 6). The peak discharge of this event, recorded on 4 June 2002 at a U.S. Geological Survey gauging station 1 km upstream from the reach, was about 10 $m^3 s^{-1}$, an event that corresponds to a recurrence interval of about 1.3 years on the annual flood series (Figure 7). On 5 June, during the receding limb of this event when discharge declined from 2.7 to 2.0 $m^3 s^{-1}$, measurements of velocities were obtained in the downstream (U), cross-stream (V), and vertical (W) directions using two acoustic Doppler velocimeters at five cross sections within a pool-riffle structure toward the downstream end of the project reach (Figures 5 and 8). During the period of measurements, flow depths were about 0.35 to 0.45 m over the riffles and about 1.05 m in the deepest part of the pool.

Overall, the patterns of streamwise velocities and 3-D velocity vectors for the field prototype are quite comparable to the patterns for flow through the experimental version of the pool-riffle structures for a similar scaled volumetric flow rate (Figure 9). At this relatively low stage, the highest streamwise velocities (>1 $m s^{-1}$) are found over the riffles. Also, as expected, velocity vectors for the field measurements indicate that flow converges into the pool downstream of the rock rib. Although the magnitude of the measured flow was not sufficient to promote pool maintenance, the documented similarity between patterns of measured velocities for experimental and

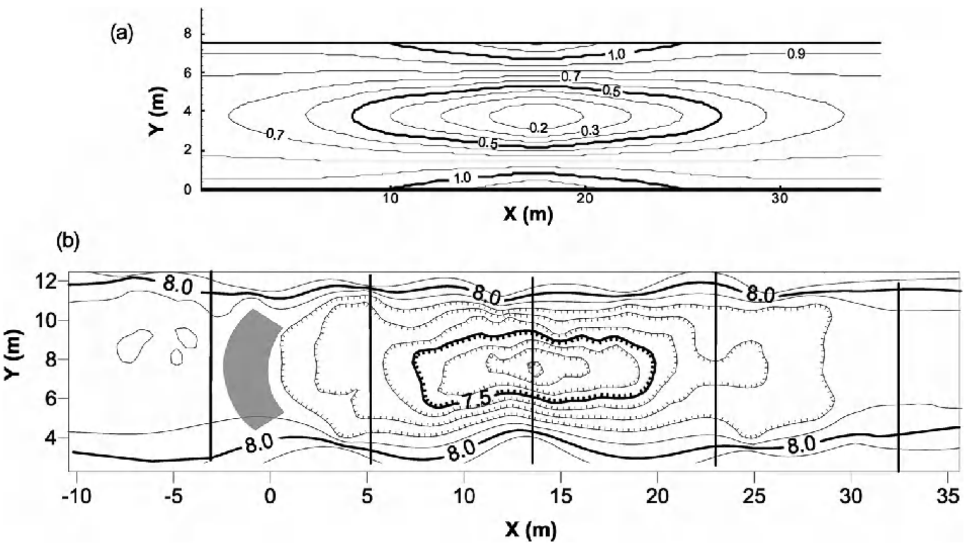

Figure 5. Comparison of topography of pool-riffle structures: (a) as designed (contour interval 0.1 m, arbitrary datum) and (b) as constructed (pool-riffle unit 3 in reach 3, WFNBCR, Northbrook, Illinois; contour interval 0.1 m, arbitrary datum; lines indicate cross sections for flow measurements; shading is location of rock rib; flow from left to right).

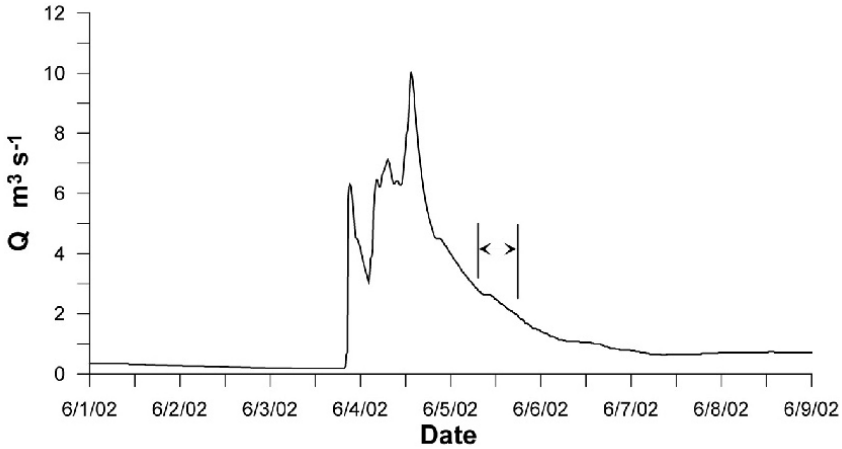

Figure 6. Hydrograph of flow event in WFNBCR in early June 2002. Brackets with arrows show interval of flow measurements in pool-riffle unit 3.

prototype conditions suggests that experimental hydraulic conditions for high flows, when pool maintenance is expected to occur, should be reproduced at least approximately in the prototype.

Since implementation of the pool-riffle structures in May 2002, the longitudinal profile of the project reach has been surveyed each year to determine whether any infilling of pools or erosion of riffles has occurred. These surveys reveal

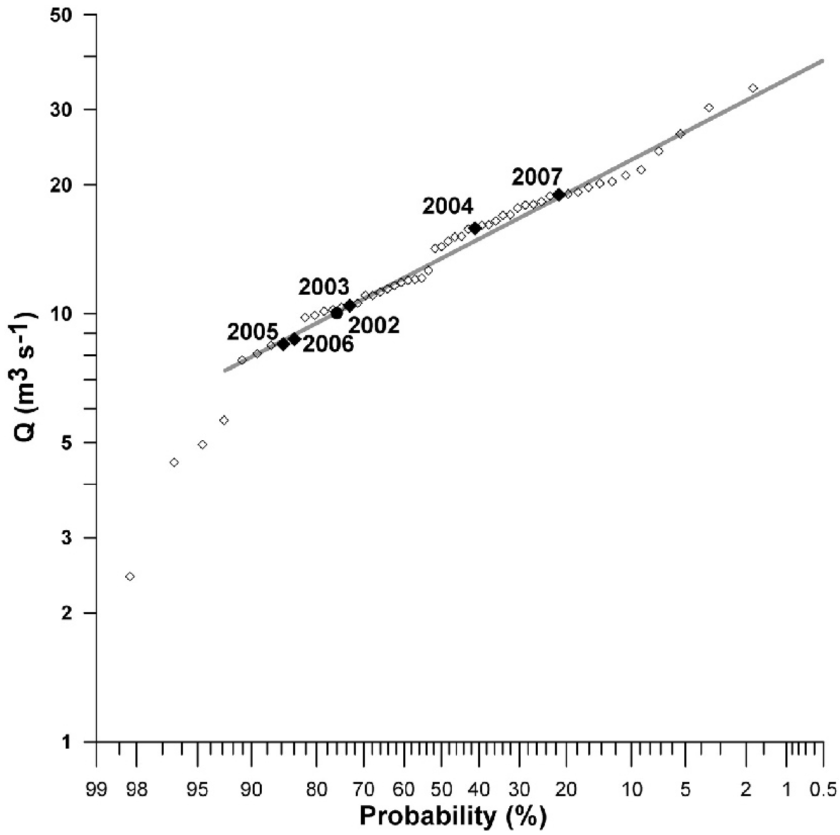

Figure 7. Annual series flood frequency curve for WFNBCR showing magnitudes of annual peaks from 2003 to 2007 (black diamonds) and the flow event immediately following construction in June 2002 (black dot).

Figure 8. Flow measurements in pool-riffle unit 3, 5 June 2002.

that after some minor adjustments immediately following construction, pool depths and riffle heights have remained virtually unchanged over a 6 year period (Figure 10). Total bed relief of about 0.65 to 0.75 m between the tops of riffles and the bottoms of pools still exists throughout the reach. Annual peak discharges during this time have ranged from a high of 19 m^3 s^{-1} with a recurrence interval of about 5 years to a low of 8.5 m^3 s^{-1} with a recurrence interval of about 1.15 years (Figure 7). None of these peaks constitutes an extreme event, but the structures do appear to be self-sustaining under the influence of floods of moderate magnitude. Moreover, a discharge of 19 m^3 s^{-1} is greater than the bankfull capacity (≈15 m^3 s^{-1}) of the straight trapezoidal channel of the WFNBCR in Northbrook. Floods greater than this capacity spill onto the floodplain of the adjacent urban landscape and are unlikely to produce substantial increases in the erosive potential of flow within the channel.

In terms of ecological benefits, implementation of the pool-riffle structures has increased fish abundance, biomass, and diversity within the project reach [*Schwartz and Herricks*, 2007]. Nevertheless, fish metrics are still in the low range compared to rural streams in the area, indicating that local habitat enhancement has only a limited capacity for improving fish community composition given the influence of intense urbanization on watershed-scale conditions, such as water quality, hydraulic stresses, and barriers to fish movement. Such findings support the notion that major ecological benefits in highly urbanized landscapes may be possible only through attempts to naturalize entire watersheds, rather simply enhancing physical habitat in short sections of streams [*Palmer et al.*, 2010]. Unfortunately, the political and economic will to implement naturalization projects at the watershed scale requires coordination among a complex array of stakeholders with diverse and often conflicting interests, a level of coordination that typically is difficult to achieve.

4. STREAM NATURALIZATION OF STRAIGHT CHANNELS IN EAST CENTRAL ILLINOIS: REFINEMENT OF NATURALIZATION DESIGN

Stream naturalization has also emerged as an environmental concern in the agricultural landscape of east central Illinois where streams throughout entire headwater drainage

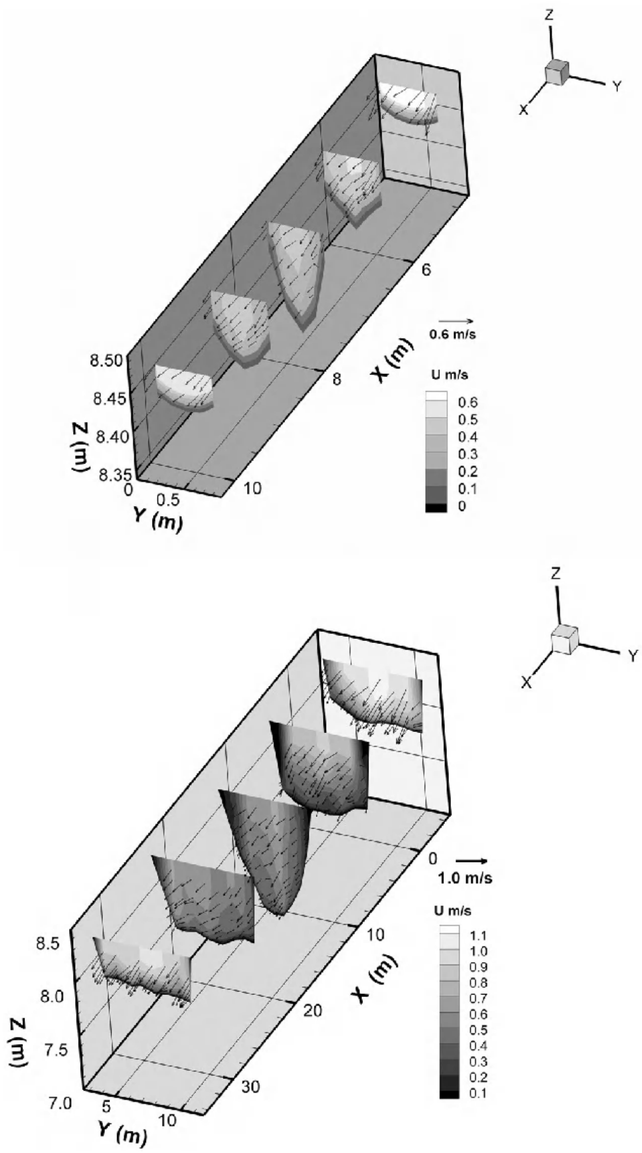

Figure 9. Downstream velocities (contours) and 3-D velocity vectors in the pool-riffle unit: (top) flume experiment, with $Q = 0.02$ m^3 s^{-1}, and (bottom) unit 3 in the WFNBCR, Northbrook, Illinois, on 5 June 2002, with $Q \approx 2.1$ to 2.6 m^3 s^{-1}.

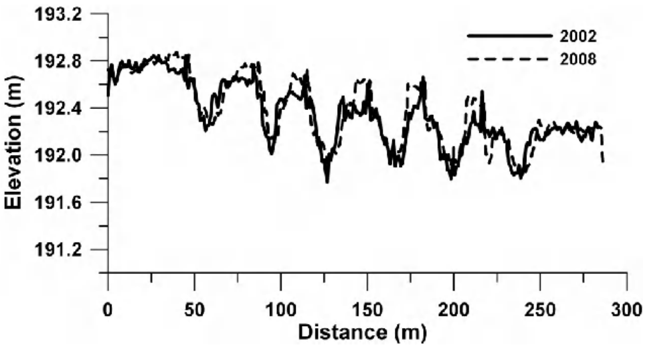

Figure 10. Longitudinal profiles through pool-riffle structures in the downstream portion of the project site, WFNBCR, immediately after excavation of pools in March 2002 and in August 2008. Slight difference in profiles at riffles is due to addition of coarse rock on riffle crests in May 2002.

networks have been channelized for the purpose of land drainage [*Frothingham et al.*, 2002; *Rhoads and Herricks*, 1996]. As in urban settings, many streams have been simplified by converting them into straight trapezoidal channels with flat, uniform beds. Studies relating fish community characteristics to geomorphological conditions have shown that enhanced reach-scale variability in channel depth and secondary-flow patterns are associated with increased fish biomass, species richness, and Index of Biological Integrity scores [*Frothingham et al.*, 2001; *Rhoads et al.*, 2003]. Preservation of previously channelized reaches that have reverted to meandering forms through long-term absence of channel maintenance represents one approach to naturalization of human-modified headwater streams [*Rhoads and Herricks*, 1996; *Rhoads et al.*, 1999]. However, the potential for such efforts to achieve widespread benefits is constrained by ongoing channel-maintenance practices and by the limited capacity of straightened headwater streams to recover a meandering pattern [*Urban and Rhoads*, 2003]. The economic demand for adequate land drainage precludes stream naturalization involving extensive remeandering of straight ditches; instead, naturalization in many cases will entail habitat enhancement in straight agricultural channels.

4.1. An Attempt to Create Pools as Part of Channel Maintenance in the Embarras River

A rudimentary attempt to implement pools in a straight agricultural channel occurred in the summer 2001 along a headwater section of the Embarras River in east central Illinois. This section of the river, which has a gradient of 0.0008 m m^{-1}, was deemed in need of maintenance by the local drainage district. As part of the process of obtaining a permit for the maintenance project under section 404 of the Clean Water Act, the drainage district submitted a permit application to the U.S. Army Corps of Engineers (USACE) district office in Louisville, Kentucky. The USACE had concerns about the impact of the project on fish habitat and required the drainage district to include habitat enhancement in the project design as part of the permitting agreement. Studies of fish habitat in the upper Embarras River have revealed that the lack of abundant deep pool habitat, caused mainly by frequent channel maintenance, is a critical factor limiting fish abundance and diversity [*Frothingham et al.*, 2001; *Rhoads et al.*, 2003; *TerHaar and Herricks*, 1989]. Geomorphological research demonstrated that development and maintenance of pool-riffle morphology is certainly possible in the Embarras River as evidenced by the existence of well-developed pools and riffles in meandering sections of this river [*Frothingham and Rhoads*, 2003]. The UI research team had worked previously with the drainage district commissioners on stream naturalization [*Rhoads and Herricks*, 1996; *Rhoads et al.*, 1999] and offered to be on site to help supervise excavation of pools by the contractor hired to perform the channel maintenance.

Material that the drainage district wanted to remove from the channel consisted mainly of coarse sand and fine gravel that had accumulated in the form of vegetated bars on the bottom of the ditch (Figure 11). Underlying this accumulated sediment was cohesive glacial till into which the pools were excavated (Figure 11). Prior to channel maintenance, the channel had a bankfull width of approximately 10 m. The constructed pools were spaced six channel widths, or 60 m, apart. In total, five pools were excavated over a reach about 270 m long.

Because the drainage district wanted to minimize the cost of extra time required for the contractor to dig the pools and because the contractor had to use a backhoe bucket with teeth to excavate pools into the glacial till, compared to the flat-edged bucket typically used to excavate unconsolidated sand and gravel, all five pools were constructed in a single day (1 June 2001). The rapid construction of the pools limited the time available to shape them into a form similar to the design used in the WFNBCR at Northbrook, Illinois. Digging of the pools was performed by a backhoe operating from the top of one bank of the channel (Figure 11). The reach of the backhoe arm was limited, and it was difficult to excavate deeply into the bed toward the side of the channel opposite the backhoe. Excavation proceeded from downstream to upstream: an approach that is less than ideal because material mobilized by disturbance of the channel bed upstream can potentially fill in downstream pools. The decision to adopt this approach was made by the contractor, who preferred to work in the upstream direction based on prior experience performing channel maintenance. Also, more effort was devoted to excavation of the first three pools than to the last two because of time

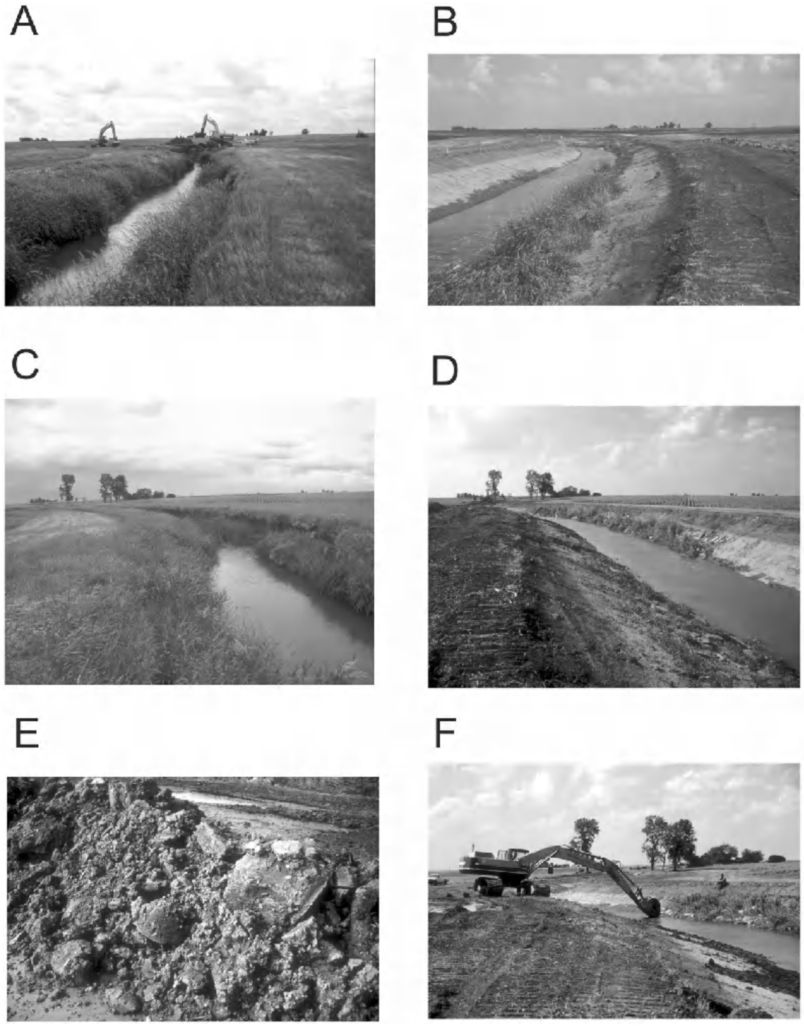

Figure 11. Upper Embarras River: (a) looking downstream before maintenance, (b) looking downstream after maintenance, (c) looking upstream before maintenance, (d) looking upstream after maintenance, (e) spoil pile containing cohesive glacial till excavated from the channel bottom, and (f) excavation of a pool.

constraints; as a result, the first three pools were deeper than the last two. A reconnaissance walk down the center of the channel at the end of the day on 1 June 2001 indicated that the pools amounted to not much more than isolated holes dug into the channel bed.

On 25 June 2002, cross-section surveys were performed along the reach to determine the morphology of the maintained channel at the locations of the pools and at intervening locations. These surveys indicated that the first three pools were 0.5 to 0.8 m deeper than intervening portions of the newly maintained channel, whereas the other two pools were only 0.3 to 0.5 m deeper than adjacent nonpool locations. Repeat surveys were conducted in September 2001 and January 2002 to document evolution of the pools. The survey data show that between these two dates, the pools filled with coarse sand and fine gravel, sediment indistinguishable from other bed material in this river (Figure 12). This period included several high flows that should have been capable of "flushing" sediment from pools if these structures had appropriate hydraulic conditions. The amount of sediment accumulation roughly equaled the difference in bed elevation between the pool bottoms and the bed at intervening locations, resulting in elimination of local relief on the channel bed associated with the pools. Net aggradation occurred only in the pools, not at intervening locations (Figure 12), suggesting that sediment-transport capacity in the pools was not adequate to prevent deposition of material transported into them from upstream, even at high flows.

The outcome of this project indicates that simply digging holes into the bottom of a straight trapezoidal channel with

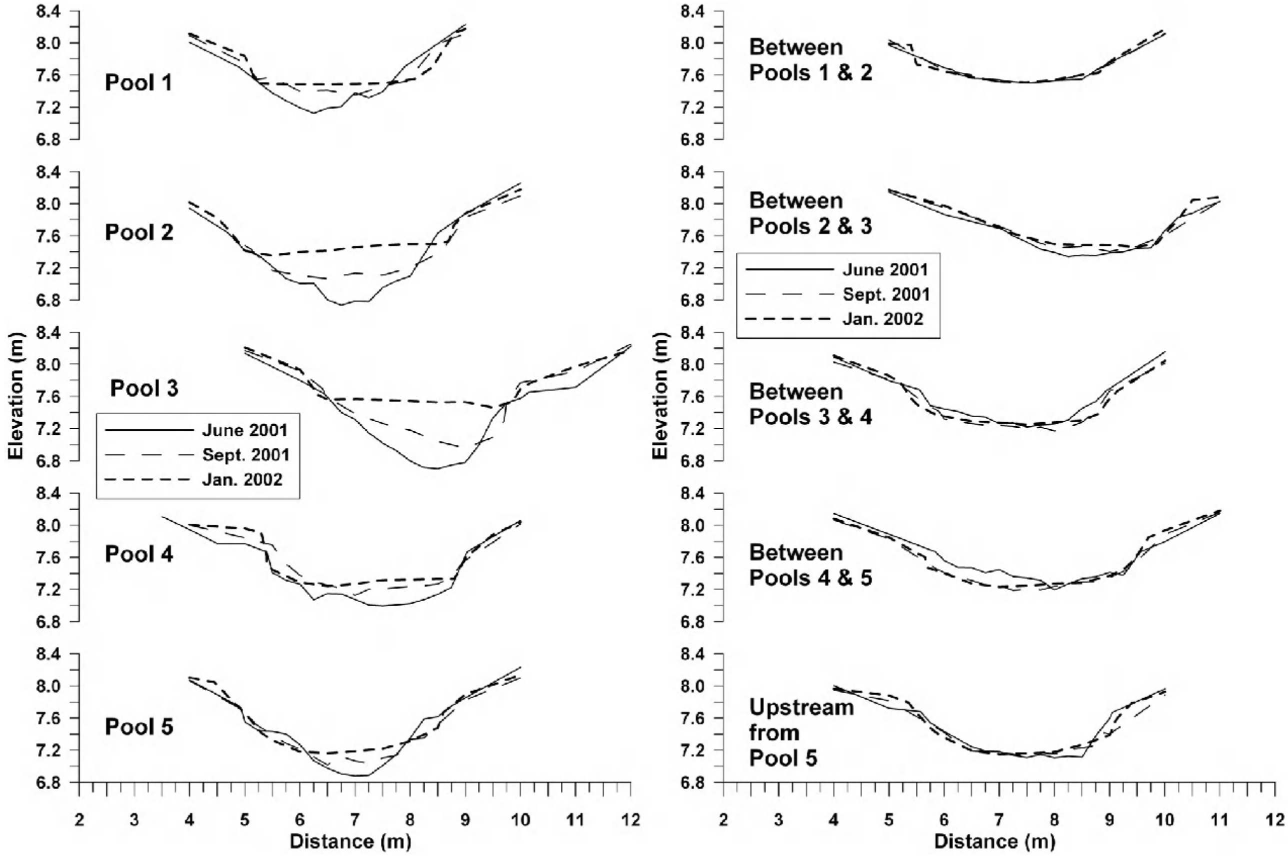

Figure 12. Repeat surveys of channel cross sections (left) at pools and (right) between the pools along the maintained reach of the Embarras River.

abundant coarse sand and fine gravel is not a viable approach to creating self-sustaining pool habitat. The large amounts of mobile bed material in the Embarras River, combined with the crude construction method, led to rapid eradication of the pools via infilling by sediment. The lesson to be learned is that preservation of pools in straight channels depends on the establishment of a morphologic configuration that generates the hydraulic conditions required to transport sediment through the pools at high flows, i.e., a reversal of shear stress wherein the shear stress in a pool exceeds the shear stress in the adjacent riffles.

4.2. Design and Modeling Analysis of Pool-Riffle Structures in the Copper Slough

Another application of the pool-riffle design is being considered for the Copper Slough, a drainage channel on the urban-rural fringe of Champaign-Urbana, Illinois (Figure 13). Implementation of the structures in this setting is in response to ecological damage caused by a chemical spill in 2000 that killed hundreds of thousands of fish along the Copper Slough and Kaskaskia River. The Illinois Department of Natural Resources (IDNR) is coordinating the project and sought the advice of the UI team on project design. A primary goal is to enhance fish habitat. Past work in the Copper Slough has shown that pool habitat is sparse in this channelized system and that the few pools that do exist attract large numbers and diverse species of fish [*Jayjack*, 1993]. Therefore, creation of pool habitat is a high priority for the Copper Slough.

The project reach consists of an approximately 300 m stretch of straight, trapezoidal drainage channel with a flat bottom (Figure 13). The channel is 4 to 5 m deep with a bottom width of 6 to 7 m. The banks have slopes of 1.0 m vertical to 1.3–1.5 m horizontal and are heavily vegetated by tall grasses. The average gradient of the channel bed in the reach is 0.0014 m m^{-1}. In contrast to the WFNBCR, but much like the upper Embarras River, the Copper Slough transports abundant sand and fine gravel. Also, glacial till lies beneath the shallow alluvium on the bottom of the drainage ditch and is exposed locally in zones of scour.

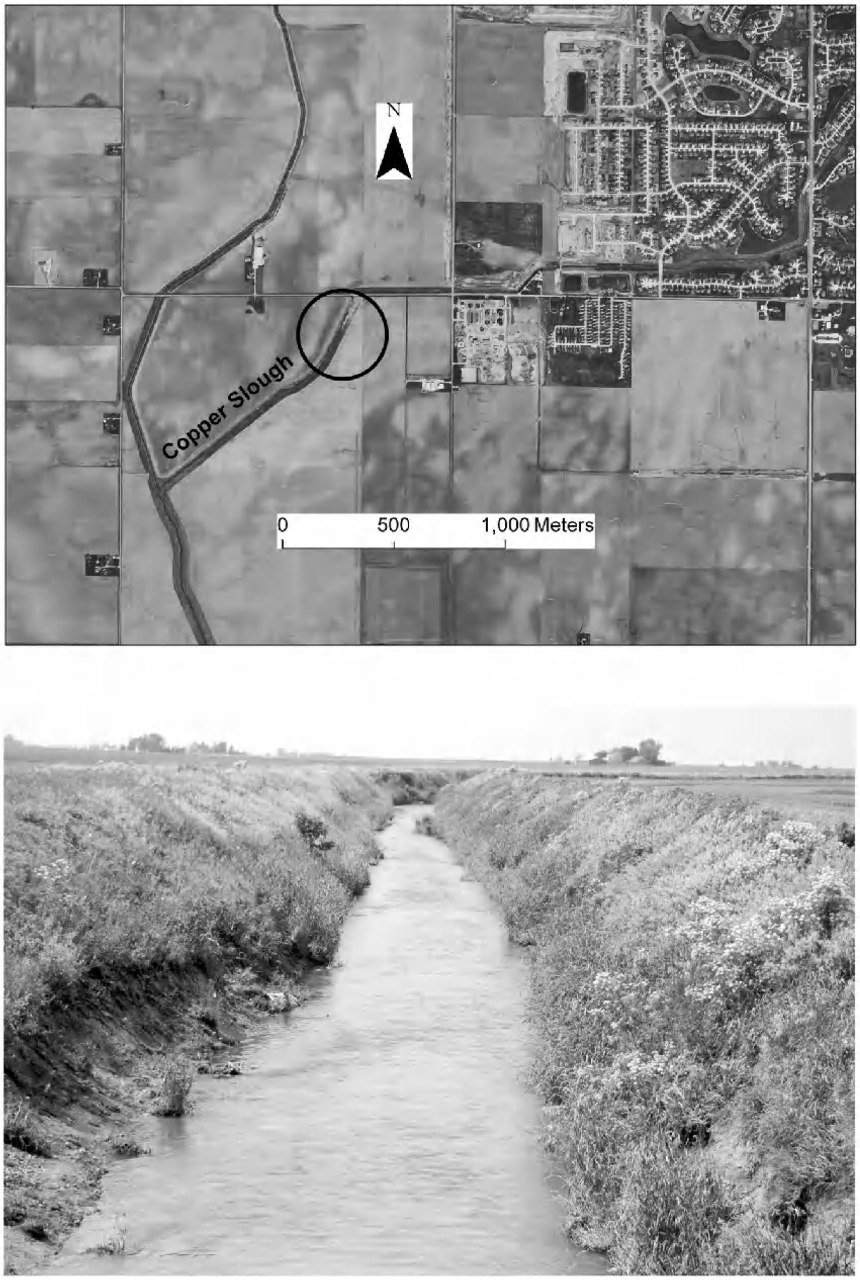

Figure 13. (top) Aerial image of Copper Slough showing project reach (circled). Suburban development from southwest fringe of Champaign, Illinois, is visible in the upper right. (bottom) Copper Slough project reach looking downstream.

The engineering firm retained by the party responsible for the spill, after consulting with the UI team, proposed a design based on the WFNBCR project but without any modifications of the lower channel banks (Figure 14). This design, referred to as design 1, consisted of elliptical pools, 0.92 m deep and 55 m long, separated by flat riffles with lengths of 4.6 m. IDNR staff were concerned that design 1 did not take into account concerns raised by the UI team about the need to generate pronounced flow convergence in pools for fluvial systems like the Copper Slough that transport substantial amounts of bed material load. Infilling of pools in the upper Embarras River served as a cautionary warning that without appropriate hydraulic conditions, excavated pools in channels with high sediment loads will not be maintained. In

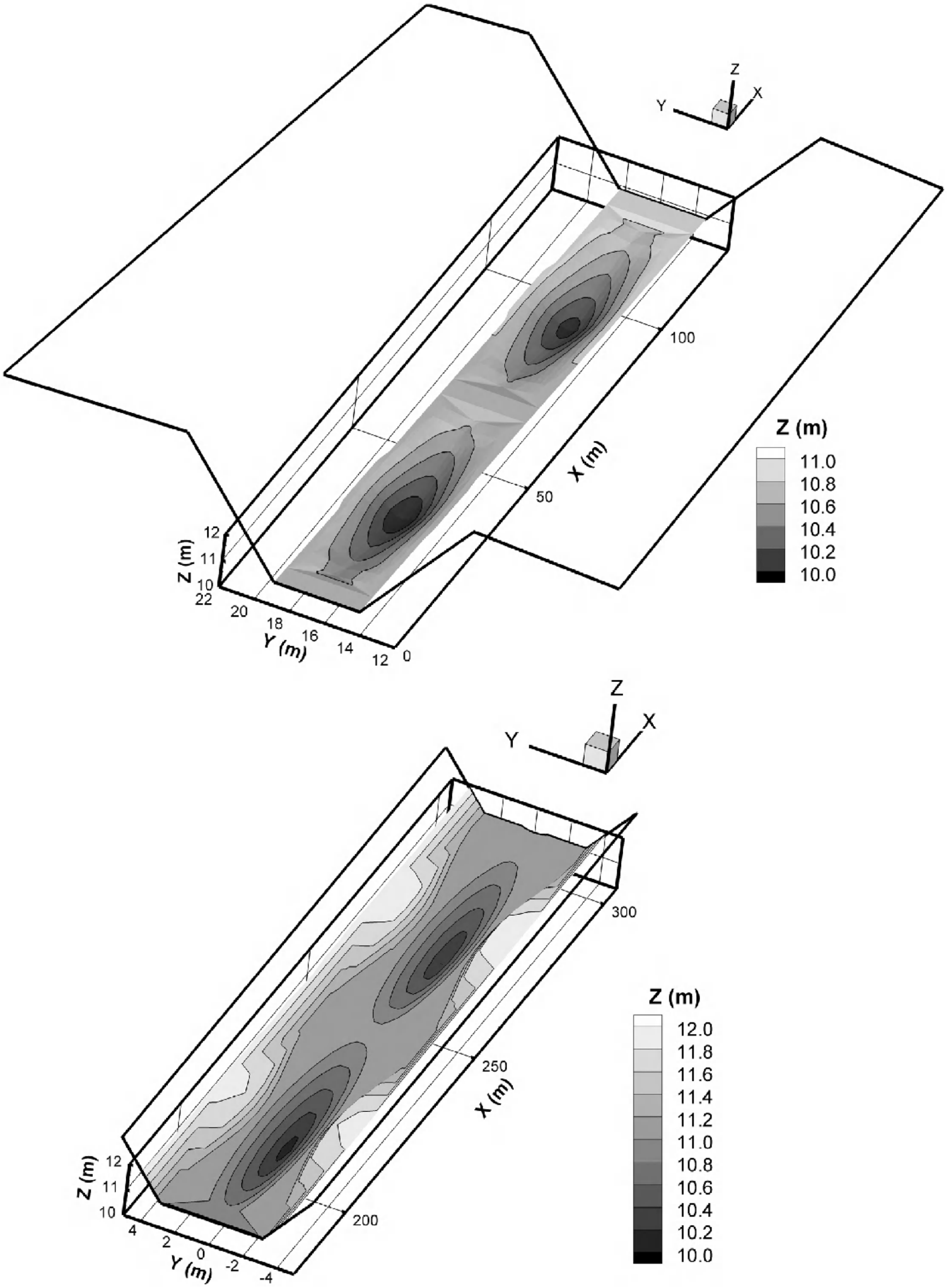

Figure 14. Three-dimensional views of pool-riffle designs for the Copper Slough. (top) design 1, view through pool-riffle units 3 and 4 without bank constrictions. (bottom) design 2, view through pool-riffle units 3 and 4 with bank constrictions.

response to these concerns, the UI team proposed a modified version of the WFNBCR design that increases the amount of flow convergence in the pools through modification of the lower channel banks (Figure 14). In this design, referred to as design 2, the lower portions of the channel banks form a sinusoidal pattern along each side of the trapezoidal channel such that the bottom width reduces to 3.5 m adjacent to pool centers and increases to 6 m at riffle crests. Pools have a depth of 1 m below the elevation of the riffle crests, and the maximum height of the sinusoidal constrictions adjacent to the pools is 1 m. For both designs, a sequence of five pool-riffle structures was proposed for the project reach.

Evaluation of designs 1 and 2 were performed using HEC-RAS, a modeling package that determines hydraulic conditions for steady, gradually varied flows in open channels through a step-backwater solution of the 1-D energy equation. Inputs to the model include cross sections defining the channel geometry, estimates of flow resistance (Manning's n), and a stage-discharge relation for a downstream boundary condition. The model produces estimates of water-surface profiles, energy losses between the cross sections, and mean velocity and mean bed shear stress at each cross section. Mean velocities (U) are estimated using the Manning equation

$$U = R^{0.67} S^{0.5} / n$$

and the bed shear stress (τ) is computed as

$$\tau = \gamma R S,$$

where γ is the specific weight of water (N m^{-3}), R is the hydraulic radius (m), S is the energy gradient, and n is Manning's resistance coefficient. Cross sections characterizing the five pool-riffle sequences were developed for designs 1 and 2 at a spacing equal to the width of the bottom width (6 m). For both designs, an average Manning's n value of 0.04 was used throughout the reach. The range of test discharges varied from 0.5 to 10 m^3 s^{-1}, where the latter value approximates "ditchfull" flow at the study site. Bankfull discharge for the Copper Slough cannot be accurately determined given that the stream is contained within the bottom of a human-made ditch, and no long-term hydrologic records are available for the site; however, given the small size of the Copper-Slough drainage basin ($\approx$40 km^2), a bankfull discharge representative of natural channel dimensions certainly is much less than the estimated conveyance of the drainage ditch.

Results of the HEC-RAS modeling are expressed as plots of mean velocity and bed shear stress over distance for different discharges (Figure 15). These results for design 1 clearly show that without modification of the lower channel banks, mean velocities and bed shear stresses are much higher over the riffles than in the pools over the entire range

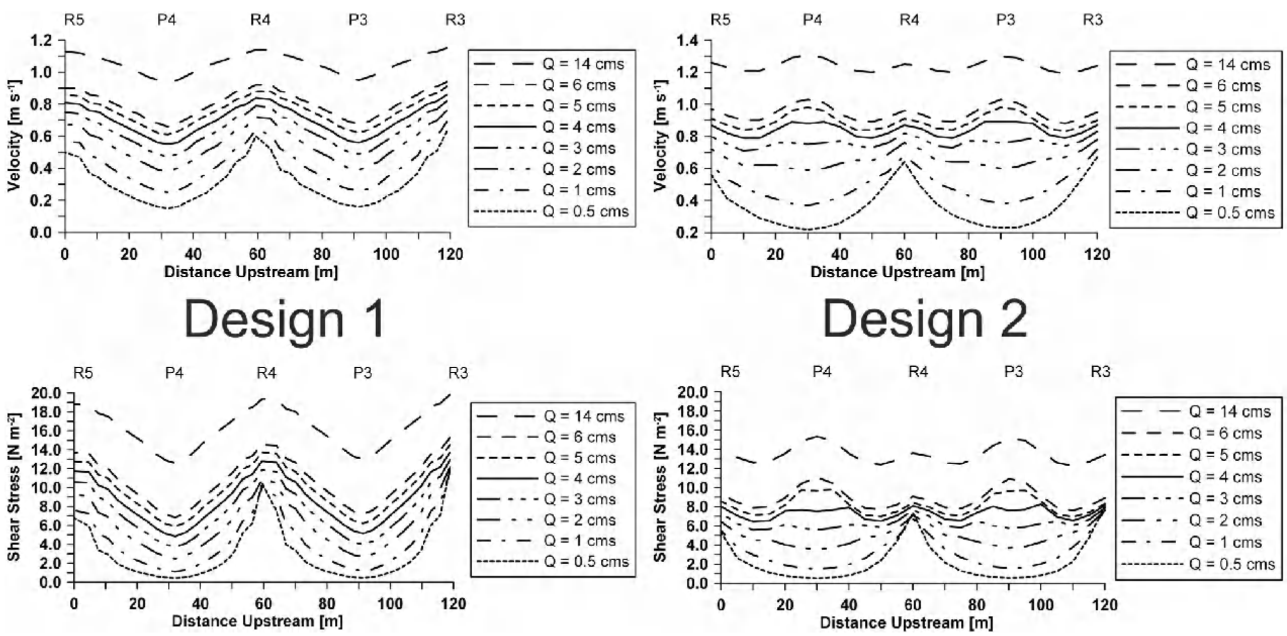

Figure 15. Changes in mean velocity and mean bed shear stress with increasing discharge in pool-riffle units 3 (R3-P3-R4) and 4 (R4-P4-R5) for (left) design 1 and (right) design 2 (flow is from right to left; cms is cubic meters per second).

of modeled discharges. Assuming that sediment-transport capacity is a function of bed shear stress, the patterns of shear stress suggest that the pools will consistently have less transport capacity than the intervening riffles. Thus, sediment transported over the riffles will not be able to move through the pools, resulting in accumulation of sediment at these locations. In other words, the pools will likely infill with sediment.

Results for design 2, which includes modification of the lower banks to constrict flows within the pools, reveal that at low stages, mean velocities and bed shear stresses are higher at riffles than at pools. However, as discharge increases, the pattern reverses, and mean velocities and bed shear stresses are higher in the pools than at the riffles (Figure 15). This reversal occurs for discharges in the range of 4 to 6 $m^3\ s^{-1}$ when the level of the flow is high enough to be strongly influenced by the constriction adjacent to the pool. The water surface elevation through pools three and four for a flow of 5 $m^3\ s^{-1}$ is about 11.75 m (Figure 16), which is close to the elevation of the tops of the constricting structures flanking the pools (Figure 14). By reducing the cross-sectional area of flow within the pool, the constriction (design 2) greatly increases the energy gradient and mean velocity compared to values of these hydraulic metrics without the constriction (design 1) (Figure 17).

To further evaluate the performance of design 2, simulation of 3-D flow through the pool-riffle structures was performed using FLOW-3D 9.3.1 (Flow Science, Inc., Santa Fe, New Mexico), a computational fluid dynamics model that solves the fully 3-D transient Navier-Stokes equations using a finite-volume, finite-difference method in a fixed Eulerian rectangular grid. FLOW3D employs the volume of fluid technique to define free surface boundaries and fluid interfaces [*Hirt and Nichols*, 1981]. It also incorporates different turbulence closure schemes; in this study, the RNG k-ε model was used because past work has shown it provides reliable results for field-scale simulations of flow in rivers [*Abad et al.*, 2008; *Bates et al.*, 2005; *Lane et al.*, 1999; *Rodriguez et al.*, 2004; *Sinha et al.*, 1998]. The computational domain for design 2 consisted of straight, uniform reaches upstream and downstream of a sequence of five consecutive pool-riffle units. The upstream and downstream reaches as well as the first and last pool-riffle sequences were characterized by 225,000 computational grid cells. The three pool-riffle units in the middle of the computational domain were represented by a dense array of 6,300,000 cells.

Results of the simulation for Q = 6 $m^3\ s^{-1}$, a discharge at which the 1-D modeling indicates the mean velocity and shear stress in pools exceed the mean velocity and bed shear stress at the riffles, show that design 2 induces strong convergence of the flow as it moves from the riffles into the pools (Figure 18). Moreover, the pattern of secondary fluid motion indicates that dual surface-convergent helical cells develop within the pool, a finding consistent with modeling of 3-D flow through the pool-riffle units in the WFNBCR [*Rhoads et al.*, 2008]. The helical motion results in near-bed fluid within the pool moving outward, away from the center of the pool, toward the adjacent channel banks (Figure 18).

Together, the results of 1-D and 3-D modeling for design 2 of the Copper Slough naturalization project suggest that changes in hydraulic conditions with increasing stage should be effective at maintaining pool-riffle morphology. As stage increases, the mean velocity and bed shear stress increase at a faster rate in the pools than in the riffles, resulting in a reversal of the magnitudes of these hydraulic metrics between pools and riffles at a discharge in the range of 4 to 6 $m^3\ s^{-1}$. This reversal should prevent sediment from accumulating in the pools. The reversal is induced by the bank constrictions, which will be constructed in such a way as to make these constrictions immobile during the highest flows in the ditch. Thus, the mechanism of reversal will be maintained, even if the pools fill in slightly with fine sediment

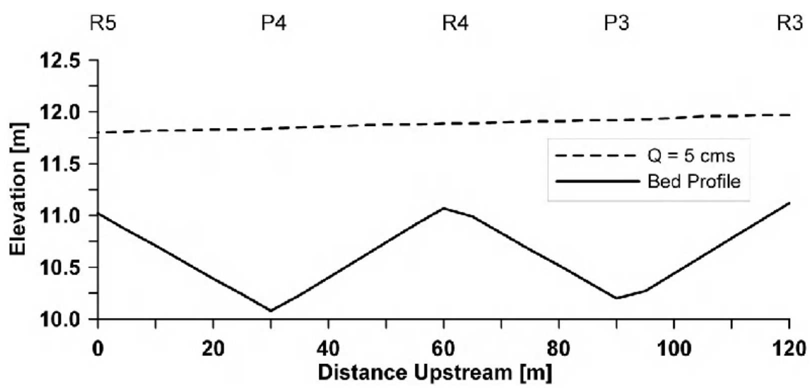

Figure 16. Water surface profile for a discharge of 5 $m^3\ s^{-1}$ through pool-riffle units 3 and 4.

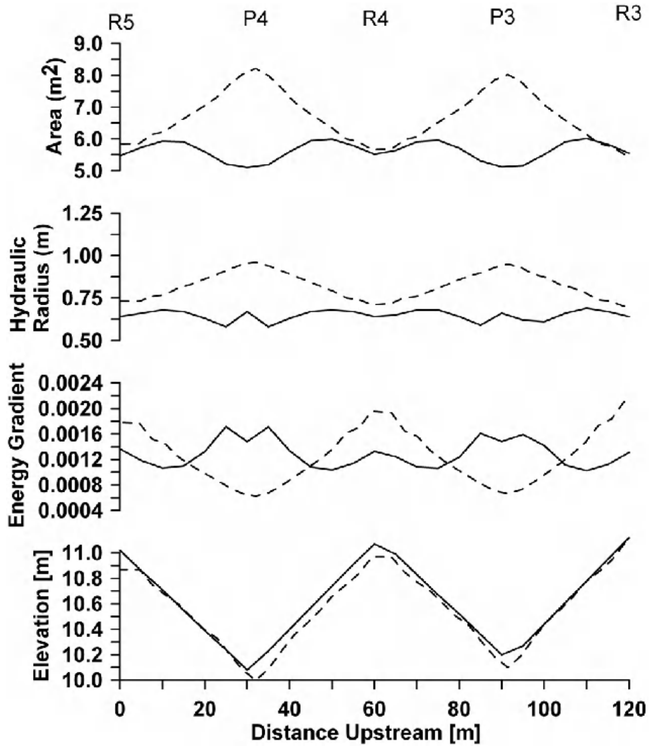

Figure 17. Spatial variation in bed profile, energy gradient, hydraulic radius, and flow cross-sectional area through pool-riffle units 3 and 4 for design 1 (dashed lines) and design 2 (solid lines) at $Q = 5$ m^3 s^{-1}.

prior to the occurrence of a relatively high-stage flushing flow. Any fine sediment that accumulates temporarily during low flows will be removed at high flows when the bed material transport capacity of the pools exceeds the transport capacity of the riffles. This capacity for fine sediment to be removed during high flows is an important aspect of pool maintenance and pool-riffle dynamics [*Jackson and Beschta*, 1982; *Lisle*, 1979; *Rathburn and Wohl*, 2003]. The 1-D hydraulic effects are reinforced by the strong three-dimensionality of the flow. Convergence of the flow into the pools results in the development of dual surface-convergent helical cells. Convergence and helical motion are viewed as important factors influencing the maintenance of forced and unforced natural pools [*Booker et al.*, 2001; *MacWilliams et al.*, 2006; *Thompson and Wohl*, 2009]. In particular, convergence generates convective acceleration of the flow and high levels of turbulence in pools, which enhance sediment mobility, whereas secondary currents tend to route sediment around the flanks of pools, rather than through the center of these zones of scour. Design 2 appears to adequately reproduce important 1- and 3-D hydraulic characteristics of natural pool-riffle sequences.

5. CONCLUSION

Society is increasingly interested in approaches to stream and river management that seek to enhance environmental quality. In many cases, the goal of restoration, or a return to a pristine condition, is not feasible in human-dominated environments, where pristine conditions may be poorly documented; few, if any, pristine reference conditions exist; and the landscape has been modified to such an extent that pristine conditions are most likely unsustainable, even if they could be known. In such environments, outcomes of environmental management will be strongly determined by complex social negotiations among competing interest groups (stakeholders), including scientists and technical experts, and by extant landscape conditions, a process referred to as stream naturalization.

Where streams have been extensively channelized, one approach to stream naturalization is to establish physical habitat defined on the basis of geomorphological principles in human-modified channels. This chapter has examined the problem of establishing pool-riffle units in straight channels. Case studies focusing on an urban channel, an agricultural channel, and a channel at the urban-agriculture interface illustrate that several factors need to be considered in designing pool-riffle sequences for straight channels.

First, the examples here all focus on situations where remeandering is prohibited by cost or by extant infrastructure adjacent to the channel, or is viewed as undesirable (e.g., the need to maintain adequate tile drainage). Thus, channel stability is a primary concern in that implementation of pool-riffle structures should not initiate meandering of the thalweg and systematic erosion of the channel banks. Because pool-riffle sequences in undisturbed streams typically are associated with meandering thalwegs (low-sinuosity channels) or meandering channels (high-sinuosity channels), geomorphological forms and functions can be adapted to, but not necessarily replicated in, straight channels where the planform alignment must be maintained.

Second, the sediment transport regime of the system, particularly the abundance of mobile bed material load, needs to be carefully considered in approaches to naturalization. Despite the importance of this issue, accurate assessments of the influence of sediment transport on project performance is limited by a variety of factors including unavailability of data on characteristics of bed material load, cost constraints associated with the collection and analysis of original data, and high levels of uncertainty in estimating sediment transport rates from predictive equations. The case studies here show that pools and riffles implemented in low-gradient channels with limited amounts of bed material (e.g., WFNBCR) are less likely to encounter problems with self-maintenance

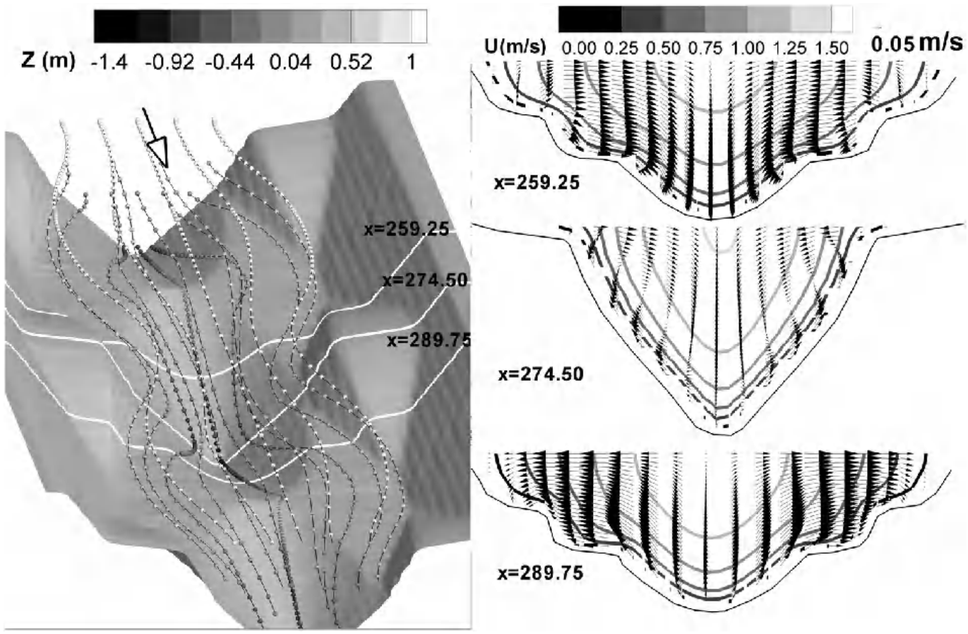

Figure 18. Three-dimensional patterns of flow through pool-riffle units in design 2 for $Q = 6\ m^3\ s^{-1}$. (left) Patterns of converging and diverging streamlines. (right) Three cross sections (shown in white on left) through the pool showing pattern of surface-convergent and bed-divergent secondary flow indicative of dual helical cells.

compared to those implemented in channels with abundant amounts of highly mobile bed material (e.g., Embarras River).

Concern about the sediment-transport regime raises a third issue, which is the main focus of this chapter: the importance of establishing adequate hydraulic conditions to promote self-maintenance of the pool-riffle structures within the context of the extant transport regime. In the relatively fine-grained streams examined in the case studies, a wide range of flows is competent to mobilize bed material. Under such conditions, maintenance of bed morphology depends mainly on the spatial pattern of transport capacity and how this pattern changes across a range of flow events. Downstream increases in transport capacity will result in erosion, whereas downstream decreases in capacity will result in deposition. This study has shown how 1-D modeling of spatial variation in hydraulic metrics, such as mean velocity and bed shear stress, can be used to assess spatial variation in transport capacity and whether these patterns are consistent with maintenance of the desired bed morphology. In pool-riffle sequences, reversals in the magnitudes of mean velocity and bed shear stress with increasing discharge indicate that sediment transport capacity in pools exceeds the capacity in riffles at moderate to high stages, which should flush accumulated bed material out of the pools and maintain scour. Results of 1- and 3-D modeling confirm that forced convergence of flow is a critical factor for accelerating fluid into the pool, for increasing the bed shear stress, and for generating patterns of helical motion that inhibit accumulation of sediment in the center of the pool.

The need for forced convergence of the flow raises a fourth issue in the naturalization of straightened channels: the requisite degree of morphological adjustability. Natural pool-riffle sequences develop through unconstrained process interactions among flow, sediment transport and bed morphology. In contrast, the simultaneous need in naturalization projects to preserve channel stability while also creating a fluvial environment that is self-sustaining often necessitates constraints on morphological adjustability. Design components that induce convergence of the flow into the pools should be comprised of coarse immovable material, thereby protecting the base of the banks from erosive forces and ensuring that forced convergence will be maintained to promote scouring of pools, even if the pools partially fill with sediment.

The approaches to naturalization of straight channels presented in this chapter are not necessarily appropriate for all channelized streams. Environmental context is critically important in determining an appropriate naturalization scheme for a particular river [*Simon et al.*, 2007]. Many channels respond to human modification through an evolutionary series of adjustments involving incision, widening, and stabilization [*Simon*, 1989]. The pool-riffle structures described here were developed for relatively low-gradient (<0.01 m m^{-1}) headwater channels that are not undergoing active erosional adjustments and are either inappropriate for actively evolving channelized streams or will require substantial modification to accommodate the excess erosional energy in such streams.

Future analysis will focus on using computational approaches to evaluate the sensitivity of hydraulic conditions in the basic pool-riffle design for straight channels (design 2) to variations in the configuration of specific design elements. In particular, additional 3-D simulations will be conducted to examine in detail the spatial patterns of boundary shear stress through the pool-riffle units as discharge, pool-riffle spacing, the asymmetry of pool entrance and exits slopes, and the degree of flow constriction through the pool vary. In addition, models capable of simulating bed material transport and evolution of the channel bed will be employed to explore directly patterns of erosion and deposition in relation to variations in discharge. Finally, it is anticipated that the Copper Slough project based on design 2 will be implemented sometime in 2012, providing an opportunity to evaluate the performance of the design directly through a systematic field measurement campaign.

Finally, most naturalization and restoration projects ultimately are aimed at achieving ecological objectives, especially enhancement of fish or macroinvertebrate communities [*Palmer et al.*, 2005]. The geomorphological approach to naturalization highlighted in this chapter focused mainly on improvement of physical habitat at the scale of individual projects extending over short reaches of human-modified streams. The effectiveness of this approach has been questioned, especially in contexts where other factors such as source areas for organisms, connectivity to source areas, water quality, invasive species, and altered hydrological regimes/sediment loads are important, and these factors may impede attainment of ecological goals through improvement of physical habitat alone [*Palmer et al.*, 2010]. In such cases, multiscale naturalization efforts focusing on entire watersheds will be necessary through interdisciplinary consideration of ecological, geomorphological, hydrological, biogeochemical, and social/political/economic factors.

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Controlling Debris at Bridges

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Woody debris from upstream areas of wooded watersheds is often transported to streams during heavy rainfall events. If this debris accumulates on bridge piers, the result can be increased erosion around the bridge foundations, flooding, and loading on the bridge structure. There are a variety of structural and nonstructural methods that are available for use in managing debris at existing bridges. However, the effectiveness of these measures has been limited, and their use is fraught with uncertainty. Application and design of debris countermeasures is largely based on engineering judgment and experience. Thus, as part of a larger restoration or stream management program, knowledge and guidance regarding which methods are applicable and cost effective under a range of bridge and stream conditions, uncertainty, and functionality of the countermeasures are needed. Given the uncertainty and lack of success in managing debris accumulation at bridges and the potentially significant impacts on bridge safety, controlling debris production and transport upstream of the bridge through stream and watershed restoration may be an important management alternative.

1. INTRODUCTION

More than 60% of bridge failures in the United States are caused by water flowing around the bridge foundations, including flooding, scour, and debris [*Wardhana and Hadipriono*, 2003]. Debris accumulation is a significant problem at bridges because it tends to exacerbate both flooding and scour around the bridge foundations, as well as loading on the structure [*Parola et al.*, 2000; *Lagasse et al.*, 2010]. Woody debris from upstream areas of forested or wooded watersheds is often transported to streams during heavy rainfall events. If the debris reaches a bridge pier, it may catch and accumulate on the pier, effectively narrowing the waterway opening. As debris continues to accumulate during subsequent high-water events, problems of flooding, scour, and loading on the pier are often intensified. In some cases, the accumulated debris can block most or all of an entire span.

Chang and Shen [1979] determined a regional distribution of the severity of debris problems at bridges (see Figure 1) based on surveys of highway departments across the country. They found that the most severe problems were in the Pacific Northwest and the Mississippi River valley. More recently, *Lagasse et al.* [2010] conducted extensive surveys and site visits across the country to provide input for developing guidelines to predict the size and geometry of debris accumulations at bridge piers and to quantify scour at piers when debris is present. Their assessment of debris accumulation observations at bridges across geographic regions of the United States showed a similar distribution to that determined by *Chang and Shen* [1979] and found that unstable upstream banks were the primary source of debris at bridges.

The most common solution for dealing with debris accumulation at bridges is regular removal of the debris. Considerable maintenance costs and effort, as well as disruptions to the aquatic system, can be associated with the continual removal of debris from the bridge piers. Other options for

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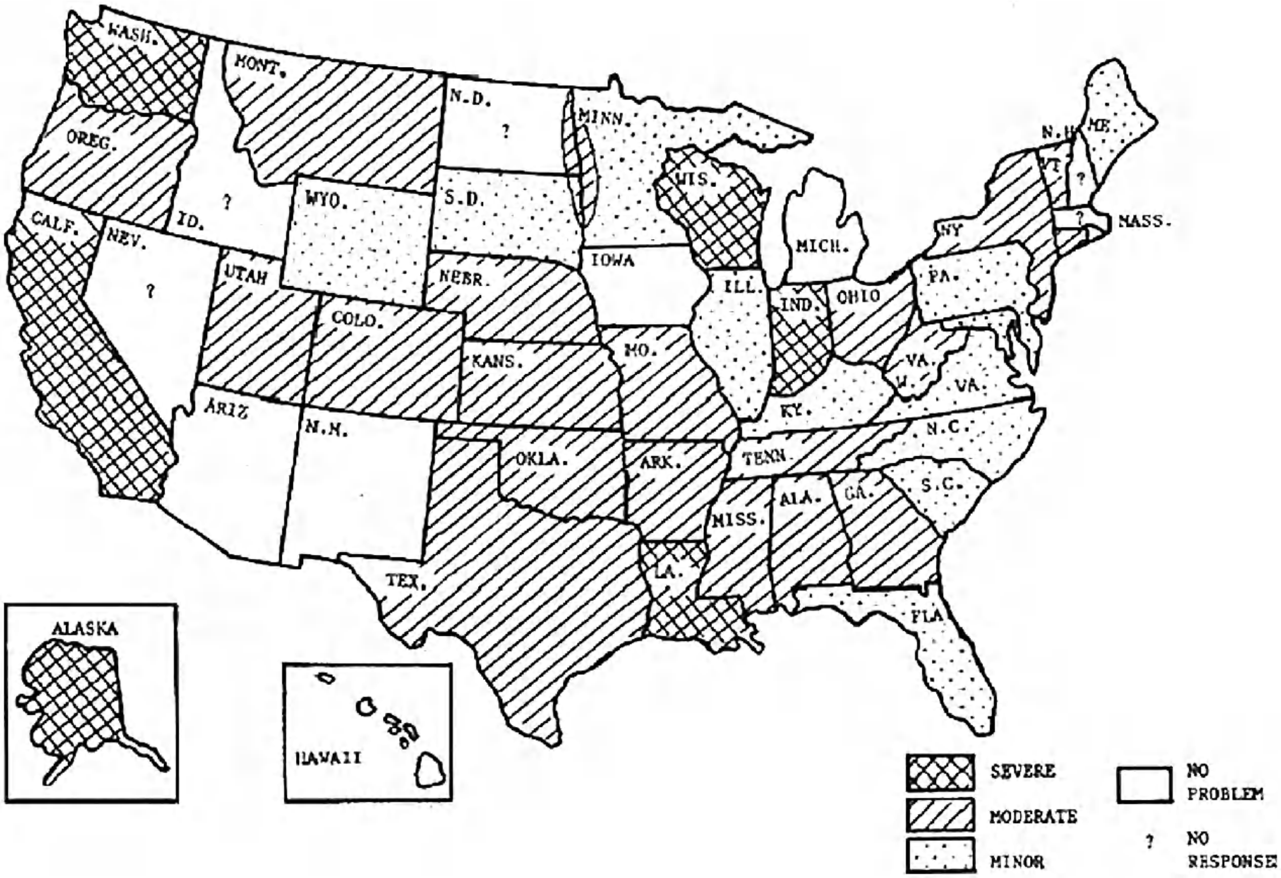

Figure 1. Debris problem distribution [after *Chang and Shen*, 1979; *Lagasse et al.*, 2010].

maintaining the waterway openings at bridges include structural countermeasures, such as deflectors, and nonstructural measures, such as channel modifications. These methods are typically applied only in the bridge right-of-way; the right-of-way length varies but typically does not exceed several bridge widths upstream and downstream of the bridge; this short length is often insufficient to affect a sustainable solution.

Given the uncertainty and lack of success in managing debris accumulations at bridges and the potentially significant impacts of debris in inducing scour and flooding at bridge foundations, and thus unsafe conditions, controlling debris production and transport upstream of the bridge may be an important management alternative. In this chapter, we present the current practices in controlling debris at bridges, limitations in current approaches, and the potential for restoration practices to assist in debris management. Although bridge replacements or alterations that would allow for debris to be more readily transported through the bridge openings may be desirable, this option is not considered here.

2. WOODY DEBRIS TRANSPORT AND ACCUMULATION AT BRIDGES

2.1. Characteristics of Supply, Transport, and Accumulation

At existing bridges, debris conditions are typically known, as the transportation departments have identified maintenance problems regarding the accumulation of debris over the life of the bridge [*Bradley et al.*, 2005]. However, it may be desirable to estimate the potential for additional debris transport and accumulation during higher flows. Such an understanding could result in watershed or stream system level approaches to reduce the amount of debris that enters the stream, rather than controlling it at the bridge. A summary of the state of the art in assessing debris supply, transport, characteristics, and accumulation is provided by *Bradley et al.* [2005] and *Lyn et al.* [2006] and is summarized below based on those documents and additional literature.

Factors affecting the accumulation of debris at bridges fall into three primary categories.

2.1.1. Stream and watershed characteristics influencing debris supply. The following are seven characteristics that influence the debris supply.

2.1.1.1. Alteration of runoff. As land use changes, the hydrologic response may intensify resulting in higher peak flows [*Hollis*, 1975; *Booth*, 1990; *Barker et al.*, 1991; *Booth*, 1991].

2.1.1.2. Stream order. First- and second-order streams rarely transport large debris, whereas third- and fourth-order streams commonly do transport large debris that may become lodged at bridges [*Diehl*, 1997].

2.1.1.3. Channel confinement. The degree to which a channel is confined, or entrenched, is related to stream bank stability [*Diehl*, 1997].

2.1.1.4. Channel sinuosity. Meandering streams that tend to have a high rate of lateral movement may induce bank erosion and introduction of debris to streams [*Chang and Shen*, 1979; *Gilje*, 1979; *Lagasse et al.*, 2001; *Pangallo et al.*, 2002; *Lyn et al.*, 2006].

2.1.1.5. Stream bank stability. Stream bank erosion is a dominant factor in the introduction of debris to the river environment [*Diehl*, 1997; *Murphy and Koski*, 1989; *Diehl and Bryan*, 1993].

2.1.1.6. Stream restoration and mitigation. In some regions, large wood placement is a common mitigation practice for stream-related impacts. In addition, direct placement and recruitment of large wood into stream systems is often a primary goal of many stream-restoration projects.

2.1.1.7. Logging and other watershed disturbances. Such disturbances are a major source of debris to streams [*Chang and Shen*, 1979; *Bryant*, 1980, 1983; *Lagasse et al.*, 2001].

2.1.2. Stream characteristics influencing debris transport. Three influential characteristics for debris transport are the following:

2.1.2.1. Stream bed irregularities. Irregularities that affect debris transport include bridge-related channel modifications with a widened (shallow) section upstream and at the bridge, as well as bars and sediment accumulation upstream of bridge piers [*Hickin*, 1984; *Wallace and Benke*, 1984; *Abbe and Montgomery*, 1996; *Lyn et al.*, 2006; *Newlin*, 2007].

2.1.2.2. Bank irregularities. Eddies provide dead zones for debris to catch and collect, including at widened sections at bridges [*Diehl*, 1997; *Newlin*, 2007].

2.1.2.3. Upstream infrastructure. Bridges, culverts, and abandoned infrastructure upstream from the bridge may be locations for debris to collect, causing larger amounts of debris to move downstream toward the bridge during subsequent higher flows [*Lyn et al.*, 2006].

2.1.3. Bridge characteristics affecting debris accumulation. The following are four characteristics of bridges that affect the accumulation of debris.

2.1.3.1. Span length. If the debris is longer than the span between adjacent piers or piers and abutments, the debris may become lodged between them [*Diehl*, 1997].

2.1.3.2. Pier placement. Debris accumulation will occur with the highest frequency and greatest severity at locations where bridge piers and abutments are located in the path of floating debris [*Gilje*, 1979; *Chang and Shen*, 1979; *Diehl*, 1997; *Bradley et al.*, 2005; *Lyn et al.*, 2006].

2.1.3.3. Pier shape and alignment. In general, solid, round-nosed piers, aligned parallel with stream flow decrease the likelihood of debris accumulation [*Chang and Shen*, 1979; *Lagasse et al.*, 2001; *Diehl*, 1997]. Open pile designs are likely to trap more debris than alternative configurations, since debris can become wedged in between the support members.

2.1.3.4. Bridge deck height and design. Bridge deck height can influence debris accumulation if flood waters approach the bottom of the superstructure. If the vertical gap between the low chord of the bridge and the stream bed is less than the length of the debris member, debris can become lodged between the bridge superstructure and the stream bed, effectively decreasing span lengths and leading to accelerated accumulation of debris at the bridge site [*Diehl*, 1997; *Bradley et al.*, 2005].

Of all the factors listed above for debris supply, the most important sources of debris identified through a study in northern Mississippi [*Wallerstein et al.*, 1997; *Wallerstein and Thorne*, 2004] were outer bank erosion in channel bends and bank mass wasting in degrading reaches. These two sources accounted for approximately 75% of all debris entering the streams in the studied system. Note that these sources are typically well outside of the departments of transportation (DOTs) right-of-way for a given bridge. Other identified sources included windthrow, preserved debris from older deposits, float from upstream, and beaver dams.

2.2. Field Observations and Case Studies

Few studies have been conducted to monitor debris accumulation and the effectiveness of countermeasures at bridges. In particular, no studies have directly related flow conditions to the accumulation of debris at bridges or the effectiveness of countermeasures under various flow conditions. *Lyn et al.* [2003] conducted a limited monitoring program at two bridges in Indiana over a 1.5 year period. Both of these bridges had debris deflectors installed upstream, and the purpose of the monitoring study was to determine the effectiveness of the deflectors particularly at different flow depths, as this factor had been found to be significant as a contributor to debris accumulation at piers during a laboratory study.

(a)

(b)

Figure 2. Debris accumulated (a) at debris deflectors upstream of bridge piers and (b) at the SR59 crossing of the Eel River, Indiana [after *Lyn et al.*, 2003].

Table 1. Summary of Debris Accumulation Countermeasure Activities[a]

State	Affiliation	Response
Wisconsin	Wisconsin DOT	Wisconsin had a second-generation debris sweeper installed; it failed due to a bent support structure and was removed from the bridge.
Ohio	Ohio DOT	Installed a debris deflector upstream of a large box culvert. Effective for a few years until erosion caused the pile to fail.
New York	FHWA, New York	No debris countermeasures currently installed or planned.
Tennessee	Tennessee DOT	There are several debris sweeper installations within the state. The installations are functioning correctly. Debris fins have been used on culverts in Tennessee.
Maryland	Maryland State Highway	No debris countermeasures currently installed or planned.
Virginia	Virginia DOT	Provided a list of 12 sites where chronic debris accumulation is a problem. Generation one and two debris sweepers were installed at three of these sites but were unsuccessful due to failure of pier mounting system.
Vermont	Vermont DOT	No debris countermeasures currently installed or planned.
Indiana	Indiana DOT	There are many sites in the state with chronic debris accumulation problems. Debris countermeasures exist at several of these sites.
Delaware	FHWA, Delaware	No debris countermeasures currently installed or planned.

[a]Information is based on responses from Federal Highway Administration and state DOT personnel.

Figure 2 shows debris accumulated at both the pier and the deflector prior to the start of the monitoring program. The deflector appeared to collect debris, but had limited effectiveness in deflecting debris.

Throughout the United States, individual state DOTs have installed countermeasures to alleviate the effects of debris accumulation at bridges. Their experiences are critical to determining applicability, reliability, uncertainty, and sustainability of the various countermeasures. *Bradley et al.* [2005], *Lyn et al.* [2003], and *Lyn et al.* [2006] provide observations and survey results. More recently, *Lagasse et al.* [2010] gathered those data as well as more recent observations and photos from across the country, primarily through surveying the state DOTs and conducting numerous

Table 2. Summary of Site Observations

Bridge	Channel Widening at Bridge (%)	Bars/Deposits	Pier Type	Location and Size of Debris	Estimated Span Width (ft)
		Pennsylvania			
PA 103 over Juniata River	none	none	solid, round nosed	pier 3 from left bank; S-M	50
PA 3019 over Clearfield Creek	150	at bridge in widened section; mod. width; gravel	solid, round nosed	M-L debris; not centered on pier	45
PA 3012 over Clearfield Creek	100	at bridge in widened section; mod. width; gravel	solid, round nosed	at pier; S-M-L	35
PA 153 over Bennett Branch	90	at bridge in widened section; mod. width; gravel	solid, round nosed	at pier; S-M-L	40
PA 3017 over French Creek	none	none	solid, round nosed	at pier; S, L	110
US Route 6 over French Creek (PA)	none	none	single, round columns	upstream at left bank; mostly L	115
PA 2013 over East Sandy Creek	180	none	solid, round nosed	none	50
PA 588 over Connoquenessing Creek	none	gravel/cobble bars in vicinity of bridge	solid, round nosed	none	70
PA 3031 over Chartiers Creek	none	none	solid, round nosed	none	70
PA 351 over Beaver River	none	none	solid, rip rap around base	at pier closest to right bank; S-M-L	85
		Virginia			
US Route 360 eastbound and westbound over Staunton River	none	sediment deposition across width of channel, very shallow in vicinity of bridge	solid, round nosed	at piers; M-L	70
VA 45 over James River	none	none	solid, round nosed	at piers; M-L	115
VA 58 eastbound over Hyco River	none	none	single, round columns; square, columns	span-blocking accumulation, between piers; S-M-L	55
VA 716 over Banister River	none	none	multiple, exposed pier columns	at pier; L	45
US Route 29 southbound over Dan River	none	none	solid, round nosed	at piers; S-M-L	120
		Tennessee			
TN 249 bridge over the Harpeth River	none	none	solid, round nosed	at pier; S-M-L	80
TN 250 bridge over Harpeth River	none	point bar, at inside of meander upstream of bridge	solid, round nosed	at pier; M-L	45–60
TN 005 (US 45W) bridge over Obion River	none	none	rounded columns, 2 per pier	at pier, between columns; S-M-L	55–70
TN 007 (US 31) bridge over Elk River	none	none	solid, round nosed	at pier; M-L	50
TN 274 bridge over Elk River	none	midchannel bars	solid, round nosed	at pier; S-M	70

site visits. They provide a list of 142 sites in 31 states. Although the Midwest, Pacific Coast, south, and west are well represented, the east accounted for only nine sites. *Sheeder and Johnson* [2008] contacted state DOT personnel located in additional eastern and Midwestern states, including Wisconsin, Ohio, New York, Tennessee, Maryland, Virginia, Vermont, Indiana, and Delaware, to identify additional locations where debris accumulation is actively being addressed. Table 1 summarizes the responses received from these states. In only four of these nine states had the DOT installed debris countermeasures. The primary management strategy for these states is debris removal.

Photographs and brief descriptions of bridge sites observed or surveyed by *Lagasse et al.* [2010] can be viewed at http://onlinepubs.trb.org/onlinepubs/nchrp/nchrp_rpt_653.pdf. Field observations were conducted by *Sheeder and Johnson* [2008] at 12 bridges in Virginia, 6 bridges in Tennessee, and 7 bridges in Pennsylvania, where state agencies are actively addressing debris accumulation problems at bridges; Table 2 provides a summary of the observations that were made at each bridge. Several common conditions were observed in the field as well as in the literature regarding the accumulation of debris at bridge piers.

1. Riprap is frequently mounded at bridge piers as a countermeasure for scour. Mounded riprap at piers often provides a rough surface for debris to become caught and accumulate.

2. Artificially widened stream sections just upstream of bridges result in lower stream velocities and promote sediment deposition. As these areas become shallower, they provide ideal locations for debris to collect. To concentrate the flow through the widened areas and through the bridge openings, a system of vanes and similar structures can be used to reduce bar formation hence promoting transport of debris through the bridge opening.

3. Where there are stream bank irregularities immediately upstream of the bridge, vanes and similar structures can also promote the stabilization of the banks and eliminate eddies and flow obstructions that tend to promote debris accumulation.

4. Piers designed with solid walls, a rounded pier nose, and pile caps that are below the stream bed elevation are less likely to trap floating debris than nonstreamlined configurations. Thus, for bridges with these less streamlined configurations, in-stream hydraulic structures, particularly submerged (Iowa-type) vanes, can be used to guide the flow and debris around the bridge piers.

3. ESTIMATING SCOUR AT BRIDGES WHERE DEBRIS ACCUMULATES

In order to estimate the effects of debris accumulations on scour at bridge piers, *Lagasse et al.* [2010] conducted site visits across the country to observe debris deposition (described in section 2.1) and scale model experiments to provide data and insight for the development of a method for estimating an equivalent bridge pier width. The scale model laboratory experiments were conducted in medium sand under clear-water conditions (i.e., no transport of bed material upstream of the bridge). A range of shapes and sizes were used for both the bridge pier configurations and the debris mass accumulations. The ratio of the flow velocity to critical velocity for sediment motion was set at either 0.7 or 1.0 (threshold conditions). Flow discharges were varied to provide the desired flow velocity and depth and to determine the scour response.

The equivalent pier width is a function of the shape of the debris pile, the intensity of the plunging flow around the debris, the length of the debris pile, and the depth of flow in the channel for the design discharge. The pier width is a primary factor in determining how deep a scour hole will potentially develop [*Richardson and Davis*, 2001]. For bridge scour calculations, the 100 year discharge is typically the minimum discharge of concern. Of course, debris can be transported and accumulate at much lower discharges; however, bridge foundations are required to withstand at least the 100 year flood. Thus, scour is calculated for only these large events. Based on the experimental evidence, the effective pier width is given by

$$a^* = \frac{K_{d1}(TW)\left(\frac{L}{Y}\right)K_{d2} + (y - K_{d1}T)a}{y} \quad \text{for} \quad L/y > 1.0 \qquad (1)$$

$$a^* = \frac{K_{d1}(TW) + (y - K_{d1}T)a}{y} \quad \text{for} \quad L/y \leq 1.0, \qquad (2)$$

where a^* is equivalent pier width, L is length of debris upstream for pier face (m), y is depth of approach flow (m), T is thickness

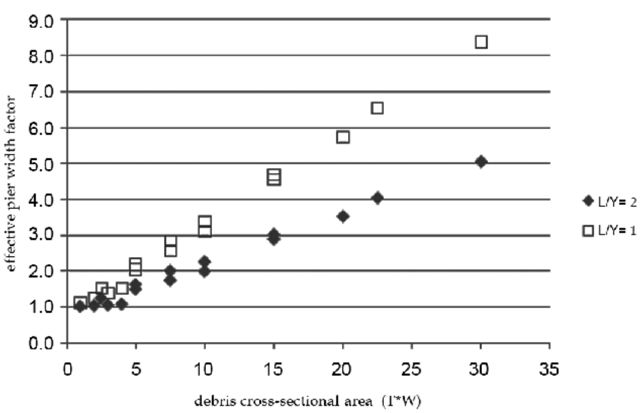

Figure 3. Effective pier width as a function of debris length, thickness and width, and flow depth based on equations (1) and (2).

of debris accumulation (m), W is width of debris accumulation (m), K_{d1} = 0.79 for rectangular debris and 0.21 for triangular debris, K_{d2} = −0.79 for rectangular debris and −0.17 for triangular debris, and a is pier width (m).

The effective pier width accounts for the debris accumulated at the upstream side of the pier. The minimum value for a^* is 1.0; that is, no or little debris is present at the pier. Figure 3 shows a range of values of a^* for a flow depth of 3 m, debris pile length to flow depth ratio (L/Y) of 2.0 for equation (1) and 1.0 for equation (2), and rectangular cross-sectional areas of debris ranging from 1 to 30 m^2. The cross-sectional area is taken as the product of the debris thickness (T) and the debris width (W). In this example, Figure 3 shows that uncertainty in determining the value of L/Y can result in a range of effective pier widths, a^*, which in turn influences the scour calculation. For example, if T^*W = 15, then for L/Y = 1.0, a^* = 4.7 and for L/Y = 2.0, a^* = 3.0. Thus, the effective pier width is either 3.0a (where a is pier width) or 4.7a. These results lead to an estimated scour depth that is 33% different, depending on which a^* is used.

The determination of debris accumulation dimensions, required for equations (1) and (2), are based on engineering judgment. Equations (1) and (2) also assume that worst case debris conditions are present for the 100 year or higher flows. This provides a conservative estimate of scour.

(a)
(b)


Figure 5. Vanes (a) during and (b) following installation on the Wapsipinicon River, Iowa. Figure 5b shows the same bend 2 years after installation [after *Odgaard*, 2005].

4. MANAGING WOODY DEBRIS ACCUMULATIONS AT BRIDGES

Woody debris accumulations at bridges are most commonly managed by simple removal; however, nonstructural approaches or structural countermeasures may be used. Nonstructural approaches typically involve changes to the channel to improve bank stability or channel alignment with the bridge. In terms of bridge maintenance, these methods are often applied close to the bridge, within the bridge right-of-way. Given the close proximity

Figure 4. Debris fin [after *Bradley et al.*, 2005].

to the bridge in which these changes are typically constructed, the methods may be effective in providing improved alignment and channel geometry, which impact the transport path of debris; they are typically not effective in decreasing the amount of debris that is transported to the bridge [*Lagasse et al.*, 2010].

Structures designed to deflect or redirect woody debris so that it travels between piers have been investigated in a number of studies [*Saunders and Oppenheimer*, 1993; *Richardson and Davis*, 2001; *Bradley et al.*, 2005]. Countermeasures include debris fins, vanes and similar structures, debris deflectors, crib structures, and debris sweepers. In a recent survey of state departments of transportation, it was found that the most commonly used structural countermeasures are debris deflectors and fins [*Lagasse et al.*, 2010]. These structures are described elsewhere, so only a brief description of each is given below. Given the recent field testing and improvements in debris sweepers, as well as a high level of interest in these countermeasures from transportation departments, a lengthier discussion is provided on this countermeasure.

4.1. Debris Fins and Angled Debris Walls

Debris fins are walls built in the stream channel immediately upstream of a bridge. They are designed to align debris to pass unimpeded through a bridge opening [*Bradley et al.*, 2005]. Angled debris walls are similar structures that allow the debris to move upward for easier removal (see Figure 4).

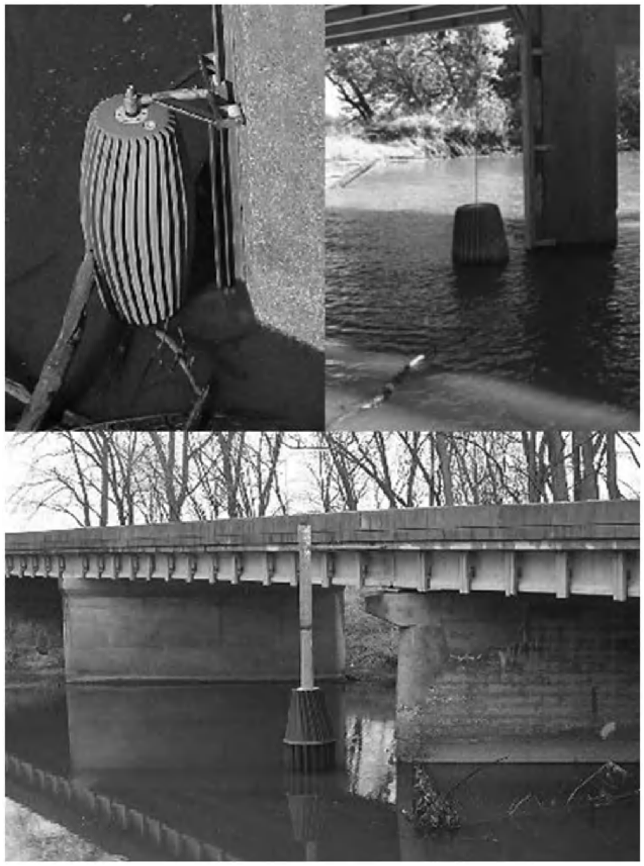

Figure 7. (clockwise from top left) Debris sweepers installed using the first-, second-, and third-generation pier attachment systems [from *Sheeder and Johnson*, 2008; Debris Free, Inc.].

Figure 6. Debris deflectors upstream of bridge [after *Bradley et al.*, 2005].

4.2. Vanes and Similar Structures

Submerged (Iowa) vanes, weirs, and vanes can be used in different configurations to protect eroding stream banks, alter the course of a channel, promote sediment movement, and/or alter flow patterns around infrastructure [*Odgaard and Kennedy*, 1983; *Odgaard and Lee*, 1984; *Odgaard and Mosconi*, 1987; *Lauchlan*, 1999; *Johnson et al.*, 2001, 2002; *Newlin and Johnson*, 2009; *Newlin*, 2007]. Evidence presented by *Diehl* [1997] suggests that secondary flow characteristics influence debris transport pathways. Therefore, it is reasonable to assume that while the exact effects of secondary circulation on debris transport have not been determined, the alteration of these flow paths will affect preferred debris transport pathways (see Figure 5).

Table 3. Benefits, Disadvantages, and Qualitative Cost Ranges for Debris Countermeasures

Option	Benefits	Disadvantages	Annual Cost Over 10 Year Period
Maintenance removal of debris	simple operation	Required over life of bridge; disruptive to aquatic ecosystem; accumulation during flood events may jeopardize bridge, cause scour and flooding	low to moderate
Vanes/weirs	Alleviates obstructions, such as bars and deposits, upstream and at bridge that may be causing debris to become lodged.	Will lead to alteration of sediment transport processes; if designed incorrectly, can cause aggradation or erosion in the vicinity of the bridge. Debris may tend to hang up on structure. Will not likely affect floating debris itself.	low
Debris deflectors	simple design and construction	May trap debris upstream of bridge requiring maintenance removal; known to fail under high lateral forces of trapped debris; failure of piles may jeopardize bridge	low
Debris sweepers	Active system; following installation, may alleviate additional maintenance requirements; requires little disturbance of the stream channel	Failure of system can increase potential for debris accumulation; not appropriate for certain bridge configurations (if span is less than maximum length of debris)	low to moderate
Crib walls, debris fins	May not require stream modification or continuing maintenance	Low reliability; problem may continue; maintenance required	moderate
Reconfiguration or removal of riprap at piers	Provides more effective scour protection as well as debris alleviation	New scour patterns may develop	moderate
Channelization or reconfiguring the channel	Prevent debris and sediment accumulation by increasing rate of flow through steepened reach	Higher velocity could cause scour at bridge; may cause erosion of bed and banks upstream; potential maintenance costs associated with erosion; not sustainable if river system is unstable	high
Removal of unneeded upstream infrastructure or other obstructions	Eliminates obstructions that may be causing sediment and/or debris-related problems in the vicinity of the bridge	Relief may be temporary; removal of obstructions may create new flow patterns, which may lead to additional sediment and/or debris-related problems	moderate to high

4.3. Crib Structures

Crib structures are walls built parallel to streamflow between exposed piles to prevent debris from becoming trapped between the piles [*Bradley et al.*, 2005].

4.4. Debris Deflectors

Debris deflectors are sacrificial piles placed upstream of bridge piers to deflect debris away from the bridge pier and guide the debris through the bridge opening [*Bradley et al.*, 2005]. While cylindrical pile debris deflectors have been widely used throughout the United States, their effectiveness as a debris accumulation countermeasure is questionable, and they may exacerbate the problem under certain conditions [*Brice et al.*, 1978; *Lyn et al.*, 2006] (see Figure 6).

4.5. Debris Sweepers

Debris sweepers are polyethylene devices that rotate around a vertical axis, under the force of flowing water. The debris sweeper device is designed and marketed by Debris Free, Inc., a California-based company specializing in bridge maintenance solutions. The rotating structure floats at the water surface and travels up and down the vertical axis as water levels rise and fall. Floating debris is realigned and guided through the bridge opening by the vortices surrounding the rotating sweeper. Three revisions have been made to the debris

Table 4. Descriptions of Categories for Channel and Bridge Characteristics

Situation	N (Not Significant)	M (Moderately Significant)	S (Significant)
Channel widening at bridge	Channel width at bridge has not been modified to accommodate flow; channel width at the bridge is approximately the same at the bridge as the average width in the channel reach upstream of the bridge.	Minor to moderate expansion of channel width at the bridge to accommodate. Expanded width is not more than 120% of the average upstream width [*Newlin*, 2007].	Significant expansion of channel width at the bridge to accommodate flow beneath the bridge. Expanded width is more than 120% of the average upstream width [*Newlin*, 2007].
Channel obstructions	No evidence of sediment bars at upstream side of bridge opening; no exposed pile caps at channel invert; no riprap at piers or riprap placed below channel invert.	Small to moderate sediment deposits or bars exist at upstream side of bridge opening; riprap is slightly mounded at base of pier(s).	Large sediment deposits or bars exist at upstream side of bridge opening; riprap is mounded at base of pier(s); a pile cap is exposed at the base of one or more piers.
Bridge/ stream configuration	Solid, round-nosed piers aligned parallel with stream flow; debris is rarely or never longer than the span between adjacent piers or piers and abutments	Piers are solid, but blunt nosed; aligned parallel with stream flow; debris is occasionally longer than the span between adjacent piers or piers and abutments	Piers are blunt nosed or pile groups exist, and not aligned with flow; debris is frequently longer than the span between adjacent piers or piers and abutments.

sweeper since its inception. Figure 7 shows debris sweepers installed using first-, second-, and third-generation pier attachment technologies. In each of these product revisions, the structure used to support the rotating polyethylene cylinder was modified to address design shortcomings identified during field tests. In the first two generations, the sweeper was attached directly to the bridge pier. In generation one, the sweepers were mounted directly to the bridge pier using a variety of bracket designs that slid up and down the pier nose with rising and receding water levels. Generation two improved upon this design by installing brackets on the top and bottom of the pier, then using a tensioned cable to secure the sweeper while allowing the device to travel up and down the cable. The third and current design involves driving a pile into the streambed to support the sweeper. While not required, the top of the pile can be secured to the bridge pier or superstructure if feasible, based on the location of the pile in relation to the bridge (see Figure 7).

Table 5. Applicability of Countermeasures as a Function of Channel and Bridge Characteristics[a]

Countermeasure	Channel Widening at Bridge	Channel Obstructions	Bridge/Stream Configuration
Debris sweeper (third generation)	N, M	N	N
Deflectors	N, M, S	N	N, M, S
Fins	N, M	N	N, M
Crib structures	N, M, S	N, M, S	M, S
Vanes, weirs	M, S	M, S	M, S
Annual and emergency maintenance	N, M, S	N, M, S	N, M, S

[a]See channel and bridge characteristics in Table 4. Each countermeasure is applicable (but not necessarily effective) for the conditions listed. Effectiveness is addressed in Table 6.

Bradley et al. [2005] identified four states where debris sweepers had been installed at that time: Oklahoma, Virginia, Tennessee, and Oregon. *Lyn et al.* [2006] investigated several additional locations in Indiana where debris sweepers were installed using the generation one and two mounting systems. Several of these structures had failed and had been removed at the time of the study. Others were still functional; however, the data collected by *Lyn et al.* [2006] at these sites were insufficient to draw conclusions regarding the factors influencing success or failure of the structures. Additional information on the debris sweeper design and failure rate statistics was provided by Debris Free, Inc. (C. Collier, personal communication, 11 January 2008). First-generation debris sweeper installations began in 2001. At present, the majority of these installations have failed. Approximately 50% of the second-generation installations, which began in 2003, are still operable. Several of these installations have survived several major hurricanes and are, therefore, expected to remain functional through lesser events. Debris Free began installation of the third-generation devices in 2005. The more robust pile support application appears to have eliminated many of the failures previously caused by insufficient support design. To date, only 10% of the installed third-generation structures have failed, and several of the failures that have occurred can be attributed to improper installation.

Based on the literature review, transportation personnel queries, and field observations, the benefits, disadvantages, and relative costs of debris countermeasures were determined, as given in Table 3. This table includes vanes and weir structures. Although these are not considered to be exclusively debris countermeasures, they do assist in creating conditions to move debris downstream away from piers and so are included here.

Table 4 provides qualitative descriptions of three primary categories of factors affecting debris accumulation at existing bridge piers with an associated significance of each factor. These three factors are combined from the literature review provided in section 2.1 for debris accumulation at bridges, as well as those provided by observations made by *Sheeder and Johnson* [2008] in section 2.2. Based on Table 4, Table 5 lists the applicability of each selected countermeasure. Again, this includes vanes and similar structures, even though they are not directly considered to be debris countermeasures, but rather important structures to alleviate conditions that may cause debris accumulation. In addition to bridge and stream characteristics, other factors are also important in the effectiveness and final selection of countermeasures, including costs (from Table 3), environmental impact, maintenance needs, reliability, aesthetics, and uncertainty in the functionality of the countermeasures. Table 6 provides relative ratings or applicability for each of these factors. Factors, including aesthetics, environmental impacts, and maintenance needs are based largely on HEC-9 [*Bradley et al.*, 2005], with modifications based on other literature reviewed above, field observations, discussions with transportation officials, and information obtained from Debris Free, Inc. Uncertainty and longevity (or sustainability) are primarily based on the survey of states conducted by *Bradley et al.* [2005], which resulted in a listing of countermeasures used in various states (i.e., countermeasure experience), as well as findings by *Lyn et al.* [2006], field observations conducted in this study, discussions with transportation personnel, and information obtained from Debris Free, Inc.

5. THE CASE FOR STREAM RESTORATION IN MANAGING DEBRIS AT BRIDGES

The estimation of the amount and dimensions of debris that will accumulate at bridges, as well as the design and effectiveness of the various debris management countermeasures and approaches, are sources of considerable uncertainty. Expected debris dimensions are based on experience, past observations, and judgment. As discussed previously, this uncertainty can lead to uncertainty in the effect of the debris on scour at the design flood. Structural countermeasures have been met with limited success. Although there are guidelines available for the applicability of the various countermeasures, design guidelines for this purpose are vague and are not directly related to the design discharge or accompanying sediment conditions. Thus, there remains significant uncertainty in how effective these countermeasures will be during a large flood in which debris is transported to the bridge or has already accumulated at the bridge.

In many cases, the best solution would be replacement or modification of the bridge substructure to permit debris passage through the bridge opening. Unfortunately, funds do not exist to replace or modify every bridge where debris is an issue. In most cases, the DOT focus will be, at least initially, on effective countermeasures that will reduce the costs and efforts of ongoing maintenance and will more effectively prevent debris accumulation during various flow events. Removal of debris after it has accumulated, while a common maintenance practice, is often not effective in

Table 6. Factors Influencing Effectiveness and Final Selection of Countermeasures[a]

Countermeasure	Debris Size			Maintenance Needs	Longevity/ Sustainability	Aesthetics	Environmental Impact	Performance Uncertainty	Installation Costs
	Small	Medium	Large						
Debris sweeper (third generation)		✓	✓	L	M	A	L	L-M	M-H
Deflectors		✓	✓	M-H	L-M	U	L	M-H	M
Fins		✓	✓	M-H	M	A	L	M-H	M
Crib structures		✓	✓	M-H	M	U	L	M	M
Vanes, weirs	✓	✓	✓	L	M-H	A	M	M	M
Annual and emergency maintenance	✓	✓	✓	H	L	U	M	M	L-M

[a]Modified from *Bradley et al.* [2005]. Abbreviations are as follows: L, low; M, moderate; H, high; A, acceptable; U, unacceptable.

preventing the problems associated with debris accumulation, such as flooding, scour, and loading, since these processes are active during the high flows while the debris is in place or accumulating.

Given the high level of uncertainty related to the prediction of the impacts of debris that most debris is supplied by unstable channel banks (as described previously), then the most effective practices to alleviate debris accumulation are likely those that involve modification of the stream channel or watershed characteristics upstream of the bridge. Such management and restoration practices require strong communication between state departments of transportation, federal and state permitting agencies, and restoration designers.

6. CONCLUSION

The accumulation of woody debris from upstream areas is an ongoing problem at many existing bridges. Given the expense of retrofitting a bridge substructure for debris problems and the continual maintenance issues associated with debris removal, it is important for stream restoration practitioners and land use managers to have a sense of the impact of debris at bridges as well as methods for reducing that impact. This chapter provides the state of the art in countermeasure applications for woody debris accumulation at existing bridges. The FHWA manual, HEC-9 [*Bradley et al.*, 2005] provided an excellent summary of the processes involved for the purpose of estimating debris size and accumulation tendencies, primarily for designing new bridges. Since that time, additional experiences in the field and advances in selected countermeasures have occurred and are addressed in this chapter. The relative uncertainty in the use of debris countermeasures was summarized in this chapter based on experiences provided by transportation personnel. The tables constructed herein reflect the combined experiences and observations made by numerous researchers and are a means for selecting countermeasures based on the best available information. In addition to the constraints and applications listed in Tables 3–6, other limitations at the site, such as access to the bridge, should also be considered in countermeasure selection.

Site observations and engineering experience show that the ability to manage debris at bridges or within the bridge right-of-way is often quite limited. Guidelines for the design of countermeasures are vague, and qualitative relationships between debris accumulation, flow discharge, and sediment discharge have not been developed. Case studies that could provide longer-term data and evidence for proper management of debris at bridges and effectiveness of structural or nonstructural countermeasures do not exist. Thus, currently the best solution for debris management at bridges lies within stream restoration practices, particularly repair or restoration of unstable banks. However, in order for stream restoration to be an effective solution, practitioners, regulatory agencies, and funding agencies need to be aware of the problems at bridges related to debris, and the possible options for management. A willingness to cooperate across transportation, environmental, and regulatory agencies is a critical part of the successful application.

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Seeing the Forest and the Trees: Wood in Stream Restoration in the Colorado Front Range, United States

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This chapter reviews the processes that influence wood dynamics in mountain streams: recruitment, storage, and transport. These processes are incorporated within the numerical value of wood load or volume of wood per area of stream. Spatial variations in wood load within a stream can be substantial as a result of spatial and temporal variations in the processes that influence wood dynamics. Such variation makes it challenging to define either the historical range of variation in wood loads or targets for restoring in-stream wood loads. Considering wood dynamics in the context of geomorphic setting, as delineated in process domains, helps to constrain the relative importance of individual processes influencing wood dynamics, as well as spatial variations in wood load. Taking the Colorado Front Range as a case study, information from reference sites, regional data, and mechanistic models is used to illustrate how partial information from multiple sources can be assembled to estimate historical range of variation in wood loads and to develop targets for restoration of in-stream wood. Although the details will vary between regions, this approach should be applicable to any mountain stream network.

1. INTRODUCTION

One of the challenges of stream restoration is to define the conditions to which a stream is to be restored. Because restoration attempts to return an ecosystem to its historic trajectory, historic conditions are the ideal starting point for restoration design [*Society for Ecological Restoration International Science and Policy Working Group*, 2004] (accessed 19 April 2010). If land use has altered a portion of a stream, or the entire channel, a reference stream with similar control variables such as geology, climate, and basin physiography can be used to characterize historic or desirable conditions for the altered stream [*Fox et al.*, 2003; *Brierley and Fryirs*, 2005]. This approach is limited, however, where entire stream networks have been highly altered and where reference sites are limited or nonexistent. The high spatial and temporal variability of process and form in many streams also makes it problematic to use a "snapshot" of reference stream characteristics over limited time and space to define desired conditions for stream restoration [*Hughes et al.*, 2005; *Jaquette et al.*, 2005; *McAllister*, 2008] (Figure 1). Perhaps most importantly, it may be impractical to strictly conform to reference conditions in a future of changing climate and increased land use [*Pierce et al.*, 2004; *Safford et al.*, 2008].

Given these challenges, a more appropriate approach to defining reference conditions and using them to inform restoration is to characterize the historical range of variability for specific aspects of stream channels [*Collins et al.*, 2003; *Reeves et al.*, 2005; *Wohl et al.*, 2005]. Historical range of variability is defined as the magnitude and frequency of fluctuations in the stream process or form of interest (e.g., annual peak flow, bed load yield, channel sinuosity, and wood load) over a reference time span. Typically, this time span extends as far back as historical, botanical, or geological records allow for climate conditions similar to the

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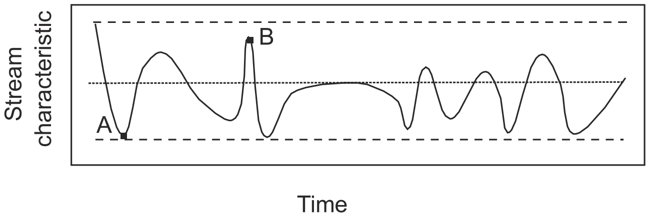

Figure 1. Schematic illustration of range of variability in a stream network. Although the characteristic of interest may have a consistent mean through time (dotted line), it can also exhibit considerable fluctuations at varying periodicities. Use of this hypothetical stream reach would provide very different values for reference conditions at time A versus time B, and it is more appropriate to think in terms of historical range of variability, as indicated by the dashed lines.

present and ends with intensive land use effects that alter channel form or water and sediment yield to the channel. Because of the potential for substantial fluctuations in yields of water, sediment, and wood at timescales of 10^1–10^2 years in association with events such as extreme storms or wildfires, adequate characterization of historical range of variability requires records of at least a century. Such records, which are difficult to generate, can be based on reference streams or on empirical or theoretical models of process [*Gregory et al.*, 2003; *Hassan et al.*, 2005; *Shields et al.*, 2006]. Examples of models for wood in streams include: *Murphy and Koski* [1989], *Bragg et al.* [2000], and *Welty et al.* [2002], who simulated wood recruitment based on forest growth and yield models, but assumed equilibrium between recruitment and in-stream wood depletion, and did not model in-stream wood processes; *Downs and Simon* [2001], who modeled wood recruitment based on bank stability; and *Meleason et al.* [2007], who linked recruitment with stream dynamics and the likelihood of wood breakage, movement, and decay. These models are ultimately tested against empirical data, which reinforces the need for understanding the historical range of variability in forest and stream characteristics that influence wood loads.

This chapter uses the case study of in-stream wood in mountain streams of the Colorado Front Range to illustrate the methods and limitations of defining targets for stream restoration based on the concept of historical range of variability. The streams of this region experienced substantial changes from various land uses starting with beaver trapping during the first decades of the 19th century [*Wohl*, 2001] and continuing with placer mining, flow regulation, timber harvest and tie drives, and construction of roads and railroads during the latter half of the 19th century continuing up to the present [*Wohl*, 2001]. One of the consequences of this land use history is a reduction in in-stream wood, defined here as wood greater than 10 cm in diameter and 1 m in length that is present within the bankfull channel. Although quantitative values of in-stream wood load prior to 19th century land uses do not exist, historical descriptions [*James*, 1823; *Fremont*, 1845] suggest that contemporary values of wood load are much lower than historical values in the Front Range [*Wohl and Jaeger*, 2009]. As resource managers with the U.S. Department of Agriculture (USDA) Forest Service and the National Park Service restore habitat diversity that can support desirable native aquatic and riparian organisms, they seek to understand how much wood was present prior to intensive human alteration of these streams and where the wood was concentrated in channel networks. This information can help to prioritize sites for restoration when it is not feasible to restore an entire stream system. Given the well-established correlations between wood load, pool volume, and fish biomass [*Harmon et al.*, 1986; *Bragg and Kershner*, 1999; *Fausch and Young*, 2004], for example, identifying geomorphic settings that historically contained the greatest wood loads can provide a basis for giving these sites higher priority for protection and restoration.

1.1. In-Stream Wood in Mountain Streams

Numerous studies document the geomorphic and ecological importance of wood in mountain streams. Geomorphic effects of wood include increased hydraulic roughness of channel boundaries [*Keller and Tally*, 1979; *Curran and Wohl*, 2003; *MacFarlane and Wohl*, 2003; *Wilcox and Wohl*, 2007], greater storage of sediment and organic matter on the streambed [*Faustini and Jones*, 2003], enhanced localized bed and bank scour [*Berg et al.*, 1998], and altered local streambed gradient and channel morphology [*Keller and Swanson*, 1979; *Baillie and Davies*, 2002; *Montgomery et al.*, 2003; *Curran and Wohl*, 2003]. Ecological effects of wood include increased retention of organic matter and nutrients [*Bilby and Likens*, 1980; *Raikow et al.*, 1995], greater habitat diversity associated with diversity of substrate and hydraulic variables [*Bisson et al.*, 1987; *Maser and Sedell*, 1994], and food and habitat for many species of microbes and invertebrates [*Maser and Sedell*, 1994]. Studies such as these indicate that in-stream wood plays a critical role in stream form and process and is particularly effective in promoting channel diversity and stability in forested mountain streams of the temperate zones. More limited studies suggest similar functions in tropical streams [*Cadol et al.*, 2009; *Wohl et al.*, 2009].

The processes of wood recruitment, storage, transport, and decay, as well as the geomorphic and ecological functions of wood, vary downstream throughout a network, and among channel reaches (channel segments tens to hundreds of meters in length with consistent geometry). Recruitment describes processes that introduce wood to the stream channel. As conceptualized by *Benda and Sias* [2003], recruitment to a specified length of stream channel occurs via transport from upstream and lateral recruitment L_i. Lateral recruitment occurs via several processes:

$$L_i = I_m + I_f + I_{\mathrm{be}} + I_s + I_e, \quad (1)$$

where I_m is chronic forest mortality, I_f is tree topple from wildfire and windstorms, I_{be} is bank erosion, I_s is mass movements, and I_e is exhumation of buried wood, with all variables in units of volume wood (length stream)$^{-1}$ (yr)$^{-1}$. The relative importance of each of these factors varies greatly between catchments and between different channel segments within a catchment. Wildfires exert a particularly important control on recruitment in semiarid regions [*Young*, 1994; *Zelt and Wohl*, 2004; *Jones and Daniels*, 2008], for example, and bank erosion and exhumation of buried wood are likely to be more important in portions of the channel network with well-developed floodplains and sinuous channels [*Benda and Sias*, 2003]. Changes in the volume of wood stored within a length of stream channel reflect recruitment and losses of wood:

$$\Delta S_c = [L_i - L_o + Q_i/\Delta x - Q_o/\Delta x - D + B]\Delta t, \quad (2)$$

where ΔS_c is change in wood storage (volume wood (length stream)$^{-1}$) within a reach of length Δx over time interval Δt,

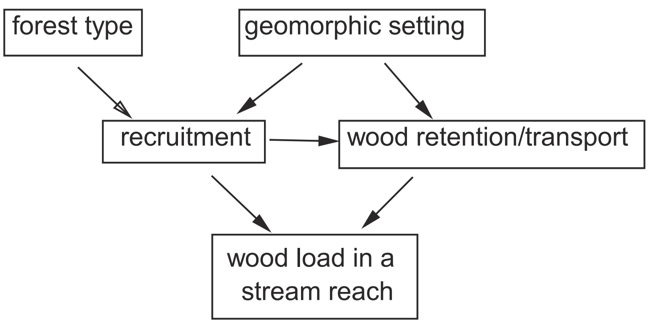

Figure 2. Schematic illustration of factors influencing wood load in a stream reach. “Forest type” refers to size of individual trees, stand density (species and seral stage), and species (rates of mortality and decay). “Geomorphic setting” includes valley confinement (connectivity to valley side slopes, lateral mobility of channel, overbank flooding, and width of riparian zone), flow and sediment regimes, and channel geometry (width, depth, and roughness as these influence wood transport). “Recruitment” includes individual mortality (via bank erosion, tree death, overbank flooding, and channel avulsion) and mass mortality (via insect infestation, wildfires, blowdown, and hillslope mass movements). “Wood retention/transport” includes individual pieces, logjams, and different piece types (bridge, ramp, unattached, and buried). “Wood load in a stream reach” represents volume and changes through time.

L_o is loss of wood to overbank deposition and channel movement, Q_i is fluvial transport of wood into the stream segment, and Q_o is fluvial transport out of the segment (both in volume wood yr^{-1}), D is in situ decay (volume wood yr^{-1}), and B is storage in beaver dams (volume wood (length stream)$^{-1}$), which are distinguished from other, passive types of storage (modified from *Benda and Sias* [2003]). As for equation (1), the relative importance of each factor varies between sites and regions; beavers exert little influence at present outside of North America, for example, and decay is much greater in tropical than in temperate regions [*Cadol et al.*, 2009]. The factors influencing each of these terms are conceptualized in Figure 2.

1.2. The Use of Process Domains as an Organizing Framework

Although mountain streams are typically longitudinally segmented, with abrupt longitudinal changes in gradient and associated characteristics such as grain size and channel geometry [*Wohl*, 2000], these streams can also exhibit overriding downstream trends in forest type and geomorphic setting, and thus in wood load. Downstream variations in wood dynamics, expressed using equation (2), are schematically illustrated in Figure 3. This figure is organized around four different types of geomorphic setting. These settings constitute process domains that designate distinct zones where spatial variability in geomorphic processes governs temporal patterns of disturbances that influence ecosystem structure and dynamics [*Montgomery*, 1999]. The process domains illustrated in Figure 3 do not necessarily occur in a regular downstream progression; steep, confined reaches, for example, can interrupt the overall downstream trends of progressively wider valley bottom relative to channel width and lower stream gradient. The variables in equation (2) vary in relative importance among the process domains, as indicated by difference in font size in Figure 3.

For this analysis, I define confined headwater channel segments as occurring where valley bottom width ≤3 times channel width. In this setting, steep, coarse-grained, cascade, step-pool, or plane-bed channels with relatively low flow depths and narrow channels have limited lateral mobility and transport capacity and few beaver dams; hence, L_o, Q_i, Q_o, and B are less important than in other geomorphic settings. In contrast, lateral recruitment from steep, adjacent slopes can be very important, particularly where mass movements occur [*Wohl et al.*, 2009]. These channel segments are likely to have the largest wood loads within the network, in part because the length of individual wood pieces is likely to be greater than channel width and the depth of flow necessary to float wood pieces of a given diameter is less likely to be

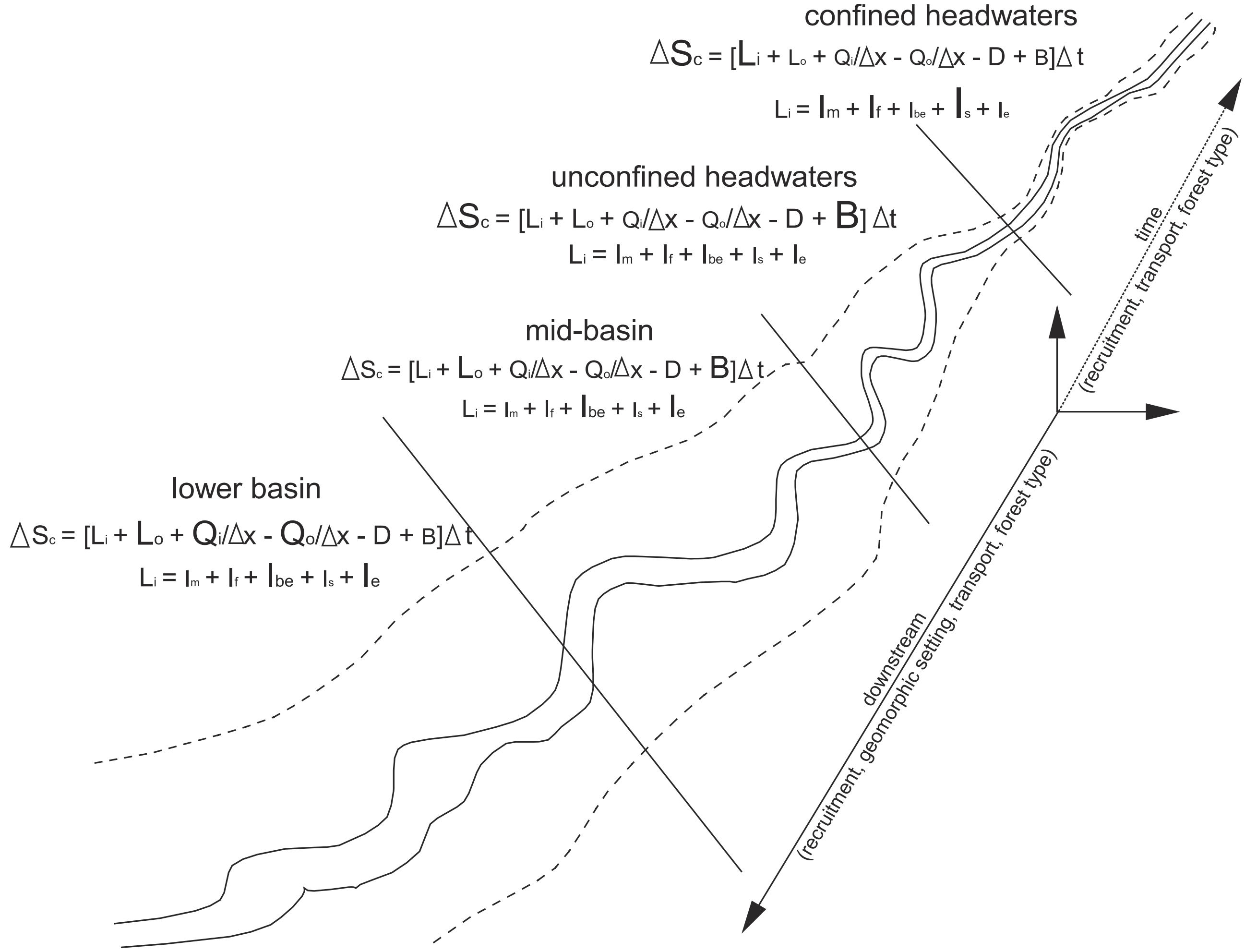

Figure 3. Schematic illustration of longitudinal variations in the variables of equation (2) in relation to changes in process domain. Solid lines indicate lateral boundaries of active channel; dashed lines indicate lateral boundaries of valley bottom. The axes running parallel to the stream trend at right indicate sources of variation in wood load through time (top right) and space (bottom right). Font size indicates the relative importance of each variable in the four types of mountain stream geomorphic setting illustrated here.

reached [*Braudrick and Grant*, 2000, 2001; *Haga et al.*, 2002; *Bocchiola et al.*, 2008], limiting the mobility of wood entering the stream via lateral recruitment.

Unconfined headwater channel segments occur where valley bottom width is 4–10 times channel width. This typically corresponds to lower gradient, finer-grained, more sinuous channels with pool-riffle bed forms. Lateral recruitment from adjacent valley side slopes is limited, but wood can be recruited from adjacent valley bottom surfaces via I_{be} and I_e. Wood load is likely to be high relative to downstream segments because downstream transport of wood remains limited by relatively narrow channels, shallow flows, and lower discharges resulting from smaller drainage areas. In temperate zones of North America, beavers are more likely to colonize these channel segments than those in confined headwaters [*Olson and Hubert*, 1994], and beaver dams could substantially increase wood load at the reach-scale.

In midbasin channel segments, valley bottom width is >10 times channel width. Discharge, channel width, and flow depth, and thus transport capacity (Q_i, Q_o) for wood, increase relative to headwater segments, as does lateral recruitment via I_{be} and I_e, and the presence of beaver dams. Increasing fluvial transport capacity for wood can result in more wood aggregation in jams, although the total wood load is likely to be less than in headwater segments [*Keller and Swanson*, 1979; *Beechie et al.*, 2000; *Wohl and Jaeger*, 2009].

Lower basin channel segments are near or beyond the mountain front. The downstream rate of increase in valley bottom width is greater than that in channel width, resulting in minimal wood inputs from valley side slopes, although recruitment from the riparian zone continues. Transport capacity reaches the greatest values because of increasing flow depth and width relative to wood size, so that Q_i and Q_o exert greater control on wood load than in upstream segments. Lateral recruitment via I_{be} and I_e remain important. Values of wood load are typically the lowest within the channel network, although wood stored along channel margins can still perform important geomorphic and ecological functions.

The details of wood recruitment and transport in this idealized downstream progression vary with hydroclimatology, latitude, and elevation. In semiarid mountains such as the Colorado Front Range, for example, lateral recruitment of wood from valley side slopes ends at the mountain front, where forests give way to chaparral and steppe vegetation. Climate also varies substantially with elevation in the Front Range and in many mountainous regions, and in turn influences forest type, flow regime (transport capacity), and decay rates.

The conceptual model illustrated in Figure 3 provides a framework for using historical range of variability in each of the relevant variables and process domains to estimate reference values of in-stream wood throughout a mountain channel network. The Colorado Front Range provides an example of the strengths and limitations of this approach.

2. MOUNTAIN STREAMS OF THE COLORADO FRONT RANGE

2.1. Overview of the Front Range

The Colorado Front Range encompasses the upper portion of the drainage basin of the South Platte River, which extends 275 km south from the Colorado-Wyoming border to the divide between the South Platte and Arkansas rivers and 100 km east from the Continental Divide to the base of the mountains (Figure 4). More than 10 primary streams flow from headwaters at about 4300 m elevation east toward the base of the range at 1520 m elevation before joining to form the South Platte River.

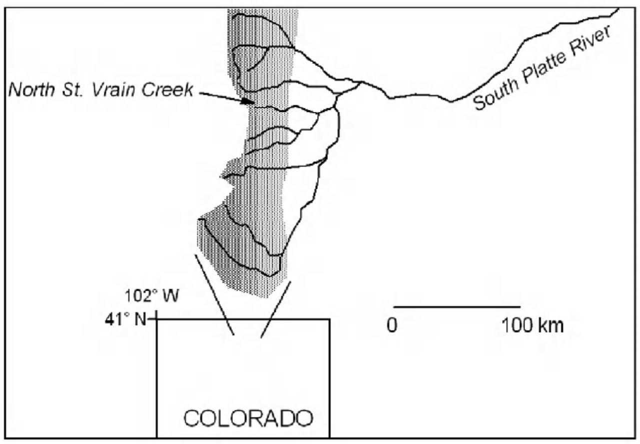

Figure 4. Location map of Front Range (shaded), showing principal streams tributary to the South Platte River.

The lithology of the Front Range is dominated by Precambrian-age granites, gneiss, and schist [*Tweto*, 1979]. The Front Range has been relatively tectonically quiescent since the early Tertiary [*Crowley et al.*, 2002; *Anderson et al.*, 2006]. Pleistocene valley glaciers extended down to approximately 2300 m elevation [*Madole et al.*, 1998]. Narrow, glaciated spines form the range crests at 4000 m elevation, below which lie widespread surfaces of low relief at 2300–3000 m elevation. Fluvial canyons are deeply incised into these low-relief surfaces [*Anderson et al.*, 2006]. Most bedrock outcrops in the region are densely jointed, and joint spacing and valley geometry correlate with the location of shear zones of Precambrian and Laramide age [*Abbott*, 1976]; wider, lower gradient portions of fluvial valleys typically correspond to more closely spaced joints and the location of shear zones [*Ehlen and Wohl*, 2002]. Variations in joint density, glacial history, and other large-scale controls create pronounced downstream variations in valley and channel geometry.

Climate in the Front Range varies with elevation. Mean annual temperature varies from 1°C at the highest elevations to 11°C at the base of the range. Mean annual precipitation decreases from approximately 100 cm at the highest elevations to 36 cm at the mountain front, and the percentage of precipitation falling as snow also decreases with elevation. Numerical simulations of climatic changes within the next century consistently predict decreased precipitation and stream flow and increased temperatures [*Nash and Gleick*, 1991; *Mote et al.*, 2005; *Stewart et al.*, 2005].

2.2. Forest Characteristics

Steppe vegetation on the plains gives way with increasing elevation to montane forest (1830–2740 m elevation), subalpine forest (2740–3400 m), and eventually alpine vegetation [*Veblen and Donnegan*, 2005]. The montane zone includes ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*). Open ponderosa pine woodlands dominate the lower montane zone (approximately 1830 to 2350 m), often forming sparse stands with larger surface areas covered by grasses and forbs than by trees. Narrow bands of riparian forests of plains and narrow-leaf cottonwood (*Populus sargentii* and *Populus angustifolia*, respectively) extend from the eastern plains into the foothills of the

Front Range along river corridors, which are dominated by broadleaf deciduous species of cottonwood and willow (*Salix* spp.) [*Veblen and Donnegan*, 2005].

Stand density and species composition vary substantially within the montane zone in relation to aspect. Xeric, south facing slopes have relatively open stands of ponderosa pine and sometimes Rocky Mountain juniper (*Juniperus scopulorum*), whereas mesic, north facing slopes have denser forest cover with more Douglas-fir. Stand density and the abundance of Douglas-fir increase with elevation. Douglas-fir becomes gradually less important in the upper montane zone (2440–2740 m).

The subalpine zone includes Engelmann spruce (*Picea englemannii*), subalpine fir (*Abies lasiocarpa*), lodgepole pine (*Pinus contorta*), aspen (*Populus tremuloides*), and limber pine (*Pinus flexilis*), as well as blue spruce (*Picea pungens*) as a riparian species. Lodgepole pine forests become increasingly important with elevation in the upper montane zone and dominate large areas of the subalpine zone, forming the most extensive forest type in the Front Range [*Veblen and Donnegan*, 2005]. More mesic subalpine sites are dominated by Engelmann spruce and subalpine fir, whereas lodgepole dominate more xeric sites and are successional to the spruce-fir community. Riparian communities include large numbers of conifers such as Douglas-fir and spruces, as well as aspen.

Age and size of individual trees varies greatly with site-specific conditions, but typical characteristics are listed in Table 1. Decay rates for standing dead trees or fallen logs of individual species have received relatively little attention in the Front Range, but available estimates are also summarized in Table 1. Decay rates tend to be higher for pines and at lower elevations, likely because the long, cold winters at higher elevations inhibit decomposition [*Arthur and Fahey*, 1990; *Kueppers et al.*, 2004].

Disturbance has the potential to alter forest dynamics and, thus, rates and mechanisms of wood recruitment and storage in mountain streams. Disturbance is defined by ecologists as any relatively discrete event in time that disrupts ecosystem, community, or population structure and changes resources, substrate availability, or the physical environment [*White and Pickett*, 1985]. Disturbance in Front Range forests takes the form of wildfire, persistent drought, insect outbreak, blowdown, hillslope mass movements, such as debris flows, and floods. Of these, fire and insect outbreaks are the most significant in terms of extent, severity, and frequency, and

Table 1. Characteristics of Dominant Tree Species in the Colorado Front Range

Tree Species	Height	Diameter	Age	Decay Rates[a]	Reference
Ponderosa pine	rarely >30 m	can be >100 cm, but mostly <50 cm	oldest known is just over 600 years, but most trees <500 years	Trees can persist >160 years after death or fire.	*Huckaby et al.* [2003], *Hall et al.* [2006]
Lodgepole pine	rarely >24 m	typically <65 cm	establish over 100–120 years following disturbance such as fire, can live >200 years, but most trees <120 years	Turnover time is 630 ± 400 years in midelevation forest (3300–3400 m) and 340 ± 130 years in lower elevation forest (3000–3300 m).	*Knowles and Grant* [1983], *Veblen* [1986a], *Kueppers et al.* [2004]
Engelmann spruce	typically <37 m	typically <75 cm	establish over 60–100 years following disturbance, can live >500 years	Turnover time is 650–920 years; dead trees can stand >190 years.	*Knowles and Grant* [1983], *Veblen* [1986a], *Mast and Veblen* [1994], *Brown et al.* [1998], *Kueppers et al.* [2004]
Subalpine fir	typically <30 m	typically <45 cm	establish over 90–100 years following disturbance, can live >350 years but mostly 200–350 years	Fallen trees may require >150 years to completely disappear; dead trees can stand >150 years.	*Knowles and Grant* [1983], *Veblen* [1986a], *Roovers and Rebertus* [1993], *Mast and Veblen* [1994], *Brown et al.* [1998]
Limber pine	typically <18 m	<65 cm	can live >300 years		*Knowles and Grant* [1983], *Veblen* [1986a]

[a]Decay rates are for terrestrial wood not in-stream wood. Continuously wet wood decays more slowly, but wood alternately wetted and dried and abraded by sediment in transport may decay more rapidly than terrestrial wood.

time since fire appears to be the single most important control on volume of dead wood in a stand [*Mast and Veblen*, 1994; *Ehle and Baker*, 2003; *Hall et al.*, 2006].

Three general types of historic fire regimes occur in the Front Range [*Veblen and Donnegan*, 2005]. Frequent, low-severity fires that burn mainly the ground surface over areas of approximately 100 ha recur at intervals of 5–30 years in the lower elevations of the montane zone. Infrequent, high-severity fires that kill all canopy trees over areas of hundreds to thousands of hectares recur at intervals greater than 100 years in the subalpine zone. A complex pattern of both low- and high-severity fires that burn areas of approximately 100 ha and recur at intervals of 40 to 100 years occurs in the middle and upper montane zone.

Tree ring records extending back to the late 16th century have been used to reconstruct hydroclimatic indicators such as average annual streamflow [*Woodhouse*, 2001; *Gray et al.*, 2003] and the Palmer Drought Severity Index [*Cook et al.*, 2007] as well as disturbances including wildfire and insect outbreaks [*Veblen and Donnegan*, 2005]. These long-term records indicate that regional atmospheric patterns, including the El Niño–Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), and Atlantic Multidecadal Oscillation (AMO), significantly influence the occurrence of widespread wildfire in the Front Range. Warmer and drier spring-summers are strongly associated with years of widespread fire, especially when preceded by 2 to 4 years of wetter-than-average spring conditions, a pattern typically associated with ENSO events [*Veblen et al.*, 2000]. Decadal- to centennial-scale fluctuations in regional precipitation associated with the ENSO, PDO, and AMO circulations created a high degree of variation in fire regimes and forest conditions of the Front Range prior to fire suppression, which started in 1920 [*Veblen and Donnegan*, 2005].

Forest insects capable of producing widespread tree kills include the mountain pine beetle (*Dendroctonus ponderosa*), which attacks primarily ponderosa and lodgepole pine by boring through the bark and infecting the tree with fungi that block the movement of water and nutrients through the tree [*Veblen and Donnegan*, 2005; *Romme et al.*, 2006]. Mountain pine beetle outbreaks recur in the same general region within about 20 years and in the same stand within 50–100 years and are capable of killing nearly all of the overstory trees.

Disturbances associated with blowdown and hillslope instability are much more localized than those caused by wildfire, drought, and insects. Blowdown is more important in the subalpine zone, where shallow-rooted trees and extreme wind speeds typically produce blowdowns up to several hectares in size [*Veblen et al.*, 1991]. Little is known of the spatial extent or recurrence interval of hillslope mass movements in the Front Range, but it is reasonable to assume that debris flows and landslides typically occur in association with high-severity wildfires [*Moody and Martin*, 2001] and

Figure 5. Photograph taken 30 years after wildfire along a section of Ouzel Creek in Rocky Mountain National Park at elevation 3050 m. Note the very slow regrowth of coniferous trees and the standing dead trees.

intense convective storms [*Shroba et al.*, 1979] with recurrence intervals of several decades to more than a century in any headwater watershed.

Regrowth of woody plants following a disturbance is slow in the semiarid Front Range relative to other temperate forests (Figure 5). Recruitment period following disturbance varies with site conditions, seed sources, and climate, but is typically 30–60 years for both the montane and subalpine zones [*Veblen and Donnegan*, 2005]. Old-growth characteristics, however, typically do not emerge for at least 200 years in montane [*Kaufmann*, 1996] or subalpine forests [*Veblen*, 1986b]. Wood recruitment to streams flowing through the disturbed area can thus decrease substantially for a period of several decades following a disturbance and may require two centuries to reach pre-disturbance conditions.

2.2.1. Stream characteristics. The major streams of the Colorado Front Range are perennial, with a snowmelt peak in late spring and early summer. Convective storms also generate flash floods below approximately 2300 m elevation. These rainfall floods can generate a peak discharge as much as 40 times the magnitude of snowmelt flood peaks [*Jarrett*, 1989].

Streams in the Front Range tend to have a very steep gradient (>0.01 m m^{-1}) and a narrow valley bottom with a limited floodplain. Channels are likely to have step-pool, plane-bed, or pool-riffle bed forms [*Montgomery and Buffington*, 1997], but channel and valley morphology has substantial longitudinal variation as a result of downstream changes in geology, glacial history, and beaver activity [*Wohl et al.*, 2004]. Most of the larger streams alternate downstream between narrow, high-gradient canyons and wider, lower-gradient reaches that coincide with Precambrian shear zones [*Ehlen and Wohl*, 2002]. As a result of this longitudinal variability, valley segments are distinctly different with respect to gradient, substrate type, degree of lateral confinement, wood load, frequency of disturbances associated with floods and debris flows, and the diversity and stability of aquatic and riparian habitat [*Wohl et al.*, 2004].

Most stream segments have a coarse bed formed from cobble- to boulder-sized sediment. Widespread movement of the abundant sand and gravel underlying the stream bed does not occur during the average annual snowmelt flood, but does occur infrequently during summer rainfall floods that occur below 2300 m. Only these floods generate sufficient stream power to mobilize the coarse-surface stream bed and to substantially reconfigure the morphology of the channel and valley bottom [*Shroba et al.*, 1979]. Flooding can also be exacerbated by a hillslope disturbance, such as a wildfire, that introduces large quantities of sediment into the river via debris flows and landslides [*Elliott and Parker*, 2001]. First- and second-order channels have more frequent and longitudinally extensive impacts from debris flows. Infrequently, with recurrence intervals >100 years, these flows create localized deposits, such as partial levees, along the larger streams. Streams of the Front Range are thus typically stable, with relatively low sediment loads, although they periodically exhibit dramatic responses to disturbance from floods and hillslope instability.

2.2.2. Land use history. Although people have lived in the Colorado Front Range for at least 12,000 years [*Benedict*, 1992], there is no evidence that population densities or land use patterns produced changes in the region's rivers until the first decades of the 19th century. Fur trappers of European descent began to trap beavers (*Castor canadensis*) in the western United States shortly after the 1804–1806 Lewis and Clark expedition. Trapping quickly became so intensive that John Charles Frémont noted the rarity of active beaver lodges during his 1842–1843 travels in the Front Range. Beavers influence water and sediment movement, channel form, and wood storage along a stream by building low dams of wood and sediment [*Naiman et al.*, 1986, 1988]. These dams create ponds that act as sediment traps, gradually filling to create floodplain wetlands or meadow environments. The dams and ponds enhance the depth, extent, and duration of flood inundation and elevate the water table during high and low flows [*Westbrook et al.*, 2006]. Beaver dams were breached with the removal of beavers, allowing some of the streams in the Front Range to incise rapidly into the accumulated fine sediment, increasing fine sediment loads, reducing flood hydrograph attenuation and groundwater recharge, and altering patterns of wood recruitment and storage. Although beaver have recolonized some streams in the Front Range, population levels remain severely depressed relative to pre-trapping levels [*Naiman et al.*, 1986].

Removal of placer metals such as gold and silver from valley-bottom sediments began in Colorado in 1859, followed by a series of mineral rushes spread across the mountainous portion of the state during succeeding decades. Miners typically used hydraulic systems and sluices to mechanically separate precious metals, processing 2–4 m^3 of sediment in 10 h. Commercial operators installed dredge boats that could process 6000–6600 m^3 of sediment in the same timespan [*Silva*, 1986]. The typical practice in either case was to remove and process valley-bottom sediment down to the bedrock contact and back to the valley side slopes [*Wohl*, 2001]. Placer mining disrupted the coarse surface layer present in many channel segments in the Front

Range, dramatically increasing sediment mobility, which resulted in alterations to water quality, aquatic habitat, grain-size distribution, channel form, and flooding [*Hilmes and Wohl*, 1995; *Wohl*, 2001].

Deforestation and construction of roads and railroads in association with mineral rushes also substantially increased sediment yields to channels and reduced the extent of forest cover and opportunities for wood recruitment to channels. High demand for railroad ties during the 1860s–1890s was satisfied mainly with timber cut in the mountains during winter and then floated downstream to collection booms during snowmelt peak flows. Naturally occurring obstructions such as wood and large boulders were removed along streams used for tie drives, overbank areas and floodplain wetlands were separated from the main channel by dikes, and meanders were artificially straightened using cutoffs. Floated logs acted as massive scouring brushes as they moved downstream, so that analogous streams with and without tie drives have statistically significant differences in riparian vegetation, channel form, and wood loads a century after the last tie drives occurred [*Young et al.*, 1990].

Flow diversions from Front Range streams began with placer mining in 1859. The magnitude and extent of diversions and water storage increased dramatically during subsequent decades of the 19th century as irrigated agriculture and urban communities grew along the base of the range, and nearly half of the regional streamflow was diverted from the western side of the Continental Divide. Flow regulation has altered woody riparian vegetation along Front Range streams [*Merritt and Wohl*, 2006] and thus wood recruitment, as well as wood transport. The common occurrence of diversion intakes and other infrastructure, along with bridges, also causes contemporary resource managers to routinely remove large wood recruited to main stem streams such as the Poudre and Big Thompson rivers. Recreational uses including whitewater rafting and kayaking are also extremely popular along some Front Range streams, and boaters sometimes remove individual pieces of wood or logjams where possible.

The cumulative effect of 19th and 20th century land uses has thus been to alter stream form and function in the Front Range. Forest characteristics and geomorphic setting have been altered relative to pre-19th century average conditions, as have processes of wood recruitment, retention and transport, and wood loads. Fire suppression has likely resulted in forests with greater coarse woody debris than historical levels [*Robertson and Bowser*, 1999], although wood entering streams is likely to be smaller in size than it was historically because of the smaller dimensions of second-growth trees. Streams tend to be less geomorphically diverse, with fewer sites of wood storage along channel margins and in beaver dams. The lack of very large trees, reduction of wood recruitment via fire and mass movements, and removal of beaver dams and logjams create a net effect of lower wood loads along most Front Range streams. Similar land use histories and consequent reductions in wood loads have been documented for rivers in diverse settings around the world [*Hedman et al.*, 1996; *Piégay and Gurnell*, 1997; *Berg et al.*, 2003; *Brooks et al.*, 2006; *Gomi et al.*, 2006].

2.2.3. Wood loads. Results for the four studies that have documented wood loads for the Colorado Front Range are summarized in Table 2. *Richmond and Fausch* [1995] showed that streams with timber harvest or other land uses, even though these occurred prior to 1900 A.D., have much lower wood loads than those streams with old-growth forest. Following a 10 year study, *Wohl and Goode* [2008] documented 16–23% mobility of pieces within a stream each year. Residence time of a piece of wood correlates with piece length and discharge; they estimated an average residence time of only 3.4 years for wood in small headwater streams. This is one to two orders of magnitude less than residence times estimated for analogous streams in the Pacific Northwest [*Swanson et al.*, 1976, 1984; *Keller and Tally*, 1979; *Lienkaemper and Swanson*, 1987; *Gregory*, 1991; *Hyatt and Naiman*, 2001; *Gurnell et al.*, 2002], a difference that *Wohl*

Table 2. Characteristics of Wood in Streams of the Colorado Front Range

Drainage Area (km^2)	Elevation (m)	Bankfull Width (m)	Wood Load (m^3/100 m stream)	Wood load (m^3 ha^{-1})	Description[a]	Reference
2.4–29.1	2719–3204	3.7–10.2	6.6–27.1	93.3–254.3	old-growth	*Richmond and Fausch* [1995]
6.9–28.9	2730–2925	4.0–5.1	0.6–6.6	–	disturbed	
8–270	1990–3040	3.5–13.9	0.02–9.8	0.06–218	disturbed	*Wohl and Jaeger* [2009]
9.2–32.3	2740–2960	4.3–6.5	1.2–15.2	18–211	disturbed	*Wohl and Goode* [2008]
7.8–20	2850–3140	5–17	3.8–18.3	64–415	old-growth	*Wohl and Cadol* [2011]
12.1–82.2	2600–3010	7–20	7.7–18.5	12–378	disturbed	

[a]Old-growth indicates forests >200 years old; disturbed indicates forests with timber harvest prior to 1900 A.D.

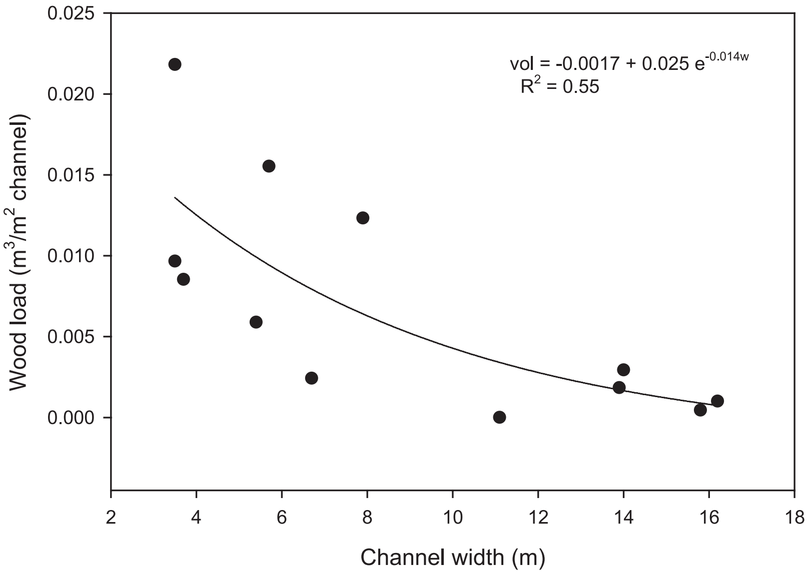

Figure 6. Plot of wood load in relation to channel width for streams of the Colorado Front Range. After *Wohl and Jaeger* [2009, Figure 3B].

and Goode [2008] attribute to the lower wood loads and lack of jams in Front Range streams. Wood within jams in the Front Range study sites had a slightly longer residence time of 4.2–4.5 years. Wood loads decrease systematically downstream in relation to drainage area and channel width [*Wohl and Jaeger*, 2009] (Figure 6), but the proportion of wood within a jam is greatest in the middle portion of the stream network. *Wohl and Jaeger* [2009] interpret these patterns as reflecting transport-limited conditions for wood in the smallest channels, and supply-limited conditions in the largest channels, analogous to interpretations of mountain stream networks elsewhere [*Marcus et al.*, 2002]. Working in headwater streams draining old-growth forest stands, *Wohl and Cadol* [2011] documented substantial longitudinal variation in wood load as process domain varied downstream. They found the greatest wood loads in slightly lower (0.04–0.06 m m^{-1}) gradient stream segments with wider valley bottoms that were located immediately downstream of steep, narrow gorges. Continuous surveys of up to 9 km of stream length at a smaller range of drainage areas than covered by *Wohl and Jaeger* [2009] indicated that downstream trends such as that in Figure 6 are overwhelmed at smaller scales by substantial interreach variation in wood loads associated with variations in recruitment, storage, and transport [*Wohl and Cadol*, 2011]. This result, in particular, supports the idea that spatially distinct process domains are useful in understanding within-network variations in wood dynamics that can help to set appropriate restoration targets for in-stream wood.

3. SETTING RESTORATION TARGETS FOR MOUNTAIN STREAMS IN THE COLORADO FRONT RANGE

Lacking quantitative estimates of wood load in Front Range streams prior to land use changes in 1800, two approaches can be used to estimate the historical range of variability in wood load at a site and relative magnitudes of wood load between sites; extrapolation from reference sites or from regional data sets [*Fox et al.*, 2003] or mechanistic models of the variables in equation (2) based on empirical data.

3.1. Reference Sites

Beavers were trapped along all of the streams in the Colorado Front Range during the 19th century, and there is no indication that beavers have returned to historical population densities. The streams that otherwise most closely approximate reference conditions are those with no historical tie drives or placer mining, no historical or contemporary flow regulation or transportation corridors, and with old-growth (>200 year old) trees. The great majority of these streams drain less than 30 km^2. North St. Vrain Creek (drainage area 240 km^2) is the only larger stream that meets most of these criteria, but the forests along the larger streams in this channel were disturbed prior to 1900 A.D., primarily by forest fires [*Sibold et al.*, 2006], and only streams draining <20 km^2 represent old-growth forest.

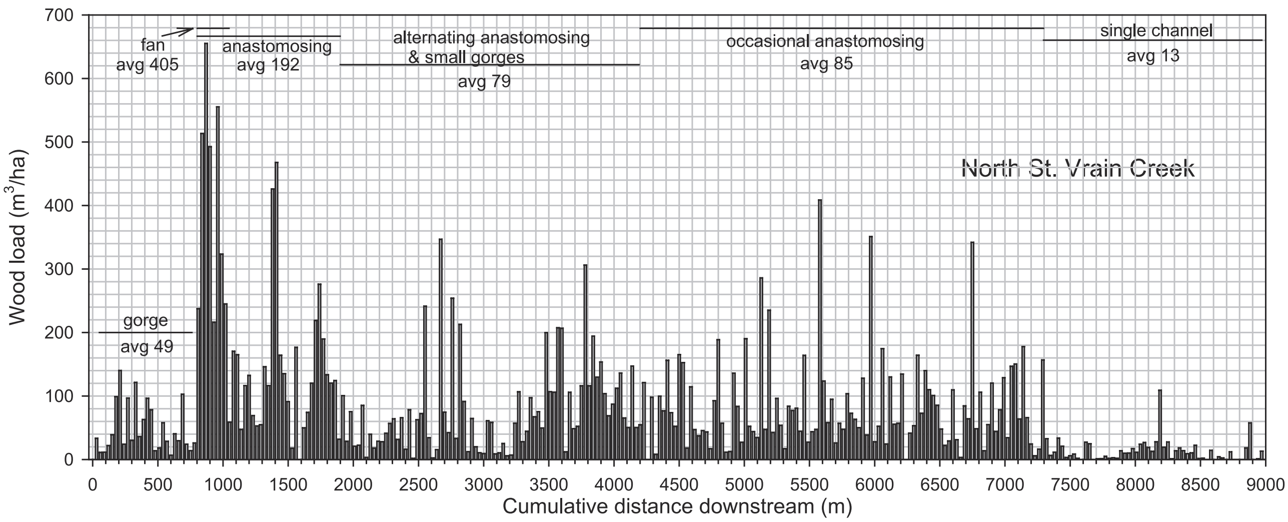

Figure 7. Bar graph showing longitudinal distribution of wood load by 30 m channel increments. Important aspects of channel geometry are noted on each graph, as are average values for *y* axis variable. After *Wohl and Cadol* [2011, Figure 5A].

Empirical data from sites in the North St. Vrain drainage indicate substantial spatial variability in wood load (Figure 7). Average values of wood volume per unit channel surface area (m^3 wood m^{-2} channel surface) exceed 400 in lower-gradient reaches with anastomosing channels and drop to as low as 13 in single-channel reaches with bedrock valley walls that limit lateral recruitment. There are no data for temporal variability in wood load at timespans greater than a decade. *Wohl and Goode* [2008] found that wood load remained relatively constant at channel lengths of a few tens of meters during the course of a decade, although most individual pieces of wood entered or left the specified channel length during that time span. Empirical data extending the length of North St. Vrain Creek also indicate large spatial variability in wood load, but can be used to estimate average wood load in relation to channel width (Figure 6):

$$\mathrm{WL} = -0.0017 + 0.025\mathrm{e}^{-0.014w}, \tag{3}$$

where WL is wood load (m^3 wood m^{-2} channel surface area) and w is channel width (m). Given the known history of changes in forest and stream characteristics in the Front Range, the values in Figure 6 and for disturbed forests in Table 2 are assumed to be lower limits for historical range of variability in wood load, whereas values for old-growth forests are likely to approximate upper limits.

These inferences are supported by application of an analogous equation developed by *Bragg et al.* [2000] for old-growth spruce fir forests in northwestern Wyoming:

$$\mathrm{WL} = 0.0561\mathrm{e}^{-0.0843w}, \tag{4}$$

with variables as defined in equation (3). Application of this equation to streams draining disturbed forests in the Colorado Front Range overpredicts actual wood load by as little as a factor of two at smaller drainage areas to as much as a factor of 13 at larger drainage areas. The use of reference sites to infer historical range of variability in wood loads for the Colorado Front Range is thus limited by the lack of old-growth forests along streams draining areas exceeding approximately 30 km^2.

3.2. Regional Data Sets and Models

Worldwide, coniferous forests average 240 m^3 of wood per ha of channel [*Gurnell et al.*, 2002]. Streams in the temperate coniferous rainforest of the northwestern United States average 812 m^3 ha^{-1} [*Gurnell et al.*, 2002]. Streams in the drier conifer forests of the Intermountain West range from relatively low values of 15–175 m^3 ha^{-1} in northern Wyoming [*Nowakowski and Wohl*, 2008] to higher values of 178–368 m^3 ha^{-1} in northwestern Wyoming [*Zelt and Wohl*, 2004]. Those surveyed in the Front Range vary from 0.1 to 415 m^3 ha^{-1} (Table 2) but fall mostly in the range of 100–200 m^3 ha^{-1} along streams with old-growth forest [*Richmond and Fausch*, 1995; *Wohl and Cadol*, 2011] and <100 m^3 ha^{-1} in streams disturbed prior to 1900 A.D. [*Wohl and Jaeger*, 2009]. Many of the streams elsewhere in the Intermountain West, however, were subject to the same historic land uses as the Front Range streams and thus likely have lower wood loads than were present prior to 1800 AD.

Front Range forests also have some unique characteristics. Values of live and dead coarse woody debris in Front Range forests are consistently lower than those of other coniferous forests in the western United States and have high variability [*Robertson and Bowser*, 1999; *Baker et al.*, 2007]. Standing dead volume composes a relatively low (7.5%) percentage of total coarse woody debris in ponderosa pine stands [*Robertson and Bowser*, 1999]. Standing dead volume increases to 12% in subalpine forests, which also have greater total biomass than montane forests and a greater total dead biomass (60%) [*Arthur and Fahey*, 1990; *Kueppers et al.*, 2004].

Given these caveats regarding regional uniqueness, wood loads predicted with existing models developed from other regions can be compared to those observed in Front Range streams. The great majority of models have been developed for very different forest types, most of which are in the Pacific Northwest region of North America [*Gregory et al.*, 2003]. The exception comes from *Bragg et al.* [2000], who used data from 13 plots in old-growth spruce-fir forest of northwestern Wyoming to develop a model of riparian wood recruitment. This model, which is considered one of the most complete representations for the ecological processes of riparian stand dynamics and in-stream processes influencing wood [*Gregory et al.*, 2003], is based on simulations of forest growth and yield, snag residency and failure, and assumptions that wood enters streams primarily through chronic forest mortality and that in-stream wood follows a long-term steady state with losses roughly equal to inputs. Their simu-

Table 3. Empirical Knowledge of Variables in Equations (1) and (2) for the Colorado Front Range

Variable	Characteristics	References
I_m, chronic forest mortality	Assume following life spans: ponderosa pine 200–300 years, lodgepole pine 100–120 years, Engelmann spruce and subalpine fir 200–350 years. Dead trees can stand >100 years; 5–10% of total biomass are dead in montane forests, 25–60% are dead in subalpine forests; 3–8% of total biomass standing are dead in montane forests, 10–20% standing are dead in subalpine forests. This can be quantified using the Forest Vegetation Simulator of *Wykoff et al.* [1982].	*Kueppers et al.* [2004]
I_f, tree topple from fire and blowdown	About 75% of fire-killed trees fall within 10 years after burn; total dead wood in ponderosa and Douglas-fir stands peaks 10–19 years after fire, reaches a minimum of 61–85 years after fire, may stabilize at >150 years. Assume all of montane forest is affected by fire within 300 year time span and 60% of subalpine forest. This could be quantified using stochastic treatment of data from historical wildfires and blowdowns.	*Pearson et al.* [1987], *Veblen et al.* [1994], *Harrington* [1996], *Ehle and Baker* [2003], *Hall et al.* [2006]
I_{be}, bank erosion	There are no measured or simulated rates of bank erosion or resulting wood recruitment.	
I_s, mass movements	There are no measured or simulated rates.	
I_e, exhumation of buried wood	Assume negligible in all but unconfined headwaters; there are no measured or simulated rates.	
L_o, lateral loss	This is negligible in confined headwaters. Personal observations suggest <10% of total wood in other stream types; there are no measured or simulated rates.	
Q_i, fluvial transport in	There are no measured or simulated rates.	
Q_o, fluvial transport out	There are no measured or simulated rates.	
D, decay	Rate decreases with elevation; assume complete decay requires ~200 years in montane zone and 600 years in subalpine zone based on decay rates on forest floor (Table 1).	*Kueppers et al.* [2004], *Hall et al.* [2006]
B, storage in beaver dam	Range is 1–5 dams km^{-1} in subalpine valleys; no information is available on volume of wood in beaver dams, which might be negligible in dams composed of many very small pieces of wood in unconfined headwaters or substantial in dams composed of larger pieces.	*Ruedemann and Schoonmaker* [1938], *Westbrook et al.* [2006]

lation predicted a relatively constant wood load of 8.6 m^3 $(100\ m)^{-1}$ of channel for northwestern Wyoming. Front Range streams with a similar range of drainage areas (<10 km^2) have values of 6.6–20.7 m^3 $(100\ m)^{-1}$ in old growth [*Richmond and Fausch*, 1995] and 3.1–7.6 m^3 $(100\ m)^{-1}$ in areas disturbed prior to 1900 [*Wohl and Jaeger*, 2009]. This suggests that the *Bragg et al.* [2000] model provides a reasonable approximation of wood load in old-growth Front Range streams, although it may only apply to very small drainage areas, and it does not account for potentially substantial spatial and temporal variations in wood load.

3.3. Mechanistic Models of Wood Processes in the Front Range

Regression equations of wood load in relation to another parameter such as channel width implicitly incorporate all of the mechanisms of wood recruitment, storage, and transport in equations (1) and (2). These mechanisms can also be more explicitly estimated from empirical data. Table 3 summarizes what is known about each of the variables in equations (1) and (2) for the Front Range. The substantial gaps in this table indicate that it is not yet feasible to develop a quantitative mechanistic model for wood dynamics in most stream segments of the Colorado Front Range analogous to that developed for the Oregon Coast Range by *Lancaster et al.* [2001]. Of the 10 variables listed in Table 3, only the first, chronic forest mortality, can be simulated quantitatively at this time. Each of the 10 variables in this table would ideally be expressed as wood volume $(\text{channel area})^{-1}$ $(\text{time})^{-1}$, but more research is required to define these quantities.

Based on knowledge of forest and stream dynamics, it is reasonable to conceptualize historical range of variability in wood recruitment and retention within streams, as illustrated in Figures 8 and 9. Although these figures do not provide precise numerical values of wood load, they can be used to constrain relative range of variability in wood load in different portions of the stream network, analogous to Figure 3.

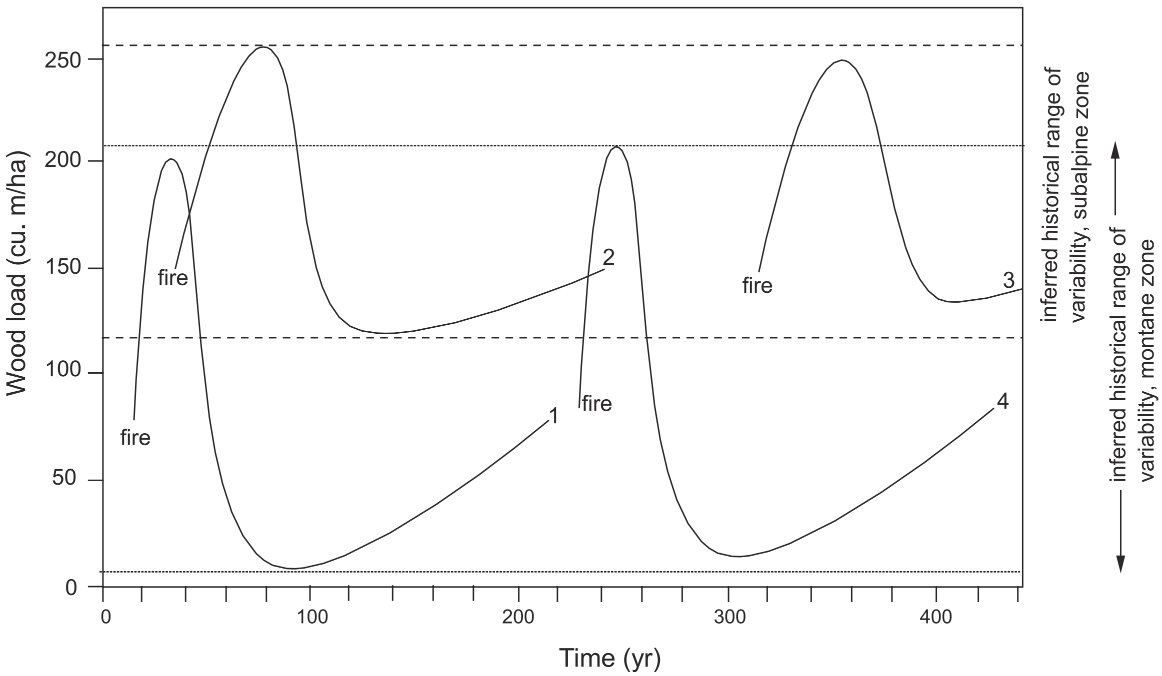

Figure 8. Conceptual model of inferred temporal variations in wood availability and resulting wood load on the forest floor in different forest types of the Colorado Front Range. This conceptualization is based on forest dynamics and does not account for transport or retention of wood in the stream or for differences in geomorphic setting as these influence recruitment mechanisms. Trajectories are as follows: 1, stand-killing fire followed by no further disturbance, montane zone; 2, stand-killing fire followed by no further disturbance, subalpine zone; 3, next stand-killing fire in subalpine forest; and 4, next stand-killing fire in montane forest. Assumptions include greater totals and less temporal variation in wood loads in subalpine forests and fire as the dominant source of disturbance. Magnitude of fluctuations is inferred from studies of dead wood in forests; periodicity of fluctuations is inferred from fire recurrence intervals; values of wood load are from data representing a point in time on old-growth streams. Inferred historical range of variability for subalpine zone is shown by dashed lines and by dotted lines for the montane zone.

3.4. Process Domains

The uncertainty in estimates of the historical range of variability in wood load based on reference sites, regional data sets, and mechanistic models can be reduced by applying the concept of process domains [*Montgomery*, 1999]. Identification of geomorphic setting (Figures 3 and 9) and forest type can constrain the relative importance of the different process of recruitment and retention (equations (1) and (2)) as well as the geomorphic and ecological roles of instream wood in that process domain. Wood-forced alluvial channel reaches or pool-riffle sequences [*Montgomery et al.*, 1995] are unlikely to occur in unconfined headwaters, for example, just as beaver dams are unlikely to occur in confined headwaters. Process domains delineate portions of the stream network between which the parameters used in numerical simulations likely differ significantly. Process domains can also be used to identify the portions of the channel network that historically had the greatest wood loads and the greatest abundance of associated aquatic habitat; in the Front Range example, these would be channel segments in the uppermost portions of the network with lower gradients and wider valley bottoms than channel segments immediately up- and downstream.

3.5. Limiting Factors

The relatively small supply and associated slow rate of recruitment of large pieces of wood capable of trapping other wood and forming stable jams is one of the primary factors

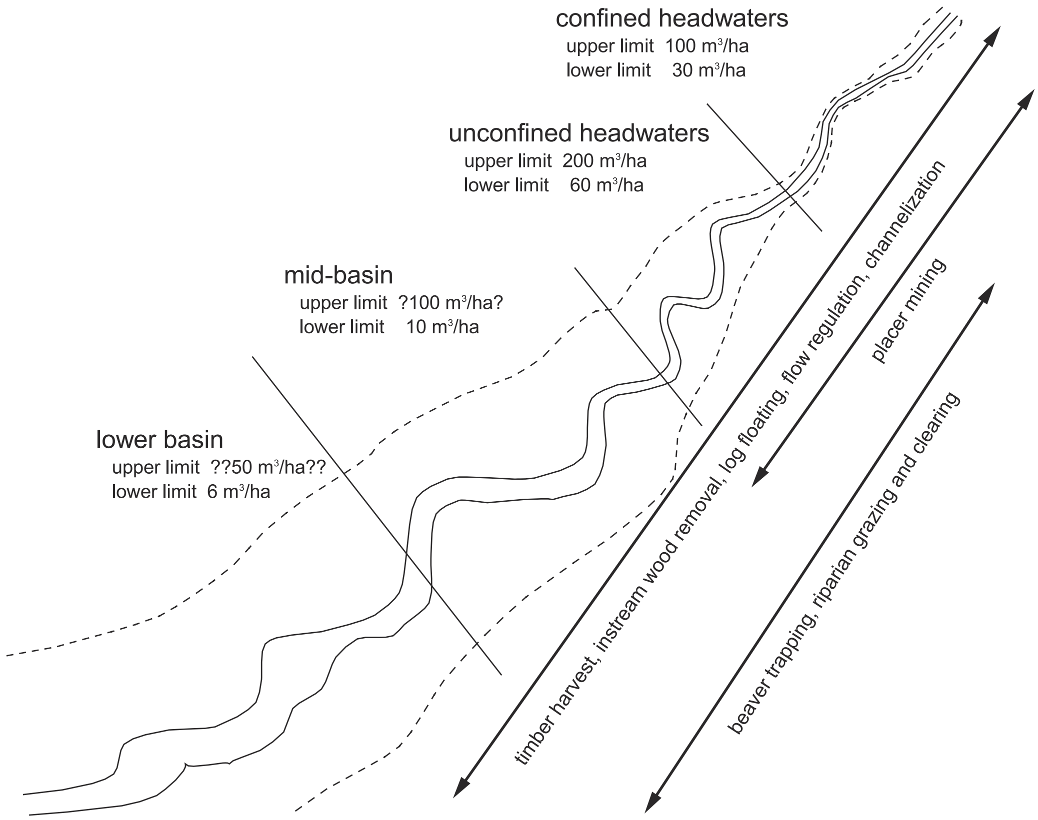

Figure 9. Inferred ranges of wood loads in streams of the Colorado Front Range for the geomorphic process domains shown in Figure 3. In each process domain, the upper and lower limits describe the range of in-stream wood load likely to result from nonhuman disturbances that influence recruitment (e.g., land cover (forest, meadow, bedrock valley walls, etc.), forest fires, insect infestations, blowdowns, hillslope mass movements, and individual tree mortality) and from instream processes of breakage, decay, and transport. Values for upper and lower limits are based on data from *Wohl and Jaeger* [2009] and *Wohl and Cadol* [2011]. The question marks for the upper limits in the midbasin and lower basin categories reflect the lack of old-growth sites in these portions of the catchment that would facilitate estimation of an upper limit for wood load. Key anthropogenic processes altering wood dynamics in each process domain are shown at the right. Some of these, such as channelization, did not occur in the Colorado Front Range. Others, such as beaver trapping, do not apply to mountainous areas outside of North America.

limiting wood retention in Front Range streams. This will not change in the short term because of relatively slow rates of tree growth and forest regeneration to old-growth conditions following disturbance such as timber harvest. Naturally recruited large trees can be artificially fixed in place or large trees cut elsewhere in the forest can be placed in the stream in order to initiate sites of jam formation and longer wood retention [*Keim et al.*, 2000; *Brooks et al.*, 2006].

A second factor limiting wood retention is the removal of naturally recruited wood in order to avoid damage to infrastructure or to maintain recreational boating. Large wood fixed in place along stream segments away from infrastructure and not typically used by boaters can alleviate some of these concerns while still providing geomorphic and ecological functions.

An example of the results from emplacement of artificially anchored wood comes from a study undertaken by USDA Forest Service scientists along the South Platte River at Elevenmile Canyon (drainage area 2400 km^2) during 1996–1998. Placement of logs and other structures along a portion of river dominated by relatively uniform runs enhanced local bed scour and increased biomass of trout by a factor of four in the vicinity of the structure within 5 years (D. Winters, personal communication, September 2008). Placement of anchored logs also substantially increased available habitat and fish abundance in smaller, headwater streams [*Gowan and Fausch*, 1996].

3.6. Setting Priorities

It is not feasible to restore wood loads to historical levels throughout streams of the Colorado Front Range because of the extensive historical and contemporary alterations of processes controlling wood recruitment and retention. Resource managers are thus faced with decisions as to which stream segments might produce the greatest return (e.g., increased fish habitat and fish abundance) for an investment of stream restoration, whether to focus on relatively pristine or on heavily compromised sites, and whether to use passive (restoration of forest and stream processes) or active (artificial recruitment and/or retention of in-stream wood) restoration practices. As with any stream restoration, it is most effective to clearly establish target outcomes for restoration and then monitor the restored stream to evaluate achievement of targets [*Wohl et al.*, 2005]. The objectives and location of the restoration effort will likely determine whether passive or active measures are most appropriate.

The information summarized in this chapter suggests that primary limiting factors in the streams of the Colorado Front Range are relatively low wood loads and few jams in the middle and lower portions of the stream network as a result of timber harvest and other disturbances prior to 1900 A.D., continuing removal of wood in highly regulated stream reaches, and possibly relatively low population densities of beaver throughout the region. The most effective short-term response might be active reintroduction of large logs that are fixed in place, facilitating the formation of jams, and active reintroduction of beaver along stream segments capable of supporting beaver colonies (in some streams, such as those in Rocky Mountain National Park where riparian willows are heavily grazed by elk, this will also require grazing exclosures). Over the longer term, minimizing anthropogenic disturbance of riparian corridors and stream channels will facilitate gradual accumulation of dispersed wood and logjams in the rivers of the Front Range.

4. CONCLUSIONS

Forest and stream characteristics in the Colorado Front Range continue to experience widespread variability that will likely substantially impact in-stream wood loads. Climate change manifested in more rapidly melting snowpacks is causing the peak annual snowmelt flow to occur earlier each year [*Stewart et al.*, 2005], which may alter several variables in equations (1) and (2) (L_o, Q_i, Q_o, I_e, and I_{be}), and warmer winters may alter wood decay rates. Nearly a century of fire suppression has contributed to widespread and severe fires during the past decade and, along with warmer winters, may be exacerbating ongoing insect outbreaks that are killing thousands of hectares of conifers in the Front Range [*Romme et al.*, 2006]. Timber harvest in response to insect outbreaks is further altering forests that have not been harvested in more than a century. All of these recent changes will cause substantial spatial and temporal variation in wood recruitment over the coming decades, which must be considered when restoring in-stream wood.

Although the development of mechanistic models of in-stream wood dynamics in the Colorado Front Range is limited by lack of field data to parameterize key processes influencing in-stream wood load, the use of equations (1) and (2) to identify these processes, and their relative importance in different process domains helps to prioritize further research and to conceptualize longitudinal differences in wood dynamics and constraints to restoring wood loads. This approach can be combined with spatial information on fish populations, stream degradation, and stream recreational use or water supply to determine which portions of a stream network can be most effectively targeted for restoration of in-stream wood.

The types of field data, numerical simulations, and conceptual models outlined here can be applied to any stream network to estimate historical range of variation and to develop targets for stream restoration. An important component of this approach is to identify process domains relevant to in-stream wood dynamics and to document (via reference sites) or infer (via simulations) the magnitude of differences in wood loads among stream segments in these process domains. This information can then inform targets for restoration of in-stream wood.

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Geomorphic, Engineering, and Ecological Considerations When Using Wood in River Restoration

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This chapter provides an overview of wood in rivers, focusing on wood stability in rivers and design considerations for the reintroduction of wood to larger alluvial channels. Wood debris is a common component of the particulate matter in streams and rivers and has been recognized throughout most forested portions of the globe as an important factor influencing stream geomorphology and ecology. The stability and preservation of wood in large channels is primarily a function of its embedment in the streambed. The ecological benefits of wood are evident at several scales ranging from the wood surface to the complex interstitial space of wood accumulations (logjams), to the role of wood on altering bed textures and bed forms, to the influence of wood on channel planform, particularly creating multichannel systems. A logjam can increase available surface area for invertebrates and cover for fish by more than four orders of magnitude. A logjam can split flow and increase edge habitat severalfold. Logjams create pools and bars and raise water elevations to increase floodplain connectivity and have been placed in rivers with basal shear stress values of 166 Pa. Regardless of whether wood is included in a restoration design, as long as riparian trees grow along a stream, wood will end up in the channel; hence, it is also important to understand how naturally recruited wood behaves in rivers. Reintroducing wood to rivers brings up many other issues, from flood conveyance to public safety, all of which should be considered in the design process.

1. INTRODUCTION

Wood is a common component of the particulate matter in streams and rivers throughout the world. In many areas, wood comprises the largest individual particles found in the stream. In low-order streams, a single piece of wood can have dimensions easily exceeding those of the channel itself and create steps that can account for majority of the vertical drop of a channel [e.g., *Keller and Tally*, 1979; *Montgomery et al.*, 1995, 1996; *Montgomery and Buffington*, 1997; *Abbe*, 2000; *Abbe and Montgomery*, 2003]. In larger-order channels, a piece of wood can form the nucleus of much larger accumulations (i.e., logjams) that can redirect currents, alter channel planform, or even completely block the channel

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[e.g., *Abbe and Montgomery*, 1996, 2003]. Recognition of the geomorphic and ecologic role of wood has led to large-scale efforts to restore riparian forests and reintroduce wood into restoration and bank protection projects. Understanding the mechanics, dynamics, and persistence of wood in the fluvial environment is critical, not only in understanding how the system will respond to wood placements but also for the consequences to riparian forests and carbon storage in alluvial valleys. What then are the key variables contributing to the stability of wood debris? Understanding wood stability is central to understanding the ultimate fate of trees once they fall into a stream.

The geomorphology of a fluvial system is largely a function of its flow regime and sediment load. Of all the components of the particulate load, wood debris remains the least predictable with regard to the implications of how changes in the size distribution and supply of wood debris influence the system. We know that when individual pieces of wood are large enough, they can form stable obstructions that alter a river's course and can last for centuries [e.g., *Muir*, 1878; *Wolff*, 1916; *Guardia*, 1933; *Montgomery and Abbe*, 2006]. However, it is well established that wood alters rivers on a range of scales (Plate 1) and that changes in wood loading can alter sediment transport capacity, bed textures and channel morphology, and sediment transport [e.g., *Lisle*, 1995; *Abbe and Montgomery*, 1996, 2003; *Buffington and Montgomery*, 1999a, 1999b; *Manga and Kirchner*, 2000; *Brooks and Brierley*, 1997, 2004; *Cordova et al.*, 2006; *Magilligan et al.*, 2007]. Also, extensive literature exists on the role in-stream wood plays on aquatic ecosystem dynamics (see *Harmon et al.* [1986] and *Maser and Sedell* [1994] for an overview) and on some of the indirect relationships among channel morphology, wood, and processes such as hyporheic exchange flow [*Boulton*, 2007; *Stofleth et al.*, 2008; *Wondzell et al.*, 2009].

Since the 1990s, wood has become a significant component of river rehabilitation efforts [*Gerhard and Reich*, 2000; *Brooks et al.*, 2006; *Chin et al.*, 2008]. However, with the increased interest in wood reintroductions as a core river management activity within many government agencies come increasing concerns about appropriate design principles and appropriate monitoring of wood reintroduction activities and, indeed, all river management activities [*Dolloff*, 1994; *Bernhardt et al.*, 2005; *Wohl et al.*, 2005; *Mehan et al.*, 2006].

This chapter will review some of the attributes of wood in rivers before describing some of the key aspects of wood debris to consider in river restoration. Drawing on over a decade of experience in reintroducing wood to rivers on two continents, we will outline the basic elements of wood stability and design for controlling stream grade and flow patterns, present several large river examples, and offer guidelines for the reintroduction of wood into rivers, including its role in carbon sequestration. The approach to wood reintroduction that we outline is one that is strongly founded in understanding the role that wood has played in natural systems. However, we also show that it is possible to understand and analyze the role and performance of individual logs and log accumulations (logjams) through the common language of mathematics and physics.

1.1. Geologic and Human History of Wood in Rivers

The affinity between trees and rivers predicates the delivery of wood debris to the channel network. Wood, or evidence of wood, can be found in fluvial sediments deposited since trees appeared about 360 million years ago. During this time, they have not only left abundant evidence of their presence in the geologic record, but they have played an important role in the evolution of landscapes and biota. The geologic record shows that logjams began forming from the time woody plants first evolved [e.g., *Gastaldo and Degges*, 2007], contributing to the vast deposits of fossil fuels upon which human civilization is built. The fluvial sediments containing evidence for ancient wood also demonstrate that some of the wood stays within the fluvial system where it gradually breaks down or is preserved over long periods of time [e.g., *Hyatt and Naiman*, 2001; *Montgomery and Abbe*, 2006; *Guyette et al.*, 2008].

Trees found in both modern and ancient alluvium demonstrate that wood debris has been a part of fluvial systems at least through the Pleistocene and potentially has been a key mechanism for long-term carbon sequestration. *Guyette et al.* [2008] radiocarbon dated 200 tree boles exposed in eroding banks of eight streams in North Missouri, United States, and found that oak trees have been accumulating in alluvial sediments since the late Pleistocene 14,000 years ago. The median age of oak boles was 3515 years B.P. They found that the mean residence time for carbon was about 1960 years due to decreases in wood density over time as a result of reductions in cell wall thickness. The implications from this and other work documenting the longevity of wood in alluvium [e.g., *Brakenridge*, 1984; *Nanson et al.*, 1995; *Brooks and Brierley*, 2002; *Abbe*, 2000; *Montgomery and Abbe*, 2006] are important for considering the carbon-sequestering role of wood debris in floodplain management and wood reintroduction projects.

1.2. Influence of Wood Debris on Alluvial Systems

Integrating wood into river restoration involves an understanding of all aspects of fluvial geomorphology that

Plate 1. Wood debris acts on a wide range of scales from substrate and cover to channel planform and floodplain morphology.

Plate 2. Changes in the size of wood recruited to rivers influence geomorphic processes. (top) Recruitment of old growth along the Queets River in Olympic National Park introduces key pieces capable of redirecting flow. (bottom) Trees falling in the river from a forest plantation along the Hoh River outside Olympic National Park are easily washed away by the river. Both photos are looking upstream. The recruitment of key members provides a means of increasing bank roughening that reduces shear stress and erosion.

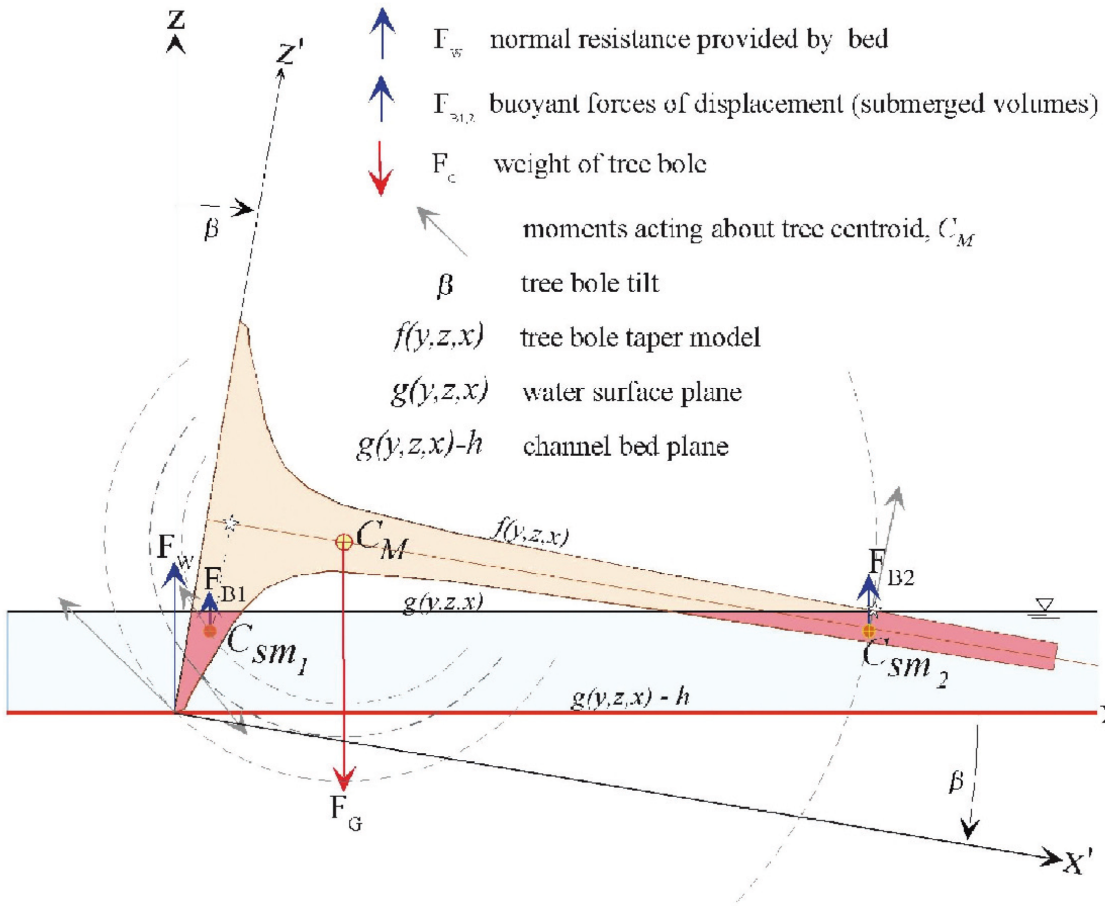

Plate 3. Free-body diagram for a snag. A root wad does several important things: (1) raises the center of mass, (2) increases the normal stress imposed on the streambed by reducing the log's footprint area, and (3) creates a bluff body flow obstruction that creates scour around the root wad that begins embedment in the streambed.

influence the recruitment, stability, transport, and effects of wood, including basin hydrology, channel hydraulics, sediment transport, channel dynamics, and riparian vegetation.

It has long been recognized in fluvial geomorphology that flow and sediment supply control channel morphology [e.g., *Lane*, 1955], but it has become increasingly apparent in some rivers that wood debris can be the dominant factor controlling channel morphology. Observations of early European settlers in North America did recognize the important role of wood debris, such as the large complex logjams of the southeastern United States that often created vast networks of impounded waters and bayous [e.g., *Lyell*, 1830; *Catlin*, 1832; *Veatch*, 1906; *Russell*, 1909; *Dacy*, 1921] to the role of a single fallen *Sequioadendron giganteum* log in creating the habitat that nurtured these trees high in the Sierra Nevada of California [*Muir*, 1878]. A great deal of effort was exerted by the U.S. government over more than a century to clear wood from rivers [e.g., *Ruffner*, 1886; *Sedell and Frogatt*, 1984; *Collins and Montgomery*, 2002]. Similar efforts were expended in other New World countries such as Australia, where active wood removal programs persisted from the 1800s up to the 1990s [*Erskine and Webb*, 2003; *Brooks et al.*, 2006]. The impact of these actions was to alter the energy gradient and morphology of rivers subjected to this treatment [e.g., *Guardia*, 1933; *Hartopo*, 1991; *Brooks et al.*, 2003].

The wood from riparian forests was an essential resource in the development of every human civilization, providing energy and the fundamental building material for shelter, transportation, and industry [e.g., *Williams*, 2003]. Human development was often focused in river valleys, and riparian forests were often the first to be cleared. In a wide range of climates, it is these areas along streams and rivers where trees thrived and attained the most impressive size. It was these same trees that were the source of timber recruited to streams and rivers via a range of mechanisms [*Abbe*, 2000; *Collins and Montgomery*, 2002; *Abbe and Montgomery*, 2003; *Benda et al.*, 2002; *Fox and Bolton*, 2007]. Hence, not only has wood been historically removed from channels for navigation and flood conveyance, but riparian sources of wood have been significantly altered or eliminated. Historic changes in the characteristics of riparian trees recruited to rivers have also had an influence on the stability of wood in rivers, with the general trend of much smaller, more mobile large wood debris (LWD) loading (Plate 2). Stable LWD accumulations directly affect the retention of smaller mobile LWD and, thus, the overall wood budgets of rivers.

2. WOOD STABILITY

About 100,000 species of trees, making up 25% of all vascular plants [*Raven and Crane*, 2007], are estimated. The vast range of trees come in many shapes and sizes and have adapted to a wide range of environmental conditions. Trees are the largest individual pieces of debris entering most streams and rivers and can have a pronounced influence on the conveyance of water and sediment. The shape and size of wood is a key attribute that contributes directly to its stability and function in streams and rivers [*Abbe and Montgomery*, 1996; *Abbe*, 2000; *Braudrick and Grant*, 2000; *Manners et al.*, 2007]. So how is it that a material that often has a specific gravity of less than unity (i.e., it floats) can remain stable in a river for long periods of time?

In circumstances where single-stem trees dominate, simplifying the shape of the tree can be a useful way of evaluating the forces that act on a piece of wood. Using a simple cylinder may be adequate for evaluating some wood used in restoration projects, but all trees have a tapered trunk (or bole) due to buttressing near the ground (Plate 3). Tree trunk buttressing can have a significant influence on the shape of a log and the centroid location of a snag [*Abbe et al.*, 1997; *Abbe*, 2000]. The presence of a root wad is one of the most important factors influencing wood stability. A simple expression for a snag taper exponent, a, using the bole radius, R_b, measured distance X_i from the base of the root wad with a radius of R_{rw} is given in equation (1):

$$a = \frac{\log(R_b/R_{\text{rw}})}{\log(X_i - 1)}. \tag{1}$$

To estimate how far the bole's center of mass is from the root mat (distance along x axis, x_c) the moment of volume with respect to x (M_x) is divided by bole's volume, V:

$$x_{c_m} = \frac{M_x}{V} = \frac{(2a+1)(x^{2a+2}-1)}{(2a+2)(x^{2a+1}-1)}, \tag{2}$$

$$M = \pi\int_{x=1}^{x_n} x(R_{\text{rw}}x^a)^2 \mathrm{dx} = \frac{\pi R_{\text{rw}}^2}{2(a+1)}(x_n^{2(a+1)}-1), \tag{3}$$

$$V = \pi\int_{x=1}^{x_n} (R_{\text{rw}}x^a)^2 \mathrm{dx} = \frac{\pi R_{\text{rw}}^2}{2a+1}(x_n^{2a+1}-1). \tag{4}$$

Assuming the log is resting on a level surface, its tilt will be a function of its length and radii at either end (root wad, R_{rw}, and crown, R_n). The centroid elevation, z_{c_m}, for this simple model is

$$z_{c_m} = \left\{R_n\left(\frac{R_{\text{rw}}-R_n}{x_n-1}\right)x_n - x_{C_m}\right\}\sin\left\{\tan^{-1}\left(\frac{R_{\text{rw}}-R_n}{x_n-1}\right)\right\}. \tag{5}$$

Centroid locations of submerged portions of a log upon which buoyant forces act can be determined through a numerical integration of the volume defined by the log's intersection with the relevant water surface elevation. A basic hydrostatic analysis is the first step in evaluating the stability of a piece of wood or tree bole. The water depth at which a log becomes fully buoyant, $F_B = F_G$, is referred to as the buoyant depth, h_{b}, and commonly corresponds to the log's maximum draft, d_m. The draft of the log relative to a particular flow depth is critical since it will influence the frictional resistance the log encounters along the channel boundaries. A bed form or roughness element upon which a log comes in contact can provide a resisting force equal to the driving forces, thus stabilizing the log. The hydrostatics of a tree lying on its side is a very different situation than a tree stump sitting upright. In the case of a tree stump, a large portion of the wood volume is displaced with relatively little water depth and the centroid (center of mass) is relatively low to the ground. Thus, a tree stump has a relatively shallow buoyant depth. But in the case of a snag with an intact root wad lying on its side, the centroid is typically situated higher above the ground. In the latter case, rising water displaces a relatively small volume of the snag because the root wad elevates the bole above the ground, so a snag has a greater buoyant depth when lying on its side versus sitting upright. The relatively high buoyancy and low potential for embedment make stumps a poor choice for LWD placements.

With the stem or bole taper determined, the volume of the buttressed end of the snag can be estimated:

$$V_{\text{rw}} = \frac{\pi R_{\text{rw}}^2}{2a+1}(X_i^{2a+1}-1). \tag{6}$$

This estimate of volume can work for the "stump" portion of the logs. For a single-stem straight tree the volume (V_b) above the stump (above the basal radius of the bole, R_b) can be estimated assuming a truncated cone or frustum of length L_b:

$$V_b = \frac{\pi L_b}{3}(R^2 + Rr + r^2). \tag{7}$$

The total snag volume, V_w, is the sum of V_{rw} and V_b. The dry weight of a snag is determined using the dry density of the wood.

$$W_{wd} = g\rho_w V_w \tag{8}$$

where

W_{wd} weight of log;
g gravitational constant;
ρ_w wood density;
V_w volume of wood.

Buoyancy will be defined on the submerged volume of the snag, Vs_w, which can be used to estimate the snag's buoyancy relative to the amount of submergence.

$$W_{ws} = g[(\rho_w V_w) - (\rho_f Vs_w)], \tag{9}$$

where W_{ws} is submerged weight of log, ρ_f is fluid density, and Vs_w is submerged volume of wood (displacement).

A negative value for the submerged weight indicates the log is buoyant. As the size of wood increases, so do its weight, strength, and the height of its centroid. Thus, it takes deeper water to float it and stronger currents to drag or break it. But size also means greater buoyant forces, and thus, a greater extent of burial is required if the snag is to remain stable. Since buoyancy depends on the weight of the water displaced, a large tree can exert significant buoyant forces if submerged, depending on its specific gravity (if greater than 1, the tree will sink). Even if buoyant, a tree may not move down the river if it encounters sufficient resistance along the riverbed, just like a grounded ship. The partially buried root wad of a buoyant tree (just like the keel of a sailboat) will encounter passive earth pressures that can be sufficient to halt its movement [*Abbe et al.*, 2003a].

After a tree falls into a river, the key to its stability will rely on whether it becomes embedded into the channel. Thus, the snag has to remain stable after bed load transport has been initiated and have sufficient weight to sink into the riverbed. The presence of a root wad and elevated centroid are critical for this process to proceed. A snag is most stable with its root wad facing upstream, forming a bluff body in the river flow. The root wad of a snag adds significant draft to the wood and, thus, drags upon the riverbed. The floating tip of a snag will be most stable in the lee of the root wad [*Abbe and Montgomery*, 1996]. Thus, the stable configuration of a snag with the root wad facing upstream forms a bluff body to incident flow [*Abbe*, 2000]. A bluff body is the opposite of an aerodynamic form. As flow goes around a bluff body, it separates around the edges to form a turbulent zone called a Von Karmen vortex street. Between each vortex street is the flow separation envelope commonly referred to as an eddy. Within this eddy, bed material can accumulate and begin to bury the back side of the root wad, adding passive earth pressure resistance to the drag acting on the snag. If a snag remains stable under the flow conditions (depth and velocity) that mobilize the substrate, the root wad will settle into the adjacent scour hole. Only a small amount of burial is required for the snag to become stable in flows that would otherwise have caused mobilization [*Abbe*, 2000; *Abbe et al.*, 2003b]. Since the stems of the key pieces initiating a wood accumulation [*Abbe and Montgomery*, 1996] are typically located within the flow separation envelope, they become buried (Plate 4). A buried snag initiates a flow obstruction that can trap mobile debris and lead to bar formation that can go on to develop into a floodplain island [*Abbe and Montgomery*, 1996]. The natural process by which a snag embeds itself into the riverbed is fundamental for understanding wood stability in large channels [*Abbe*, 2000; *Abbe et al.*, 2003a].

Despite the vast number of tree species, the range of specific gravity is relatively low when compared to rock. The specific gravity of all woods can never exceed cellulose and lignin creating the solid wood material, which is 1.54 [*Skaar*, 1988]. Since wood originates as living tissue, it must have some porosity to transmit water and nutrients. Thus, the specific gravity of the densest woods (dry) does not exceed 1.37 for Lignum vitae (*Guajacum sanctum*) and can be as low as 0.16 for balsa (*Ochroma lagopus*). The relatively low specific gravity of wood (often <1) when compared to rock (>1) is one of the principal perceptions that can influence the application and management of wood in rivers. To assume that all wood floats, however, would be a mistake, just as equating buoyancy with instability would be a mistake.

The specific gravity of a piece of wood depends on its porosity and moisture content. If wood is completely saturated, the specific gravity must of course be greater than water and less than the wood substance. Moisture content varies the greatest within the long cell cavities (lumen) and the least in the cell walls that comprise the wood or xylem. The maximum possible moisture content is dependent on density of the wood structure, which is reflected in the basic specific gravity of different species (Figure 1). Determining the weight, volume, specific gravity, and moisture content of wood is a fundamental step in designing with wood.

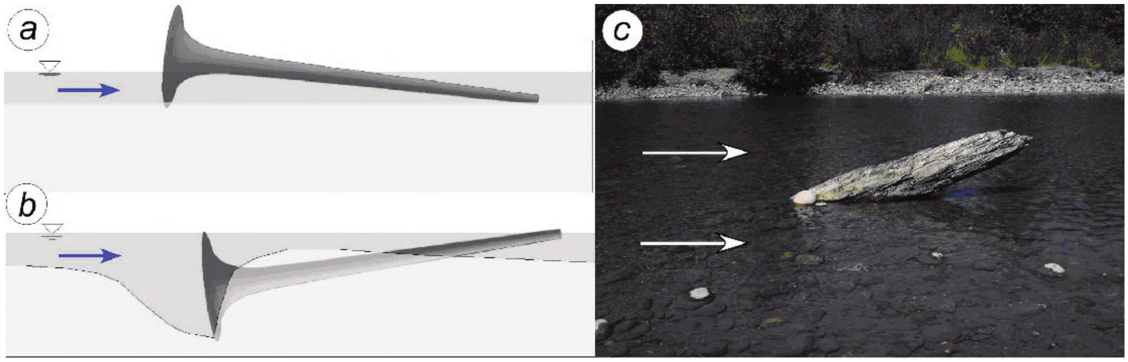

Plate 4. (a) If a snag remains stable after bed load transport has been initiated, (b) scour can begin process by which the snag is buried into the streambed. (c) Buried snags will always have their tips pointed downstream and create formidable obstructions within the river.

$$W_g = W_d\left(1 + \frac{M}{100}\right) \tag{10}$$

$$M = \left(\frac{W_g}{W_d} - 1\right)100, \tag{11}$$

where W_g is weight of green wood, W_d is oven dry weight of wood, and M is moisture content of wood.

Wood density, ρ_w, is determined by

$$\rho_w = \left(\frac{W_d}{V_g}\right)\left(1 + \frac{M}{100}\right), \tag{12}$$

$$G_b = \left(\frac{W_d}{V_g}\right)\left(\frac{1}{p_w}\right), \tag{13}$$

$$\rho_w = G_b\rho\left(1 + \frac{M}{100}\right), \tag{14}$$

where ρ is density of water, V_g is green volume of wood, and G_b is "basic" specific gravity [*Simpson*, 1993].

The maximum moisture content (%), $M_{\max}$, is expressed as a function of the wood porosity $(1 - \gamma_b/\gamma_w)$. Thus, the denser the wood, the lower the maximum moisture content.

$$M_{\max} = (100/\gamma_b)(1 - \gamma_b/\gamma_w), \tag{15}$$

where γ_b is basic specific gravity of tree species and γ_w is specific gravity of the wood material (cellulose and lignin) equal to 1.54.

The maximum moisture content for western red cedar (*Thuja plicata*) has a relatively low specific gravity (γ) of about 0.35 and has maximum moisture content of about 220%. Douglas-fir (*Pseudotsuga menziesii*) has a specific gravity of about 0.55 and a maximum moisture content of about 120%. Australian red mahogany (*Eucalyptus resinifera*) has a maximum moisture content of only 40% due to its high specific gravity of 0.96. The maximum moisture content of one the world's hardest woods (γ = 1.35) Lignum vitae (*Guaiacum* spp.) is only 10%. When saturated, even low-density woods can sink, as reflected in the many logs found at the bottom of mill ponds throughout the world. When designing with wood, the conservative assumption is to use the dry or green density of the wood in force balance calculations. If it is known that the timber will remain submerged; a stability analysis can be undertaken using a less conservative value for the timber density.

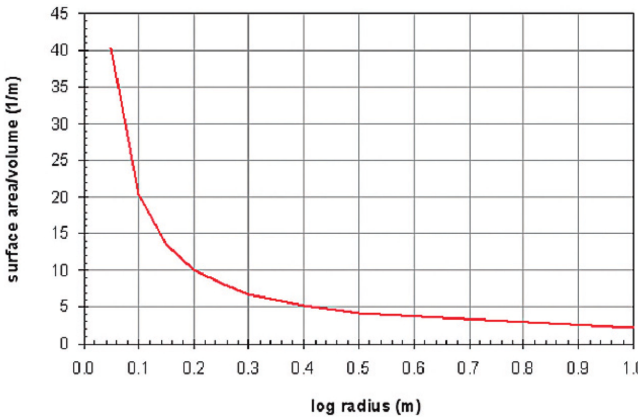

Plate 5. The ratio of surface area to volume declines dramatically as a function of the log radius (independent of log length). The greater the surface area to volume, the more rapid the exposure of the log to decay.

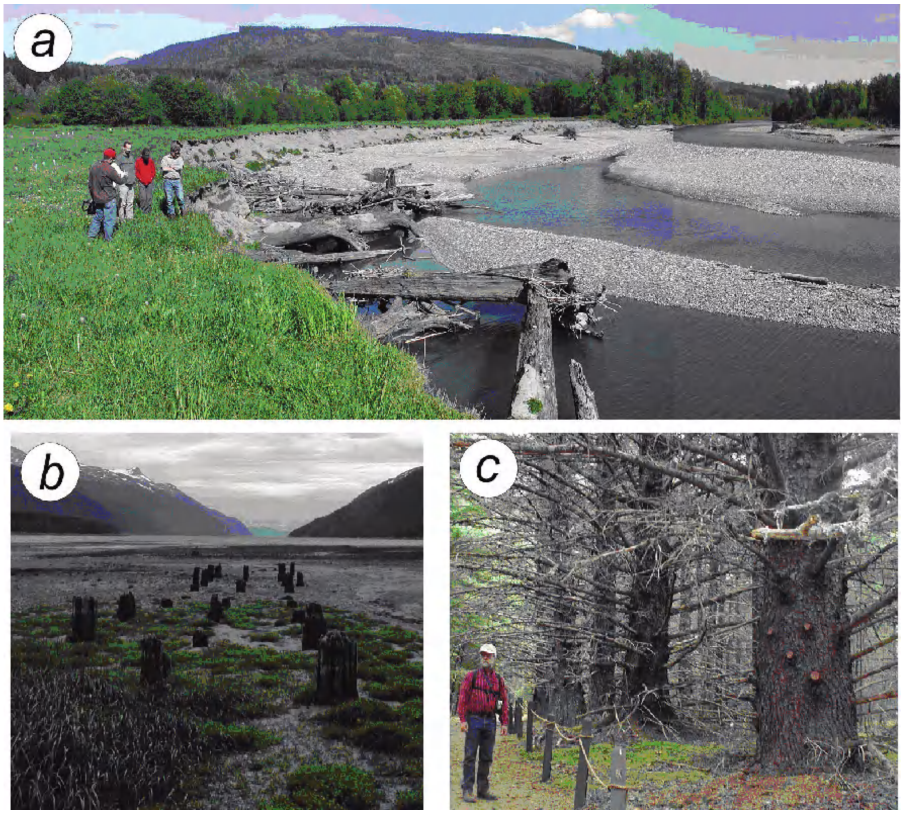

Plate 6. Examples of wood longevity: (a) 2003 exposure of buried logjam along South Fork Nooksack, Washington, over 118 years old based on the fact that a river never historically occupied this area [*Collins and Sheikh*, 2004], (b) remains of 110 year old timber piles in Dyea, Alaska, at outlet of Taiya River (2002), and (c) 110 year old Sitka spruce trees in Dyea (2002).

It should also be recognized that wood density will vary through time, both on short time scales of hours to weeks, with wetting and drying cycles, and over periods of years as the timber decays. The short-term variation in density is a function of the difference in the moisture content of the wood, which is a function of the proportion of intercellular pore spaces that contain either water or air. The longer-term variation in density associated with timber decay is a function of an increase in porosity as the lignin and cellulose decays. Figure 2 shows some experimental data using pieces of river red gum (*Eucalyptus camaldulensis*) showing the wood density under field conditions during drought (considered to be the worst-case conditions) and then after oven drying for 24 h (the basic dry density) as well as the following 6 and 18 h saturation. These data show that even in the case of relatively dense Australian eucalypts, when it dries out in the field, this timber can become buoyant. However, even after 6 h of immersion, it is possible for these timbers to increase their specific gravity to the extent that they are no longer buoyant. Hence, it is clear that the moisture content is a critical variable for understanding the stability of individual logs in a river. Furthermore, the same species of timber might behave very differently in two rivers depending on the hydrologic regime. Timber within a river having a stable base flow and in which flood waters rise gradually may never dry out and will consequently be more stable than the same tree in a very flashy ephemeral channel where a piece of wood may be completely dry and then completely submerged in a matter of hours.

3. WOOD LONGEVITY

Two types of stability exist with regard to wood: mechanical and biogeochemical. The first involves the ability of a piece of wood to resist the forces that would move or break it. The second involves the decay or breakdown of the wood material. Both types of stability lead to common and legitimate questions in river engineering and restoration, and both can be addressed. Mechanical stability can be evaluated using a force balance approach. Decay can be addressed based on a set of assumptions. The certainty to which predictions can be made regarding either condition depend on

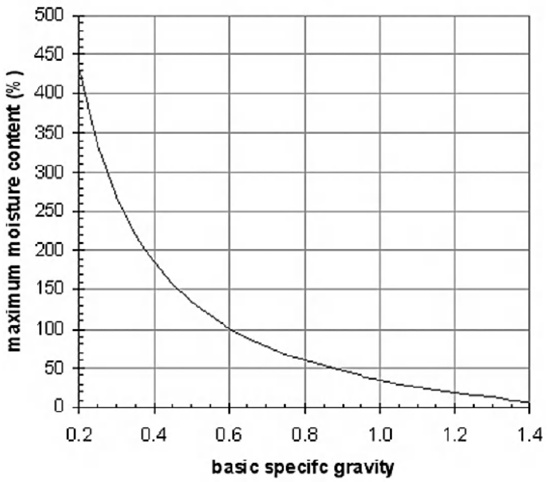

Figure 1. Maximum moisture content as a function of basic specific gravity [*Simpson*, 1993].

the quality of the input data and validity of the assumptions. An individual piece of wood ("log") is stable in fluvial environments under one of two conditions: (1) the wood is large enough to be locked into place within the channel either between banks or against preexisting obstructions such as boulders or trees or (2) the wood is embedded within alluvial sediments.

In the first case, the wood must be situated such that it cannot rise above the obstruction during high water or break under the drag force imposed by flowing water. The first condition is commonly found in low-order headwater areas where wood is large relative to the channel. The second condition is found in large alluvial channels where an individual piece of wood is small relative to the channel geometry.

An important question in any wood debris restoration work regards the longevity of the wood. Wood can last virtually indefinitely under two scenarios, when kept under anaerobic (submerged) condition or perfectly dry. Obviously, the latter condition will not be found in rivers, but the former does occur, although the most common state is likely to be one of wetting and drying. When wood is saturated year round, it can be remarkably well preserved and lasts for hundreds and even thousands of years and plays an important role in structuring alluvial rivers and forested floodplains [e.g., *Nanson et al.*, 1995; *Abbe*, 2000; *O'Connor et al.*, 2003; *Montgomery and Abbe*, 2006; *Fox and Bolton*, 2007; *Magilligan et al.*, 2007]. Because wood floats and is subject to decay, it is often believed that wood should not be put in rivers; this common perception has hampered the integration of wood in restoration. This perception fails to take into account the geologic history of wood in rivers, including the last 6000 years in which wood has been an integral part of aquatic environments. Wood in rivers can last for a very long

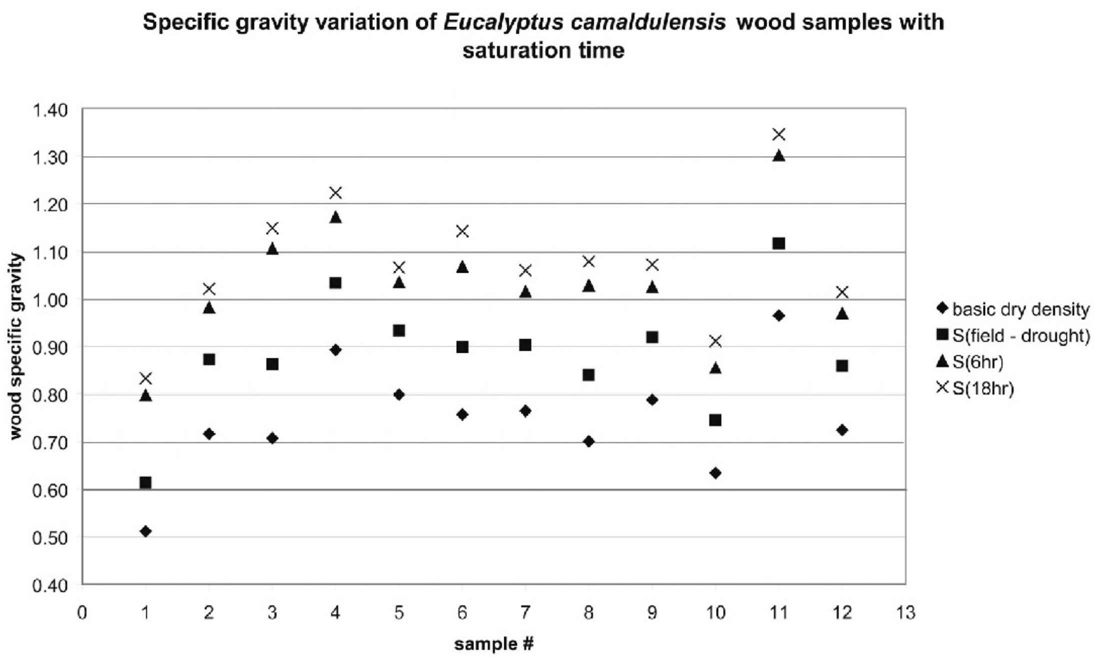

Figure 2. Some experimental data using pieces of river red gum (*Eucalyptus camaldulensis*) showing the wood density under field conditions during drought (considered to be the worst-case conditions) and then after oven drying for 24 h (the basic dry density) as well as the following 6 and 18 h saturation.

time, depending on the tree species and the conditions of preservation. Wood situated above the base water surface will be subjected to biological decay and physical breakdown associated with wetting and drying and abrasion by transported sediment. Thus, the type of wood and size of log play a dominant role in its decay. Certain woods are chemically predisposed to excellent preservation. In general, however, the larger the log, the greater the longevity, since the ratio of surface area to volume decreases as log diameter increases (Plate 5). Conversely, the higher the ratio of surface area to volume, the faster the decay [e.g., *Spanhoff et al.*, 2001], so it is always advantageous to use larger logs for the structural foundation of any in-stream wood structure.

Many examples can be found throughout the world to illustrate the preservation of wood relative to the water table. Wood in alluvial sediments can be subjected to a wide range of decay agents that can break down the structural integrity of a log. But in the right depositional conditions, wood debris can last for thousands of years. In the case of restoration, field inspections within a project area can reveal evidence of relic logjams exposed in eroding banks (Plate 6a). These ancient structures typically consist of the key pieces that initially formed the logjam. The piles beneath St. Mark's in Venice, Italy, were so well preserved below the ground water level after 1002 years; they were left in place to support the reconstructed tower and determined to have an "indefinite" life [*Jacoby and Davis*, 1941, p. 81]. A similar phenomenon can be observed where old pilings are exposed in rivers and estuaries, such as the wharf pilings of the ghost town of Dyea, Alaska, constructed in 1898. The pilings below ground level remain in good condition after more than a hundred years, sufficient time for trees planted on the river's floodplain to obtain substantial size (Plates 6b and 6c). A simple model of wood decay can provide a basic guideline for estimating longevity. The model assumes cylindrical log geometry with homogeneous decay and is very sensitive to an assumed decay exponent (Figure 3), which varies substantially between tree species and the depositional setting. The mass of a log at time t can be predicted using the following:

$$M(t) = M(0)e^{k(t-t_0)}, \tag{16}$$

where

$M(t)$ mass at future time t;
$M(0)$ mass at time of placement;
k decay coefficient;
t future time;
t_0 starting time.

On the basis of the assumed decay rate, log mass and diameter can be predicted for a given time frame, thereby

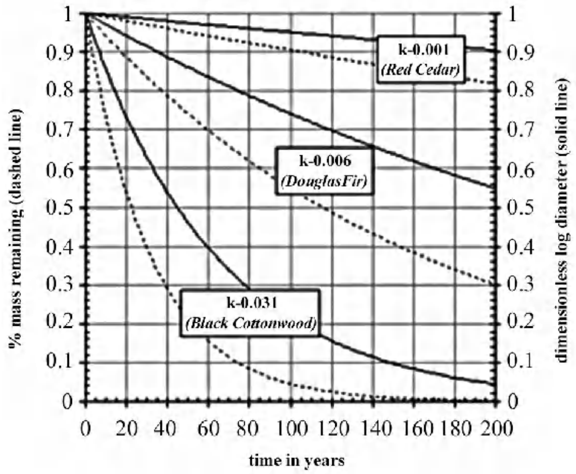

Figure 3. Simple decay model for cylindrical logs with spatially uniform decay. Decay rates (k) are taken from *Harmon et al.* [1986] for forest floor logs and thus are conservative for timber situated in a stream or river. The curves are for three common tree species in the Pacific Northwest that show a wide range in susceptibility to decay, ranging from western red cedar (*Thuja plicata*) to Douglas-fir (*Pseudotsuga menzisii*) to black cottonwood (*Populus trichocarpa*). After 120 years, a Douglas-fir log would lose about 50% of its mass and have an effective diameter of about 70% of its original.

allowing an assessment of the structure's integrity for specific design lives. Forest floor decay rates, k, of common Pacific Northwest species range from 0.001 for red cedar (*Thuja plicata*) to 0.006 for Douglas-fir (*P. menziesii*) to 0.031 for black cottonwood (*Populus trichocarpa*) [*Harmon et al.*, 1986]. It is likely that decay rates are higher in warmer climates but may be offset in the case of timbers that are more resistant to decay such as gum trees, mahogany, and ironwood. Decay rates are very much dependent on a variety of environmental settings, physical condition of the wood, and agents of decay (e.g., bacteria, fungus, and termites). When exposed to these agents, wood may only last several decades [e.g., *Hyatt and Naiman*, 2001]. In soils where wood is susceptible to wetting and drying, restoration design should carefully consider potential biochemical degradation and whether the wood will achieve the desired design life. As wood decomposes, it rapidly loses strength, which may be important in using posts or piles to provide lateral resistance. The role of wood decay in the failure of natural and engineered wood structures is unknown and, thus, an important area for additional research. The current engineering practice in restoration assumes structurally sound timber, an assumption that while valid today, may not be valid in 25 years. As wood decays, strength is lost more rapidly than mass [*Abbe*, 2000], so it is wise to err on the

Plate 7. In small headwater streams of the Puget Sound Lowlands, wood can account for much of the creeks' head loss and sediment storage. In this 12% gradient segment of Schmitz Creek in west Seattle, Washington, historic "relic" wood buried in the alluvium accounts for over 90% of the head loss and helps stabilize banks despite heavy foot traffic (July 2009 photo).

side of larger timber whenever possible. In critical sites, it may be worth considering environmentally sensitive wood treatments for structural elements such as piling. The key to rehabilitating wood in rivers is ensuring riparian forest conditions will ultimately negate the need for in-stream wood placements. Logjams can play an important role creating riparian forest refugia within active channel migration zones [*Abbe and Montgomery*, 1996; *O'Connor et al.*, 2003]. The design life of wood structures built in rivers and floodplains should allow sufficient time to reestablish functional wood recruitment on and adjacent to LWD structures.

The other critical aspect of designing longevity for restoration projects involves replacement; once an individual structure reaches its design life, will its function be adequately replaced by the restored riparian forest? Replacement should be the long-term goal of all restoration projects. That is, we are restoring the process role of wood, not simply building engineered structures. If riparian conditions cannot be restored to conditions that will sustain the function of the original wood structures, then it should be made clear that future work, from maintenance to new construction will be required. In situations where it is impossible to fully restore riparian forest conditions with associated wood recruitment, and longevity of individual structures is critical, the focus should preferentially be on function rather than materials. In difficult environments, such as urban creeks, where a high factor of safety and longevity are paramount, materials such as concrete logs or steel piles can be used, as long as the completed structure emulates the desired function of natural wood. Real wood debris (particularly racking material) can be integrated with these other materials to provide the desired biological attributes and visual aesthetic. In some urban environments "relic" wood, comprised of large old logs, is all that prevents some creeks from undergoing severe incision and bank erosion, even despite dramatic increases in the magnitude and frequency of peak flows (Plate 7). These relic logs

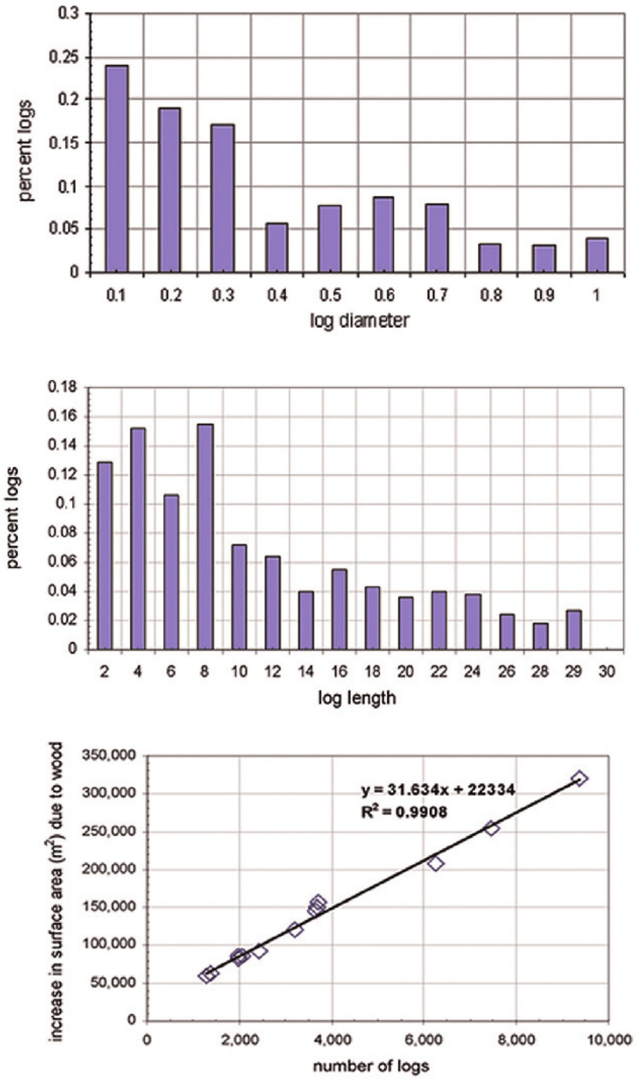

Plate 8. (top and middle) Example distribution of racking logs in a logjam by diameter and length and (bottom) the resulting increase in surface area based on number of racked logs. The last chart shows how this distribution of log sizes creates surface area within the river as the number of logs increase in the jam. This example is conservative since it does not include fine organic debris (diameters <0.1 m), which would increase the surface area significantly.

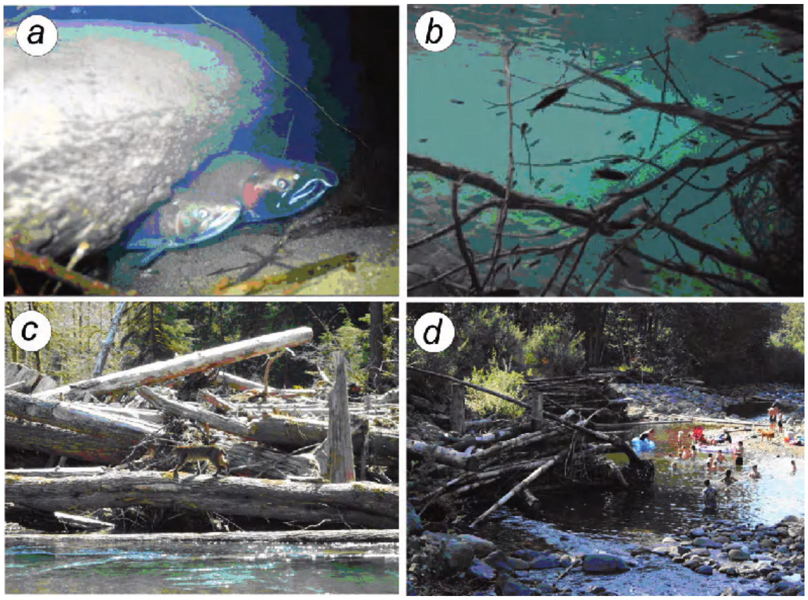

Plate 9. Logjams form the most complex habitat found in rivers, forming pools, bars, and cover for all sorts of species. The interstitial spaces within the structures offer (a and b) cover for fish and (c) river access for predators and (d) create pools that humans enjoy during hot summers (Mashel River engineered logjams (ELJs) in Smallwood Park, Eatonville, Washington). Plate 9a courtesy of G. Pess, Plate 9b courtesy of Wild Fish Conservancy, and Plate 9c courtesy of P. Caton.

are typically large and composed of wood that is more resistant to decay (e.g., cedar or tight grain old growth). If these logs are not ultimately replaced, the creek may be at serious risk of incision. This same principle is needed in restoration projects where engineered structures are placed.

4. WOOD COMPLEXITY AND HABITAT

By adding wood roughness to a channel, shear stress is partitioned among the channel form, sediment, and wood, thereby reducing the effective shear stress available for

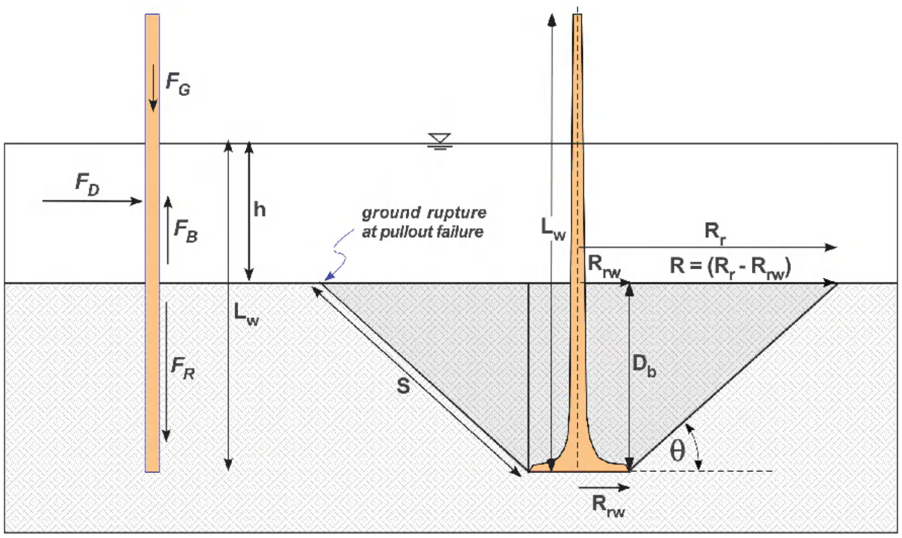

Plate 10. Timber piles and posts. A traditional driven pile (left) consisting of a vertical cylinder that relies only on skin friction for resisting buoyancy when fully submerged. A buried root wad post benefits from additional surcharge of overlying alluvium. Forces acting on an embedded root wad pile or post (right) are the same as a simple pile (left), with the addition of the geostatic load of the alluvium.

sediment transport, which consequently reduces the overall bed grain size [*Manga and Kirchner*, 2000]. Even small amounts of wood debris can have a significant effect on bed textures, thereby modifying aquatic habitat [*Buffington and Montgomery*, 1999a, 1999b]. On a larger scale, logjams form bluff bodies that alter flow patterns within a channel [*Abbe and Montgomery*, 1996]. A logjam structure introduces a unique substrate to the stream ecosystem (wood) in concentrated and complex assemblages that have been found to be heavily used by invertebrates [*Coe et al.*, 2006] and fish [*Peters et al.*, 1998].

The surface area of individual pieces of wood and accumulations has important implications with regard to biological productivity that can affect wood decay and the amount of habitat availability [*Wondzell and Bisson*, 2003; *Coe et al.*, 2006]. Two simple principles apply with regard to surface area available to invertebrates and other crucial organisms: (1) the smaller the tree, the greater the surface area relative to the tree's volume (Plate 5), and (2) the more trees in a logjam, the greater the surface area. Large tree stems with attached root wads are key to structure stability and longevity, but small debris is key to creating the complexity, substrate, and cover to enhance the food web [*Coe et al.*, 2006]. Smaller wood has a higher ratio of surface area to volume and, thus, is prone to higher decay rates since decay is proportional to both variables (increases with surface area and decreases with volume). When accumulations of small debris form against larger key pieces, they not only greatly enhance the ecologic functions of the structure but they can also improve stability by reducing scouring flow through the key members. The largest logjams form on larger rivers where massive accumulations of smaller, more mobile debris accumulate (Figure 4). The accumulations of wood debris not only split up the river flow, but they also create entire ecosystems within the river.

Logjams can introduce a tremendous amount of physical complexity and organic substrate within a river. Modeling debris as simple cylinders, and assuming a random distribution of sizes representative of the material entering the river, a logjam of 1000 logs can have a surface area of over 60,000 m^2, while an accumulation of 10,000 logs will have a surface area of over 300,000 m^2 (Plate 8).

Because the wood in a logjam is composed of a broad distribution of sizes and shapes, it creates a complex matrix with a wide range of interstitial spaces that can accommodate a commensurate range of organisms of various sizes. In addition to the range of interstitial area is a range of hydraulic conditions and lighting. A logjam is similar to a densely populated urban area of many different tenements. *Peters et al.* [1998] observed that both juvenile and adult fish seek refuge within logjams during the day (Plate 9). *Coe et al.* [2006] found that invertebrate densities are much greater within logjams when compared to alluvial banks. Moreover, because logjams extend above the water, they form excellent habitat for birds and mammals (Plate 9). *Harvey et al.* [1999] found that fish holding at large woody debris accumulations were less likely to move away from the wood over varying flows as opposed to fish using portions of a stream without obstructions. In the Williams River wood reintroduction experiment in southeastern Australia, the numbers of Australian bass (*Macquaria novemaculeata*) found within the confines of a single logjam on one sampling occasion exceeded those found within the entire 1.1 km study reach, over seven sampling occasions, across several years [*Brooks et al.*, 2006].

5. DESIGNING WOOD DEBRIS STRUCTURES

Engineered logjams (ELJs) have been widely used in the Pacific Northwest of North America over the past decade, as

Figure 4. Wood accumulation on a constructed logjam in the Hoh River in 2008, 4 years after construction. Person in foreground gives scale of logjam. The ELJ has accumulated several thousand of pieces of debris ranging in length from 1 to over 20 m and diameters from 0.1 to 1 m. The logjam increases the surface of cover and organic substrate by over 100,000-fold.

well as in Australia, as an alternative, more sustainable approach to river management [*Brooks et al.*, 2004, 2006; *Brooks*, 2006]. In particular, they have been used for bank protection and habitat enhancement in high-energy gravel bed rivers supporting migratory species of Pacific salmon. Two general types of wood structures exist: (1) grade control and (2) flow deflection. The focus of this chapter is the latter. Grade control structures are predominantly found in small to moderate-sized channels where log lengths equal or exceed channel widths. In these systems, wood can be a very important structural component in dissipating energy and capturing sediment. Flow deflection structures are typically used in large alluvial systems where channel widths exceed log lengths.

Distinct types of logjams, or in-stream woody debris accumulations, are found in different parts of a channel network [*Abbe et al.*, 1993; *Wallerstein et al.*, 1997; *Abbe and Montgomery*, 2003; *Comiti et al.*, 2006; *Andreoli et al.*, 2007; *Baille et al.*, 2008]. Using observations from the Queets River basin on the Olympic Peninsula in Washington, distinct types of logjams have been classified according to the presence or absence of key members, source and recruitment mechanism of the key members, logjam architecture (i.e., log arrangement), geomorphic effects of the logjam, and patterns of vegetation on or adjacent to the logjam [*Abbe et al.*, 1993; *Abbe*, 2000; *Abbe and Montgomery*, 2003]. Six of these logjam types provide naturally occurring templates for ELJs intended for grade control and flow manipulation. Logjam types primarily applicable for grade control include log steps and valley jams; types more applicable for flow manipulation include flow deflection, bankfull bench, bar apex, and meander jams [*Abbe et al.*, 1993; *Abbe and Montgomery*, 2003; *Abbe et al.*, 2003b, 2003c].

The number of different architectures for these structures is infinite, but all will be subjected to similar processes, and their structural integrity is based on the same set of principles. We have compiled a planning framework for wood projects including objectives, opportunities, constraints, and project elements. We then briefly describe some of the key factors influencing wood stability and wood function to consider in designing each type of structure and present examples.

5.1. Project Planning

Before delving into the specifics of designing an individual wood structure, it is critical to assess the site and understand the geomorphic, hydrologic, hydraulic, ecological, and human context of the project. This assessment will all go into clearly defining the project goals and constraints, which in turn will influence structure design. Wood and wood structures are just one part of river restoration and management; hence, a much more comprehensive view of the system, from the physical processes to the politics, will be crucial to implement successful projects. All projects should be designed to accommodate the physical and biological processes to which the project will be subjected and emulate natural self-sustaining structures. ELJ technology [*Abbe et al.*, 1997, 2003c] was developed out of recognition of the natural role of logjams, particularly in their ability to form “hard points” in large alluvial rivers and was applied to river management. The philosophic elements of this approach are mirrored in the emerging field of “biomimicry” [*Benyus*, 2002]. A better understanding of natural processes and structures, exemplified by wood in rivers, offers plenty of opportunity in civil engineering and landscape architecture to develop much more sustainable long-term approaches to land management.

5.2. Project Design

The design process used for ELJ structures follows a formal geotechnical and civil engineering design approach similar to that used in traditional infrastructure development. The design process includes a formal quality assurance and quality control program, a reach analysis, data collection and verification, the establishment of a design basis, modeling, iterative design development with risk assessment, constructability and cost, public relations efforts and education, regulatory approval, and contract package development.

A reach analysis provides the necessary background information on historic and current conditions including channel geometry, substrate, hydrology, hydraulics, wood loading, and disturbance processes. Risk assessments can be relatively brief for projects with no risks to property, infrastructure, or life and can be extensive for projects with potential risks. In either case, a risk assessment should include all aspects of the project (Plate 10). Initially, the results of the reach analysis (including a geomorphic analysis and a review of field data) serve as the platform for determining the risk associated with the preliminary conceptual plan. The description of historical channel dynamics and flooding formulated during the reach analysis is essential for documenting preexisting conditions and risks at the project site if no ELJs were constructed. A reach analysis must be performed at spatial and temporal scales that are adequate for describing these relationships. Conceptual design alternatives are prepared, and a feasibility analysis is performed to compare the habitat benefits, cost, and initial risk associated with achieving the performance objectives of the project with each of the design alternatives.

If the results of the risk assessment indicate that the preliminary conceptual plan falls within an acceptable range of risk and meets the goals of the project, the preliminary conceptual plan then undergoes a hydraulic and scour analysis. Hydraulic modeling is done to evaluate flow regimes under current conditions and under potential build-out scenarios. In a geomorphic reach analysis, areas of physical constraints are identified and demarcated. These areas are then incorporated into the design alternatives; for example, differentiating areas within the channel migration zone where the main stem channel can freely move, areas in the channel migration zone where only secondary channels are acceptable, areas that can tolerate inundation but no channels, and areas in which no erosion or inundation is acceptable. Hydraulic modeling and scour analysis are an iterative process that allow for changes in the number and location of proposed structures. Hydraulic modeling should include a one-dimensional (1-D) model of the project reach to determine potential backwater effects of the project [e.g., *Brummer et al.*, 2006] and 2-D modeling as needed to evaluate the effects of structures on flow deflection and localized water elevations. Scour estimates should include all aspects of relative scour, including general, contraction, pier, and abutment scour [e.g., *Liu et al.*, 1961; *Johnson and Torrico*, 1994; *Hoffmans and Verheij*, 1997; *Fischenich and Landers*, 2000; *Melville and Coleman*, 2000; *Federal Highway Administration (FHWA)*, 2001; *Chase and Holnbeck*, 2004; *Fael et al.*, 2006]. The designs are modified to achieve the goals of the project and to minimize the risk associated with the designs. With this understanding, ELJs can be designed and placed in such a way that they achieve the desired goals, accommodate natural processes, and even diminish risks to infrastructure and property.

After a thorough understanding of the project reach and watershed, a clear definition of project opportunities and constraints, and the selection of appropriate natural analogs, the engineering design can proceed. Design development begins by refining the conceptual plan on the basis of the performance goals of the project. The results of the initial geomorphic analysis, risk assessment, hydraulic modeling, and scour analysis are incorporated into the preliminary design plans to refine the number of structures, structure archetypes, orientation, and predicted channel response.

5.3. Structure Stability Assessment

For wood debris that is held in place by burial or ballasting, it is critical to estimate the buoyant force acting on individual logs and the total structure. Stability is commonly quantified using a factor of safety (FS) estimate taking the ratio of resisting forces to driving forces. So for hydrostatic conditions, the ratio will be the gravitational force acting downward over the buoyant force acting upward. If FS is greater than 1, the wood should be stable under the set of assumptions built into the calculation. For engineered structures, a minimum FS of 1.5 or greater is used. One of the key assumptions in estimating a FS for embedded wood is that the surcharge material, typically native alluvium or imported rock, remains in place. Thus, if bank erosion or scour removes the surcharge, it could impact the long-term stability of the wood structure. So it is important to determine whether or not the surcharge material will be a risk in the future. For example, burying a log into the bank and then placing boulders on top of the log assumes that the boulders will not roll off the log, which may not be a safe assumption if the log is otherwise set within native alluvium that can be eroded by the stream. Placement of ballast should be designed to ensure it functions as desired, which will require an understanding of channel dynamics and structure performance. Structures such as embedded bendway weir logs could be put at risk if localized bank erosion exposes the buried portion of the logs. More complex structures, such as timber cribs, can be designed to retain their ballast even when completely exposed to the stream since the material is situated within the interior of the crib. Here again it is important to understand the architecture of the structure with regard to ballast retention. If the crib has an open bottom and scour gets beneath the structure, ballast can "bleed" out and compromise stability. Bleeding can also occur along the flanks of the structure if gaps between log layers of the crib are larger than the surcharge material, which is commonly the case when native alluvium is used. Both of these conditions (bleeding through base or sides of a wood structure) can be solved in multiple ways, which will be discussed further under structure design.

The final design plans should include plans for temporary erosion and sedimentation control, construction sequencing, surveyor control, traffic access, ELJ locations, grading for the ELJ structures, and planting, as well as detailed cross sections of the ELJ structures.

As outlined above, many types of ELJ structures exist, and the selection of a specific set of materials and architecture depends on the particular site, project goals, acceptable levels of risk, costs, and constraints. Experiences with ELJs to date suggest that, in certain circumstances, they can provide an economical method of bank protection and help in managing debris (especially mobile wood) that may be hazardous to bridges and culverts. At the same time, installation of an ELJ can reestablish important habitat elements of forest streams that have been degraded by conventional river engineering and management. While the situations in which ELJ technology can provide a sound engineering solution that delivers measurable environmental and esthetic benefits are numerous, in some situations, an ELJ structure would be inappropriate.

Natural accumulations of wood debris exhibit distinct size, shape, and orientation, which combine to create various hydraulic and geomorphic effects in different portions of mountain channel networks. Therefore, the design of an ELJ project should include careful scoping of the types of logjams that are likely to prove stable and meet the design objectives in the local geomorphic context.

Assessment of whether ELJs represent an appropriate approach and, ultimately, the final design specifications for a site, depend on both the geomorphic and hydraulic characteristics of the stream reach and floodplain, as well as human objectives and constraints. Consequently, investigations and analyses associated with ELJ design need to address (1) potential local and watershed disturbances that might influence the project, (2) historical planform characteristics and changes in channel, floodplain, and forest patterns in the valley bottom, both upstream and downstream of the project site, (3) results of topographic, geomorphic, geotechnical, and hydraulic analyses of the project reach and the sub-reach-scale area where the ELJ structures will be built, and (4) size, position, spacing, and architecture of the ELJs and constituent logs. ELJ stability is based upon the composite framework of large key members and stacked logs that provide a foundation for smaller stacked and racked pieces.

Consistent with the objective of imitating natural processes, ELJs are typically built of native wood debris and alluvial soils. However, imported and engineered materials such as piles, rock, or concrete logs have been used in the core structure, so long as the complete structure still looks and acts like a natural structure, thus understanding the range of natural structures is directly applicable to restoration design. All ELJ designs should be based on local conditions.

5.4. Structural Design

ELJ structures are designed to be stable against lateral velocity (drag) and vertical lift and buoyancy forces. The parameters used as input to the calculations of these forces include coefficients of drag and lift, cross-sectional area of the part of the structure projection that is perpendicular to flow, volume of wood material in the structures, density of water, specific weight of alluvium and wood, active and passive earth pressures, flow velocities as noted above, and water surface elevations. ELJ structures are used in a variety of situations and can be subjected to a wide range of loading. The structures are engineered to allow changing load paths by the strategic orientation and interlacing of individual structure members. One way to increase the structural stability and the FS is to incorporate inclined or vertical timber or steel piles.

For example, piling is designed for bending loads rather than axial loading. Drag loads are treated as a point load acting at the midpoint of the pile, and the piles (or column) are treated as cantilevered beams fixed at selected scoured bed elevation. The pile loading consists of static head, velocity head, and drag load. The angle at which forces from the river would act on the logjam is based on historical channel planforms and the channel migration zone. The worst-case flow, perpendicular flow, is used in the load calculations. The calculations are based on two separate conditions: (1) maximum probable scour with the pile exposed and (2) predicted scour with one third of the pile exposed.

5.4.1. Static head. Static head is used in the calculations, assuming water is backed up behind the entire height of the structure, which would cause the largest load (height of the ELJ compared to water elevations during the design flood event, e.g., 100 year flood).

5.4.2. Velocity head. Velocity head is based on hydraulic modeling, typically using flows from the 25 and 100 year flood events.

5.4.3. Drag load. Drag load is induced by flowing water that impinges upon the upstream face of the ELJ. The ELJ must resist overturning and sliding.

5.4.4. Lift. Lift consists of upward forces to consider for individual logs that will experience overtopping flow, particularly relevant in grade control structures using log weirs.

ELJ design can include quantitative assessment of failure modes for each structural element (log) and the entire structure. Looking at how each element contributes to the stability of the completed structure and what kind of forces or changes it may be subjected to is critical. For example, one of most common failure mechanisms for wood structures is scour. Important questions include, but are not limited to, the following: (1) How will the structure fare if a deep scour pool forms? (2) What type of scour will the structure experience? (3) What is the incident flow direction, and if that changes, how will the stability and performance of the structure be impacted? (4) Is structure stability based on pilings or ballast? (5) If the structure is dependent on ballast, will the ballast stay intact if the structure is undercut by scour?

Stable ELJs can be built without the use of cable, earth anchors, chain, imported rock, or steel piling, but all these structural elements have been used in construction of some ELJs. These types of anchoring should not be depended upon without thorough consideration of their purpose, the forces to which they will be subjected, and how they will perform as the channel deforms. For example, one of the most misapplied anchoring techniques is cable earth anchors. If a log starts to move, so will the cable. Once a cable starts oscillating, it is similar to a cable saw and is prone to do more damage than good. Therefore, if cable anchors are used, they should be arranged to prevent any log displacement, which typically means at least three points of attachment at each end of a log. If scour undercuts the log and it settles, the cables will slack and be subject to motion. Given this tendency, it is always best to embed wood into the channel bed as much as possible, which can preclude any need for cable anchors. Cable or chain can still be used in attaching logs to one another, but slack or loose cable should always be avoided. The following section discusses the structural advantages of embedded wood.

5.5. Piles, Posts, and Embedment

Embedment, or burial, is the single most important factor for wood stability in an alluvial channel. A pile can remain stable even when totally submerged with only minimal burial depth and despite having no surcharge. Under hydrostatic conditions, a pile is held in place by its skin friction, which is a function of the earth materials, pile composition, and burial depth. Piles illustrate a means of introducing stable wood into rivers that have been around for thousands of years. Several ways exist to get embedment in a timber structure: (1) driven vertical or inclined ("batter") piles, (2) excavated posts with or without root wads, (3) excavating the entire structure to the maximum scour depth, (4) creating a self-settling gravity structure, or (5) a combinations of these. Piles and buried posts are a cost-effective element for stabilizing ELJ structures. To keep descriptions clear, we define piles as driven straight timbers and posts as excavated timbers with or without attached root wads (Plate 11). The mechanics of pile and post stability are linked to how they interact with the substrate, skin friction, geostatic loads, and passive earth pressures. Estimates of log buoyancy are critical in designing buried timber structures. Also important is estimation of the lift forces acting on a log weir. Logs protruding from the streambed will be subjected to significant drag forces and should also be assessed for likelihood of breakage.

5.6. Skin Friction

The unit skin friction or shaft resistance of a buried pile (q_s) is equal to the product of the angle of wall friction (δ), the earth pressure coefficient (K_s), and the average vertical effective stress (σ'_v) [*Broms and Hellman*, 1970]:

$$q_s = (K_s \sigma'_v \tan\delta). \tag{17}$$

The angle of wall friction is dependent on the pile material and the angle of internal friction of the substrate (ϕ'). The total skin friction resistance is given by the sum of layer resistances, with A_w is vertical area of embedded wood in each soil layer:

$$\Phi_s = \sum(K_s \sigma'_v \tan\delta A_w). \tag{18}$$

The average vertical effective stress acting on the pile is the difference of the normal stress of the soil, σ_s, and the pore pressure within the soil, u:

$$\sigma'_v = \sigma_S - u. \tag{19}$$

The average pore pressure is proportional to the depth of water, h, and water density, ρ_f:

$$u = 0.5h\rho_f. \tag{20}$$

Normal stress of soil is proportional to the depth of soil, d_s, and soil density, ρ_s.

$$\sigma_S = 0.5 d_S \rho_S. \tag{21}$$

Unit skin friction resistance is a function of the earth pressure coefficient, K_s, the vertical effective stress, σ_v', and the wall friction angle, δ. The wall friction angle is dependent on the pile material and the friction angle of the soil, ϕ', for timber $\delta = 2/3\ \phi'$ (for concrete it is $J\ \phi'$ and 20° for steel). The ultimate unit skin friction is expressed as

$$q_S = K_s \sigma'_v \tan(\delta). \tag{22}$$

The coefficient K_s depends on the pile material and soil density. K_s values for timber range from 1.5 to 4.0, for low- to high-density soils, respectively. From equation (22), we can simplify equation (18) to the sum of the product of q_s and A_w for each soil layer:

$$\Phi_s = \sum(q_s A_w). \tag{23}$$

The FS for a simple pile coming out of the riverbed under hydrostatic loading is

$$\mathrm{FS} = \frac{\Phi_s + W_W}{F_B}, \tag{24}$$

where W_w is dry weight of pile and F_B is buoyant force.

A pile 9.1 m in length, 0.3 m in diameter, and situated in 3 m of water would have to be buried at least 1.5 m to stay in place (Figure 5). If the burial depth is doubled to 3 m, the FS increases eightfold. Based on empirical studies, skin friction resistance can reach a maximum at depths of between 10 and 20 pile diameters [*Broms and Hellman*, 1970]. With regard to

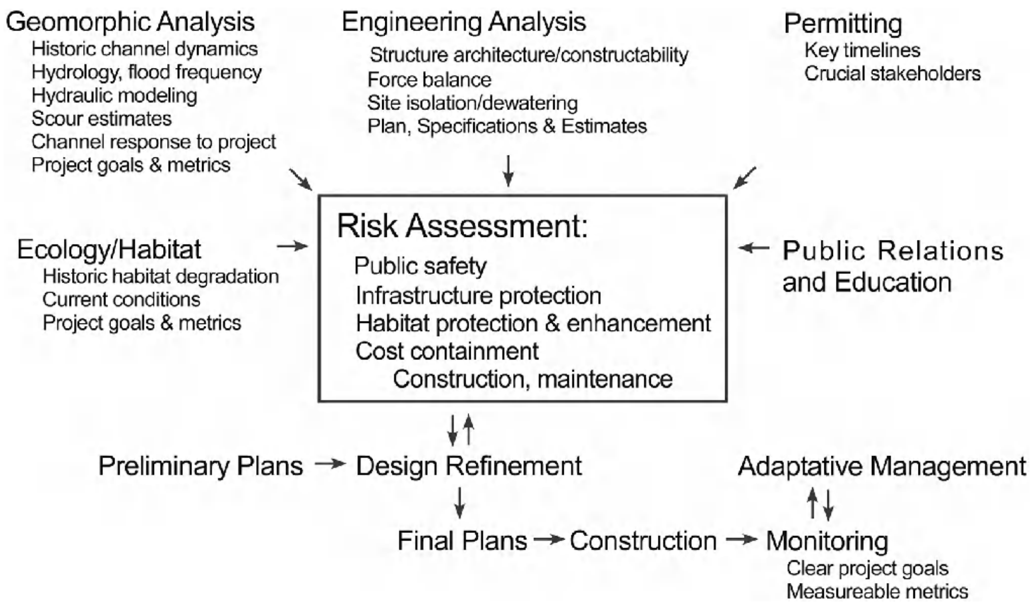

Figure 5. General risk assessment process for designing, constructing, and managing wood in rivers.

lateral loading on a pile, passive earth pressures continue to increase proportional to depth [*Abbe et al.*, 2003b].

For a vertical timber post with its root wad buried, the pull-out resistance will be proportional to the volume of overlying soil (as defined by internal friction angle), which can be expressed as the difference between the frustum defining the soil volume from the root wad to the ground surface and the buried volume of the timber (Figure 6):

$$V = \left[\frac{\pi D_b(R^2 + Rr + r^2)}{3} + \pi R_{\mathrm{rw}}^2 D_h\right] - \left[\frac{\pi R_{\mathrm{rw}}^2}{2t+1}(X_i^{2t+1} - 1)\right]. \tag{25}$$

The effectiveness of burying a post with attached root wad as compared to burial of simple cylindrical post can be illustrated by burying scale models similar to those depicted on Figure 6. The results for dry soil show the significant increase

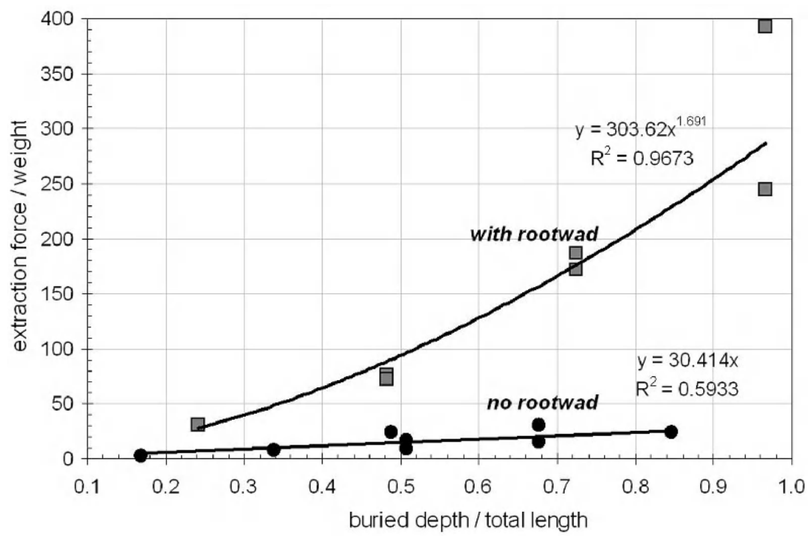

Figure 6. Experimental results of buried piles with and without root wads relative to snag dry weight and length. Simple piles without root wads increase resistance with burial length by a factor of 30 of the log weight. Root wad piles exhibit a nonlinear increase in the extraction force relative to log weight as burial depth is increased.

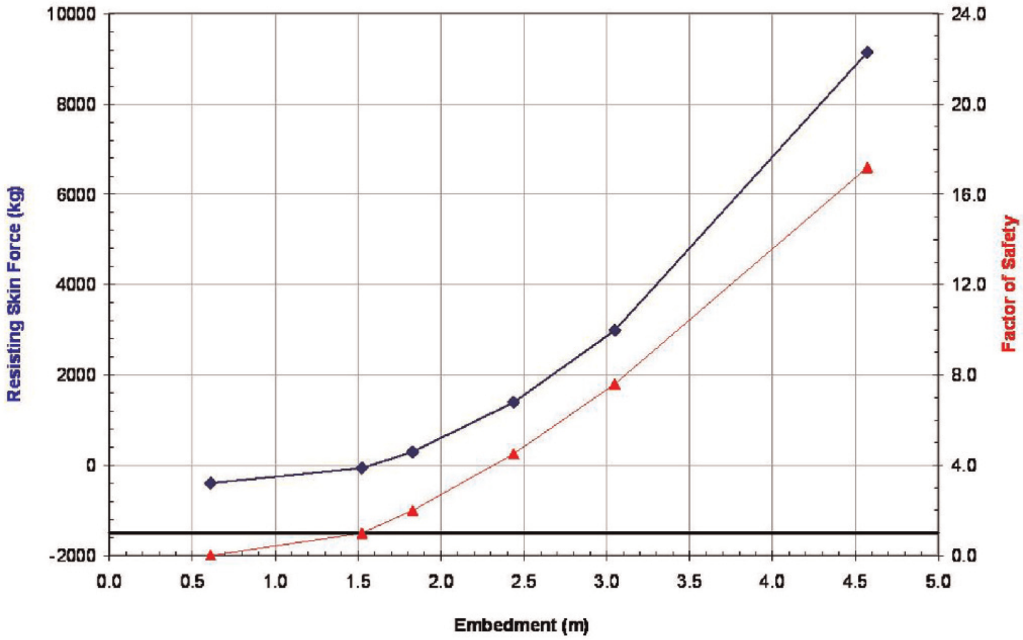

Plate 11. Resistance and factor of safety attributed to skin friction under hydrostatic conditions as a function of embedment depth (9.1 m pile, 0.3048 m in diameter, submerged in 3 m of water).

in vertical force necessary to pull out a buried root wad as opposed to a simple pile (Figure 6). At burial depths of 20%, the post length addition of a root wad doubles the resistance, and at depths of half the post length, resistance increases sixfold. Basic analysis elements for evaluating loads on piles and posts are presented below in summary of force balance calculations.

5.7. Example of an Engineered Flow Deflection Logjam

Engineered wood placements designed to redirect flow such as bar apex and meander jams [*Abbe and Montgomery*, 2003] include a core structure with a facing of racked logs. The number of architectures for the structure core is infinite, but the purpose is always to ensure the structure's stability

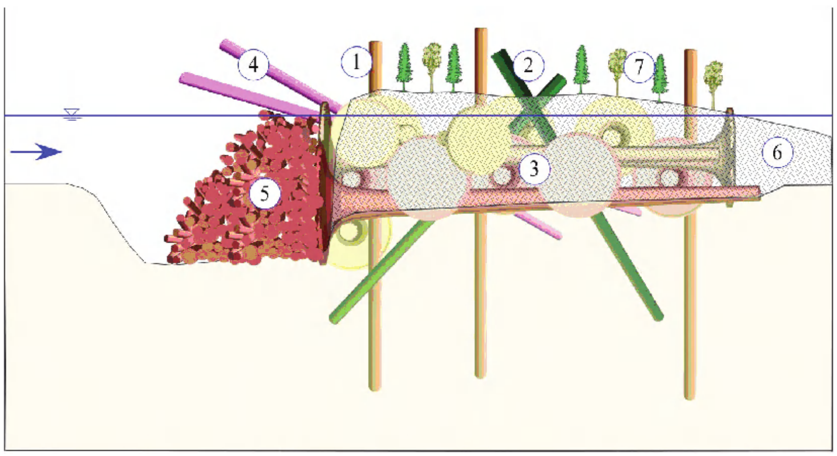

Plate 12. Some of the basic components that can be used in a flow deflection engineered wood placement: (1) driven vertical piles, (2) inclined or "batter" piles, (3) core structure of ELJ, (4) racking retention logs embedded into core, (5) racking debris, (6) compacted backfill in core, and (7) reforestation on top of core. The architecture and types of materials used to create the core can vary substantially.

Plate 13. Project example of introducing ELJs to increase channel complexity by creating channel anabranching in the Upper Mashel River near Eatonville, Washington. Air photos of preproject (2005) and postproject (2009) conditions at restoration project constructed in 2006 and 2007 (consisting of six ELJs delineated on 2009 photo). With addition of side channels activated by ELJS, total bankfull channel length increased approximately 180% from about 890 to 1610 m. Floodplain connectivity almost doubled from about 3.6 to 6.9 ha (yellow dashed lines). The project successfully experienced a peak flow greater than 50 year recurrence flood in January 2009 with estimated velocities of over 4 m s^{-1} and maximum shear stress of 166 Pa. Bottom photo is ELJ 2 in 2008 with side channel to left.

over its design life. The most effective means of creating a stable core that can resist lateral and vertical loads is to use piles or buried posts. While timber piles are typically used, other materials could be used if warranted. Piles can be driven or excavated. When pile depth is limited, burying root wads can significantly add to the structure's integrity as explained earlier. Designing the size and spacing of ELJ structures placed along riverbanks can draw from the literature on spur dikes [e.g., *Copeland*, 1983].

Typical elements of an engineered flow deflection logjam (Plate 12) include the following: (1) vertical piles or posts, either with root wads (excavated) or without (driven), (2) batter or inclined piles/posts, (3) key and stacked logs comprising crib box at core of structure, (4) inclined "retention" logs protruding from internal crib to hold racking logs in place, (5) racked logs, (6) internal lining of small wood debris (small logs and slash applied to plug gaps in crib and prevent "bleeding" of backfill, live stake bundles can be used higher in the structure, as long as the stake tips reach the water table), (7) backfill surcharge filling crib box, and (8) revegetation on surface of structure.

5.8. *Basic ELJ Force Balance Analysis*

A force balance analysis is an important part of design that should be done for any engineered wood placement in streams and rivers. Owing to the many types of engineered wood structures (including the wide variety of ELJ types), the force balance should be structure specific. In the force balance analysis, it is important to clearly describe assumptions and data sources regarding water depths, flow velocities, substrate material, incident angle of flow, and scour depths. Free-body diagrams help with understanding the forces acting on individual pieces and entire structures. A comprehensive treatment of force balance is beyond the scope of this chapter, but we provide basic outline of formulae to consider in evaluating buoyancy and horizontal forces acting on an ELJ structure.

5.8.1. Buoyancy analysis. The following steps should be undertaken for buoyancy analysis.

1. Calculate volumes of logs.

$$V_i = \pi\left(\frac{D_i}{2}\right)^2 L_i n_i, \tag{26}$$

where

V_i volume of log type i (m^3);
D_i diameter of log type i (m);
L_i length of log type i (m);
n_i number of logs of type i.

$$V_{\text{tot}} = \sum_{i=1}^{k} \pi\left(\frac{D_i}{2}\right)^2 L_i n_i, \tag{27}$$

where

V_{tot} total volume of logs (m^3);
i log type (identifier) (number);
k total number of log types;
n_i number of logs of type i.

With root wad,

$$V_i = \left(\pi\left(\frac{D_i}{2}\right)^2 L_i + \pi\left(\frac{D_{\text{rw},i}}{2}\right)^2 L_{\text{rw},i}(1-e_{\text{rw}})\right) \times n_i, \tag{28}$$

where

D_i diameter of log stem (m);
$D_{\mathrm{rw},i}$ diameter of root wad (m);
L_i length of log stem (m);
$L_{\mathrm{rw},i}$ length of root wad (m);
e_{rw} ratio of voids in root wad.

2. Calculate buoyant forces on submerged logs. If all logs are of uniform density and submerged, then

$$F_B = (\gamma_{\mathrm{lwd}} - \gamma)V_{\mathrm{lwd}}, \tag{29}$$

where

F_B buoyant force (N);
γ_{lwd} unit weight of wood piece (N m^{-3});
γ unit weight of water (9810 N m^{-3});
V_{lwd} submerged wood volume.

For different densities per log type,

$$\sum F_B = \sum_{i=1}^{n} (\gamma_{\mathrm{lwd}(i)} - \gamma)V_i, \tag{30}$$

where $\sum F_B$ is total buoyant force of all LWD (N), $\gamma_{\mathrm{lwd}(i)}$ is unit weight of log type i (N m^{-3}), and n is total number of submerged pieces of LWD.

3. Calculate downward forces of submerged fill (sediment).

Determination of volume of interior to be filled by soil

$$V_J = l_J w_J h_J, \tag{31}$$

where

V_J interior volume of ELJ (m^3);
l_J interior length of ELJ (m);
w_J interior width of ELJ (m);
h_J interior height of ELJ filled with alluvium (m).

Determination of volume of submerged soil inside the interior

$$V_{\mathrm{ss}} = \{(1-k)V_J\} - \sum V_{\mathrm{LWD}}, \tag{32}$$

where V_{ss} is volume of submerged soil in the ELJ (m^3), k is void ratio of the soil, and $\sum V_{\mathrm{LWD}}$ is volume of LWD in the interior (m^3)

Determination of weight of submerged soil

$$W_{\mathrm{ss}} = V_{\mathrm{ss}}(\gamma_{\mathrm{ss}} - \gamma), \tag{33}$$

where W_{ss} is weight of submerged soil (N) and γ_{ss} is saturated unit weight of soil (N m^{-3}).

Determination of weight of submerged boulder ballast (if used)

$$W_{\mathrm{sb}} = \pi \frac{D_b^3}{6}(\gamma_b - \gamma)n, \tag{34}$$

where

W_{sb} submerged weight of boulder (N);
D_b diameter of boulder (m);
γ_b unit weight of boulder (N m^{-3});
n number of boulders submerged.

4. Calculate downward forces of unsubmerged ELJ components (cover sediment, boulders, and logs).

Determination of volume of alluvium/soil above waterline

$$V_{\mathrm{soil}} = A_{\mathrm{soil}} h_{\mathrm{soil}}, \tag{35}$$

where V_{soil} is volume of soil above waterline (stage of design flow, m^3), h_{soil} is depth of soil above water (m), and A_{soil} is area of soil cover (m^2).

Determination of weight of cover soil

$$W_{\mathrm{soil}} = V_{\mathrm{soil}} \gamma_{\mathrm{soil}}, \tag{36}$$

where W_{soil} is weight of dry alluvium/soil and γ_{soil} = bulk weight of dry soil.

Determination of weight of boulder ballast (if relevant)

$$W_b = \pi \frac{D_b^2}{6} \gamma_b n, \tag{37}$$

where

W_b dry weight of boulder (kg);
D_b diameter of boulder (m);
γ_b unit weight of boulder (kg m^{-3});
n number of boulders above waterline.

Determination of weight of logs above water

$$W_{\mathrm{dry\ lwd}} = \sum_{i=1}^{n} V_i \gamma_{\mathrm{lwd}(i)}, \tag{38}$$

where

$W_{\mathrm{dry\ lwd}}$ dry weight of boulder (kg);
V_i volume of LWD piece i (m^3);
$\gamma_{\mathrm{lwd}(i)}$ unit dry weight of LWD piece i (kg m^{-3});
n number of LWD pieces above waterline.

5. Find net force.

$$F_n = \sum(F_B) + W_{ss} + W_{soil} + W_{sb} + W_b + W_{dry\ lwd}. \quad (39)$$

Determination of FS (do layer by layer)

$$FS = \frac{F_n}{\sum F_B}. \quad (40)$$

5.8.2. Horizontal forces calculations. The following steps should be undertaken for horizontal forces calculation.

1. Calculate the force due to water velocity.

$$F_D = C_D A \rho \left(\frac{U^2}{2}\right), \quad (41)$$

where

F_D force due to the velocity of the water (N);
C_D drag coefficient;
A area projection equals ELJ width times channel depth (m^3);
ρ density of water (1000 kg m^{-3});
U design flow in channel (typically associated with 100 year flood event) (m s^{-1}).

2. Calculate difference in hydrostatic force upstream and downstream of ELJ.

Calculation of hydrostatic force upstream of the ELJ

$$F_{H0} = A_0 P_0, \quad (42)$$

where F_{H0} is hydrostatic force upstream of the ELJ face, A_0 is cross-sectional area of upstream face of ELJ, and P_0 is pressure of water on the upstream face of the ELJ (see below).

$$A_0 = (h_0 + d_{S0}) w_{J0}, \quad (43)$$

where h_0 is depth of channel upstream of logjam, d_{S0} is depth of scour on upstream side of logjam, and w_{J0} is upstream width of ELJ.

$$P_0 = \left(\frac{h_0 + d_{S0}}{2}\right)\gamma, \quad (44)$$

where γ is unit weight of water (N m^{-3}).

Calculation of hydrostatic force downstream of the ELJ

$$F_{H1} = A_1 P_1, \quad (45)$$

where F_{H1} is hydrostatic force downstream of the ELJ, A_1 is ELJ area on the downstream side, and P_1 is pressure of water on the downstream side of the ELJ.

$$A_1 = (h_1 + d_{S1}) w_{J1}, \quad (46)$$

where h_1 is depth of channel downstream of logjam, d_{S1} is depth of scour on downstream side of logjam, and w_{J1} is downstream width of ELJ.

$$P_1 = \left(\frac{h_1 + d_{S1}}{2}\right)\gamma. \quad (47)$$

Calculation of difference in hydrostatic force

$$\Delta F_H = F_{H0} - F_{H1}. \quad (48)$$

3. Calculate net horizontal force.

$$F_x = F_V - \Delta F_H + F_D. \quad (49)$$

4. Determine force per pile (assuming equal distribution)

$$F_{x(\text{pile})} = \frac{F_x}{n}, \quad (50)$$

where $F_{x(\text{pile})}$ is net horizontal force per pile (N), and N is number of piles.

5.9. Scour Analysis

Bed deformation around wood accumulations is an essential means by which the structures create important habitat, whether deep pools with adjacent cover or shallow riffles and bars in depositional areas. Scour is the primary failure mechanism for in-stream structures such as bridge piers, abutments, or bank protection. Predicting the depth and dimensions of scour is critical to designing wood structures. Different types of scour are linked to the hydraulic conditions induced by the structure, including plunging scour (such as flow over a weir), contraction scour (concentrated flow channel constriction), pier scour (flow around either side of an obstruction), and abutment scour (flow around one side of an obstruction). Scour is cumulative, so if two ELJs are placed opposite one another, they can induce both pier and constriction scour. Scour equations are largely dependent on laboratory experimentation and empirical coefficients, so the results of various equations can vary considerably, which requires a great deal of professional judgment and clear assumptions when applying results to a particular situation and design. Included here are some examples of different equations to estimate maximum scour depths for designing in-stream structures. Because these equations are primarily based on empirical data from laboratory flume experiments, they should be used in the context of professional judgment and actual on-site evidence of scour depths. Existing residual pool depths within a project reach provide a minimum estimate of potential scour, so close attention should be given to the maximum pool depths and their causal mechanisms within the project area. Recent advances in scour predictions around wide piers [*Sheppard et al.*, 2011] and around piers with wood accumulations [*Lagasse et al.*, 2010] offer refinements and new insights into predicting scour around structures similar to a large ELJ.

5.9.1. Local Pier Scour. The Colorado State University (CSU) equation was developed for the U.S. Federal Highway Administration for local pier scour under either clear water (no bed load input) or live-bed (active bed load) conditions [*Hoffmans and Verheij*, 1997; *FHWA*, 2001; *Melville and Coleman*, 2000]. The CSU equation includes a correction factor to adjust for bed material for cases where the $D_{50} \geq$ 2 mm and the $D_{90} \geq 20$ mm.

$$\frac{d_{\mathrm{ps}}}{y_1} = 2.0K_1K_2K_3K_4\left(\frac{a}{y_1}\right)^{0.65} Fr^{0.43}, \tag{51}$$

where

d_{ps} maximum scour depth (m);
y_1 flow depth immediately upstream of pier (m);
a pier width (m);
K_1 correction factor for pier nose shape (for square $K_1 = 1.1$);
K_2 correction factor for flow angle of attack $[\cos\theta + (L/a)\sin\theta]^{0.65}$
L pier length;
θ incident angle of flow on pier (0 = hitting straight on, parallel to channel);
K_3 correction factor for bed condition for clear water/plane bed conditions $K_3 = 1.1$, for dunes $K_3 = 1.1$ to 1.3);
K_4 correction factor for bed armoring (minimum value is 0.4) $0.4\ V_r^{0.15}$ (Meuller K-4 correction [*FHWA*, 2001]);

where

$$V_r = \frac{V - V_{icD_{50}}}{V_{cD_{50}} - V_{icD_{95}}} > 0, \tag{52}$$

$$V_{icD_X} = 0.645 V_{cD_X}\left(\frac{D_X}{a}\right)^{0.053}, \tag{53}$$

$$V_{cD_X} = 6.19 y_1^{1/6} D_x^{1/3} \tag{54}$$

where

V velocity of upstream approach flow (m s^{-1});
V_{icDx} approach velocity necessary to initiate scour of grain size D_x (m s^{-1});
V_{cDx} critical velocity for incipient motion of D_x (m s^{-1});
Fr Froude number = $V/(gy_1)^{0.5}$;
g gravitational acceleration, 9.81 (m s^{-1}).

5.9.2. The Johnson and Torrico Correction Factor for Wide Piers. FHWA [2001] recommends application of the Johnson and Torrico correction factor [*Johnson and Torrico*, 1994] in the CSU equation when the ratio of flow depth to pier width is less than 0.8, the ratio of pier width to D_{50} is greater than 50, and when $Fr < 1$ (subcritical flows). In many ELJ situations, these conditions would apply, but in cases where $Fr > 1$, predictions using the Johnson and Torrico correction factor will underpredict scour.

$$\frac{d_{\mathrm{ps}}}{y_1} = 2.0K_1K_2K_3K_4K_w\left(\frac{a}{y_1}\right)^{0.65} Fr^{0.43}. \tag{55}$$

For cases where $V/V_c < 1$,

$$K_w = 2.58\left(\frac{y}{a}\right)^{0.34} Fr^{0.65}. \tag{56}$$

For cases where $V/V_c \geq 1$,

$$K_w = 1.0\left(\frac{y}{a}\right)^{0.13} Fr^{0.25}. \tag{57}$$

5.10. The Modified Froehlich Equation for Abutments in Sand Bed Rivers

Contraction scour is not directly accounted for in the modified Froehlich equation, so a safety factor of +1 is added [*Fischenich and Landers*, 2000]. The equation was derived for scour at abutments in sand bed channels and has input parameters for abutment shape, incident flow angle, and abutment length perpendicular to flow:

$$y_s = y_1 2\left(\frac{\theta}{90}\right)^{0.13}\left(\frac{W_0}{y_a}\right)^{0.43} Fr^{0.61} + 1.0, \tag{58}$$

where

y_s scour depth below water surface (m);
y_1 depth of flow at structure (m);
W_0 length of structure projected perpendicular to flow (m);
θ angle of embankment to flow (degrees);
Fr Froude number of flow upstream of structure = $V/(gy_1)^{0.5}$.

5.11. Simplified Chinese Equation for Live-Bed Scour in Coarse-Bedded Channels

Chase and Holnbeck [2004] present the simplified Chinese equation for live-bed scour that is applicable to examining the effect of large ELJ flow deflection structures in coarse-bedded rivers. The Chinese equation was developed from laboratory and field data for both live-bed and clear water scour situations. Only the live-bed formulae for situations in which the critical velocity exceeds the approach velocity are presented here.

For live-bed scour (when $V_o > V_c$),

$$y_s = 0.6495 K_s b^{0.6} y_0^{0.15} D_m^{-0.07} \left(\frac{V_0 - V_{ic}}{V_c - V_{ic}}\right)^c. \tag{59}$$

For clear-water scour,

$$y_s = 0.834 K_s b^{0.6} y_0^{0.15} D_m^{-0.07} \left(\frac{V_0 - V_{ic}}{V_c - V_{ic}}\right)^c, \tag{60}$$

where

- y_s depth of pier scour (m);
- K_s pier shape coefficient (dimensionless), equal to 1.0 for cylinders, 0.8 for round-nosed piers, and 0.66 for sharp-nosed piers;
- b width of pier normal to flow (m);
- y_0 depth of incident flow upstream of pier (m);
- D_m mean particle diameter of substrate (m);
- V_0 approach velocity upstream of pier (m s^{-1});
- V_c critical velocity for incipient motion of bed material (m s^{-1}), assuming density of water is 1000 kg m^{-3} and gravity of bed material is 2.65.

$$V_c = \left(\frac{y_0}{D_m}\right)^{0.014} \left(29.035 D_m + 6.05E^{-7}\left[\frac{10 + y_0}{(D_m)^{0.72}}\right]\right)^{0.5}. \tag{61}$$

The approach velocity corresponding to the critical velocity at the pier is calculated as

$$V_{ic} = 0.645 \left(\frac{D_m}{b}\right)^{0.053} V_c, \tag{62}$$

where

$$c = \left(\frac{V_c}{V_0}\right)^{8.20 + 2.23 \log D_m}. \tag{63}$$

5.12. Contraction Scour

FHWA [2001] recommends the modified Laursen equation for estimating contraction scour under live-bed conditions. The equation was developed for sand-bedded channels and is, thus, likely to overpredict scour in gravel-bedded channels.

$$d_{cs} = y2 - y0, \tag{64}$$

where

$$\frac{y_2}{y_1} = \left(\frac{Q_2}{Q_1}\right)^{6/7} \left(\frac{W_1}{W_2}\right)^{k_1}, \tag{65}$$

where

- d_{cs} average depth of contraction scour (m);
- y_0 existing depth in contracted channel segment prior to scour (m);
- y_1 average depth upstream of contracted channel segment (m);
- y_2 average depth in contracted channel segment after scour (m);
- Q_1 flow upstream of contracted channel segment (m^3 s^{-1});
- Q_2 flow in contracted channel segment (m^3 s^{-1});
- W_1 channel bottom width upstream of contracted channel segment (m);
- W_2 channel bottom width in contracted channel segment (m);
- k_1 average depth in contracted channel segment after scour (m), where $u^*/\omega < 0.5$, $k_1 = 0.59$ (most sediment moving as bed load) and $0.5 < u^*/w < 2.0$, $k_1 = 0.64$ (some suspended sediment transport), $u^*/\omega > 2.0$, $k_1 = 0.69$ (most sediment moving as suspended load);
- u^* shear velocity in upstream channel segment (m s^{-1}), equal to $(g y_1 S)^{0.5}$;
- ω fall velocity of D_{50} of bed material (m s^{-1}) equal to $[(G - 1) g D_{50}]^{0.5}$.
- G specific gravity = (sediment density/water density);
- S energy slope of flow in channel upstream of contracted segment (m m^{-1}).

5.13. Abutment Scour

ELJs placed along a bank and intended to act like flow deflection groins are similar to bridge abutments. *Melville and Coleman* [2000] and *FHWA* [2001] recommend the modified *Froehlich* [1989] equation for live-bed scour around a local abutment. The equation was based on regression results of laboratory flume experiments.

$$\frac{d_{as}}{y} = 2.27 K_1 K_2 \left(\frac{L'}{y}\right)^{0.43} Fr^{0.61} + 1.0, \tag{66}$$

where

- d_{as} depth of scour (m);
- K_1 coefficient for abutment shape;
- K_2 coefficient for angle of abutment relative to flow, equal to $(\theta/90)^{0.13}$, $\theta < 90°$ if abutment points downstream, $\theta > 90°$ if abutment points upstream;

L' length of abutment projected perpendicular to flow (m), $L \cos \theta'$ if $\theta > 90°$ then $\theta' = \theta - 90$, if $\theta < 90°$ then $\theta' = \theta$;
y flow depth (m).

For sand-bedded channels, *Hoffmans and Verheij* [1997] recommend the Liu equation for abutment scour [*Liu et al.*, 1961], which was developed based on dimensional analysis,

$$d_{\mathrm{as}} = K_L y \left(\frac{L}{y}\right)^{0.4} Fr^{0.33}, \qquad (67)$$

where

d_{as} depth of scour (m);
K_L coefficient for abutment shape, streamlined, $K_L = 1.1$, blunt, $K_L = 2.15$;y flow depth (m);
L abutment length perpendicular to flow (m).

6. PERFORMANCE OF ELJS AND LESSONS LEARNED

While ELJ projects have been constructed in western Washington state since 1995 and have performed well through many large floods, ELJs remain an experimental technology. The projects constructed to date confirm that postconstruction inspections and maintenance are needed as an essential component of ELJ projects that are designed to control bank erosion. Whereas these ELJ demonstration projects show the technology to be an environmentally and economically viable alternative to traditional river engineering in certain applications, inappropriate design and application of ELJs can result in locally accelerated bank erosion, unstable debris, or channel avulsion. Care should be taken to understand local hydraulic, geologic and geomorphic, and sociopolitical conditions for every site, particularly the effect of spatial and temporal variability. Continued research and experimental applications of ELJs in a variety of topographic and climatic settings are needed to help refine the design guidelines for their use in rehabilitating and managing river systems. Integrating the elements discussed above and drawing from various guidelines and publications [e.g., *Abbe et al.*, 1997, 2003b, 2008; *Brooks*, 2006; T. B. Abbe et al., Bank protection and habitat enhancement using engineered log jams: An experimental approach developed in the Pacific Northwest, unpublished report, Natural Resources Conservation Service, 2005, hereinafter referred to as Abbe et al. unpublished report, 2005], we have put together a general checklist for reintroducing wood to rivers and restoration projects (Figure 7).

Hundreds of engineered wood structures have been built for river restoration throughout North America and elsewhere. The beneficial effects of ELJ projects on channel morphology and habitat has been widely recognized, from creating pools and cover, to increasing floodplain connectivity and creating more complex channel planform [e.g., *Abbe and Montgomery*, 1996, 2003; *Brooks and Brierly*, 2004; *Brooks*, 2006; Abbe et al., unpublished report, 2005]. The Upper Mashel River restoration project offers an example of how ELJs were used to increase natural wood debris retention, channel length, pool frequency, cover, and floodplain connectivity (Plate 13). The Mashel project transformed an incised single thread plane bed channel reach into a multichannel pool-riffle complex.

Thus far, ELJs have performed remarkably well in a variety of streams and rivers, though no impartial scientific investigation that takes into account the many different site locations, flow conditions, or distinct design conditions has been performed to date. Table 1 is a compilation of a small sample of ELJ projects that illustrate how different types of ELJs have fared through a range of project sites and flow events. Structural complexity of these projects varied from minimal engineering to high levels of engineering (e.g., steel H piles, scour aprons, and rock ballast), which has definitely influenced structure performance. Damages to ELJ structures appear primarily to be associated with scour and turbulence along the flanks of the structures, though overtopping flows have also resulted in loss of some backfill and revegetation of one structure. Failures have resulted from structures being "plucked" apart piece by piece, as opposed to the downstream transport of an intact ELJ, which has not been observed.

7. SUMMARY OF RECOMMENDED ELJ DESIGN PROTOCOL

7.1. Reach Analysis

Research analysis attempts to answer questions such as (1) Why is the road or infrastructure at risk? (2) What are the processes causing the damage? (3) Are things getting worse or better?

The analysis should document historical channel changes, sediment transport and deposition, bank materials and stability, hydrology and hydraulics, ecologic and biological conditions and opportunities, riparian conditions, and infrastructure constraints. The reach analysis should provide sufficient information to make predictions about the river's future under various scenarios so that sustainable logjam designs can be developed that emulate natural conditions and processes.

7.2. Feasibility Study

A feasibility study evaluates actions that should be considered and assesses solutions that are realistic from a cost

Guideline Checklist for Re-introduction of Wood in Rivers, version 2.0

Project Information

Project:	
Owner:	
Location:	
River System:	
Date:	

Project Team:	**Initials**
Reviewed:	
Approved:	

Project Definition

Topic	**Considered Yes/No/NA**	**Date**	**Initials**	**Notes**	**Review Date**	**Initials**
1. Identification of Project Goals						
a. Clear description of goal(s), e.g.:						
i. Increase in pool frequency						
ii. increase/sustain side channels						
iii. control incision of bank erosion						
iv. Promote specific bed substrate						
v. increase productivity						
introduce and trap organic matter						
fish cover, invertebrate substrate						
vi. attenuate downstream flooding						
b. Quantification of project goals						
i. Identify success factors						
ii. Identify project requirements and constraints						
funding						
landowner						
technical						
regulatory (solicit agency input)						
c. Assess project sustainability (concept screening)						
i. Environmental						
(geomorphic/aquatic/riparian habitat, natural processes)						
ii. Social (human community)						
iii. Financial						
(cost/benefit relative to environmental and social factors)						
2. Existing/Historical site conditions						
a. Channel gradient						
b. Channel type(s)						
c. Hydrology, flow regime						
d. Is wood part of system? Was it once? What changed?						
e. Historical channel mapping (HCMZ)						
f. Geologic controls						
g. Floodplain &riparian conditions						
h. Hydrology and flow regime (base flow, water table, peak flows)						
i. Hazard identification						
3. Watershed Disturbance						
a. Natural (e.g., fire, dam breaks, debris flows)						
b. Development (e.g., timber harvest, urbanization, incision, trends)						
c. Climate Change						
d. Historical channel response (aggradation, incision, brands)						
4. Identification of Opportunities and Constraints (O&C)						
a. Consistent with existing and future system changes						
b. Hazard identification						
i. Flood inundation, real and jurisdictional						
ii. Channel Migration Zone / erosion hazard areas						
iii. Wood debris loading (include future riparian projections)						
iv. Projected channel response to watershed disturbances						
c. Upstream or downstream impacts						
i. flooding						
ii. erosion						
d. Critical infrastructure						
e. Public Safety						
f. Environmental						
g. Social (human community)						
h. Financial						
i. Technical						
j. Construction (feasibility, O&C)						

Figure 7. General guidelines for reintroduction of wood to rivers.

	k. Political (public and stakeholder)						
	l. Regulatory						
	m. Legal						
	5. Concept Development						
	a. Hazard Identification						
	b. Opportunity and constrain review						
	c. Develop concept alternatives						
	6. Stakeholders and the community						
	a. Identify (landowners, interest groups, regulatory)						
	b. Identify interest groups						
	c. Outreach and coordination						
	Reach Analysis Report						
Design Development	**7. Design flows**						
	a. Hydrology and flow regime (base flow, water table, peak flows)						
	b. Hydraulics (roughness, depths and velocities)						
	i. Measured						
	ii. Modeled						
	c. Flow scenarios (anticipated response)						
	d. Fish passage considerations						
	8. Substrate						
	a. Substrate composition (clay/silt/sand/gravel/cobble/bedrock)						
	b. Critical shear stress to initiate motion & associated discharge						
	c. Geotechnical properities (internal angle of friction)						
	d. Depth and stratigraphy of alluvium and subsurface geology						
	e. Scour depths (general and local)						
	f. Sediment transport and budget						
	9. Wood debris transport and budget						
	a. Wood inventory (length, DBH, rootwad diameter, species)						
	b. Piece mobility (force-balance)						
	c. Wood recruitment potential (riparian conditions)						
	d. Future conditions associated with riparian management						
	10. Hazard delineations						
	a. Flood inundation, real and jurisdictional						
	b. Channel Migration Zone / erosion hazard areas						
	c. Wood debris loading (include future riparian projections)						
	d. Projected channel response to watershed disturbances (5)						
	11. Alternatives Assessment						
	a. Establish comparison criteria consistent with project goals						
	b. Identify concept alternatives						
	c. Identify risks						
	d. Identify costs						
	e. Identify benefits						
	f. Compare concepts and select preferred alternative						
	12. Hazard and Risk Assessment						
	a. Define hazards and probabilities						
	e.g., floods, debris flows, dam breaks, channel migration						
	b. Define risks (consequences)						
	e.g., impacts to habitat, public safety, property, infrastructure						
	c. Estimate risk, evaluate with owner						
	Preliminary Design Report (Preferred alternative)						
	13. Structure design						
	a. Location						
	b. Architecture						
	c. Scour (general, local – e.g., pier, abutment)						
	d. Force Balance						
	i. Buoyancy						
	ii. Drag/lift						
	iii. Ice loading (impact, lift, shear, jacking)						
	iV. Impact						
	v. Embeddment / skin friction						
	VI. Passive Earth Pressure						
	VII. Surcharge						
	v. Member strength and size						
	e. Material alternatives (timber, steel, rock, etc.)						
	f. Structure performance to:						
	ii. channel change						
	ii. predicted wood loading scenarios						
	ii. hydraulic conditions						
	g. Factors of Safety (consistent with Hazard and Risk Assessment)						

Figure 7. (continued)

	ii. sensitivity analysis						
	ii. redundancy						
	h. Structure design life						
	i. Required design life						
	ii. Decay analysis						
	i. Revegetation (root cohesion, surcharge, erosion protection)						
	j. Prediction of future channel conditions (5, 10, 50 yrs)						
	k. Concept 30% Plans						
	l. Stakeholder presentation of plan						
	14. Construction Planning						
	a. Contracting (format, specifications, contract type)						
	b. Sequencing						
	c. Timing: staging/construction/planting						
	d. Site access, traffic control						
	e. Fish protection/exclusion						
	f. Construction period peak flow analysis and contingency plan						
	g. Material specifications						
	h. Excavation and shoring						
	i. Piling (excavated, impact, vibratory)						
	j. Diversions, dewatering, crossings						
	k. Turbidity and erosion control						
	l. Construction risk assessment and plan for flood response						
	m. Revegetation and treatment of disturbed areas						
	n. Permitting, Permit Level Plans (60-70%)						
	15. Public Safety and Signage						
	a. Recreational Safety Checklist						
	b. Public education and notification						
	c. Signage						
	16. Basis of Design or Design Documentation						
	a. Design Criteria						
	b. Modeling						
	c. Scour						
	d. Design Calculations						
	e. Alternatives Assessment						
	f. Hazard and Risk Assessment						
	g. Signage, Public Education, and Recreational Safety Checklist						
	h. Risk Management measures and responsibilities						
	Basis of Design Report						
	Contract Documents: 100% Plans, Specifications and Estimates (PS&E)						
Implementation	**17. Construction**						
	a. Contract type						
	b. Final PS&E, Final Permits						
	c. Contract prep and bidding support						
	d. Pre-construction meeting						
	e. OVERSIGHT; daily reports w/ photo documentation,						
	f. Stakeholder meetings (weekly)						
	g. Pay estimation based on work complete (monthly)						
	h. Site winterization / project close-out						
	i. BMP & turbidity monitoring						
	j. As-built & construction report						
	k. Design team & stakeholder debrief						
	18. Project performance monitoring						
	a. Stability, scour, wood accumulation						
	b. Habitat diversity and area habitat utilization (fish and wildlife)						
	c. Fish and wildlife						
	d. Wood longevity						
	19. Adaptive management						
	a. Identify management criteria						
	b. Adaptive management plan						
	c. Maintenance						
	d. Documentation						

DISCLAIMER: This is a general checklist for stream and river wood projects. Specific sites may involve different elements not provided here, nor does this list provide the details of each element. Please send comments to Tim Abbe at tim.abbe@gmail.com.
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Figure 7. (continued)

Table 1. Performance of Selected Engineered Logjams as of 2009

Project (River)	Type[a]	Year Built	Number of ELJs	Intact in 2009	Stabilization[b]	Gradient (MM^{-1})	1 Year Q (cm)	100 Year Q (cm)	Maximum Q as of 2007 (cm)	Recurrence Interval of Maximum Peak Flow to Date (years)	Estimated Basal Shear Stress Experienced to Date (Pa)	Estimated Basal Shear Stress for 100 Year Q (Pa)
Upper Cowlitz	P[c]	1995	3	1	none	0.0044	127	1186	932	28	50	130
NF Stillaguamish RM 21	R/P/B	1998	5	4	E,[d] A	0.0028	108	643	679	>100	90	90
NF Stillaguamish RM 22	R	1999	3	3	E, A	0.0028	108	643	679	>100	90	90
Lower Elwha[e]	R	1999	21	21	E, A, Pt, C	0.0048	116	1176	920	27	70	110
Cispus RM 19	P/R	1999	7	7	E, A	0.007	37	528	419	25	120	180
Methow Tennis Family	P	2003	3	3	E, A	0.013	40	291	223	8	150	270
Hoh RM 14	P	2004	12	12	E, A, Ps, C, Sa	0.002	319	1902	1671	25	50	120
SF Nooksack Hutchinson	R	2006	6	6	E, A, Pt	0.0125	97	682	222	10	100	260
Mashel	R/P	2006	6	6	E, A, Pt	0.0057	34	159	~127	50	166	170
Williams (Australia)	R/G/P	2000	20	17	E, A., Pt	0.002	170	~800	270	8	100	120
Stockyard Ck (Australia)	R/P/G	2002	19	18	E, A, Pt, C	0.0025	7	190	116	50	65	76
Hunter River (Australia)	R/P/G	2004	33	33	E, A, Pt, C	0.001	170	3760	350	2	50	120

[a]Types are defined as follows: R restoration; P, bank protection; B, bridge protection; and G, grade control.

[b]Stabilization abbreviations are as follows: E, embedment; A, alluvial backfill; Pt, timber piles; Ps, steel H piles; C, chain used to attach logs to one another or piles (no earth anchors); Sa, scour apron; and R, rock ballast.

[c]Upper Cowlitz was emergency response constructed in December 1995.

[d]Excavation for NF Stillaguamish did not get to estimated scour depths.

[e]Lower Elwha structures constructed from 1999 to 2004.

and constructability perspective. The feasibility study should help answer important questions such as (1) Can the threatened infrastructure be relocated? (2) How much of the channel migration zone can be preserved or regained? (3) Can habitat be enhanced as part of solving traditional problems, such as bank protection and flood control? (4) Are local construction materials available? (5) Will partnerships with other stakeholders benefit the project?

7.3. Risk Assessment

Risk assessment evaluates and predicts how the project will perform under both normal and adverse conditions and evaluates the accuracy of the scientific data to be used in the project design. The risk assessment should also determine the potential effects on changes in the river channel (including flood levels, scour, sedimentation, and bank erosion) and evaluate potential short- and long-term impacts on humans, infrastructure, and natural habitat. The assessment should include appropriate public outreach and involvement, during which project stakeholders and affected groups and individuals are educated about the project and provide project managers and experts with feedback, insights, and ideas. Liability of building structures in rivers is becoming a major issue in some areas where the recreational community and flood protection districts have a long history of channelized rivers. Restoration advocates must take time to educate stakeholders and ensure their projects are compatible with local communities.

7.4. Design

The design of a project builds in factors of safety that are equivalent to those applied to any other civil engineering project. In doing so, geomorphologists and engineers should determine the type, size, location, and strength of the structures needed to withstand maximum forces and achieve the highest level of public and environmental protection.

7.5. Construction

Construction entails preparation of the site and delineation of the specific construction sequence, including site access, flow diversions and dewatering, major excavation and grading, careful placement of structural elements, fish removal and protection, water quality and erosion control, and revegetation. Construction of ELJs can range from relatively simple placement of large woody debris directly into a stream or river to more complex structures. The construction can be accomplished in many different ways, which can greatly affect the cost, regulatory compliance, and final outcome. Based on the complexity of these structures, it is essential that the designer be integrated into construction inspection.

7.6. Monitoring and Maintenance

Monitoring and maintenance provide periodic monitoring and maintenance of the structures. Monitoring should include an evaluation of structural integrity, scour, drift accumulation, and their ecological effects, such as surveys of fish and invertebrate use [e.g., *Abbe et al.*, 2003b, also unpublished report, 2005; *Brooks et al.*, 2004; *Brooks*, 2006]. Maintenance can include culling, repairing any structural damage, and revegetating, as needed. Too often, this phase is underemphasized or ignored.

Many things need considering in restoring and managing rivers, particularly when considering the reintroduction and management of wood debris. Figure 7 presents a checklist for the design of wood in river restoration [*Abbe et al.*, 2008], which is offered as a set of guidelines and reminders of the many factors for restoration design and river management.

8. CONCLUSION

Wood debris has been a natural part of the sediment load in rivers since woody vegetation appeared on Earth 360 million years ago. Both alive and dead trees have a significant influence on the morphology and habitat complexity of streams and rivers. Wood accumulations attenuate flood peaks, dissipate energy, trap sediment, deflect flows, and create anabranching channels, pools, and cover. Reintroducing wood to rivers is a critical component of habitat restoration in a wide range of environments throughout the world. We have presented some of the many issues to consider when designing wood structures in fluvial systems. Properly designed, wood debris structures, such as ELJs, have been very successful components of river restoration, whether used for grade control, flow deflection, pool formation, or increasing channel complexity and floodplain connectivity. Many issues need considering when doing any river restoration project, particularly with regard to wood debris. This consideration is even more important regarding the potential liability of placing flow obstructions in a river or structures that may be washed downstream if not properly designed and constructed. A great deal needs to be learned about ELJs including the hydraulics, longevity, influence on wood budgets, and effects of natural wood accumulation. For many river systems, wood is an essential element of any restoration and management planning. The key consideration should always be process when incorporating wood into river restoration planning. Wood structures should not simply be seen as yet another structural measure for controlling rivers.

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Development and Application of a Deterministic Bank Stability and Toe Erosion Model for Stream Restoration

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The Bank Stability and Toe Erosion Model (BSTEM) is a spreadsheet tool used to simulate stream bank erosion in a mechanistic framework. It has been successfully used in a range of alluvial environments in both static mode to simulate bank stability conditions and design of stream bank stabilization measures and iteratively over a series of hydrographs to evaluate surficial, hydraulic erosion, bank failure frequency, and the volume of sediment eroded from a bank over a given time period. In combination with the submodel RipRoot, reinforcing effects of riparian vegetation can be quantified and included in analysis of mitigation strategies. The model is shown to be very useful in testing the effect of potential mitigation measures that might be used to reduce the frequency of bank instability and decrease sediment loadings from stream banks. Results of iterative BSTEM analysis are used to extrapolate volumes of bank-derived sediment from individual sites to reaches when used with observations of the "percent reach failing." Results show that contributions of suspended sediment from stream banks can vary considerably, ranging from 10% in the predominantly low-gradient, agricultural watershed of the Big Sioux River, South Dakota, to more than 50% in two steep, forested watersheds of the Lake Tahoe Basin, California. Modeling of stream bank mitigation strategies shows that toe protection added to eroding stream banks can reduce overall volumes of eroded sediment up to 85%–100%, notwithstanding, that hydraulic erosion of the toe in this case makes up only 15%–20% of total bank erosion. BSTEM is available to the public free of charge at http://www.ars.usda.gov/Research/docs.htm?docid=5044.

1. INTRODUCTION

Sediment is one of the leading contributors to water-quality impairment in surface waters of the United States through its adverse effects on water supply and aquatic life-support processes. Stream bank erosion by mass failure represents an important form of channel adjustment and a

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significant source of sediment in disturbed streams, often contributing 60%–80% of the suspended-sediment load [*Simon and Rinaldi*, 2006].

Given the relatively important role of stream bank erosion in watershed sediment yields and channel adjustment, it is surprising that little, if any, quantitative information is available on the effectiveness of bank treatments on reducing erosion. Further, mechanistic bank stability analyses are rarely conducted in restoration activities or sediment load estimates as part of restoration or erosion-control activities. Bank failures generally occur by a combination of hydraulic processes that undercut the base of the bank and geotechnical processes causing bank collapse by gravity. The variables and processes that control stream bank erosion need to be predicted accurately under existing and remediated conditions to evaluate bank stabilization designs, existing stream bank-derived sediment loads, and the potential to alter sediment loads from stream banks. The fundamental premise to reduce loadings from stream bank erosion is, therefore, to either reduce the hydraulic and downslope forces and/or to increase the resistance of the bank toe to hydraulic forces and the resistance of the bank mass to downslope (gravitational) forces. Mitigation measures to reduce bank erosion might include some combination of bank toe protection to increase resistance to hydraulic forces, planting of vegetation on the bank top and face to increase the cohesive strength of the bank materials thereby making them more resistant to mass failure, or regrading the bank slope to a flatter angle to reduce the overall driving downslope force. All of these processes and conditions can be simulated with the deterministic Bank Stability and Toe Erosion Model (BSTEM) [*Simon et al.*, 2000].

BSTEM has been used statically to test for relative stability of a bank under given pore water pressure and vegetation conditions [*Pollen and Simon*, 2005; *Pollen-Bankhead and Simon*, 2009], to test for stable bank slope designs [*Simon et al.*, 2008], and to determine the importance of seepage undercutting relative to bank strength, bank angle, pore water pressure, and root reinforcement [*Cancienne et al.*, 2008]. With time series pore water pressure data, the model has been used quasi-dynamically to evaluate the important variables controlling bank stability [*Simon et al.*, 2000] and the mechanical and hydrologic effects of riparian vegetation [*Simon and Collison*, 2002; *Simon et al.*, 2006]. Most recently, BSTEM has been used iteratively to simulate hydraulic erosion at the bank toe and bank stability during a series of flow events for the purpose of evaluating current (existing) and potential changes in failure frequency and stream bank-derived sediment loads [*Simon et al.*, 2010].

The purpose of this chapter is to demonstrate the application of BSTEM as a viable, mechanistic tool for three typical stream restoration objectives: (1) determining stable bank conditions under a variety of environmental conditions and erosion-control strategies, (2) quantifying bank-widening rates and sediment loads emanating from stream banks, and (3) determining potential reductions in widening rates and sediment loads under a range of mitigation techniques. The model is applied herein under static conditions to design a bank stabilization project and iteratively over a series of annual hydrographs in diverse environments to predict sediment loads and potential load reductions from stream bank erosion.

2. BANK STABILITY AND TOE EROSION MODEL

BSTEM is a mechanistic bank stability model specifically designed for alluvial channels. It is programmed in Visual Basic and exists in the Microsoft Excel environment as a simple spreadsheet tool. Data input, along with the various subroutines are included in different worksheets including Input Geometry, Bank Material, Bank Vegetation and Protection, Bank Model Output, and Toe Model Output. The user is able to move freely between worksheets according to their needs at various points of model application. BSTEM is available to the public free of charge at http://www.ars.usda.gov/Research/docs.htm?docid=5044.

2.1. General Model Capabilities

The original model developed by *Simon et al.* [1999, 2000] is a limit equilibrium analysis in which the Mohr-Coulomb failure criterion is used for the saturated part of the stream bank, and the *Fredlund et al.* [1978] criterion is used for the unsaturated part. The latter criterion indicates that apparent cohesion changes with matric suction (negative) pore water pressure, while effective cohesion remains constant. In addition to accounting for positive and negative pore water pressures, the model incorporates complex geometries, up to five user-definable layers, changes in soil unit weight based on water content, and external confining pressure from streamflow. Current versions combine three limit equilibrium-method models that calculate factor of safety (F_s) for multilayer stream banks. The methods simulated are horizontal layers [*Simon et al.*, 2000], vertical slices with tension crack [*Morgenstern and Price*, 1965], and cantilever failures [*Thorne and Tovey*, 1981]. The model can easily be adapted to incorporate the effects of geotextiles or other bank stabilization measures that affect soil strength.

The version of BSTEM used throughout this chapter (version 5) includes a submodel to predict bank toe and bank surface erosion and undercutting by hydraulic shear. This is based on an excess shear stress approach that is linked to the geotechnical algorithms. Complex geometries resulting from simulated bank toe erosion are used as the new input geometry for the geotechnical part of the bank stability model. The geometry of the potential failure plane can be input by the user or can be determined automatically by an iterative search routine that locates the most critical failure-plane geometry. If a failure is simulated, that new bank geometry can be exported back into either submodel to simulate conditions over time by running the submodels iteratively with different flow and water table conditions. In addition, the bank stability submodel automatically selects between cantilever and planar-failure modes. The mechanical, reinforcing effects of riparian vegetation [*Simon and Collison*, 2002; *Micheli and Kirchner*, 2002] can be included in model simulations. This is accomplished with the RipRoot model [*Pollen and Simon*, 2005] that is based on fiber-bundle theory and included in the Bank Vegetation and Protection worksheet. The current version of BSTEM (version 5) also includes new features that can account for enhanced hydraulic stresses on the outside of meander bends as well as reduced, effective hydraulic stress operating on fine-grained materials in a reach characterized by a rougher boundary.

2.1.1. Bank-Toe Erosion Submodel. The Bank-Toe Erosion submodel is used to estimate erosion of bank and bank toe materials by hydraulic shear stresses. The effects of toe protection are incorporated into the analysis by changing the characteristics of the toe material in the model. The model calculates an average boundary shear stress from channel geometry and flow parameters using a rectangular-shaped hydrograph defined by flow depth and the duration of the flow (steady, uniform flow). The assumption of steady, uniform flow is not critical insomuch as the model does not attempt to rout flow and sediment and is used only to establish the boundary shear stress for a specified duration along the bank surface. The model also allows for different critical shear stress and erodibility of separate zones with potentially different materials at the bank and bank toe. The bed elevation is fixed because the model does not incorporate the simulation of bed sediment transport. Toe erosion by hydraulic shear is calculated using an excess shear approach. The average boundary shear stress (τ_o) acting on each node of the bank material is calculated using

$$\tau_o = \gamma_w RS, \tag{1}$$

where τ_o is average boundary shear stress (Pa), γ_w is unit weight of water (9.81 kN m^{-3}), R is local hydraulic radius (m), and S is channel slope (m m^{-1}).

The average boundary shear stress exerted by the flow on each node of the bank profile is determined by dividing the flow area at a cross section into segments. A line is generated that separates the bed- and bank-affected segments (starting at the base of the bank and extending to the water surface) at an angle equal to the average of the bank and bank toe angles. The hydraulic radius (R) of the flow on each segment is the area of the segment (A) divided by the wetted perimeter of the segment (P_n). Thus, the shear stress varies along the bank surface according to equation (1) as parameters comprising the segmented areas change.

An average erosion rate (in m s^{-1}) is computed for each node by utilizing an excess shear stress approach [*Partheniades*, 1965]. This rate is then integrated with respect to time to yield an average erosion distance in centimeters. This method is similar to that employed in the CONCEPTS model [*Langendoen*, 2000], except that here, erosion is simulated to occur normal to the local bank angle and not horizontally:

$$E = k \quad \Delta t(\tau_o - \tau_c), \tag{2}$$

where E is erosion distance (cm), k is erodibility coefficient (cm^3 (N-s)$^{-1}$), Δt is time step (s), τ_o is average boundary shear stress (Pa), and τ_c is critical shear stress (Pa).

Resistance of bank toe and bank surface materials to erosion by hydraulic shear is handled differently for cohesive and noncohesive materials. Originally, for cohesive materials, the relation developed by *Hanson and Simon* [2001] using a submerged jet test device [*Hanson*, 1990, 1991] was used

$$k = 0.2\tau_c^{-0.5}. \tag{3a}$$

This relation has been recently updated based on hundreds of tests on stream banks across the United Sates [*Simon et al.*, this volume]:

$$k = 1.62\tau_c^{-0.838}. \tag{3b}$$

The *Shields* [1936] criterion is used for resistance of noncohesive materials as a function of roughness and particle size (weight) and is expressed in terms of a dimensionless critical shear stress:

$$\tau_c^* = \tau_o/[(\rho_s - \rho_w)gD], \tag{4}$$

where τ_c* is critical dimensionless shear stress, ρ_s is sediment density (kg m^{-3}), ρ_w is water density (kg m^{-3}), g is gravitational acceleration (m s^{-2}), and D is characteristic particle diameter (m).

2.1.2. Bank Stability Submodel. The bank stability submodel combines three limit equilibrium methods to calculate a factor of safety (F_s) for multilayered stream banks. The methods simulated are horizontal layers [*Simon and Curini*, 1998; *Simon et al.*, 2000], vertical slices for failures with a tension crack [*Morgenstern and Price*, 1965], and cantilever failures [*Thorne and Tovey*, 1981].

For planar failures without a tension crack, the factor of safety (F_s) for both the saturated and unsaturated parts of the failure plane is given by the ratio of the resisting and driving forces:

$$F_s = \frac{\sum_{i=1}^{I} \left(c_i' L_i + S_i \tan \phi_i^b + [W_i \cos \beta - U_i + P_i \cos(\alpha - \beta)] \tan \phi_i'\right)}{\sum_{i=1}^{I} \left(W_i \sin \beta - P_i \sin[\alpha - \beta]\right)}, \quad (5)$$

where c_i' is effective cohesion of ith layer (kPa), L_i is length of the failure plane incorporated within the ith layer (m), S_i is force produced by matric suction on the unsaturated part of the failure surface (kN m^{-1}), ϕ^b is angle representing the rate of increase in shear strength with increasing matric suction (degrees), W_i is weight of the ith layer (kN), U_i is the hydrostatic-uplift force on the saturated portion of the failure surface (kN m^{-1}), P_i is the hydrostatic-confining force due to external water level (kN m^{-1}), β is failure-plane angle (degrees from horizontal), α is bank angle (degrees from horizontal), ϕ' is angle of internal friction (degrees), and I is the number of layers.

The cantilever shear failure algorithm is a further development of the method employed in the CONCEPTS model [*Langendoen*, 2000]. BSTEM can utilize the different failure algorithms depending on the geometry and conditions of the bank. Determining whether a failure is planar or cantilever is based on whether there is undercutting and then comparing the factor of safety values. The failure mode is automatically determined by the smaller of the two values. The model is easily adapted to incorporate the effects of geotextiles or other bank stabilization measures that affect soil strength. This current version (5) of the model assumes hydrostatic conditions below the water table. Matric suction above the water table (negative pore water pressure) is calculated by linear extrapolation.

2.1.3. Root Reinforcement (RipRoot) Submodel. Waldron [1977] extended the Coulomb equation for root-permeated soils by assuming that all roots extended vertically across a horizontal shearing zone and that the roots act like laterally loaded piles, with tension transferred to them as the soil is sheared. In the *Waldron* [1977] model, the tension developed in the root as the soil is sheared is resolved into a tangential component resisting shear and a normal component increasing the confining pressure on the shear plane. ΔS can be represented by

$$\Delta S = T_r(\sin\theta + \cos\theta \tan\phi)(A_R/A), \quad (6)$$

where T_r is the average tensile strength of roots per unit area of soil (kPa), A_R/A is the root area ratio (dimensionless), and θ is the angle of shear distortion in the shear zone.

Gray [1974] reported that the angle of internal friction of the soil appeared to be affected little by the presence of roots. Sensitivity analyses carried out by *Wu et al.* [1979] showed that the value of the first angle term in equation (6) is fairly insensitive to normal variations in θ and ϕ (40°–90° and 25°–40°, respectively) with values ranging from 1.0 to 1.3. A value of 1.2 was therefore selected by *Wu et al.* [1979] to replace the angle term, and the simplified equation becomes

$$\Delta S = 1.2 T_r (A_R/A). \quad (7)$$

According to the simple perpendicular root model of *Wu et al.* [1979], the magnitude of reinforcement simply depends on the amount and strength of roots present in the soil. However, *Pollen et al.* [2004] and *Pollen and Simon* [2005] found that these perpendicular root models tend to overestimate root reinforcement due to the inherent assumption that the full tensile strength of each root is mobilized during soil shearing and that the roots all break simultaneously. This overestimation was largely corrected by *Pollen and Simon* [2005] by constructing a fiber-bundle model (RipRoot) to account for progressive breaking during mass failure. Validation of RipRoot versus the perpendicular model of *Wu et al.* [1979] was carried out by comparing results of root-permeated and nonroot-permeated direct-shear tests. The direct-shear tests revealed that accuracy was improved by an order of magnitude by using RipRoot estimates [*Pollen and Simon*, 2005; *Mickovski et al.*, 2009].

A later paper by *Pollen* [2007] investigated the forces required to pull out roots in a field study, and the RipRoot model was modified to account for both root failure mechanisms. The addition of pullout forces allowed for estimations of spatial variability in root reinforcement with changes in soil texture and temporal changes with changes in soil water. In the RipRoot model currently embedded in BSTEM 5, a vegetation assemblage can be created by accessing the species database contained in the submodel; the user enters species, approximate vegetation ages, and approximate percent cover of each species at each site to estimate root density. This database includes tests performed across the United States. Root reinforcement values are then calculated automatically using RipRoot's progressive breaking algorithm.

Table 1. Required User-Input Parameters for BSTEM[a]

Driving Forces			Resisting Forces		
Parameter	Purpose	Source	Parameter	Purpose	Source
Hydraulic Processes: Bank Surface					
Channel slope (S)	boundary shear stress (τ_o)	field survey or design plan	particle diameter (D) (cohesionless)	critical shear stress (τ_c)	bulk sample particle size (cohesionless); default values in model
			critical shear stress (τ_c) (cohesive)	critical shear stress (τ_c)	jet test (cohesive); default values in model
Flow depth (h)	boundary shear stress (τ_o)	field survey, gauge information, design plan	particle diameter (D) (cohesionless)	erodibility coefficient (k)	bulk sample particle size (cohesionless); default values in model
			critical shear stress (τ_c) (cohesive)	erodibility coefficient (k)	jet test (cohesive); default values in model
Unit weight of water (γ_w)	boundary shear stress (τ_o)	considered constant, 9810 N m^{-3}			
Geotechnical Processes: Bank Mass					
Unit weight of sediment (γ_s)	Weight (W), Normal force (σ)	core sample in bank unit; default values in model	unit weight of sediment (γ_s)	weight (W), normal force (σ)	core sample in bank unit; default values in model
Bank height (H)	Shear stress	field survey or design plan	effective cohesion (c')	shear strength (τ_f)	borehole shear, direct shear, triaxial shear; default values in model
Bank angle (α)	Shear stress	field survey or design plan	effective friction angle (ϕ')	shear strength (τ_f)	
			pore water pressure (μ_w)	shear strength (τ_f)	interpolated from water table

[a]Default values for geotechnical parameters are shown in Table 2.

2.2. Data Requirements

As BSTEM is a mechanistic model, the data required to operate the model are all related to quantifying the driving and resisting forces that control the hydraulic and geotechnical processes that operate on a stream bank. Input-parameter values can all be obtained directly from field surveying and testing. If this is not possible, the model provides default values by material type for many parameters. It has been our experience that all of the data needed to run BSTEM can be collected at a site by a crew of four within 1 day. Required data fall into three broad categories: (1) bank geometry and stratigraphy, (2) hydraulic data, and (3) geotechnical data. A summary of the required input parameters is provided in Table 1. The default geotechnical values that are included in the model are provided in Table 2.

2.3. General Model Limitations

BSTEM can simulate the most common types of bank failures that typically occur along alluvial channels. Once failure is simulated, the failed material is assumed to enter the flow. The model does not simulate rotational failures that generally occur in very high banks of homogeneous, fine-grained materials characterized by low bank angles. Although potentially damaging with regard to the amount of

Table 2. Default Values in BSTEM for Geotechnical Properties[a]

Soil Type	Statistic	c' (kPa)	ϕ' (°)	γ_{sat}(kN m^{-3})
Gravel (uniform)[b]		**0.0**	**36.0**	**20.0**
Sand and gravel[b]		**0.0**	**47.0**	**21.0**
Sand	75th percentile	1.0	32.3	19.1
	median	**0.4**	**30.3**	**18.5**
	25th percentile	0.0	25.7	17.9
Loam	75th percentile	8.3	29.9	19.2
	median	**4.3**	**26.6**	**18.0**
	25th percentile	2.2	16.7	17.4
Clay	75th percentile	12.6	26.4	18.3
	median	**8.2**	**21.1**	**17.7**
	25th percentile	3.7	11.4	16.9

[a]Data derived from more than 800 in situ direct-shear tests with the Iowa Borehole Shear Tester except where indicated. BSTEM values are indicated in bold.
[b]Data from *Hoek and Bray* [1977].

Table 3. Potential Alternative Means to Control the Two Primary Processes That Control Stream Bank Stability

Means of Control	Hydraulic Protection	Geotechnical Protection
Increase critical shear stress	bank toe and face armoring with rock, large wood, live vegetation	
Decrease applied shear stress	redirect flows, reduce channel slope (remeandering), increase bottom width, live vegetation (increased roughness)	
Increase bank shear strength		pole and post plantings, bank top vegetation, brush layers, drainage
Decrease driving, gravitational forces		reduce bank height, terraces, flatten bank slope; buttress bank toe

land loss, these failures are not common along alluvial streams. Another limitation of the current version of BSTEM is that it cannot simulate a dynamic water table and, therefore, dynamic pore water pressure distributions. The elevation of the phreatic surface must be input by the user. Vertical distributions of pore water pressure (below the water table) and matric suction (above the water table) are then calculated by the model through linear interpolation. Bank undercutting by seepage erosion is similarly not included in the version described herein. Finally, the hydrologic effects of riparian vegetation, including interception, evapotranspiration, and the accelerated delivery of water along roots and macropores cannot be simulated at this time. A research version of BSTEM currently used by scientists at the U.S. Department of Agriculture Agricultural Research Service National Sedimentation Laboratory does include a near-bank groundwater submodel that permits dynamic adjustment of pore water pressures over extended hydrographs. This dynamic version of BSTEM will be made available to the public in the near future.

3. BANK STABILITY MODELING FOR STREAM RESTORATION

Bank stability modeling is an important, if not critical, component of stream restoration or erosion-control activities that pertain to excess sediment loads or potential risk of adjacent lands and infrastructure. There are at least three restoration objectives that can benefit greatly from the use of a mechanistic tool to reliably predict sediment loadings and widening rates from stream bank erosion. These include the following: (1) determining bank stability conditions under a range of hydraulic and geotechnical conditions and erosion-control strategies, which includes designing sustainable bank stabilization measures and determining unstable bank conditions to assure continued delivery of sediment to the channel (in cases where there is insufficient supply [i.e., *Wyżga et al.*, this volume]), (2) quantifying bank-widening rates and sediment loads emanating from stream banks, and (3) determining potential reductions in widening rates and sediment loads under a range of mitigation techniques.

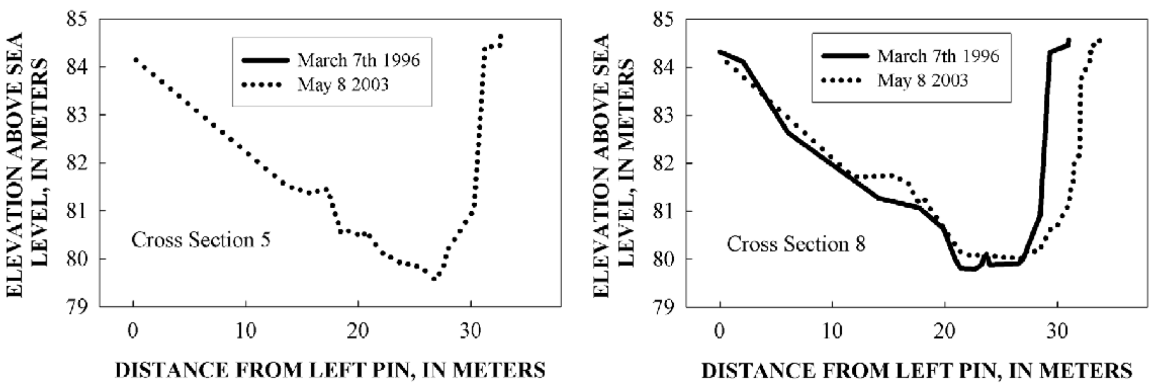

Figure 1. Changes in geometry between March 1996 and May 2003 at two cross sections along the Goodwin Creek bendway showing continued bank retreat. Modified from the work of *Simon et al.* [2008], reprinted with permission from American Society of Civil Engineers (ASCE).

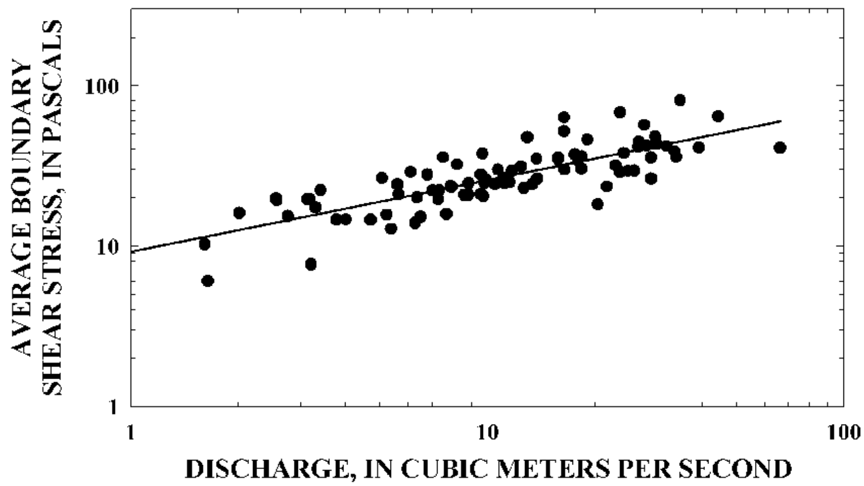

Figure 2. Relation between discharge and average boundary shear stress (τ_o) in the Goodwin Creek bendway. Data are calculated by multiplying measured peak flow depths with bed slope and unit weight of water.

Any restoration objective that requires reduction of sediment loads from stream banks must focus on mitigation measures that directly affect the processes that control stream bank stability, namely, hydraulic erosion and geotechnical instability (Table 3). Protection from hydraulic processes must either reduce the available boundary shear stress and/or increase the shear resistance to particle detachment, thereby reducing the likelihood and magnitude of bank toe steepening. Protection from geotechnical instability must focus on increasing soil shear strength and/or decreasing the driving (gravitational) forces to reduce the likelihood of mass failure of the upper bank.

Implementation of any design plan requires the analysis of the hydraulic and geotechnical processes likely to exist at the site, particularly during worst-case conditions. For hydraulic processes, these occur at peak flows when boundary shear stresses are greatest. For geotechnical processes, these generally occur during a wet period and following recession of peak stage when pore water pressures in the bank are at a maximum, and the confining pressure provided by the flow on the bank has been lost. This is referred to as the "drawdown" condition.

To address the first objective (above), BSTEM can be run for worst-case conditions under existing conditions to test

Table 4. Data Requirements for Using Riprap Sizing Spreadsheet[a]

Parameter (Symbol)	Units	Comments/Values
Channel width (W)	m	from field survey or design plan
Flow depth (h)	m	worst-case design flow or gauge information
Bed slope (S)	m m^{-1}	from field survey or design plan
Bank angle (θ)	degrees	from field survey or design plan
Specific gravity (G)	dimensionless	considered constant: 2.65
Angle of repose of rip rap (ϕ)	degrees	from literature[b]
Critical Shields parameter (τ^*_c)	dimensionless	typical: 0.032, 0.047, 0.06
Specific weight of water (γ_w)	N m^{-3}	considered constant: 9810 N m^{-3}

[a]Data requirements are by *Julien* [2002].

[b]See the work of *Simons and Sentruk* [1992, p. 413] and/or the work of *Selby* [1982, p. 54].

Table 5. Riprap Sizing Results Using *Julien*'s [2002] Spreadsheet Tool[a]

Flow Depth (m)	Slope	τ_c*	Size (cm)
3	0.003	0.032	*38.0*
3.5	0.003	0.032	*44.3*
4	0.003	0.032	*50.6*
4.5	0.003	0.032	*56.9*
3	0.003	0.047	25.9
3.5	0.003	0.047	30.2
4	0.003	0.047	34.5
4.5	0.003	0.047	38.8
3	0.003	0.06	20.3
3.5	0.003	0.06	23.6
4	0.003	0.06	27.0
4.5	0.003	0.06	30.4

[a]Implementation was based on the most conservative results (in italics); maximum stone sizes calculated at τ_c* of 0.032.

Table 6. Measured Geotechnical Parameters Used in Bank Stability Modeling

Layer	Friction Angle (deg)	Effective Cohesion (kPa)	Saturated Unit Weight (kN m^{-3})
1	33.1	1.4	16.9
2	28.1	2.7	19.3
3	28.1	2.7	19.3
4	27.0	6.3	20.0
5	36.0	0.0	20.0

whether "no action" is a viable option. Perhaps a single failure episode which will result in a flatter bank angle may be sufficient to reduce widening rates and sediment loads. If this is not the case, BSTEM can be run testing various combinations of mitigation strategies (Table 3) again under worst-case conditions. Restoration activities that involve one of the other two objectives (above) need to rely on iterative simulations with BSTEM over some specified period. This may be an annual hydrograph, series of annual hydrographs, or a selection of annual hydrographs representing the range of the annual flow series. Results from this latter approach can then use weighted load values (based on frequency of occurrence) to obtain mean annual loading rates. The following case studies will provide example applications for each of these restoration objectives.

3.1. Restoration Objective 1: Designing a Sustainable Bank Stabilization Project

Periodic channel surveys of a 4.7 m high bendway on Goodwin Creek, Mississippi, United States, during the period 1977 to 1996, in conjunction with dating of woody vegetation growing on the channel banks and bars at the study site, disclosed that rates of bank failure and channel migration over the period were relatively uniform at about 0.5 m yr^{-1}. Surveys conducted after every major flow event between 1996 and 2007 showed that migration rates continued at about 0.5 m yr^{-1}, resulting in about 20 m of bank retreat between 1966 and 2006 and the types of channel changes shown in Figure 1.

Based on fundamental processes of stream bank stability and knowledge attained from field observations, it was apparent that both hydraulic and geotechnical protection would probably be required to stabilize the 100 m long reach (Table 3). Hydraulically, the concept was to provide greater roughness and erosion resistance to the bank toe region. Bank toe protection was to be conducted using rock for two primary reasons. First, the use of engineered log jams in deeply incised streams of the southeastern United States has not been successful in some cases [*Shields*, 2003; *Shields et al.*, 2004]. Second, the cooperator on the project from the U. S. Army Corps of Engineers had great experience and success with using rock at the bank toe to resist hydraulic erosion and undercutting. Options for the design of the upper part of the bank were limited due to landowner constraints requiring the plan to retain a field road whose edge was located 5 m from the bank top edge. Thus, if required, bank slopes could only be flattened from the preproject slope of 75°–80° to 45° (1:1). Because the top bank edge could only be moved about 2 m landward, construction of the 1:1 slope would have to take place by filling, using material derived from the point bar on the inside part of the bend. In addition, if additional bank material strength was required to increase the shearing resistance of the bank mass, a planting scheme was devised using a range of woody riparian species to provide root reinforcement.

Stone size was selected based on a simple one-dimensional hydraulic analysis [*Julien*, 2002] such that the stone would not be mobilized at peak flows where average boundary shear stresses can reach 60–80 N m^{-2} (Pa) (Figure 2). Data required for this analysis can be obtained in the field or, for

Table 7. Summary of Simulation Results Using BSTEM for Goodwin Creek Bendway Representing Existing, After Initial Bank Failure, and Designed 1:1 Slope Geometries for the Case of Low-Flow and Drawdown Conditions[a]

Case	Geometry	Dominant Vegetation	Groundwater Elevation	F_s	Interpretation
1	existing	none	at flow level	1.10–1.56	stable
2	existing	none	moderate	0.87	unstable
3	after failure	none	moderate	1.44	stable
4	after failure	none	high	0.45	unstable
5	1:1 slope	none	moderate	2.10	stable
6	1:1 slope	none	high	0.67	unstable
7	1:1 slope	black willow	high	0.81	unstable
8	**1:1 slope**	**eastern sycamore, river birch**	**high**	**1.28**	**stable**

[a]Case 8 (bold) represents most stable design case.

Figure 3. View of the bendway looking upstream (left) from January 2006 and immediately following construction in March 2007. (middle) Note edge of constructed rock riffle in lower right and (right) stone-toe protection with three bendway weirs. From the work of *Simon et al.* [2008], reprinted with permission from ASCE.

the case of angle of repose for rip rap, from literature values [*Simons and Sentruk*, 1992, p. 413; *Selby*, 1982, p. 54] (see Table 4). Calculations were made for 3.0 to 4.5 m deep flows at a slope of 0.003 using typical Shields parameter values (τ_c*) of 0.032, 0.047, and 0.06 (Table 5). The most conservative results were obtained using the lowest value of τ_c* (0.032). Results for this case showed recommended stone sizes of 38 cm for the 3 m deep flow to 57 cm for the 4.5 m deep flow. From this analysis, it was determined that a combination of R-200 and R-650 stone, graded from 2.5 to 40 and 60 cm, respectively, would be sufficient.

Geotechnical data on bank material shear strength were collected during earlier phases of the bendway research (see Table 6); BSTEM version 5 was employed to simulate stability for preconstruction and initial (1:1 slope) design conditions. Effective cohesion and friction angle were obtained in situ using a borehole shear test device [*Lohnes and Handy*, 1968; *Lutenegger and Hallberg*, 1981]. Simulation of existing bank stability conditions supported observations over the past 10 years where, under low-flow conditions and a relatively deep near-bank groundwater table, banks were stable but become unstable with higher levels of saturation (Table 7). Keeping the geotechnical properties of the banks constant, the simulations were repeated with the designed 1:1 geometry. Much like the results for the existing geometry, the designed slope would be stable at low-flow conditions but unstable for the drawdown case (Table 7). In an attempt to increase the factor of safety under drawdown conditions, simulations were conducted to include root reinforcement provided by common riparian species in the top 1.0 m of the bank [*Simon and Collison*, 2002; *Pollen and Simon*, 2005]. This was attempted initially using black willow because this is one of the most commonly used woody riparian species in restoration work. Results, however, produced a F_s of 0.81, still indicative of instability. Simulations were repeated with eastern sycamore, which has been shown along with river birch to provide the greatest amount of root reinforcement over the top 1.0 m of the bank [*Simon and Collison*, 2002]. In this case, the F_s for the critical, drawdown case increased to 1.28, at the upper limit

Figure 4. Views of the Goodwin Creek bendway looking downstream during (left) February 1997 (pre project) and (right) July 2009 (post project). Construction took place 26 February to 2 March 2007.

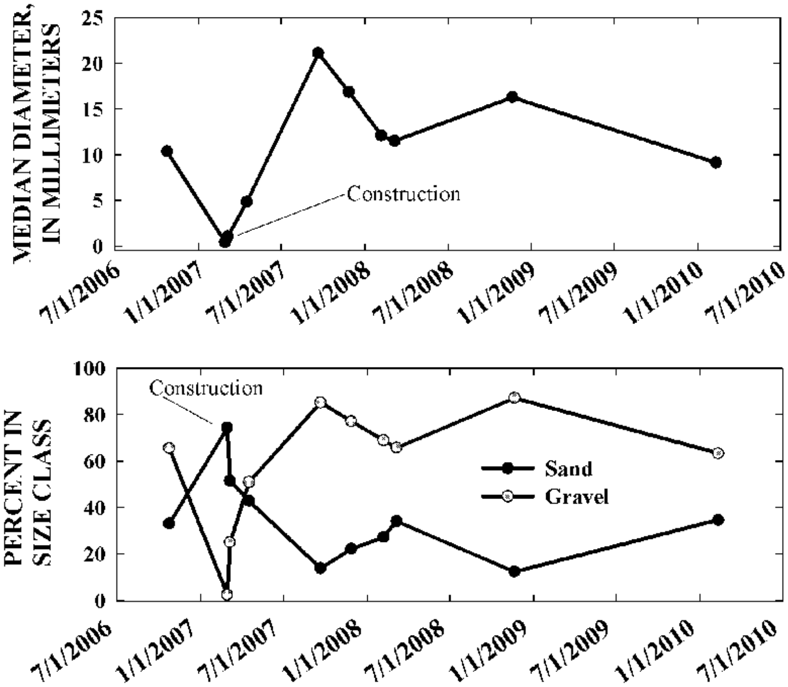

Figure 5. Variation in bed material particle size. Note the shift to sand bed during construction and the return to gravel bed following flushing of the fine material following construction.

of conditional stability. It is important to note, however, that it may take up to 3 years for riparian plants to start providing significant root reinforcement to the bank.

3.1.1. Final Design and Implementation. Based on the hydraulic and geotechnical analysis described in the preceding section, the overall design plan was implemented. About 275 t of both R-200 and R-650 stone costing between \$27 t^{-1} and \$33 t^{-1} were delivered to the site in late February 2007. Stone-toe protection along with three bendway weirs were constructed and placed. Material from the point bar was excavated and used to build the 1:1 slope on the left bank. All woody material removed from the bar was reused on the constructed bank. Vegetation that was excavated whole was

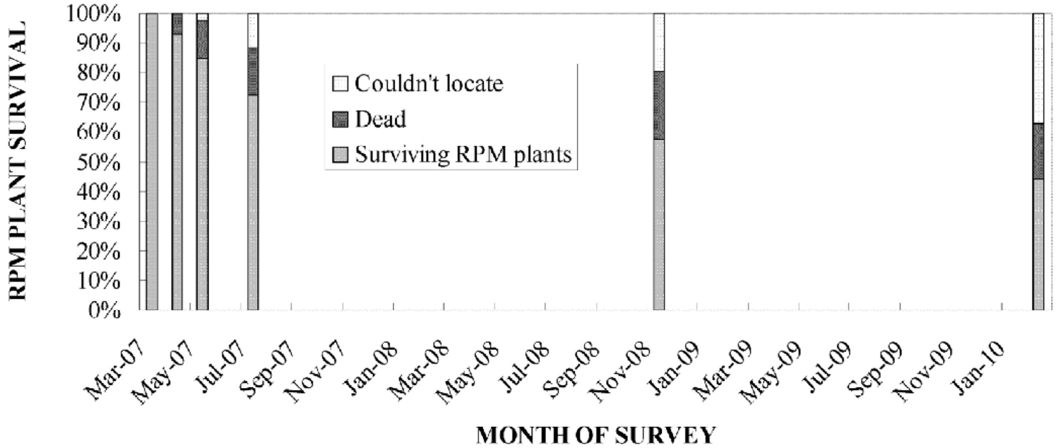

Figure 6. Survival rate of purchased root production method (RPM) plants between construction (March 2007) and February 2010. Note RPM indicates root production method, which produces faster growth and greater root biomass.

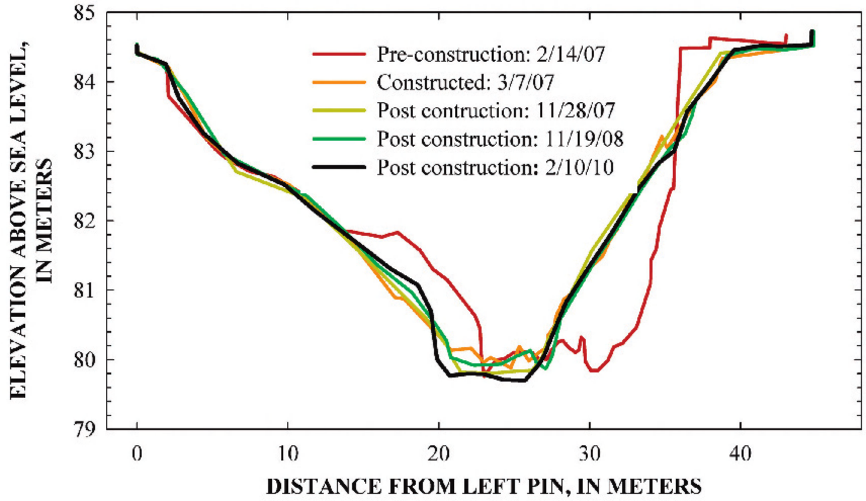

Plate 1. Comparison of channel geometry at cross section 8 (see Figure 1). Differences in bank geometry are largely a function of rod placement during surveys. Note streambed scour between March and November 2007. Dashed line represents highest stage over the period.

replanted on the constructed bank (including mature trees). Native species were selected and purchased based on the dominant species of surrounding riparian buffers, with assemblages largely composed of sycamore, river birch, and sweetgum trees. Other vegetation was cut and trimmed with the branches used for post plantings, and stems were placed on the ground along the contour to intercept overland flow that might be generated from the field road. The entire bank was then seeded with grass and overlain with straw (Figure 3). Construction took place between 26 February and 2 March. The cost of the entire project was $33,000 or $330 m^{-1}.

3.1.2. Postproject Monitoring. To evaluate the performance of the design scheme, a limited monitoring program was put in place following construction. Immediately following construction, the reach was surveyed, and samples of bed material were taken at numerous cross sections. Each of the purchased plants was tagged, their diameter was noted, and they were located with a GPS unit. Discharge was monitored continuously at the stream gauges just upstream of the reach.

Over the period March 2007 to February 2010, there was no hydraulic erosion at the bank toe and no mass failures of the upper part of the bank (Figure 4). The most significant change in the channel was up to 0.5 m of scour along some parts of the streambed (Plate 1). This was expected because of (1) the redirection of flows into the center of the channel and (2) the temporary fining of the streambed in some places due to construction activities. For example, at one of the cross sections, prior to construction, the streambed was composed of 80% gravel compared to 13% gravel immediately after construction. Two postconstruction storm events flushed much of the sand-sized material out of the cross section, and by November 2007, the streambed again was composed of 80% gravel (Figure 5). Similar trends occurred for all cross sections. Because 2007 was a relatively dry year, plants had to be watered periodically during the first growing season. The survival rate of the purchased plants is shown to decrease to about 75% through July 2007, to 55% through November 2008, and to 45% through February 2010 (Figure 6). Survival rates may, in fact, be greater by about 40% representing those plants that could not be located.

3.2. Restoration Objectives 2 and 3: Iterative Modeling to Quantify Sediment Delivery From Stream Banks and Potential Reductions using Different Mitigation Strategies

To address restoration objectives that require a determination of gross amounts of sediment delivery from stream banks, simulations must be performed over a range of hydraulic and geotechnical conditions representing series of flow hydrographs. Quantifying stream bank erosion is not a matter of developing a simple relation (i.e., power function) between flow and sediment delivery. Moderate flows may undercut the bank toe but still not cause mass failure unless bank saturation causes sufficient loss of matric suction and generation of pore water pressure to weaken the bank mass to result in failure. High flows that are often effective at eroding

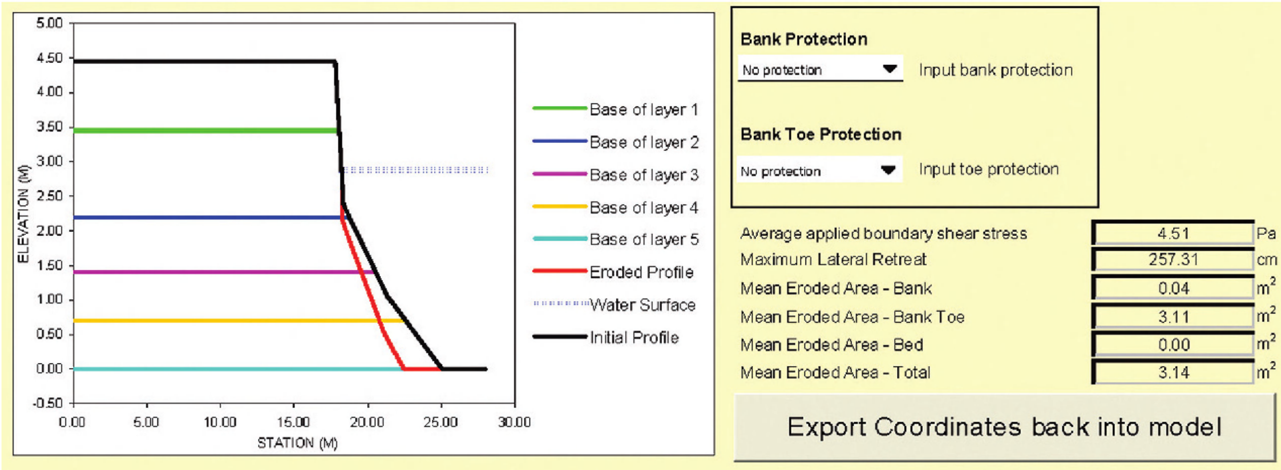

Plate 2. Example results from toe erosion submodel of first flow event and resulting hydraulic erosion.

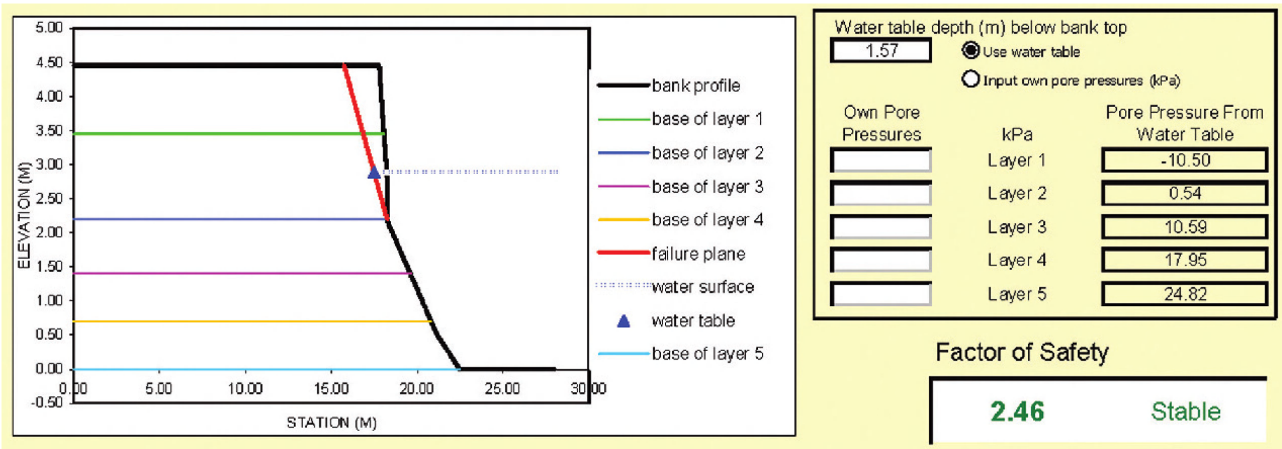

Plate 3. Example results from the bank stability submodel following the first flow event. This simulation shows a stable bank.

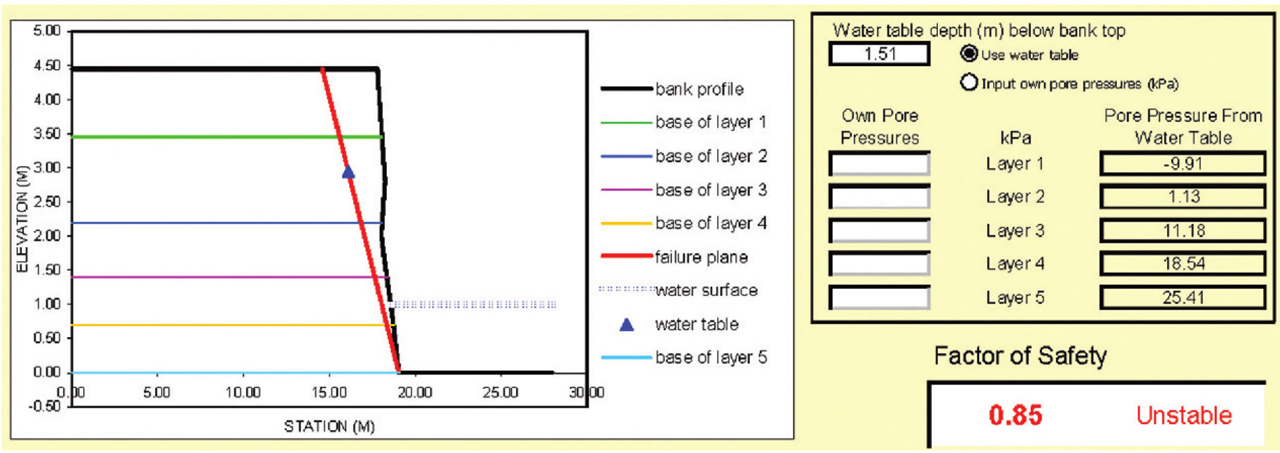

Plate 4. Example results from the bank stability submodel showing an unstable bank under drawdown conditions. In this case, the bank geometry exported to simulate the next flow event is represented by the failure plane (in red) and the original bank toe.

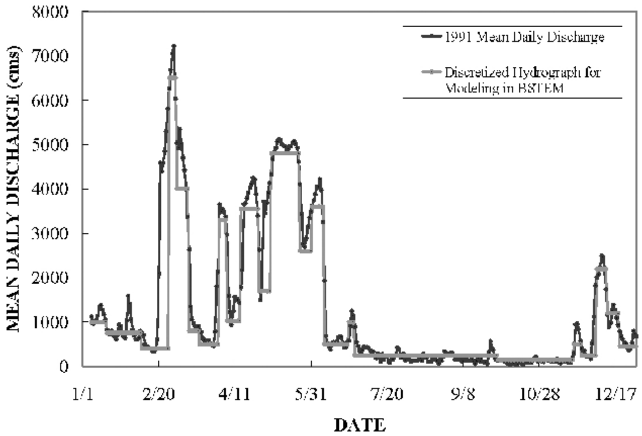

Figure 7. Example of discretized hydrograph for the Lower Tombigbee River, Alabama (below Coffeeville Dam) for the 90th percentile flow year (1991). The numbers 1–6 indicate the storms with discharges that exceeded the critical shear stress of the toe material and were modeled iteratively at this site. From the work of *Bankhead et al.* [2008].

the bank toe may prevent, or at least delay, mass failure because of the confining force provided by the flow that buttresses the bank. In fact, bank failures commonly occur on the recessional limb of storm hydrographs when the banks have lost geotechnical strength due to the effects of pore water pressure and the confining force provided from streamflow. It is for these reasons that analysis, therefore, must be conducted iteratively for a series of hydrographs so that variations in pore water pressure and surface water stage can be accounted for.

Depending on the project needs, iterative model runs were made to represent stream bank dynamics for a range of flow years: a 90th or 99th percentile flow year is used to represent a very wet year and, therefore, potential worst-case conditions for erosion, bank failures, and suspended sediment loadings. A range of flow years was also modeled in two of the case studies presented herein so that the suspended sediment loadings from an average annual year could be calculated. In addition, BSTEM was run with different mitigation strategies to see how, for example, the effect of placing rock at the bank toe or growing different types of vegetation on the banks might affect bank stability and sediment delivery to the channel. The following sections outline the general methods and results of three studies carried out using iterative runs of BSTEM. The flow years and mitigation strategies modeled varied according to the river system being studied and the project objectives.

3.2.1. Lower Tombigbee River, Alabama. Stream bank erosion is prevalent along the Lower Tombigbee River, Alabama. Aerial reconnaissance using GPS-linked video indicated that more than 50% of all banks along the study reach between river kilometer (RKM) 115 and 417 have experienced recent bank failures [*Bankhead et al.*, 2008]. Associated with this erosion is the loss of land and property. Taking the average widening rate of 1.2 m yr^{-1} over the 29 year period of air photo analysis and multiplying by the length of the study reach (301 km) provided an estimate of the total land loss over the period. This was equivalent to about 1040 ha. Given this considerable amount of land loss from bank instabilities, potential strategies to reduce the magnitude and

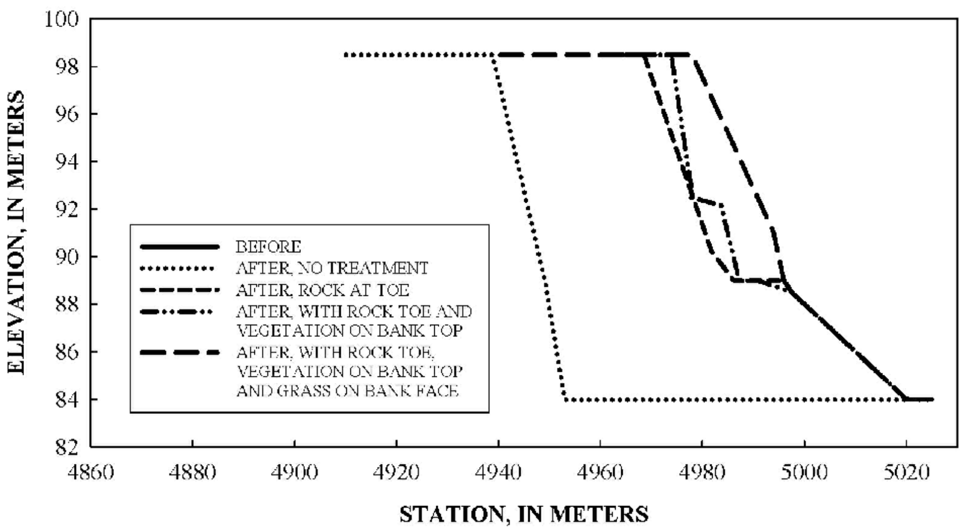

Figure 8. Summary of iterative modeling results for alternative mitigation strategies showing the volume of failures for each bank condition. Modified from the work of *Bankhead et al.* [2008].

Table 8. General Site Characteristics for the Modeled Stream Banks

Stream	River Kilometer (RKM)	Bank Height (m)	Special Characteristics
Blackwood Creek	1.94	3.0	no top bank vegetation
	2.39	2.4	Lemmon's willow (moderate)
Upper Truckee River	4.51	2.6	meadow vegetation
	8.45	1.9	mixed meadow and woody vegetation
	13.1	2.7	golf course with lodgepole pine
Ward Creek	2.48	14.9	14.9 m steep, terrace slope adjacent to channel; coarse material at toe; mature conifers
	3.60	1.3	meadow vegetation

frequency of bank failures along the study reach were investigated.

As an example, a series of alternative strategies to reduce the magnitude and frequency of bank failures was simulated for the site at RKM 184.5. Given that the BSTEM model simulates failures in two dimensions (height and width), a reach length of 100 m was assumed to provide results in m^3. Simulations were conducted in such a way as to be able to quantify the reduction in the frequency of failures and the volume of material delivered to the channel by bank failures. This was accomplished by running the toe erosion and bank stability submodels iteratively for a high-flow year (1991, 90th percentile flow year) to represent worst-case flow conditions. The 1991 flow year contained six major flow events. Mean daily discharge data were used, converted to daily stage, and used in conjunction with a surveyed bed slope of 0.000088 m m^{-1} for the toe erosion submodel. Geotechnical data were collected in situ. To test for the effectiveness of reducing stream bank loadings, iterative modeling was carried out for (1) existing conditions, no mitigation; (2) rock placement along the bank toe; (3) rock placement at the bank toe and 5 year old woody vegetation on the bank top; and (4) rock placement at the bank toe, 5 year old woody vegetation on the bank top, grading the bank to a 45° (1:1) slope, and 5 year old woody vegetation on the regraded slope.

The selected annual hydrograph(s) for the river were discretized into a series of steady state rectangular-shaped discharge events (Figure 7), where the peak of each rectangular hydrograph was set to 90% of the actual hydrograph peak for each storm event. The reason for this reduction in flow peak for each part of the discretized hydrograph was so that the discretized peaks represented the average value occurring over that time period.

Discharge values for each flow event were converted to a series of flow depths, based on a stage-discharge relation developed for the closest U.S. Geological Survey (USGS) gauge to each site. As water table information was unavailable, it was assumed that water table height equaled the flow height at the peak of each hydrograph. For the critical, drawdown condition, the water table was simulated to be equal to the peak flow elevation. For each flow event, the discretized hydrograph was input into BSTEM as a flow depth and peak duration using the following approach:

1. The effects of the first flow event were simulated using the toe erosion submodel to determine the amount (if any) of hydraulic erosion and the change in geometry in the bank toe region (Plate 2).
2. The new geometry was exported into the bank stability submodel to test for the relative stability of the bank. (1) If the F_s was greater than 1.0, geometry was not updated, and the next flow event was simulated (Plate 3). (2) If F_s was less than 1.0, failure was simulated, and the resulting failure plane became the geometry of the bank for simulation of toe erosion for the next flow event in the series. (3) If the next flow event had a stream stage elevation lower than the previous one, the bank stability submodel was run again using the new lower stream stage elevation and higher groundwater table elevation to test for stability under drawdown conditions. If F_s was less than 1.0, failure was simulated, and the new bank geometry was exported into the toe erosion submodel for the next flow event (Plate 4).
3. The next flow event in the series was simulated.

For each set of conditions, the total number of bank failures and the volume associated with each failure was summed and then compared to the other alternatives to quantify the effectiveness of each treatment. For the initial case with existing bank conditions, 11 failures were simulated, resulting in about 55,000 m^3 of eroded bank sediment. Although the number of bank failures was only reduced by 1 (to 10) for the case with toe protection, the amount of lateral retreat and volume of failed material was drastically reduced (by about 500%) to about 9500 m^3. This was because the toe protection did not allow the bank to be undercut at its base, thereby reducing the size of subsequent failures. The addition of bank top vegetation provided additional cohesive strength to the top 1.0 m of the bank and resulted in a further reduction of failure frequency (to 8) and failure volume (8500 m^3). This effect would probably be more pronounced if older specimens were simulated because of greater root density and diameters. Alternative 4, which included rock at the bank toe, grading the bank slope to 1:1 and placing woody vegetation on the bank top and face, greatly reduced failure frequency (to 3) and showed the smallest failure volume of all cases (about 3200 m^3). Results from each of the alternative strategies are shown in graphic form in Figure 8.

Application of these treatments represents a broad range of options and costs. It is important to recognize, however, that both the absolute frequency and volume of failures likely represents an overestimate of what actually took place during 1991. This is because once failure is simulated, the model does not account for the fate of this material, which may be deposited at the bank toe, providing a buttressing (stabilizing) effect and serving to build up the bank toe region. What is relevant are the relative differences between the exiting case (no mitigation) and the various alternatives.

3.2.2. Lake Tahoe Basin, California and Nevada. In this case study, the project objectives were to determine the type of load reductions that could be realized by applying select mitigation measures to stream banks of several streams in the Lake Tahoe Basin, California and Nevada. Cooperators were interested in reducing the delivery of fine-grained sediment from the three main contributors (Upper Truckee River and Blackwood and Ward Creeks) to Lake Tahoe due to decreasing

Table 9. Iterative Modeling Results for the Upper Truckee River (RKM 13.1) for Existing Conditions With Toe Protection[a]

				SW=GW			Drawdown						
Event	Shear Stress (Pa)	Toe Erosion	Volume (m^3)	FS	Failure	Volume (m^3)	FS	Failure	Volume (m^3)	Shear Emergence (m)	Failure Angle (deg)	Total Volume (m^3)	Total Fines (m^3)
						Existing Conditions (Assuming 100 m Reach)							
1	6.57	yes	0.70	1.22	no	0.00	1.21	no	0.00	1912.03	40	0.70	0.13
2	6.32	yes	8.50	0.95	yes	362			0.00	1911.88	40	371	67.4
3	8.12	yes	1.40	1.56	no	0.00	1.49	no	0.00	1911.91	34	1.40	0.25
4	5.34	yes	0.30	1.47	no	0.00	1.45	no	0.00	1911.88	34	0.30	0.05
5	2.53	yes	0.20	1.29	no	0.00		no	0.00	1911.88	34	0.20	0.04
6	7.08	yes	3.50	0.99	yes	194	1.37	no	0.00	1911.88	44/32	198	35.9
7	6.55	yes	0.50	1.48	no	0.00			0.00	1911.98	32	0.50	0.09
8a	7.89	yes	64.0	0.91	yes	194			0.00	1911.88	46	258	47.0
8b	7.89	yes	8.70	0.97	yes	185	1.29	no	0.00	1911.88	44.5/32	194	35.3
9	6.46	yes	1.10	1.41	no	0.00	1.35	no	0.00	1911.94	34.5	1.00	0.20
10	3.04	no	0.00	1.51	no	0.00	1.49	no	0.00	1911.94	34.5	0.00	0.00
11	3.13	no	0.00	1.50	no	0.00	1.47	no	0.00	1911.94	34.5	0.00	0.00
12	5.18	yes	0.00	1.35	no	0.00	1.28	no	0.00	1911.91	34.5	0.00	0.00
1/1/1997	13.8	yes	1.60	1.03	no	0.00	0.35	yes	262	1911.88	34.5	264	48.0
Totals		12	90.5		3	935		1	262			1288	234
						Toe Protection (Assuming 100 m Reach)							
1	6.57	no	0.00	1.41	no	0.00	1.40	no	0.00	1912.10	40	0.00	0.00
2	6.32	no	0.00	1.44	no	0.00			0.00	1912.10	40	0.00	0.00
3	8.12	no	0.00	1.31	no	0.00	1.25	no	0.00	1912.10	40	0.00	0.00
4	5.34	no	0.00	1.36	no	0.00	1.34	no	0.00	1912.10	40	0.00	0.00
5	2.53	no	0.00	1.38	no	0.00			0.00	1912.10	40	0.00	0.00
6	7.08	no	0.00	1.27	no	0.00	1.19	no	0.00	1912.10	40	0.00	0.00
7	6.55	no	0.00	1.33	no	0.00			0.00	1912.10	40	0.00	0.00
8	7.89	no	0.00	1.26	no	0.00	1.13	no	0.00	1912.10	40	0.00	0.00
9	6.46	no	0.00	1.34	no	0.00	1.30	no	0.00	1912.10	40	0.00	0.00
10	3.04	no	0.00	1.45	no	0.00			0.00	1912.10	40	0.00	0.00
11	3.13	no	0.00	1.44	no	0.00	1.43	no	0.00	1912.10	40	0.00	0.00
12	5.18	no	0.00	1.36	no	0.00	1.32	no	0.00	1912.10	40	0.00	0.00
1/1/1997	13.8	yes	0.10	1.19	no	0.00	0.28	yes	137	1912.10	40	137	25.0
Totals			0.1		0	0.0		1	137			137	25.0

[a]Abbreviations are as follows: FS, factor of safety; SW=GW, surface water level equals groundwater level.

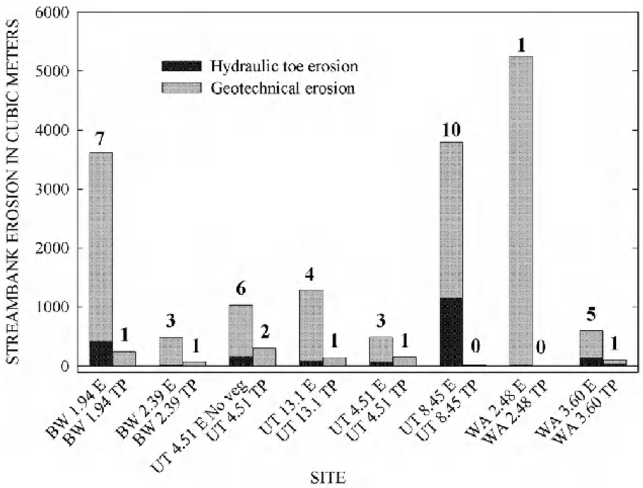

Figure 9. Volume of stream bank erosion under existing (E) conditions and with toe protection (TP) for sites on Blackwood Creek (BW), Upper Truckee River (UT), and Ward Creek (WA). Note the large reduction in total eroded volume for each site by virtually eliminating toe erosion. Bold numbers refer to the number of failure episodes during the 99th percentile flow year. From the work of *Simon et al.* [2009].

lake clarity. A previous study had identified that about 25% of the fine-grained sediment entering the lake emanated from eroding stream banks, with the Upper Truckee River being the largest contributor [*Simon*, 2008]. More than 50% of the suspended-sediment load from the Upper Truckee River and Blackwood Creek is derived from stream bank erosion. A 99th percentile flow year including 12 flow events and a sustained snowmelt period was selected to simulate stream bank erosion with and without various mitigation strategies. In addition, a rain on snow event that occurred on 1 January 1997 and was estimated at about a 50 year recurrence interval was added to the end of the hydrograph. Geotechnical characteristics of the banks and riparian vegetation were determined in situ as part of previous studies [*Simon et al.*, 2003, 2006]. Root reinforcement calculations were made within BSTEM by the RipRoot submodel according to the characteristics shown in Table 8.

Using BSTEM iteratively and under existing conditions at RKM 13.1, a total of 1288 m^3 of material was predicted to be eroded during 12 periods of hydraulic erosion and four mass failure episodes [*Simon et al.*, 2009] (Table 9). Toe erosion represented just 7% of the total bank erosion in the reach. The addition of toe protection virtually eliminated bank steepening by hydraulic erosion at the bank toe, and total bank erosion was reduced by about 89% to 137 m^3 over the same period. Similar results were obtained for all other paired simulations at additional sites (Figure 9) with an average load reduction of 86.8% using toe protection. This result highlights the important relation between hydraulic erosion at the toe that steepens bank slopes and subsequent bank instability. Under existing conditions, toe erosion accounted for an average of 13.6% of the total stream bank erosion, yet control of that process resulted in a total sediment load reduction from bank erosion of almost 87%.

To estimate the total load reduction that could be anticipated for the entire length of each stream, modeled results were combined with observations of the longitudinal extent of recent bank failures along the main stem lengths of each stream. Rapid geomorphic assessments (RGAs) that use diagnostic characteristics of channel form to infer dominant active processes were conducted along each stream as part of earlier research [*Simon et al.*, 2003]. The longitudinal fraction of banks experiencing recent failures was noted for each bank in a reach (6–20 channel widths in length) and expressed as one of five percentage ranges (0%–10%, 11%–25%, 26%–50%, 51%–75%, 76%–100%) (Table 1). The midpoint of the range for each bank (left and right) was used to determine a local mean failure extent. This was then classified as low, moderate, or high in order to apply different unit loads along each stream. Unit loads associated with each class were selected by comparing bank-derived sediment volumes estimated by the numerical simulations with the results of RGAs. For reaches classified as low, a load an order of magnitude lower than the moderate value was used. Unit loads were multiplied by a weighting factor representing the total length of banks (left and right) that had recently failed in a reach to obtain total stream bank-derived sediment loads for the stream. The average extent of bank failures (in percent) was then broken into low, medium, and high groupings to apply different unit loading rates along each stream according to the following procedure. Sediment loads were calculated for each reach by applying the appropriate total loading rate (high or moderate) to those classed as high or moderate. For reaches classified as low, a value an order of magnitude lower than the moderate rate was used. Fine-grained loads for each reach were calculated using the measured percentage of fines (<0.063 mm) for the site. Table 10 shows an example for Blackwood Creek.

To address the cost of potential management scenarios for fine-grained load reduction by toe protection, three options were considered, which included treating all reaches (All), treating only those reaches eroding at high rates (H), and treating only those reaches eroding at high and moderate rates (H+M) (Table 11). A cost for rock placement of $984 m^{-1} was used as the cost basis (obtained from local sources) that was then multiplied by the length of reach represented by each treatment option. The 86.8% average load reduction obtained for all paired simulations was used to determine the

Table 10. Example Calculation of Total Stream Bank Loads From Blackwood Creek[a]

Distance (km)	Extent of Failures (%) Left	Right	Mean	Reach Length (km) 1	Reach Failing (%) 2	Weighting Factor (1)*(2)/100	Total Volume (m^3)	Fraction <0.063 mm (%)	Fines Volume (m^3)
8.29	0–10	0–10	5.0						
8.19	0–10	26–50	21.5[b]	0.10	13.25	0.0133	62.5[b]	5.8	3.6[b]
7.69	11–25	11–25	18.0[c]	0.50	19.75	0.0987	46.6[c]	0.00	0.00[c]
7.18	11–25	11–25	18.0[c]	0.51	18	0.0918	43.3[c]	26.0	11.3[c]
7.17	11–25	76–100	53.0[d]	0.01	35.5	0.0035	128[d]	26.0	33.4[d]
6.84	0–10	11–25	11.5[c]	0.33	32.25	0.1064	50.2[c]	26.6	13.4[c]
6.51	0–10	51–75	34.0[b]	0.33	22.75	0.0751	354[b]	22.1	78.3[b]
6.03	0–10	26–50	21.5[b]	0.48	27.75	0.1332	629[b]	20.0	125.7[b]
5.55	0–10	26–50	21.5[b]	0.48	21.5	0.1032	487[b]	7.9	38.5[b]
5.08	0–10	51–75	34.0[b]	0.47	27.75	0.1304	616[b]	23.5	144.7[b]
4.15	26–50	11–25	25.5[b]	0.93	29.75	0.2767	1306[b]	3.6	47.0[b]
3.95	0–10	76–100	46.5[d]	0.20	36	0.0720	2604[d]	21.4	557.3[d]
2.80	51–75	0–10	34.0[b]	1.15	40.25	0.4629	2185[b]	12.3	268.7[b]
1.97	26–50	11–25	25.5[b]	0.83	29.75	0.2469	1165[b]	24.8	289[b]
1.77	11–25	51–75	40.5[d]	0.20	33	0.0660	2387[d]	16.6	396.3[d]
0.32	51–75	0–10	34.0[b]	1.45	37.25	0.5401	2549[b]	16.3	415.6[b]
0.00	26–50	26–50	38.0[b]	0.32	36	0.1152	544[b]	16.3	88.6[b]

[a]Results of RGAs (columns 2 to 3) permitted a mean percentage of each reach experiencing bank failures to be estimated. The mean value for the percent failing of consecutive reaches was multiplied by the reach length to calculate the weighting factor for each reach. Fine-grained loads were determined by multiplying the fraction of fines in each reach by the estimated total load.

[b]Moderate (4720 m^3 km^{-1}) stream bank–derived unit loads [see *Simon et al.*, 2009].

[c]Low (472 m^3 km^{-1}) stream bank–derived unit loads [see *Simon et al.*, 2009].

[d]High (36,170 m^3 km^{-1}) stream bank–derived unit loads [see *Simon et al.*, 2009].

reduced load for each protected reach. These reduced loads were then summed for each applicable reach to obtain the load (in t) for the entire stream under the three treatment alternatives (All, H, and H+M). These load values were then compared to the existing load (no treatment) to determine the "potential" load reduction for each of the three streams. These ranged from ranged from 33% to 87% depending on the treatment option (length treated). The unit cost (in \$ t^{-1}) of performing this type of rehabilitation similarly varied from \$267 t^{-1} to almost \$2500 t^{-1} (Table 11).

Additional simulations were carried out to quantify the effects of the addition of top bank vegetation and, in one case, a reduction in bed slope. Load reductions of about 53% were simulated for the case on the Upper Truckee River where root reinforcement was provided to the top 1.0 m of the bank. For locations with higher banks, load reductions

Table 11. Loads and Costs for Performing Bank Toe Protection Assuming a Unit Cost of \$984 m^{-1} for Placement of Stone at the Bank Toe[a]

Stream	Loads (t) Existing	Toe Protection All	H+M	H	Total Cost (\$) Toe Protection All	H+M	H	Unit Cost (\$ t^{-1} of Load Reduction) All	H+M	H
Blackwood Creek	4432	585	623	2920	8,159,449	6,840,551	403,543	2,121	1,796	267
Load reduction (%)		86.8	85.9	34.1						
Upper Truckee River	5691	751	914	3789	20,911,417	10,735,138	2,601,378	4,233	2,247	1,368
Load reduction (%)		86.8	83.9	33.4						
Ward Creek	2956	390	451	910	6,358,661	3,120,669	1,731,594	2,478	1,246	846
Load reduction (%)		86.8	84.7	69.2						
Totals	13,079				35,429,527	20,696,358	4,736,515			

[a]H+M refers to reaches designated as high and moderate.

would probably not be as significant because the limited extent of rooting depths would provide a smaller increase in overall bank strength. Load reductions from the flattening of bed slope (by the addition of meanders) and the consequent decrease in boundary shear stresses were 54% for the case of the Upper Truckee River and 42% for Blackwood Creek.

3.2.3. Big Sioux River, South Dakota. Excessive sediment transport in the Big Sioux River, South Dakota, led action agencies to consider erosion-control strategies in this agricultural basin. Before planning and design of mitigation measures could be conducted, however, it was decided to determine the contributions from stream bank and upland sources so that erosion-control measures could be focused on specific areas of high unit loadings. The objectives of this study, therefore, were to determine (1) average, annual rates of stream bank erosion; (2) the contribution of stream bank erosion to total erosion from other sources; and (3) the effects of possible bank mitigation strategies where necessary.

To obtain average, annual stream bank loads, BSTEM was run iteratively for a range of flow years representing the 10th (dry flow year) through the 90th (very wet) percentile flow years. The volumes of sediment eroded during each percentile flow year modeled were then weighted according to their likely frequency of occurrence to calculate suspended sediment load on an average, annual basis. Results were extrapolated spatially for the entire study reach and then compared to suspended-sediment loads calculated from available data at a USGS gauge along the study reach.

Bank stability and toe erosion analyses were carried out using BSTEM, at five study sites along a 300 km reach of the Big Sioux River, South Dakota, for a range of percentile flow years (90th, 75th, 50th, 25th, and 10th) [*Bankhead and Simon*, 2009]. An example of the flow years that were discretized are shown for USGS gauge 06480000 in Figure 10. Model results showed that eroded volumes of sediment emanating from stream banks decreased nonlinearly from the 90th percentile flow year to the 10th percentile flow year. Predicted volumes of sediment eroded at each site ranged from 169 to 1359 m^3 of sediment per 100 m reach during the 90th percentile year, under existing conditions where the banks have a cover of native grasses (Table 12). These volumes of eroded sediment were predicted to fall to 0 to 21 m^3 per 100 m reach during the modeled 10th percentile flow year, again, assuming a cover of native grasses.

Table 12. Unit Load Values per 100 m of Channel for the Control Case of Existing Geometry With Top Bank Grasses

	Percentile of Flow Magnitude[a]				
Site	90	75	50	25	10
Castlewood	473	42	28	2	10
Estelline	169	98	40	17	12
Brookings	972	200	125	13	10
Egan	1359	218	190	32	21
Renner	680	78	25	29	0

[a]Volume eroded in m^3 $(100\ m)^{-1}$.

Although simulations were conducted for the range of flow years, bank failures were generally predicted only during the 90th percentile flow year at each site. Loads simulated during lower-percentile flow years indicated that hydraulic scour at the bank toe was the predominant erosion process, rather than mass wasting of the banks by geotechnical failure. It followed, therefore, that the addition of toe protection (up to 1 m high) to banks with existing native grass cover greatly reduced the volume of bank material eroded at each site. Model runs indicated that even when the contribution to total erosion from toe scour was not that great (e.g., only 16% to

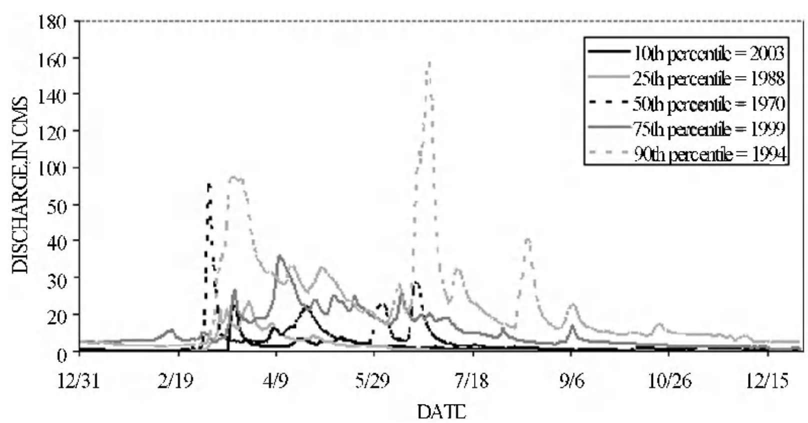

Figure 10. Flow years used for iterative modeling of selected sites on the Big Sioux River, South Dakota. Data are from USGS gauge 06480000. From the work of *Bankhead and Simon* [2009].

Table 13. Example Results of Weighting Values to Produce Average, Annual Stream Bank Loadings Expressed as a Unit Volume and a Unit Mass[a]

	Percentile of Flow Magnitude[b]					Average Annual Loadings		
Site	90	75	50	25	10	Cubic Meters per 100 m	Cubic Meters per Kilometer	Tons per Kilometer
Castlewood	47.3	10.5	14.0	1.5	9.0	82.3	823	14.3
Estelline	16.9	24.5	20.0	12.8	10.8	85.0	850	15.3
Brookings	97.2	50.0	62.5	9.8	9.0	228	2285	40.9
Egan	136	54.4	95.0	24.0	18.9	328	3282	58.1
Renner	68.0	19.5	12.5	21.8	0.0	122	1218	20.6

[a]Weighting values are from Table 12.

[b]Volume eroded in m^3 $(100\ m)^{-1}$.

50% of total erosion came from toe scour during years where bank failures did occur), if the toe scour was prevented, the overall volume of eroded bank material was reduced by 87%–100%.

Average, annual volumes of stream bank sediment emanating from each of the modeled reaches were calculated using weighted values for each percentile flow year (Table 13). These were then converted to $m^3\ km^{-1}$ by multiplying by 10 and then to $t\ km^{-1}$ by multiplying by the bulk unit weight of the material. Average bulk unit weights of the bank material for each site ranged from 16.9 to 18.0 kN m^{-3} [*Bankhead and Simon*, 2009].

Contributions of sediment from stream bank erosion along the study reach of the Big Sioux River were in the range of 10%–25% of the total suspended-sediment load. Average, annual contributions of sediment from stream bank erosion for the entire study reach (6340 t) were shown to be about 15%. During a particularly wet, high-flow year as occurred in 1994, stream bank contributions were consequently greater (27,000 t), comprising 25% of the total suspended-sediment

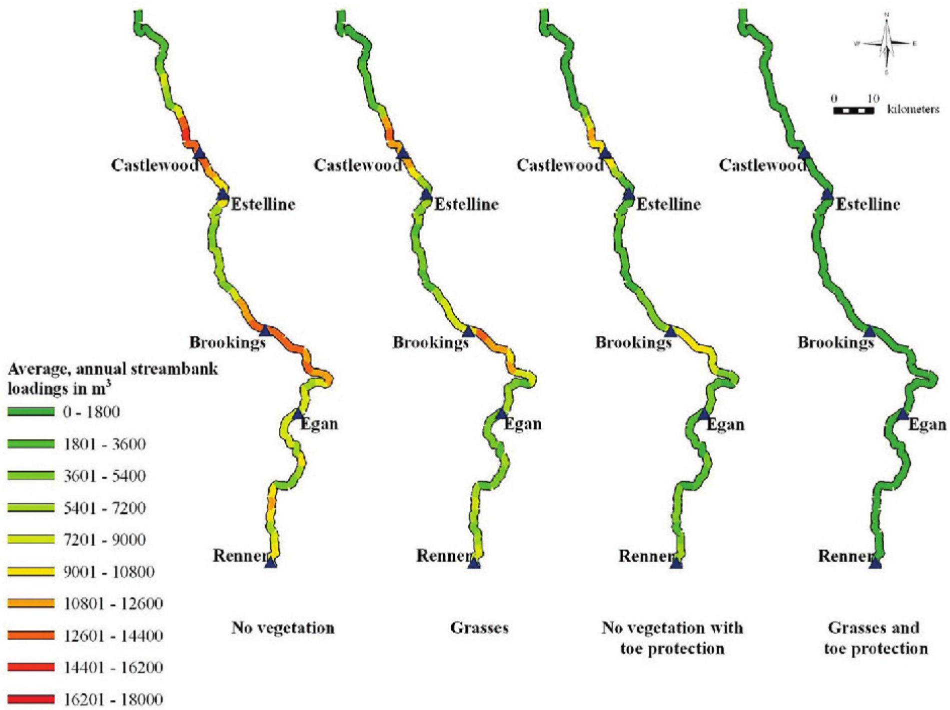

Plate 5. Illustration of spatial distribution of average, annual stream bank loads (m^3) for a range of mitigation strategies and bank conditions along the Big Sioux River, South Dakota. From the work of *Bankhead and Simon* [2009].

load over the 300 km study reach. The data further indicated that stream bank contributions were generally greater in the lower half of the study reach.

The iterative modeling results from the five sites needed to be interpolated and extrapolated over the 300 km study reach to determine total loads and potential load reductions for the mitigation strategies tested. As expected, the bare-bank simulations displayed greater average, annual loadings along the entire study reach, with total loads of 503,000 m^3 (8810 t). The effect of top bank grasses (or an assemblage of grasses and young cottonwood trees) resulted in a reduction of average, annual stream bank loads of 28% (to 362,000 m^3 or 6340 t); 20% for the 90th percentile flow. The addition of bank toe protection to the grassed bank resulted in a total reduction in average, annual loads (from the bare-bank case) of 97% (to 15,200 m^3 or 267 t). The important role of toe protection was further apparent by comparing the difference in stream bank loads between the bare-bank case and the mitigation strategy that incorporated toe protection alone. Here, average, annual stream bank loads were reduced 51% from 503,000 m^3 (8810 t) to 243,000 m^3 (4250 t); 84% for the 90th percentile flow.

Results of potential mitigation strategies can also be displayed spatially. Maps, such as those shown in Plate 5, can be used to focus stream restoration practitioners to those reaches that are the most problematic and to identify the magnitude of the potential load reductions that could be expected for a given reach and mitigation strategy. For example, in some of the yellow reaches in the map with no vegetation or mitigation, addition of grasses to the bank top may provide enough stability to reduce sediment loading to the required level in that particular reach. In the reaches that are shown in red in the first map, addition of vegetation may not be sufficient, with those reaches also requiring further mitigation, such as rock toe protection, to reduce bank erosion and resulting suspended sediment loadings.

4. CONCLUSIONS

The BSTEM is a simple spreadsheet tool developed to simulate stream bank erosion in a completely mechanistic framework. It has been successfully used in a range of alluvial environments in both static mode to simulate bank stability conditions and design of stream bank stabilization measures and, iteratively, over a series of hydrographs to evaluate surficial, hydraulic erosion, bank failure frequency, and thus, the volume of sediment eroded from a bank over a given period of time. In combination with the submodel RipRoot, the reinforcing effects of riparian vegetation can be quantified and included in analysis of mitigation strategies. In addition, the model has been shown to be very useful in testing the effect of potential mitigation measures that might be used to reduce the frequency of bank instability and decrease sediment loads emanating from stream banks. Finally, the results of iterative BSTEM analysis can be used to spatially extrapolate bank-derived volumes of sediment, from individual sites to entire reaches when used in conjunction with RGAs conducted at regular intervals along the study reach. Results of these case studies have shown that the relative contribution of suspended sediment from stream banks can vary considerably, ranging from as low as 10% in the predominantly low-gradient, agricultural watershed of the Big Sioux River, South Dakota, to more than 50% for two steep, forested watersheds of the Lake Tahoe Basin, California. Modeling of stream bank mitigation strategies has also shown that the addition of toe protection to eroding stream banks can reduce overall volumes of eroded sediment up to 85%–100%, notwithstanding that hydraulic erosion of the toe in this particular case makes up only 15%–20% of total bank erosion. Vegetation provides a stabilizing effect to the modeled stream banks, but sufficient time must be factored into any restoration design involving vegetation as a mitigation measure, to allow sufficient development of root networks.

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Bank Vegetation, Bank Strength, and Application of the University of British Columbia Regime Model to Stream Restoration

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The University of British Columbia Regime Model (UBCRM) is based on rational regime theory. A feature of the model is that it quantifies the effect of bank vegetation and its effect on channel geometry. Three bank vegetation models can be applied to gravel bed rivers with either noncohesive, cohesive, or composite banks. Simplified dimensionless equations for width and slope derived using the UBCRM are applied to a site on the Coldwater River, British Columbia. Between 1953 and 2003, there were significant land use changes that included riparian and floodplain clearing. The observed widening and steepening can be explained by a reduction in bank strength and that changes in the sediment load, discharge, or grain size do not appear to be significant. Applied correctly, the UBCRM can provide qualitative and quantitative insight into the primary causes of historic disturbance and can serve as an aid in restoration design. Because of the physically based nature of the parameters in the UBCRM, analysis and design are directly linked to fluvial processes including flow resistance, sediment transport, and bank stability.

1. INTRODUCTION

Bank or riparian vegetation plays multiple important roles in stream function including bank strength [*Smith*, 1976; *Wynn and Mostaghimi*, 2006; *Simon et al.*, 2006], flow resistance [*Masterman and Thorne*, 1992; *Darby*, 1999; *Hirschowitz and James*, 2009] shading and reduced temperatures [*Theurer et al.*, 1985; *Dewalle*, 2008], and in-stream habitat [*Beschta*, 1991; *Jowett et al.*, 2009], among others. This chapter deals only with the issue of bank stability and the influence on channel geometry, primarily the reach average channel width and slope. An overview of the background for The University of British Columbia Regime Model (UBCRM) is given, together with its potential application, both qualitative and quantitative, to the assessment and restoration design alluvial gravel rivers and streams, with particular emphasis on vegetation and bank strength.

The UBCRM [*Millar and Quick*, 1993; *Eaton et al.*, 2004] is based on rational regime theory. The primary assumption is that alluvial rivers tend to an equilibrium state, which is commonly referred to as being "in regime." A river in regime is considered stable, not necessarily in the sense that the bed and banks are nonerodible and fixed with time but in the sense that average channel dimensions are maintained over a period of years. Regime represents a dynamic equilibrium in which average channel dimensions remain more or less constant over time, despite ongoing bed and bank erosion

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and deposition, meander cutoffs, and lateral migration. The rational regime approach used in the UBCRM can serve as a useful framework from which to interpret channel change and to understand and quantify the controlling variables and as a basis for determining reach-averaged channel dimensions for river restoration of alluvial gravel rivers.

The UBCRM has been developed over a number of years in collaboration between researchers in the Department of Civil Engineering and the Department of Geography at the University of British Columbia. The model is based on the understanding that a simple model with modest data requirements is more likely to be useful than a data-intensive, numerically demanding one, especially for environmental practitioners. While simple, the model does consider the relevant controlling factors, the most important of which is the nature and erodibility of the channel banks. The goal of our research on this topic is to determine which simplifying assumptions about river channel behavior are reasonable to make and to identify the underlying physical processes.

Rational regime theories relating stream channel conditions to the external driving forces have a long history [*Yang*, 1976; *Chang*, 1979; *Ferguson*, 1986; *Kirkby*, 1977; *White et al.*, 1982; *Davies and Sutherland*, 1983]. There are two main impediments to the general acceptance of rational regime models: the first is the development of a scientifically reasonable understanding of the extremal hypotheses used in the models and the second is the incorporation of a bank stability analysis in the model. Researchers at the University of British Columbia (including M. Church, B. Eaton, R. Millar, and M. Quick) have made significant progress on these two issues. The extremal hypotheses have been reformulated in such a way as to make the underlying principle more easily understood [*Eaton et al.*, 2004; *Millar*, 2005]. The UBCRM has been tested against observed channel adjustments in the laboratory and in the field and observed behavior that is consistent with our generalized extremal hypothesis [*Eaton and Church*, 2004; *Eaton and Millar*, 2004; *Eaton and Church*, 2007]. Various bank strength formulations have been incorporated into the UBCRM, which results in a general agreement between model predictions and observed channel dimensions, overcoming the long-standing criticism that regime models consistently underpredict channel width [*Millar and Quick*, 1993, 1998; *Millar*, 2005; *Eaton*, 2006]. The UBCRM is gaining greater recognition, and the paper by *Eaton et al.* [2004] was awarded the Wiley Award by the British Geomorphological Research Group for best paper in the journal *Earth Surface Processes and Landforms* for 2005. It is now being tested by various researchers and environmental consultants in British Columbia who are looking for practical tools for making better decisions about stream channel management. There are numerous potential applications for this model, including the replacement of purely empirical hydraulic geometry relations in channel design and restoration. A feature of the UBCRM is that it quantifies the effect of bank vegetation and the effect on channel geometry using a variety of bank vegetation models.

2. BANK VEGETATION MODELS

There are three basic models that are used in the UBCRM to quantify the effect of bank vegetation on bank stability: (1) the noncohesive ϕ' model [*Millar and Quick*, 1993], (2) the cohesive τ_{crit} model [*Millar and Quick*, 1998], (3) and the composite bank H_{max} model [*Eaton*, 2006]. These are described below.

2.1. Noncohesive ϕ' Model

The noncohesive ϕ' model returns the critical bank shear stress necessary for fluvial erosion of essentially noncohesive gravel bank sediment. It is based on the U.S. Bureau of Reclamation algorithm [*Lane*, 1955] and accounts for bank vegetation effects by increasing the effective angle of repose, ϕ':

$$\frac{\tau_{c\ \mathrm{bank}}}{\rho g(s-1)d_{50}} = 0.048\tan\phi'\sqrt{1-\frac{\sin^2\theta}{\sin^2\phi'}}, \qquad (1)$$

where $\tau_{c\ \mathrm{bank}}$ is the critical shear stress for the bank sediment (Pa); ρ is the density of water (1000 kg m^{-3}), s is specific gravity of bank sediment (2.65 assumed), g is gravitational acceleration (9.8 m s^{-2}), d_{50} is the median grain diameter of the gravel bank sediment (m), and θ is the bank angle (°) measured from the horizontal. This algorithm is derived by considering a force balance acting on gravel particles in the bank surface and is relevant only in terms of fluvial erosion or detachment of individual bank particles. It does not consider mass failure processes, which are important for fine-grained, cohesive banks [*Darby and Thorne*, 1996; *Simon and Collison*, 2002].

For loose, noncohesive gravel with no significant vegetation effects (Figure 1a), ϕ' takes a value equal to the angle of repose of the sediment, or 30° to 40°, depending on grain diameter and angularity. Where strengthening by vegetation is significant (Figure 1b), ϕ' takes higher values. The maximum value of ϕ' = 90° represents nonerodible banks. In general, as bank vegetation density increases, higher values of ϕ' are obtained, which correspond to narrower and deeper channels [*Millar and Quick*, 1993]. Alternatively, when bank vegetation is removed, this generally is reflected in lower values of ϕ', weaker banks, and a wider and shallower

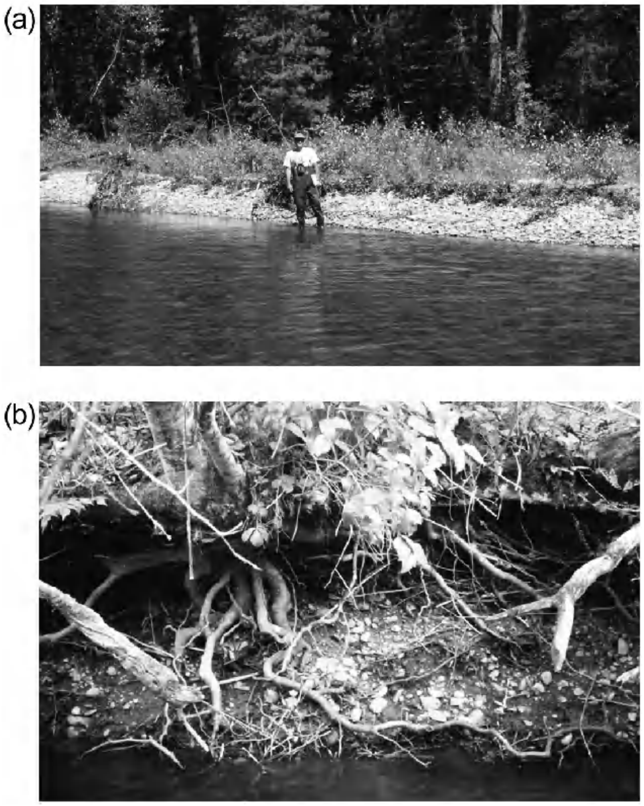

Figure 1. (a) Loose, noncohesive gravel banks with no significant vegetation effects, West Kettle River, British Columbia. (b) Binding and strengthening effect of bank vegetation, Keogh River, British Columbia.

channel. *Millar* [2000] gives an example where following logging of the riparian forest, a river increased in width by a factor of about five times and was transformed from a single-thread to a multiple-thread braided river [*Millar*, 2000]. The value for ϕ' cannot be measured directly but must be obtained from model calibration or estimated from experience. It is also subject to scale effects, where different bank heights with the same vegetation can exhibit a markedly different value of ϕ' [*Eaton and Millar*, 2004; *Eaton*, 2006].

The noncohesive model can also be expressed in terms of relative bank strength, μ' [*Millar*, 2005]:

$$\mu' = \tau_{c\ \mathrm{bank}}/\tau_{c\mathrm{bed}} = \tan\phi'/\tan\phi, \tag{2}$$

where ϕ = the angle of repose of the bed sediment. A value of μ' = 1 indicates that the bed and banks are equally erodible (Figure 1a). Banks stabilized by vegetation take values of μ' greater than one (Figure 1b). For instance, a value of μ' = 2 indicates that bank strength, or more correctly the critical bank shear stress, is twice that of the bed material.

2.2. Cohesive τ_{crit} Model

The cohesive τ_{crit} model applies to fluvial erosion of cohesive sand/silt/clay banks. The requirement for bank stability is

$$\tau_{\mathrm{bank}} \leq \tau_{c\ \mathrm{bank}}, \tag{3}$$

where $\tau_{c\ \mathrm{bank}}$ is the critical shear stress for fluvial erosion of the cohesive bank sediment. $\tau_{c\ \mathrm{bank}}$ is a function of clay, silt, and sand content and may also be influenced by bank vegetation and roots. The value of $\tau_{c\ \mathrm{bank}}$ can potentially be measured directly in situ [e.g., *Hanson and Simon*, 2001]. A major difficulty is relating local or point $\tau_{c\ \mathrm{bank}}$ measurements to the reach scale and to incorporate heterogeneous vegetation effects. Alternatively, reach-averaged estimates can be obtained through model calibration in a manner similar to obtaining estimates of ϕ', that is, by valuing the input value of $\tau_{c\ \mathrm{bank}}$ until the computed and observed channel dimensions match. *Millar and Quick* [1998] showed that rivers with vegetated banks take higher values of $\tau_{c\ \mathrm{bank}}$ and are narrower and deeper than unvegetated counterparts.

2.3. Composite Bank H_{max} Model

A new bank stability was proposed by *Eaton* [2006] that applied to composite banks with a lower noncohesive gravel base (ϕ' = 40°), overlain by a cohesive silt/clay layer (Figure 2). In this model, the vegetation effects are limited to the upper cohesive unit. Increased bank vegetation and root reinforcement increases the effective root cohesion, c', which increases H_{max}, the maximum stable height of the cohesive layer. The stability of the cohesive layer is modeled using a simple slab-failure analysis and can include pore water pressure effects. Stability of the lower gravel layer is assessed using the noncohesive ϕ' model. Unlike the noncohesive ϕ', the composite bank model does not appear to suffer from the same scaling effects. Furthermore, the observed H_{max} can be measured in the field and used in place of c' for simulation, making this model relatively straight forward to apply to design calculations.

3. UNIVERSITY OF BRITISH COLUMBIA REGIME MODEL

The basis of the UBCRM is that an equilibrium river channel will tend to adjust its reach-averaged width and slope over time to an optimum configuration, which can be defined as the maximum sediment transport efficiency [*Millar*, 2005]:

$$\eta = G_b/\rho QS, \quad (4)$$

where G_b is bed material transport rate at the formative discharge (kg s^{-1}), ρ is the density of water (kg m^{-3}), Q is formative discharge (m^3 s^{-1}) and assumed here to be the morphological bankfull discharge, S is the reach-averaged channel gradient (which is presumed representative of the energy slope), and ρQS is the total stream power in mass units (kg s^{-1}). Note that maximization of η is equivalent to maximizing G_b under conditions of constant ρQS [*White et al.*, 1982] and is equivalent to minimization of S or ρQS under condition of constant G_b [*Chang*, 1979, 1980].

Because $G_b = \rho QC$, where C is the sediment concentration (dimensionless), equation (4) can be further simplified to provide an alternate definition of efficiency:

$$\eta = C/S. \quad (5)$$

Thus, for a specified C, the optimum or maximum efficiency corresponds to a minimization of S. The assumption that alluvial rivers adjust to an optimum state defined by equation (5) is an extension and restatement of earlier work by *Yang* [1976], *Yang and Song* [1979], *Yang et al.* [1981], *Chang*, [1979, 1980], *White et al.* [1982], *Huang et al.* [2002, 2004], and others. Essentially, the UBCRM and similar extremal-based models, assume that over time, an alluvial river will tend to adjust to an optimum condition, which can be defined by a number of criteria, including equation (5). *Eaton et al.* [2004] provide a full discussion of this background theory.

Evidence for an optimum was first demonstrated by *Gilbert* [1914], who performed a series of experiments in a rigid-walled (nonerodible) flume in which the width was varied. Gilbert demonstrated the existence of both an optimum channel width (W), in which C is maximized and also a minimum S under conditions of constant C. The existence of a minimum-slope optimum can be readily simulated by simultaneously solving equations for flow resistance, continuity, and a sediment transport relation. This system of equations is indeterminate with infinite possible solutions because more unknown dependent variables exist than equations available for solution. Solving for a range of trial W

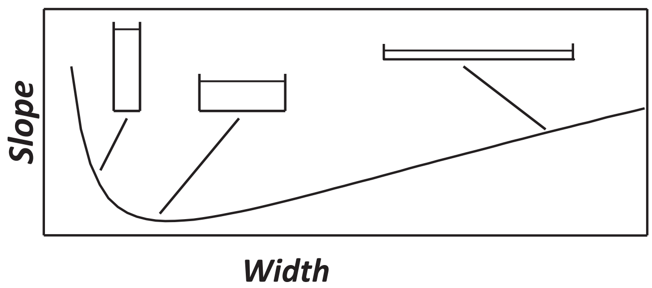

Figure 3. A single solution curve corresponding to all possible combinations of W and S for specified values for Q, C, and bank strength. The optimum width corresponds to the minimum slope.

Figure 2. Example of a composite bank from Carmanah Creek, British Columbia. A vertical cohesive layer overlies a lower, non-cohesive gravel/cobble layer. Vegetation effects are limited to the upper, cohesive layer. The thickness of the upper layer is H_{max}.

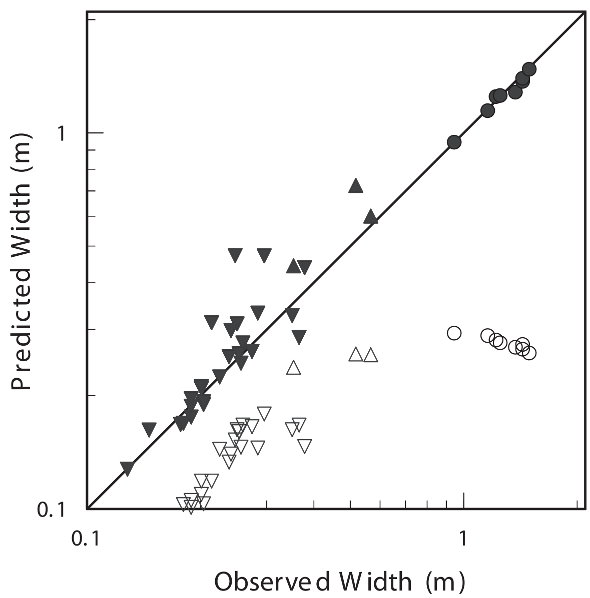

Figure 4. Calculated channel widths for straight (triangles) and meandering laboratory channels (circles). Data are from *Wolman and Brush* [1961] and *Schumm and Khan* [1972]. The open symbols are calculations performed with no consideration of bank strength. The solid symbols correspond to The University of British Columbia Regime Model (UBCRM) using $\phi' = 30°$. Note the improved prediction, particularly for the meandering channels. After *Eaton and Millar* [2004].

values with constant bankfull discharge (Q) and sediment concentration (C) yields a single solution curve (Figure 3) that is characterized by a distinctive broad and asymmetric optimum: the minimum S. *Gilbert* [1914, p. 67] presents this same curve, which he derived experimentally.

The fundamental and important difference between our work and others is that the UBCRM explicitly includes the bank stability as a constraint. The importance of this was demonstrated by *Eaton and Millar* [2004], who calculated the width of laboratory scale, self-formed alluvial channels and showed that good prediction of channel width was only possible if the bank stability was accounted for. A version that ignored the bank stability predicted channels that were about half of the observed width (Figure 4).

When bank strength is varied, a family of solution curves is produced where the optimum W becomes wider and the optimum S becomes steeper as the bank strength (represented by μ') is reduced (Figure 5). A similar family of solution curves with wider and steeper optima can also be derived by holding bank strength constant and increasing sediment concentration, C (Figure 5). Changes in the value of any one of the independent variables, Q, C, d_{50} or bank strength will result in a new solution curve with a new optimum or regime geometry. This is an important consequence of this theory as the hydrology, sediment supply, and bank stability at a restoration site may be substantially different compared to historic conditions. The UBCRM can therefore potentially be used to determine the reach-averaged channel dimensions based on historic, current, or even future catchment and channel conditions.

The UBCRM is usually implemented as a numerical model (B. C. Eaton, The University of British Columbia user's manual, unpublished report, 35 pages, University of British Columbia, Vancouver, British Columbia, Canada, 2007). However, for illustrative purposes, simplified dimensionless

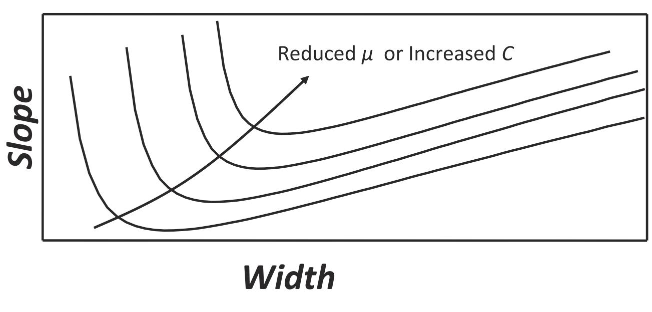

Figure 5. A family of theoretical solution curves for different combinations of Q, C, and bank strength. Each curve has one optimal solution (the minimum slope). Wider and steeper optima are generally associated with reduced bank strength and/or an increase in sediment load.

equations for W and S generated using the UBCRM [*Millar*, 2005] are used here:

$$W^* = 28.1 Q^{*0.50} C^{*-1.12} \mu'^{-1.66} \tag{6a}$$

$$S = 1.98 Q^{*-0.33} C^{*-1.86} \mu'^{-0.93}, \tag{6b}$$

where $W^* = W/d_{50}$, $Q* = Q/(d_{50}^2\sqrt{gd_{50}(s-1)})$, and $C^* = -\log_{10} C$. Note that the Q^* exponents derived theoretically in the W and S equations agree well with empirical counterparts [*Hey and Thorne*, 1986] or 0.5 and –0.33, respectively. Similar dimensionless equations were also developed by *Millar* [2005] for the depth D and the width/depth ratio W/D:

$$D^* = 0.0764 Q^{*0.37} C^{*1.16} \mu'^{1.22} \tag{6c}$$

$$W/D = 425 Q^{*0.12} C^{*-2.30} \mu'^{-2.90}, \tag{6d}$$

where the dimensionless depth, $D^* = D/d_{50}$. Similarly, the Q^* exponent in equation (6c) also agrees well with empirically derived values [*Hey and Thorne*, 1986].

Equations (6a) through (6d) reveal the dependence of channel geometry on the primary independent variable describing discharge, sediment load, and bank strength. Note that a change in the value of any one of the independent variables will result in a new optimal solution, which corresponds to new values for W, S, D, and W/D. This chapter will focus on application of the equations for W and S (equations (6a) and (6b)).

4. CONTRAST WITH EMPIRICAL REGIME EQUATIONS

Equation (6a) can be expressed in a dimensional form:

$$W = aQ^{0.5}, \tag{7}$$

where the coefficient $a = 28.1 C^{*-1.12} \mu'^{-1.66} / (d_{50} g(s-1))^{0.25}$. This indicates that for given values of C^*, μ', and d_{50}, the bankfull channel width will scale with $Q^{0.5}$ but that different values for a can be expected between different reaches and river systems due to differences in the sediment load, bank strength, and or grain size. This is supported by numerous empirical studies that have found relationships similar to equations (7), with an exponent value of 0.5 or close to it (see, for example, *Hey and Thorne* [1986] for a summary). The value of the empirical coefficient a can range from about 2 to 5, and this variation has been attributed to differences in sediment load [*Simons and Albertson*, 1963; *Hey and Thorne*, 1986], bank material type [*Simons and Albertson*, 1963; *Charlton et al.*, 1978], bank silt and clay content [*Ferguson*, 1973], bank vegetation [*Charlton et al.*, 1978, *Andrews*, 1984; *Hey and Thorne*, 1986; *Huang and Nanson*, 1997], and grain size [*Kellerhals*, 1967; *Bray*, 1982].

Empirical regime width equations similar to equation (7) have been used extensively for the design of constructed channels and canals [*Bray*, 1982]. However, because of the variation in a, application to a specific design or restoration project generally requires a survey of adjacent rivers with the same hydrogeomorphic region in order to establish the relationship between W and Q, or alternatively between W and catchment area [*Newbury and Gaboury*, 1994; *Annable*, 1996]. In these empirical studies, a has no obvious physical basis, but it is simply a parameter that is derived from regression analysis. The value of the empirical coefficient a for a region represents the "average" value for the rivers used in the particular study. Individual rivers or reaches within a region can diverge greatly from the average value, as evidenced by the typical scatter in the data about the regression line in empirical studies. Thus, regionally derived averages may not be appropriate for any one specific reach if the conditions at that reach, particularly sediment load or bank strength, are substantially different than the regional average values. In contrast, the parameters in the UBCRM can be determined for a specific reach, have a more physical basis, and are directly related to controlling physical processes. This is demonstrated using the case study below.

5. APPLICATION OF THE UBCRM TO THE COLDWATER RIVER, BRITISH COLUMBIA

Application of the UBCRM is demonstrated by applying equations (6a) and (6b) to a reach on the Coldwater River in southwestern British Columbia. The study reach is located at 49°59′15″N 120°55′25″W, approximately 27 km upstream of the confluence with the Nicola River at Merritt, British Columbia. Along this section of the Coldwater River, the valley flat has been developed for agricultural and cattle grazing. This has resulted in extensive floodplain clearing, grazed woody riparian vegetation and trampled stream banks. Geomorphic and hydrologic data for the reach are from *Bailey* [2003].

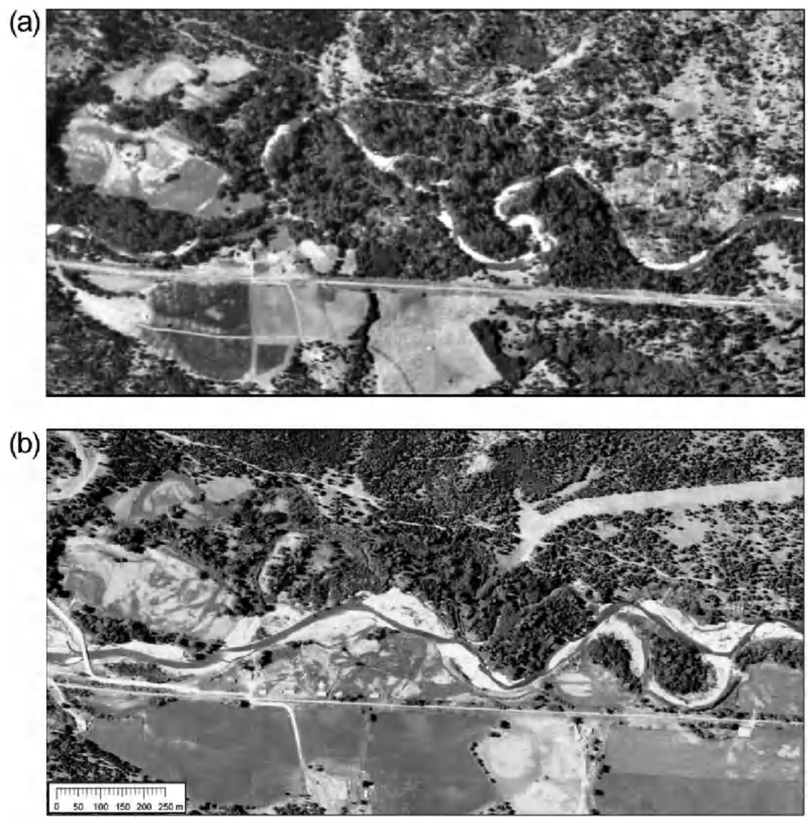

Figure 6. Coldwater River, British Columbia: (a) 1953 and (b) 2003. Flow direction is from left to right. British Columbia Government Aerial Photography: BC 1745:92 and BCC00036:170. From *Bailey et al.* [2005], reprinted with permission.

Table 1. Summary of Observed and Calculated Variables for the Coldwater Study Reach[a]

Variable	1953	2003
	Observed	
W(m)	26	62
S	0.0032	0.0049
Sinuosity	1.7	1.1
	Calculated	
W(m)	26	62
S	0.0032	0.0052
μ'	2.12	1.26
C^*	2.83	2.83

[a]For both 1953 and 2003, values of $Q = 78$ m^3/s and $d_{50} = 0.033$ m were used [after *Bailey*, 2003].

Air photos are presented for 1953 and 2003 (Figures 6a and 6b). In 1953, despite road construction and floodplain clearing, the channel appears relatively intact, with dense riparian vegetation. Measurements from the 1953 air photo [*Bailey*, 2003] indicate an average width of 26 m and a sinuosity of about 1.7. By 2003, extensive clearing for agriculture had been completed with significant loss of the riparian vegetation. The channel had widened by some 150% to 62 m and straightened, with a reduction in sinuosity to about 1.1, largely as a consequence of meander cutoffs. Field measurements by *Bailey* [2003] give an average channel slope of 0.0049 and a median grain diameter for the gravel banks of 0.033 m. Analysis of the annual maximum daily discharge data for the Coldwater River gauge near Merritt (08LG010) for 1961–1994 and corrected for catchment area, give a mean annual maximum daily discharge of 78 m^3 s^{-1} for the reach [*Bailey*, 2003], which is assumed to represent the channel-forming discharge, Q. Relevant variable and parameter values are summarized in Table 1.

6. INTERPRETING HISTORIC CHANGES

The UBCRM, in the form of the simplified equations for W^* and S (equations 6a and 6b), is applied here initially to first qualitatively interpret the changed conditions along the Coldwater River since 1953 and then used in a more quantitative manner. Inspection of equations (6a) and (6b) indicate that a reduction in either μ' or C^* would result in a wider and steeper channel, consistent with the observed changes. This corresponds to a decrease in bank strength and/or an increase in the sediment load. (Note that a reduction in dimensionless C^* by definition means an increase in the dimensional sediment concentration C.) An increase in Q^* would result in a wider, though less steep and more sinuous channel, which is not consistent with the observed changes. At first glance, it would therefore appear that reduced bank strength and/or an increase in sediment load may represent the primary causes of the change.

In order to quantify the changes in parameter values, equations (6a) and (6b) were adjusted to fit the 1953 channel geometry. The values of Q and d_{50} are assumed to have remained constant between 1953 and 2003, at 78 m^3 s^{-1} and 0.033 m, respectively. Trial values of μ' and C^* were input in a stepwise fashion until convergence was achieved, that is, until the computed W and S values became equal to the values obtained from air photos (Figure 6) or from field measurement or hydrologic analysis [*Bailey*, 2003]. All experience to date indicates that for given values of Q and d_{50}, there is a single combination of μ' and C^* that will provide a match for the observed W and S. In this example, the values obtained for μ' and C^* corresponding to the 1953 geometry are 2.12 and 2.83, respectively (Table 1). The value of $\mu' = 2.12$ indicates that the gravel banks are over twice as resistant to fluvial erosion than would be expected from purely noncohesive gravel, and this value is consistent with well-vegetated riparian zones.

7. CHANGE IN BANK STRENGTH OR SEDIMENT LOAD?

The extent of floodplain clearing and riparian disturbance between 1953 and 2003 suggest that reduced bank strength represents a likely factor in the observed channel changes. Holding all other parameters constant and reducing only the value of μ' from 2.12 to about 1.26 in equation (6a) replicates the observed increase in width from 26 to 62 m (Figure 7a). Looking at S, which must also adjust if W is changing, equation (6b) predicts an increase in the regime slope from 0.0032 to 0.0052 as the channel widens (Figure 7). This predicted value of S compares quite favorably with the observed channel slope of 0.0049, measured in the field by *Bailey* [2003]. Thus, the observed changes from 1953 to 2003 can be simulated reasonably well by reducing only the bank strength, and they indicate that the sediment-transporting capacities of the 1953 and 2003 channels, despite their morphologic differences, remain quite similar. The modest difference (6%) between the observed and calculated 2003 channel slope could be accounted for by a slight reduction in C. However, given the imprecision inherent in both the field measurements of channel slope and the estimation of sediment transport capacity, this 6% difference is not considered significant. Ideally, a postdisturbance value of $\mu' = 1$ would be expected for complete removal of bank vegetation effects. The value of $\mu' = 1.26$ in this example could indicate either some residual vegetation effects, the presence of some interstitial cohesive sediments, or perhaps indicate that the value of d_{50} used in the analysis is not entirely representative.

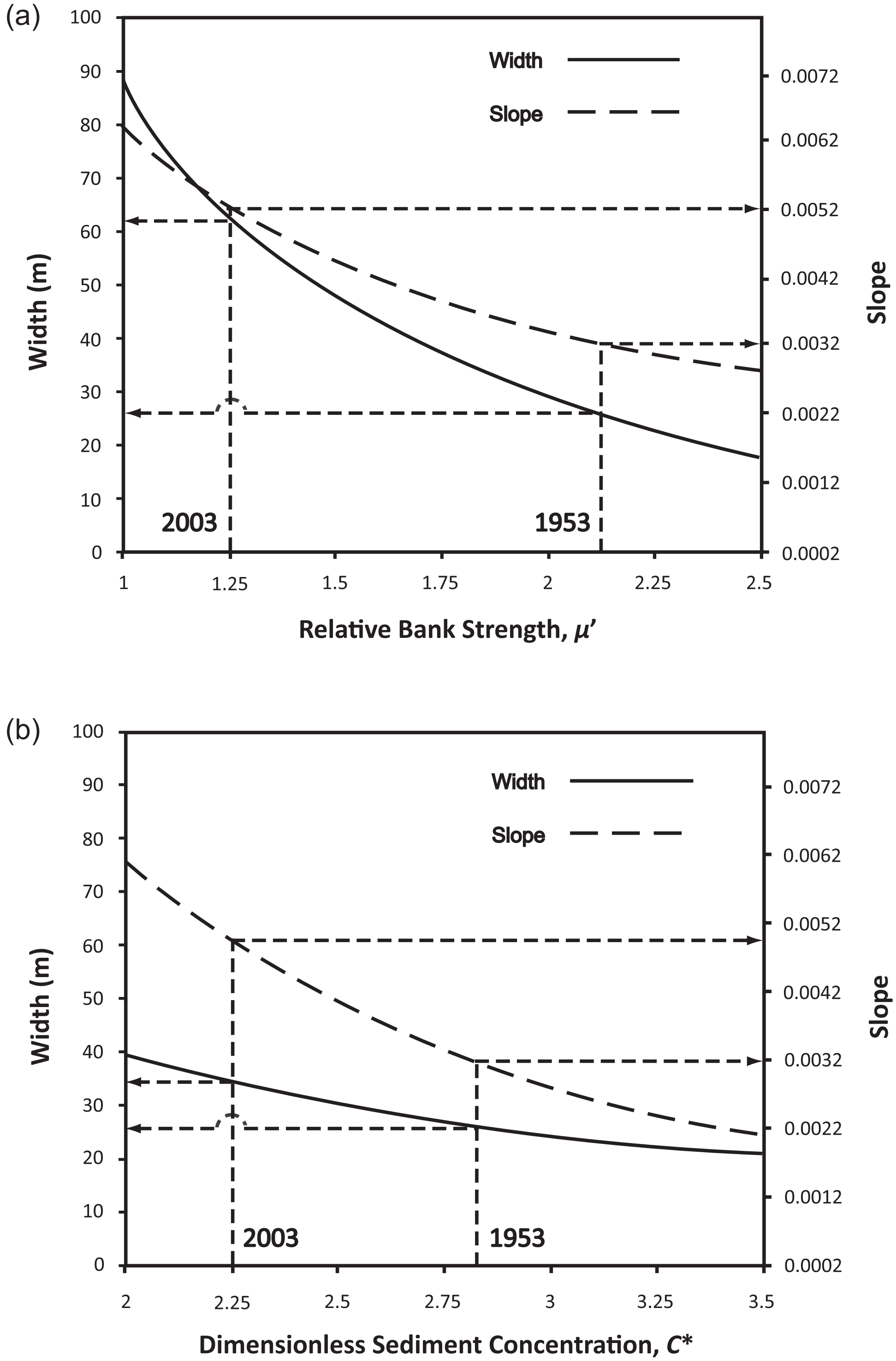

Figure 7. (a) Variation of W and S with relative bank strength μ' for the Coldwater River calculated using equations (6a) and (6b). Values of $Q = 78$ m^3 s^{-1}, $d_{50} = 0.033$ m, and $C^* = 2.83$ were held constant. The calculated 1953 dimensions are indicated for $\mu' = 1.26$, the 2003 dimensions are indicated for $\mu' = 2.12$, and they agree closely with the observed values (Table 1). (b) Variation of W and S with relative bank strength C^* for the Coldwater River calculated using equations (6a) and (6b). Values of $Q = 78$ m^3 s^{-1}, $d_{50} = 0.033$ m, and $\mu' = 2.83$ were held constant. The change in S is well replicated by reducing C^* from 2.83 to 2.25. However, note that W is relatively insensitive to changes in C^*, and the observed increase in W is not reproducible by changing C^*.

In contrast, holding the bank strength constant at the 1953 value of $\mu' = 2.12$ and changing C^* alone is not sufficient to reproduce the observed changes (Figure 7b). Compared to S, W is relatively insensitive to changes in C^* alone. Reducing C^* from 2.83 to 2.25 reproduces the observed increase in slope from 0.0032 to 0.0049 but only increases the bankfull width from 26 to 34 m, much less than the observed increase to 62 m by 2003. Without significant reduction in μ', there is no change in C (or Q or d_{50} for that matter) that can replicate the observed widening and steepening of the Coldwater River. Therefore, it is concluded that the principal impact to the reach is removal and disturbance of riparian and floodplain vegetation, which, in turn, produced a substantial weakening of the banks and a reduction in the ability to withstand fluvial erosion. This is consistent with the observed land use changes that have occurred since 1953, with an obvious reduction in the riparian vegetation (Figure 6).

It has been assumed that the values of Q and d_{50} have remained constant between 1953 and 2003. Given the land use changes, there is certainly scope for significant change. However, we have no data on which to base any analysis. Given that the observed channel changes between 1953 and 2003 could be well simulated by changing only the bank strength suggests strongly that any changes in Q and d_{50} are of second-order importance.

8. APPLICATION TO RESTORATION

The historical photo evidence (Figure 6) and the preceding analysis with the simplified UBCRM equations (6a) and (6b) suggest that the observed widening, steeping, and reduced sinuosity of the Coldwater River is most likely due to destruction and removal of the riparian vegetation and consequent reduction in the bank strength. There is no need to significantly change the values of C^*, Q, or d_{50} in order to simulate the observed widening and steepening at the study site. This does not preclude changes in these variables since 1953, only that their effect appears secondary in magnitude. Clearly, channel restoration would focus on bank vegetation and bank strength.

From a channel geometry perspective, restoration to the original (1953) narrower and more sinuous geometry would require reestablishment of the higher bank strength that existed prior to disturbance. The current channel represents that regime geometry that corresponds to the existing, weaker bank conditions. In their current state, these banks could not withstand the higher shear stresses that would develop in a narrower, deeper, and more sinuous channel. Thus, if the goal were to reestablish the 1953 geometry and morphology and to re-access the meander cutoffs, bank strengthening would be necessary. In the short term, this could only be achieved through "engineered" channel construction using rip rap or other bank revetments to provide the necessary bank strength. In this scenario, the riparian vegetation would not be necessary to provide bank strength, but as it established, it would provide for the many other ecological benefits.

Alternatively, longer-term restoration could focus on reestablishing a functioning and effective riparian forest. This could involve planting appropriate species and perhaps fencing to prevent livestock disturbance. Depending upon the forest species that were used, it could conceivably take years or even decades for the forest and root systems to develop to the point where they produced significant strengthening of the banks. Selective use of rip rap, LWD, or logjams [*Abbe and Montgomery*, 2003] or other structural approaches could be included to protect more vulnerable areas form erosion to allow vegetation to re-establish.

9. UNCERTAINTY IN THE PREDICTIONS

Application of the UBCRM assumes that the river is or will ultimately adjust to an equilibrium condition defined by the maximum sediment transport efficiency (equation (5)). This optimum solution depends on several parameters related to discharge, bank strength, grain size, and sediment transport capacity, all of which can be difficult to quantify. Estimates of

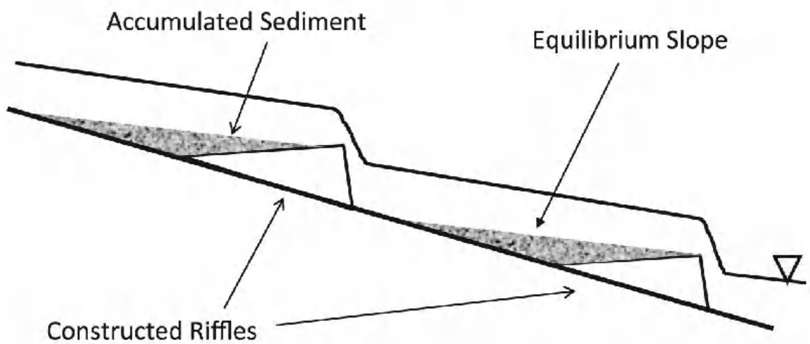

Figure 8. Constructed riffles in an engineered channel to control channel gradient.

these parameter values can introduce considerable uncertainty, which may not be readily quantified. Therefore, the quantitative results from applying the UBCRM must be viewed critically and assessed using professional judgment.

10. ADJUSTMENT OF CHANNEL SLOPE

Changes in S on the Coldwater River between 1953 and 2003 appear largely as a consequence of reduced sinuosity (Figure 6). Clearly, reducing S during restoration efforts could be achieved by increasing channel sinuosity. Alternatively, in an engineered restored channel, constructed riffles [*Newbury and Gaboury*, 1994; *Walker et al.*, 2004] could also be used to control channel gradient as an alternative to increased sinuosity. The equilibrium slope would establish over time from deposition of sediment behind the riffles (Figure 8). The use of riffles for gradient control might be particularly appropriate in many urban settings where encroachment from development precludes restoration to a highly sinuous channel.

11. SUMMARY

The UBCRM is simple to implement with modest data requirements yet offers several potential advantages over empirical regime equations: (1) It does not require regional data to develop empirical regime relations. (2) Parameters are site-specific and not based on regional averages. (3) Parameters are generally physically based.

Application to stream restoration or channel design typically involves calibration for a specific reach and then evaluation of how the reach could potentially respond to changes in the formative discharge, bank strength, or any of the other governing conditions related to channel morphology in the model. This can be used to provide quantitative design values.

Because of the physically based nature of the parameters, analysis and design is directly linked to fluvial processes including flow resistance, sediment transport, and bank stability. Applied correctly, the UBCRM can provide qualitative and quantitative insight into the primary causes of historic disturbance and can serve as an aid in restoration design.

Current formulation of the model is limited to gravel bed rivers. However, the basic theory could be extended to include sand bed rivers. Application assumes that the reach is in equilibrium with the prevailing hydrology, sediment supply, and bank strength. This may not be valid in all cases. In particular, application to incising or aggrading streams where sediment load and transporting capacity are not balanced may not be justified.

The simplified equations used here (equations (6a) and (6b)) are primarily for illustrative purposes. Actual analysis and design for restoration is more effective using numerical or spreadsheet models, which can be downloaded from http://www.geog.ubc.ca/~beaton/UBC%20Regime%20Model.html. These models offer greater flexibility and access to several bank stability models (noncohesive, cohesive, and composite banks). A detailed user's manual [*Eaton*, unpublished report, 2007] can also be downloaded; this provides an expanded description of the numerical models, data requirements, and additional case studies.

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Application of the CONCEPTS Channel Evolution Model in Stream Restoration Strategies

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The series of biennial U.S. National Water Quality Inventory surveys show no reduction in the percentage of degraded miles of streams since the early 1990s despite an exponential increase in river restoration projects to improve water quality, enhance in-stream habitat, and manage the riparian zone. This may suggest that many river restoration projects fail to achieve their objectives and could therefore benefit from using proven models of stream and riparian processes to guide restoration design and to evaluate indicators of ecological integrity. The U.S. Department of Agriculture has developed two such models: the channel evolution computer model CONCEPTS and the riparian ecosystem model REMM. CONCEPTS is a robust computational model for simulating the long-term evolution of incised and restored or rehabilitated stream corridors. REMM is a computational model for evaluating management decisions to control nonpoint source pollution in the riparian zone. These models have been integrated to evaluate the impact of in-stream, edge-of-field, and riparian conservation measures on stream morphology and water quality. This chapter presents how in-stream restoration measures are represented in CONCEPTS. Further, the capabilities of CONCEPTS and REMM are demonstrated through model applications that evaluate the long-term stability of newly constructed channels, the impact of bank protection on downstream sediment loads and streambed composition, and the effectiveness of woody and herbaceous riparian buffers in controlling stream bank erosion of an incised stream.

1. INTRODUCTION

The rapid increase in number of stream restoration projects in the United States over the last two decades [*Bernhardt et al.*, 2005] has not led to a reduction in miles of streams impaired by sediment [*U.S. Environmental Protection Agency*, 1994, 1995, 1997, 2000, 2002, 2007]. Because only 564,000 (16%) of total U.S. stream miles are assessed, it is likely that the impact of the documented restoration projects on the assessment is limited. Nevertheless, many restoration projects fail to achieve their objectives. According to *Palmer and Allan* [2006], this is due to the lack of policies to support restoration standards, to promote proven methods, and to provide basic data needed for planning and implementation.

A common reason of restoration project failure is to only focus on the reach to be restored and thereby ignoring its location within the watershed [*Palmer and Allan*, 2006]. As a consequence, projects often are exposed to flow and sediment regimes different from those used in the design phase resulting in possible flooding, bank collapse, or excessive scour and fill of the stream bed. *Palmer et al.* [2005] recommend the use of proven models of in-stream and riparian processes not only to guide the design of restoration projects but also to assess, both pre- and postproject, indicators of ecological integrity. The U.S. Department of Agriculture

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(USDA) has developed various computer models to evaluate the effects of hillslope (upland) and in-stream restoration measures on stream morphology, downstream sediment loads, and to predict key attributes of stream corridors known to control physical habitat quality, such as base flow statistics, water temperature, bed material composition, and large wood density [*Shields et al.*, 2006]. Two of these models, CONCEPTS [*Langendoen and Alonso*, 2008; *Langendoen and Simon*, 2008] and REMM [*Altier et al.*, 2002], simulate in-stream and riparian physical processes at the stream-corridor scale. CONCEPTS and REMM have been shown to successfully simulate the effects of conservation measures on the transport of water and sediment within a stream and riparian zone [e.g., *Lowrance et al.*, 2000; *Wells et al.*, 2007; *Langendoen et al.*, 2009b] and therefore form a proven method to assess the impact of stream restoration strategies on channel morphology. A third model, BSTEM, simulates the effects of restoration measures on sediment loadings from stream banks at individual cross sections and is discussed elsewhere in this book [*Simon et al.*, this volume].

This chapter presents: (1) an overview of the CONCEPTS and REMM models, (2) guidelines for evaluating stream restoration measures using CONCEPTS, and (3) a range of applications to demonstrate the capabilities of the CONCEPTS and REMM models.

2. SIMULATION OF STREAM CORRIDOR PHYSICAL PROCESSES

2.1. Simulated Processes

Restoration of the functions of a stream is needed when a natural reequilibration either is not physically possible or is very slow. The complex dynamics of physical and ecological processes and their interactions make designing effective restoration practices difficult. Computer models that can properly simulate these processes can guide professionals in the planning and design phases of restoration practices. The discussion below of the CONCEPTS and REMM models is limited to the simulation of physical processes that affect stream morphology. However, these models can be used to calculate indicators that assist in ecological restoration [*Shields et al.*, 2006].

The forces acting on the stream boundary and the resistance to erosion of the boundary materials govern stream morphology. In general, the force exerted by the flowing water on the channel boundary depends on flow velocity distribution and boundary roughness. The resistance to erosion is a function of boundary material properties such as texture, density, erodibility, and shear strength. These properties are significantly affected by the presence of riparian vegetation. Stream restoration measures are designed to affect both the forces exerted by the flow (e.g., lowering near bank velocities) and the resistance to erosion of the channel boundary (e.g., reducing its erodibility). The CONCEPTS and REMM computer models simulate these processes and their controls. The following sections very briefly discuss the science included in the models and their integration. More details are given by *Altier et al.* [2002], *Langendoen and Alonso* [2008], *Langendoen and Simon* [2008], and *Langendoen et al.* [2009a].

2.2. CONCEPTS: Computer Model of In-Stream Processes

The CONCEPTS computer model has been developed to simulate the evolution of incised streams and to evaluate the long-term impact of rehabilitation measures to stabilize stream systems and reduce sediment yield [*Langendoen and Alonso*, 2008; *Langendoen and Simon*, 2008]. CONCEPTS simulates unsteady, one-dimensional (1-D) flow, graded sediment transport, and bank erosion processes in stream corridors. It can predict the dynamic response of flow and sediment transport to in-stream structures.

2.2.1. Hydraulics. CONCEPTS models streamflow as 1-D along the channel's centerline. Hence, it is limited to fairly straight channels; it cannot predict bar formation and channel migration. CONCEPTS simulates gradually varying flow (described by the Saint-Venant equations) as a function of time along a series of cross sections representing stream and floodplain geometry. The governing system of equations are solved using the generalized Preissmann scheme, allowing a variable spacing between cross sections and large time steps conducive to long-term simulations of channel evolution. The implementation of the solution method contains various enhancements to improve the robustness of the model, particularly for flashy runoff events.

2.2.2. Sediment transport and bed adjustment. Alluvial stream banks are typically composed of fine-grained deposits containing clays, silts, and fine sands (hereafter referred to as fines), which may overlay coarser relic point bars. Streambeds are more commonly composed of sands and gravels, resistant clay layers, or bed rock. Therefore, the range in particle sizes being transported in alluvial streams may be quite large, and the composition of the sediment mixture in transport may be quite different from that of the bed material if a majority of the sediments are fines transported in suspension. CONCEPTS therefore calculates sediment transport rates by size fraction for 14 predefined sediment size classes ranging from 10 μm to 64 mm.

CONCEPTS uses a total-load evaluation of bed material transport and treats movement of clays and fine silts (<10 μm) as pass-through background wash load. The differences in transport mechanics of suspended and bed load movement are accounted for through nonequilibrium effects. The composition of bed surface and substrate is tracked, enabling the simulation of vertical and longitudinal fining or coarsening of the bed material.

2.2.3. Stream bank erosion. CONCEPTS simulates channel width adjustment by incorporating the two fundamental physical processes responsible for bank retreat: fluvial erosion or entrainment of bank material particles by flow and bank mass failure due to gravity. Bank material may be cohesive or noncohesive and may comprise numerous soil layers.

The detachment of cohesive soils is calculated following an excess shear-stress approach. An average shear stress on each soil layer is computed. If the critical shear stress of the material is exceeded, entrainment occurs. CONCEPTS is able to simulate the development of overhanging banks.

Stream bank failure occurs when gravitational forces that tend to move soil downslope exceed the forces of friction and cohesion that resist movement. The risk of failure is expressed by a factor of safety, defined as the ratio of resisting to driving forces or moments. CONCEPTS performs stability analyses of wedge-type failures and cantilever failures of overhanging banks. The effects of pore water pressure and confining pressure exerted by the water in the stream are accounted for.

2.3. REMM: Computer Model of Riparian Processes

REMM has been developed as a tool to aid natural resource agencies and others in making decisions regarding management of riparian buffers to control nonpoint source pollution [*Altier et al.*, 2002]. The structure of REMM is consistent with buffer system specifications recommended by the U.S. Forest Service and the USDA Natural Resources Conservation Service as national standards [*Welsch*, 1991]. The specified riparian buffer system consists of three zones parallel to the stream, representing increasing levels of management away from the stream. Although REMM is designed to simulate this type of buffer system, the model can be used with different types of vegetation within each zone. Processes simulated in REMM include storage and movement of surface and subsurface water, sediment transport and deposition, transport, sequestration, and cycling of nutrients, and vegetative growth.

2.3.1. Hydrology. Water movement and storage is characterized by processes of interception, evapotranspiration (ET), infiltration, vertical drainage, surface runoff, subsurface lateral flow, upward flux from the water table in response to ET, and seepage. The storage and movement of water between the zones is based on a combination of mass balance and rate-controlled approaches. Vertical drainage from a soil layer occurs when soil water content exceeds the field capacity. The amount drained from a soil layer also depends on the capacity of the receiving layer and is set equal to the lesser of the hydraulic conductivities of the draining and receiving layers. When a shallow groundwater table is present, soil water content above the groundwater table is assumed to be in equilibrium with the water table. The matric potential or pressure head is approximated by the height above the water table. Soil water content is related to pressure head using Campbell's equations [*Campbell*, 1974].

Subsurface lateral movement is assumed to occur when a water table builds up over the restricting soil layer. The lateral water movement is simulated using Darcy's equation. Rates of lateral subsurface movement between zones are constrained by the lesser of the respective hydraulic conductivities of the soil layers in each zone. If rates of soil water movement for the upslope zone exceed the transmission rates for the downslope zone, the soil water excess is accumulated in the upslope zone until it is saturated. A seep will then occur to the surface of the downslope zone.

2.3.2. Plant growth. REMM simulates the growth of several types of herbaceous and woody vegetation in two canopy layers for even-aged forest stands. Individual species present in a particular buffer system may be characterized through the parameterization of various variables, which represent values for the initial sizes of the plants, rates of photosynthesis, respiration requirements, rates of growth and mortality, sensitivity to light and temperature, response to nutrients, and timing of phenostages.

2.4. CONCEPTS-REMM Integration

The physical process modules of CONCEPTS and REMM have been integrated to study the interactions between in-stream and riparian processes. A daily feedback of several parameters has been established to calculate: (1) daily stream loadings of water, sediments, and nutrients emanating from the riparian buffer; (2) effects of water surface elevation on soil water in the riparian zone (seepage and recharge); (3) effects of pore water pressure and root biomass on stream bank stability; and (4) in case of bank failure, stream loadings of sediments, nutrients, and plant/tree biomass contained by the failure block.

The bank stability analysis performed by CONCEPTS accounts for soil water content and root biomass in the bank. The groundwater table and vertical distribution of soil water computed by REMM in the zone nearest to the channel are used to calculate pore water pressure. The pore water pressure is assumed hydrostatic below the groundwater table. Soil water content above the groundwater table is converted to suction values using *Campbell*'s [1974] equation. The mechanical effect of roots is to enhance the confining stress and resistance to sliding and increase the shear strength of the soil/root mass through the binding action of roots in the fiber/soil composite [e.g., *Coppin and Richards*, 1990; *Gray and Sottir*, 1996]. The vertical distribution of root biomass concentration calculated by REMM is converted to a root-area-ratio (RAR) and used to modify soil shear-strength using *Wu et al.*'s [1979] equation.

3. INPUT DATA REQUIREMENTS

3.1. CONCEPTS

CONCEPTS uses two types of input data: (1) input data that control the execution of the model (e.g., simulation start and end dates, simulated processes, and requested output) and (2) input data that characterize the modeled stream corridor. Different data are required to perform hydraulic routing, sediment routing, and stream bank erosion calculations.

To perform hydraulic routing the channel and floodplain geometry are required and are represented by a series of cross sections. These data are typically obtained through channel surveys using standard methods such as level or total station. Flow resistance is parameterized using the Manning n friction factor. The user can input different Manning n values for streambed, left and right banks, and left and right floodplains. Manning n values are reported in literature and can be calibrated using observed water surface profiles or flow depths. Discharge has to be specified at the inlet of the study reach and at tributary inflow points. Time series of discharges can be obtained through measurements or generated using hydrologic computer models. A boundary condition at the model outlet is optional. The model calculates a looped rating curve internally based on local flow conditions. However, if water level at the downstream boundary is controlled externally, the user can specify a rating curve or a time series of water level elevation.

To simulate sediment transport and bed adjustment initial bed material stratigraphy with grain size distribution and porosity for each stratigraphic layer is required. Bed material can vary along the stream but is assumed homogeneous across the stream. Bed material gradation can be determined by sampling the bed material. Entrainment of cohesive, fine-grained bed material is calculated using an excess shear-stress approach that requires the specification of a critical shear stress below which no erosion takes place and an erosion rate or erodibility coefficient that represents the rate at which the cohesive bed material is eroded once the critical shear stress is exceeded. The resistance to erosion can be measured in situ using portable flumes or jet testers, or samples can be collected and tested in laboratory settings using annular flumes or flumes such as the Erosion Function Apparatus [*Briaud et al.*, 2001]. At inflow locations, fractional sediment transport rates have to be specified, which can be either measured or calculated using sediment transport relations.

Stream bank erosion calculations require the specification of bank material stratigraphy, with its associated grain-size distributions, bulk density, resistance to erosion (critical shear stress and erosion-rate coefficient) values, and shear-strength (cohesion and friction angle) values. Most properties can be measured by collecting samples and consequent laboratory analysis. Resistance to erosion and shear strength properties can also be measured in situ using jet test and borehole shear test devices, respectively.

Validation and applications of CONCEPTS [e.g., *Wells et al.*, 2007; *Langendoen and Alonso*, 2008; *Langendoen and Simon*, 2008; *Langendoen et al.*, 2009a, 2009b] showed that it can satisfactorily predict and quantify (a) the temporal progression of an incised stream through the different stages of channel evolution, (b) changes in thalweg elevation, (c) changes in channel top width, and (d) bed material grain size distribution. However, bed and bank material properties representing resistance to erosion and failure must be adequately characterized. It is highly recommended to perform a geomorphic analysis of the stream system to determine channel conditions and variations in sediments and soils along the stream. Such an analysis could be performed using the Rapid Geomorphic Assessment technique [e.g., *Simon et al.*, 2002]. Differences between observed and simulated evolution are commonly largest along reaches where either model assumptions regarding flow and sediment transport (e.g., 1-D assumption) are inappropriate, as is the case in the late stages of channel adjustment, or assumptions regarding input data (e.g., channel geometry, water inflows, or bed and bank material properties) are required. The use of median and average values of critical shear stresses and effective cohesion generally provide good results. Because critical shear stresses typically vary greatly both between different soils and within a soil, users of the model should measure an adequate number of critical shear stress values for each soil in the bed and banks.

3.2. REMM

REMM's input data are related to model execution, the physical description of the riparian buffer, and boundary conditions such as weather and upland inputs. The buffer characteristics comprise its physical dimensions (length, width, slope, etc.), the fraction of the area covered by vegetation, and physical descriptors of litter and soil layers (such as initial carbon and nutrient levels, and hydrologic properties). Vegetation data includes information on the plant, factors related to photosynthesis, transpiration characteristics, nutrient content of plant part pools, and the initial size of the plants. Regional databases are available that describe typical plant characteristics for various species.

Daily weather input consists of rainfall amount and duration, minimum and maximum air temperature, incoming solar radiation, and wind velocity. Gridded data sets, such as VEMAP (http://www.cgd.ucar.edu/vemap/V2.html), are available that cover the United States if these data are not available from nearby weather stations. Daily field inputs include surface runoff and subsurface drainage volumes and associated eroded soil material, inorganic and organic materials, and plant nutrients.

Calibration and testing procedures of the REMM submodels are reported in the works of *Altier et al.* [1998], *Bosch et al.* [1998], and *Inamdar et al.* [1998a, 1998b].

4. IMPLEMENTATION OF RESTORATION MEASURES

This section describes how in-stream and riparian restoration measures can be represented in CONCEPTS. CONCEPTS is capable of evaluating restoration measures at individual cross sections and along entire reaches. This allows, for example, the determination of restoration measure placement or the length of protection needed. It should be noted that because CONCEPTS is a 1-D model, it cannot simulate the complex 3-D flow near in-stream structures and the resulting local channel morphology. The 3-D effects are averaged over the distance between two consecutive cross sections. However, a 1-D approach can adequately assess the long-term impact of restoration measures on channel stability.

4.1. Streambed Restoration Measures

Streambed restoration measures are typically employed to stabilize the streambed and control channel grade. Common grade control measures are sills or drop structures that can be constructed of large stones, logs, or sheet pile weirs.

There are two methods to evaluate grade control measures using CONCEPTS. Both methods assume that the grade control measures are stable under the full range of imposed flow conditions. First, if the designed drop in bed elevation at the structure is rather small, such that the flow drowns the structure for medium to large runoff events, the bedrock elevation can be set to the level of the bed surface at the cross section with the grade control structure. This will prevent erosion below this elevation. Deposition is possible, and the deposited material can be eroded in the future, but the extent of erosion is then limited to the top of the grade control structure.

The second method uses a drop structure element in CONCEPTS. This method should be used if the drop in bed elevation at the structure is significant. In this case, free-fall conditions cause a significant energy head loss that may not be simulated adequately by the above method. This method simulates both free fall and drowned conditions at drop structures. Bed load will be captured by the structure as long as its invert exceeds the upstream bed elevation. Once bed elevation exceeds structure invert, all sediment will pass the structure and no further deposition will occur upstream of the structure. The drop structure geometry is limited to a symmetrical trapezoidal cross section with a horizontal bottom.

4.2. Stream Bank Restoration Measures

Stream bank restoration practices can be placed anywhere on the bank by introducing layers that represent the erodibility of the stabilization measure (Figure 1). Hence, these bank protection measures could cover the toe only or protect the entire bank face. Similarly, the effects of riparian vegetation on top of the bank on stream bank erosion can be evaluated using different soil layers.

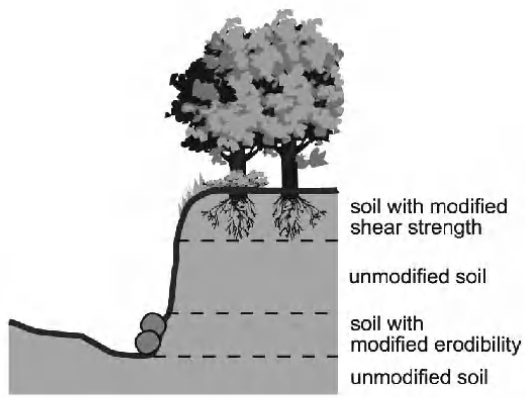

Figure 1. Use of soil layers to characterize stream bank protection and stabilization measures in CONCEPTS. The shown stream bank comprises four soil layers and three soils. The top layer is the unmodified soil with an increased cohesion value representing the added reinforcement provided by the tree roots and a reduced erodibility coefficient. The third layer is the unmodified soil with an increased critical shear stress value and reduced erodibility coefficient representing the resistance to fluvial erosion provided by the rock serving as toe protection.

4.2.1. Protection against fluvial erosion. A bank material must be introduced to represent the protected portion of the bank. The critical shear stress and erodibility coefficient for this bank material layer should characterize the resistance to erosion of the stream bank protection measure. For example, the critical shear stress could be set to the allowable shear stress used in tractive channel design. The *Natural Resources Conservation Service* [2007, Chapter 8] tabulates allowable shear stress values for many bank protection measures.

A number of protection measures, for example, vegetation, root wads, or vanes, deflect the flow away from the bank thereby reducing shear stresses exerted by the flow, which cannot be simulated accurately by a 1-D model such as CONCEPTS. However, this could be represented by an equivalent increase in critical shear stress of the affected bank soils.

4.2.2. Bank stabilization measures. Bank stabilization measures typically enhance soil shear strength. This could be done, for example, by improving drainage or by mechanical reinforcement provided by roots of riparian vegetation. The vertical distribution of root biomass of riparian vegetation is represented by introducing bank material layers with varying cohesion values. The Riproot model of *Pollen-Bankhead and Simon* [2009] can be used to calculate the added cohesion due to plant roots.

5. MODEL APPLICATION

This section presents three sample applications in which the CONCEPTS model was used to assess the performance of stream restoration measures at the stream corridor scale. The first application evaluates the long-term stability of a channel constructed within a reservoir deposit to minimize bank erosion and downstream sediment load. In the second example, CONCEPTS is used to assess the impact of urbanization on channel morphology and the potential benefits of stream bank protection measures. The last example presents the capabilities of the combined CONCEPTS and REMM model to evaluate vegetative riparian management strategies.

5.1. Kalamazoo River Dam Removal

5.1.1. History. Between the mid-1800s and the early 1900s, four dams were constructed on the Kalamazoo River between Plainwell and Allegan, Michigan. The impoundments have been the depositories of upstream sediment and industrial waste materials containing polychlorinated biphenyl (PCB) and kaolinite clays. During the 1960s, water levels behind the decommissioned hydroelectric dams were lowered, exposing the previously inundated material. In response to the lowering of water levels, the river began to erode the sediments and transport them downstream, but much of this waste clay remains impounded behind the dams mainly as floodplain deposits [*Rheaume et al.*, 2002, 2004]. The state of Michigan is interested in removing the dams while minimizing impacts locally and to downstream reaches, and to provide for improved fisheries. CONCEPTS was used to simulate sediment loadings from PCB-contaminated stream banks and channel changes for a section between Plainwell and Otsego, which contains the Plainwell and Otsego City Dams, under three different scenarios: (1) dams in (DI) or baseline, (2) dams out (DO), and (3) design (D). The design scenario evaluates a redesigned stream-riparian corridor to minimize the adverse local and downstream impacts of the dam removal.

5.1.2. Study reach. The study reach of the Kalamazoo River is 8.8 km long, from river kilometer (rkm) 82.4 (cross-section OC8), to cross-section P3, at rkm 91.2 (Figure 2). The model of the study reach is composed of 52 cross sections and contains both Plainwell and Otsego City Dams. The Plainwell Dam is 172 ft (52.4 m) wide and 14 ft (4.3 m) high. The Otsego City Dam is 151 ft (46.0 m) wide and 13 ft (4.0 m) high. The study reach can be separated into three distinct subreaches based on location relative to the Plainwell and Otsego City Dams. The Otsego (OC) reach extends from rkm 82.4 to the Otsego City Dam at rkm 85.3. The Plainwell-Otsego (POC) reach extends from the upstream end of the Otsego City Dam to the Plainwell Dam at rkm 88.3. The Plainwell reach extends from the Plainwell Dam to the upstream boundary of the study reach at rkm 91.2.

5.1.3. Input data. Flows for all three simulation scenarios are based on a 17.7 year discharge record (October 1984 to June 2002) from the USGS gauge on the Kalamazoo River at Comstock, Michigan (04106000). The 17.7 year flow record was created using daily data from 1984 to 1989 and hourly data from 1989 to June 2002 to account for changing hydraulic conditions and instantaneous peaks. The Gunn River flows into the POC section of the study reach from the north between cross-sections G5 and G6 (Figure 2). Because there is no flow data for this tributary, the flow from the Gunn River (296 km^2) was estimated using a drainage area comparison with the flow record from the Kalamazoo River at Comstock (04106000; 2740 km^2). Given the respective drainage areas, the Gunn River discharge record was 17% of the Kalamazoo River at Comstock discharge record.

A sediment rating curve for fines (clays, silts, and very fine sands) was derived from 51 suspended-sediment samples collected by the USGS at the Plainwell gauge. For coarse sediment particles transported as bed load, the sediment

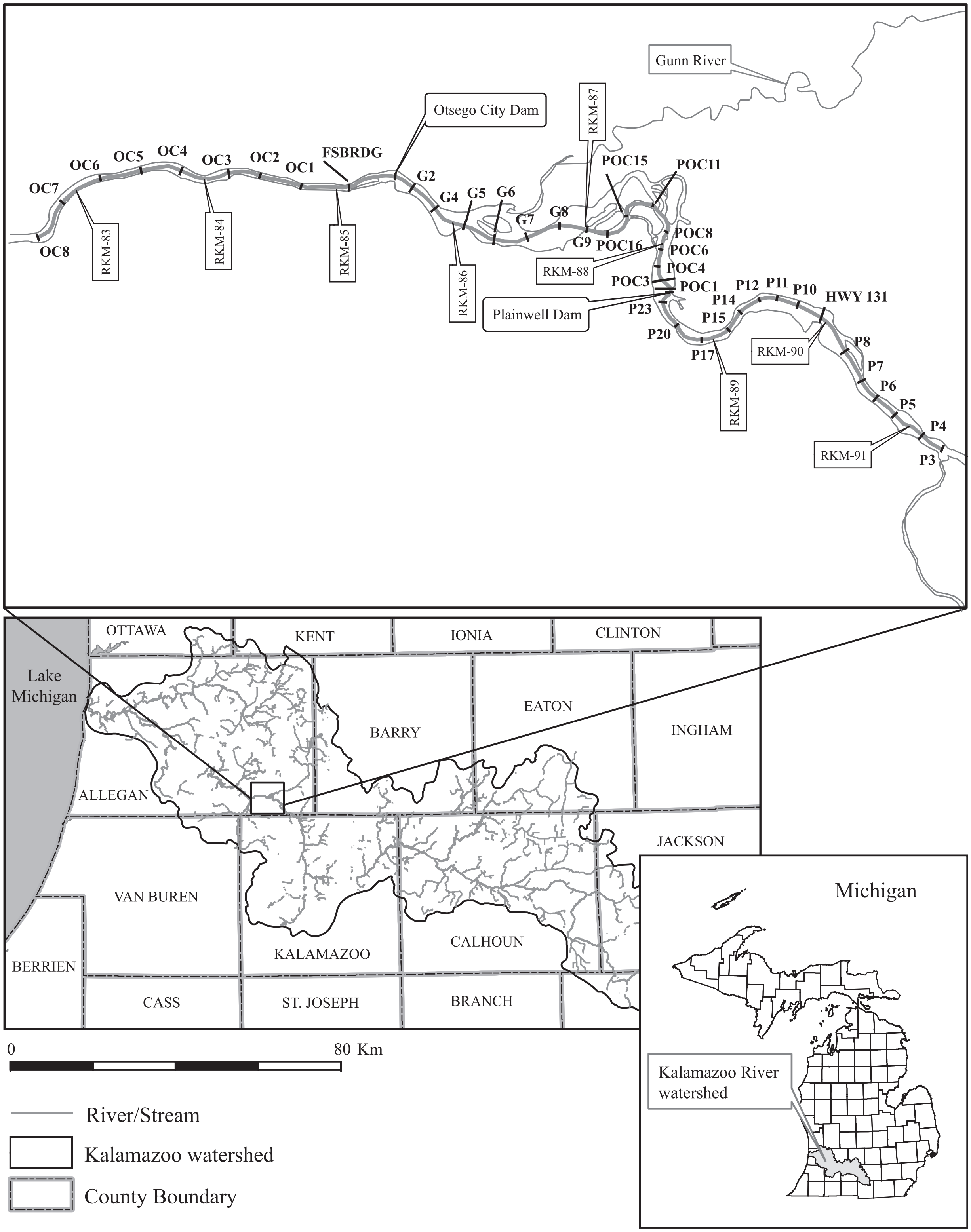

Figure 2. Map of Kalamazoo River study reach (85°40′W, 42°28′N) showing modeled cross sections and locations of the Plainwell and Otsego City Dams.

transport rates at the inlet are assumed to equal the local sediment-transport capacity of the flow.

The simulation period is August 2000 through November 2037. The start date coincides with the first cross-section surveys by the USGS [*Rheaume et al.*, 2002]. The inflow record of water and sediment consists of the observed flow through June 2002 followed by two sequences of the 17.7 year flow record discussed above. The simulation period is long enough for channel adjustments to reach equilibrium for the DO and D scenarios.

Bed material stratigraphy and composition were determined at 101 transects covering the study reach [*Rheaume et al.*, 2002, 2004] and were directly used in the model simulations. Data on bank material stratigraphy, composition, and properties were collected at 27 locations. Regions with similar bank material were identified, and data collected in these regions were aggregated. Critical shear stress of the bank material ranges from a minimum of 1.3 Pa along the POC reach to a maximum of 70 Pa along the left bank immediately upstream of the Plainwell Dam. Effective cohesion ranges from a minimum of 0 Pa for sandy bank material to a maximum of 6.8 kPa for the right bank of the most upstream cross sections. A comprehensive report of the measured values and those assigned to each model cross section is provided in the work of *Wells et al.* [2004].

5.1.4. Modeling scenarios. The DI scenario assumes current channel geometries and boundary sediments as initial conditions. This simulation is used as a baseline by which to compare the two alternative scenarios in terms of gross amounts of channel change, the mass of material eroded from channel banks, and fine-grained sediment transport. The DO scenario also assumes current channel geometries as initial conditions but with the Plainwell and Otsego City Dams no longer in place, leaving 3–4 m high knickpoints. Finally, the design scenario also assumes that the two dams are no longer in place; however, design channel geometry is used instead of the current channel geometry for initial conditions [*Rachol et al.*, 2005].

For the D scenario, channel geometry, channel location, floodplain area, and channel elevation were modified between the Otsego City Dam (rkm 85.3) and cross-section P15 (rkm 89.0) to minimize potential flooding, erosion, or sedimentation problems after removal of the dams [*Rachol et al.*, 2005]. Cross sections in the impounded area upstream of the Plainwell Dam were mainly modified by lowering the channel to its predam elevation and removing impounded sediment to increase floodplain area. The slope through this reach is similar to that for predam conditions. In the POC reach, the slope of the designed channel is steeper than that for predam conditions. In the anastomosing part of the reach, valley cross sections were modified by simplifying the multiple channel system into one or two main channels. Downstream of the multichannel reach, the channel elevation was lowered below its predam elevation to provide a smooth transition to the incised reach downstream of the Otsego City Dam and impounded sediment removed to create a floodplain area. Streambeds of excavated cross sections were assigned material composition and properties found at the level of excavation [*Rheaume et al.*, 2002, 2004].

5.1.5. Results and discussion. The DI modeling scenario represents a baseline condition with existing channel geometries (including the low-head dams) and boundary characteristics. In general, the simulation predicted aggradation in the Plainwell reach with sediment deposited in the backwaters caused by U.S. Highway 131 bridge and the Plainwell Dam (Figure 3a). The main branch of the POC reach is slightly erosional, whereas the OC reach is mainly a transport reach. Results show that over the entire study reach, there is a net annual deposition of material (4100 t yr^{-1}). However, silts and clays are eroded primarily from the bed at an average annual rate of 1990 t yr^{-1}.

For the DO scenario, large-scale erosion of the deposits upstream of the dams occurred very quickly as the fine-grained particles were unable to resist the increased shear (Figure 3b). The channel incises down to its parent bed material (predam elevations), limiting the extent of erosion to the depth of the reservoir deposits. In the Plainwell reach, bed deposition of 6400 t yr^{-1} for the baseline (DI) scenario turned to erosion of 289 t yr^{-1} for the DO scenario. Net bed erosion in the POC reach increased 1346% to 6580 t yr^{-1} for the DO scenario compared with the DI scenario (455 t yr^{-1}). Bank erosion also increased greatly (1645%) in the POC reach from about 157 to 2740 t yr^{-1} on average, due to higher shear stresses exerted by the flow caused by the initial steepening of the channel, especially upstream of the Otsego City Dam location.

Figure 3c shows the differences between the current thalweg profile and that of the design channel for the D scenario. Simulation shows the POC and OC reaches are fairly stable because of the coarse-grained bed material. Channel deposition (2570 t yr^{-1}) simulated under this scenario is 37% lower than the DI scenario (4100 t yr^{-1}). Erosion of stream bank materials <63 μm (112 t yr^{-1}) is 28% greater than that for the DI scenario (87.7 t yr^{-1}).

Over the simulation period, the DI/baseline scenario provides the smallest load passing the outlet (Figure 4 and Table 1). The total load is the largest for the DO scenario; however, the silt and clay fraction is smallest for the DO and D scenario. The increase in sand-sized sediment transport

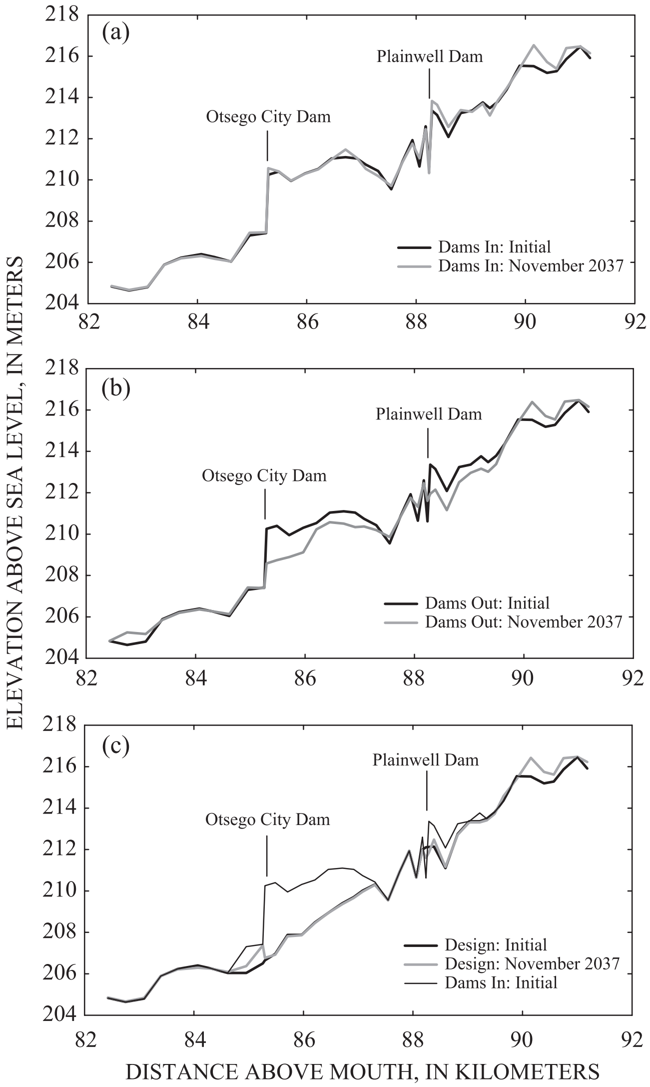

Figure 3. Initial and final thalweg profiles for the (a) dams in, (b) dams out, and (c) design scenarios.

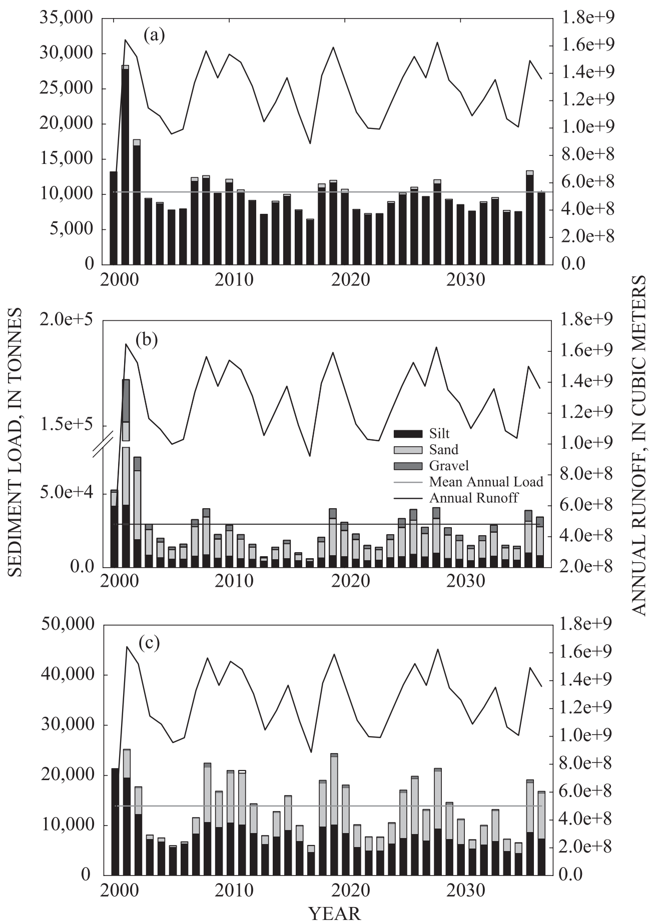

Figure 4. Sediment loadings at the outlet of the study reach for the (a) dams in, (b) dams out, and (c) design scenarios.

appears to limit the amount of fines being transported. Sediments eroded from the channel boundary and downstream sediment load are similar and fairly low for the DI and D scenarios, indicating a stable stream system. Removal of the low-head dams induces severe channel bed and stream bank erosion upstream of the former dam locations, significantly increasing sediment load. However, most of these sediments are eroded in the first 3 years (Table 1). The quantities of fine-grained material (<63 μm) transported past the downstream boundary over the last 35 years of the simulation are similar to those of the DI and D scenarios. Therefore, most of the channel adjustment due to dam removal occurs in the first 3 years of the simulation.

Although the DI (baseline) case clearly provides the smallest loads for total sediment transport, in order to improve navigation and fisheries within this reach of the Kalamazoo River, the removal of the low-head dams and implementation of the design proposed by the USGS provides reduced loadings in materials less than 63 μm, and total loads passing OC8 are comparable with the existing DI loadings.

Table 1. Simulated Average Annual Sediment Load Passing the Downstream Boundary of the Kalamazoo River Study Reach

	Sediment Yield in Kilotons per Year		
Scenario	<63 μm	<2 mm	Total
Dams in (DI)	10.4	10.5	10.5
Dams out (DO)	8.9	25.9	30.1
Dams out (DO, year 1–3)	43.7	114	127
Dams out (DO, year 4–38)	6.5	20.0	23.6
Design (D)	8.4	13.9	14.2

5.2. Shades Creek Bank Stabilization

5.2.1. Overview. The Shades Creek watershed is located near Birmingham, Alabama, in an area experiencing rapid urbanization (Figure 5). Nearly the entire length of Shades Creek is listed as impaired due to sediments. Surveys conducted between 1990 and 1993, and again in 1997, indicated impairment caused by collection system failure, road and bridge construction, land development, urban runoff, removal of riparian vegetation, and bank/shoreline modification. *Simon et al.* [2004] carried out a study to determine bed material composition, sediment yields, and sources in the Shades Creek watershed and to compare these to "reference" sediment yields for unimpaired streams in the region.

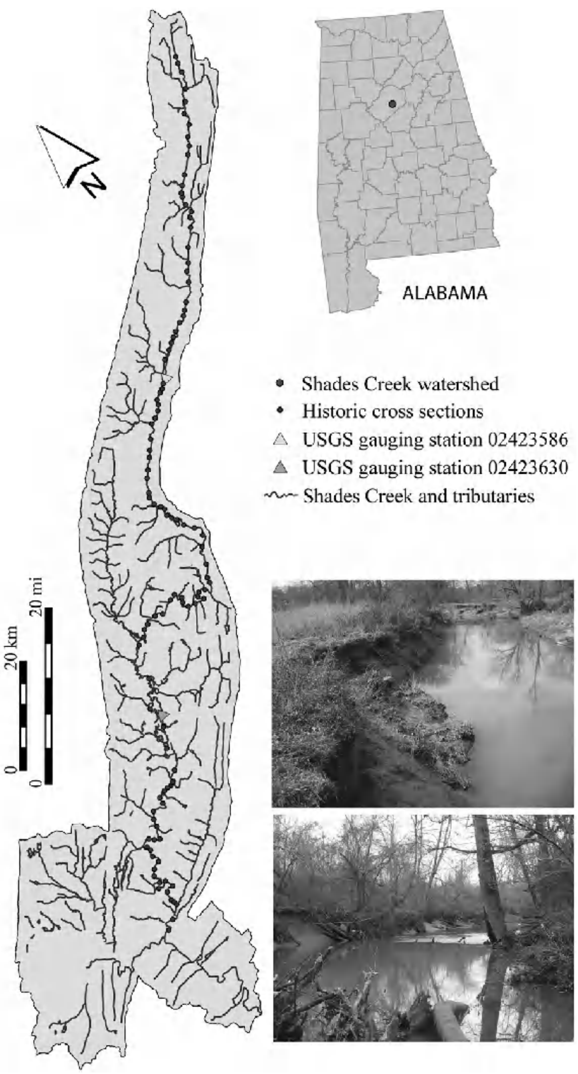

Figure 5. Map of Shades Creek, Alabama (86°51′W, 33°22′N), with photos indicating the degree of stream bank erosion.

As part of the study, CONCEPTS was used in combination with the watershed model AnnAGNPS [*Bingner and Theurer*, 2001] to evaluate, among others, (1) the effects of urbanization on channel erosion and bed material gradation and (2) the potential reduction in fine-grained sediment yield provided by stream bank stabilization measures. AnnAGNPS provides peak flow discharge, runoff volume, and clay, silt, and sand mass for each runoff event for reaches and cells draining into the modeling reach. These data are then converted into triangular-shaped hydrographs. The presented results below describe three simulation scenarios using (1) current (2001) land use (70% forest, 16% pasture, 11% urban, and 3% water), (2) current land use with selected stream bank protection (hereafter referred to as 2001LURP), and (3) land use change from forest to urban, that is, 81% urban and 0% forest (2001LUFU).

5.2.2. Study reach. The Shades Creek modeling reach extends from approximately 10.0 km above the confluence with the Cahaba River to approximately 86.5 km above the confluence with the Cahaba River. The modeling reach is composed of 156 cross sections. Bed and bank material composition and geotechnical properties at each cross section were obtained from sediment samples and in situ testing. Stream bank materials have an average silt/clay content of 15%, an average sand content of 81%, and an average gravel content of 4%. Bank toe materials have an average silt/clay content of 13%, an average sand content of 67%, an average gravel content of 5%, and an average boulder/cobble content of 15%. The streambed materials have an average silt/clay content of 1%, an average sand content of 24%, an average gravel content of 28%, and an average boulder/cobble content of 47%. Measured effective cohesion values were adjusted for root reinforcement by riparian vegetation by adding 2 to 4 kPa to the top 1 m of the bank soils depending on riparian vegetation density and species. Measured critical shear stresses were adjusted for shielding of bank face material by riparian vegetation.

5.2.3. Results. A summary of the simulation results are listed in Table 2. Both runoff and average annual suspended-sediment load showed a discernible increase for the modeling scenario where all forest land was changed to urban (2001LUFU). Increases in sediment load are a direct result of greater runoff rates. This is manifest in the number of cross sections experiencing width adjustment greater than 2.0 m, which increased from 11 for the 2001 land use scenario to 23 for the 2001LUFU scenario. Stream banks are the greatest source of sediments to suspended load, except for the 2001LURP scenario, which simulated protected banks (see Table 3). Uplands were the main source of fines

Table 2. Simulated Annual Runoff, Suspended Sediment Load, Average Widening, and Average Change in Bed Elevation for Shades Creek, Alabama

Scenario	Average Annual Runoff[a] (mm yr^{-1})	Average Annual Sediment Load[a] (t yr^{-1}r)	$\overline{\Delta T}$[b] (cm yr^{-1})	$\overline{\Delta z_b}$[c] (cm yr^{-1})
2001 land use	457	19,700	2.83	0.172
2001LURP	457	19,500	1.62	0.117
2001LUFU	702	29,200	4.20	0.276

[a]Numbers are given at the mouth of Shades Creek with the Cahaba River.
[b]Average annual change in top width along the modeling reach.
[c]Average annual change in bed elevation along the modeling reach.

for the 2001LURP scenario because of the 10,200 t yr^{-1} or 40% reduction in contributions from the banks. This 40% reduction was the result of protecting 11% of the stream length. The 46% (12,300 t yr^{-1}) increase in loads for the 2001LUFU originated mainly from the stream banks (8950 t yr^{-1}) as opposed to uplands (3460 t yr^{-1}).

CONCEPTS was also used to determine the change in bed material composition caused by land use changes. Embeddedness is used to characterize bed material composition. Embeddedness is defined as the percentage of bed material finer than 2 mm (sand, silt, and clay) in gravel or gravel/cobble-dominated streambeds. Shades Creek is located in the Ridge and Valley ecoregions, which reference median embeddedness value is 4% and the reference third quantile embeddedness value is 13.4% [*Simon et al.*, 2004]. Along Shades Creek, there are 53 sections with a coarse-grained streambed, 42 of which are located within stable reaches. The embeddedness of 10 cross sections is smaller than 4%, and the streambed of 26 cross sections has an embeddedness value smaller than 13.4%.

For the 2001 land use scenario, the number of coarse-grained cross sections has reduced to 24 due to aggradation. Only three sites have an embeddedness value smaller than 4%. There are seven sites with an embeddedness value smaller than 13.4%. The number of sites with coarse-grained streambeds between rkm 45 and 55 has reduced from ten to only one, indicating significant deposition of fines.

For the 2001LURP scenario, the number of coarse-grained cross sections has reduced to 29; however, this is five more than for the 2001 land use scenario. Only three sites have an embeddedness value smaller than the reference median of 4%. There are eight sites with an embeddedness value smaller than the reference third quartile of 13.4%. The average embeddedness is slightly smaller for the 2001LURP scenario than that for the 2001 land use scenario.

Table 3. Relative Source Contributions of Uplands and Stream Banks to Suspended Sediment for Shades Creek, Alabama

	Uplands (%)		Stream Banks (%)		Total (t yr^{-1})	
Scenario	Fines	Sands	Fines	Sands	Fines	Sands
2001 land use	40.3	31.2	59.7	68.8	18,700	8,000
2001LURP	88.7	33.8	11.3	66.2	8,500	7,390
2001LUFU	37.2	27.6	62.8	72.4	27,200	11,800

For the 2001LUFU scenario, the number of coarse-grained cross sections has reduced to 26, two more than for the 2001 land use scenario. Only one site has an embeddedness value smaller than the reference median of 4%. There are nine sites with an embeddedness value smaller than the reference third quartile of 13.4%.

The above modeling scenarios show that targeted bank protection is needed to prevent the fining of coarse-grained beds caused by ongoing urbanization of the watershed. For example, a 40% reduction in fine-grained sediment loadings from stream banks can be realized by protecting 11% of the stream length.

5.3. Evaluation of Vegetative Bank Stabilization Treatments

5.3.1. Overview. The integrated CONCEPTS and REMM models were used to study the effectiveness of woody and herbaceous riparian buffers in controlling stream bank erosion along an incised reach of the Goodwin Creek, Mississippi (Figure 6). Between 1996 and 2006, extensive research on stream bank failure mechanics was conducted along this reach. The following data were collected at the study site: cross-section geometry, water surface elevations, bank material properties, pore water pressures in the bank, precipitation, root mapping and tensile strength, canopy interception, and plant stem flow. Two flow measuring flumes in upstream tributaries provide continuous discharge and fine sediment data. A NOAA SURFRAD station located in the watershed collects the following weather and climate input data for REMM: incoming solar radiation, air temperature, relative humidity, wind speed and wind direction.

Major failure episodes have occurred, resulting in up to 5.5 m of top bank retreat along the right bank between March 1996 to March 2001, which increased channel top width

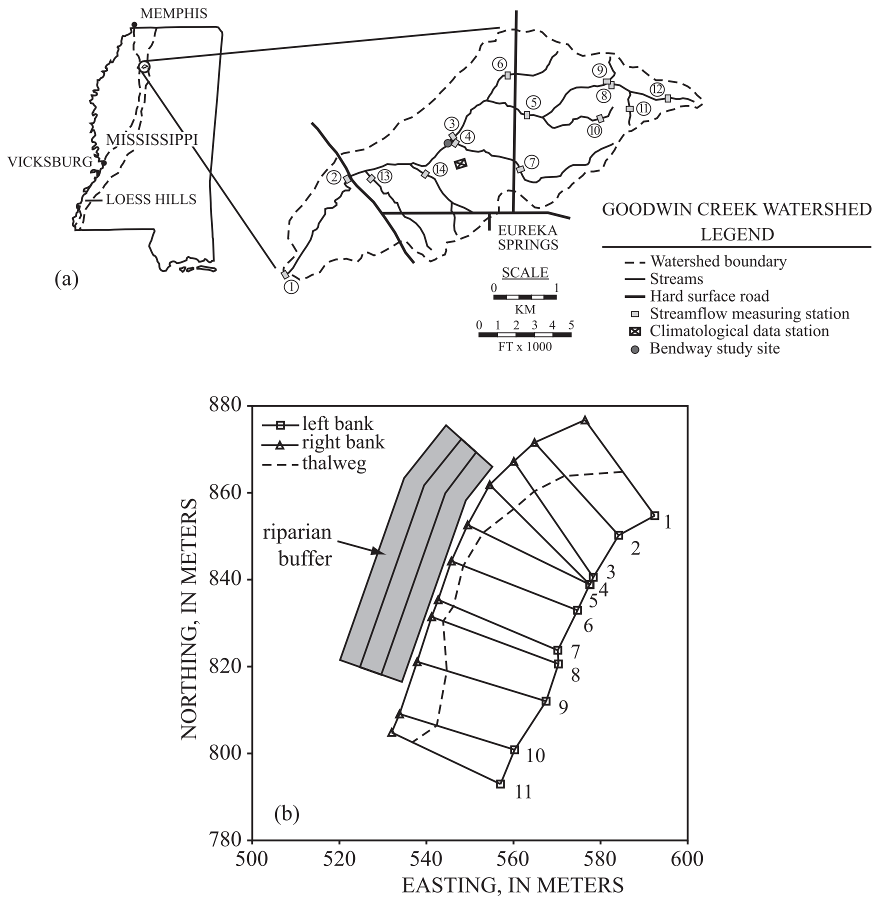

Figure 6. Goodwin Creek Bendway study site (89°52′W, 34°15′N): (a) location map and (b) plan view showing surveyed cross-section locations.

from 26 to 32 m approximately. Planar and cantilever failures were relatively common along the steepest section of the 4.7 m high banks. Cantilevers were formed by (1) preferential erosion of sands and silts by fluvial undercutting about 3.0 to 3.5 m below the top bank and (2) by sapping and small pop-out failures in the region of contrasting permeabilities of the stream bank material about 1.6 to 2 m below the top bank. It was observed that the loss of matric suction from infiltrating precipitation and subsequent seepage significantly contributes to mass bank instability [*Simon et al.*, 2000].

Bank material consists of about 2 m of moderately cohesive, brown clayey-silt of late Holocene (LH) age overlying 1.5 m of early Holocene (EH) gray, blocky silt of considerable cohesion and lower permeability, which perches water. These units are separated by a thin (0.1 to 0.2 m) layer containing manganese nodules. These materials overlie 1 m of sand and 1.5 m of packed (often weakly cemented) sandy gravel. Cohesion and friction angle were measured in situ with effective cohesion values ranging from 0 to 6.3 kPa. Core samples were also analyzed for bulk density, porosity, and particle size distribution.

Pore water pressure data were collected using tensiometers along the right bank of the bendway at (1) an open plot (short cropped turf/bare) since December 1996; (2) a mature riparian tree stand (a mixture of sycamore (*Platanus occidentalis*), river birch (*Betula nigra*), and sweetgum (*Liquidambar styroflora*)) since July 1999; and (3) an eastern gamagrass (*Tripsacum dactyloides*) buffer since December 1999 [*Simon and Collison*, 2002]. Data were recorded every 10 min at depths of 30, 100, 148, 200, and 270 cm (corresponding to different layers within the bank profile). For model comparison, these data were time-averaged over a 24 hour (daily) interval.

5.3.2. Simulation results. The effect of the riparian tree stand and gamagrass buffer on stream bank erosion was simulated for the period of January 1996 to September 2003. The riparian buffer in both scenarios had a width of 15 m (three

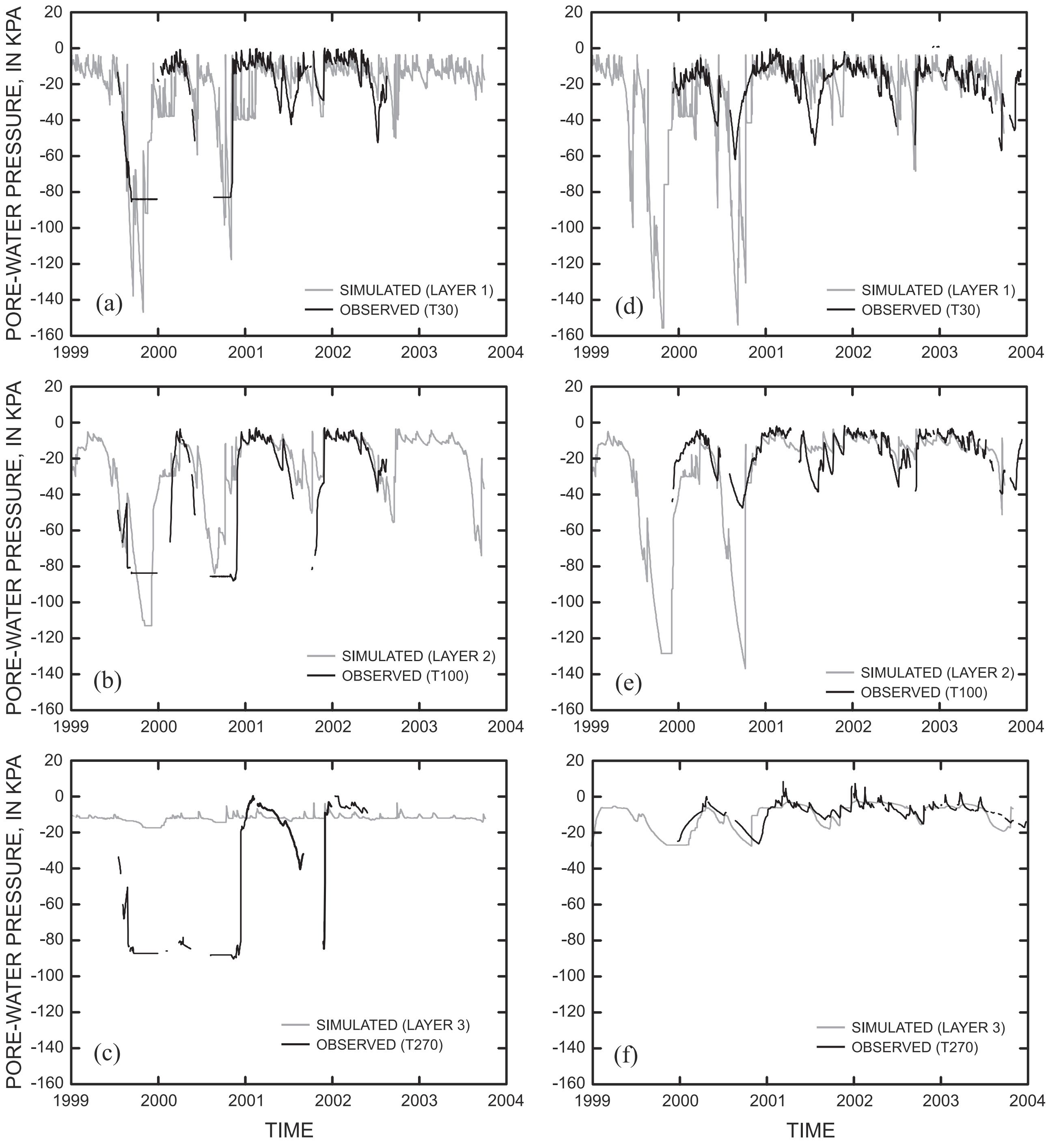

Figure 7. Comparison of simulated and observed pore water pressures (PWP) within the right bank of the Goodwin Creek Bendway study site for (a–c) a deciduous tree stand and (d–f) an eastern gamagrass buffer. Figures 7a and 7d compare the simulated PWP in layer 1 (0–0.5 m) to the observed tensiometer data at a depth of 0.3 m. Figures 7b and 7e compare the simulated PWP in layer 2 (0.5–1.7 m) to the observed tensiometer data at a depth of 1.0 m. Figures 7c and 7f compare the simulated PWP in layer 3 (1.7–3.2 m) to the observed tensiometer data at a depth of 2.7 m.

zones of 5 m) and four layers (two layers spanning the LH unit, one layer spanning the EH unit, and a fourth layer representing the sand unit). The properties of the trees at the start of the simulation were height of 21 m, root depth of 1.0 m, a biomass of coarse roots of 48,000 kg ha^{-1}, and a biomass of fine roots of 15,500 kg ha^{-1} (mean RAR ≈ 1%). The properties of the grass at the start of the simulation were height of 0.1 m, root depth of 1.0 m, and biomass of fine roots of 4000 kg ha^{-1} (mean RAR ≈ 0.1%). The biomass values of fine roots are suitable values for woody and herbaceous riparian buffers along Goodwin Creek.

The temporal and spatial distributions of pore water pressure reflect the effects of infiltrating rainfall and evapotranspiration (Figure 7). For the grass buffer, the simulated pore water pressures agree well with those observed in the LH and EH layers (Figure 7d, 7e, and 7f). Peak suction values in the fall and the temporal variation of pore water pressure are accurately simulated, except for the fall of 2000 where suction values are overpredicted in the LH unit (Figure 7d and 7e). For this time period, the planted grasses were in their first year of development, whereas they were already well established in the model simulation. For the riparian tree stand, the simulated pore water pressure distribution agrees well in the LH unit (Figure 7a and 7b) but does not compare well in the EH unit (Figure 7c).

Figure 8a compares the simulated increase in channel top width for the two riparian buffer scenarios to that observed. The woody buffer greatly reduced stream bank erosion by preventing any planar failures. The anchoring effects of coarse roots in the upper 1 m of the stream bank significantly increased factor of safety, though undercutting of the stream bank produced some cantilever failures along the central part of the bendway, leading to near vertical stream banks at the end of the simulation (Figure 8b). With progressive undercutting, the bank will eventually fail in case of the riparian tree stand.

Simulated top-bank retreat for the gamagrass buffer is similar to that observed. The added cohesion due to the grass roots did not noticeably contribute to total shear strength due to the height of the stream bank with respect to rooting depth. That is, only the soil shear-strength along the top 1 m of the failure plane is affected by the grass roots. Further, the grass buffer does not have a coarse root system that can act as anchors.

To summarize, the deciduous tree stand significantly reduces stream bank erosion rates. However, the simulation period is too short to accurately calculate the reduction percentage. The effect of the eastern gamagrass buffer on the rate of stream bank erosion is negligible. The ratio of the rooting depth to bank height (<0.5) in combination with the absence of a coarse root system minimizes any contributions of the gamagrass buffer to the stability of the stream bank. This modeling exercise shows that for the Goodwin Creek Bendway, a coarse rooting system, e.g., as provided by trees, may significantly reduce bank erosion rates. The failure of the gamagrass buffer to reduce erosion rates shows that the hydrologic benefits, that is a reduction in pore water pressure provided by vegetation, is of secondary importance to the long-term rate of stream bank retreat for the studied incised stream.

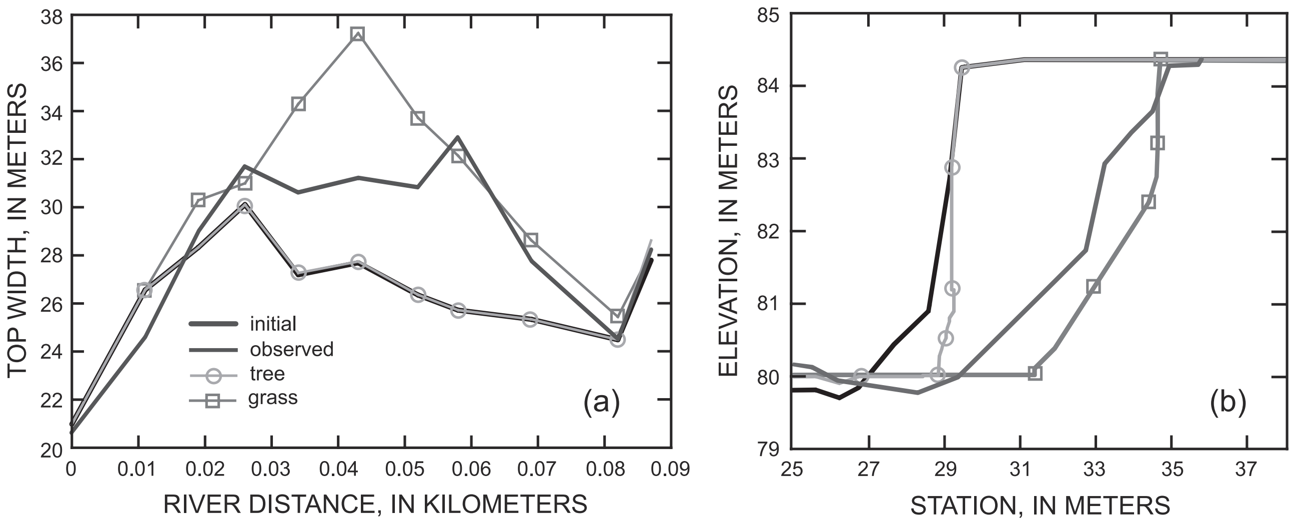

Figure 8. Comparison of simulated bank retreat at the Goodwin Creek Bendway study site between January 1996 and September 2003 for the two vegetative treatment scenarios against those observed: (a) change in channel top width and (b) stream bank erosion at the cross section located at river distance 0.058 km.

6. SUMMARY

The channel evolution model CONCEPTS and the riparian zone management model REMM were developed to (1) simulate the long-term evolution of incised channel systems, (2) evaluate the effectiveness of stream restoration designs, and (3) assess management decisions to control nonpoint source pollution in the riparian zone. The models simulate the processes and controlling factors that shape streams: hydraulics, sediment transport and bed adjustment, stream bank erosion, and riparian zone hydrology and plant growth.

Restoration measures to control streambed and stream bank erosion are represented by adjusting bed and bank material properties. Three example applications of the model at the stream corridor scale demonstrated its capabilities to evaluate stream restoration measures that stabilize streambeds and stream banks or the evolution of newly constructed channels to replace highly disturbed existing channels.

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Practical Considerations for Modeling Sediment Transport Dynamics in Rivers

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Sediment transport dynamics are some of the most important aspects to consider in river restoration and management projects. Restoring a river usually involves the manipulation of its flow conditions, channel cross sections, channel alignment, sediment supply, bed material composition, and riparian conditions, all of which directly or indirectly affect sediment transport dynamics. Because a river will be reshaped through sediment transport process following restoration, a lack of or an inadequate consideration of postrestoration sediment transport dynamics may result in poor performance or failure of the project. Here we discuss some practical considerations in sediment transport modeling as a guide for resource managers overseeing river restoration projects as well as sediment transport practitioners. The discussion is not intended as a "how to" guide or a thorough review of the scientific literature pertaining to sediment transport. Instead, the project examples discussed herein are intended to illustrate some of the lessons learned from our experiences in conducting sediment transport analyses for applied projects. The examples are not necessarily river restoration projects, but the practical considerations discussed should generally apply to any sediment transport analysis, including those for river restoration projects.

1. INTRODUCTION

One of the common misunderstandings in sediment transport analysis is the belief that more complicated tools and methodology yield more accurate and more dependable results. While more complicated methods and models usually produce more detailed results, they do not necessarily produce more accurate or reliable results due to limitations of sediment transport theory, the stochastic nature of sediment transport, and often a limited understanding of the system to be analyzed. Moreover, a more complicated methodology or model will inevitably require more input data, which usually introduces additional uncertainty associated with the input data. As a result, it is not uncommon for an excessively complicated model to produce less satisfactory results than a simpler model for conditions where the simpler model can still achieve the project goal. Thus, matching the approach for modeling sediment transport to the project goals is an important first step in an analysis. Two of the project examples

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presented in this chapter are devoted to demonstrate the importance of selecting appropriate methods and models: (1) an analysis of sediment delta progradation in Slab Creek Reservoir as an example where a very simple approach was sufficient and (2) the simulation of fine-sediment dynamics in the gravel-bedded Lagunitas Creek as an example of applying a more complex model. While selecting an appropriate method and tool is the first step toward a successful sediment transport analysis, it is also imperative to have a good understanding of the limitations of different methods and tools. This understanding will help the modeler make the correct decisions during modeling, provide accurate interpretations of modeling results, and recommend appropriate contingency plans to lower the risks associated with the uncertainties. We present the numerical modeling of sediment pulse dynamics in a flume with forced pool-riffle morphology to demonstrate limitations of one-dimensional (1-D) numerical models. We then present a sediment transport study for Marmot Dam removal on the Sandy River, Oregon, to demonstrate some of the important practical considerations in sediment transport modeling, including boundary conditions, the zeroing process as part of model calibration, different approaches to address uncertainties, and to further our discussion on the application of multidimensional numerical and scaled physical models. Finally, we provide a brief discussion of generic flume experiments as a useful but often ignored tool to understand sediment transport dynamics using two examples.

2. SLAB CREEK RESERVOIR DELTA PROGRADATION

2.1. Project Background

Slab Creek Reservoir is located on the South Fork American River in California, impounded by a 71 m dam that raises water levels for power generation. The owner of the reservoir is designing a 400 MW pumped-storage facility as part of a development project that includes the construction of an upper storage reservoir (Iowa Hill storage). Upon completion, water will be pumped from Slab Creek Reservoir into the upper storage reservoir during low power demand periods and released back into Slab Creek Reservoir for power generation during peak energy demand. As part of the permitting process, stakeholders wanted to know whether the repeated water pumping from and releasing into Slab Creek Reservoir would produce persistent high turbidity events. The key to this question is how fast the deltaic front of the sediment deposit in Slab Creek Reservoir will advance during the lifespan of the project because pumping-related turbidity will only potentially occur if the Slab Creek deltaic front reaches the vicinity of the pump intake. While stakeholders were interested in developing a 1-D numerical model to answer the question, our examination indicated that a much simpler mass conservation analysis based on basic physical principles would achieve the project goal without the time and expense of additional field data collection. The results are briefly discussed below while a detailed description of the analysis is given by *Stillwater Sciences* [2008] (accessed 9 July 2010).

2.2. Analysis and Results

Slab Creek Reservoir bathymetry with reasonable resolution is available for 1992 and 2007 (thalweg elevations of the two data sets are presented in Figure 1). The analysis required identification of three deltaic features shown in the longitudinal profile presented in Figure 1 (labeled as “A,” “B,” and “C” in Figure 2). The feature labeled “A” is not a delta deposit but rather an old dam (American River Intake Dam) that was submerged upon the completion of Slab Creek Dam. The deposit upstream of American River Intake Dam indicates that it was completely filled with sediment prior to the construction of Slab Creek Dam, which is reasonable given its relatively small size. The sediment deposited upstream of American River Intake Dam prior to the construction of Slab Creek Dam is indicated in the shaded area in Figure 2, and has a topset slope of 0.0016 (the dashed line labeled “D” in Figure 2), which is identical to the topset slope of the deltaic deposit marked “C” (discussed below). This slope represents the equilibrium slope of the sediment deposit if the reservoir pool level is kept within the normal range of operating conditions (i.e., minimal reservoir drawdown). The deltaic front (i.e., foreset bed) labeled “B” is considerably lower than the normal drawdown pool level. This deltaic deposit was most likely formed during the last week of October 1991 when the

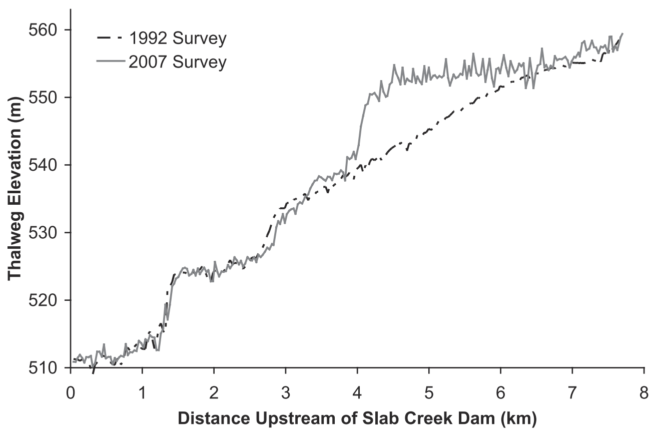

Figure 1. Thalweg elevation within Slab Creek Reservoir, surveyed in 1992 and 2007. Slab Creek Dam, located at 0 km in the diagram, is 71 m tall and was constructed in 1967.

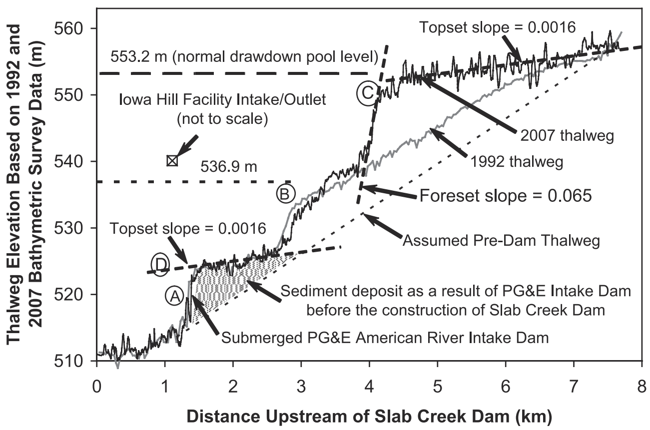

Figure 2. Characteristic features used to estimate long-term sediment supply to Slab Creek Reservoir and to produce a mass conservation model to estimate future advancement of the delta front of the reservoir deposit. Submerged American River Intake Dam is indicated at point A; sediment deposit attributed to the October 1991 drawdown event is indicated at point B; 2007 delta front, with a foreset slope of 0.065 and a topset slope of 0.0016, is shown at point C; and the topset of American River Intake Dam deposit is indicated at point D, which is identical to that of the 2007 deposit shown at point C. Also depicted are the approximate location and elevation of the proposed intake/outlet structure of the Iowa Hill Facility as well as the normal pool elevation of Slab Creek Reservoir.

reservoir was lowered to a pool level of 536.9 m due to an outage at an upstream power plant. This represents approximately a 15 to 18 m drawdown from normal operation that presumably mobilized and transported the sediment deposit previously stored further upstream to form the odd-shaped deposit labeled "B." Because the drawdown did not last long enough for the deposit to reach an equilibrium configuration, the resulting sediment deposit upstream of deltaic front "B" is considerably steeper than the equilibrium slope of 0.0016. The deltaic front "C" is interpreted as the deposit formed after the October 1991 event, as subsequent operational rules were implemented to maintain the Slab Creek Reservoir above 551.7 m at all times.

While topography of the Slab Creek Reservoir area prior to dam construction is not available, a predam thalweg elevation can be reasonably estimated by connecting the bottom of the American River Intake Dam and the upstream end of the current sediment deposit with a straight line (Figure 2). The longitudinal information shown in Figure 2, in combination with a cross-sectional area to depth relation developed based on a typical cross section located approximately 0.8 km upstream of Slab Creek Dam where sediment deposition is minimal, was used to derive Slab Creek Reservoir sedimentation volumes and rates (Table 1).

Note in Table 1 that the Slab Creek sedimentation rate for the period of 1992 to 2007 is significantly higher than the period of 1967 to 1992. This is, in part, attributable to a large landslide that occurred in Mill Creek, a tributary to South Fork American River, on 24 January 1997 that delivered a significant volume of sediment to the main stem [*Sydnor*, 1997] (accessed July 2010). The majority of this sediment pulse quickly transported into the Slab Creek Reservoir and subsequently elevated the sedimentation rate for the period of 1992 to 2007. An average sedimentation rate of 29,000 m^3 yr^{-1} for the period of 1967 to 2007 (includes the high sediment production from the 1997 landslide event) was used to estimate the future advancement of the deltaic front. This analysis was simply accomplished by drawing two straight lines that represent the future topset and foreset locations, then calculating the volume below the two lines and dividing it by the sedimentation rate to obtain the time needed for the sediment deposit to reach this level. Based on the 2007 profile, the sediment deposit in Slab Creek Reservoir has a foreset bed slope of 0.065 and a topset bed slope of 0.0016, and the foreset/topset break point is located approximately 2 m below the normal pool level. The predicted deltaic front advancement using the above information is presented in Figure 3, indicating that the deltaic front will not reach the intake within the facility design life of 100 years, and thus pumping operations within the reservoir are not expected to produce turbidity spikes.

2.3. *The Alternative: Predicting Deltaic Front Advance With a 1-D Numerical Model*

A 1-D numerical model would have accomplished the same goal but with significantly more effort without gaining additional confidence in the results. To estimate the reservoir sedimentation rate for input data to run a 1-D numerical model, one would have conducted the same or similar exercises as discussed above. Additional efforts would have

Table 1. Estimated Sediment Accumulation Rate in Slab Creek Reservoir Based on Topographic Survey Data in 1992 and 2007

Period	Bulk Volume of Sediment Deposited During the Period (m^3)	Sediment Accumulation (Bulk Volume) Rate (m^3 yr^{-1})
Pre-Slab Creek Dam[a]	176,000	NA
1967–1992	577,000	22,000
1992–2007	593,000	37,000
1967–2007	1,170,000	29,000

[a]Amount of sediment accumulated behind American River Intake Dam.

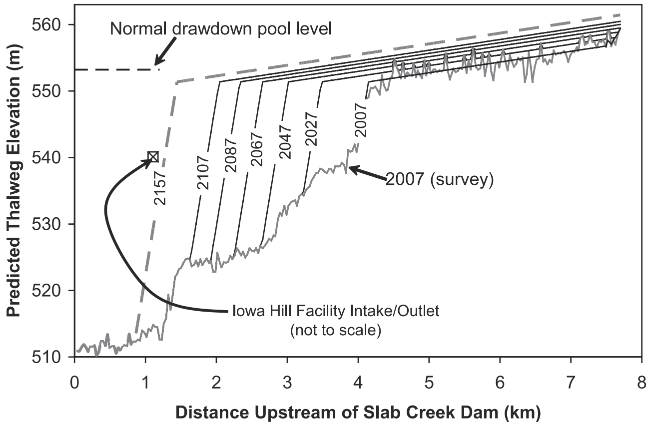

Figure 3. Predicted future delta advancement within Slab Creek Reservoir using a simple mass conservation model. These results indicate that the delta front will not reach the proposed intake in the designed project lifetime of 100 years as the delta front is predicted to be almost 1 km upstream of the intake in 150 years (i.e., year 2157).

included: collection of sediment samples for grain size analysis, analysis of discharge records to select typical hydrologic years for model input, numerical model set up, and a rigorous model calibration to reproduce the observed sediment deposit's volume and topset and foreset slopes. However, due to limitations in sediment transport theory, it is likely that the observed foreset slope could not have been replicated with a numerical model, and the final calibrated model would only have matched the volume of sediment deposit and the topset slope. Because the derivation of sedimentation rate is similar or identical to the simple method discussed earlier, and model calibration also tries to replicate the observed deposit (i.e., to match topset and foreset slopes), the results of a 1-D numerical model would at best have the same confidence as the simple method. As a result, the simple mass conservation exercise presented earlier is the most appropriate method for this particular project.

3. LAGUNITAS CREEK FINE-SEDIMENT DYNAMICS

3.1. Project Background

Lagunitas Creek, located in Marin County, California, provides regionally important habitat for coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*). However, the watershed's high fine-sediment yield (i.e., sand and finer (<2 mm)) potentially causing excessive fine-sediment deposition that reduces the survival rate of the salmonid eggs is a concern for resource managers. If a numerical model is used to examine how the fraction of fine sediment within spawning gravel deposits will change under various potential measures for fine-sediment reduction, it must have the ability to simulate the transport dynamics of both coarse (i.e., gravel and coarser (>2 mm)) and fine sediment and the interaction between the two size fractions and be able to track the fraction of fine sediment in gravel deposit through time. A 1-D sediment transport numerical model called The Unified Gravel Sand (TUGS) model [*Cui*, 2007a] has the ability to simulate these criteria and was utilized to predict potential outcomes under different approaches to reduce fine sediment.

In addition to equations that govern the flow of water in river channels and the particle size-based Exner equations of sediment continuity (including the abrasion of gravel during transport), the fundamental components of TUGS model include the surface-based bed load equation of *Wilcock and Crowe* [2003] that links the local sediment transport capacity to the local boundary shear stress, the gravel transfer function of *Hoey and Ferguson* [1994] and *Toro-Escobar et al.* [1996] that links the subsurface and surface gravel grain size distribution with the bed load, a sand transfer function that links the sand fraction in the subsurface to that on the bed surface, and relations for sand entrainment and infiltration into the subsurface bed material. Specific details of TUGS model can be found in the work of *Cui* [2007a] and case studies demonstrating satisfactory results of TUGS model application in other projects are available in the works of *Cui* [2007b] and *Gomez et al.* [2009]. TUGS model is significantly more complicated than computer models developed by the same author and his colleagues used for other purposes (e.g., DREAM-1 and DREAM-2, as presented in the work of *Cui et al.* [2006a, 2006b]) because it simulates the interaction between coarse and fine sediments. Unsurprisingly, the corollary of this added complexity is that TUGS model is more difficult to set up and requires more input data, some of which is often impractical to obtain (e.g., grain size distribution of sediment supply) and has to be assumed during the model calibration process based on more readily available data (such as surface or subsurface grain size distribution). The increased complexity is necessary for this particular project because without these model capabilities, simulating how the fraction of fine sediment in the bed changes through time would not be possible.

3.2. Analysis and Results

The following data were available and were used either as model input or for model examination/calibration: (1) daily discharge records from two stations, (2) a sediment budget analysis that provided estimated sediment supply at each tributary junction within the study reach. (3) a longitudinal profile of the river, including locations of nonerodible, geologic, and anthropogenic controls such as bedrock outcrops and concrete weirs, (4) bankfull channel width estimated in the field during the longitudinal profile survey, and (5) surface grain size distributions at various locations obtained through pebble counts and bulk samples.

Of particular note, TUGS requires a comprehensive grain size distribution of the sediment supply, which is impractical to obtain in many projects including this one, and a reasonable estimate of an abrasion coefficient, which predicts how fast gravel particles will break down into finer particles while transported downstream. The grain size distribution of the sediment supply and a gravel abrasion coefficient were derived during a "zeroing process," where assumptions were made with these unavailable parameters and adjusted iteratively until the model reproduced key parameters observed in the field, including the longitudinal profile and surface grain size distribution. Further discussion of the importance of the zeroing process is provided later in this chapter. A comparison of simulated and observed grain size distributions of the postzeroing model is provided in Figure 4, indicating that predicted surface median size (D_{50}) (Figure 4b), surface D_{84} (Figure 4c), surface sand fraction (Figure 4d) generally fell within the measured range, while surface D_{16} (Figure 4a) was underpredicted. The level of agreement between comparisons shown in Figure 4, especially the key sand fraction result (Figure 4d), is generally acceptable in sediment transport modeling exercises, and the model with postzeroing input data was used to simulate fine-sediment fractions in the gravel bed under different measures. Among the measures, the most practical one is to augment clean spawning gravel into one of Lagunitas Creek's tributaries as a measure

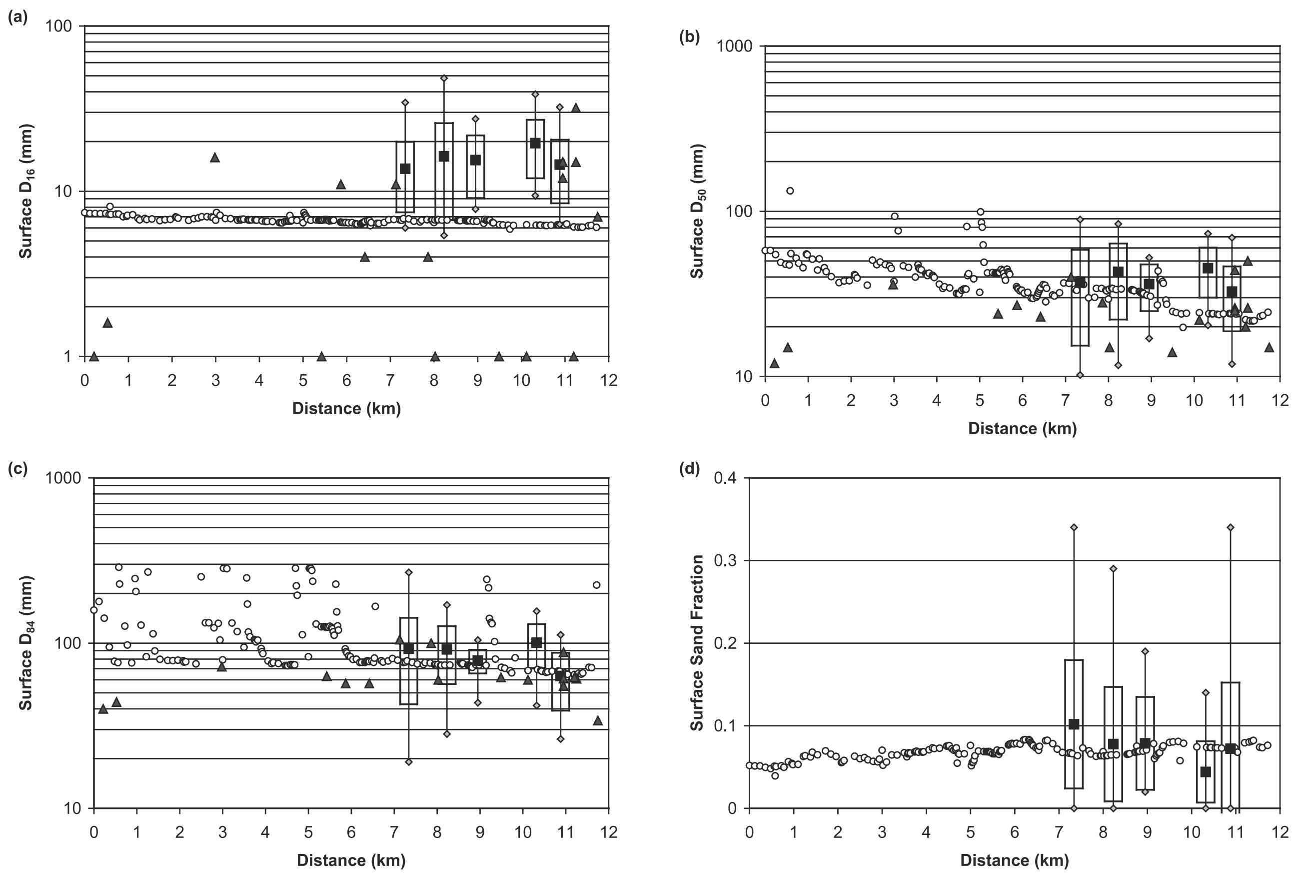

Figure 4. Simulated surface characteristic grain size and surface sand fraction under current conditions in comparison with field observations: (a) surface D_{16}, (b) surface D_{50}, (c) surface D_{84}, and (d) surface sand fraction. Simulated TUGS model results are depicted with open circles. Solid triangles are pebble count results by M. O'Connor (personal communication, 2006) and Stillwater Sciences staff in 2008. Additional field data are bulk samples from Balance Hydrologics (2008): the solid squares are mean values, the diamonds are the maximum and minimum values, and the large open rectangular boxes represent mean value ±1 standard deviation.

to decrease the fraction of fine sediment in the sediment supply. Simulated surface sand fraction averaged over the study reach downstream of the gravel augmentation point with various rates of gravel augmentation is presented in Figure 5 in comparison with the current condition (0 t yr^{-1} gravel augmentation), indicating decreased fine-sediment fraction with increasing rate of gravel augmentation. The degree of fine-sediment reduction, however, is rather small for the range of gravel augmentation rates examined. While a final decision as to what action to implement in Lagunitas Creek to reduce the amount of fine sediment on channel bed has not been made, the modeling exercises at least provide some idea to management agencies as to whether the examined measures will be effective, and thus, potentially eliminating some possible trial-and-error actions prevalent in many river management or restoration projects.

4. ONE-DIMENSIONAL MODELING OF SEDIMENT TRANSPORT DYNAMICS IN A FLUME WITH FORCED POOL-RIFFLE MORPHOLOGY

One-dimensional numerical sediment transport models are widely used for sediment transport evaluations in rivers due to their relative simplicity compared to other tools such as multidimensional numerical and scaled physical models. Implicit in their formulation, 1-D numerical sediment transport models are not capable of simulating detailed local topographic features such as pools and riffles in rivers, and their applications generally involve extended river reaches over a long period of time [e.g., *Thomas and Chang*, 2008; *Spasojevic and Holly*, 2008; *Cui et al.*, 2008]. As a result, the spatial resolution of 1-D numerical sediment transport models is generally one wave length of the dominant features (e.g., a pool-riffle sequence) or longer, usually implying distances on the order of several channel widths [*Cui et al.*, 2008]. As a consequence, 1-D numerical sediment transport model results pertaining to sediment transport characteristics at a scale smaller than several channel widths should usually be viewed as extrapolations beyond model resolution. Because 1-D numerical models average parameters over the entire cross section, any results describing how a particular cross section changes (e.g., amount of erosion or deposition near a particular bank) should also be viewed as beyond the resolution of the model, even though many 1-D numerical models provide such detailed results in their outputs. Moreover, because of the inherent uncertainties associated with input parameters such as sediment supply and future hydrologic conditions, the temporal resolution of most 1-D model applications is generally on the order of a year or more, unless a model is specifically set up to examine a particular event. Thus, 1-D numerical sediment transport models must be applied and interpreted on a reach-averaged and time-averaged basis. An informative demonstration of the reach-averaged nature of 1-D numerical sediment transport modeling is the numerical simulation of a series of flume experiments given by *Cui et al.* [2008], summarized below.

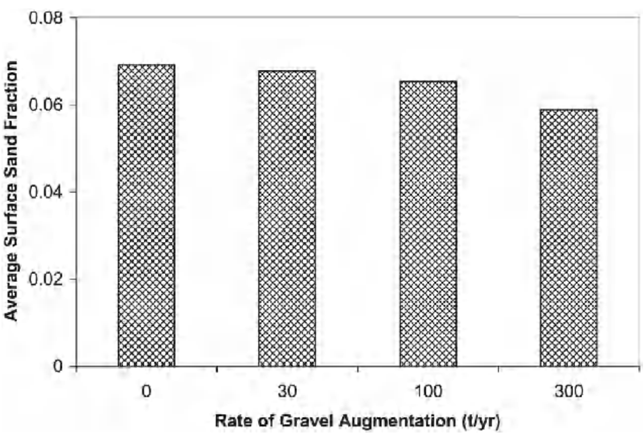

Figure 5. Simulated surface sand fraction averaged over the study reach downstream of the gravel augmentation point, indicating a slightly decreased surface sand fraction with increasing rate of gravel augmentation.

The experiments were conducted in a 0.86 m wide, 28 m long flume at the Richmond Field Station (RFS) of the University of California, Berkeley, to examine coarse- and fine-sediment pulse movement in an armored gravel-bedded flume with forced pool-riffle morphology (Figures 6a, 6b, and 6d) (Figure 6c will be discussed later). To create an equilibrium sediment deposit, gravel (4.2 mm median grain size) was fed into the flume at a constant rate of 40 kg h^{-1}, along with a constant discharge of 20 L s^{-1} until an equilibrium profile was realized. Prior to the experiment, sand bags were placed in the flume at a frequency of approximately five channel widths, alternating between left and right banks to force the formation of pool-riffle morphology. Once an equilibrium deposit was formed, evidenced by the observation that the rate of sediment exiting the flume became approximately the same as the sediment feed rate, the sediment feed was terminated but the constant water discharge continued, until a new equilibrium condition was reached, determined by negligible sediment exiting the flume. This new equilibrium deposit had an armored surface layer and is similar to the condition of a gravel-bedded river downstream of a large dam, where gravel supply is trapped in the reservoir upstream of the dam. Following the formation of this armored equilibrium deposit, coarse- and fine-sediment pulses were introduced to the flume, and their evolution was monitored by measuring bed surface elevation and sediment transport rate at the exit of the flume. The experimental data were used

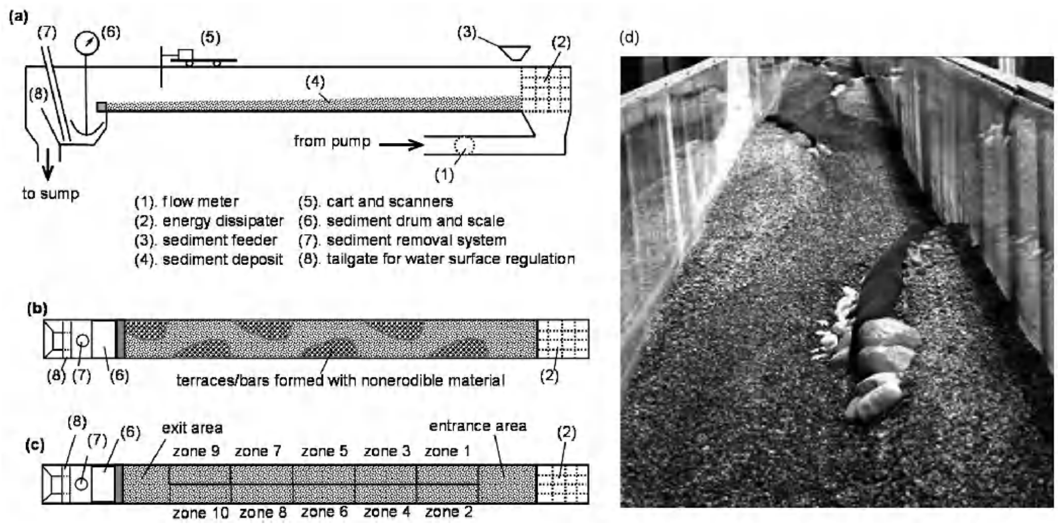

Figure 6. Experimental flume Richmond Field Station, the University of California, Berkeley, used for two generic experiments: (1) sediment pulse dynamics in rivers with pool-riffle morphology and (2) fine-sediment infiltration into gravel deposit. (a) Sketch of the flume and its associated facilities. (b) Plan view of the setup for sediment pulse experiments. (c) Plan view of the setup for fine-sediment infiltration experiments. (d) Photograph showing sediment pulse experiment, looking upstream.

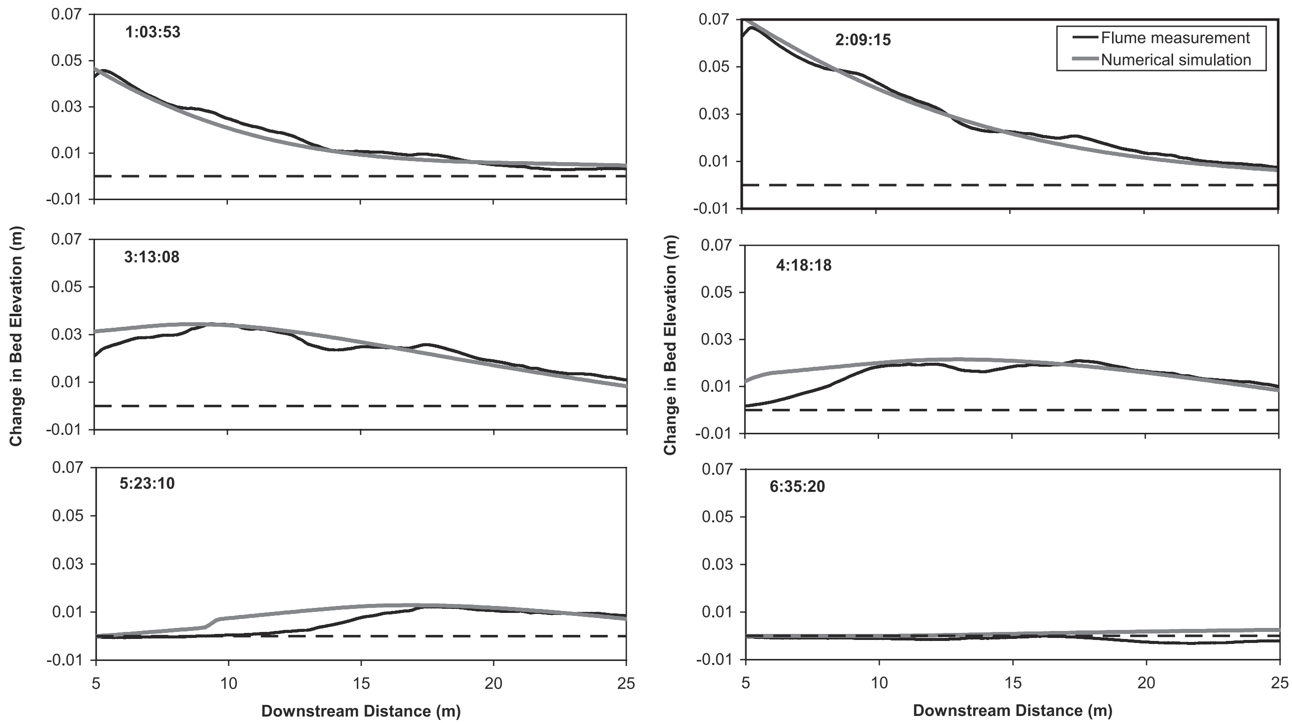

Figure 7. Comparison of measured and simulated change in reach-averaged bed elevation relative to the initial reach-averaged bed profile for run 7 (large fine-sediment pulse run). Time steps in the diagrams reference time relative to the start of sediment pulse feed, in hours:minutes:seconds. DREAM-1 was used for numerical simulation. Diagram adapted from the work of *Cui et al.* [2008], reprinted with permission from ASCE.

to determine whether large fine- and coarse-sediment pulses in rivers would result in an oversimplified channel bed (i.e., a less complex channel with flatter bed), presented in more detail later in this chapter where generic flume experiments are discussed.

The data were also used by *Cui et al.* [2008] to examine the performance of two 1-D numerical models, DREAM-1 and DREAM-2 [*Cui et al.*, 2006a, 2006b], for simulation of sand-sized and gravel-sized sediment transport, respectively, focusing on how the models performed with respect to the pool-riffle morphology. First, due to the reach averaged nature of 1-D sediment transport numerical models, *Cui et al.* [2008] did not use the surveyed initial bed profile, which is rather undulating due to the presence of pool-riffle sequences, as the initial condition but instead used a planer bed with a slope identical to the surveyed slope averaged over five channel widths (one wavelength of the pool-riffle morphology). This practice produced a good match between numerical simulation and experimental data for both models even though DREAM-1 was uncalibrated, and DREAM-2 was calibrated simply by adjusting one coefficient within the model so that it reproduced the observed bed slope under 40 kg h^{-1} sediment feed rate and 20 L s^{-1} discharge. Comparisons of numerical modeling results and flume observations for two runs, one each for DREAM-1 and DREAM-2, are presented in Figures 7 and 8, respectively, both indicating good agreement between observations and predictions. Then, to demonstrate that trying to use 1-D numerical models to produce results at a scale finer than a reach-averaged resolution would produce undesirable if not completely invalid results, *Cui et al.* [2008] also simulated the same two runs using the surveyed thalweg profiles as the initial profile input to the models (Figures 9 and 10). Results in Figures 9 and 10 indicate that the models poorly reproduced the channel aggradation and degradation at most locations, especially in the area of pools, although the simulated general patterns of sediment pulse movement are visible and bear some similarities with the observations. Further analysis of the simulation results indicate that the median errors in bed elevation for DREAM-1 and DREAM-2 simulations are approximately 8 and 2 times higher, respectively, for runs where the observed thalweg elevations were used directly as model initial conditions compared to using a planer bed, which represents the reach average of the observed bed elevation.

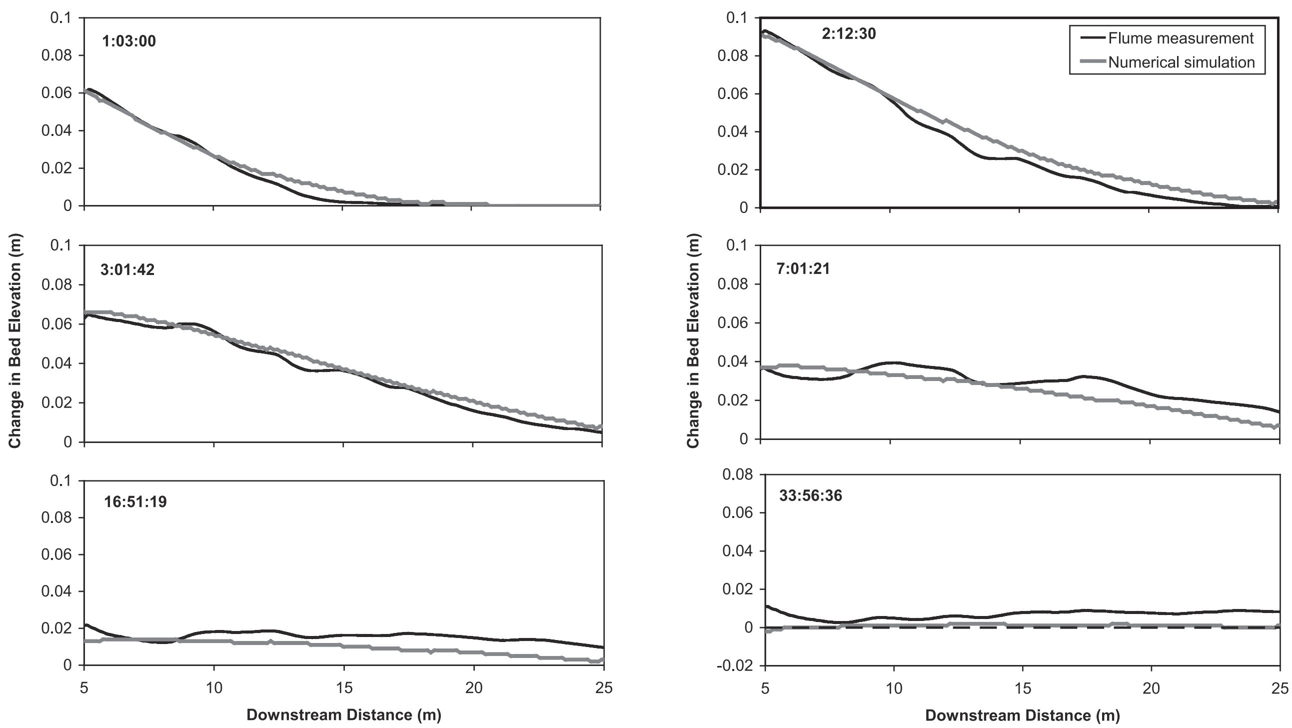

Figure 8. Comparison of measured and simulated change in reach-averaged bed elevation relative to the initial reach-averaged bed profile for run 8 (the large coarse pulse run). Time steps in the diagrams reference time relative to the start of sediment pulse feed, in hours:minutes:seconds. DREAM-2 was used for numerical simulation. Diagram adapted from the work of *Cui et al.* [2008], reprinted with permission from ASCE.

5. MARMOT DAM REMOVAL SEDIMENT TRANSPORT STUDY, SANDY RIVER, OREGON

5.1. Project Background

The 14 m tall Marmot Dam was located on the Sandy River, Oregon approximately 48 km upstream of its confluence with Columbia River. Based on economic and environmental considerations, Portland General Electric (PGE), the owner of the dam, decided to remove the dam and decommission the associated hydropower project, which reestablished continuity for physical and biological processes throughout the system including fish passage for three listed species of anadromous salmonids. In order to obtain a permit for dam removal and select an appropriate removal alternative, a pair of 1-D numerical sediment transport models was developed in 1999 to understand the fate of approximately 750,000 m^3 of gravel and sand deposited upstream of Marmot Dam during its 80+ years of operation (see *Stillwater Sciences* [2000] (accessed May 2010) and *Cui and Wilcox* [2008]). In 2002, stakeholders agreed on the alternative that removed the dam in a single season with minimal sediment excavation, at least in part based on the modeling results. This option, referred to as "blow-and-go" alternative hereafter, involved releasing almost all of the sediment stored upstream of the dam to downstream reaches, making it the most economical removal alternative considered as well as the one with the highest potential for causing downstream impacts. Downstream concerns included potential burial of spawning habitat, blockage of secondary channels, and simplification of channel geometry that could potentially hamper upstream

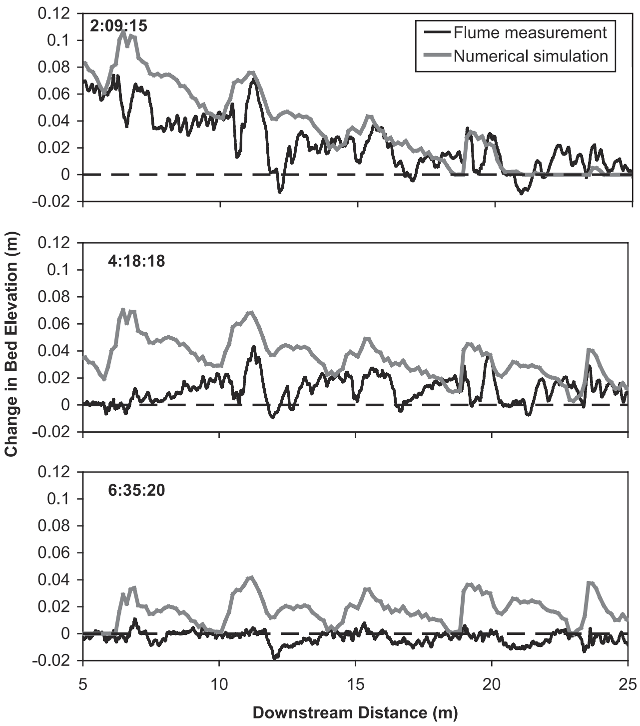

Figure 9. Comparison of measured and simulated change in bed elevation for run 7 (the large fine-sediment pulse run) without reach averaging, demonstrating decreased model performance following mishandling of the initial condition relative to the reach-averaged results presented in Figure 7. Numerical simulation used the initial thalweg elevation without averaging as model input, and the measured change in bed elevation is calculated based on the surveyed thalweg elevation data. Time steps in the diagrams reference time relative to the start of sediment pulse feed in hours:minutes:seconds. DREAM-1 model was used for simulation. Diagram adapted from the work of *Cui et al.* [2008], reprinted with permission from ASCE.

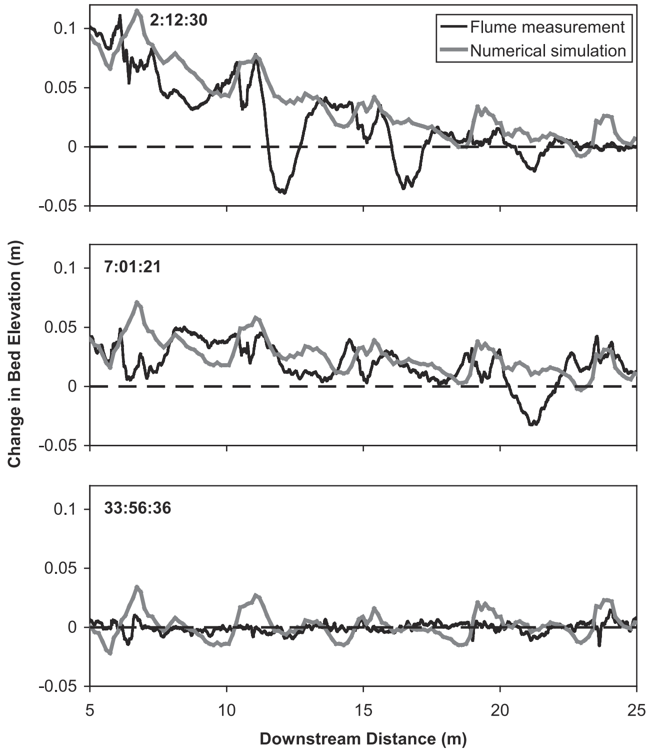

Figure 10. Comparison of measured and simulated change in bed elevation for run 8 (the large coarse-sediment pulse run) without reach averaging, demonstrating decreased model performance following mishandling of the initial condition relative to the reach-averaged results presented in Figure 8. Numerical simulation used the initial thalweg elevation without averaging as model input, and the measured change in bed elevation is calculated based on the surveyed thalweg elevation data. Time steps in the diagrams reference time relative to the start of sediment pulse feed in hours: minutes:seconds. DREAM-2 model was used for simulation. Diagram adapted from the work of *Cui et al.* [2008], reprinted with permission from ASCE.

migration of adult salmonids, fine-sediment deposition in spawning habitat, and sediment deposition in the delta area (Sandy River confluence with Columbia River) that might block adult salmonids from entering the Sandy River.

Modeling results, however, indicated that major coarse-sediment deposition would occur only within the first few kilometers downstream of the dam and within a short distance downstream of the Sandy Gorge approximately 8 km downstream of the dam, and fine sediment would pass through most of the Sandy River with little deposition except within a few kilometers of the confluence with the Columbia River where the river is already sand bedded under current conditions. In addition, modeling results indicated that multiple-year staged removal would provide no advantage over the blow-and-go alternative and that dredging a portion of the stored sediment during one dry season would provide only minimal benefit over the blow-and-go alternative (i.e., minimal reduction in the thickness of sediment deposition downstream of the dam).

As a result, the blow-and-go option was implemented in the summer of 2007, and the cofferdam protecting the working area and preventing erosion of the sediment deposit was breached in the fall following the first storm event of the season. Figure 11 details simulated erosion and deposition processes in the Sandy River upstream and downstream of Marmot Dam following dam removal in comparison with field data collected after dam removal, indicating that field observations generally fall within the range of numerical model predictions. The model also predicted a low daily-averaged concentration of total suspended solids (TSS) with a maximum increase in daily averaged TSS under 500 ppm at all times. Following cofferdam breaching, there was a significant short duration spike in TSS (almost 35,000 ppm higher than background condition derived based on sampling data provided by J. Major (personal communication, 2009)) that resulted in a daily average TSS approximately 2000 ppm higher than background conditions for the first 24 h period following removal. After approximately 10 h following cofferdam breaching, however, TSS concentrations were observed to be similar to background levels as predicted.

5.2. Volume and Grain Size Distribution of the Deposit

Coring and shallow pit bulk sampling within the Marmot Dam impoundment was conducted by *Squier Associates* [2000] in order to better understand the volume and grain size distribution of the deposit (key sediment transport model inputs) and to determine potential toxic chemical concentrations, which could disallow the release of the reservoir deposit downstream if concentrations exceeded environmental standards. The volume of sediment deposited in the Marmot impoundment as estimated by *Squier Associates* [2000] was approximately 750,000 m^3, of which approximately 490,000 m^3 was gravel ($\geq$2 mm) and 260,000 m^3 sand (<2 mm), deposited in two distinctive layers as shown in Figure 12. The grain size distributions of the deposits were also used to approximate the grain size distribution of the sediment supply as discussed below.

5.3. Sediment Supply

Based on a review of pertinent sediment production literature from the same region with similar geological and climatic conditions, Stillwater Sciences geologists estimated a sediment production rate in the Sandy River basin upstream of Marmot Dam between 100 and 600 t km^{-2} yr^{-1}, which translates to approximately 70,000 to 300,000 t yr^{-1} sediment supply at the Marmot Dam site that has a catchment area of approximately 680 km^2 [*Stillwater Sciences*, 2000] (accessed May 2010). The coring results of the impoundment deposit (Squier Associates 2000, as discussed above) provided an excellent approximation of the grain size distribution of the sediment supply. The volume estimate of the impoundment deposit, however, was not beneficial for calculating the sediment supply rate in the Sandy River because the duration that completely filled Marmot Dam impoundment was unknown (e.g., the reservoir was completely full of sediment and when this occurred was unknown). Although the 70,000 to 300,000 t yr^{-1} sediment supply was a rough estimate, a more accurate assessment was determined neither practical nor necessary, and the rough supply estimate would be adequate to conduct sediment transport modeling, especially with the grain size distribution information provided through the coring exercise. Because most of the reach of interest (i.e., Sandy River downstream of Marmot Dam) is gravel bedded, the gravel supply rate is the primary consideration for modeling. Using the assumption that 5% to 10% of the sediment supply is gravel, the 490,000 m^3 of gravel deposited within the Marmot Dam impoundment would represent at least 15 times more than the long-term averaged gravel supply in the Sandy River. As a result, the sediment supply used as model input was not expected to significantly affect the simulated sediment transport dynamics following Marmot Dam removal. Nevertheless, the assumed sediment supply rate is important because an imbalance between the sediment supply rate (and the associated grain size distribution) with the channel geometry, slope, and hydrology would result in persistent channel aggradation or degradation for long-term simulations such as those following dam removal. A sediment supply rate that is in balance with sediment transport capacity in the Sandy River was determined as part of the zeroing process discussed later in this section.

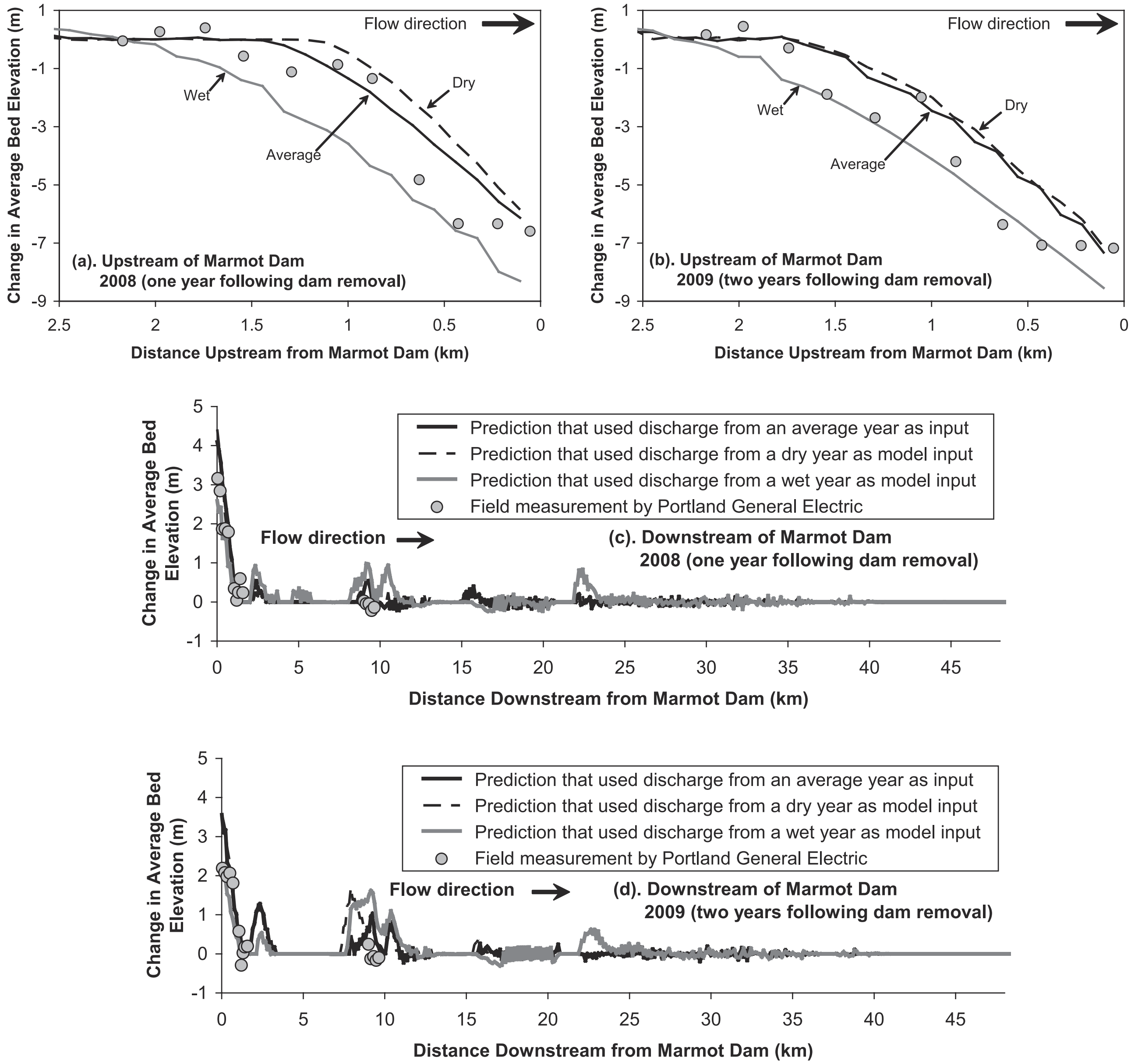

Figure 11. Predicted channel aggradation and deposition following Marmot Dam removal in comparison with predam-removal and postdam-removal survey data. (a) Upstream of Marmot Dam 1 year following removal. (b) Upstream of Marmot Dam 2 years following removal. (c) Downstream of Marmot Dam 1 year following removal. (d) Downstream of Marmot Dam 2 years following removal. Predictions used three typical hydrologic conditions as input for the first year of modeling: a wet, an average, and a dry year. Predictions were made in 1999 and presented by *Stillwater Sciences* [2000] and *Cui and Wilcox* [2008]. Marmot Dam was removed during the summer of 2007, and predam- and postdam-removal field data were collected by Portland General Electric (PGE) during the summers of 2005 through 2009. More details of the survey data are given by *Stillwater Sciences* [2010].

5.4. Simulating Discharge

Hydrologic conditions vary considerably from year to year in the Sandy River, and using different hydrologic conditions will influence modeling results. While some practitioners advocate using a Monte Carlo method to simulate sediment transport dynamics (e.g., treating discharge and sediment supply as stochastic parameters in the model using random

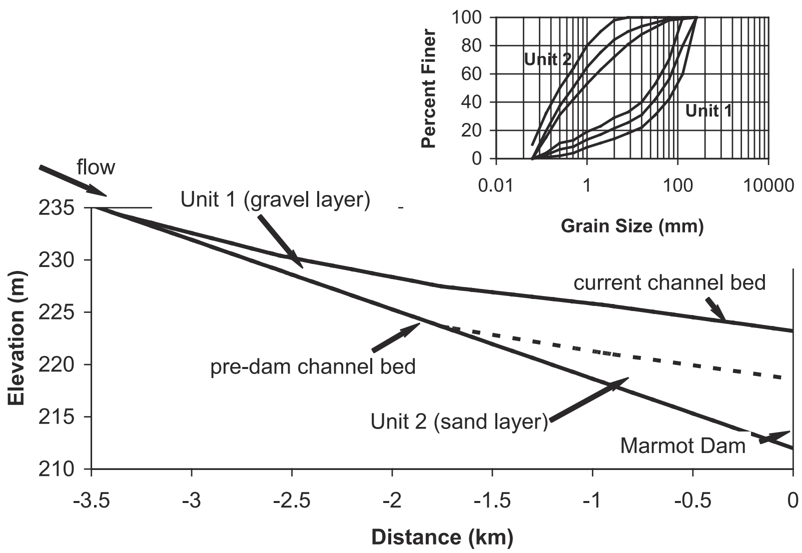

Figure 12. Marmot Dam upstream sediment deposit, showing two layers: a top layer composed primarily of gravel and pebble (unit 1) and a bottom layer composed primarily of sand (unit 2). Grain size distributions of the two layers are provided as average and ±1 standard deviation. Diagram was developed based on information provided by *Squier Associates* [2000] and previously published in the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

functions following predetermined distributions), simulations for Marmot Dam removal project used a much simpler method that provided confident predictions with far less effort. Recognizing that postdam-removal sediment transport would become less active through time, three model runs were conducted for each removal alternative, each run using a different hydrologic condition to represent the first year following dam removal, while the hydrologic conditions starting in year 2 after dam removal were selected randomly from the existing record and were consistent for all runs. The three hydrologic conditions were represented by three typical years selected from the daily discharge record based on exceedance probabilities for the annual peak series and annual runoff: a wet year with exceedance probability of approximately 0.1, an average year of approximately 0.5, and a dry year of approximately 0.9. Three sets of results were provided for each alternative, with the sediment transport characteristics expected to fall within the envelope of values included in the three scenarios. This practice proved to be effective, as evidenced by the comparison between preremoval modeling results and postremoval field data shown in Figure 11. A Monte Carlo simulation would likely have achieved similar results, but the number of runs for each alternative would need to be several hundreds, if not thousands, in order to achieve meaningful statistics for the simulated results.

5.5. Determining Downstream Boundary Conditions

The downstream boundary of the simulation was set at the Sandy River-Columbia River confluence approximately 48 km downstream of Marmot Dam as a fixed bed elevation and a normal flow condition. A mathematically correct downstream boundary condition would involve using a series of water surface elevations in the Columbia River at the confluence, but this is impractical because future water surface elevations are unknown and have no direct relation with the discharge in the Sandy River. The downstream boundary condition was not expected to have any impact to the modeling results near Marmot Dam, where significant erosion (in the impoundment) and deposition (downstream of the dam) would occur. Because there was considerable amount of sand in the Marmot Dam deposit, and sand was expected to transport rapidly downstream following dam removal, however, it was unclear whether the approximate nature or uncertainty in downstream boundary condition would have some influence on predicted sand deposition near the confluence. This uncertainty could have potential impacts for local fish passage if greater amounts of sand were deposited than predicted, especially during critically dry flow conditions. The intractability of this concern was addressed with a contingency plan (discussed later) instead of trying to increase the confidence and precision of the modeling, which was likely not possible to achieve.

5.6. Setting the Initial Channel Geometry

Marmot Dam removal sediment transport modeling was unique in that channel geometry and longitudinal profile for modeling input were all obtained through remote techniques: channel geometry was assumed to be rectangle, and the width of the channel was measured from a set of 1:6000 scale aerial photos; longitudinal profile was measured through photogrammetry analysis based on aerial photographs obtained during low flow periods. Only limited field data were collected, which was used only for validation purposes.

We used rectangles to approximate detailed channel geometry because the width to depth ratio of a natural river is usually large during high flow events when there is active sediment transport, and a rectangle usually provides a good approximation of the cross section if floodplains are neglected. Neglecting floodplains is often acceptable because (1) overbank flow events usually occur only for a small fraction of time, and thus, the cumulative sediment transport during overbank flow periods usually accounts for only a small part of the total sediment transport despite the fact that overbank flow events are always associated with significant sediment transport and (2) potential simulation errors introduced by omitting floodplain are usually collectively accounted for with other modeling uncertainties (e.g., hydrology and sediment supply) in the calibration process that includes a period of time with different flow events, including overbank flow events [*Cui et al.*, 2008].

We did not subtract water depth from the longitudinal profile obtained through photogrammetry analysis that are more representative of the water surface elevation instead of the thalweg elevation. Adjusting the longitudinal profile by subtracting water depth would have required on-the-ground cross-section surveys that are expensive and time consuming. More importantly, such adjustment would have provided no additional value for the modeling because the longitudinal profile of the river had to be adjusted during a zeroing process in order to serve as the initial condition for the modeling of sediment transport dynamics following dam removal.

5.7. Simulating the Baseline Condition: The Zeroing Process

The objective of a zeroing process is to adjust the model input parameters so that they approximately reproduce the existing quasi-equilibrium long profile under the assumed background conditions. One assumption, therefore, is that the modeled reach is in a quasi-equilibrium condition, in which some aggradation or degradation may occur following flood events, but the long-term cumulative channel aggradation or degradation is minimal. The zeroing process

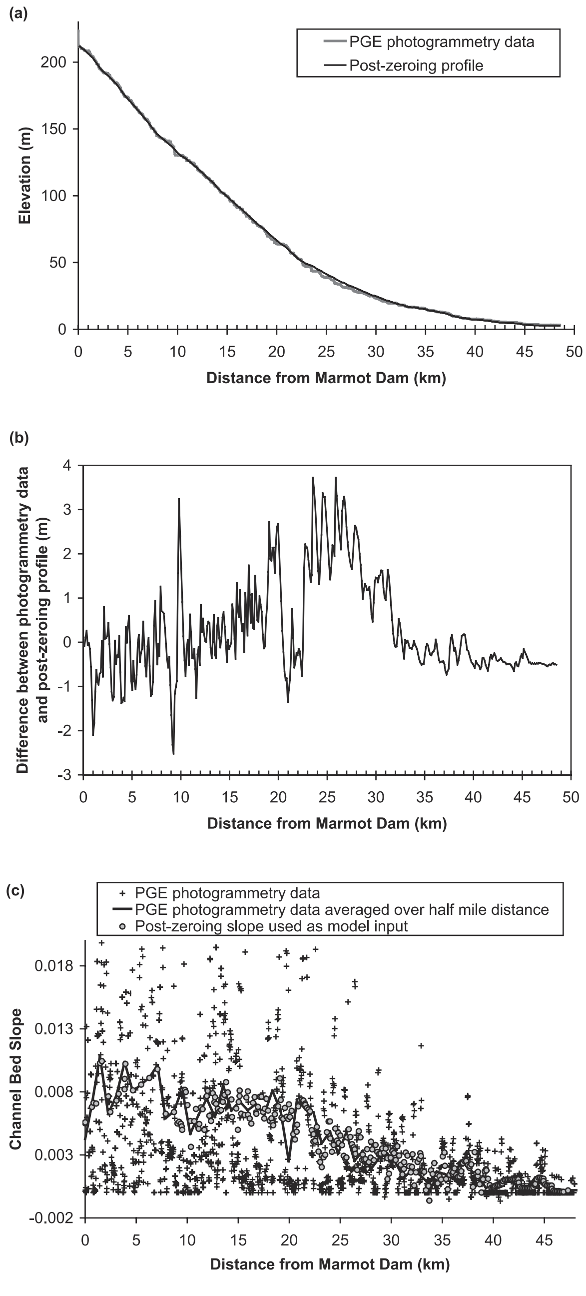

Figure 13. Comparison of the Sandy River photogrammetry-derived longitudinal profile to the postzeroing profile (i.e., the initial profile for Marmot Dam removal used for sediment transport modeling). (a) Longitudinal profile, showing reasonable agreement at least at the scale presented. (b) Net difference between the two profiles, showing up to 3.7 m differences at certain locations. (c) Channel gradient, which serves as the driving force for sediment transport. Figure 13c is from the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

involves running the model repeatedly with the surveyed longitudinal profile as the initial condition and the recorded hydrologic condition and best estimate of sediment supply rate and grain size distribution as boundary conditions. During this process, certain input data such as channel width, sediment supply rate, and/or grain size distribution are adjusted iteratively until the model reproduces a quasi-equilibrium profile similar to that observed. The reproduced quasi-equilibrium longitudinal profile is then used as the initial profile for modeling future conditions such as evaluating sediment transport dynamics following dam removal herein or following channel reconstruction for restoration. Because this initial profile is in quasi-equilibrium state within the model, any deviations from this condition in the subsequent simulations are considered to result from the perturbation injected into the model input (e.g., the release of the sediment deposit in the impoundment area in case of dam removal).

During the zeroing process for Marmot Dam removal sediment transport study, the following adjustments were made: channel width was adjusted by narrowing some of the excessively wide cross sections, long-term averaged sediment supply rate was adjusted within the range found during the literature review, and the abrasion coefficient of gravel particles was also adjusted based on published range so that the predicted grain size distribution and longitudinal profile under the current conditions were similar to observations. The resulting postzeroing equilibrium profile and channel gradient are shown in Figure 13 in comparison with the surveyed data, and the simulated annual and cumulative changes in bed elevation postzeroing process are shown in Figures 14 and 15, respectively. Note in Figure 13a that the

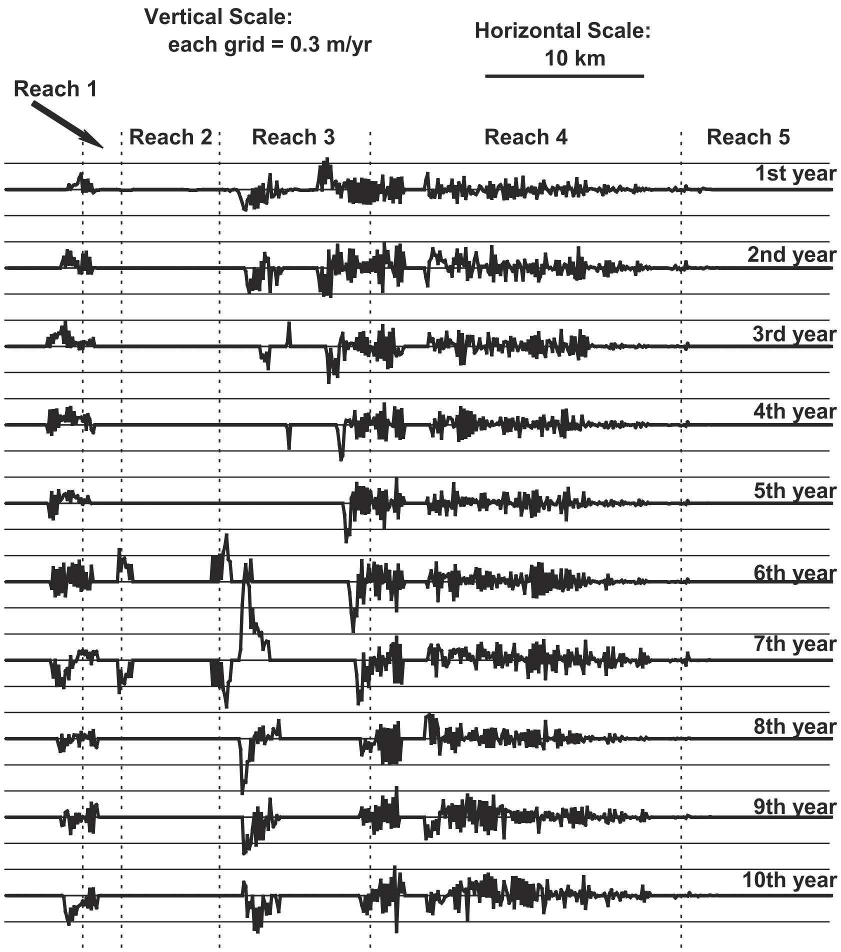

Figure 14. Simulated annual change in bed elevation under the assumed background condition, showing up to 0.6 m of annual aggradation or degradation due to sediment deposition or erosion at certain locations. Flow is from left to right; Marmot Dam was located at the upstream end of reach 1; and the Columbia River confluence is at the downstream end of reach 5. From the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

postzeroing profile is very similar to the photogrammetry data, at least at the scale shown, and the channel gradient is also similar between the two sets of profiles, as shown in Figure 13c. However, there was actually up to 3.7 m of elevation difference between the two profiles, as shown in Figure 13b. The simulated channel aggradation and degradation under background conditions using the postzeroing profile as initial condition indicated that although there is annual aggradation or degradation up to 0.6 m at certain locations (Figure 14), the cumulative change in bed elevation is minimal (Figure 15). As such, any changes simulated following dam removal would indeed be the consequences of the release of stored sediment.

It should be noted that there are professional judgment to be made during the zeroing process, and the experience of the modeler may play an important role in successfully conducting the zeroing process.

5.8. Interpreting Model Results

Running a sediment transport model is only part of the process of analyzing sediment transport dynamics for river

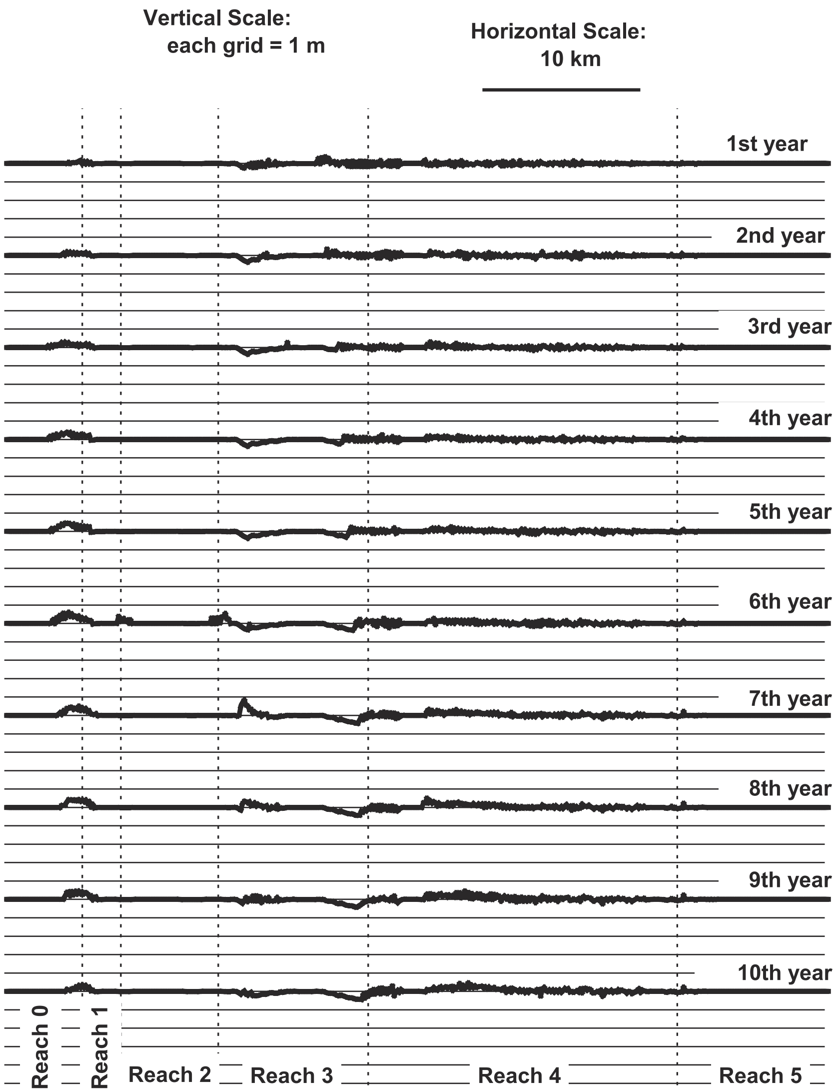

Figure 15. Simulated cumulative change in bed elevation under the assumed background condition, showing minimal cumulative change in the 10 year period of simulation. Flow is from left to right; Marmot Dam was located between reaches 0 and 1, and the Columbia River confluence is at the downstream end of reach 5. From the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

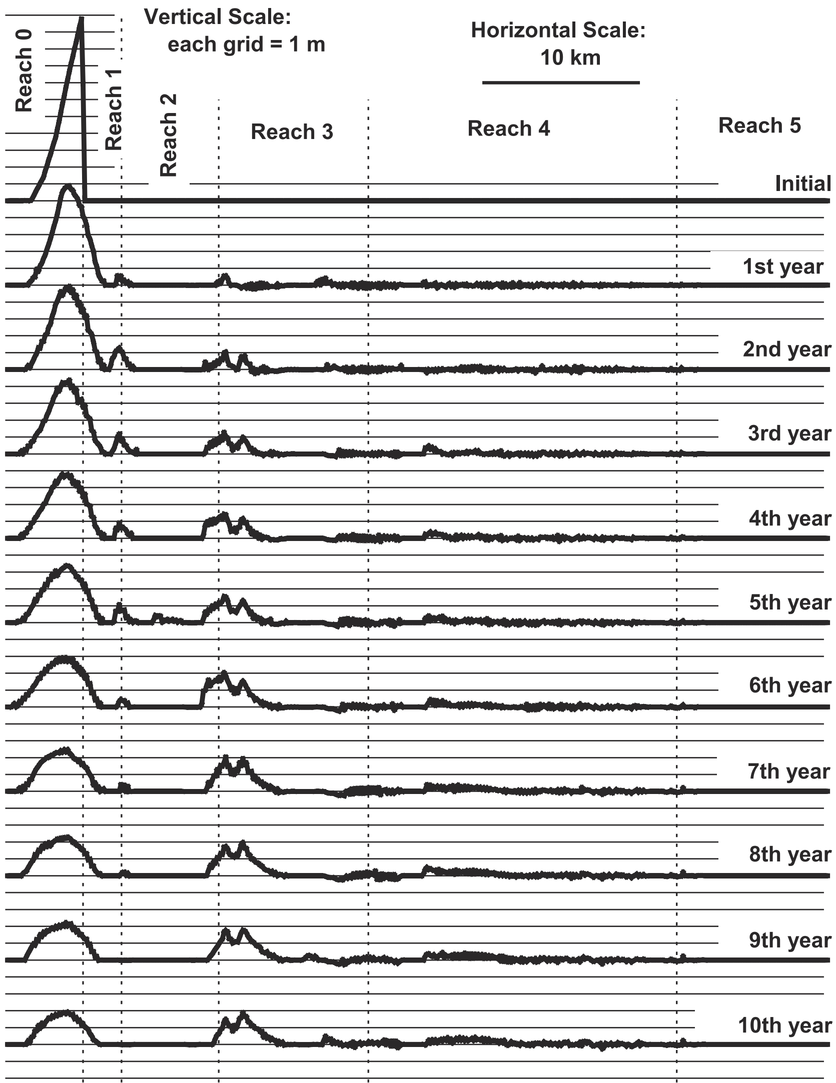

Figure 16. Simulated channel aggradation and degradation following Marmot Dam removal for the "blow-and-go" scenario under the average hydrologic condition. Flow is from left to right; Marmot Dam was located between reaches 0 and 1, and the Columbia River confluence is at the downstream end of reach 5. From the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

restoration. Critically, model results need interpretation by experienced professionals for any number of reasons, of which three are highlighted here. First, there is a need to identify whether all the model results are explainable and, if not, whether there are potential errors in the model and/or input data. During the Marmot Dam removal sediment transport modeling, for example, the first set of results showed an unlikely scenario of more sediment deposition in the narrow and steep Sandy Gorge (reach 2 in Figures 14 and 15), than in neighboring reaches. This led to the discovery and correction of an error in the model, a single line error within thousands of lines of FORTRAN code that would not have been discovered otherwise. Figure 16 illustrates the results from one run of the corrected model showing minimal change in bed elevation within the Sandy Gorge following dam removal, far more consistent with an understanding of how sediment behaves in similar conditions. Second, there is a need to ensure that results are interpreted by those with sufficient experience in sediment transport processes and sediment transport modeling, and with a good understanding of the river system to be modeled. Bed elevation and thickness of sediment deposition as a function of distance, for example, can easily be interpreted inaccurately because results are often presented (by necessity) on plots with very compressed longitudinal scales and measured from arbitrary datum. Using the results presented in Figure 16 as an example, it is very easy to think that there would be substantial amount of reservoir sediment left in the impoundment area (reach 0) and downstream even 10 years after dam removal, evidenced by the "bumps" in reaches 0 and 3, raising concerns about potential long-term impact to spawning habitat in the river. However, for those with knowledge of the river and modeling practice, at least part of the bump in the impoundment area was to involve predam topography rather than the reservoir deposit because the thickness of sediment deposit in that reach was measured from arbitrary bench values (i.e., if the thickness is measured from a higher elevation in that area, the bump would have been smaller). In addition to cumulative thickness of sediment deposition presented in Figure 16, an experienced modeler would also examine the annual change in bed elevation (Figure 17). Results in Figure 17 show that annual change in bed elevation for reaches 2, 3, 4, and 5 is only slightly higher than background conditions (shown in Figure 14) at all times, and reach 0 and 1 becomes similar to background conditions (i.e., similar to downstream reaches) after approximately year 2, indicating that the potential impact to spawning habitat would be minimal except in the reach immediately upstream and downstream of the dam (i.e., the downstream portion of reach 0 and upstream portion of reach 1), where the impact was expected to last for a couple of years at maximum. Third, river projects often involve multidisciplinary issues so requiring a team of experienced experts to understand and interpret the issues related to sediment transport. In the Marmot Dam removal project,

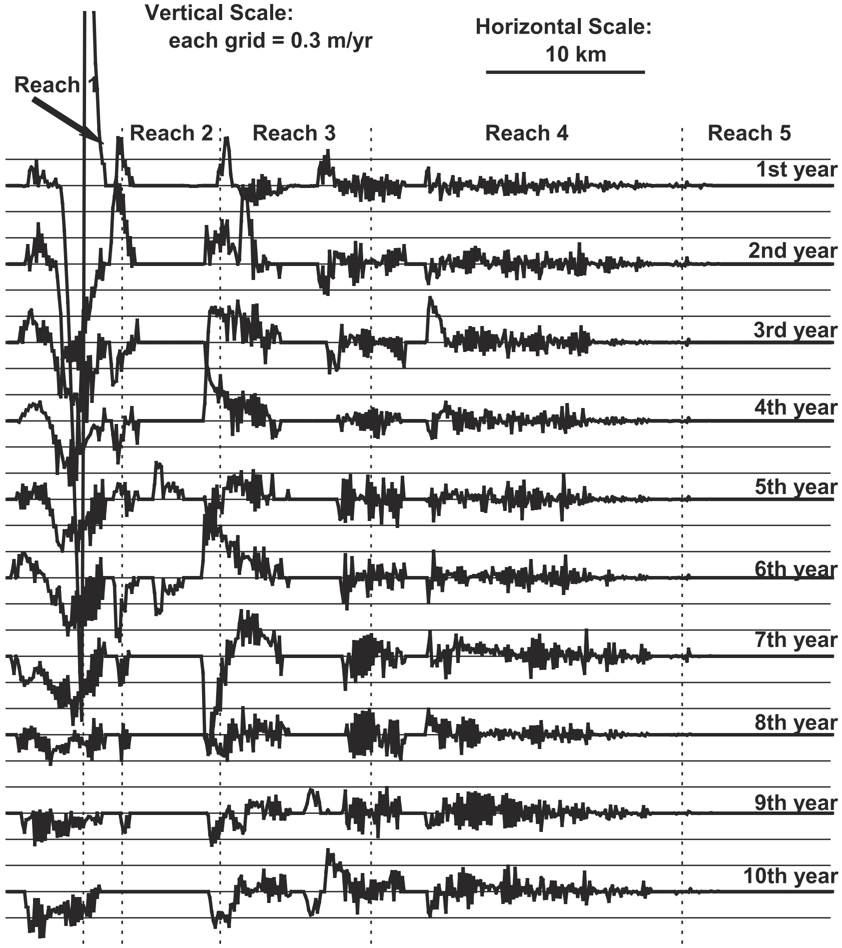

Figure 17. Simulated annual change in bed elevation following Marmot Dam removal for the "blow-and-go" scenario under the average hydrologic condition. Flow is from left to right; Marmot Dam was located between reaches 0 and 1, and the Columbia River confluence is at the downstream end of reach 5. From the work of *Cui and Wilcox* [2008], reprinted with permission from ASCE.

for example, a team of geomorphologists, engineers, fisheries biologists, and riparian biologists were required to help interpret the modeling results. The team of scientists not only provided interpretations of modeling results, but also identified sensitive issues and, in association with uncertainties of the model, designed contingency plans (discussed below) to ensure that the project would be successful even if certain aspects of sediment transport dynamics were predicted less accurately or incorrectly.

5.9. Accommodating Uncertainties

Numerical sediment transport models are far from perfect representations of the prototype rivers, and there are always uncertainties associated with their predictions. In addition to unknown future hydrologic conditions, the most common sources of uncertainties include the following: (1) model resolution lower than what is required for a particular project, (2) areas of concern not represented or inadequately represented by the model, and (3) uncertainties associated with upstream and downstream boundary conditions. These uncertainties must be identified by the modeler and the team of professionals that provide interpretations to modeling results. Several techniques can make the numerical modeling results useful despite the uncertainties associated with the modeling results: (1) to conduct sensitivity test runs and provide a potential range of outcomes rather than providing a single set of prediction so that the "true results" will be most likely within the predicted range, (2) to use the model results comparatively between different alternatives, (3) to better understand the specific concern and potential outcome through more in-depth research, and (4) to provide appropriate contingency plans to address the concerns. These techniques are discussed below.

5.10. Conducting Sensitivity Test Runs

As discussed earlier, sediment transport modeling for Marmot Dam removal project used discharge record from three different hydrologic years to serve as input data during the first year following dam removal and, thus, providing three sets of results for each scenario. This practice proved to be effective, evidenced by the comparison of observed and predicted channel aggradation and degradation shown in Figure 11, where observations generally fell within the predicted range. In addition to considerations of uncertainties in future discharge, the modeling also conducted other sensitivity test runs, including (1) potential errors in estimated grain size distributions of the reservoir deposits (assumed coarser- and finer-grain sizes as sensitivity test runs) and (2) potential errors in sediment transport equations in case of steep slopes (applied slope adjustment to predicted sediment transport capacity as sensitivity test runs). Detailed descriptions of Marmot Dam removal study sensitivity test runs can be found in *Stillwater Sciences* [2000] and the work of *Cui and Wilcox* [2008], and sensitivity runs that examined a variety of parameters for dam removal sediment transport modeling can be found in the work of *Cui et al.* [2006b].

5.11. Using Modeling Results Comparatively

Despite the many uncertainties, modeling results are generally much more reliable if used for comparisons of different alternatives. For example, modeling results shown in Figure 16 indicated that there would be increasing sediment deposition near the upstream end of reach 3 in the first few years following dam removal, reaching a maximum value of approximately 0.6 m in year 6. This result could only be interpreted as that there would likely be sediment deposition in that area, with relatively low confidence level in the predicted value of 0.6 m. If, however, a second dam removal alternative was simulated under the same assumptions, and the results indicated that there would be only up to 0.3 m of sediment deposition in the same area, then we would be able to say confidently that the second dam removal alternative would result in lower sediment deposition in reach 3, although the level of confidence toward the predicted value of 0.3 m was equally low. The concept of using modeling results comparatively was fully utilized during Marmot Dam removal sediment transport study, and the model was used to examine several dam removal alternatives, providing important information for the stakeholders to select a preferred alternative confidently based on comparisons of modeling results between the alternatives. For example, modeling results indicated that a 2 year staged removal would produce no benefit in terms of reducing the amount of downstream sediment deposition, and dredging sediment during one dry season (summer and fall) prior to dam removal would produce minimal benefit compared to the most cost-effective "blow-and-go" alternative. Had these results not been available at the time, it was most likely that regulating agencies would have demanded dredging prior to dam removal or a removal alternative that releases sediment gradually, and PGE might have chosen to abandon the dam for some other interested parties to continue to operate instead of dam removal due to the high cost.

5.12. Detailed Research for Specific Questions

Although none of the concerns raised during Marmot Dam removal study was addressed through in-depth research during the project, a couple of follow-up research projects that benefit dam removal project, in general, were inspired by Marmot Dam removal project. These studies are (1) flume experiments investigating sediment pulse dynamics in a channel with pool-riffle morphology, inspired by the need to understand whether a large pulse of sediment release following dam removal project would result in reduced channel complexity (i.e., filled pools and flattening of channel bed) and (2) flume experiments investigating fine-sediment infiltration into a gravel deposit, inspired by the need to understand the impact to salmonid spawning habitat due to fine-sediment release following dam removal. These two studies are briefly discussed as examples of generic flume experiments later in this chapter.

5.13. Contingency Plans

Instead of trying to answer each concern with greater confidence, appropriate contingency plans can often be used as a relatively economical way to address the uncertainties associated with sediment transport numerical modeling. Three contingency plans were developed by PGE to address the concerns where numerical modeling and professional judgment were unable to provide adequate resolution to the issues: (1) PGE would dredge the channel and/or install large woody debris if postremoval monitoring indicated upstream passage difficulties for adult salmonid in the vicinity of the dam, (2) PGE would dredge open the entrances to secondary channels if they were blocked by sediment deposition, and (3) PGE would contract a local miner to dredge a channel to facilitate upstream migration of adult salmon if monitoring indicated fish passage difficulty in the Sandy River delta. The concern for upstream fish passage near the Marmot Dam site was caused by the fact that numerical modeling was unable to answer the question as to whether the channel bed will become less complex following rapid sediment deposition, which in turn, could result in relatively shallow water depths. Similarly, numerical modeling was unable to provide information as to whether the sediment deposition at a specific location such as the entrance to a side channel will occur after dam removal. The fish passage concern near Sandy River delta was largely prompted by the fact that there was a historical upstream fish passage blockage during an extremely dry year, compounded with the fact that it is the downstream end boundary of the numerical modeling, and thus, there is relatively low confidence in modeling results in that specific area. Postremoval monitoring indicated none of the concerns were realized, and as a result, no contingency plan was put into action. The Marmot Dam removal project shows that providing contingency plans can be an efficient and economic way of addressing some of the uncertainties in numerical sediment transport modeling or other sediment transport analysis, especially where potential consequences are serious, and modeling results were unable to answer questions beyond a reasonable doubt.

5.14. Potential Multidimensional Numerical Modeling of Sediment Transport Following Marmot Dam Removal

Because 1-D numerical sediment transport models can only reliably predict sediment transport dynamics on a reach-averaged basis, as demonstrated earlier, it is intuitive to think that multiple dimensional numerical models or scaled physical models should be used in place of a 1-D model when more detailed results are desired. The objective of producing more detailed and reliable results using multidimensional numerical models or scaled physical models, however, may not be always achievable due to their respective limitations. A comprehensive description of multidimensional numerical sediment transport models can be found in *Spasojevic and Holly* [2008], where the authors summarized capability requirements of different numerical models, discussed modeling techniques, and provided model examples. Here we briefly discuss some of more important limitations facing the use of multidimensional models.

There are three main limitations for multidimensional numerical models. First, while multidimensional modeling can usually realistically reproduce the flow field, the detailed relation between sediment transport and movement of sediment particles is not fully understood, and all the current sediment transport equations were developed based on data collected on a cross-section averaged basis. As such, topography predicted as a direct result of flow field (such as scour due to river bend) can often be realistically modeled, but the topography associated with complex sediment transport dynamics, such as the formation and development of alternative bars in a straight channel, may not be realistically reproduced. Second, attempting to model sediment transport dynamics in detail requires the collection of detailed field data, some of which are critical to the modeling but impractical to obtain in many situations or at a large scale. For example, simulating detailed topography in an area subject to channel erosion will require the knowledge of detailed grain size distributions and information with regard to where nonerodible material (such as bedrock and large boulders) is located and how deep it is beneath the surface. While it is possible to make some generalized assumptions about grain size distributions based on observations of the surface or bulk samples, it is impractical to know the locations and depth of the bedrock and large boulders, and without such information, the modeling results with regard to future topography would not have the desired resolution. Third, limitations in available computer resources will set upper bounds on the number of nodes permissible in a multidimensional model simulation, and because computational meshes cannot be overly distorted (i.e., the longitudinal dimension of the meshes cannot be too much larger than the lateral dimension), there are practical limits on the length of the river that can be simulated. This latter issue can introduce a further problem in that modeling short reaches may make the entire simulation domain dependent on boundary (especially downstream boundary) conditions, making the simulation results unreliable. This is generally not an issue in 1-D modeling because a 1-D model can be set to a significantly longer reach so that the interested area is beyond the influence of the model boundary.

Multidimensional numerical modeling was not proposed during the Marmot Dam decommission process, although there was some interest from academicians to test run a 2-D model in a short reach up- and downstream of the dam. The primary reason that multidimensional numerical modeling was not proposed at the time was that 1-D numerical modeling had satisfactorily answered all the important questions that the stakeholders and regulating agencies needed to know, with a few uncertainties addressed with contingency plans discussed earlier, allowing the stakeholders to reach an agreement. In addition, multidimensional simulations would have been limited to only a short period of time following dam removal, which was not the primary interest of the stakeholders. Technically, setting up a 2-D model a short distance up- and downstream of the dam will face the difficulty of correctly assigning the downstream end boundary conditions because there would potentially be channel aggradation and subsequent degradation that was not known prior to modeling. As discussed earlier, modeling results in a short river reach will be dictated by boundary conditions, especially the downstream boundary condition. As a result, an independent 2-D numerical sediment transport model would not have been feasible, and 1-D modeling results would have to be used to serve as the downstream boundary condition for a 2-D model. If a 2-D numerical sediment transport modeling simulation was conducted using 1-D model results as downstream boundary condition, it may have yielded results of some interest that the 1-D model did not offer, but it should be noted that the 2-D modeling results would not have been more reliable than that of the 1-D model due to its dependence on 1-D modeling results. That is, seeking more reliable modeling results should not have been the reason to conduct a 2-D modeling under these circumstances.

5.15. Scaled Physical Model of Marmot Dam Removal Sediment Transport

Scaled physical models, which provide powerful visualizations for the modeled projects and events, are subject to similar limitations as multidimensional numerical sediment transport models. Because a scaled physical model usually cannot be constructed to represent a long river reach, for example, project managers and professionals using scaled physical models for river restoration and other river projects should pay particular attention to impact of possible errors introduced through the setup of boundary conditions and treat the results cautiously, as discussed below.

Prior to Marmot Dam removal, a physical model was constructed at St. Anthony Falls Laboratory (SAFL), the University of Minnesota by *Marr et al.* [2007] to provide direct observations of sediment transport dynamics following dam removal and to examine how to best breach the cofferdam following dam removal. The experiment provided powerful visualization effect as to how sediment would likely be eroded from the Marmot impoundment following cofferdam breaching and suggested that the cofferdam should be artificially breached near its left bank to allow for more efficient erosion of the reservoir deposit, which was adapted by the engineers in the field. The model also demonstrated the major limitations for scaled physical models in that it cannot cover an adequately long reach, and thus, modeling results are likely significantly affected by the downstream boundary set up, making a direct scale up of modeling results rather difficult. The SAFL's Marmot Dam removal model scaled approximately 305 m (1000 feet) of the impoundment area [*Marr et al.*, 2007] and an even shorter reach downstream of the dam, and as a result, the downstream boundary was located in a reach that received rapid and substantial deposition following dam removal (Figure 11). The experiment, however, could only use a fixed water surface elevation as downstream boundary condition. As a result, the model likely overpredicted the rate of sediment erosion from the impoundment, evidenced by a quick evacuation of all the reservoir deposit in the model. This notwithstanding, scaled models such as this are very useful to answer some of the specific questions, as long as their limitations are fully considered, and model results are not overinterpreted.

6. PRACTICAL USES OF GENERIC PHYSICAL MODELS

One of the most powerful but often ignored tools for understanding of sediment transport issues is generic physical/flume modeling (i.e., modeling not scaled according to a prototype). Generic flume experiments are usually not designed to solve a site-specific sediment transport problem. Instead, they most often attempt to answer fundamental sediment transport questions and therefore are used to develop universal theories and validate numerical models that can be applied to other projects in similar fluvial environments. Because of their wide-ranging applicability, generic flume experiments are probably one of the most economic ways for addressing sediment transport issues on a per project basis, even though conducting a successful generic flume experiment can be expensive.

The flume experiments conducted for sediment pulse evolution in rivers [e.g., *Lisle et al.*, 1997, 2001; *Cui et al.*, 2003a; *Sklar et al.*, 2009], for example, are typical generic experiments that established that sediment pulses in rivers evolve by a combination of dispersion and translation, with dispersion always being the dominant process. The research established theories and provided insight between the evolution of sediment pulses, the flow parameters, and the relative grain size distributions of bed material and pulse sediment; the measurements collected during the experiments became a critical data set for the examination and validation of numerical models developed thereafter [e.g., *Cui et al.*, 2003b], which were the predecessors of the 1-D sediment transport model used both for the Marmot Dam removal project [*Stillwater Sciences*, 2000; *Cui and Wilcox*, 2008] and, subsequently, the DREAM and TUGS models [*Cui et al.*, 2006a, 2006b; *Cui*, 2007a].

Two generic model experiments conducted at RFS inspired by Marmot and other dam removal projects can provide some insights with regard to the considerations and values of flume experiments. During the Marmot Dam removal project, it was not well understood if the rapid sediment deposition following dam removal would result in an oversimplified channel that would potentially impair holding, rearing, and spawning habitat for native salmonids. Although, from a project perspective, this concern was addressed satisfactorily through development of a contingency plan (discussed earlier in this chapter) during the project's permitting process, several flume experiments were conducted to better understand this issue because its potential impact on downstream biological processes is of interest in most dam removal projects. The key experimental results, illustrating the potential change in channel complexity following the introduction of sediment pulses, were briefly discussed in the work of *Downs et al.* [2009] and are presented in Figure 18. Experimental observations indicated that pools did not ubiquitously fill with sediment and maintained water depths similar to their initial depths in areas of higher shear stress while contracting in aerial extent as sediment accumulated in areas of lower shear stress areas. These data suggest that pool filling and topographic simplification of the channel bed are likely not issues of concern when considering the impact of the large amount of sediment release after dam removal, provided there is enough flow to transport the released sediment. This conclusion was also confirmed by field observations following the removal of Marmot Dam, where it was found that there was no substantial change in channel complexity index, defined here as the standard deviation of bed elevation [*Stillwater Sciences*, 2010] (accessed May 2010) (Figure 19). Lateral variations in channel bed elevation persisted following the dam removal, thus preserving a reasonable water depth even under low flow conditions.

In addition to bed burial, fine sediment (sand and finer) infiltrating into downstream gravel deposits and impacting habitat conditions and biological processes (e.g., salmonid spawning) has also been a concern associated with dam removal projects. To address these issues, a series of intensive flume experiments were conducted at the RFS to test

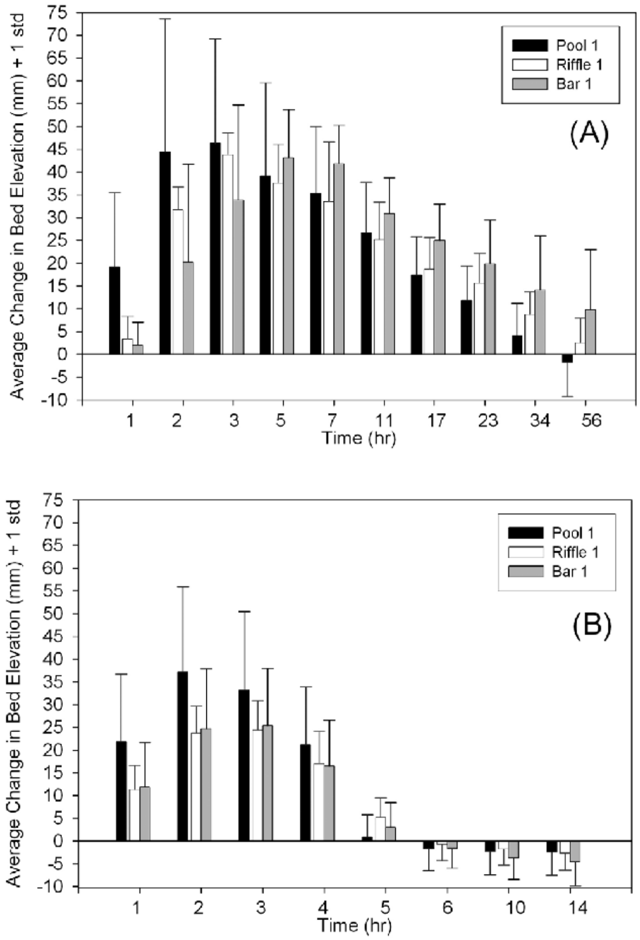

Figure 18. Average change in bed elevation +1 standard deviation for different morphologic units in the experimental runs. (a) Coarse-sediment pulse and (b) fine-sediment pulse. Diagram adapted from the work of *Downs et al.* [2009], reprinted with permission of IAHR.

ideas about the interaction between infiltrating sediment and the bed sediment framework [*Wooster et al.*, 2008]. Prior to the experiments, it was hypothesized that the amount of fine-sediment infiltration and the vertical profile of fine-sediment concentration in a gravel deposit following infiltration are functions of the grain size distributions of the gravel deposit and the infiltrating fine sediment. To examine this hypothesis, the flume was set at a moderate slope, divided into 10 zones (Figures 6a and 6c), and filled with gravel of nine different grain size distributions (two zones were filled with gravel of identical grain size distribution for comparison purposes). Water was than released into the flume at a constant discharge, and sand was fed from the upstream until the 10 zones appeared to be saturated with fine sediment (i.e., no more fine sediment could be infiltrated into the deposit). Upon termination of the experiment, multiple sediment samples were collected from each of the 10 zones at different depths, and grain size distribution of each sample was analyzed. Corroborating the experimental data with basic geometric relations, *Wooster et al.* [2008] were able to derive semiempirical relations linking the amount of fine sediment in a gravel deposit due to infiltration with grain size distributions of the gravel deposit and the infiltrating fine sediment. The experiments showed that fine-sediment concentration decreases exponentially in depth for the case of fine sediment infiltrating a gravel bed initially devoid of fine sediment, confirming observations that fine-sediment infiltration into gravel deposit is generally shallow. The implication of these results is that impact from fine-sediment release on a gravel bed following dam removal will be mostly on the channel

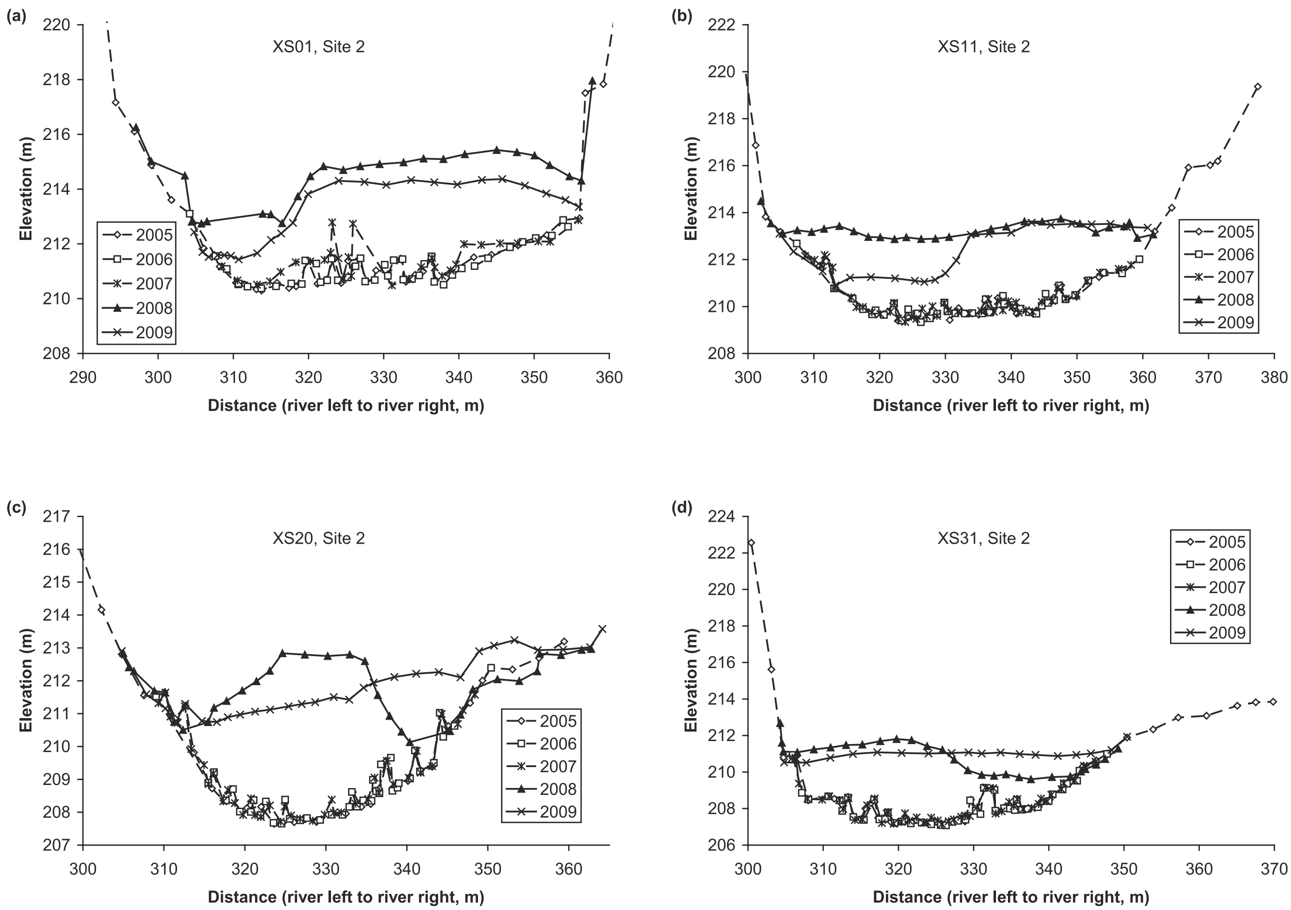

Figure 19. Selected cross sections in the Sandy River before (2005, 2006, and 2007) and after (2008 and 2009) Marmot Dam removal, showing persistent lateral variations in topography following substantial channel aggradation postdam removal. All four cross sections are located within 600 m downstream of the dam. Field data were collected by PGE.

surface, which can be helpful for the design of future dam removal projects. For example, in some dam removals, it may be comparatively beneficial to encourage the quick release of all impounded fine sediments prior to winter high flows (or a planned high flow release from upstream dams). The high flow will act to remove much of the fine sediment accumulated on channel surface and in the shallow depth of the deposit so limiting the duration of the impact from fine-sediment deposition. Figure 20 provides a comparison of the experimental data with the relations derived by *Wooster et al.* [2008], which also contains an account of experimental details.

7. SUMMARY AND CONCLUSIONS

In this chapter, we have discussed several practical issues that are important for sediment transport evaluations, including the importance of selecting appropriate methods and tools, the importance of understanding the limitations of the tools to be used for the analysis, consideration of issues

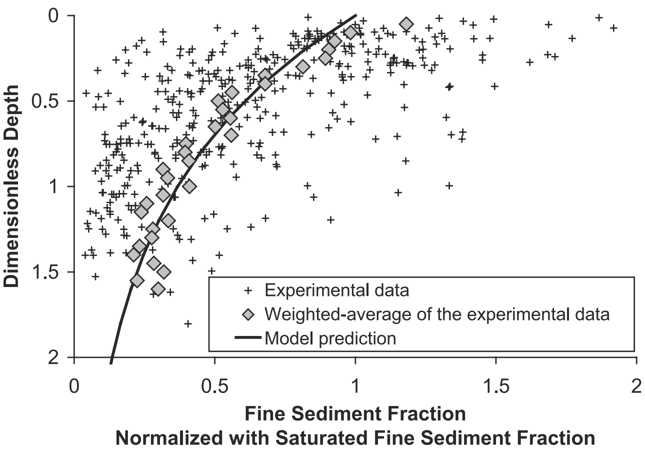

Figure 20. Fine-sediment infiltration experiments: Comparison of experimental data with the predictive relations derived from the experiments, showing good agreement when compared on a weighted-averaged basis. The solid line is predicted with the following relation derived by *Wooster et al.* [2008]: $\frac{f}{f_s} = \exp\left\{-0.0233\sigma_{gg}^{1.95}\left[\ln\left(\frac{D_{gg}\sigma_{gg}}{D_{sg}}\right)-2.44\right]\left(\frac{z}{D_{gg}}-2\right)\right\}$, where f denotes fine-sediment fraction; f_s denotes saturated fine-sediment fraction; σ_{gg} denotes geometric standard deviation of gravel deposit; D_{gg} denotes geometric mean grain size of gravel deposit; D_{sg} denotes geometric mean grain size of the infiltrating fine sediment; z denotes depth into the deposit; and z/D_{gg} is the dimensionless depth or the y axis of the diagram. Saturated fine-sediment fraction (f_s) is predicted with $f_s = \frac{0.621(1-0.621\sigma_{sg}^{-0.659})\sigma_{gg}^{-0.659}}{1-0.621^2(\sigma_{gg}\sigma_{sg})^{-0.659}}\left[1-\exp\left(-0.0146\frac{D_{gg}}{Dy}+0.0117\right)\right]$. More details of the research are given by *Wooster et al.* [2008].

relating to data use, setting baseline conditions, interpreting the results and accommodating uncertainties in numerical models, and the potential utility of physical models. Because sediment transport is inherently complex and contingent on boundary conditions specific to each project, and because the discussions are drawn mostly from our past experiences, the information presented here is far from comprehensive. As such, this chapter is intended as a vehicle to promote critical thinking before, during, and after sediment transport evaluations rather than as a specific guide for sediment transport modeling. The key experiences that we recommend are as follows:

1. Select appropriate methods and tools and avoid the one-tool-fits-all practice that we see often. This is usually the first step to ensure that the analysis achieves the project goal.

2. Understand the limitations of the tools utilized for the analysis, reduce uncertainties through appropriate techniques, and if necessary, provide contingency plans to safeguard the success of the project. None of the tools used in sediment transport analysis are perfect, but modeling results are very useful if their limitations are realized and appreciated in interpreting the results.

Set up models using the correct techniques. Tools for sediment transport analyses, particularly numerical models, require the modeler to have extensive modeling experience, comprehensive knowledge of sediment transport theories and principles, and an analytical understanding of specific geomorphic conditions of the river in order to make certain critical decisions. Important steps in the modeling process include (1) determining the appropriate input data and the level of accuracy of input data; (2) adjusting the model and input data through model calibration and/or other techniques (such as the zeroing process discussed in this chapter); and (3) simplifying the model inputs as much as is feasible (e.g., approximating channel cross-section dimensions using a simple rectangular channel form can often be sufficient).

3. Assemble a team of experienced professionals to guide the sediment transport analysis and to help the modeler correctly interpret the results. This is especially important given that numerical tools are becoming more user friendly, and users without adequate experience or knowledge can easily abuse them to generate outputs that may be inappropriate or incorrect. An incorrect modeling approach or an incorrect interpretation of model results will ultimately do more harm than good.

4. In addition to evaluating sediment transport dynamics on a project by project basis, generic flume experiments can be used to develop generalized theories related to sediment transport and habitat condition that can be used in a variety of restoration projects and fluvial settings. Despite the possible

relatively high expense to conduct a successful generic flume experiment, it is almost always more cost effective on a per project basis compared to dealing with the same issue in all the projects due to the fact that results obtained through a successful generic analysis are general in nature and can be applied to subsequent projects.

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